Critical Habitats and Biodiversity: Inventory, Thresholds and Governance

10

Abbreviations

ABNJ	Areas beyond national jurisdiction
AUV	Autonomous underwater vehicle
eDNA	Environmental DNA (molecular tool for assessing biodiversity)
BBNJ	Biodiversity beyond national jurisdiction (refers to negotiations to establish an international legally binding instrument under UNCLOS on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction)
BEF	Biodiversity and ecosystem function
B_{msy}	Biomass at maximum sustainable yield
CBD	Convention on Biological Diversity
CCAMLR	Convention for the Conservation of Antarctic Marine Living Resources
CCRF	Code of Conduct for Responsible Fishing
CoML	Census of Marine Life
COPEPOD	Coastal and Oceanic Plankton Ecology Production and Observation Database
CPUE	Catch per unit effort
DD	Data deficient (Red List category)
EBSA	Ecologically and biologically significant area
EEZ	Exclusive economic zone
EOV	Essential Ocean Variable
EuroGOOS	European Global Ocean Observing System
FAIR	Findable, accessible, interoperable and reusable (principles for data sharing)
FAO	Food and Agricultural Organization of the United Nations
GEO	Group on Earth Observations Biodiversity Observation Network
GOBI	Global Ocean Biodiversity Initiative
GOOS	Global Ocean Observing System
GOOS	BioEco Biology and Ecosystems Panel of the Global Ocean Observing System
GDP	Gross domestic product
IMOS	Integrated Marine Observing System (Australia)
IOC	Intergovernmental Oceanographic Commission
IODE	International Oceanographic Data and Information Exchange
IOOS	Integrated Ocean Observing System (United States)
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISA	International Seabed Authority (UN agency charged with managing mining in the area; seabed in ABNJ)
ITIS	Integrated Taxonomic Information System
IUCN	International Union for the Conservation of Nature
IUU	Illegal, unregulated and unreported
KBA	Key biodiversity area
MBON	Marine Biodiversity Observation Network (part of the GEO BON program)
MPA	Marine protected area
NAGISA	Natural Geography in Shore Areas (CoML project)
NCP	Nature's contribution to people

Originally published in:
Rogers, A., O. Aburto-Oropeza, et al. 2020. *Critical Habitats and Biodiversity: Inventory, Thresholds and Governance*. Washington, DC: World Resources Institute. Available online at www.oceanpanel.org/blue-papers/critical-habitats-and-biodiversity-inventory-thresholds-and-governance.
Reprint by Springer International Publishing (2023) with kind permission.
Published under license from the World Research Institute.

J. Lubchenco, P. M. Haugan (eds.), *The Blue Compendium*, https://doi.org/10.1007/978-3-031-16277-0_10

NEAFC	North-East Atlantic Fisheries Commission
NEOLI	No-take, enforced, old, large, isolated (refers to MPAs; Edgar et al. 2014)
NGO	Non-governmental organisation
OBIS	Ocean Biogeographic Information System
OECM	Other effective area-based marine conservation measure
OSPAR	Oslo Paris Commission
PR	Performance review (in the context of fisheries management organisations)
RFMO	Regional Fisheries Management Organisation
RLS	Reef Life Survey
ROV	Remotely operated vehicle
SDG	Sustainable Development Goal
TURF	Territorial use rights for fishing programs
UN	United Nations
UNCLOS	United Nations Convention on the Law of the Sea
UNEP	United Nations Environment Programme
UNESCO	UNESCO
VME	Vulnerable marine ecosystems
WCMC	World Conservation Monitoring Centre
WoRMS	World Register of Marine Species

Highlights

- Evidence suggests that ocean biodiversity at all levels is being lost as a result of the direct and indirect impacts of human pressures. The main drivers of biodiversity loss are overexploitation and human pressures in coastal environments (development, habitat loss, pollution, disturbance). Increasingly, climate change and ocean acidification are and will be drivers of biodiversity loss especially in sensitive coastal ecosystems.
- Despite advances in understanding the distribution of species and habitats in the ocean, many aspects of marine biodiversity remain poorly understood. As a result, changes in marine biodiversity are difficult to ascertain and there is a critical need to establish current baselines and trends through survey and monitoring activities.
- There needs to be a concerted effort to increase funding and capacity for marine biodiversity research, especially in developing countries which are rich in biodiversity. There also needs to be an increase in collaboration across scientific disciplines and other data users and measures to make data collection and analysis interoperable and repeatable to ensure that we can enjoy the benefits of ecosystem services which underpin the blue economy whilst ensuring that biodiversity is conserved. These efforts should be focused on the already established international networks for biodiversity monitoring that include the Biology and Ecosystems Panel of the Global Ocean Observing System (GOOS BioEco), the Group on Earth Observation Biodiversity Observation Network (GEO BON), the Marine Biodiversity Observation Network (MBON), and global data integrators such as the Ocean Biogeographic Information System (OBIS) of the International Oceanographic Data and Information Exchange (IODE) programme of the Intergovernmental Oceanographic Commission of the United Nations Educational, Scientific and Cultural Organization (UNESCO-IOC) and the Ocean Data Viewer of the United Nations Environment Programme's World Conservation Monitoring Centre (UNEP-WCMC).
- There has been a significant apparent increase in the coverage of marine protected areas (MPAs). However, most MPAs are only lightly to minimally protected, with many lacking even management plans and very few classified as fully protected. Maximum environmental and societal benefits accrue only when 30–40% of key marine ecosystems are represented in fully or highly protected and implemented MPAs. We estimate that only 3% of the key habitats explored in this study lie in fully protected MPAs, and for some habitats, no countries have placed them in fully protected MPAs. Hence, opportunities abound to strengthen protection in existing MPAs and create new highly to fully protected MPAs, paying close attention to positive enabling conditions, good design principles and adequate enforcement and funding.
- It is critical to establish a legal framework for the conservation of biodiversity in the whole ocean, including areas beyond national jurisdiction. For this reason, reaching a strong agreement for the new international legally binding instrument on the conservation and sustainable use of biological diversity of areas beyond national jurisdiction (BBNJ) is essential.
- The ability of wealthier countries to implement conservation measures within their exclusive economic zones (EEZs) is higher and might need to compensate for less wealthy countries with higher biodiversity and higher pressures. Achieving the 30–40% target in fully or highly protected areas, especially in developing countries, will be greatly enhanced by capacity building, financial support and development of alternate economically viable options for employment.
- Marine ecosystems often exhibit tipping points where pressures lead to a major regime shift that results in an alternative and less productive state. Recognising such tipping points and incorporating them as reference points in fisheries management can greatly improve marine species conservation as well as the functioning and resilience of marine ecosystems.
- Accelerated and expanded reform of fisheries management practices are required if the food and nutritional needs of a growing human population are to be met with-

out permanent and long-lasting biodiversity loss resulting in the erosion of ecosystem services. It is especially important that these reforms include greatly improved monitoring of catch and bycatch in fisheries; the elimination of illegal practices in industrial fisheries through improved enforcement; a reduction in the fishing capacity where it is contributing to overfishing and/or damage to biodiversity whilst ensuring that basic needs for food, nutrition and livelihoods are met in coastal communities; and better incorporation of biodiversity considerations into all levels of fisheries management and the fishing industry. There must be better collaboration with the environmental sector for government departments and also with intergovernmental and non-governmental organisations.

1 Overview

Marine habitats are extremely valuable in many ways (e.g., economically, culturally or for subsistence) and provide many necessary services for humans (Costanza et al. 1997, 2014). Despite their importance, coastal and oceanic habitats are increasingly threatened by fishing, climate change, oil and gas exploration, pollution and coastal development (Jackson et al. 2001; Halpern et al. 2008, 2019; Heery et al. 2017; Harris 2020). Habitat degradation and loss from these threats are not uniformly distributed and are cumulative with poorly understood interactions between pressures (Halpern et al. 2008). Despite the enormous impacts humans have had on marine ecosystems in the global ocean over the past 50 years, they tend to appear not as the complete extinction of individual species (Dulvy et al. 2003) but rather as changes in ecosystem composition and in the relative abundance and ecological status of individual species, along with more regional or local extirpations (Worm and Tittensor 2011). A species need not become globally extinct to radically alter the composition of the ecosystem ('ecological extinction'), disappear from the local environment ('local extinction') or become commercially non-viable ('commercial extinction'). Biodiversity loss is a globally significant symptom of unsustainable exploitation of Earth's natural environment and a major threat to the ecosystem services on which we, and future generations, depend.

The ocean's natural capacity to provide ecosystem services such as food, coastal protection and carbon sequestration are being eroded as a result of the above changes (Cheung et al. 2010, 2013; Barange et al. 2014; Spalding et al. 2014; Arias-Ortiz et al. 2018). Over 500 million people worldwide live in the coastal zone and are afforded protection by ecosystems such as coral reefs, mangroves forests, seagrass beds and kelp forests. In the case of coral reefs, the reduction in damage to terrestrial assets conferred through coastal protection is estimated at US $4 billion annually (Beck et al. 2018). For the top five countries that benefit from reef protection, this is the equivalent benefit of $400 million annually in mitigated damage to society (Beck et al. 2018). Without reefs, the economic impact of flooding would more than double, with the area of land affected increasing by 69% and people affected by 81% (Beck et al. 2018). The loss of this critical ecosystem, which is estimated to result in a 1–10% reduction of its former range under the most optimistic future scenarios (IPCC 2018), is a looming crisis of vast ecological and social dimensions.

In response to habitat degradation, losses in biodiversity and associated impacts, there has been an international effort towards conserving marine ecosystems. The Strategic Plan for Biodiversity 2011–2020 from the Convention on Biological Diversity (CBD) has resulted in an accelerated effort to increase the protection of marine areas. Specifically, Aichi Biodiversity Target 11 calls for the conservation by 2020 of 'at least 10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services … through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures'. A body of scientific literature suggests that the Aichi Biodiversity Target should be a first step. More ambitious targets of ocean protection (e.g., 30%), have been proposed and discussed in the scientific literature for many years (Gell and Roberts 2003; Balmford et al. 2004). Recent meta-analyses indicate that maximum environmental and societal benefits do not accrue until 30–40% of representative marine ecosystems are protected (Gell and Roberts 2003; Gaines et al. 2010; O'Leary et al. 2016; Sala et al. 2018a). This call for an enhanced scope for protection was endorsed by Resolution 50 of the International Union for the Conservation of Nature (IUCN) at the World Conservation Congress in 2016 'to designate and implement at least 30% of each marine habitat in a network of highly protected MPAs and other effective area-based conservation measures, with the ultimate aim of creating a fully sustainable ocean'. This call included specific reference to implementing protected areas in the exclusive economic zone (EEZ) of countries and in areas beyond national jurisdiction (ABNJ) (IUCN 2016).

Spatial conservation measures such as marine protected areas (MPAs) are one way of addressing these problems and have become the most recognised area-based marine conservation measure worldwide. An abundance of evidence suggests that if they are well designed, enforced and financed, fully protected MPAs can provide an abundance of benefits, including increases in biodiversity, size and abundance of previously targeted species (Halpern 2003; Lester and Halpern 2008; Lester et al. 2009; Edgar et al. 2014; Sala and Giakoumi 2017); enhanced spillover of juveniles and adults to adjacent fished areas (Halpern et al. 2010; Di Lorenzo et al. 2016); and restoration of ecological interactions within the protected area (Micheli et al. 2004; Mumby et al. 2007). More recent studies report additional benefits, including enhanced resilience to environmental and climate changes (Mumby and Harborne 2010; Micheli et al. 2012; Roberts et al. 2017; Bates et al. 2019). It is important to note here that biodiversity may benefit even further if more than 30–40% of representative habitats are protected by networks of MPAs. However, because of trade-offs between ocean conservation and uses such as fisheries, placing 30–40% of habitats in highly or fully protected MPAs is viewed as the optimal balance between protection of biodiversity and ecosystem service provision (Gaines et al. 2010). Also, to attain a representative coverage of 30% of marine habitats in fully or highly protected MPAs, a larger area may be required than 30% of the ocean to attain representativeness (O'Leary et al. 2018; see Jones et al. 2020 for an assessment based on species ranges lying within MPAs). Other effective area-based marine conservation measures (OECMs), such as locally managed marine areas, territorial use rights for fishing programs (TURFs), fisheries restricted areas, particularly sensitive sea areas, and areas of particular environmental interest, among others, have proven successful in conserving important areas for biodiversity and ecosystem services that include food security and poverty alleviation, such as in Northern Mozambique (Diz et al. 2018). The IUCN has created guidelines to recognise and report OECMs (IUCN-WCPA 2019) to incentivise robust long-term conservation and management of biodiversity. OECMs are an important but complementary tool to supplement an existing MPA network; however, they are not necessarily (or generally) mandated with a biodiversity conservation objective (Tittensor et al. 2019).

Therefore, this Blue Paper focuses on MPAs because they are supported by an important body of peer-reviewed literature indicating their effectiveness as fisheries management and conservation tools. Furthermore, MPAs can protect biodiversity but can also restore ecosystem structure, function and potentially services (Cheng et al. 2019) that mitigate and promote adaptation to climate change (Mumby and Harborne 2010; Micheli et al. 2012; Roberts et al. 2017). Therefore, implementing MPAs preserves habitats and their biodiversity and allows the maintenance of valuable ecosystem services (Costanza et al. 2014). We can roughly divide MPAs into no-take areas (where no fishing is allowed) and multiuse areas. Although, in some cases, the latter category does generate some benefits, in others, MPAs fail to reach their conservation objectives completely (Agardy et al. 2011). Scientific evidence is now accumulating in favour of fully protected MPAs (also known as marine reserves), which are dubbed most effective in environmental management (McClanahan et al. 2008; Edgar et al. 2014; MacNeil et al. 2015; Sala and Giakoumi 2017). Fully protected marine reserves, besides prohibiting fishing activities, also remove or minimise other human pressures that enable species to maintain or recover their abundance, biomass and diversity (Lester et al. 2009). It is notable, however, that MPAs are often not well designed, enforced or financed (Gill et al. 2017; Dureuil et al. 2018), which impacts their effectiveness, and there is particular concern for regions of high marine biodiversity, such as the marine biodiversity hot spot in Southeast Asia, where many species are reduced and destructive exploitation is expanding largely unchecked even within MPAs.

The High Level Panel for a Sustainable Ocean Economy has a vision of a productive and protected ocean, which will play a major role in achieving the Sustainable Development Goals (SDGs). Continued loss of marine biodiversity will undermine our ability to achieve a number of the SDGs, especially SDG 14 (to conserve and sustainably use the ocean), but also other goals (e.g., SDG 2, hunger and food security; SDG 9, resilient infrastructure). This Blue Paper addresses the topic of critical habitats and marine biodiversity with the following specific aims:

- Synthesise knowledge presenting the most recent inventory of marine habitats and biodiversity in the global ocean.
- Provide a brief overview of the impacts of habitat degradation and biodiversity loss in reducing ecosystem services.
- Review evidence of how biodiversity relates to ecosystem function and exploitation/degradation tipping points.
- Identify the range of measures undertaken by governments and industrial sectors to monitor, protect and address loss of marine biodiversity and their effectiveness.
- Determine opportunities for action to improve the sustainability of blue economic activities with respect to maintaining, and, where possible, restoring, the ocean's habitats and biodiversity.

We use the Convention on Biological Diversity's definition of biodiversity as the variability among living organisms, including diversity within species, between species and of ecosystems. The topics of marine biodiversity and ecosystem integrity are complicated by a lack of data, which pervades almost all aspects of our understanding of its distribution and trends. By necessity, therefore, we have been driven to examine specific aspects of the topic, such as well-studied groups of organisms or habitats as well as particular case studies.

This underlines the need for more scientific work on many aspects of ocean biodiversity, from variation within species and connectivity of populations to processes at the level of habitats and entire ecosystems, the sum of which underpin the functioning of Earth.

2 An Inventory of Marine Habitats and Biodiversity

2.1 Species

Globally, it is estimated that only 10–25% of marine species have been described (Mora et al. 2011; Appeltans et al. 2012), and some of the least known groups are likely to have thousands to over a hundred thousand undescribed species (e.g., Isopoda, Gastropoda, Tanaidacea). The geographic distributions of even fewer species are known (Gagné et al. 2020). Genomic approaches, coupled with large-scale sampling of the upper layers of the ocean (e.g., the Tara expedition), have also revealed tens of thousands of uncharacterised microbes, including eukaryotes, prokaryotes and viruses (de Vargas et al. 2015; Sunagawa et al. 2015). However, it is estimated that about half of the major taxonomic groupings (e.g., Vertebrata) have identified more than 50% of their known species already, and with the current rate of description of new species (average of 2000 new species described per year), those groups might have all their species described by the end of the century (Appeltans et al. 2012).

Knowledge of marine biodiversity varies markedly across regional, national and, more importantly, trophic levels (Costello et al. 2010). Data from the Census of Marine Life (CoML) programme is available in the ever-growing Ocean Biogeographic Information System (OBIS)[1] of the Intergovernmental Oceanographic Commission (IOC) of the United Nations Educational, Scientific and Cultural Organization (UNESCO). The CoML data suggest that, in relative terms, China, Australia and Europe have the best knowledge base of marine species with the tropical western Atlantic, tropical eastern Pacific and Canadian Arctic regions being poorly studied (Costello et al. 2010). Ecosystems that are particularly poorly known include the deep sea, coral reefs, icecovered areas and chemosynthetic habitats (Costello et al. 2010). Knowledge of the identity and distribution of commercially exploited taxa is greater than that of non-extracted taxa, and larger organisms tend to be better known than smaller organisms (Fautin et al. 2010; Worm and Tittensor 2018). Currently, only a handful of species are considered to have enough independent records that describe their full geographic distribution (about 50,000 species; Gagné et al. 2020). Emblematic (mammals, corals or fish) and exploited species (fish and invertebrates) are among the most well-documented spatially. Other patterns of biodiversity, including intraspecific genetic variation and habitat diversity, are also not well described (Fautin et al. 2010; Blasiak et al. 2020), with some exceptions. The Global Ocean Biodiversity Initiative (GOBI), which uses CoML and OBIS as primary sources of data, has participated in the CBD effort to identify ecologically and biologically significant areas (EBSAs) in the ocean.[2] These areas can be characterised by high biological diversity, but they also include a number of other criteria, including unique or rare species or communities; importance for the life history stages of marine species; importance for threatened or endangered species or habitats; vulnerability, fragility or slow recovery; biological productivity; and naturalness (CBD 2009). Geographic areas with the best knowledge of marine biodiversity do not match well with areas of highest diversity, reflecting both historical and present-day scientific capacity for taxonomy. Historically, highly sampled regions are often located in the Northen Hemisphere in the coastal regions around developed countries. It is crucial to account for such sampling bias when examining the distribution of biodiversity (Tittensor et al. 2010; Gagné et al. 2020). The common approaches to provide an unbiased picture of marine biodiversity consist of (1) removing species with not enough records to describe their full distribution and (2) applying statistical methodologies on known species records to correct for bias. The main hot spots of marine biodiversity have been recognized in the Indo-Pacific Coral Triangle and a lower peak in the Caribbean (Briggs 2007; Worm and Tittensor 2018; see Box 10.1). A general decline in biodiversity from the tropics to the polar latitudes has also been hypothesised, although there is debate on whether some taxa show more bimodal patterns (Thorson 1952, 1957; Fischer 1960; Stehli et al. 1967, 1972; Clarke and Crame 1997; Williamson 1997; Roy et al. 1998; Tittensor et al. 2010; Edgar et al. 2017; Worm and Tittensor 2018; Box 10.1). Hypothesised explanations include speciation and extinction rates over geological timescales as correlated with latitude (Crame and Clarke 1997; Jablonski et al. 2006, 2013) and ecological drivers such as habitat area, land versus ocean area by latitude, sea surface temperature (Worm and Tittensor 2018), and intrinsic biological traits such as larval development mode and interspecies interactions (Roy et al. 1998; Pappalardo and Fernández 2014; Edgar et al. 2017).

[1] For more information, see the OBIS website, https://obis.org

[2] To learn more about GOBI, visit its website, http://gobi.org/

Box 10.1 Estimating Global Patterns of Biodiversity

Using the biodiversity data found in Reygondeau (2019) and Gagné et al. (2020), the authors developed a standardised database drawing on online websites with records of the global distribution of marine species with sufficient records to have a robust distribution. Specifically, the database was populated with species data for which at least 10 spatially informed occurrences were available. Occurrence data originated from the Ocean Biogeographic Information System (OBIS);[a] Intergovernmental Oceanographic Commission (IOC) of the United Nations Educational, Scientific and Cultural Organization (UNESCO);[b] the Global Biodiversity Information Facility (GBIF);[c] Fishbase;[d] the Coastal and Oceanic Plankton Ecology Production and Observation Database (COPEPOD);[e] the Jellyfish Database Initiative;[f] and the International Union for the Conservation of Nature (IUCN).[g] The full filtering methodology can be found in Gagné et al. (2020).

From the initial data set (more than one billion entries), we removed records (1) with spatial location as “not assigned” (NA) or null values, (2) not identified to species level and (3) replicated among databases (i.e., records with the same species name, coordinates, and sampling details). The remaining records (731,329,129 records; more than 101,000 species) were assigned full taxonomic information using the Taxize library4 in R Studio. We also used this procedure to update all species’ synonyms to valid names, as officially recognised by the Integrated Taxonomic Information System (ITIS)[h] and the World Register of Marine Species (WoRMS).[i] Next, we explored the relationship between the number of independent records (independent in time and area of sampling) and latitudinal range and thermal range for species with well-known global coverage and ecology (number of observations greater than 2000; 1196 species). For each known species, we randomly selected *n* records (number of observations from 1 to 1000) within the global pool, and for each selected number of records (*n* = 1 to 1000 records), we computed the species’ latitudinal range and thermal range. The procedure was replicated 1000 times. We then confronted the simulated latitudinal range and thermal range (1000 simulations) to values obtained using all the information gathered on the species. We computed an interval of confidence of known range by quantifying the difference between the 1st and the 99th percentile of observed latitude coordinates and thermal value, and we assumed that the acceptable number of records to capture the latitudinal and thermal range was obtained when more than 950 randomly selected records were included within the confidence interval determined from the global pool of records. The median number of points found to capture the latitudinal range was 33 ± 4 records and 41 ± 3 records for thermal range. All species with less than 41 independent records were removed from further analysis.

Thus, the final data set on which all analyses presented in this study are based comprises up-to-date taxonomic information and filtered occurrences for 41,625 species, for a total of 51,459,235 records representing 17% of all accepted marine and non-fossil species.

Notes:

[a] *OBIS, http://www.iobis.org*

[b] *UNESCO-IOC, http://ioc-unesco.org/*

[c] *GBIF, http://www.gbif.org*

[d] *FishBase, http://www.fishbase.org*

[e] *COPEPOD, http://www.st.nmfs.noaa.gov/plankton*

[f] *Jellyfish Database Initiative, http://people.uncw.edu/condonr/JeDI/JeDI.html*

[g] *IUCN, http://www.iucnredlist.org/technical-documents/spatial-data*

[h] *ITIS, http://www.itis.gov*

[i] *WORMS, http://www.marinespecies.org*

[j] *For more information see WORMS*

The distribution of biodiversity in the global ocean has been described for numerous taxa, particularly in recent years as more observations have been synthesised into large-scale patterns (Tittensor et al. 2010; Reygondeau 2019). While there is consistency across many groups, it is important to bear in mind that there remains a significant taxonomic bias in our understanding.

There are some groups that we know well (typically those species in which we have a keen commercial interest or which are charismatic, such as vertebrates, or those which form biogenic habitats such as corals and seagrasses), but there are many for which we have very limited information (numerous invertebrate groups, most deep-sea taxa, and much of the microbial biosphere). In Box 10.1 we present a new analysis of the global pattern of marine biodiversity which is aimed at reducing bias from the issue of uneven sampling of species from different parts of the ocean.

At a global scale, the biodiversity distribution estimated from our study appears to be relatively consistent with other studies, resolutions and analyses (Fig. 10.1; Tittensor et al. 2010; Asch et al. 2018; Reygondeau 2019). The pattern across multiple taxa is primarily tropical to subtropical peaks

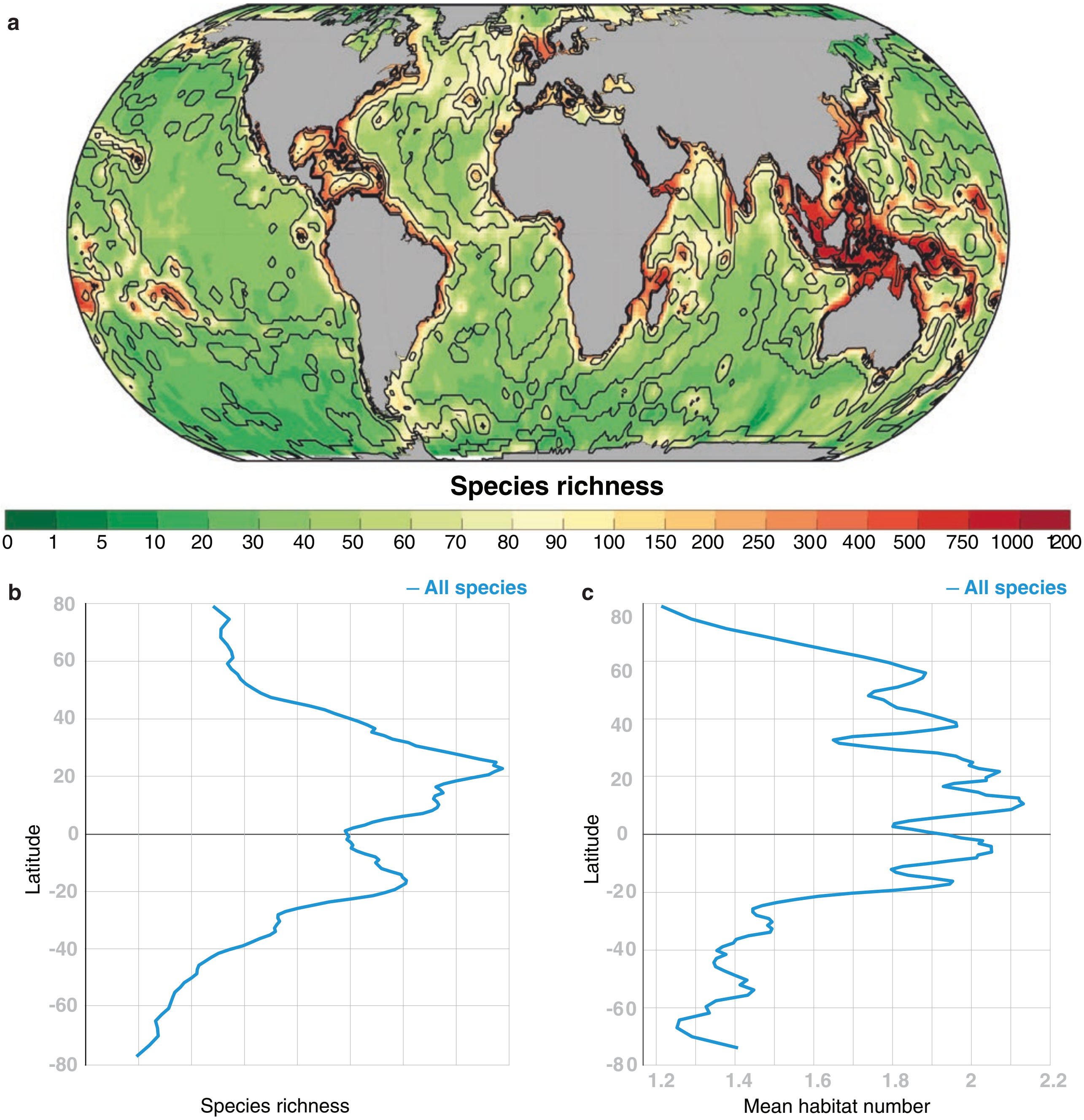

Fig. 10.1 Global patterns of biodiversity and habitat richness. *Notes: Map of species richness* (**a**) *is on a 100 × 100 km equal-area grid with a superimposed contour map of the number of habitats per geographical cell (see Sect. 2.2). Latitudinal gradient of species richness* (**b**) *is of all marine species.* (**c**) *Plot of the average number of habitats versus latitude). (Source: Authors)*

in species biodiversity, particularly for coastal species; but there are steep longitudinal gradients in diversity, with an increase from both east and west towards Southeast Asia, and from east to west in the tropical Atlantic. The Indo-Pacific Coral Triangle, central and western Indian Ocean, Red Sea, South West Pacific Islands (i.e., the Bismarck Archipelago, the Great Sea Reef of Fiji, New Caledonia, New Guinea, the Solomon Islands, Vanuatu) and Southeast Asia show the highest levels of species richness as indicated in previous studies (e.g., Selig et al. 2014). The Caribbean also has a relatively high species richness, but not as high as the aforementioned areas and parts of the northeast Atlantic, such as the North Sea, are as diverse. This latter result may reflect the high number of species records in the northeast

Atlantic, introducing some bias into the overall picture of the distribution of species richness given the exclusion of species with less than 41 samples. Also, small areas, such as tropical or subtropical islands, which are characterized by a high species diversity may be unresolved because of the spatial resolution of this analysis (as for Selig et al. 2014). Individual taxonomic groups and different parts of the ocean (coastal, pelagic, deep sea) can show differing distributions. Taxa that follow the general pattern, albeit with some variation in relative intensity of hot spots, include reef-building corals, coastal fishes, shallow-water ophiuroids (brittle stars), cone snails, mangroves, coastal cephalopods, lobsters and gastropods. Seagrasses have a more temperate-skewed distribution of richness, perhaps reflecting their improved ability to tolerate cold water, relative to reef-building corals and mangroves.

Macroalgae (seaweeds such as kelp) are less well-known in terms of distribution at the species level, but at the genus level again appear to peak at more temperate or subtropical latitudes (Gaines and Lubchenco 1982; Kerswell 2006; Short et al. 2007; Tittensor et al. 2010; Keith et al. 2014; Worm and Tittensor 2018). Coastal sharks show a similar pattern to other coastal fishes, but their distribution is more centered around temperate latitudes (Lucifora et al. 2011). Deviations from the general patterns described include coastal marine mammals, whose endothermy has enabled them to develop a metabolic advantage in colder waters (Pompa et al. 2011; Grady et al. 2019). Pinnipeds (seals, sea-lions and walrus) show an inverse pattern with peak species diversity in subpolar and polar environments (Tittensor et al. 2010; Pompa et al. 2011).

Biodiversity in the open ocean shows a generally bimodal pattern (Chaudhary et al. 2016), with pelagic zooplankton such as foraminifera, copepods and euphausiids, open ocean fishes such as tuna and billfishes, pelagic sharks, and cetaceans all showing a mid-latitudinal peak in species richness, generally between latitudes 30 and 40° (Tittensor et al. 2010). Some differences between these taxa are apparent, including cetaceans being widely distributed in terms of richness peaks across latitudinal bands, whereas pelagic shark hot spots tend to skew towards the coast. Marine bacteria and phytoplankton diversity patterns remain much less well-known at a global scale, though modelling has predicted an intermediate latitude peak in phytoplankton, and there may be a similar gradient in bacteria, though more data and analyses are needed to confirm this for both groups (Worm and Tittensor 2018). Pelagic cephalopods are undersampled, but they appear to show a similar intermediate latitudinal peak, albeit only in the Northern Hemisphere (Tittensor et al. 2010). Pelagic seabirds (such as albatross and petrels) show a mid-latitude peak, but only in the Southern Hemisphere (Davies et al. 2010).

Deep-sea biodiversity is far less known, and whilst regional patterns have been described for multiple groups (Rex and Etter 2010), global patterns are far less well understood at the species level (though model predictions of habitat suitability are available at higher taxonomic levels for other taxa, such as cold-water corals; Tittensor et al. 2009). A global pattern has been described only for the ophiuroids (brittle-stars), which, as mentioned above, show a relatively typical shallowwater pattern of a peak in diversity at low latitudes on the continental shelf and slope, but they have a markedly different distribution in deep waters (more than 2000 m; Woolley et al. 2016). Deep-water ophiuroids show maximum richness at temperate latitudes (between latitudes 30 and 50°), with diversity higher in regions closer to continental margins where particulate organic material export from the surface, used as a food source by most deepsea organisms, is higher. The deep sea is an extremely food-limited, lightless environment, with relatively shallow gradients of temperature over large distances horizontally, and these environmental factors may shape different patterns, though more information is needed to ascertain whether these patterns hold across multiple taxonomic groups.

Biodiversity metrics, other than species richness, that have been assessed at a global scale are few. The global distribution of functional richness in fishes appears similar to species richness, but evenness shows an opposite pattern (increasing with latitude), and functional diversity appears highest in the tropical eastern Pacific (Stuart-Smith et al. 2013). The fish food web is globally connected and suggests a higher vulnerability to species extinctions in the open ocean compared to coastal areas (Albouy et al. 2019).

In summary, known patterns (based on a biased sample of taxonomic groups) indicate that species biodiversity appears to peak in the tropical Indo-Pacific, with a secondary peak in the Caribbean, and a general tropical or subtropical peak in richness. Coastal species tend to match this pattern more closely than oceanic species, which tend to show bimodal peaks at intermediate latitudes; yet whilst deep-sea taxa remain poorly known, one group (brittle stars) shows a markedly different distribution with temperate peaks close to continental margins and in areas of high food export from the surface ocean.

2.2 Habitats

Using previously published spatial data sets (Table 10.1), we synthesised information at the global level to produce patterns of habitat diversity (see Fig. 10.2). Because of their ecological and socio-economic importance, and the relative availability of information, we focused on the following marine habitats ordered from their distance to the coast: estuaries, mangroves, saltmarshes, seagrasses, coral reefs, kelp

Table 10.1 Spatially referenced habitat data for coastal and oceanic ecosystems included in the habitat diversity analysis

Habitat	Time span	Data type	Source
Estuaries	2003	Polygon	Alder (2003)–updated by UNEP-WCMC
Mangroves	1997–2000	Polygon	Giri et al. (2011)–updated by UNEP-WCMC
Saltmarsh	1973–2015	Point	McOwen et al. (2017)–updated by UNEP-WCMC
Seagrasses	1934–2015	Polygon	UNEP-WCMC and Short (2018)
Coral reefs	1954–2018	Polygon	UNEP-WCMC et al. (2018)
Kelp forests	NA	Point	Jorge Assis, research in progress
Shelf valley and canyons	1950–2009	Polygon	Harris et al. (2014)
Cold coral reefs	1915–2014	Point	Freiwald et al. (2017)–updated by UNEP-WCMC
Seamounts and guyots	1950–2009	Polygon	Harris et al. (2014)
Trenches	1950–2009	Polygon	Harris et al. (2014)
Hydrothermal vents	1994–2019	Point	Beaulieu and Szafranski (2018) (InterRidge Vents Database)
Ridges	1950–2009	Polygon	Harris et al. (2014)

Source: Authors

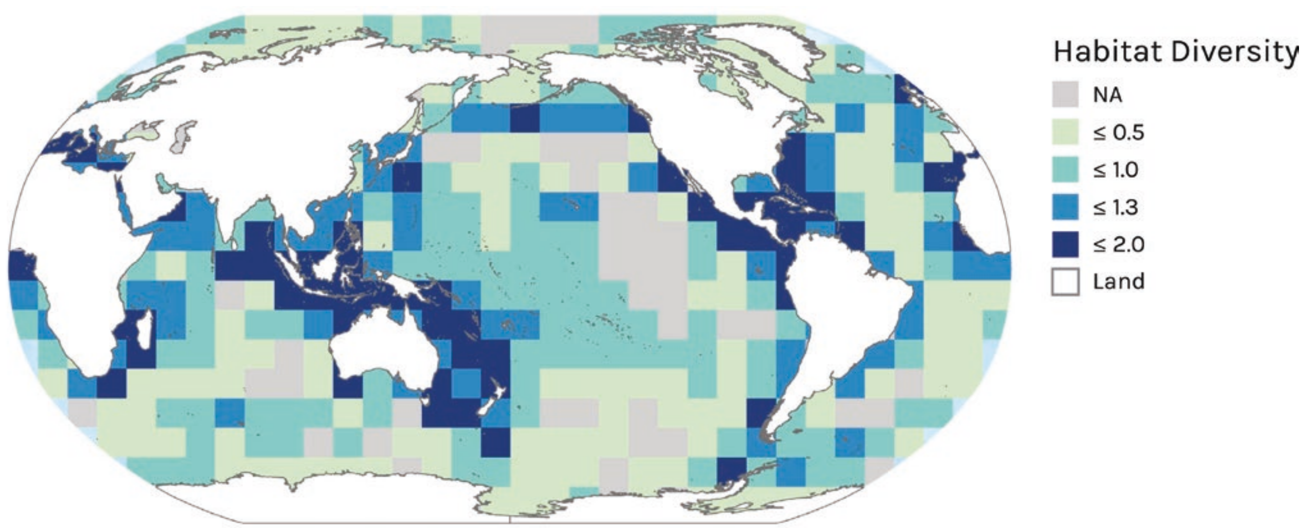

Fig. 10.2 Global habitat diversity. *Note: Habitat diversity calculated with Shannon-Wiener diversity index for habitats studied. Habitat diversity is displayed for 1000-km pixels. (Source: Authors)*

forests, shelf valley and canyons, cold-water corals (deep sea corals), seamounts and guyots, trenches, hydrothermal vents and ridges (Table 10.1).

The global habitat diversity index was based on the 12 habitats in Table 10.1. First, these habitats were converted into binary rasters at a 1-km resolution and projected into the World Robinson projection. A constant raster was created at a resolution of 1000 km by 1000 km. Next, these rasters were imported into R Studio. The packages 'raster', 'sp', 'rgdal', and 'tidyverse' were used to work with the data. Within each cell of the constant raster, the number of 1 km pixels that contained a habitat were summed. Each of the cells of the constant raster was then viewed as a community, and the Shannon Index of diversity was used to calculate a diversity value for each cell using the number of cells of each habitat as species counts. These values were then transformed into a raster and were uploaded into ArcGIS Pro 2.4 to create Fig. 10.2.

Coastal areas had a much higher diversity, because of the occurrence of 6 of the 12 habitats considered. The other 6 habitats occur in deeper waters, where many areas remain understudied. Although our technological capability is increasing through efforts like the global Seabed 2030 mapping project,[3] there are still large gaps in our understanding of deepwater habitat distribution (Rogers et al. 2015). Hence, although the data considered (Table 10.1) are the current best-available representation of the extent of global habitats, the progressive use of improved large-scale mapping technologies will improve our knowledge of global habitat diversity patterns.

Based on the habitat diversity analysis, the Indo-Pacific Coral Triangle, the eastern seaboard of Australia and the Caribbean are hot spots for habitat diversity (Fig. 10.2), a

[3]Information about the Seabed 2030 project can be found at https://seabed2030.gebco.net/

pattern which is similar to that for species diversity (Fig. 10.1). The distribution of these data skews to the right, with fewer areas with higher diversity. The United States, Australia and Indonesia have the highest area of analysed habitats with an average of 6.94%, 5.81% and 5.05% of the global total, respectively. Unsurprisingly, there is a strong and significant correlation with EEZ area, explaining 63% of the variation. Russia, which also has a very large EEZ, does not seem to follow this trend—probably because much of its coastline lies at polar latitudes.

3 Biodiversity Loss

3.1 Evaluating the Loss of Species

The dominant pressures on the ocean are direct exploitation by fisheries, followed by land and sea use change (Costello et al. 2010; IPBES 2019). These pressures were identified by the Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) and by previous studies. Of the three other main drivers considered, invasive species, climate change and pollution are growing in importance. Climate change impacts arise from ocean warming, acidification, deoxygenation, changes in currents and circulation, and sea level rise (IPCC 2019). Temperature rise is correlated with global shifts in distribution, generally away from the tropics but influenced by regional and local oceanography (Cheung et al. 2009; Burrows et al. 2011, 2014; Poloczanska et al. 2013, 2016; Humphries et al. 2015; Molinos et al. 2016). This is driving the large-scale alteration of marine communities at middle to high latitudes (e.g., the Atlantification of the Barents Sea; Fossheim et al. 2015; Oziel et al. 2017; Vihtakari et al. 2018) and may be exacerbated by geographic patterns of thermal tolerance in marine species (Stuart-Smith et al. 2015). Deoxygenation of the ocean has already caused a shift in the vertical and horizontal distribution of pelagic species such as marlins and squid (Stramma et al. 2012; Stewart et al. 2013; reviewed in Breitburg et al. 2018). Climate change is also a significant driver of ecosystem damage, including on coral reefs (Hoegh-Guldberg et al. 2007; Gattuso et al. 2015; Hughes et al. 2018a) and seagrass beds (Thomson et al. 2015; Arias-Ortiz et al. 2018).

To evaluate such impacts on biodiversity, we analysed the IUCN Red List for 12 marine invertebrate and vertebrate taxa. This list comprises analyses of the current status of populations of species with respect to extinction risk, and it considers population decline, negative changes in range (e.g., range of occupancy and/or levels of fragmentation of populations), and whether populations of a species are very small (IUCN 2017). For marine invertebrates and vertebrates, data were extracted from the IUCN online Summary Statistics.[4]

To reduce bias, the assessment was restricted to taxa with more than 10 species assessed. Whilst these taxa represent a relatively small proportion of those living in marine environments, they are the best studied to date; therefore, they present a good (if taxonomically biased) data set on which to assess the threat of extinction and its causes across a range of marine ecosystems (Webb and Mindel 2015). Only around 3% of the roughly 240,000 described marine species have been assessed for the Red List (Sullivan et al. 2019).

3.2 Invertebrates

There are 3081 marine invertebrate species in seven classes across four phyla that have had some representative assessment on the IUCN Red List (see Fig. 10.3 and Table 10.2). The numbers reflect the extremely low level of assessment of marine invertebrates, a total of 2.6% of species across these four phyla, from as low as 0.5% for Arthropoda to 7.5% for Cnidaria (Table 10.2). Furthermore, these samples are biased: 839 species of hard corals (order Scleractinia) and 16 fire corals (genus *Millepora*) make up 97% of the cnidarians assessed, all from a single assessment (Carpenter et al. 2008), and the 686 Cephalopoda species represent 44% of all marine Mollusca assessed but likely less than 1% of all marine Mollusca. By their nature, Red List assessments tend to focus on relatively well-described taxa for both marine and terrestrial species (Webb and Mindel 2015).

With these caveats and the challenge of data deficiency, the proportion of species threatened ranges from a lower bound of 11% to an upper bound of 46%. The most speciose invertebrate classes (Anthozoa, Gastropoda, Malacostraca) as well as the Cephalopoda show the lowest levels of threat. The criteria used for assessment are indicative of marine species characteristics: of the 326 species listed in one of the three 'threatened' categories (vulnerable, endangered, and critically endangered), over 75% (254) are listed on the basis of estimated population decline (Criterion A, for the past, present and/or future), 14% were listed on the basis of small range and decline (Criterion B), and 7% were listed for their very small population size or range (Criterion D). Only 5 species were listed under more than one criterion.

3.3 Vertebrates

Compared to invertebrates, marine vertebrates are relatively well represented in the IUCN Red List (Fig. 10.3). Reptiles, birds and mammals have been fully assessed, and among

[4] See IUCN Red List, https://www.iucnredlist.org/search

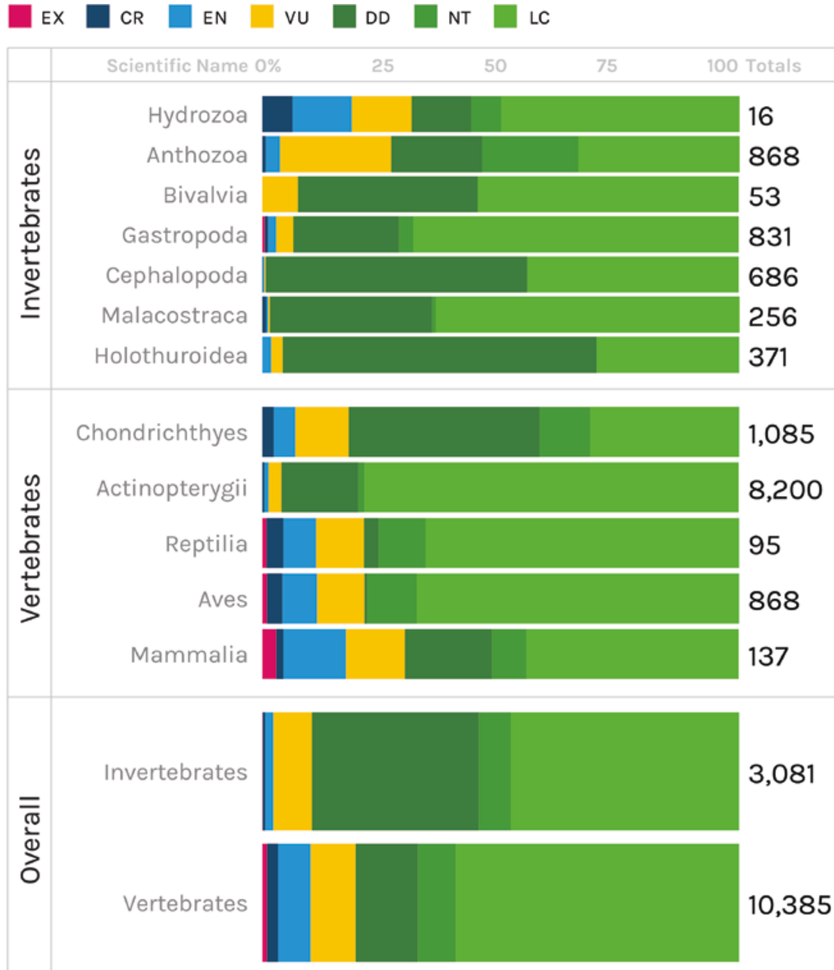

Fig. 10.3 IUCN Red List threat categories for marine species. *Note: These taxa have more than 10 species assessed. Data deficient (DD) species are depicted between the threatened categories (CR critically endangered, EN endangered, VU vulnerable) and non-threatened categories (NT not-threatened, LC least concern). EX extinct in the wild. Numbers on the right of the bars represent the total number of species assessed per taxon group. (Source: Authors)*

Table 10.2 Proportion of invertebrate species assessed on the IUCN Red List compared to the total number of species currently described on the World Register of Marine Species

Phylum	Number of species described	Number of species assessed	% Assessed
Arthropoda	56,479	266	0.5
Cnidaria	11,744	884	7.5
Echinodermata	4408	372	5.0
Mollusca	48,275	1570	3.3
TOTAL	120,906	3092	2.6

Source: WoRMS (n.d.)

marine fishes, of the approximately 18,000 described to date, just over 50% have been assessed (9285 species of sharks, rays and bony fish). Of these, there are 8200 marine actinopterygians, from 30 different orders, for which at least 10 species have been assessed. The two fish classes included in this analysis make up 79% of all assessed marine vertebrates and compose 70% of marine vertebrates listed as threatened. However, the actinopterygians have the lowest overall proportion of threatened species (4%) compared to other marine vertebrate taxa (20–30%). The chondrichthyan extinction risk at this taxonomic level of analysis is substantially higher than for most other vertebrates, and only about one-third of species are considered safe (Dulvy et al. 2014). We note that all species of marine turtles are currently threatened with extinction.

The actinopterygians are less well understood than marine reptiles, birds and mammals, and, as a result, have by far the highest proportion (and number) of species listed as 'data deficient' (DD; see Fig. 10.3); some of these DD species may also be threatened but the lack of data prohibits this assessment from being made.

This situation highlights the poor overall understanding we have of many fish species, even some that are heavily exploited, such as many deepwater and coral reef fishes; examples include the deepwater orange roughy (*Hoplostethus atlanticus*), coral reef groupers and snappers (Epinephelidae and Lutjanidae), coastal and estuarine groups such as croakers (Sciaenidae), and cold-water wolf-fishes (*Anarhichas*). The documentation of these species should be a priority from the perspective of population (status, distribution and trends) and use (i.e., fisheries catches). However, for all taxa there is also a need to collect data on less well-understood aspects of impacts on populations, such as from unintentional catch/

bycatch or through destruction of key life history areas such as spawning or nursery grounds. Such data are collected for some fisheries but by no means all, and data are often aggregated at higher taxonomic levels that render them useless for species-level assessments.

3.4 Drivers of Species Decline

We analysed the identified drivers of extinction risk for species listed as critically endangered, endangered or vulnerable for the 12 groups in Fig. 10.3. This was achieved by looking at each threatened species in the IUCN Red List and recording the drivers of extinction risk. Whilst many of the IUCN drivers of biodiversity decline are relatively straightforward to interpret, the category 'biological resource use' requires some explanation. This refers to the effects that harvesting activities have on the extinction risk, including those caused by targeted catch and bycatch for commercial and artisanal fisheries, the aquarium trade, marine curio trade, shell collecting and traditional medicine. We also note a controversy that began in the 1990s regarding the use of the IUCN extinction threat categories for commercially fished species (Rice and Legacè 2007). The main policy instruments used for fisheries management such as the United Nations Convention on the Law of the Sea (UNCLOS), the United Nations Fish Stocks Agreement and the Code of Conduct for Responsible Fishing (CCRF) by the Food and Agricultural Organization of the United Nations (FAO) all highlight biomass at maximum sustainable yield (B_{msy}) as a target for sustainable fisheries management. Under a sustainable management regime, it is possible to reduce a stock size to below levels which would trigger categorising a species or stock as threatened with extinction under the IUCN Red List criterion of decline in population size while other fisheries management reference points indicate the stock can still be exploited (Rice and Legacè 2007).

Whilst this has been a subject of debate (see Rice and Legacè 2007), more recent studies have demonstrated that conservation metrics as assessed by Red List criteria align well with fisheries assessments of stock status (e.g., Davies and Baum 2012; Fernandes et al. 2017). Thus, it can be concluded that threat categories identified through the Red List criteria do not exaggerate extinction or extirpation risk and occurrences of disagreement between the two approaches are rare (Davies and Baum 2012; Fernandes et al. 2017). The IUCN has specifically identified this issue in the guidelines for applying extinction risk criteria (IUCN 2017).

For invertebrates, the most significant threat for mobile taxa was biological resource use (Fig. 10.4), including overexploitation of populations through directed fishing (Holothuroidea), bycatch (Cephalopoda) or for shell collecting (Gastropoda). For sessile taxa, Anthozoa and Hydrozoa, drivers of extinction risk are evenly distributed amongst multiple drivers, reflecting a range of anthropogenic stressors in coastal ecosystems. The assessed Gastropoda are also predominantly coastal, and this is reflected in the broader range of drivers of extinction risk in this taxon. Other contributing factors to extinction risk included small geographic range (e.g., cone shells; Peters et al. 2013), life history factors (e.g., Cephalopoda, Holothuroidea; Bruckner et al. 2003; Collins and Villanueva 2006) and high commercial value (e.g., Holothuroidea; Purcell et al. 2014). We also note that the first assessment of threat from deep-sea mining has just occurred, with the first of 14 hydrothermal vent invertebrates (a snail) being listed as 'endangered' (Sigwart et al. 2019). This assessment was on the basis of the small geographic range and number of populations of this species, an attribute shared by other vent-endemic taxa. Deep-sea mining is currently controversial, and regulations for environmental management of this activity are still being formulated by the International Seabed Authority (ISA) of the United Nations. Whether these measures will be sufficient to protect vent-endemic species with small ranges from the effects of exploitation of seabed massive sulphides remains to be seen (Durden et al. 2018; Washburn et al. 2019).

Across the marine vertebrate taxa assessed (except birds), the major driver of extinction risk is resource use, including by both small- and large-scale fisheries and both targeted and incidental catch (Fig. 10.4). This is in general agreement with the key messages of the IPBES Global Assessment Report (2019). In particular, larger species at higher trophic levels have been heavily reduced by exploitation whether as high-value target species or because they are taken incidentally as bycatch, and many have shown a sharp decline (Christensen et al. 2014; Suazo et al. 2014; Fernandes et al. 2017). However, the full impacts of incidental catch are little understood for smaller fish species and many invertebrates, because catch data poorly documents them at the species level. Despite little evidence that overexploitation or bycatch have caused global extinctions, local extinctions and commercial extinctions (in which a species is reduced to a level at which it is no longer commercially viable) are much more common (Dulvy et al. 2003). In addition, overexploitation has dramatically reduced the abundance of numerous species worldwide, both large and small (McCauley et al. 2015), caused large range contractions (Worm and Tittensor 2011) and impacted body mass (Ward and Myers 2005). At the ecosystem level, overexploitation has triggered trophic cascades (Worm and Myers 2003; Frank et al. 2005; Daskalov et al. 2007), reduced total community biomass (Ward and Myers 2005) and degraded habitat structure (Thrush and Dayton 2002; Clark et al. 2016). Within species, it has also affected

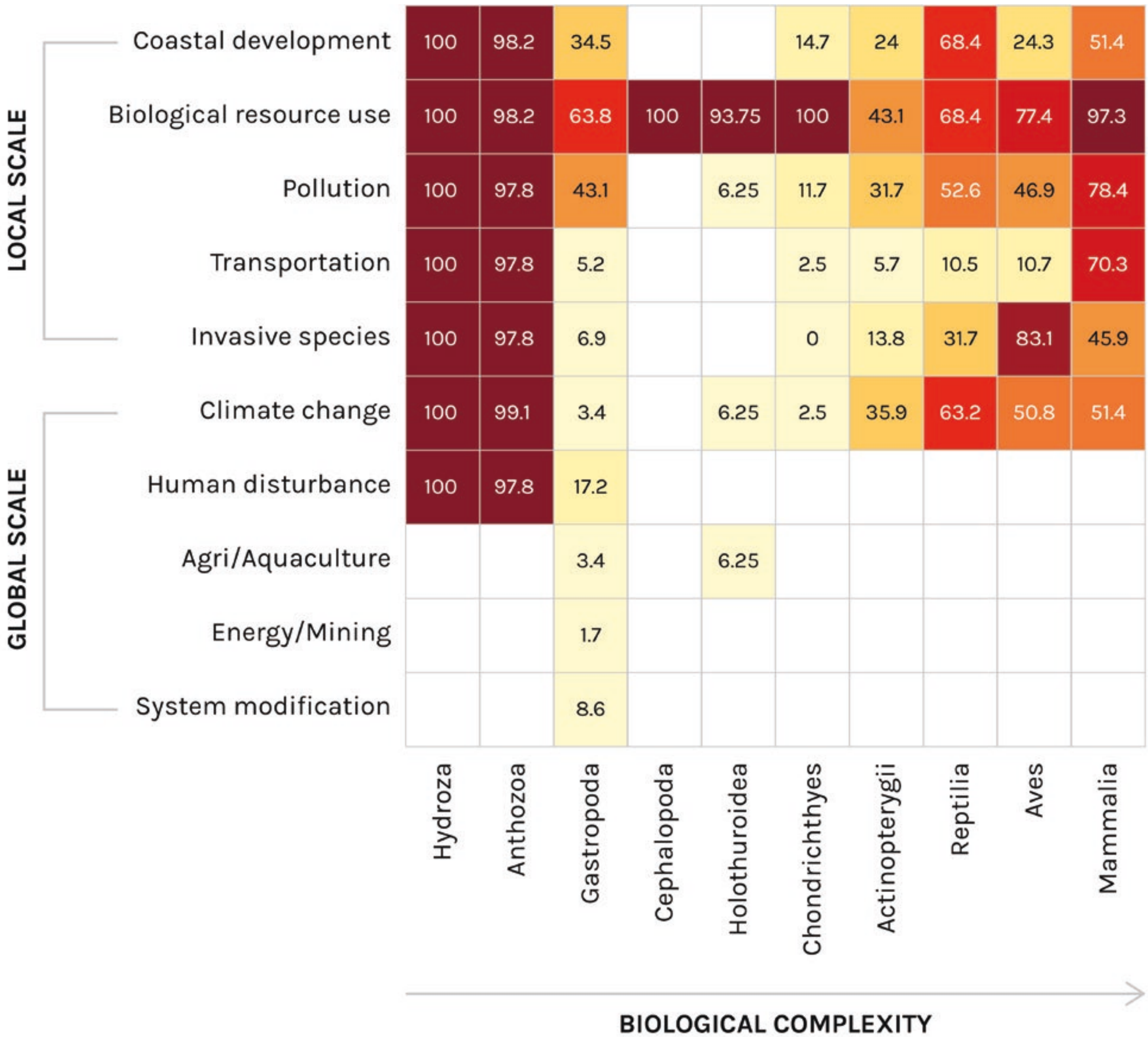

Fig. 10.4 The proportion of the threatened species of each taxon affected by different drivers of extinction risk. *Note: The percentage is reported within each cell. Threatened species out of those assessed for each taxon were: 5 out of 16 Hydrozoa; 226 out of 868 Anthozoa; 58 out of 831 Gastropoda; 5 out of 52 Cephalopoda; 16 out of 371 Holothuroidea; 197 out of 1085 Chondrichthyes; 334 out of 8200 Actinopterygii; 19 out of 95 Reptilia; 177 out of 868 Aves; 37 out of 137 Mammalia. Note that drivers are drawn from the IUCN (2019) Red List. Several drivers are often listed for an individual species. (Source: IUCN Red List)*

genetic diversity and induced evolutionary effects (Pinsky and Palumbi 2014; Heino et al. 2015; Kuparinen and Festa-Bianchet 2017), both of which can potentially reduce the capacity of populations to adapt to threats such as climate change (Blasiak et al. 2020).

A growing number of species are part of high-value consumer markets. As with the Holothuroidea (Purcell et al. 2014), greater rarity pushes their value even higher, which means that they continue to be sourced even if they become more difficult to procure (Courchamp et al. 2006; Sadovy de Mitcheson et al. 2018). Examples of this include shark fins and fish swim bladders, exotic pet species and a range of animals highly valued as luxury food, traditional medicines or ornamentals.

Box 10.2 Fish Spawning Aggregations as Key Biodiversity Areas

To illustrate the importance of key biodiversity areas (KBAs), we selected fish spawning aggregations to contextualise the term "site" in the KBAs,a a seascape unit that (1) can be delimited on maps, (2) encompasses the important habitat used by the species of conservation concern and (3) can actually or potentially be managed as a single unit for conservation. Fish spawning aggregation 'timing' is also part of the context of KBAs. Unlike the conspicuous and betterunderstood breeding colonies of birds and mammals, or the well-known turtle nesting beaches, spawning aggregations of fish are relatively less well-known. But like bird colonies and turtle nesting beaches, they can remain consistent from year to year in time and space and are often appealing targets for fishing because catchability can be particularly high.

Many medium- to large-sized demersal and benthopelagic species in the global ocean form temporary aggregations solely for the purpose of reproduction; these gatherings are the only occasions known for locating a mate and spawning. In the case of tropical groupers (Fig. 10.5a, b) and snappers, many aggregations are highly predictable both spatially and temporally; typically, they form for a week or two over several consecutive months each year. Among temperate species, of the top 25 fishes by weight supplying global fisheries,b many undergo regular spawning migrations, aggregate to spawn for short or extended periods in small or extensive areas, and are exploited at these times. Examples range from Alaska (walleye) pollock (*Theragra chalcogramma*), Atlantic cod (*Gadus morhua*) and Atlantic mackerel (*Scomber scombrus*) to largehead hairtail (*Trichiurus lepturus*) and European pilchard (*Sardina pilchardus*).

Overfishing of spawning aggregations, or of migrations towards these, was a major factor in several fishes declining to threatened status, including the Nassau grouper (*Epinephelus striatus*), the totoaba croaker (*Totoaba macdonaldi*) and the 74 sparid, *Polysteganus undulatus* and other species, none of which were effectively managed prior to declines. Aggregation fishing is likewise implicated for certain populations of orange roughy (*Hoplostethus atlanticus*) (Fig. 10.5c), barred sand bass (*Paralabrax nebulifer*) and large yellow croaker (*Larimichthyes crocea*).[c] Spatial concentration from spawning was also identified by fuzzy logic as an intrinsic extinction vulnerability factor in marine fishes.[d] A global assessment of the known status of 948 spawning aggregations (mainly reef fishes) shows that 26% are decreasing (as determined by reduced catches or underwater visual census counts), 13.5% are unchanged and 3–4%, each, are either increasing or have disappeared entirely; the remaining 53% are of unknown status (Fig. 10.6). These aggregations occur in the global ocean, in over 50 countries, in almost 50 families and in more than 300 fish species.

As productivity hot spots that support a massive proportion of fish biomass, spawning aggregations are key components of the marine ecosystem. Because they are particularly vulnerable to fishing—yet are important to fisheries—they need more conservation and management attention than they have attracted to date, especially from spatial and/or seasonal protective measures.[e] Although conventional management controls may be used for aggregating species—such as minimum sizes, fishing effort or gear controls—and assessments consider maximum sustainable yield or recruitment overfishing, the spawning aggregations themselves are not often explicitly the focus of management, partly because they are so appealing to target. Their management, for example, is not included as a criterion in the Marine Stewardship Council fishery assessment Principle 1, except in relation to habitat protection or access to spawning grounds. However, given issues such as hyperstability and possible depensatory effects at low population levels associated with assessing and managing exploited aggregating species, a specific focus on protecting spawning fish deserves higher priority and special management consideration, especially for species forming large aggregations.[f] On the other hand, well-managed spawning aggregations can support valuable fisheries and contribute to food security as well as conserving biodiversity.

Sources:

[a] Edgar et al. (2008)

[b] FAO (2018)

[c] Sadovy de Mitcheson (2016)

[d] Cheung et al. (2005)

[e] Erisman et al. (2015); Sadovy de Mitcheson (2016)

[f] van Overzee and Rijnsdorp (2015); Sadovy de Mitcheson (2016)

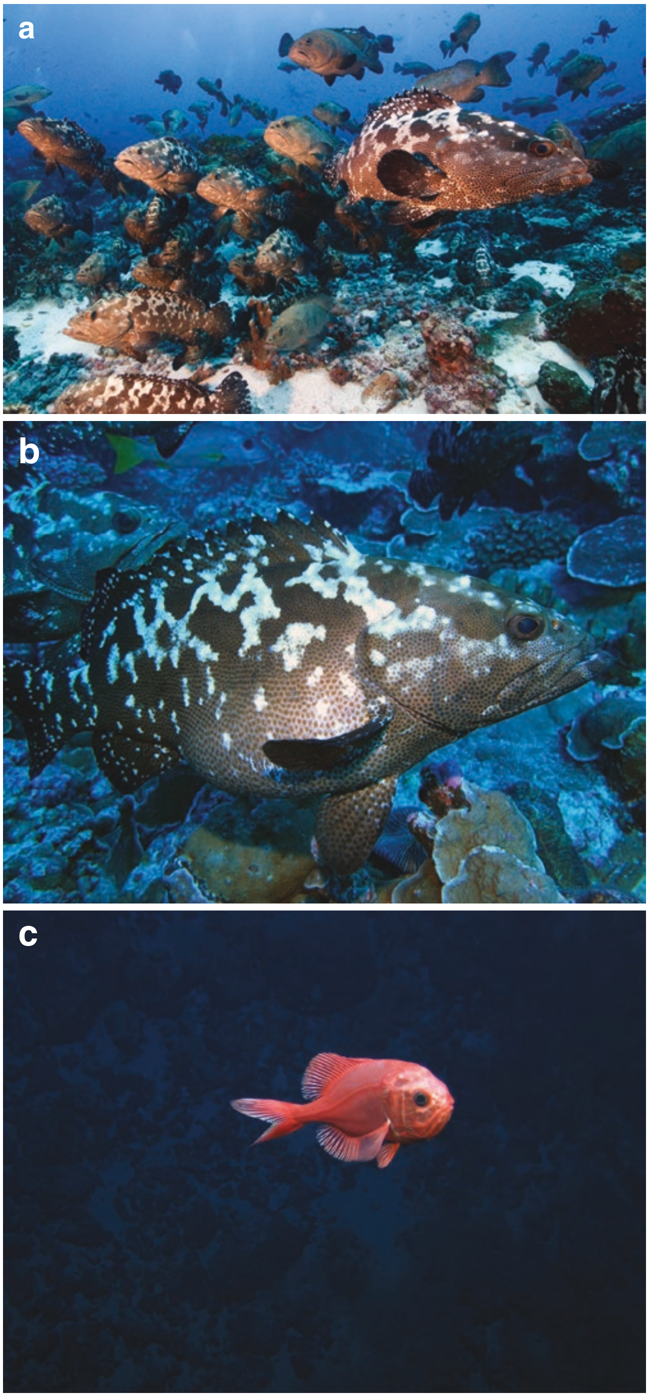

Fig. 10.5 (**a**) Spawning aggregation of the camouflage grouper, *Epinephelus polyphekadion* in French Polynesia. (Photo © Yvonne Sadovy-Micheson). (**b**) Gravid female camouflage grouper at spawning site. (Photo © Stan Shea). (**c**) Orange roughy, *Hoplostethus atlanticus*, a deep-sea species which aggregates around the summits and upper flanks of seamounts for spawning when it is targeted for fishing. (Photo © IUCN Seamounts Project, AD Rogers)

Loss or compromise of key biodiversity areas (such as key egg-laying, nesting, pupping or mating grounds) can quickly reduce populations (see Box 10.2). The finding that biological resource use is the number-one driver of species decline, both in this study and in the IPBES Global Assessment Report (2019), suggests that Aichi Biodiversity Target 6[5] of the Strategic Plan for Biodiversity 2011–2020 has not been attained across the fisheries sector. This is a surprise considering the reported stabilisation and rebuilding of many fish stocks resulting from improved management practices in recent decades (Fernandes and Cook 2013; Hilborn and Ovando 2014; Fernandes et al. 2017; Hilborn et al. 2020). Findings of stabilisation of fisheries are also in contrast to observations that the overall trend, globally, is one of increased overfishing (Pauly and Zeller 2016; FAO 2018). One explanation of the global trends of fisheries declines is the massive increase in the size of the global fishing fleet from 1950 to the present (2015 figures) from 1.7 to 3.7 million vessels (Rousseau et al. 2019). As a result of improving technology (e.g., vessel power) over this period, fishing effort has increased almost exponentially, and catch per unit effort (CPUE) has declined exponentially (Rousseau et al. 2019). The catches from artisanal fishing fleets are often not reported in official government figures, and yet globally the total power levels of these fishing fleets are comparable to those of industrial fishing fleets; they are also less well managed (see below; Rousseau et al. 2019). Asian fishing fleets, in particular, have increased dramatically in both numbers of vessels and fishing power (Rousseau et al. 2019).

Fishing fleets in Europe and North America were reduced in the 2010s, and evidence suggests that it is in these regions CPUEs have stabilised and the decline has also decreased in Oceania as a result of improved fisheries management (Rousseau et al. 2019). Despite a continued increase in overfishing and the decline in CPUEs, global fishing fleets have continued to increase in size and power (Rousseau et al. 2019). If past trends continue, a million more motorized vessels could appear in global marine fisheries in the coming decades.

Both small-scale fisheries and those undertaken by developing states are performing worse than those of developed

[5] By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem-based approaches, so that (i) overfishing is avoided, (ii) recovery plans and measures are in place for all depleted species, (iii) fisheries have no significant adverse impact on threatened species and vulnerable ecosystems, and (iv) the impacts of fisheries on stocks, species and ecosystems are within safe biological limits.

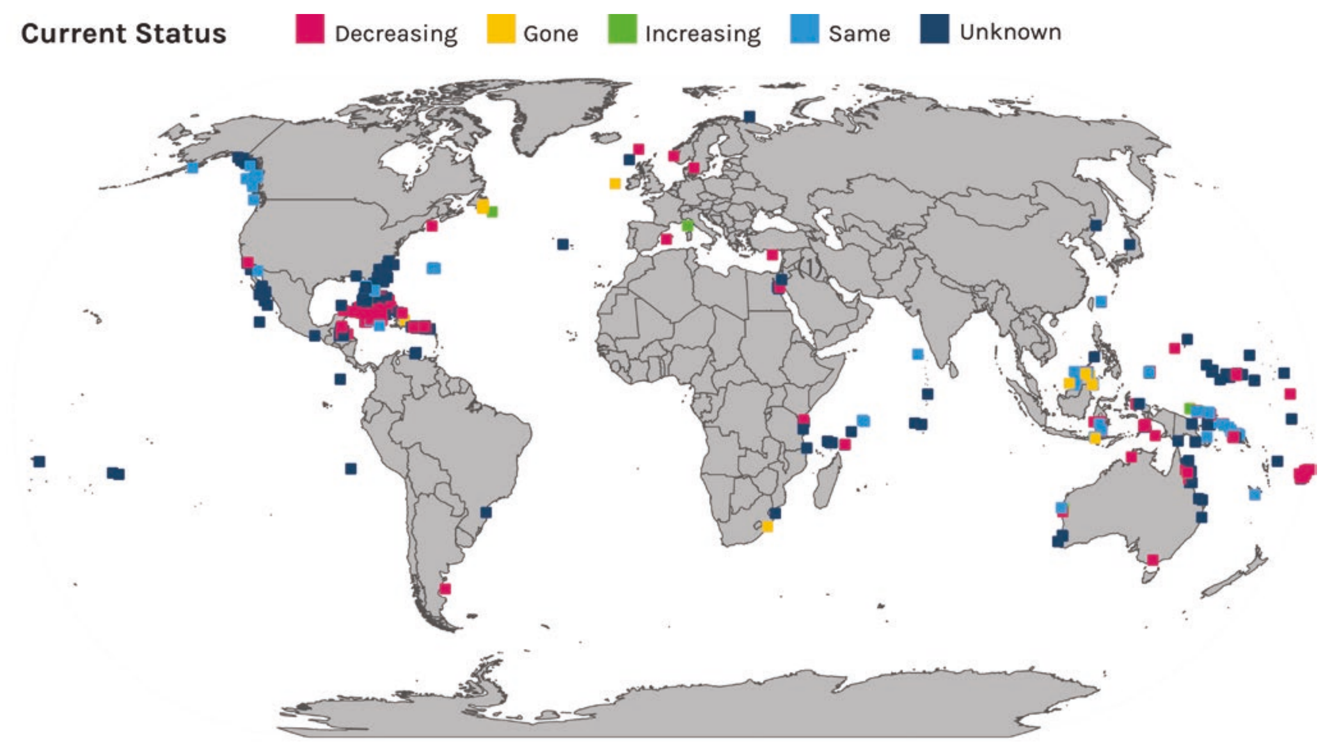

Fig. 10.6 Proportion of invertebrate species assessed on the IUCN Red List compared to the total number of species currently described on the World Register of Marine Species. *Note: A total of 948 documented spawning aggregations are shown. The database is weighted towards tropical reef fish species and underrepresents non-reef and temperate or polar regions.* (*Source: Science and Conservation of Fish Aggregations (database),* https://www.SCRFA.org. *Accessed 14 July 2019*)

states (Hilborn and Ovando 2014; Ye and Gutierrez 2017). A conservative estimate that 23% of global fish catch comes from unassessed fisheries indicates that the lack of data gathering is a significant barrier to sustainable management of target and non- target (bycatch) species (Costello et al. 2012; Gilman et al. 2014; Rousseau et al. 2019). Unassessed fisheries perform poorly in terms of sustainable management compared to those which are subject to scientific stock assessment (Hilborn and Ovando 2014). A large proportion (though not all) of the unassessed fisheries are small, mostly coastal and often artisanal, and many of them are located in the developing world. The costs of scientific fisheries assessments are high and therefore may be uneconomical for implementation in small fisheries, particularly for developing coastal states. In such cases, methods for data-poor fisheries assessment—which rely on broader life history characteristics and/or catch trends, including catch-per-unit-effort estimates—may be a more cost-effective and practical means of management (Hilborn and Ovando 2014), although less reliable (Edgar et al. 2019). Ecosystem-based fisheries management may also be appropriate for small-scale, multispecies fisheries but there is a challenge between the need for complex data with that of practical implementation (Hilborn and Ovando 2014).

Studies that have found standards of fisheries management to be generally poor amongst coastal states with many fisheries exhibiting overcapacity, capacity-enhancing subsidies, problems with foreign access agreements and issues around the transparency of management and decision-making, show that such problems are worse within developing states (Mora et al. 2009; Pitcher et al. 2009). This emphasises the lack of capacity to manage fisheries in these countries (Pitcher et al. 2009; Hilborn and Ovando 2014; Ye and Gutierrez 2017). This situation is magnified because developed countries either import fish from other regions of the world or establish fisheries partnership agreements, effectively externalising their costs for fisheries management (Ye and Gutierrez 2017). As with smallscale fisheries, investment in management methods that are appropriate for developing countries are needed to establish more even standards for global fisheries sustainability. However, this may need reciprocal arrangements between developed and developing countries, especially where the former benefit from the fisheries resources of the latter, to enhance fisheries management capacity through finance, training and technology transfer (Ye and Gutierrez 2017). Seafood trading mechanisms that promote sustainability may also be useful for addressing the management of fisheries in developing countries. Carrot-and-stick approaches may be useful as well, such as marketbased measures (e.g., certification or eco-labelling) which promote sustainable fishing or impose import restrictions on overfished stocks (Ye and Gutierrez 2017).

We also point out that overfishing is by no means restricted to developing states, and a cursory examination of the literature indicates that even in the waters of regions such as Europe, a significant number of stocks are in decline or are overfished, especially smaller stocks (Fernandes et al. 2017). Studies of fisheries sustainability also often neglect to acknowledge that even modern fish stock assessment methods have levels of uncertainty associated with them and relatively few use, or are validated by, fisheries independent data (Edgar et al. 2019). Improvements in catch efficiency in fleets may also be difficult to represent in stock assessments (Edgar et al. 2019). An increasing issue is also that stock assessments are often based on historical assessments when current climate change means that the environment is changing rapidly and such data may not reflect alterations in stock dynamics or distribution (Edgar et al. 2019). Stock assessments also concentrate on management of single species or stocks, ignoring interspecies interactions (e.g., with predators and prey) and other aspects of ecosystem structure, function and health (Edgar et al. 2019).

There have been increasing measures to incorporate biodiversity considerations into fisheries management (Garcia 2010; Rice and Ridgeway 2010; Friedman et al. 2018). These measures can be seen as part of a broader shift in societal views on the use of natural resources from one of straightforward economic exploitation to one of sustainable development whereby the use of ecosystem goods and services must be traded off against the resilience of the environment (Garcia 2010; Friedman et al. 2018). These concepts were introduced into the arena of resource management following World War II, but they were significantly strengthened through the adoption of the World Conservation Strategy in the 1980s, the outcomes of the United Nations Conference on Environment and Development and the Brundtland Commission (1983–87), culminating in the CBD which entered into force in 1993 (Friedman et al. 2018). UNCLOS and the subsequent 1995 Fish Stocks Agreement both included specific provisions with respect to sustainability of both target fish stocks and the wider ecosystem. At this point, states began to incorporate increasing measures to address sustainability and to decrease the environmental impact of fishing. These measures have been reviewed on a regular basis through the United Nations General Assembly, and biodiversity considerations have been gradually mainstreamed in fisheries management through a variety of voluntary agreements and measures by the FAO (e.g., the CCRF; international plans of action to reduce fishing impacts on sharks, seabirds and turtles; see Friedman et al. 2018 for a more comprehensive list). Likewise, the fisheries management and environmental sectors have increased their collaboration to improve the environmental performance of fisheries (Friedman et al. 2018). However, given the impact on extinction risk in marine species (this study and the IPBES Global Assessment Report 2019), there is clearly a long way to go in improving the environmental sustainability of marine capture fisheries. It is also notable that reducing overfishing would in itself reduce impacts on threatened species affected by bycatch (e.g., mammals, seabirds and turtles; Burgess et al. 2018).

Uneven implementation at the global level is also an issue with measures to conserve biodiversity from the destructive effects of fishing. For example, the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), which manages fisheries in the Southern Ocean, has worked with the nongovernmental organisation (NGO) Birdlife International to massively reduce interactions (often fatal) of albatrosses and petrels with longline fishing in the region by 67,000 per annum (Friedman et al. 2018). However, at present it is estimated that seabird bycatch in longline fisheries globally range from an average of 160,000 to an upper range of 320,000 per annum and is a major driver of the decline of albatrosses and petrels (Anderson et al. 2011; Dias et al. 2019). Technical measures for longline fishing, including setting lines at night, are known to decrease bycatch and have been successful at reducing this source of mortality in albatrosses and petrels in areas of the Southern Ocean such as South Georgia (Anderson et al. 2011; Phillips et al. 2016). Yet recent analysis of the behaviour of pelagic longline fishing vessels in the southern Atlantic, Indian and Pacific Oceans indicate that less than 5% of vessels may be complying with requirements south of latitude 25° south by setting in the daytime (Winnard et al. 2018). We point out that obtaining data on fisheries bycatch is problematic for many fisheries, especially on the high seas and where observer coverage is low and reporting mechanisms are weak (Anderson et al. 2011; Gilman et al. 2014; Phillips et al. 2016), while the impact of purse-seine fisheries, such as for forage fish, have not been properly evaluated.

Another example of uneven implementation of actions to conserve biodiversity has been in the uptake of the FAO's International Guidelines for the Management of Deep-Sea Fisheries in the High Seas (FAO 2008). These guidelines were established to protect vulnerable marine ecosystems (VMEs), such as deep-sea cold-water coral reefs and seamounts, from the impacts of bottom trawling as well as to improve the management of low-productivity deepwater fisheries. The guidelines have resulted in significant actions to protect biodiversity by regional fisheries management organisations (RFMOs) or agreements through the use of spatial conservation measures, gear restrictions and encounter rules, which require a vessel to move away from an area where VMEs are encountered and to report the encounter (Rogers and Gianni 2010; Wright et al. 2015; Bell et al.

2019). There have also been efforts to implement biodiversity conservation measures using RFMOs and Regional Seas Agreements to implement biodiversity conservation measures (Friedman et al. 2018).

A good example is the action by the North-East Atlantic Fisheries Commission (NEAFC) and the Oslo Paris (OSPAR) Commission to initiate MPAs in areas beyond national jurisdiction, such as the Mid-Atlantic Ridge (Wright et al. 2015). The collaboration between the NEAFC and the Oslo Paris (OSPAR) Commission was formalised first through a memorandum of understanding (NEAFC/OSPAR 2008) and then through a collective arrangement (NEAFC/OSPAR 2014). However, implementation of the FAO guidelines has progressed much more slowly and unevenly with other RFMOs and agreements (Rogers and Gianni 2010; Wright et al. 2015) with some showing poor progress even to the present (Bell et al. 2019). In some cases, this seems to be linked to a lack of capacity and financial support to achieve a better performance of fisheries in areas beyond national jurisdiction in terms of sustainability of stocks and protection of biodiversity (Bell et al. 2019).

For birds, the major threats are invasive species for breeding colonies and unintentional bycatch at sea, with the latter being solely responsible for many species becoming threatened (Paleczny et al. 2015; Dias et al. 2019). For mammals, it is notable that transportation corridors are a major threat given the increasing impacts of ship strikes on cetacean populations (Ritter and Panigada 2019). Climate change and extreme weather are also significant threats for four of the five vertebrate groups assessed. Additional threats include coastal activities such as residential and commercial development and pollution.

Box 10.3 The Global Risk to Marine Biodiversity

In order to estimate the patterns of global risk to biodiversity, we overlaid spatial data on human impacts from Halpern et al. (2008) onto the data on species diversity used to generate Fig. 10.1. Human impact index data were regridded on a 110-by-110-km equal area grid and overlaid with the species richness data (Fig. 10.7a). The relationship between species richness and the corresponding human impact index was assessed by computing the centroid of the relationship in a log-log dimension (Fig. 10.7b). Based on the position of the geographical cell, we established four categories: high richness and high impact in red, low richness and high impact in violet, high richness and low impact in green and, finally, low richness and low impact in blue. Then the Euclidean distance among each geographical cell to the centroid of each category was computed, and the shades of colour in Fig. 10.7b represent these distance intervals.

The multitude of impacts from human society on the ocean have been summarised at a global level, showing alteration of all marine ecosystems.[a] The examination of the relationship between biodiversity[b] and anthropogenic pressures[c] (Fig. 10.7a, b), reveal four different scenarios:

1. Regions where the level of biodiversity and human impact are very high include the Indo-Pacific Coral Triangle; Southeast Asia, including the seas off Thailand, China and Korea; northern Australia; the western Indian Ocean; the Mediterranean; the coasts of northern Europe (North Sea); and some western Pacific Islands. Although this analysis specifically aimed to reduce sampling bias, the levels of sampling for species from different regions of the ocean vary dramatically. Therefore, it is likely that sampling bias has resulted in some areas being classified as having a high biodiversity as a result of intense sampling rather than in having a high inherent species richness; the North Sea is the most obvious example. Some areas have been identified as high impact with a high diversity in other studies (e.g., Indo-Pacific Coral Triangle, northern Australia, some of the western Pacific Islands, areas of the Indian Ocean).[d] In some cases, they are also in locations where there is a rapid increase in human pressures (e.g., Australia and parts of the Indo-Pacific Coral Triangle).[e] The explanation for some areas of the ocean having high levels of diversity and impact vary. In some cases, there is a high coastal population and/or high levels of direct (e.g., fisheries) and indirect (e.g., pollution) exploitation of coastal and offshore marine ecosystems. These waters include those of both developed and developing coastal states.
2. Areas where human pressures and biodiversity are moderately high include the central Indian Ocean and Caribbean, the eastern seaboard of the United States and Canada, and the western coast of the United States as well as northern Brazil. Some of these areas have been identified as high impact with a high diversity in other studies (i.e., the Caribbean and parts of the Indian Ocean).[f] The recent rapid increase in human pressures has also been observed for the coast of Brazil.[g]
3. Areas of high biodiversity and low human pressure include some of the islands in the western and central tropical Pacific, parts of Hawaii, the Galapagos Islands, the Seychelles and areas of the open ocean, Russian Arctic and Alaska. Some regions with a high diversity and a low level of human threat include those in which

significant fully or highly protected MPAs have been established and have reduced pressures as well as being relatively remote (e.g., Kiribati and the Galapagos Islands).

4. We note that there is also a lack of areas which have a lower diversity which are highly impacted (i.e., points in the lower right quadrant of Fig. 10.7b). This may be explained by relatively low observed impacts in polar and open ocean ecosystems, regions with a lower diversity than the tropics and coastal ecosystems. Lack of data on human impacts may be a factor here.

In conclusion, more than half of the ocean is considered to be heavily perturbed by human activities; this includes more than half of the hot spots of marine species richness.

Sources:

[a] Halpern et al. (2008, 2019)
[b] Reygondeau (2019)
[c] Halpern et al. (2019)
[d] Jenkins and Van Houten (2016)
[e] Halpern et al. (2019)
[f] Jenkins and Van Houten (2016)
[g] Halpern et al. (2019)

Climate change, especially increasing temperature and habitat impacts, is predicted to become an increasing threat to many invertebrate and vertebrate species (IPCC 2019), but there are uncertainties in terms of projections. The upper thermal tolerance limits for shallow tropical reef-building corals have been exceeded in multiple global stress events from 1998 to 2017 (Hughes et al. 2018a; Stuart-Smith et al. 2015), resulting in large-scale coral loss, local and regional scale shifts in species composition and ultimately reef function. This impacts ecosystem function and the provisioning of ecosystem services (Hughes et al. 2017), and as waters warm, such thermal limits will be more frequently exceeded. There is already evidence that reproductive synchrony in broadcast-spawning corals is breaking down (Shlesinger and Loya 2019), and in fish species, spawning times could be radically affected; these are often temperature-associated changes, and they may impact reproductive success (Asch and Erisman 2018). Some fish appear to go deeper, tracking cooler waters in warming seas, illustrating the rapid responses of marine life to ocean warming (Burrows et al. 2019). It is also stressed here again that the taxa assessed for the IUCN Red List are a biased sample often focusing on those which are heavily exploited (e.g., the Holothuroidea for the invertebrates). Many groups of organisms, especially poorly known invertebrates, are likely to be significantly impacted by climate change either directly as their environmental tolerances are exceeded or indirectly as their habitat is destroyed. The overall impacts of climate change on marine biodiversity is therefore likely to be currently underestimated.

Particularly in the coastal zone, development and pollution, which are often connected, have been the other major drivers of species declines. As with the lack of information on small-scale fisheries, it is notable that the monitoring of biodiversity within the waters of coastal states is weak despite it being a requirement in several of the conventions and agreements reviewed in this report. An indicator of this is the number of species categorised as DD in Red List assessments (see Fig. 10.3). There is overwhelming evidence from a broad range of taxa that loss of habitats formed by foundation species, including corals, mangrove forests, seagrass beds, saltmarshes and kelp forests, continues unabated in many regions of the world (see Sect. 3.5), despite specific agreements or conventions which are aimed at conserving such ecosystems (e.g., the Ramsar Convention on Wetlands).

In summary, biodiversity impacts in the ocean have generally manifested as population declines, habitat degradation and loss, and ecosystem-level changes rather than as global extinctions. Although overexploitation has been the primary driver of loss to date for many groups, it is notable that habitat destruction through coastal development and pollution are major contributors to species being added to the Red List's threatened categories. Climate change impacts are expected to grow in the future.

Although few marine extinctions have been observed (Dulvy et al. 2003), in the best-assessed groups of marine species around 11–46% are judged to be at risk of extinction, a range that spans the proportion of threatened terrestrial species in well-assessed groups (20–25%; Webb and Mindel 2015) with individual groups falling above and below this range. Global extinctions in the marine environment are relatively rarely documented (Dulvy et al. 2003; McCauley et al. 2015), and trends in the species richness of local communities can be relatively flat, though with turnover in species composition (Dornelas et al. 2014). OBIS currently holds over 50 million occurrence records of 125,000 marine species; about half of the total number of marine species described to date according to the World Register of Marine Species (WoRMS). Given this, extinction rates in marine species may be higher than previously estimated because they have simply not been documented.

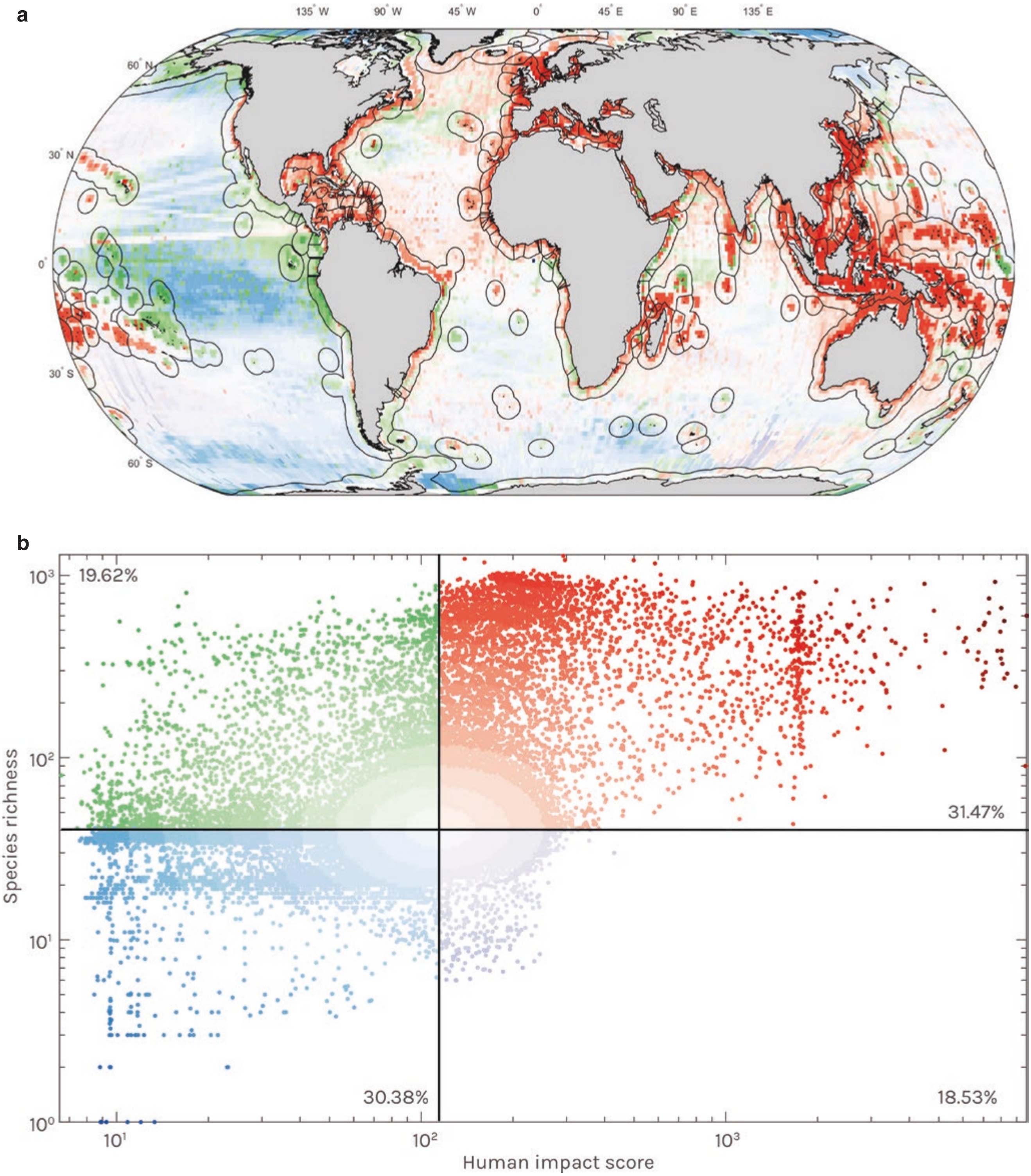

Fig. 10.7 Marine biodiversity in relation to human impacts. *Note: Map* (**a**) *and scatter plot* (**b**) *of the relationship between marine biodiversity and the human impact score. Each quadrant has been computed based on the centroid of the relation in a log-log dimension. Colour shades are computed as the Euclideian distance of the geographical pixel from the centroid of the relation. (Sources: Based on Halpern et al.* 2008 *and Reygondeau* 2019*)*

3.5 Habitat Degradation and Its Drivers

The IPBES Global Assessment Report (2019) summarised key threats to the ocean. Overall, 66% of the ocean is experiencing increasing cumulative impacts (Halpern et al. 2015). The area of ocean still classified as 'wilderness', characterised by having a low impact across a range of anthropogenic stressors, is as low as 13% (Jones et al. 2018), and the area with no discernible human footprint is down to 3% (Halpern et al. 2015). Seagrass meadows decreased in extent by over 10% per decade from 1970 to 2000, the global cover of mangroves has declined about 40% (Thomas et al. 2017) and that of saltmarshes by an estimated 60% (Gedan et al. 2009). Regionally, kelp forests have also shown significant declines in distribution, such as in the Great Southern Reef area of Australia, where they are associated with a high level of endemism (species restricted to a specific geographic location) (Bennett et al. 2016). However, kelp forests are highly dynamic ecosystems, and globally the picture is more complicated; whereas in some areas no trends are apparent, in others, kelp forests are extending their range (Krumhansl et al. 2016).

The role of coral reefs as a flagship ecosystem is characterised by their high biodiversity (Fisher et al. 2015) and their benefits to people (Wilkinson et al. 2016; IPBES 2019). Coral reefs have lost half of their live coral cover since the 1870s, and losses have accelerated over the last two to three decades as a result of the direct effects of climate change (Wilkinson et al. 2016; IPCC 2019) and the indirect effects of other drivers, such as predator outbreaks or disease epidemics, some of which are exacerbated by climate change (Wilkinson et al. 2016; IPBES 2019, BG 4, 5).

Projections for coral reef loss—even at the most optimistic climate change scenarios—are dire: corals could be reduced to 10–30% of their former abundance at warming of 1.5°C, and they could be reduced to only 1% at 2 °C (IPCC 2018). Estimates of coral loss generally conflate loss of cover with loss of reefs. Most reefs will endure, but coral cover on them will decline.

Marine habitats have experienced significant reductions in area in the past century. Coastal reclamation and land-use change, together with pollution and, more recently, climate change, have led to the vast loss of many valuable coastal habitats, estimated at an average of 30–50% (Pandolfi et al. 2003; Polidoro et al. 2010; Waycott et al. 2009; Barbier 2017; Duarte et al. 2020). The first large-scale loss of coastal habitats was documented in China more than a millennium ago and in Europe around the 14th century, when seawalls were built to prevent tidal inundations and to transform saltmarshes into agricultural land (Loke et al. 2019). Such coastal development sprawls throughout much of the world, leading to significant saltmarsh losses in Europe, the United States, Canada and Asia. In China, for instance, more than 60% of the coastline is now artificial (Liu et al. 2018). Land reclamation and conversion to aquaculture ponds and rice paddies has led to much of the observed mangrove loss (Richards and Friess 2016).

Eutrophication and physical impacts, such as dredging, are responsible for much of global seagrass losses (Waycott et al. 2009). It is important to note that as well as causing the loss of ecosystems such as mangroves and saltmarshes, coastal engineering can also prevent such ecosystems from adapting to climate change by preventing the landward migration of such habitats as sea level rises which is known as transgression (Hughes 2004; Alongi 2015; Lovelock et al. 2015).

The first losses of coral reefs were driven by siltation derived from the deforestation of tropical watersheds, overfishing and reduced water quality from sewage and excess nutrient inputs from agricultural land (Pandolfi et al. 2003; MacNeil et al. 2019; Williams et al. 2019). Recent global bleaching events, driven by El Niño warming events exacerbated by anthropogenic ocean warming (Hughes et al. 2017, 2018a, b; Claar et al. 2018; Lough et al. 2018), have now emerged as a major driver of present, and future, coral loss. Under such a multiplicity of detrimental anthropogenic stressors, coral reefs have a tendency to convert to alternative stable states, such as dominance by fleshy algae or cyanobacterial mats (Ford et al. 2018a). This can be associated not only with loss of positive ecosystem services, such as coastal protection or fisheries, but also the potential for negative impacts on coastal human communities (e.g., an elevated risk of ciguatera or ciguatera-like diseases; Ford et al. 2018a).

Upwelling regions, where most of the fisheries for forage fish are located, have also been degraded by overfishing. This results in food chain alterations and the risk of trophic structure breakdown, particularly when small pelagic fish—which are the link between primary producers and the secondary consumers in the typical wasp-waist trophic structure—are removed from the food web (Cury et al. 2000). Such examples have already been observed affecting top predators and lower trophic levels (Velarde et al. 2015a, b).

Overfishing of small pelagic or forage fish results in increased population fluctuations (Cisneros-Mata et al. 1996; Hsieh et al. 2006) and reduces their resilience to natural environmental periodic changes such as the El Niño Southern Oscillation and the Pacific Decadal Oscillation, rendering these forage fish populations more vulnerable to these natural variations, risking their final collapse. Furthermore, more than one million square kilometres (km^2) of the seabed are subject to bottom trawling each year (about 14% of the total trawlable area of 7.8 million km^2 which lies shallower than 1000 m depth; Amoroso et al. 2018). This degrades seabed communities through physical impact, affecting biodiversity and ecosystem function (Thrush and Dayton 2002; Pusceddu et al. 2014; Ashford et al. 2018) and significantly alters ecosystem processes such as sedimentation at large scales (Puig et al. 2012; Pusceddu et al. 2014). Deep-sea ecosystems can be especially vulnerable to the effects of fishing. Seafloor ecosystems are fragile and have low resilience (Clark et al. 2016; Rogers 2018) and the targeting of deep-sea fish species and the effects of bycatch have been observed to rapidly overexploit stocks (Norse et al. 2012; Victorero et al. 2018). The deep sea is increasingly contaminated with litter (Pham et al. 2014; Woodall et al. 2015) and in the near future, it will experience increased temperatures, stratification, decreased oxygen concentrations, and ocean acidification (Rogers 2015; Sweetman et al. 2017). The increasing demand for raw metals and minerals, coupled with the depletion of terrestrial resources, is making deep-sea mining more attractive economically (Petersen et al. 2016; Miller et al. 2018). The impacts of this industry are likely to be extremely severe (Niner et al. 2018).

3.6 Reducing the Provisioning of Ecosystem Services

Biodiversity plays a significant role in ecosystem functioning, which underpins nature's contribution to people (NCP). The concept of NCP is elaborated in the IPBES Global Assessment Report (2019), as the positive and negative contributions of living nature to people's lives. Here, we focus specifically on positive ecosystem services, 'the benefits people obtain from ecosystems' (MA 2005), a subset of NCP. This is because we focus on the potential negative consequences of biodiversity loss in the ocean, and the positive provisioning of ecosystem services has been widely discussed in the context of the marine environment. The benefits of ecosystem services include provisioning services; the production of goods and materials such as food, raw materials and pharmaceuticals; regulatory services; the control of climate, atmosphere and other aspects of the environment that maintain the Earth system; supporting services; those that enable the provision of direct and indirect ecosystem services to humankind; and cultural services, including recreation, tourism, inspiration for art, culture, spiritual experience and cognitive development (de Groot et al. 2012; Costanza et al. 2014; Barbier 2017).

There have been various attempts to estimate a monetary value for marine ecosystem services (Costanza et al. 1997, 2014; WWF 2015; Martin et al. 2016), demonstrating that conservation of species and ecosystems is economically advantageous (Costanza et al. 2014). Specific examples include the use of natural ecosystems for coastal defence (Narayan et al. 2017; Hooper et al. 2019) and for sustainable fisheries management (Costello et al. 2016, 2019; World Bank 2017). Valuations for ecosystem services have been developed for land-based systems where the 'value' of natural capital (abiotic and biotic elements of nature) can easily be estimated from the areas of different types of habitat. Such valuation methods run into difficulties when applied to marine ecosystems, which are three-dimensional; contain many mobile elements, both spatially and temporally; are highly connected and often data-poor (Hooper et al. 2019). Ecosystem services are also provided at different scales—from the individual to human society as a whole (Pendleton et al. 2016; Small et al. 2017)—and, as such, are often public goods or the product of common assets that lead to problems with simplistic systems of monetisation (Costanza et al. 2014). Also, whilst ecosystem services are generally positive, nature can also generate negative impacts on people depending on spatial, temporal, social and cultural contexts (IPBES 2019). This is particularly complicated by the fact that many ecosystem services are strongly linked; thus, enhancing provisioning services, for example, can have a negative impact on regulating services (Raudsepp-Hearne et al. 2010). This can be assessed through analysis of bundles of ecosystem services and the trade-offs between them (Raudsepp-Hearne et al. 2010). The cost-benefit analysis approach inherent in the monetary valuation of ecosystem services can be useful in some contexts, but a more comprehensive methodology is required to establish a value for ecosystem services that takes into account more than just instrumental values (Colyvan et al. 2010; Hooper et al. 2019). One way of counteracting some of the difficulties in valuing ecosystem services can be the development of a risk register, which identifies those ecosystems and their services in danger of loss (Mace et al. 2015).

The relationship between Biodiversity and Ecosystem Functioning (the BEF curve), and thus the provisioning of ecosystem services, is not well understood but is generally observed to be positive (Hector and Bagchi 2007; Harrison et al. 2014; Lefcheck et al. 2015), including in marine ecosystems (Stachowicz et al. 2007; Danovaro et al. 2008; Gamfeldt et al. 2014; Duffy et al. 2016). The shape of the BEF curve has major implications for the impacts of biodiversity loss on ecosystem functioning and service provision and can be saturating, linear or accelerating (Fig. 10.8). These studies provide some scientific understanding of the mechanisms that may underlie the degradation of ecosystem services when biodiversity is lost, including biomass production, resilience to disturbance and biological invasions (Stachowicz et al. 2007; Duffy et al. 2016).

The impacts of biodiversity loss on ecosystem services are multi-faceted. Regional changes in biodiversity have been shown to affect fisheries and other services and generate risks, including harmful algal blooms, oxygen depletion, coastal flooding, and species invasions (Worm et al. 2006).

High biodiversity may also result in greater resistance to climate change impacts, potentially mitigating the effects on fishery yields (Duffy et al. 2016). On coral reefs, ecosystem

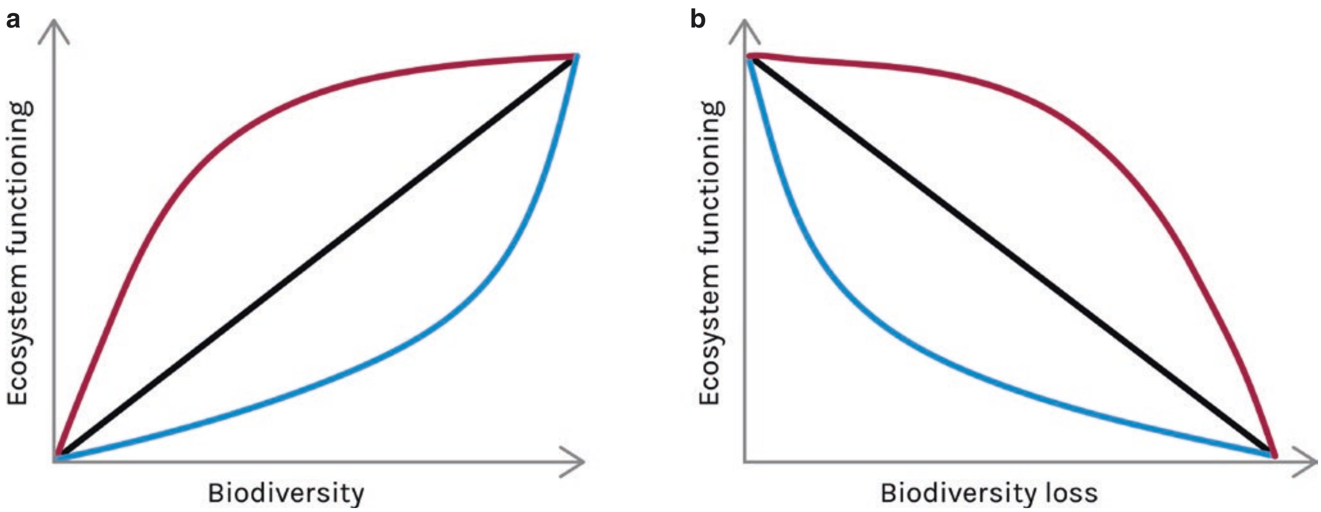

Fig. 10.8 Three types of positive biodiversity-ecosystem functioning relationships. *Notes:* (**a**) *Ecosystem functioning relationships: saturating (red), linear (black), and accelerating (blue).* (**b**) *Relationship between biodiversity loss and the three types of biodiversity-ecosystem functioning relationships. (Source: Modified from Naeem 2002; Strong et al. 2015)*

functioing has been suggested to scale with biodiversity, with human population density impacting both biodiversity and ecosystem functioning (Mora et al. 2018). The loss of coastal habitats renders coastlines more vulnerable to flood risks from sea level rise (Guannel et al. 2016) and cyclones (Barbier 2017; Hochard et al. 2019). In the case of coral reefs, the reduction in damage to terrestrial assets conferred through coastal protection is estimated at $4 billion annually (Beck et al. 2018). For the top five countries that benefit from reef protection (Indonesia, Philippines, Malaysia, Mexico, Cuba), this is the equivalent benefit of $400 million annually in mitigated damage (Beck et al. 2018). Annual expected damage from flooding would double, and costs from frequent storms would triple without coral reefs (Beck et al. 2018). The global loss of coral reefs has been estimated to have an economic impact of more than $10 trillion per annum (Costanza et al. 2014). Coastal habitats are important habitats and nursery sites for many species, so their losses result in reductions in fisheries and coastal food production (Aburto-Oropeza et al. 2008; Barbier 2017; Robinson et al. 2019; Unsworth et al. 2019), and they increase threats to species with a fragile conservation status.

Seagrasses, saltmarshes and mangroves are the three internationally recognised blue carbon habitats that actively sequester and store organic carbon from the environment (Nellemann et al. 2009; Duarte et al. 2013a, b). Mangroves are able to sequester more organic carbon on average than seagrasses and slightly more than saltmarshes (Mcleod et al. 2011). However, seagrasses have an area of around 180,000 km^2 globally, more than twice the area of mangroves, highlighting their importance as a significant carbon sink in comparison to mangroves. However, some of the carbon that is stored in these marine macrophytes has an allochthonous source from other habitats. Kelp beds and other macroalgae communities (Wernberg and Filbee-Dexter 2019) are only recently being considered important in blue carbon storage (Trevathan-Tackett et al. 2015; Krause-Jensen and Duarte 2016; Krause-Jensen et al. 2018). This may not only be through the existence of natural kelp and macroalgal communities but also through kelp aquaculture, where a significant amount of carbon is sequestered prior to harvesting (Duarte et al. 2017). Therefore, it is critical to focus on filling the gaps in knowledge of the extent, distribution and role of macroalgae in a global context, for both climate mitigation and adaptation, and as providers of crucial ecosystem goods and services.

Projected reductions in overall marine biomass associated with climate change may further impact ecosystem services such as fishery yields (Lotze et al. 2019). Any impact on fishery yields may have knock-on effects on food security. It is possible that some countries are likely to face a 'double jeopardy' of impacts on both agricultural and fisheries sectors as a result of climate change (Blanchard et al. 2017).

4 Thresholds and Tipping Points

There are ecological thresholds and other reference points that—if exceeded through the alteration of marine habitats, the exploitation of living marine resources or other human impacts on marine ecosystems—could result in negative and irreversible changes to ecosystems and the broader services they provide (Rockström et al. 2009; Lenton 2013).

The ecosystem approach to management of marine resources aims to preserve the integrity and resilience of marine ecosystems by reconciling conservation and exploitation (Pikitch et al. 2004). Under heavy fishing and climate pressures, many ecosystems are facing severe and abrupt regime shifts. This results in alternate ecosystem states that are most often less productive for fisheries, more prone to booms and busts, weakly reversible and thus less manageable (Pine et al. 2009; Estes et al. 2011, Travis et al. 2014). In this context, a major challenge for research and management is understanding evolving species interactions while identifying critical thresholds and tipping points involved in the disruption of marine ecosystems.

4.1 Changes in Marine Ecosystems

Climate patterns have long been recognised as responsible for regime shifts in both pelagic and benthic marine ecosystems. Empirical evidence has accumulated to indicate that shifts in species composition are initiated by large environmental changes, such as in the California Current (Hooff and Peterson 2006), the Gulf of Alaska (McGowan et al. 1998), the northern Pacific (Hare and Mantua 2000), the northern Atlantic (Aebischer et al. 1990) or the Humboldt Current (Chavez et al. 2003). Likewise, regime shifts between tropical coral reefs and algal-dominated reefs have been reported in response to thermal anomalies associated with El Niño events (Hughes et al. 2007; Diaz-Pulido et al. 2009), now compounded with anthropogenic ocean warming (Graham et al. 2015).

Long-term ocean warming and acidification—as well as extreme events that are becoming more frequent, more intense and longer lasting—alter the structure of ecosystems and cause mortality and community reconfiguration. This is particularly noticeable for sessile organisms that are impacted by discrete, prolonged, anomalous warm-water events known as marine heat waves (Hobday et al. 2018). The widespread bleaching and mortality of reef-building corals (e.g., in the Great Barrier Reef, the Caribbean Sea and the Gulf of Mexico), seagrass meadows and kelp forests have been strongly affected by localised, extreme warming of the ocean (Smale et al. 2019). The density and diversity of corals on reefs are declining, leading to vastly reduced habitat complexity, loss of biodiversity and domination by macroalgae that form stable communities relatively resistant to a return to coral domination (Wilson et al. 2006; Hoegh-Guldberg et al. 2007).

Climate change reinforces the frequency and strength of ecosystem shifts by affecting the distribution of marine life. Geographical shifts in marine species, from plankton and fishes to mammals and seabirds, occur as the result of ocean warming and have changed the distribution by hundreds of kilometres or more since the 1950s (Poloczanska et al. 2013, 2016; IPCC 2019). Ocean warming and heat waves also cause a poleward expansion of corals, leading to a phase shift from kelps to corals in South West Australia, facilitated by the poleward expansion of tropical herbivorous fish that prevent kelp from reestablishing (Wernberg et al. 2016). A poleward shift in species distributions is the most commonly observed pattern; it leads to changes in community structure, resulting in cascading impacts on ecosystem structure (IPCC 2019). The tropics may be particularly sensitive to this phenomenon as well as the transition zone between tropical and temperate communities, where the rate and magnitude of change will be highest. However, in the Humboldt upwelling system off the coast of Chile, most fish species do not show expansion of their southern endpoint because of a weak warming trend, reinforcing the hypothesis that temperature is a major determinant of species range dynamics (Rivadeneira and Fernandez 2005).

A global decrease in abundance and biodiversity of marine species driven by ocean warming is projected to diminish the catch potential for global fisheries in the 21st century (Britten et al. 2017; IPCC 2019). Global rates of biomass production as well as standing stocks are projected to decrease in ocean ecosystems at all depths, from the surface to the deep seafloor. The large-scale redistribution of global fish and invertebrate species biomass is expected to occur by 2055, with an average increase of 30–70% in high-latitude regions and a drop of up to 40% in the tropics under climate change scenarios (Cheung et al. 2010).

These changes in distribution are already affecting the species composition of catches. Fisheries are catching an ever-increasing percentage of warm-water marine species, a phenomenon identified as the 'tropicalisation' of the world catch (Cheung et al. 2013). Displacement of tropical herbivorous fish to temperate habitats also drives a similar tropicalisation of benthic habitats (Vergés et al. 2014; Wernberg et al. 2016). Using an ensemble of multiple climate and ecosystem models, it is projected that even without considering fishing impacts, mean global marine animal biomass will decrease by 5% (±4% standard deviation) under low emissions and 17% (±11% standard deviation) under high emissions by 2100, with an average 5% decline for every 1 °C of warming (Lotze et al. 2019).

In ecosystems stressed by overexploitation and climate change, cascading effects that have promoted regime shifts have been thoroughly documented in diverse marine ecosystems, ranging from upwelling systems to coral reefs. In the upwelling system of Namibia, following the collapse of the forage fish during the 1970s, namely sardines (*Sardinops sagax*) and anchovies (*Engraulis encrasicolus*), the ecosystem became dominated by two species of very low caloric value: the bearded goby (*Sufflogobius bibarbatus*) and a jellyfish (Cnidaria, Medusozoa). The latter reached a biomass estimated at more than 40 million tonnes during the 1980s and 12 million tonnes during the

2000s. As a consequence, the predators of these forage fish, the African penguin (*Spheniscus demersus*) and the Cape gannet (*Morus capensis*), suffered a lack of adequate prey and declined by 77% and 94%, respectively. Juvenile penguin survival was found to be approximately 50% lower than in proximate areas that were not food depleted, revealing the extent and effect of marine ecological traps. Cape hake (*Merluccius capensis*) and deepwater hake (*Merluccius paradoxus*) catches declined from 295,000 tonnes in 1972 to 150,000 tonnes since 1990, and the production of Cape fur seal (*Arctocephalus pusillus*) pups was strongly affected (Roux et al. 2013; Sherley et al. 2017).

In the Gulf of California, elegant terns (*Thalasseus elegans*) experience low or failed breeding and nesting distribution changes during years of positive sea surface temperature anomalies associated with increased sardine fishing effort by the local industrial fleet (Velarde et al. 2015b). In the Black Sea ecosystem, intense fishing of large predators and eutrophication of the ecosystem resulted in an outburst of an invasive comb jelly, *Mnemiopsis leidyi*, in a system-wide trophic cascade (Daskalov et al. 2007). Likewise, Wanless et al. (2005) observed that the major reproductive failure of birds in the North Sea during the 1990s was caused by a change in the dominant trophic pathway, which forced the birds to feed on sprats rather than sand eels, with the latter constituting higher-energy feed. A comprehensive fishery-independent data set of North Pacific seabird tissues was recently used to inform pelagic ecosystem trends over thirteen decades (from the 1890s to the 2010s), revealing a long-term shift from higher trophic level prey to lower trophic level prey, from fishes to squids (Gagné et al. 2018).

Most Caribbean reefs experienced a rapid shift from coral to algal dominance during the 1980s. The regime shift was initiated by a decline in the abundance of herbivorous fish caused by overexploitation. The role of herbivory was replaced by the urchin *Diadema antillarum*, but populations of this animal were severely depleted by a disease epidemic. Macroalgae proliferated over the reefs, thereby reducing reef coral recruitment.

Key interactions among four major tropical taxa—coral, macroalgae, fish and urchins—have created a self-perpetuating process that locked reef ecosystems into an alternative, nearly coral-free state (Travis et al. 2014), sometimes together with increased nutrients, to cause and perpetuate regime shifts cascading down to microbial components (Bozec et al. 2016; Haas et al. 2016; Zaneveld et al. 2016). Similarly, in the Humboldt upwelling system, the influence of overfishing of carnivores has favoured the increase in the biomass of herbivores, which subsequently changed the structure of kelp forests (Pérez-Matus et al. 2017).

4.2 Quantifying Tipping Points

The above examples illustrate the need to quantify connectivity in food webs, particularly the strength of predator-prey interactions in order to identify thresholds that push marine ecosystems past their tipping points.

Small pelagic fish exert a major control on the trophic dynamics of upwelling ecosystems and constitute mid-trophic level, ‘wasp-waist’ populations (Cury et al. 2000; Bakun 2006). These small- and medium-sized pelagic species are the primary food source of many marine mammals, larger fishes and seabirds, transferring energy from plankton to larger predators. They also are grazers/predators in marine ecosystems, feeding upon phytoplankton, zooplankton, and, in some cases, the early life stages of their predators. Using 72 ecosystem models, a global meta-analysis quantified the required forage fish biomass to sustain all fish predators in marine ecosystems, including marine mammals (Pikitch et al. 2012). A minimum precautionary biomass of 40% of forage fish is required to sustain predators.

The cascading effect of the overexploitation of forage fish is particularly detrimental to seabirds. The global and substantial overlap and competition between small pelagic fisheries and seabirds represents 48% of all marine areas, notably in the Southern Ocean, Asian shelves, Mediterranean Sea, Norwegian Sea, and California coast (Grémillet et al. 2018). Behind all of the diversity and complexity of the world’s marine ecosystems and the multitude of adverse drivers in bird declines, a striking pattern relating seabird breeding success and their fish prey abundance was found for 14 bird species within the Atlantic, Pacific, and Southern Oceans (Cury et al. 2011). A threshold in prey (fish and krill, termed forage fish) abundance, equivalent to one-third of the maximum prey biomass, was found below which there is the occurrence of consistently reduced and more variable seabird breeding success. This threshold is also equivalent to the long-term average prey abundance and constitutes an evolutionary stable strategy for marine birds. This empirically derived guiding principle embraces the ecosystem approach to management aimed at sustaining the integrity of predator-prey interactions and marine food webs. In well-documented ecosystems, this universal threshold can be revisited and sometimes adapted according to specific ecological and environmental constraints, such as the quality of food or the existence of specific reproductive habitats that are accessible to birds (Guillemette et al. 2018).

Coral bleaching events resulting from global warming and ocean acidification will compromise carbonate accretion, with corals becoming increasingly rare on reef systems (Hoegh-Guldberg et al. 2007). Consequently, policies that result in atmospheric levels of carbon dioxide above 500 parts per million, appear extremely risky for the future of coral reefs and should be strongly avoided. Moreover, near-future increases in local sea temperature of as little as 0.5°C will result in the protective mechanism of coral reefs being lost, which may increase the rate of degradation of local coral reefs (Ainsworth et al. 2016). The loss of ecological resilience occurs because coral cover increases more slowly after disturbances but also when competitive interactions with macroalgae become more frequent and longer in duration. To reduce those interactions, coral reefs require higher levels of grazing to exhibit recovery trajectories (i.e., about 40% of the reef being grazed; Hoegh-Guldberg et al. 2007). Maintaining resilient coral reefs similarly requires harvest limitations and maintaining the minimum biomass of grazing fish species playing a key role, such as parrotfish (with a harvest limitation of less than 10% of virgin fishable biomass combined, with an enforceable size restriction of more than 30 cm) (Bozec et al. 2016).

4.3 Fisheries Management Perspective

With climate change and overexploitation, ecosystems are more vulnerable to changes that previously could be absorbed and may suddenly shift from desired to less desired states in their capacity to generate ecosystem services (Folke et al. 2004). Recovering ecosystems that have experienced regime shifts and have moved past their tipping points appears very difficult, to almost impossible (Haas et al. 2016), so that adaptive practices work only poorly or not at all (IPCC 2019).

For sustainable exploitation and conservation, it is crucial to fully appreciate the fact that ecosystems have tipping points, identify the potential thresholds, and implement them into management (Suding and Hobbs 2009; Travis et al. 2014). In a global change context, multiple and confounding factors influence the state of marine ecosystems. Reliable detection and attribution appear to be fundamental to our understanding of ecosystem changes (IPCC 2019), however, the confident attribution of tipping points in ecosystem dynamics remains challenging. Overexploitation and climate change can promote tipping points and can potentially act in synergy within ecosystems, increasing the risk of irreversible changes. Marine conservation and adaptive management approaches must consider long-term persistent warming and acidification as well as consequent discrete extreme events that are pivotal in shaping ecosystems. The limitation of CO_2 emissions appears to be a strong constraint in the preservation of marine ecosystems, despite the difficulty in reaching the Paris Agreement targets. However, the growing threat of abrupt and irreversible climate change must compel political and economic action on carbon emissions (Lenton et al. 2019).

Fisheries management will have to consider the structuring role of key species, such as small pelagics in upwelling systems or herbivorous fishes in coral reef ecosystems. To avoid regime shifts, the ecosystem approach would greatly benefit from the integration of readily available limit reference points, defined by predator-prey interactions between species, into fisheries management strategies. Examples of such ecosystem-based management approaches which go beyond the traditional single-species stock assessment are plentiful. For example, the CCAMLR has the principle embodied in its articles to ensure that target stocks and their dependent and related species are all maintained at productive levels (Constable 2011). This has steered the management of krill fisheries in the Southern Ocean to ensure that stocks are fished sustainably but also that the predators of this keystone species are supplied with ample prey (Constable 2011). Similar approaches are used to manage finfish in the Antarctic (Constable 2011). Other successes of the CCAMLR ecosystem approach include technical measures to prevent the mortality of albatrosses and petrels in longline fisheries for Patagonian and Antarctic toothfish (*Dissostichus eliginoides* and *Dissostichus mawsoni*; Friedman et al. 2018). Many fisheries, including those in the CCAMLR, employ observer programmes to estimate the bycatch of endangered species or non-target species which may be vulnerable to fishing mortality and to alter fishing practices to reduce such impacts should they be detected (see Gilman et al. 2017). Integration of such ecosystem-based indicators will help to sustain desired ecosystem states while protecting marine species.

5 Monitoring

Humans and climate change continue to impact the marine world and its resources. Thus, when evaluating policy and management approaches, it is vital to be guided by indicators that can capture the status, trends and drivers of ocean health (Block et al. 2011; Miloslavich et al. 2018b; Cubaynes et al. 2019). The main indicators used in marine conservation planning relate to habitat extent, species diversity and extinction risk. Nevertheless, quantifying habitat extent and its associated diversity is difficult because of the high technical and logistical requirements as well as funding constraints; therefore, results are limited in statistical power and often fail to provide the required spatial- temporal dimension (Palmer et al. 2002).

5.1 How Can We Effectively Monitor and Manage Biodiversity and Enjoy the Benefits of a Sustainable Blue Economy in a Changing World?

Ocean monitoring and surveillance have been identified as components of the blue economy needed to respond to ocean health challenges (EIU 2015). The Framework for Ocean Observing (Lindstrom et al. 2012; Tanhua et al. 2019) provided key concepts based on the delivery of a multidisciplinary system, focused on the use of Essential Ocean Variables (EOVs). EOVs act as the common focus for observations to generate data and information products based on the scientific and social requirements. Biological EOVs, which are highly focused on understanding biodiversity trends, were identified based on their relevance to address such social and scientific requirements and their feasibility for global measurement in terms of cost, available technologies and human capabilities (Miloslavich et al. 2018a). The sustained observation of these EOVs will serve as the foundation for implementing management and policy based on science to promote a healthy and sustainable ocean, from local to regional to global scales. These biological EOVs also support the global climate observing system as plankton communities and some coastal ecosystems (e.g., coral reefs, seagrass beds, and mangrove forests) are considered to be essential climate variables (WMO 2016). Planning is currently underway for the internationally coordinated and global networks that measure these biological EOVs. Such planning includes (1) identifying existing data sets for each EOV at all geographical scales; (2) reviewing technological monitoring approaches and standard operating procedures along with the capacity needed to use them; and (3) recommending approaches for data and metadata consolidation in findable, accessible, interoperable and reusable (FAIR) systems. Building the system required to achieve the sustainability of marine diversity and ecosystems, which is critical for the blue economy, will require governance, broad communication and establishing partnerships. It will also require the development of new technologies and of human capacity. Investing in people and their institutions, particularly for developing countries, is required to build infrastructure and long-term support networks with enhanced access to data, tools and technologies. Additionally, collaborations that combine multiple knowledges, including indigenous knowledge, can provide an important role in understanding species distribution (Skroblin et al. 2019) and may play an increasing role in enhancing our capacity to have a more holistic understanding of ecology (Ens et al. 2015).

This can be facilitated by international initiatives, but it will require the long-term engagement of national institutions and local communities as well as funding, including major contributions from philanthropists and the private sector if it is to be sustained (Bax et al. 2018; Miloslavich et al. 2018b).

5.2 What Are the Technological Tools for Biodiversity Monitoring?

The methods for monitoring marine biodiversity are quite extensive and specific to the taxonomic group, type of ecosystem and/or spatial scale of the monitoring effort. Some of the persistent technical challenges of marine biodiversity monitoring include the need for clearly defined and standardised best practices and interoperable observation technologies. Data are collected through a combination of remote sensing and in situ observations (see Canonico et al. 2019 for a recent review). Remote sensing allows for observations at broad, global scales repeatedly, with a resolution highly dependent on the sensor. It provides information on functional phytoplankton groups and on the cover and distribution of some coastal habitats, such as coral reefs, seagrass beds, mangroves and macroalgae, and some structured habitats such as floating macroalgae (e.g., *Sargassum*). In situ observations include a variety of methods, from simple visual survey and/or sample collection to the use of sensors, instruments, and platforms. At the most basic level these observations rely on survey and/or sampling either on shore or in shallow water using scuba divers. Large-scale application of such methods can be used to tackle global questions about spatial differences in coastal marine communities or for monitoring over time if protocols are standardised (e.g., the Natural Geography in Shore Areas, or NAGISA, sampling protocol used in the CoML; Iken and Konar 2003; Cruz-Motta et al. 2010). Some of the most-used newer technologies include acoustic monitoring, which supports biomass and abundance estimates among other parameters; animal telemetry for animal movement in combination with environmental descriptions; 'omic' approaches to report on biodiversity across scales and taxa; and video/photo imagery from automated underwater vehicles (AUVs), remotely operated vehicles (ROVs), submersibles and divers. These technologies are already generating big data, which will require the use of artificial intelligence and machine learning processes, improved (real-time) quality control and enhanced data capabilities (Edgar et al. 2016). In the next decade, it will be critical to develop technologies that enable increasingly automated real-time biological observations.

In this context, satellite based remote sensing is frequently proposed as a cost-effective tool to lower the costs of obtaining spatially and temporally relevant information and monitoring changes (Mumby et al. 1999, 2004; Green et al. 2000). As technology continues to advance, improving the resolution and accuracy of satellite imagery, our knowledge of the distribution of habitats is improving. Although there has been a progression in monitoring a number of coastal habitats (Mumby et al. 2004; Giri et al. 2011), remote sensing has certainly not reached its full potential (Andréfouët 2008) because of technical limitations and difficulties classifying habitats (Zoffoli et al. 2014). Often there is a need to supplement this with existing field data and/or expert knowledge to obtain a more complete picture (Andréfouët 2008). Moreover, only the shallower component of subtidal critical habitats, such as seagrass meadows and algal stands, can be resolved by even the most advanced remote sensing technologies (e.g., hyperspectral satellite imaging; Wicaksono et al. 2019). Likewise, important habitats, such as deep-sea corals, are beyond the reach of existing or future airborne remote sensing technologies. The mapping of seabed topography at a relatively coarse scale can be undertaken using satellite gravity mapping (e.g., for seamounts; Yesson et al. 2011).

Habitats at shelf depths and in the deep sea were traditionally mapped by using plumb lines which had a wad of tallow in a cavity at the bottom of the plummet (the weight at the end of the line). The tallow would pick up fragments of whatever was on the seabed and a notation of the seabed type was added to nautical charts, providing a navigational aid for mariners.

As modern oceanographic science developed in the 19th century, habitat mapping was undertaken by trawling, dredging or other forms of seabed sampling. A significant advancement in seabed mapping was the development of single-beam sonar. Using this technology, Bruce Heezen and Marie Tharp constructed the first global topography maps of the seafloor. In the present day, the main tool of habitat mapping in coastal and deep waters is multibeam acoustic survey (Harris and Baker 2012; Lamarche et al. 2016). These sophisticated sounders not only accurately measure the depth of the seafloor but also give information on the hardness of substrata through the strength of acoustic return as well as seafloor microtopography (roughness) and volume heterogeneity, which relates to sediment grain size and composition (Harris and Baker 2012; Lamarche et al. 2016). This information can be used to identify seafloor texture, whether it is made of rock or sediment, for example, and can be used to classify habitat (Lamarche et al. 2016). Coupled with the use of seabed sampling using surface deployed gear (e.g., trawls or cores) and/or image-based surveying using towed cameras, ROVs, AUVs or submersibles for groundtruthing, these methods can provide accurate maps of seabed habitats (Harris and Baker 2012; Lamarche et al. 2016). An issue with this approach is that it is time consuming and expensive, and coverage tends to be restricted to areas targeted for specific study for scientific or industrial purposes. The global Seabed 2030 mapping project is currently collecting multibeam data to produce a more comprehensive map of seafloor topography than previously available.

Although it will certainly allow the identification of larger-scale geomorphological structures such as seamounts, canyons and plains, the extent this will be used in mapping of finer-scale habitats is unclear. An alternative technology to multibeam bathymetry is side-scan sonar. This produces a photograph-like sonar image of the seabed and can be particularly useful in imaging small objects and finer-scale structures on the seabed (e.g., sand waves; Lamarche et al. 2016). This technology is cheaper than multibeam systems but has a poor georeferencing capability, and backscatter calibration is usually not possible (Lamarche et al. 2016). A relatively new technology now being carried by AUVs is synthetic aperture sonar which provides very high resolution imagery but at a longer range than side-scan sonar (Hansen 2011). AUVs with hyperspectral imaging capabilities are now being developed to extend remote sensing capabilities to deeper waters for high-resolution habitat identification (Bongiorno et al. 2018; Foglini et al. 2019).

Many marine habitats and areas of the world still remain under-studied at larger scales, such as rocky reefs, algae beds, and large areas of the deep ocean for which there are no publicly available global distribution maps at present (Rogers et al. 2015). For the habitats where spatially referenced and processed information are available, often data sets relate to one point in time with very little indication of changes through time (Halpern et al. 2015). This limits their utility in understanding how, where and when the natural world is changing. As new technology is made available, such as the Google Earth Engine platform (Gorelick et al. 2017; Traganos et al. 2018; Nijland et al. 2019),[6] and barriers for information sharing are removed, there is a great opportunity to increase our capacity to understand, monitor and develop evidence-based policies and management plans to protect marine ecosystems.

Satellite remote sensing has had a significant impact on assessing the levels of fishing effort in the global ocean. Access to fisheries data is often denied for reasons of commercial confidentiality, but in a world where fisheries are sustainably managed, it is not necessary to hide what is taken or conceal the location, whether in national waters or in ABNJ. Satellite surveillance is increasingly useful as a means of spotting problems such as illegal fishing and transhipments; it is also a useful way to assess patterns of fishing even in the remotest parts of the ocean (Eigaard et al. 2017; Amoroso et al. 2018; Boerder et al. 2018; Elvidge et al. 2018; Ford et al. 2018b, c; Kroodsma et al. 2018;

[6]For more information about the Google Earth engine, see https://earthengine.google.com/

Longépé et al. 2018; Rowlands et al. 2019). The development of online platforms such as the Global Fishing Watch has exposed the industry to societal oversight where previously it did not exist, especially in waters far from the coast.[7]

The new model of fisheries surveillance has been taken up by several coastal states, such as Chile, Indonesia and Panama. These countries have now committed to making the tracking data of vessels carrying their flags available to public scrutiny. Such data can only improve the sustainability of fishing; it will not only identify where and when fishing is taking place but also provide insight into the enforcement of MPAs (Rowlands et al. 2019) and destructive fishing practices (Winnard et al. 2018).

5.3 Overseeing the Monitoring of Biodiversity

At the intergovernmental level, two major organisations provide a governance framework for marine biodiversity observations.

The first, the IOC of UNESCO, through the Global Ocean Observing System (GOOS), has led the implementation of the Framework for Ocean Observing (Lindstrom et al. 2012) with the goal of serving users across climate, operational services and ocean health (Tanhua et al. 2019). GOOS is also co-sponsored by the World Meteorological Organization, the United Nations Environment Program, and the International Science Council. Within GOOS, marine biodiversity observations are coordinated by the Biology and Ecosystems Panel, or GOOS BioEco (Miloslavich et al. 2018a). GOOS also provides governance at the regional level through the GOOS Regional Alliances, examples of which are the Integrated Marine Observing System (IMOS) of Australia, the Integrated Ocean Observing System (IOOS) of the United States and the European Global Ocean Observing System (EuroGOOS) in Europe. Through expert panels, regional alliances, the Observations Coordination Group, and affiliated projects, GOOS supports a broad observing community, from individual scientists and research organisations to governments, UN agencies, and international programmes.

The second major organisation is the Marine Biodiversity Observation Network (MBON) framed in the Group on Earth Observations Biodiversity Observation Network (GEO BON), which facilitates the coordination between individual monitoring programmes and existing networks (Muller-Karger et al. 2018). Both MBON and GOOS BioEco share common goals and encourage the use of best practices for marine biodiversity monitoring, the contribution of data to open access data systems and provide a framework for data management, communication and applications (Canonico et al. 2019).

Based on these shared goals, these organisations have signed an agreement together with OBIS, which operates under the IOC's International Oceanographic Data and Information Exchange (IODE) programme, to work together to advance sustained, globally consistent observations of marine biodiversity with the commitment to open access and data sharing, implementing best practices and international standards and enhancing global capacity (Miloslavich et al. 2018a). Having this overarching governance in place is a major step; however, much work still needs to be done. To achieve the required level of coordination and communication across all networks, programmes and countries, the organisations need to ensure the interoperability of the data and that the data contributes to the development of indicators to address policy and management requirements. Specifically related to governance in coastal zones, an assessment carried out by the Economist Intelligence Unit across 20 countries found that the Coastal Governance Index is uneven, with developed countries doing relatively well but still requiring work. Other important factors that contribute to better coastal policies include participatory inclusion in decision-making and accountability, the level of economic development, having the capacity required for the implementation of policies, and having marine spatial planning policies (EIU 2015).

With the proper training and quality control, citizen science can be used both as a way of communication and as a way for data collection on a broad range of scales. An excellent success story of citizen science is the Reef Life Survey (RLS) programme.[8] The RLS was established in Australia in 2008 to collect data on the biodiversity of benthic and fish communities on rocky and coral reefs through trained volunteer scuba divers (Stuart-Smith et al. 2017). Since its establishment, it has expanded globally to more than 3000 sites in nearly 50 countries, providing invaluable data for ecosystem management and conservation (Stuart-Smith et al. 2018). Furthermore, the Biodiversity Indicators Partnership, which promotes the development and delivery of biodiversity indicators to measure progress on Aichi Biodiversity Targets and SDGs, has recently accepted two of the RLS indicators (the 'Large Reef Fish Indicator' and the 'Reef Fish Thermal Index') to inform Aichi Biodiversity Targets 6, 10 and 11 and also SDG 14.2 (RLS 2019).

[7] See the Global Fishing Watch, https://globalfishingwatch.org/

[8] More information about the Reef Life Survey can be found on its website, https://reeflifesurvey.com

6 Gaps and Challenges in Habitat Protection

6.1 How Much of Key Marine Habitats Are Protected?

To understand how MPAs are currently distributed across the key habitats considered (Table 10.1), the March 2020 version of the World Database of Protected Areas (UNEP-WCMC and IUCN 2020) was used to calculate the extension of all the coastal protected areas and MPAs (hereafter collectively referred to as MPAs), or the number of reported locations of each habitat, inside of an MPA within EEZs. We considered three scenarios for the analyses: (1) all areas designated as MPAs without distinction, (2) only MPAs reporting a management plan and (3) only fully protected MPAs (labelled in the database as 'no-take zones').

We estimate that 12% of the habitats considered in this study lie within an MPA. However, when we considered only the MPAs with management plans, only 6% of the habitats are included, and just 3% are in fully protected MPAs at a global level. An example of how these three scenarios overlap is provided by kelps, where more than 40% of the world extent of these habitats are recorded as protected within all forms of MPAs (Fig. 10.9a). However, kelp protection decreases to only 24% under MPAs with management plans and only 1% in fully protected MPAs (Fig. 10.9a).

The deeper habitats show a similar trend, with the habitat with most of its area protected being cold-water corals. They have 24% lying within MPAs, which drops when only managed and fully protected MPAs are considered to 14% and 4%, respectively.

It is important to consider that coastal habitats have arguably received historically higher levels of human pressures

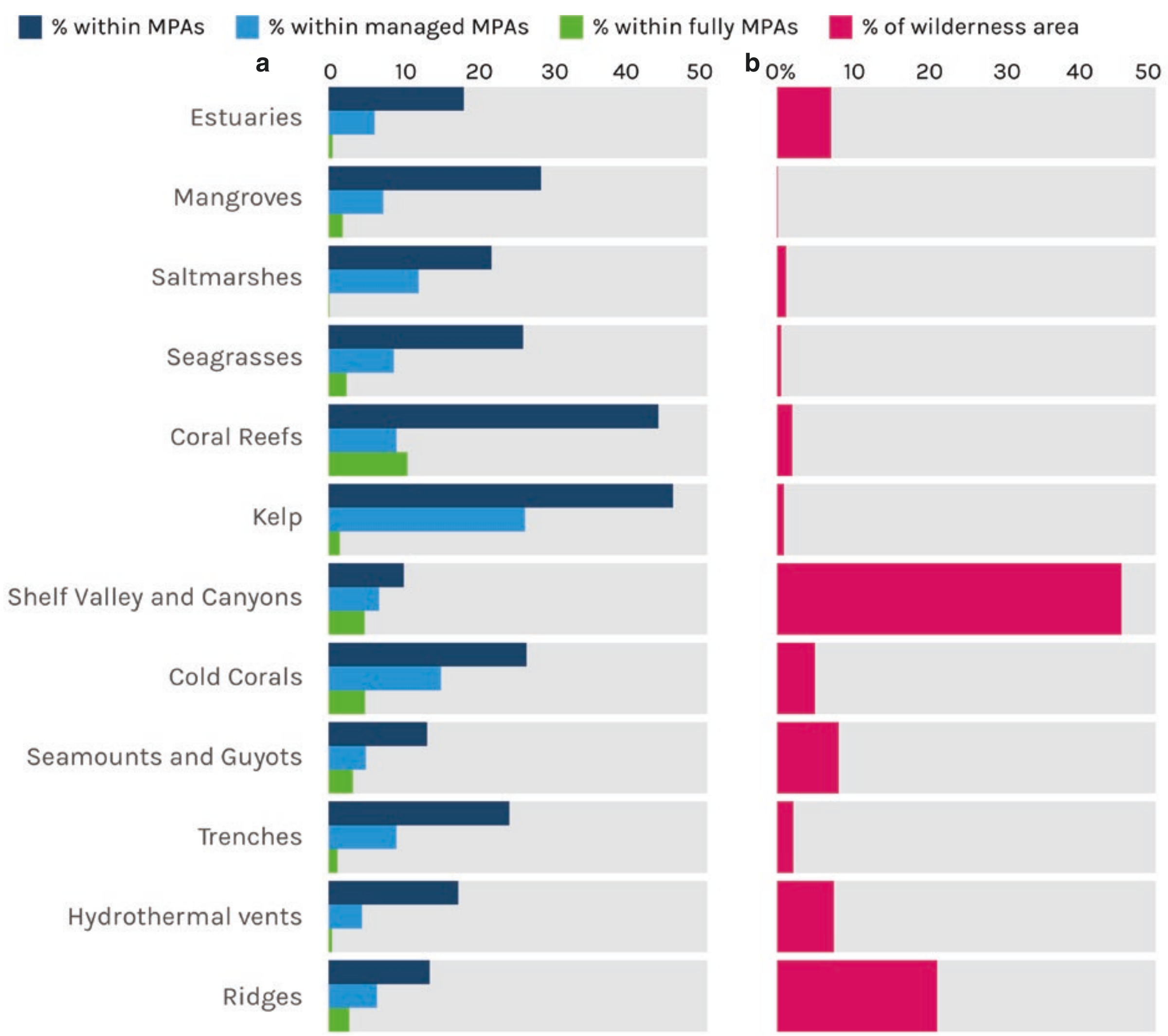

Fig. 10.9 Current conservation efforts for key selected habitats. *Notes: Habitats on the x-axis are ordered according to their distance to the coast, as a proxy for their average depth.* (**a**) *The bars represent the percentage of the habitat within MPAs, within MPAs with a management plan, and fully protected MPAs.* (**b**) *The percentage of wilderness inside the habitat area. (Source: Authors)*

compared to oceanic habitats. Evidence of the destruction of coastal habitats (see Sect. 3.5, Habitat Degradation and Its Drivers), which has already severely reduced their original distributional area, should be taken into account when considering the percentage of the current habitat extent in MPAs.

Estuaries and saltmarshes are the coastal habitats with the lowest proportion in fully protected MPAs (Fig. 10.9a) despite their importance in habitat provision for a wide range of species and ecosystem services (e.g., carbon sequestration, nutrient cycling, coastal protection; Barbier 2017).

The area of the selected habitats lying within designated MPAs declines moving from the coast to offshore (Fig. 10.9a). However, this pattern is much less obvious for MPAs with management plans and non-existent for fully protected MPAs (Fig. 10.9a). This suggests that both coastal and offshore habitats are equally poorly represented within fully protected MPAs. The offshore habitats had on average a higher proportion in marine wilderness (based on the area estimated by Jones et al. 2018); most likely the result of decreased accessibility (Fig. 10.9b). At present the global coverage of MPAs is 7.43%, with 17.22% of national waters designated as MPAs, but this figure falls to 1.18% in ABNJ (UNEP-WCMC and IUCN 2020; accessed on 30 March 2020). The discrepancy between the coverage of MPAs in EEZs and ABNJ results from the lack of a coherent international legal framework for the establishment of marine protected areas on the high seas, putting at risk largely unknown biodiversity (O'Leary et al. 2012; Rogers et al. 2015). International efforts towards protecting habitats such as seamounts in ABNJ have been made in regional or sub-regional organisations such as RFMOs (e.g., New England seamounts protected from bottom trawling by the Northwest Atlantic Fisheries Organization), and the ongoing negotiations to manage marine biological diversity in ABNJ, which are aimed at establishing a new legal framework for protection of biodiversity in international waters and on the seafloor.

Additionally, the existence of a habitat inside of an area designated as an MPA does not mean it is protected. As can be seen from the above analyses, many MPAs lack a management plan, and even where such plans exist, MPA objectives and management might not involve the habitat, and permitted activities may even be destructive and/or poorly enforced (e.g., trawling in MPAs; Dureuil et al. 2018). In many meta-analyses of MPA effectiveness, there are benefits to conservation even where protection is partial (i.e., MPAs where not all activities are banned; e.g., Lester and Halpern 2008; Sciberras et al. 2013; Gill et al. 2017; Sala and Giakoumi 2017). Our analyses suggest that despite the apparent progress reported in MPA designation (UNEP-WCMC and IUCN 2020), reaching the Aichi Biodiversity Target 11 of having 10% of representative habitats of our oceans being well protected is still a remote target, as has been found in other studies (Klein et al. 2015; Jenkins and Van Houten 2016; Sala et al. 2018a; Jones et al. 2020). Key shortfalls and key features that can hinder and enhance MPA effectiveness, respectively, have been recognised in current literature (Edgar et al. 2014; Gill et al. 2017). In particular, the NEOLI features identified the most important characteristics of an MPA: being No-take (i.e., fully protected), well Enforced, Old (more than 10 years), Large (more than 100 km^2) and Isolated. The main issue is that MPAs that fulfill some or all of these features, are not common globally (Edgar et al. 2014; Sala et al. 2018a). Although, most existing MPAs could improve in some of the NEOLI features by increasing the no-take area, fostering compliance and enforcement, and extending the boundaries to isolate key habitats to protect, these features are difficult to achieve. Our analyses indicate that to reach international goals and markedly increase the conservation benefits of the global MPA network, it is important to improve existing MPAs while also creating new ones.

6.2 Protection Gaps in EEZs

Humans are exerting pressures on marine habitats throughout the world, often leading to significant damage to them as well as loss of associated biodiversity (Halpern et al. 2015). To understand this on a global scale, we calculated the average biodiversity value for each EEZ, using biodiversity data from Reygondeau and Dunn (2018), and found the sum of ecological and social factors that decrease the health of the ocean. This analysis reveals that countries that have higher biodiversity also experience higher pressure (p-value <0.001, $R^2 = 0.165$; see Fig. 10.7). One might expect that countries with high gross domestic product (GDP) would be capable of protecting a larger fraction of their EEZ. Although we found a significant relationship, GDP explains very little of the variation in the area of MPAs that are implemented in the national waters of each country (p-value <0.001, $t = 0.11$; see Fig. 10.10a). We would expect that countries with higher investment capacities (i.e., GDP) would show a higher relative area of MPA coverage, especially because EEZs and GDP tend to be related. Furthermore, although there are considerable conservation efforts and investments—reflected in MPA coverage—biodiversity and the relative MPA area to each country's EEZ are not correlated (p-value >0.05; see Fig. 10.10b). These results indicate that areas with high biodiversity should be prioritised for protection not only for their biodiversity per se but also to create resilience from the high pressures they experience.

However, representative biodiversity from all regions must be included in a global network of fully or highly protected MPAs, and this must be complemented by sustainable management of all human activities in the ocean (see below; Margules and Pressey 2000). The lack of correlation between

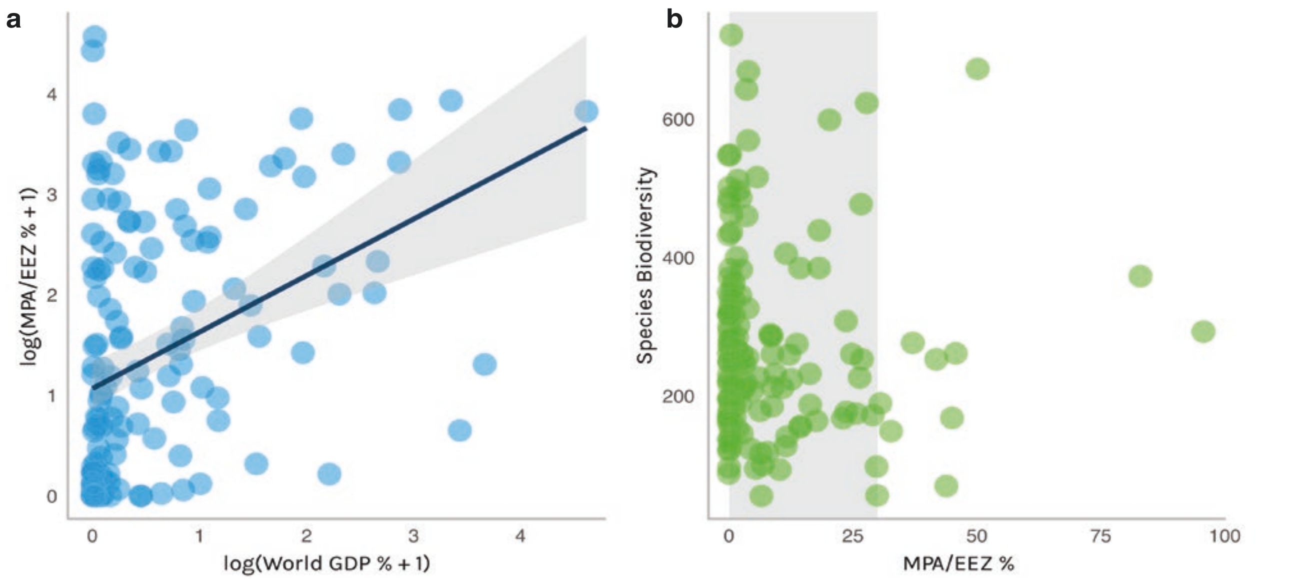

Fig. 10.10 Relationships between biodiversity, GDP and MPA extent. *Notes: Panel* (**a**) *shows the gross domestic product (GDP) that a country has relative to the world and the amount of their exclusive economic zone (EEZ) that is covered by marine protected areas (MPAs). Panel* (**b**) *reveals that the relative size of a country's MPAs are not correlated with their biodiversity. The grey region in Panel* (**b**) *represents the countries with less than 30% of their EEZ with MPA coverage. (Source: Authors)*

the biodiversity within an MPA and the amount of the EEZ that is protected by a coastal state suggests that biodiversity-rich countries do not develop more MPAs than biodiversity-poor countries.

Further, Kuempel et al. (2019) found that MPAs with the strictest protection were 6.3 times more likely to be found in low-threat ecoregions, indicating that countries focus conservation efforts in the least threatened areas as opposed to areas with high threats to biodiversity. Additionally, areas with lower biodiversity can still be highly productive and valuable in terms of ecosystem services provision to coastal states as well as in ABNJ.

Even when considering the best-case scenario, using all the MPAs reported and assuming that these have at least some benefit to protect habitats, it is possible to see that between 45% and 90% of countries are protecting less than 30% of habitat extent (Table 10.3). The numbers worsen, in area terms, when the two other scenarios are considered, with at best 23.3% of countries with 30% or more of a habitat lying within a managed MPA (saltmarsh) and 4.2% in fully protected MPAs (hydrothermal vents; Table 10.3).

For saltmarshes and estuaries, no countries include 30% of the area of habitat in fully protected MPAs (Table 10.3). Indeed, if we break down the conservation effort for each country for the category of all MPAs, there is a large gap where some countries are committing more effort whereas others are not performing as well. Here, we propose to measure the proportional conservation efforts amongst countries by using measures of central tendency, the mean and the median percentage of habitat protected globally, as an alternative to absolute measures of habitat area. The overall protection effort is 'fair' when the mean and median percentage of habitat protected globally coincide to form a normal distribution of the conservation efforts (Fig. 10.11). The mean and the median percentages are reported as blue and red circles, respectively, which show that for most habitats there is a wide gap between area present and area protected. This indicates that current global conservation efforts are inadequate. Most countries are protecting very little (less than 1%) of the habitats they could protect, and conservation efforts are unevenly distributed. If MPAs with management plans are considered, for some habitats the 'effort gap' metric is even worse (e.g., saltmarshes, kelps and coral reefs; Table 10.3). In other cases, the effort gap appears to decrease, but this is mainly because the amount of habitat in managed MPAs is so small compared to all MPAs. A very small amount of habitat lies within fully protected MPAs, rendering the effort gap metric very small as all states are equally performing badly. Through this effort gap metric, we see that for fair habitat conservation globally, countries need to cooperate to reach international goals, thereby compensating for the effort gap either by increasing their MPAs and/or aiding conservation programmes in less wealthy countries or regions. The effort gap highlights how even if some countries are contributing towards achieving a 'total conservation target', the majority of countries are under-performing.

This proportional conservation approach could also be applied to properly measure the effort each country should give to the protection of the high seas. This approach can be useful in a context where the use of ABNJ is emerging and

Table 10.3 Summary of the habitat protection target proposed

Habitat	Percentage of countries below 30%	Percentage of area below 30%	Mean percentage effort	Median percentage effort	Effort gap
Saltmarshes	51.2	87.9	41.0	28.1	12.9
	(76.7/100)	(92.0/100)	(21.1/0.5)	(1.8/0)	(19.3)
Kelps	45.3	37.2	36.3	37.7	−1.4
	(77.4/98.1)	(52.6/100)	(17.0/2.88)	(1.0/0)	(16)
Coral Reefs	61.6	44.5	30.7	17.5	13.2
	(86.6/97.3)	(91.7/95.8)	(22.3/1.1)	(0/0)	(22.3)
Hydrothermal vents	64.6	62.6	29.5	0.0	29.5
	(85.4/95.8)	(94.1/99.3)	(13.9/3.2)	(0/0)	(13.9)
Mangroves	59.1	59.2	29.3	19.9	9.4
	(86.0/98.9	(92.3/97.3)	(9.3/1.1)	(0/0)	(9.3)
Seagrasses	70.3	58.6	24.4	6.68	17.7
	(89.0/99.2)	(86.6/98.5)	(9.0/0.8)	(0/0)	(9)
Estuaries	72.8	76.2	20.2	5.7	14.5
	(88.8/100)	(94.8/100)	(8.5/0.1)	(0/0)	(8.5)
Cold Corals	76.6	77.5	18.7	0.0	18.7
	(91.2/98.5)	(87.3/99.8)	(7.98/1.5)	(0/0)	(7.98)
Trenches	80.4	74.8	18.3	0.0	18.3
	(93.5/97.8)	(91.4/100)	(6.59/2.34)	(0/0)	(6.59)
Ridges	82.0	73.1	16.4	0.0	16.
	(92.6/98.4)	(89.6/97.8)	(7.77/2.26)	(0/0)	(7.77)
Seamounts and guyots	81.4	59.1	14.4	0.0	14.4
	(92.0/96.5)	(84.7/86.6)	(6.3/2.5)	(0/0)	(6.3)
Shelf Valley and Canyons	90.6	97.1	11.1	0.1	11.0
	(95/98.9)	(98.0/99.9)	(5.3/1.0)	(0/0)	(5.3)

Notes: For each habitat, the percentage of countries that have granted less than 30% protection is shown ('Percentage of Countries below 30%') for all MPAs and then, in parentheses, the figure for managed MPAs/ fully protected MPAs that is below 30% protection. The 'Mean' and 'Median Percentage Effort' refers to the percentage of habitat countries protect on average. The differences between these two values is reported as the 'Effort Gap', representing the percentage by which countries below the threshold should ideally increase their protection to make a fair contribution to conservation. We did not calculate this for fully protected MPAs as the amount of habitat lying within this category of protected area is so low that the effort for all countries is equally very poor
Source: Authors

presents serious governance challenges (Merrie et al. 2014). For example, each country should deploy a conservation effort relative to its use of ABNJ across all sectors (e.g., fishing, shipping). ABNJ are a special case of global commons management. In these areas, establishing and enforcing conservation measures will require new financing mechanisms, such as a levy on the use of the resources and/or by establishing an international trust fund under the new legally binding instrument for the conservation and sustainable use of biological diversity of ABNJ. It is important that ABNJ are managed fairly by a proportional conservation measure rather than international goals with total conservation targets, which might disproportionately favour some countries over others and imperil the health of the high seas.

Whilst we have emphasised the use of MPAs in biodiversity protection mainly because their implementation can be quantified and analysed spatially to some extent, MPAs are not the only management measure that can conserve biodiversity (Duarte et al. 2020). It has been argued that the ocean can be compared to a frontier system, both to within EEZs and in ABNJ, where there is open access to resources, larger and less differentiated jurisdictions than on land and fewer laws that constrain human activity (Norse 2005). This situation has led to a free, open access scramble for resources. This has resulted in increasingly unsustainable levels of exploitation of marine living and other resources and the impacts on biodiversity that have been documented here and in other studies (Norse 2005). Marine reserves by themselves do not necessarily reduce overfishing, competition amongst fishers or the growth of global fishing fleets, and they may even increase competition amongst fishers by reducing areas available to fish, possibly even displacing fishing effort to areas where levels of fishing have been low or nonexistent (Kaiser 2005; Norse 2005; Agardy et al. 2011; FAO 2011; Hilborn 2018).

Marine reserves also provide little protection from threats such as long-range pollutants (e.g., many persistent organic pollutants; Agardy et al. 2011) or invasive species (e.g., Burfeind et al. 2013). The connectivity of populations of marine species and between habitats also means that even if fully protected MPAs are designed to ensure maximum con-

Fig. 10.11 Current conservation efforts for key selected habitats. *Notes: Best-case scenario, using all the MPAs reported. Habitats on the x-axis are ordered according to their distance to the coast, as a proxy for their average depth. Black circles represent countries hosting one of the key habitats. The y-axis represents the percentage of area that each country is protecting of that habitat within its exclusive economic zone (EEZ). Most of the countries are below the 30% target (white line), which has been identified as a threshold to ensure the maintenance of the ecosystem services of a habitat. The blue circles represent the mean percentage of all the countries' protection efforts for that habitat, whereas the red circles are the median percentage of all the countries' protection efforts. (Source: Authors)*

servation effectiveness, other measures are required outside of reserves to ensure success (e.g., Lipcius et al. 2005; Gaines et al. 2010). This concern applies also and increasingly to climate change and ocean acidification. It is therefore important that all areas of the ocean are managed, including global measures to improve the sustainability of fisheries and aquaculture (Costello et al. 2019; Widjaja et al. 2020; Duarte et al. 2020), as well as of industries extracting non-living resources. As such, it will be important to implement zoning or marine spatial planning to include all areas of EEZs and ABNJ to reduce competition between ocean uses (e.g., Norse 2005) and to reduce the occurrence of pollution from all sources (Duarte et al. 2020) as well as opportunities for alien species to invade non-native ecosystems (Molnar et al. 2008). Reducing and mitigating greenhouse gas emissions to hold global temperature increases to 1.5oC or below is also a priority (IPCC 2019; Duarte et al. 2020) in which the ocean has a role to play (Hoegh-Guldberg et al. 2019).

7 International Conventions and Agreements

We have identified 23 international conventions and agreements that relate to protection of the marine environment and biodiversity (Table 10.4). It is important to consider that these conventions and agreements are not exhaustive in terms of the binding obligations on states. Below the level of international conventions and agreements are regional and sub-regional conventions and agreements (e.g., for RFMOs) as well as voluntary actions such as the CCRF (for a list of examples, see Friedman et al. 2018). Also, decisions under the governance framework of such conventions and agreements, as well as by their implementing agencies, put further binding obligations on states. Added to this is national legislation which provides a complex and interacting web of marine legislation (for an example based on Europe, see Boyes and Elliott 2014). Therefore, the absence of a 'yes' in Table 10.4 does not necessarily mean that a signatory state is not obliged to conform to the activity in the column.

Table 10.4 Characteristics of the International conventions and agreements to protect marine biodiversity and environments

Convention/Agreement	A	B	C	D	E	F	G	H	I	J	K
1. IWC	Yes	Yes		Yes		Yes					
2.Convention on fishing	Yes									Yes	
3. Convention on high seas oil casualties								Yes		Yes	
4. Ramsar	Yes	Yes	Yes	Yes		Yes				Yes	Yes
5. Dumping convention						Yes		Yes		Yes	Yes
6.Heritage Convention			Yes	Yes						Yes	Yes
7. CITES	Yes	Yes				Yes				Yes	
8. Marine pollution (not oil)								Yes		Yes	
9. Marpol						Yes		Yes		Yes	
10. CMS	Yes	Yes	Yes	Yes		Yes				Yes	
11. UNCLOS	Yes	Yes	Yes			Yes	Yes	Yes	Yes	Yes	Yes
12. Basel						Yes		Yes		Yes	Yes
13. CBD	Yes	Yes	Yes	Yes		Yes	Yes		Yes	Yes	Yes
14. High seas fisheries compliance	Yes									Yes	Yes
15. Part XI UNCLOS							Yes			Yes	Yes
16. Straddling stocks agreement	Yes	Yes			Yes	Yes		Yes		Yes	Yes
17. Protocol marine pollution					Yes	Yes	Yes	Yes		Yes	Yes
18. Cartegena							Yes		Yes	Yes	Yes
19. Stockholm						Yes	Yes	Yes		Yes	Yes
20. Antifouling					Yes	Yes	Yes	Yes		Yes	
21. Ballast						Yes	Yes		Yes	Yes	Yes
22. Port state measures	Yes									Yes	Yes
23. Nagoya	Yes	Yes	Yes							Yes	Yes

A. Sustainable management of living resources; B. Sustainable management of unexploited species; C. Habitat management or protection; D. Implement protected areas; E. Precautionary principle; F. Monitoring of species, habitats or environment; G. Environmental impact assessment; H. Prevention of environmental pollution; I. Biosecurity; J. Encourage or impel international cooperation; K. Capacity building

Notes: a. Where trade in that species may impact on an endangered species. The conventions and agreements are as follows: (1) International Whaling Convention (1946); (2) Convention on Fishing and Conservation of the Living Resources of the High Seas (1958); (3) International Convention Relating to Intervention on the High Seas in Cases of Oil Pollution Casualties (1969); (4) Convention on Wetlands of International Importance Especially as Waterfowl Habitat (Ramsar; 1971); (5) Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (1972); (6) Convention Concerning the Protection of the World Cultural and Natural Heritage (1972); (7) Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES; 1973); (8) Protocol Relating to Intervention on the High Seas in Cases of Marine Pollution by Substances Other than Oil (1973); (9) Protocol of 1978 Relating to the International Convention for the Prevention of Pollution from Ships, 1973, (Marpol); (10) Convention on the Conservation of Migratory Species of Wild Animals (CMS or Bonn Convention; 1979); (11) United Nations Convention on the Law of the Sea (UNCLOS; 1982); (12) Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal (1989); (13) Convention on Biological Diversity (CBD; 1992); (14) Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas (1993); (15) Agreement Relating to the Implementation of Part XI of the United Nations Convention on the Law of the Sea of 10 December 1982 (1994); (16) Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks (1995); (17) Protocol to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, 1972 (1996); (18) Cartegena Protocol on Biosafety to the Convention on Biological Diversity (2000); (19) Stockholm Convention on Persistent Organic Pollutants (2001); (20) International Convention on the Control of Harmful Anti-Fouling Systems on Ships (2001); (21) International Convention for the Control and Management of Ships' Ballast Water and Sediments (2004); (22) Agreement on Port State Measures to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated Fishing (2009); (23) Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from Their Utilization to the Convention on Biodiversity (2010)
Source: Authors

Notwithstanding this, Table 10.4 provides an overview at the highest level of what what ocean management measures states have enacted to protect marine biodiversity.

The 23 international treaties to protect the marine environment and conserve marine biodiversity were analysed using clustering and were found to fall into in three hierarchal groups (Fig. 10.12a): those that aim to protect biodiversity, those dedicated to fisheries and regulation of anthropogenic activities (navigation, ballast waters, etc.) and those regulating pollution.

Beginning more than 60 years ago, the International Whaling Convention (1946) was aimed at the sustainable management of whaling but also concerns protected areas specifically targeted at whale conservation. Almost all the international treaties since then have required cooperation between countries; capacity building; monitoring of species, habitat or the environment; and the management of living resources (Fig. 10.12b). In the last three decades, they have evolved to include a wider range of considerations, including prevention of pollution, conservation of non-commercial species and habitats and biosecurity (Fig. 10.12c). However, many of these treaties focused on specific sectors (e.g., pollution or fisheries management; see Fig. 10.12a) with some specifically dealing with a narrow range of issues (e.g., the Cartagena Protocol relating to biosecurity of organisms modified through biotechnology). Of the 23 conventions, 11 represent the sustainable management of living resources in the ocean and 10 pertain to preventing damage to the marine environment by pollution. It is notable that only 8 conventions and agreements deal with managing or conserving species which are not fished commercially, and only 6 protect marine habitats. Five of the conventions or agreements specifically require the implementation of MPAs.

7.1 Fisheries Governance, Sustainability and Impacts on Biodiversity

On the face of it, the range of international and sub- international conventions and agreements would appear to adequately manage the marine environment and biodiversity. However, as outlined in Sect. 3 of this report, marine species and habitats are in decline, and this amounts to a loss in the provisioning of ecosystem services. For fisheries, this has a significant impact in economic terms; for example, the Sunken Billions report suggests that lost revenue resulting from overfishing amounted to $83 billion in 2012 (World Bank 2017).

Improved management and judicious conservation of wild fisheries would lead to increased biomass in the ocean, higher profits for fishers and greater food provision (40% more production in the future than under business as usual and 20% more than now; Costello et al. 2019; see also World Bank 2017).

No fewer than 11 conventions and agreements deal with the sustainable management of living resources, and all but 3 of them also cover non-target species (Table 10.4). This does not include the large number of regional and sub-regional agreements and additional binding measures that states are committed to for fisheries (Friedman et al. 2018). As already indicated in Sect. 6 the problem in fisheries management is one of uneven implementation of measures to increase sustainability of catches of target species and to prevent harm to biodiversity. There are many aspects of fisheries management where this unevenness of implementation is apparent. For example, compliance to the FAO's CCRF, one of the primary pillars in placing biodiversity measures in fisheries management (Friedman et al. 2018), is better in developed countries than in developing ones, but for most it falls far short of 'good' (Pitcher et al. 2009). Likewise, RFMOs have been widely criticized for their performance both in terms of managing target fish stocks on the high seas and also bycatch (Cullis-Suzuki and Pauly 2010; Polacheck 2012; Gilman and Kingma 2013; Gjerde et al. 2013; Gilman et al. 2014; Clark et al. 2015; Leroy and Morin 2018; Pentz et al. 2018). Since 2006, the United Nations General Assembly has called for the development of performance reviews (PRs) for RFMOs (Haas et al. 2019). By 2016, all RFMOs which had entered into force by 2012 had undergone PRs, and some have been reviewed twice (Haas et al. 2019). There is evidence that these reviews have led to improvements, particularly in the areas of compliance and enforcement, conservation and management and international cooperation (Haas et al. 2019). Decision-making and dispute settlement and financial and administrative issues were areas where lower improvement scores were obtained (Haas et al. 2019). Other recent reviews of RFMO performance reveal a more mixed picture of improvement (Gjerde et al. 2013; Gilman et al. 2014; Pons et al. 2018).

An analysis of the drivers of management effectiveness in tuna RFMOs identified that those with a greater number of member countries, a greater economic dependency on the fisheries, a lower mean GDP, a greater number of fishing vessels and a higher proportion of small vessels had lower levels of research, management and enforcement (e.g., the Indian Ocean Tuna Commission; Pons et al. 2018). There are multiple issues within RFMOs, but those most pertinent to biodiversity conservation include the fact that fisheries management has paid insufficient attention to the environmental management of a broader range of natural assets (Gilman et al. 2014; Hooper et al. 2019). In the analysis on tuna RFMOs by Pons et al. (2018), it was noted that scores

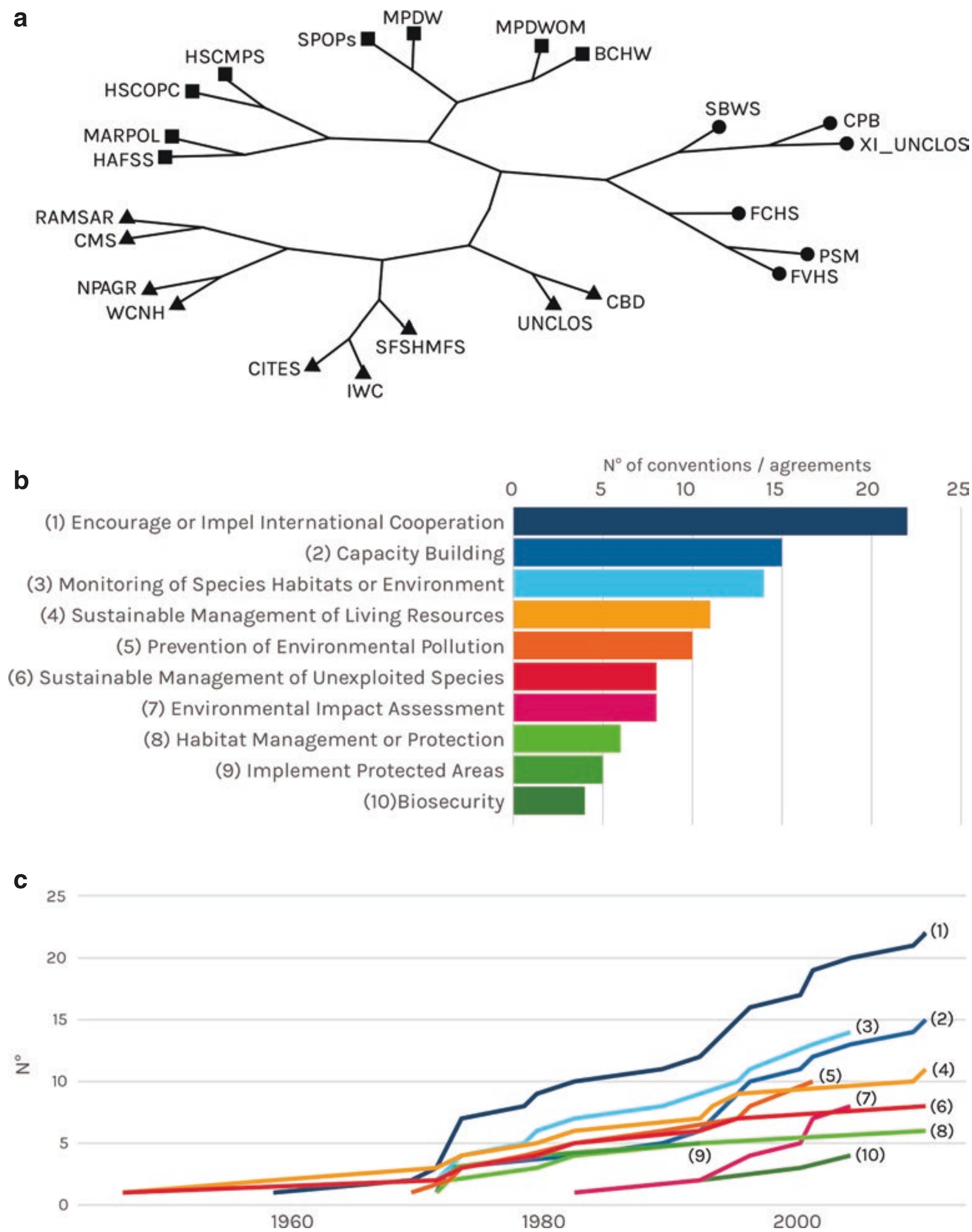

Fig. 10.12 Analysis of 23 International treaties to protect the marine environment and conserve marine biodiversity. *Notes: Panel* (**a**) *shows Ward's hierarchical clustering with Euclidean distance of international conventions/agreements according to their mission topics; the convention acronyms are as follows: BCHW basel convention on the control of transboundary movements of hazardous wastes and their disposal, CBD convention on biological diversity, CITES convention on International trade in endangered species of wild flora and fauna, CMS convention on the conservation of migratory species of wild animals (or Bonn), CPB cartegena protocol on biosafety to the convention on biological diversity, FCHS convention on fishing and conservation of the living resources of the high seas, FVHS agreement to promote compliance with international conservation and management measures by fishing vessels on the high seas, HAFSS International convention on the control of harmful antifouling systems on ships, HSCMPS protocol relating to intervention on the high seas in cases of marine pollution by substances other than oil, HSCOPC International convention relating to intervention on the high seas in cases of oil pollution casualties, IWC International Whaling Commission, Marpol protocol of 1978 relating to the International convention for the prevention of pollution from ships, MPDW protocol to the convention on the prevention of marine pollution by dumping of wastes and other matter, MPDWOM convention on the prevention of marine pollution by dumping of wastes and other matter, Ramsar convention on wetlands of international importance especially as waterfowl habitat, SFSHMFS agreement for the implementation of the provisions of the United Nations convention on the law of the sea of 10 December 1982 relating to the conservation and management of straddling fish stocks and highly migratory fish stocks, SPOPs Stockholm convention on persistent organic pollutants, UNCLOS United Nations Convention on the Law of the Sea, WCNH convention concerning the protection of the world cultural and natural heritage, XI_UNCLOS agreement relating to the implementation of Part XI of the United Nations Convention on the Law of the Sea of 10 December 1982. Panel* (**b**) *shows the number of conventions/agreements associated towards a main goal as listed in Table 10.4; Panel* (**c**) *shows how the number of each conventions/agreements changed over time for each main goal. (Source: Authors)*

for fisheries management in general were low and, in particular, for discarding and bycatch measures. This was attributed to a lack of severe consequences for exceeding bycatch quotas, with the result that non-target species such as marlins and sharks scored low for all management dimensions (Pons et al. 2018). Application of the precautionary principle can be useful in such cases, but this has been included in few international agreements or conventions (Table 10.4), although its use in RFMOs is spreading (de Bruyn et al. 2013).

Illegal, unregulated and unreported (IUU) fisheries contribute significantly to the overexploitation of fish stocks as well as impacts on biodiversity. They are a particular problem for commercial species, which acquire a high value because of their increasing scarcity. Examples of such species include several croakers, giant clams and red corals (Zhang and Wu 2017). These IUU vessels do not adopt fishing practices to avoid bycatch or other forms of environmental damage (Petrossian et al. 2018). A very sad example of this is the imminent extinction of the vaquita (*Phocoena sinus*), a porpoise found in the Sea of Cortez. The vaquita is suffering high mortality as bycatch in illegal gill nets set for the totoaba (*Totoaba macdonaldi*), a croaker whose swim bladder is prized in Chinese medicine and which is also endangered with extinction (Jaramillo-Legorreta et al. 2019).

What is less recognised is the role of state-corporate crime in marine fisheries (Standing 2015). This is an issue in developing coastal states where fisheries access agreements are used to allow foreign fishing vessels into their waters. There is ample evidence that the licensing coastal states and the vessels' flag states often ignore overfishing, corruption and the significant losses to the livelihoods and incomes of local small-scale fisher folk (e.g., Belhabib et al. 2015; Standing 2015; Zhang and Wu 2017). States can use their political and economic power to impose such agreements on countries, even where there is awareness of the likely outcome in terms of overfishing and negative societal impact (Standing 2015; Zhang and Wu 2017). There is also a significant role in such activities by business elites and global investment companies (Standing 2015). This is further exacerbated when political issues arise, such as in the disputed waters of the South China Sea (Zhang and Wu 2017).

Whilst fisheries impacts are not the only drivers of loss of species and habitats in the ocean, they illustrate the barriers to tackling the biodiversity crisis. Setting specific targets as policy objectives and then ensuring that their progress is monitored and reported on is crucial. Despite the objectives of increasing MPAs under the CBD (and other conventions and agreements), it was the adoption of Aichi Biodiversity Target 11 that has spurred the international community to reach a specific goal of 10% of coastal and marine areas, which are in ecologically representative and well-connected protected areas or other forms of spatial conservation management.

Likewise, SDG 14 has reinforced Aichi Biodiversity Target 11 by also calling for the protection of 10% of coastal and marine areas (SDG 14.5); the elimination of overfishing, IUU fishing, and destructive fishing practices (SDG 14.4); and the prohibition of fishing subsidies which enhance overcapacity and overfishing and which contribute to IUU fishing (SDG 14.6).[9] These targets also come with indicators against which progress can be monitored. By setting such clear goals and guidelines for reporting progress, coastal and flag states can better manage their ecosystems (Lidström and Johnson 2019).

Along with the clear setting of targets for achieving standards of fisheries sustainability, biodiversity and environmental protection, high seas fisheries management organisations should be operating to clear international standards and a system of monitoring progress to achieve such standards should also be put in place. Further improvement in the sustainability of fisheries can also be achieved by using innovative technologies to improve the monitoring of fishing activities and catches (Kroodsma et al. 2018; Bradley et al. 2019) as well reducing bycatch (Avery et al. 2017) and other environmental impacts of large- and small-scale fisheries. Implementing these measures will require adequate funding and increased capacity, especially amongst developing coastal states (Friedman et al. 2018).

A significant improvement in fisheries management would also be attained through the adoption of several voluntary codes and guidelines as clear international standards for management of fisheries (e.g., the FAO's CCRF, 1995, and Voluntary Guidelines for Flag State Performance, 2014), but again, without mechanisms for monitoring and reporting such standards will be slow in improving performance.

The implementation of new conventions and agreements should also be more rapid, and we note that the Agreement on Port State Measures to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated Fishing 2009) to date only has 61 parties. A new implementing agreement for UNCLOS, known as the biodiversity beyond national jurisdiction (BBNJ) agreement, currently under negotiation, represents a step forward in putting in place a framework for spatial conservation and other measures to protect biodiversity in ABNJ. The text of this agreement contains strong provisions for monitoring and reporting on progress in implementation as well as the establishment for international standards through the operation of a Scientific and Technical Committee and a decision body (e.g., a Conference of Parties

[9]For more information on SDG 14, see https://sustainabledevelopment.un.org/sdg14

in collaboration with existing agreements and implementing agencies). It also includes the precautionary principle and significant improvements in transparency and the involvement of civil society in aspects of decision-making, particularly in processes related to environmental impact assessment. The inclusion of provisions for capacity building and technology transfer among states in the BBNJ agreement may also be extremely important not just for improving the capacity of developing states to monitor and manage biodiversity in ABNJ but also within their own coastal waters.

8 Opportunities for Action

The IPBES Global Assessment Report identifies that biodiversity is declining faster than at any other time in human history, and rates of species extinction are likely tens to hundreds of times higher than any time in the last ten million years (IPBES 2019). Despite the data limitations, we have presented evidence in this paper that marine ecosystems, like their terrestrial and freshwater counterparts, are suffering from severe habitat degradation, species population reductions and ecosystem impacts at multiple levels, with significant consequences to society through loss of ecosystem services provision which is the cause of direct economic losses, impacts on livelihoods and ultimately on human health and security.

Although these findings present a gloomy prospect for the future there are notable successes in reversing the decline of marine species through strong management and conservation measures (Duarte et al. 2020). The most notable of these is the recovery of populations of the great whales following the moratorium of whaling imposed by the IWC (Duarte et al. 2020). As related in the present report, reduction of fishing fleet capacity, coupled with modern fisheries management approaches and strong monitoring, control and enforcement has led to the stabilisation and recovery of fish stocks in the waters of Europe, the United States and elsewhere (Fernandes et al. 2013, 2017; Hilborn and Ovando 2014; Rousseau et al. 2019; Hilborn et al. 2020). Some habitats have also showed some recovery from past losses, an example being the recovery in seagrass beds in northern Europe (de los Santos et al. 2019).

This recovery was attributed to management actions including those reducing coastal pollution, measures to prevent anchoring and trawling in seagrass beds, as well as natural recovery (de los Santos et al. 2019). There are also examples of habitat restoration leading to local rehabilitation of habitats such as mangrove forests in the Mekong Delta (Duarte et al. 2020). Duarte et al. (2020) suggest that strong management action could lead to substantial recovery of abundance of species and structure, function of communities with increased provision of ecosystem services by 2050. Given the evidence for strong recovery of species and some recovery of specific habitats over decadal timescales we believe that such optimism is justified. However, recovery will only take place at large scales following strong and coordinated management action. Based on this evidence and our analysis of drivers of biodiversity loss, we find these opportunities for urgent action at local to international levels.

There are opportunities to improve monitoring, increase efficiency in MPAs, and achieve sustainable ecosystem-based fisheries management. Some specific actions/ deliverables for these high-level policy decisions include no net loss of habitat; establishing a blue bond market for investing in marine environmental sustainability; marine spatial planning to identify (on a regional basis) best options to increase no-take areas, including in the vicinity of offshore renewable energy projects; moving intensive aquaculture operations offshore, where feasible; and planning conservation responses to future coastline inundations (e.g., determining where the new sea grass meadows and mangroves will exist with sea level rise). Bringing the entire ocean under sustainable management is also a critical element in reducing open access and overexploitation of resources which has led to declines in marine species and ecosystems (Norse 2005).

8.1 Technology for Mapping

Technological advancements in remote sensing, including satellites, lidar, unmanned aerial vehicles, AUVs, and the computational ability to process such multidimensional big data in the past few decades has drastically expanded our capacity to understand the world. With increasing spatial and temporal resolution of the data captured, there is a large opportunity to further enhance our understanding of the status and trends in marine habitats and ecosystems, the drivers of change and the impacts of degradation on their contribution to people and, thus, improved visualisation and maps to support the decision-making process. The advancements in the field of artificial intelligence have also paved the way for the application of data mining and natural language processing into biodiversity and ecosystem studies. Therefore, marine scientists have the unique opportunity to extract knowledge from historical and unstructured sources (e.g., text, images, audio), store complex information in machine-readable formats and connect with expert systems to set up knowledge bases—all areas of marine science that have yet to be well explored. For effective management, governments need to know where, what, why, and how much of an activity is sustainable because anthropogenic impacts expand into deeper and deeper waters (Baker and Harris 2020).

However, there are challenges to overcome with regard to harnessing the above-mentioned technological advancements into global marine studies. Utilising the technological advancements into a thematic discipline requires multidisciplinary experts, dialogue and knowledge exchange across disciplines as well as basic scientific programming skills and knowledge of machine-readable data and metadata formats. The lack of interoperable web services and a catalogue for referencing remote sensing products and geospatial data sets limits the smooth communication of needs from a thematic discipline to the technology developers.

There is an opportunity for NGOs, industry, researchers, and government institutions to collaborate to increase the application of current advancements in technological capacity. To accomplish this cross-disciplinary discussion, there needs to be an exchange of knowledge, and scientists need to be trained to make their analysis and work interoperable. Streamlined services are also needed to support the production of standard essential variables and indicators in the field, including a catalogue of key data sets, which would integrate a wide variety of primary data, and standardised processing services (i.e., web rest services), which would improve access and maintain frequently used data resources.

We envision that by 2030 a catalogue of marine habitats, including those that we currently have limited information on, such as kelp forests and rocky reefs, will have their EOVs monitored spatially and temporally, and variation and distribution changes within them will be automatically generated over time and publicly accessible. We support the development of a comprehensive ocean observing system which has been identified as a priority for the United Nations Decade of Ocean Science for Sustainable Development and GOOS. With this information accessible, organisations can effectively monitor the global distributions of economically important marine habitats, such as coral reefs, mangroves and seagrasses. On a local level, governments should collaborate with industry and NGOs to effectively map drivers of habitat degradation and ground truth the data produced from the global habitat mapping efforts. Such mapping and monitoring of marine ecosystems has been among recommendations for improved management of marine biodiversity for almost 30 years (Norse 1993).

To be able to develop the collaborations and technological capacity to make this vision a reality, we suggest the following high-priority opportunities for action:

- The present intergovernmental organisations (e.g., UNESCO-IOC), biodiversity monitoring networks (GOOS BioEco, GEO BON/MBON), databases (e.g., OBIS) and philanthropic efforts involved in gathering and making ocean data available for management purposes (e.g., Google Earth Engine; Ocean Data Foundation)[10] require a coordinated approach to face the challenge of comprehensive and global monitoring of biodiversity. These organisations, under the leadership of UNESCO-IOC, in partnership with national ocean biodiversity monitoring networks (e.g., IMOS, IOOS, EuroGOOS) and the CBD, should— through workshops or other means—create maps of both habitat extent and environmental drivers to identify conflicts and gaps in knowledge, including in the distribution of marine habitats, technological limitations and solutions with explicit goals and institutions/organisations assigned to meeting the goals. These efforts should include multidisciplinary scientists, including, but not limited to, marine, artificial intelligence and data experts.
- The Decade of Ocean Science for Sustainable Development provides an ideal jump-off point for such a coordinated approach to ocean biodiversity monitoring, especially as it recognises the importance of producing actionable data but will also produce significant new data sets on species and habitat distribution in the ocean.
- By 2025 this should culminate in collaborative research platforms where global habitat maps and EOVs can be compiled based on interoperable data sources, be visualised and be made publicly available in a way that facilitates ecosystem-based management of human activities in the ocean whilst enabling biodiversity conservation.
- By 2028, integration of novel technological developments with quality-control standards increase temporal resolution of habitat maps and drivers so that quality annual maps of habitat extent and impacts are made available.
- Throughout 2020–30, knowledge bases and technology transfer between governments is promoted to equip all countries with the tools necessary to sustainably manage and map the ocean. Capacity-building efforts are targeted at providing all countries with the expertise to access and act upon biodiversity data for meeting international targets and ocean management needs.

- By accomplishing these goals, we believe there will be numerous additional benefits past increasing our understanding of the planet, including improved environmental and biodiversity monitoring plans, technological advancements, the training of new generations of scientists from diverse backgrounds and increased collaboration between stakeholders.

8.2 Addressing the Biodiversity Data Gap

There is a pressing need for a greater coordinated effort to gather information on marine biodiversity and extinction risk, from baselines of diversity and ecosystems to the long-term monitoring of population genetics, species, habitats and

[10] Information about the Ocean Data Foundation can be found on its website, https://www.oceandata.earth/

ecosystems. Again, despite recommendations to develop such coordinated knowledge gathering on marine biodiversity, as well as improving the capacity to do so by all nations nearly 30 years ago (Norse 1993), this has not happened to date. The IUCN Red List shows that although there are a good range of assessments for marine vertebrates (fish, seabirds, marine mammals), extinction risk assessments on marine invertebrates are restricted to a few scattered groups.

There is now an opportunity for states, intergovernmental organisations, foundations and other philanthropic organisations to invest in the infrastructure, including human resources, to meet their international commitments (e.g., under the CBD) to establish baselines of biodiversity and long-term monitoring of the status of species and habitats both within their EEZs and in ABNJ, especially where their flagged vessels are or will be undertaking activities such as fishing or other extractive activities. Such an effort should focus on the already established networks for biodiversity monitoring, including GOOS BioEco and the marine component of GEO BON, MBON.

The first has developed a framework and a globally coordinated strategy for monitoring biodiversity change using biological EOVs which are complemented by the EBVs coordinated by the latter. Data repositories already exist to receive such information (e.g., OBIS; Navarro et al. 2017). GOOS BioEco is facilitating the establishment of coordinated networks to implement monitoring of these essential variables. These will be established in collaboration with MBON and will include oceanographic research centres, government institutions and universities, and natural history museums. These networks should also build on existing efforts, such as the Global Coral Reef Monitoring Network.

By establishing such networks, states will be able to establish a baseline of marine biodiversity in their waters and in ABNJ, allowing the subsequent monitoring of changes in biodiversity through time. This will enable the continual assessment of the success of measures to reduce biodiversity loss by states and allow them to actively manage their activities to mitigate or reverse biodiversity loss. For developing states, assistance in capacity building will be required. Associated benefits from such an effort will include

- maintenance or enhancement of marine ecosystem services provision (e.g., fisheries, coastal protection, tourism);
- identification of marine genetic resources (Blasiak et al. 2020);
- the training of a new generation of marine scientists;
- increased opportunities for citizen science and education; and
- increased effectiveness of investment in biodiversity conservation through specific targeting of interventions.

At present, there are no alternative measures to achieve such a goal, and without it, undocumented biodiversity loss will continue in the face of pressures arising from poverty, the increasing human population and the drive for economic development. We envision a pathway to improved biodiversity monitoring to include the following milestones:

- The identification or establishment of national centres for marine biodiversity monitoring and developed capacity in taxonomy and field ecology, including training in new taxonomic tools such as environmental DNA (eDNA) and other emerging technologies, to undertake baseline assessments and long-term monitoring.
- A baseline biodiversity inventory and the establishment of key monitoring sites as part of the GOOS BioEco networks or of an existing MBON and expanding geographic coverage through the establishment of new MBON sites/regions (2023–25).
- The coordination of biodiversity monitoring activities at a regional basis implementing best practices to exchange knowledge, deliver FAIR and open-access data and share resources where appropriate (2020–25).
- The establishment of a marine biodiversity programme that feeds into national policies and management actions to mitigate biodiversity loss as well as into regional organisations, such as RFMOs, to manage activities in a way as to protect and conserve biodiversity. Biodiversity management becomes embedded into national institutions and legislation and into regional bodies (2025–30+).

There are a range of habitats formed by foundation species that are overwhelmingly important to biodiversity because they are connected to ecosystem functions over a wider geographic area than their immediate occurrence. These include, most notably, coral reefs, mangrove forests, seagrass beds, saltmarshes, kelp forests and other coastal ecosystems. In ABNJ, these are probably strongly represented within EBSAs and may include habitats such as seamounts.

We recommend that coastal states and regional ocean management organisations should adopt a policy of zero net loss for such ecosystems. Because the costs of habitat restoration are often much higher than conservation (Friess et al. 2019), such a policy should prioritise avoidance of activities which lead to significant damage in the first place.

We believe that by establishing or further developing a national MBON coordinated at a regional level, including ABNJ, it could—if used to support effective management and conservation—help to improve and secure economic and other societal gains from the provisioning of ecosystem services. Additional benefits from developing marine genetic resources (Blasiak et al. 2020) and improving environmental awareness and education within society are difficult to estimate but would certainly be positive.

8.3 Citizen Science and Education Programmes

Citizen science provides a great opportunity to increase public participation in science, overcome significant barriers to the scientific process and improve natural resource management (Theobald et al. 2015; McKinley et al. 2017). Citizen science and environmental education programmes are also scientific projects that can produce reliable information in which members of the public directly engage in research to answer particular questions (Parrish et al. 2018; McKinley et al. 2017). Biodiversity-related projects have been shown to span greater geographic and temporal ranges than conventional academic research, engaging millions of volunteers and generating up to $2.5 billion in kind annually (Theobald et al. 2015). There are many goals and benefits for citizen science, spanning publishing results in peer-reviewed journals, education, community empowerment and personal fulfilment (Parrish et al. 2018).

Despite many long-term citizen science projects creating robust data sets,[11] many academic researchers still show a bias against citizen science (Bonney et al. 2014). Theobald et al. (2015) found that only about 12% of projects out of 388 provide data to scientific publications. Therefore, methods of quality assurance (actions taken to ensure the quality of measurements taken) and quality control (post hoc actions to ensure the quality of results) are pivotal to many projects where the primary goal is science generation and should continue to be developed (Bonney et al. 2014; McKinley et al. 2017). A participant's time and success in mastering a task is a function of the complexity of the task (Sauermann and Franzoni 2015), which supports that projects should be simply designed at scale, and projects at smaller scales, with higher complexity, can be more involved (Parrish et al. 2018).

Citizen science programmes can also generate significant social outcomes, including increasing science education, engagement in policy and collaboration. As such, they represent the following opportunities for action:

- Governments increase general science education in line with SDG 4 to 'Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all' (2020–25).
- Citizen science programmes coordinate and organise to ensure that the wealth of information gathered is accessible, usable, known to decision-makers and connected with networks of biodiversity monitoring, including GOOS BioEco and the marine component of GEO BON, MBON, starting in 2023.
- Industry and governments that benefit from this information provide increased funding for the development of community-based programmes in developing countries to increase exposure to science and raise a new generation of scientists by 2025.
- Academia generates best practices and resources to increase the amount citizen science can be used to generate robust data and science, thus removing the bias against this information by 2030.

By accomplishing the previous recommendations, we see a future defined by increased scientific literacy around the world, improved efficiency of moving conservation science into conservation action, and higher awareness and knowledge of the planet around us.

8.4 Well-Enforced, Green-Listed, Fully Protected Marine Reserves

There is strong evidence that the implementation of well-enforced, fully protected MPAs that include 30–40% of key marine habitats will conserve biodiversity, enhance biomass and abundance of marine life as well as improve the resilience of marine ecosystems (Roberts et al. 2001; Lester and Halpern 2008; Gaines et al. 2010; Sciberras et al. 2013; Edgar et al. 2014; Mellin et al. 2016; Sala and Giakoumi 2017). These MPAs can also benefit fisheries (Roberts et al. 2001; Gaines et al. 2010; Di Franco et al. 2016; Ban et al. 2017), provide coastal protection (Roberts et al. 2017) and improve the resilience of ecosystems against the impacts of climate change (Mellin et al. 2016; Roberts et al. 2017). However, poor capacity for the enforcement of MPAs (Gill et al. 2017) and poverty alleviation—specifically, the generation of jobs (Cinner et al. 2009; Gurney et al. 2014)—can undermine MPA objectives. Additionally, the social impacts of protected areas are poorly understood largely because MPA evaluations have tended to focus on one or very few outcomes, and few have had the requisite data to assess causal effects (Gurney et al. 2014). Opportunities over the next two years (e.g., the BBNJ agreement and the CBD Conference of Parties in 2021) offer the chance to adopt a new target beyond the 10% of marine protection and to accelerate the slow progress made to date. Whatever targets for biodiversity protection are set, they must represent the full range of marine ecosystems and species. The aims should include no net loss of important habitats which structure marine ecosystems, such as coral reefs, mangrove forests, seagrass beds, saltmarshes and others.

Experts, conservation practitioners, philanthropic organisations and representatives from government should come together convened by the IUCN, the United Nations Environment Programme (UNEP) and the CBD to establish

[11] See eBird (https://ebird.org/home), COASST (https://coasst.org/) and Zooniverse (https://www.zooniverse.org/)

the best strategy for increasing and improving existing MPAs on the basis of the approach we have outlined in this paper for coastal states.

Strategies tailored for each group of countries—and ultimately each individual country—can be developed, and international assistance, including economic, capacity building and technical advice, can be targeted to effectively achieve global, regional and national targets. For ABNJ, a different approach can target areas of conservation importance whilst balancing these with economic need. The framework developed by O'Leary et al. (2018), with input from the CBD EBSA process, offers a practical approach to achieve this. We envision the pathway as follows:

- The MPA targets are established internationally, at the CBD's Conference of Parties or (for the ocean) at the United Nations Ocean Conference in 2021.
- An implementation conference is initiated to identify specific targets at global, regional and national levels to protect representative marine ecosystems and the best strategic approaches and practical measures to achieve these targets. The conference should be convened by the IUCN, UNEP and the CBD, with attendance from experts and governmental, intergovernmental and non-governmental organisations as well as potential funders (Global Environment Facility, government-funding agencies, private philanthropists and foundations). The target year for the conference is 2022.
- By 2022, a large campaign and economic support should be in place to involve communities and stakeholders to implement community-based MPAs (Pollnac et al. 2001; Aburto-Oropeza et al. 2011). By 2023, a global map to implement community- based MPAs should be generated by states. In the Philippines, where government policy, international aid, universities and NGOs have invested a great effort to implement community-based MPAs, there are over 400 of these management areas. Although only 25% of them are effective in the protection of the resources, clear common factors have been described as the path to successful community-based MPAs: (1) relatively small communities, (2) community census statistics to prioritise targeted interventions, (3) overfishing challenges, (4) movement to alternative income projects, (5) increased level of community participation in decision-making, (6) strong local leadership, (7) receiving scientific and MPA-implementing advice and (8) closely working with local or municipal governments (Pollnac et al. 2001; Crawford et al. 2006; Rossiter and Levine 2014). These small but successful examples of community- based MPAs have proven that not only is it possible to recover marine biodiversity in a short time period (one decade), but they are also producing significant economic benefits for local communities. Cabo Pulmo National Park in Mexico is considered a success according to both biological and social measures: the MPA has seen significant recovery of biomass (Aburto-Oropeza et al. 2011) and demonstrable community engagement and participation, along with extensive socio-political support (and media attention) at the local, national and international levels. Cabo Pulmo has achieved a kind of symbolic power in the world of marine conservation (Anderson 2019), and it has influenced the transition of a governance system into a new, adaptive tourism model (Langle-Flores et al. 2017). There is a need for scaling up community-based MPAs to increase the social and ecological benefits for coastal areas. Evaluating approaches has demonstrated that 'opportunistic approaches' and 'donor-assisted approaches' do not create the necessary outcomes requested by global conservation targets. Rather, a systematic conservation planning approach of community-based MPAs can improve ecological and social outcomes, particularly if this planning incorporates equity for stakeholders (Kockel et al. 2019).
- The implementation conference should lay out a clear road to attaining established targets, with appropriate milestones (2023–30). We suggest that a single agency be tasked with measuring progress towards milestones and the final targets (e.g., UNEP- WCMC). Reports should be produced for the CBD's Conferences of Parties in 2024, 2026 and 2028 prior to 2030. Reporting should also extend to other relevant meetings (e.g., the Our Ocean and United Nations Ocean Conferences).

Balmford et al. (2004) estimated the costs of running a global MPA network covering 20–30% of the ocean at \$5–\$19 billion per annum. However, the potential gain in direct enhancement of fisheries and tourism and the avoided costs in environmental damage through reduction/mitigation of coastal inundation is likely to dwarf these costs. This is without accounting for other ecosystem services, such as CO_2 sequestration, nutrient cycling, waste remediation, protection of marine genetic resources and cultural services, which represent a value in the trillions of dollars overall (Costanza et al. 2014).

Furthermore, we point to the already estimated erosion in the value of marine ecosystem services as a result of the erosion of habitats which amount to a loss of more than \$10 trillion per annum in just over a decade between 1997 and 2011. Much of this loss was focused on coastal ecosystems, with coral reefs losing nearly half their value as a result of the loss of this habitat (Costanza et al. 2014).

8.5 Ecosystem-Based Fisheries Management

There is an extreme urgency to eliminate IUU fishing and accelerate the reform of fisheries management to reflect modern ecosystem-based concepts where biodiversity is managed sustainably alongside target stocks. Both the IPBES Global Assessment Report (IPBES 2019) and our own analyses indicate that overfishing, illegal fishing and destructive fishing practices are the prime drivers of biodiversity loss in the ocean. Whilst much progress has been made in sustainable ecosystem-based fisheries management (Hilborn and Ovando 2014; Friedman et al. 2018; Hilborn et al. 2020), progress remains fragmented. The fishing power of the global fishing fleet is continuing to grow and underlies overfishing in much of the global ocean (Rousseau et al. 2019). We have identified clear barriers to accelerating progress in fisheries sustainability and increasing consideration of biodiversity conservation in fisheries. These barriers include a lack of capacity and funding, whether being associated with institutions or developing states, and overwhelming pressure in some parts of the world to exploit living marine resources exacerbated by growing industrial and small-scale fishing fleets. There is also evidence that in some states, elements of the fishing industry and financial institutions are complicit in allowing overfishing and illegal fishing to continue (Standing 2015; Zhang and Wu 2017). This is not only immensely damaging to biodiversity but also leads to massive economic losses (Costello et al. 2016; World Bank 2017) and the loss of livelihoods and impacts food security (Sumaila et al. 2013; Standing 2015; Freduah et al. 2017). In the face of climate change impacts, overfishing will exacerbate these problems (Badjeck et al. 2010). If biodiversity loss in the ocean is to be halted or reversed, this elephant in the room cannot be ignored.

The reform of fisheries management practices and of the institutions charged with their management is already under way (Friedman et al. 2018). This reform process must be accelerated and driven through the adoption of appropriate targets by the competent authorities. The most important of these reforms include the following:

- Good data underlies all fisheries management both in the context of target species, bycatch species and the environmental impact of fishing. Given the development of modern technologies, from remote sensing to mobile computing and phones, there is an opportunity to greatly improve the monitoring of catches of target and bycatch species in all industrial fisheries. Given the importance of small-scale fisheries in terms of global fishing power, special measures to include these in fisheries catch statistics as well as fisheries management (including co- management/community management arrangements) is critical. Such measures will also allow an assessment of the nutritional and economic benefits of small-scale fisheries at the national level so they are accounted for in decisions on fisheries policy.
- Uniformly adopting modern principles of ecosystem-based fisheries management and the precautionary principle for all fisheries management as expressed in the UN conventions and agreements, the FAO's CCRF and other FAO guidelines and codes.
- Eliminating IUU fishing and other illegal practices in fishing through improved monitoring, control and enforcement. It is especially important that measures to eliminate IUU fishing are adopted rapidly by all fishing and port states.
- Stabilising, and then reducing, fishing pressure should be a priority in regions where growth in fishing capacity continues, undermining efforts to sustainably manage fisheries pressure and to conserve biodiversity. It is critical to ensure that measures to reduce fishing capacity protect the basic needs for food, nutrition and livelihoods in coastal communities, particularly in developing countries.

We also note the opportunities for other important reforms in fisheries management:

- Develop and fund infrastructure and human capacity to enable sustainable management of biodiversity as well as target fish stocks.
- Reform decision-making processes and adopt greater transparency by fisheries management organisations to speed up progress in eliminating overfishing.
- Make all fisheries data public, including data on vessel tracking, catch and bycatch within 12 months of collection.
- Specify measures to address issues of overfishing by developing states and in small-scale fisheries, including investment in data-poor stock assessment methods and the use of reciprocal mechanisms to enhance institutional, management and governance capacity in developing states through finance, training and technology transfer.
- Establish community-based fisheries management to assist in increasing the biological and socio-economic sustainability of fisheries.
- Continue efforts to merge and coordinate the objectives of the fisheries and environmental sectors at all levels of fisheries management (international to local).
- Develop a set of investment standards for the investment in fisheries, and especially infrastructure such as vessels, so only sustainable fisheries/fishing operations are financed.
- Initiate a formal regular review of RFMOs, ensuring they are meeting new standards of fisheries management; the

following areas specifically require attention: (1) updating conventions and agreements to implement modern standards of ecosystem-based fisheries management, including specific provisions for the conservation and protection of biodiversity; (2) further convergence between fisheries and environmental sector governance structures to integrate biodiversity considerations into fisheries management; (3) implementing mechanisms to ensure the rapid and accurate reporting of catches of target and bycatch species; (4) more rigorous target-based efforts to ensure rapid implementation of rules and recommendations; (5) a transformation of transparency for both fisheries-related data and decision-making processes; (6) reforming decision- making structures to prevent 'opt-out' or lowest- common-denominator regulations within fisheries management organisations; and (7) greater clarity on participatory rights, such as allocation of catch levels or fishing effort (Gjerde et al. 2013; Friedman et al. 2018).

- Develop a set of minimum standards for fisheries partnership agreements to ensure (1) sustainable fishing; (2) fair and equitable financial benefits for parties; (3) clear financial structures and reporting arrangements to ensure licence fees or other financial benefits flow to society; (4) adequate arrangements for monitoring, control, surveillance and enforcement of fisheries; and (5) formal structures for dispute resolution amongst partners with arbitration by an impartial third party.

Aichi Biodiversity Target 6 and SDG 14 embody specific targets for fisheries sustainability, and the measures above will clearly help to attain these goals. The SDGs are set for 2030 (with some interim targets due in 2020), but the CBD post-2020 biodiversity framework also provides a timetable for achievement of these goals and an opportunity to finally achieve the objectives of Aichi Biodiversity Target 6. We view the next decade, therefore, as critical in accelerating reforms of fisheries and biodiversity objectives to protect marine living resources.

By adopting these reforms, overfishing and IUU fishing will be eliminated, and fish stocks and associated ecosystems should be able to rebuild. The financial benefits of this just in fisheries revenue alone has been estimated at $83 billion per annum (World Bank 2017). Broader benefits will include increasing fish catches (Costello et al. 2016) and securing both livelihoods and food supplies as well as increasing their resilience to climate change impacts for the future. Given that destructive fishing impacts, such as bycatch, are the main drivers of biodiversity loss for a number of marine species, the benefits of reducing extinction risk and restoring ecosystem function and services provision will be enormous. This will also increase ecosystem resilience against climate change and other impacts.

9 Limitations of the Paper and Conclusions

As identified in several parts of this study, a lack of FAIR and open data on marine biodiversity is problematic when trying to identify patterns of species and habitat diversity as well as changes in these parameters over time. For example, in the IUCN Red List data, many species are classified as DD, and many groups of invertebrates have not been assessed at all. Without this information, it is very difficult to estimate the current state of, and trends in, marine biodiversity in the ocean.

There are significant gaps in our analyses because comparable global data sets were not available for many coastal habitats, including rocky reefs. Within the available data sets, there are many gaps and sampling biases, leading to higher diversity values in areas which likely do not correspond to species or habitat diversity. Likewise, particularly for deep-sea and offshore parts of the ocean, only large-scale oceanic habitats that can be identified through physical features (e.g., seamounts) could be identified, and the water column, the largest ecosystem on Earth, was largely neglected in this study. A trend analysis for the marine habitats examined here was not possible with the current publicly available data but should be pursued in future efforts as outlined in Sect. 8.

Despite these gaps, we have sufficient information to understand the broad state of marine species and habitat diversity to generate effective management responses. However, to reduce habitat loss and degradation, we need an increase in multi-decadal monitoring because it is essential to be able to understand, prevent future damage and monitor potential recoveries of marine ecosystems (Bayraktarov et al. 2016; Gangloff et al. 2016). Monitoring will establish baselines so that we can quantify changes in habitat extent and impacts from anthropogenic activities and use this information effectively to manage our natural resources.

A lack of adequate funding and capacity—particularly in developing countries but also in the organisations charged with sustainably managing economic activities in the ocean—is repeatedly highlighted in this study. Urgent measures are required to build capacity, transfer technology and build the global financial supporting structures so the blue economy can grow in a sustainable fashion that neither depletes marine species or habitats nor undermines the ecosystem services on which humankind relies. Current biodiversity loss in the ocean is at least partially due to a lack of equitability in states' ability to monitor biodiversity and manage activities within their EEZs and ABNJ.

The current crisis of biodiversity loss in the ocean may require developing and implementing further international agreements and national measures to protect habitats and species. A new legally binding instrument under UNCLOS to conserve and sustainably use marine biodiversity of areas beyond national jurisdiction (the BBNJ agreement) is currently being negotiated and should become an important legal framework for the conservation of 50% of Earth's surface area. In addition, new protocols could be developed as part of existing conventions, specifically the CBD, the World Heritage Convention and the Convention on Migratory Species, among others. Such protocols should include provisions that human activities should not result in the long-term or permanent loss of biodiversity in the ocean, with clear mandates for monitoring their effectiveness. They should also lay out renewed commitments for implementing biodiversity protection measures as well as monitoring and data-gathering activities which are already embodied in existing conventions and agreements. These new protocols should apply to all sectors operating in the ocean and should include the broad family of UN specialized agencies, including the FAO and associated RFMOs, the International Maritime Organization and the ISA.

The fisheries reforms described in this Blue Paper would likely cost millions to tens of millions of dollars on a state-by-state basis; yet in economic returns from fisheries alone, there is the potential for billions of dollars in return. Not undertaking these reforms will lead inevitably to commercial and/or local to wide-scale extinction of both exploited and non-target species, undermining ecosystem resilience and service provision. By extending this to the broader values to society and to the restoration of biodiversity and ecosystem services, reforms could be transformative.

The speed of the decline of marine species and habitats means that the opportunities for action we have identified should be taken up with urgency. Such an international effort, spanning all sectors involved in the blue economy as well as the implementing organisations involved in their management, may require a coordinated effort on the scale of that currently addressing climate change. A large-scale global plan of action for ocean biodiversity conservation may be required to expedite these opportunities with the speed required.

Acknowledgements The paper's technical reviewers, Miriam Fernandez Bergia, Stuart Kininmonth and Boris Worm, as well as its arbiter, Martin Stuchtey, all provided helpful technical comments.

The authors also thank the World Resources Institute for providing support as the Ocean Panel Secretariat.

While our colleagues were very generous with their time and input, this report reflects the views of the authors alone. Jorge Assis was supported by the Foundation for Science and Technology (Portugal) through the projects UIDB/04326/2020, BIODIVERSA/004/2015 and the transitional norm—DL57/2016/CP1361/CT0035.

The authors thank LSF Editorial for copyediting and Rosie Ettenheim for design.

About the Authors

Lead Authors

Alex D. Rogers is the Science Director of REV Ocean, a not-for-profit working for a healthier ocean. He is also a Visiting Professor of the Department of Zoology, University of Oxford, a Researcher of the Oxford Martin School and a Senior Research Fellow of Somerville College, University of Oxford. Prof. Rogers has a background in marine ecology, systematics and molecular ecology.

Octavio Aburto-Oropeza is an Associate Professor at Scripps Institution of Oceanography (SIO), a professional photographer associate with the International League of Conservation Photographers and a National Geographic Explorer. His research and photographs have focused on marine reserves and commercially exploited marine species and their fisheries in Latin America and the United States.

Contributors

Ward Appeltans is the marine biodiversity focal point of UNESCO- IOC and since 2012 is the head of the secretariat of OBIS. Before joining UNESCO-IOC he managed WoRMS and was a project leader at the data centre of the Flanders Marine Institute and was involved in several European marine biodiversity projects.

Jorge Assis is a research fellow at the Centre of Marine Sciences and lecturer at the University of Algarve. He has a background in marine ecology and data science and currently focuses his research in addressing the potential consequences of global climate changes (past, ongoing and future) to marine biodiversity.

Lisa T. Ballance is Director of Oregon State University's Marine Mammal Institute, Endowed Chair for Marine Mammal Research and a former marine ecologist with the National Oceanic and Atmospheric Administration (NOAA). Her research focuses on ecology and conservation biology of cetaceans and seabirds, largely in the Pacific and Southern Oceans far from shore.

Philippe Cury is senior scientist at IRD (Institut de Recherche pour le Développement; French National Research Institute for Sustainable Development) and the IRD representative for Europe in Brussels at CLORA (Club des Organismes de Recherche Associés). He is the President of the Scientific Council of the Monaco Institute of Oceanography and one of the two scientific Directors of Euromarine Consortium (www.euromarinenetwork.eu). His research focuses on sustainable management of marine resources in an ecosystem perspective.

Carlos M. Duarte is Professor of Marine Science and Tarek Ahmed Juffali Research Chair in Red Sea Ecology at King Abdullah University of Science and Technology (KAUST), and Professor at the Arctic Research Center, Aarhus University, Denmark. His research addresses marine ecology, conservation and the impacts and responses to human pressures for ecosystems ranging from polar to tropical and from coastal to the deep sea. He is a member of the Expert Group supporting the HLP commissioning this series of Blue Papers.

Fabio Favoretto is a Marine Ecologist at the Universidad Autónoma de Baja California Sur in La Paz, Mexico. He has a background in benthic ecology and currently works on remote sensing applications in coastal environments.

Joy Kumagai is a Research Associate at SIO. Her research focuses on the impacts of anthropogenic activities on ecosystem services and marine biodiversity. Her background is in marine biology and applying Geographical Information Systems to answer research questions.

Catherine Lovelock is a Professor at the School of Biological Sciences at the University of Queensland, Australia. She has investigated the physiology and ecology of mangroves and other coastal plant communities over their global distribution, focusing on climate change and its impacts as well as the role of coastal wetlands in climate change adaptation and mitigation.

Patricia Miloslavich is the Executive Director of the Scientific Committee on Oceanic Research (SCOR), an international NGO to help address interdisciplinary science questions related to the ocean. She is a retired professor from the Universidad Simón Bolívar in Venezuela. Her background is in marine biology, biodiversity and global ocean biological observations.

Aidi Niamir is a scientist at the Data and Modelling Centre of the Senckenberg Biodiversity and Climate Research Institute in Germany. His research and services mainly focus on spatiotemporal models, application of earth observation in biodiversity and ecosystem studies and open geo-information tools to support science-policy interactions. He is a lead author of the IPBES Global Assessment Report on Biodiversity and Ecosystem Services and the head of the IPBES technical support unit on data.

David Obura is a Director of CORDIO (Coastal Oceans Research and Development in the Indian Ocean) East Africa, a Kenyan research organisation focused on coral reefs of the Western Indian Ocean. From a foundation in coral reef resilience, he increasingly works to integrate conservation and development in ways that will be necessary for a sustainable future for coastal systems. He works with a number of global working groups, including the Earth Commission and GOOS BioEco, and chairs the IUCN Coral Specialist Group.

Bethan C. O'Leary is a Research Associate at the University of York and a Senior Research Fellow at the University of Salford. Her research focuses on sustainable management of marine resources, particularly through marine protected areas, and the integration of scientific research with applied environmental management and policy.

Gabriel Reygondeau is a research associate in the Changing Ocean Research Unit at the University of British Columbia and in the Department of Ecology and Evolution at Yale University. He is an oceanographer, ecologist and biogeographer who explores the relationship between climate, biodiversity and human activities.

Callum Roberts is Professor of Marine Conservation at the University of York. His research focuses on threats to marine ecosystems and species and on finding the means to protect them. His team helped design the world's first network of high seas protected areas and provided the scientific underpinning for a new ocean protection target—30% by 2030—which is gaining widespread international support.

Yvonne Sadovy was a Professor at the School of Biological Sciences of the University of Hong Kong for over 25 years after directing the government fishery lab in Puerto Rico. She founded and co-chairs the IUCN Groupers & Wrasses Specialist Group and co-founded and directs Science and Conservation of Fish Aggregations. Her specialty is reef fishes and fisheries, their conservation and management

Tracey T. Sutton is Professor in the Halmos College of Natural Sciences and Oceanography at Nova Southeastern University (Dania Beach, Florida) and Assistant Chair of the Department of Marine and Environmental Sciences. His research entails oceanic ecosystem structure, food web dynamics, organismal ecology, biogeography, fish taxonomy, and deep ocean stewardship, with emphasis on human impacts. He is a member of the Expert Panel for the first and second United Nations World Ocean Assessments.

Derek P. Tittensor is the Jarislowsky Chair in Marine Ecosystem Forecasting at the Department of Biology, Dalhousie University, Canada. He is a marine ecologist with a research focus on biodiversity and human impacts on marine ecosystems. He is also coordinator of the Fisheries and Marine Ecosystem Model Intercomparison Project (Fish-MIP).

Enriqueta Velarde is Senior Researcher/Professor at the Institute of Marine Sciences and Fisheries at the Universidad Veracruzana in Mexico and has 42 years of experience in research projects related to seabird ecology. Main areas of interest include generating a long-term time series of biological and ecological seabird parameters providing information to support demographic, trophic and fisheries predictive models; marine protected area planning as well as genetic and life history research of endangered species in natural protected areas. Other work areas focus on ethno-ornithology, indigenous knowledge and citizen science.

Interactive versions of the major figures in this paper with statistics broken down to coastal state EEZs are available at: https://octopus. zoo.ox.ac.uk/studies/critical-habitats-2020

References

Aburto-Oropeza O, Ezcurra E, Danemann G, Valdez V, Murray J, Sala E (2008) Mangroves in the Gulf of California increase fishery yields. Proc Natl Acad Sci U S A 105(30):10456–10459. https://doi.org/10.1073/pnas.0804601105

Aburto-Oropeza O, Erisman B, Galland GR, Mascareñas-Osorio I, Sala E, Ezcurra E (2011) Large recovery of fish biomass in a no-take marine reserve. PloS One 6(8):e23601. https://doi.org/10.1371/journal.pone.0023601

Aebischer NJ, Coulson JC, Colebrook JM (1990) Parallel long-term trends across four marine trophic levels and weather. Nature 347(October):753–755. https://doi.org/10.1038/347753a0

Agardy T, di Sciara GN, Christie P (2011) Mind the gap: addressing the shortcomings of marine protected areas through large scale marine spatial planning. Mar Policy 35:226–232. https://doi.org/10.1016/j.marpol.2010.10.006

Ainsworth TD, Heron SF, Ortiz JC, Mumby PJ, Grech A, Ogawa D, Eakin CM, Leggat W (2016) Climate change disables coral bleaching protection on the great barrier reef. Science 352(6283):338–342. https://doi.org/10.1126/science.aac7125

Albouy C, Archambault P, Appeltans W, Araujo MB, Beauchesne D, Cazelles K, Cirtwill AR et al (2019) The marine fish food web is globally connected. Nat Ecol Evol 3(August):1153–1161. https://doi.org/10.1038/s41559-019-0950-y

Alder J (2003) Putting the coast in the sea around us project. Sea Around Us Newsl 15:1–2. http://seaaroundus.org/newsletter/Issue15.pdf

Alongi DM (2015) The impact of climate change on mangrove forests. Curr Clim Change Rep 1(March):30–39. https://doi.org/10.1007/s40641-015-0002-x

Amoroso RO, Pitcher CR, Rijnsdorp AD, McConnaughey RA, Parma AM, Suuronen P, Eigaard OR et al (2018) Bottom trawl fishing footprints on the world's continental shelves. Proc Natl Acad Sci U S A 115(43):E10275–E10282. https://doi.org/10.1073/pnas.1802379115

Anderson RB (2019) Beyond 'Success': community, governance, and the future of Cabo Pulmo National Park. Hum Organ 78(2):147–157. https://doi.org/10.17730/0018-7259.78.2.147

Anderson ORJ, Small CJ, Croxall JP, Dunn EK, Sullivan BJ, Yates O, Black A (2011) Global seabird bycatch in longline fisheries. Endanger Species Res 14(2):91–106. https://doi.org/10.3354/esr00347

Andréfouët S (2008) Coral reef habitat mapping using remote sensing: a user vs producer perspective. implications for research, management and capacity building. J Spatial Sci 53(1):113–129. https://doi.org/10.1080/14498596.2008.9635140

Appeltans W, Ahyong ST, Anderson G, Angel MV, Artois T, Bailly N, Bamber R et al (2012) The magnitude of global marine species diversity. Curr Biol 22(23):2189–2202. https://doi.org/10.1016/j.cub.2012.09.036

Arias-Ortiz A, Serrano O, Masqué P, Lavery PS, Mueller U, Kendrick GA, Rozaimi M et al (2018) A marine heatwave drives massive losses from the world's largest seagrass carbon stocks. Nat Clim Change 8(April):338–344. https://doi.org/10.1038/s41558-018-0096-y

Asch RG, Erisman B (2018) Spawning aggregations act as a bottleneck influencing climate change impacts on a critically endangered reef fish. Divers Distrib 24(112):1712–1728. https://doi.org/10.1111/ddi.12809

Asch RG, Cheung W, Reygondeau G (2018) Future marine ecosystem drivers, biodiversity, and fisheries maximum catch potential in Pacific Island countries and territories under climate change. Mar Policy 88:285–294. https://doi.org/10.1016/j.marpol.2017.08.015

Ashford OS, Kenny AJ, Froján CRS, Bonsall MB, Horton T, Brandt A, Bird GJ, Gerken S, Rogers AD (2018) Phylogenetic and functional evidence suggests that deep-ocean ecosystems are highly sensitive to environmental change and direct human disturbance. Proc R Soc Lond B Biol Sci 285(1884):20180923. https://doi.org/10.1098/rspb.2018.0923

Avery JD, Aagaard K, Burkhalter JC, Robinson OJ (2017) Seabird longline bycatch reduction devices increase target catch while reducing bycatch: a meta-analysis. J Nat Conserv 38(July):37–45. https://doi.org/10.1016/j.jnc.2017.05.004

Badjeck M-C, Allison EH, Halls AS, Dulvy NK (2010) Impacts of climate variability and change on fishery-based livelihoods. Mar Policy 34(3):375–383. https://doi.org/10.1016/j.marpol.2009.08.007

Baker EK, Harris PT (2020) Habitat mapping and marine management. In: Harris PT, Baker EK (eds) Seafloor geomorphology as benthic habitat. Elsevier, Amsterdam, pp 17–33. https://doi.org/10.1016/B978-0-12-814960-7.00002-6

Bakun A (2006) Wasp-waist populations and marine ecosystem dynamics: navigating the 'predator pit' topographies. Prog Oceanogr 68(2–4):271–288. https://doi.org/10.1016/j.pocean.2006.02.004

Balmford A, Gravestock P, Hockley N, McClean CJ, Roberts CM (2004) The worldwide costs of marine protected areas. Proc Natl Acad Sci U S A 101(26):9694–9697. https://doi.org/10.1073/pnas.0403239101

Ban NC, Davies TE, Aguilera SE, Brooks C, Cox M, Epstein G, Evans LS, Maxwell SM, Nenadovic M (2017) Social and ecological effectiveness of large marine protected areas. Glob Environ Chang 43(March):82–91. https://doi.org/10.1016/j.gloenvcha.2017.01.003

Barange M, Merino G, Blanchard JL, Scholtens J, Harle J, Allison EH, Allen JI, Holt J, Jennings S (2014) Impacts of climate change on marine ecosystem production in societies dependent on fisheries. Nat Clim Change 4(March):211–216. https://doi.org/10.1038/nclimate2119

Barbier EB (2017) Marine ecosystem services. Curr Biol 27(11):R507–R510. https://doi.org/10.1016/j.cub.2017.03.020

Bates AE, Cooke RSC, Duncan MI, Edgar GJ, Brunoi JF, Benedetti-Cecchi L, Côté IM et al (2019) Climate resilience in marine protected areas and the 'Protection Paradox'. Biol Conserv 236(August):305–314. https://doi.org/10.1016/j.biocon.2019.05.005

Bax NJ, Appeltans W, Brainard R, Duffy JE, Dunstan P, Hanich Q, Harden Davies H et al (2018) Linking capacity development to GOOS monitoring networks to achieve sustained ocean observation. Front Mar Sci 5(September):346. https://doi.org/10.3389/fmars.2018.00346

Bayraktarov E, Saunders MI, Abdullah S, Mills M, Beher J, Possingham HP, Mumby PJ, Lovelock CE (2016) The cost and feasibility of marine coastal restoration. Ecol Appl 26(4):1055–1074. https://doi.org/10.1890/15-1077

Beaulieu SE, Szafranski K (2018) InterRidge global database of active submarine hydrothermal vent fields. Version 3.4. http://vents-data.interridge.org

Beck MW, Losada IJ, Menéndez P, Reguero BG, Díaz-Simal P, Fernández F (2018) The global flood protection savings provided by coral reefs. Nat Commun 9(June):2186. https://doi.org/10.1038/s41467-018-04568-z

Belhabib D, Sumaila UR, Lam VWY, Zeller D, Le Billon P, Abou Kane E, Pauly D (2015) Euros vs. Yuan: comparing European and Chinese fishing access in West Africa. PloS One 10(3):e0118351. https://doi.org/10.1371/journal.pone.0118351

Bell JB, Guijarro-Garcia E, Kenny A (2019) Demersal fishing in areas beyond national jurisdiction: a comparative analysis of regional fisheries management organisations. Front Mar Sci 6(October):596. https://doi.org/10.3389/fmars.2019.00596

Bennett S, Wernberg T, Connell SD, Hobday AJ, Johnson CR, Poloczanska ES (2016) The 'Great Southern Reef': social, ecological and economic value of Australia's neglected kelp forests. Mar Freshw Res 67(1):47–56. https://doi.org/10.1071/MF15232

Blanchard JL, Watson RA, Fulton EA, Cottrell RS, Nash KL, Bryndum-Buchholz A, Büchner M et al (2017) Linked sustainability challenges and trade- offs among fisheries, aquaculture and agriculture. Nat Ecol Evol 1(September):1240–1249. https://doi.org/10.1038/s41559-017-0258-8

Blasiak R, Wynberg R, Grorud-Colvert K, Thambisetty S, Bandarra N, Canário AVM, da Silva J et al (2020) The ocean genome: conservation and the fair, equitable and sustainable use of marine genetic resources. In: Blue Paper 4, High level panel for a sustainable ocean economy. World Resources Institute, Washington, DC

Block BA, Jonsen ID, Jorgensen S, Winship AJ, Shaffer SA, Bograd SJ, Hazen EL et al (2011) Tracking apex marine predator movements in a dynamic ocean. Nature 475(July):86–90. https://doi.org/10.1038/nature10082

Boerder K, Miller NA, Worm B (2018) Global hot spots of transhipment of fish catch at sea. Sci Adv 4(7):eaat7159. https://doi.org/10.1126/sciadv.aat7159

Bongiorno DL, Bryson M, Bridge TCL, Dansereau DG, Williams SB (2018) Coregistered hyperspectral and stereo image seafloor mapping from an autonomous underwater vehicle. J Field Robot 35(3):312–329. https://doi.org/10.1002/rob.21713

Bonney R, Shirk JL, Phillips TB, Wiggins A, Ballard HL, Miller-Rushing AJ, Parrish JK (2014) Next steps for citizen science. Science 343(6178):1436–1437. https://doi.org/10.1126/science.1251554

Boyes SJ, Elliott M (2014) Marine legislation: the ultimate 'Horrendogram': International Law, European Directives & National Implementation. Mar Pollut Bull 86(1–2):39–47. https://doi.org/10.1016/j.marpolbul.2014.06.055

Bozec YM, O'Farrell S, Bruggemann JH, Luckhurst BE, Mumby PJ (2016) Tradeoffs between fisheries harvest and the resilience of coral reefs. Proc Natl Acad Sci U S A 113(16):4536–4541. https://doi.org/10.1073/pnas.1601529113

Bradley D, Merrifield M, Miller KM, Lomonico S, Wilson JR, Gleason MG (2019) Opportunities to improve fisheries management through innovative technology and advanced data systems. Fish Fish 20(3):564–583. https://doi.org/10.1111/faf.12361

Breitburg D, Levin LA, Oschlies A, Grégoire M, Chavez FP, Conley DJ, Garçon V et al (2018) Declining oxygen in the global ocean and coastal waters. Science 359(6371):eaam7240. https://doi.org/10.1126/science.aam7240

Briggs JC (2007) Marine longitudinal biodiversity: causes and conservation. Divers Distrib 13(5):544–555. https://doi.org/10.1111/j.1472-4642.2007.00362.x

Britten GL, Dowd M, Kanary L, Worm B (2017) Extended fisheries recovery timelines in a changing environment. Nat Commun 8(May):15325. https://doi.org/10.1038/ncomms15325

Bruckner AW, Johnson KA, Field JD (2003) Conservation strategies for sea cucumbers: can a CITES Appendix II listing promote sustainable international trade? SPC Beche-de-mer Inform Bull 18(May):24–33

Burfeind DD, Pitt KA, Connolly RM, Byers JE (2013) Performance of non-native species within marine reserves. Biol Invasions 15:17–28. https://doi.org/10.1007/s10530-012-0265-2

Burgess MG, McDermott GR, Owashi B, Peavey Reeves LE, Clavelle T, Ovando D, Wallace BP et al (2018) Protecting marine mammals, turtles, and birds by rebuilding global fisheries. Science 359(6381):1255–1258. https://doi.org/10.1126/science.aao4248

Burrows MT, Schoeman DS, Buckley LB, Moore P, Poloczanska ES, Brander KM, Brown C et al (2011) The pace of shifting climate in marine and terrestrial ecosystems. Science 334(6056):652–655. https://doi.org/10.1126/science.1210288

Burrows MT, Schoeman DS, Richardson AJ, Molinos JG, Hoffmann A, Buckley LB, Moore PJ et al (2014) Geographical limits to species-range shifts are suggested by climate velocity. Nature 507(March):492–495. https://doi.org/10.1038/nature12976

Burrows MT, Bates AE, Costello MJ, Edwards M, Edgar GJ, Fox CJ, Halpern BS et al (2019) Ocean community warming responses explained by thermal affinities and temperature gradients. Nat Clim Change 9(December):959–963. https://doi.org/10.1038/s41558-019-0631-5

Canonico G, Buttigieg PL, Montes E, Muller-Karger FE, Stepien C, Wright D, Benson A et al (2019) Global observational needs and resources for marine biodiversity. Front Mar Sci 6(July):367. https://doi.org/10.3389/fmars.2019.00367

Carpenter KE, Abrar M, Aeby G, Aronson RB, Banks S, Bruckner A, Chiriboga A et al (2008) One-third of reef-building corals face elevated extinction risk from climate change and local impacts. Science 321(5888):560–563. https://doi.org/10.1126/science.1159196

CBD (Convention on Biological Diversity) (2009) Azores scientific criteria and guidance for identifying ecologically or biologically significant areas and designing representative networks of marine protected areas in open ocean waters and deep sea habitats. CBD Secretariat, Quebec. https://www.cbd.int/marine/doc/azores-brochure-en.pdf

Chaudhary C, Saeedi H, Costello MJ (2016) Bimodality of latitudinal gradients in marine species richness. Trends Ecol Evol 31(9):670–676. https://doi.org/10.1016/j.tree.2016.06.001

Chavez FP, Ryan J, Lluch-Cota SE, Niquen M (2003) From anchovies to sardines and back: multidecadal decadal change in the Pacific. Science 299(5604):217–221. https://doi.org/10.1126/science.1075880

Cheng BS, Altieri AH, Torchin ME, Ruiz GM (2019) Can marine reserves restore lost ecosystem functioning? a global synthesis. Ecology 100(4):e02617. https://doi.org/10.1002/ecy.2617

Cheung WWL, Pitcher TJ, Pauly D (2005) A fuzzy logic expert system to estimate intrinsic extinction vulnerabilities of marine fishes to fishing. Biol Conserv 124(1):97–111. https://doi.org/10.1016/j.biocon.2005.01.017

Cheung WWL, Lam VW, Sarmiento JI, Kearney K, Watson R, Pauly D (2009) Projecting global marine biodiversity impacts under climate change scenarios. Fish Fish 10(3):235–251. https://doi.org/10.1111/j.1467-2979.2008.00315.x

Cheung WWL, Lam VW, Sarmiento JI, Kearney K, Watson R, Zeller D, Pauly D (2010) Large-scale redistribution of maximum fisheries catch potential in the global ocean under climate change. Glob Chang Biol 16(1):24–35. https://doi.org/10.1111/j.1365-2486.2009.01995.x

Cheung WWL, Watson R, Pauly D (2013) Signature of ocean warming in global fisheries catch. Nature 497(May):365–369. https://doi.org/10.1038/nature12156

Christensen V, Coll M, Piroddi C, Steenbeek J, Buszowski J, Pauly D (2014) A century of fish biomass decline in the ocean. Mar Ecol Prog Ser 512(October):155–166. https://doi.org/10.3354/meps10946

Cinner J, McClanahan T, Daw TM, Graham NAJ, Maina J, Wilson SK, Hughes TP (2009) Linking social and ecological systems to sustain coral reef fisheries. Curr Biol 19(3):206–212. https://doi.org/10.1016/j.cub.2008.11.055

Cisneros-Mata MA, Montemayor-López G, Nevárez-Martínez MO (1996) Modeling deterministic effects of age structure, density dependence, environmental forcing, and fishing in the population dynamics of *Sardinops sagax* caeruleus in the Gulf of California. Calif Cooper Ocean Fish Investig Rep 37(October):201–208. https://www.calcofi.org/publications/calcofireports/v37/CalCOFI_Rpt_Vol_37_1996.pdf

Claar DC, Szostek L, McDevitt-Irwin JM, Schanze JJ, Baum JK (2018) Global patterns and impacts of El Niño events on coral reefs: a meta-analysis. PloS One 13(2):e0190957. https://doi.org/10.1371/journal.pone.0190957

Clark NA, Ardron JA, Pendleton LH (2015) Evaluating the basic elements of transparency of regional fisheries management organizations. Mar Policy 57(July):158–166. https://doi.org/10.1016/j.marpol.2015.03.003

Clark MR, Althaus F, Schlacher TA, Williams A, Bowden DA, Rowden AA (2016) The impacts of deep-sea fisheries on benthic communities: a review. ICES J Mar Sci 73(1):51–69. https://doi.org/10.1093/icesjms/fsv123

Clarke A, Crame JA (1997) Diversity, latitude and time: patterns in the Shallow sea. In: Ormond RFG, Gage JD, Angel MV (eds) Marine biodiversity: patterns and processes. Cambridge University Press, Cambridge, pp 122–147

Collins MA, Villanueva R (2006) Taxonomy, ecology and behaviour of the cirrate octopods. In: Gibson RN, Atkinson RJA, Gordon JDM (eds) Oceanography and marine biology: an annual review, vol 44. CRC, Boca Raton, FL, pp 277–222. https://doi.org/10.1201/9781420006391

Colyvan M, Justus J, Regan HM (2010) The natural environment is valuable but not infinitely valuable. Conserv Lett 3(4):224–228. https://doi.org/10.1111/j.1755-263X.2010.00118.x

Constable AJ (2011) Lessons from CCAMLR on the implementation of the ecosystem approach to managing fisheries. Fish Fish 12(2):138–151. https://doi.org/10.1111/j.1467-2979.2011.00410.x

Costanza R, D'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K et al (1997) The value of the world's ecosystem services and natural capital. Nature 387(May):253–260. https://doi.org/10.1038/387253a0

Costanza R, de Groot R, Sutton P, van der Ploeg S, Anderson SJ, Kubiszewski I, Farber S, Turner RK (2014) Changes in the global value of ecosystem services. Glob Environ Chang 26(May):152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002

Costello MJ, Coll M, Danovaro R, Halpin P, Ojaveer H, Miloslavich P (2010) A census of marine biodiversity knowledge, resources, and future challenges. PloS One 5(8):e12110. https://doi.org/10.1371/journal.pone.0012110

Costello C, Ovando D, Hilborn R, Gaines SD, Deschenes O, Lester SE (2012) Status and solutions for the world's unassessed fisheries. Science 338(6106):517–520. https://doi.org/10.1126/science.1223389

Costello C, Ovando D, Clavelle T, Strauss CK, Hilborn R, Melnychuk MC, Branch TA et al (2016) Global fishery prospects under contrasting management regimes. Proc Natl Acad Sci U S A 113(18):5125–5129. https://doi.org/10.1073/pnas.1520420113

Costello C, Cao L, Gelcich S, Cisneros MA, Free CM, Froehlich HE, Galarza E et al (2019) The future of food from the sea. World Resources Institute, Washington, DC. www.oceanpanel.org/future-food-sea

Courchamp F, Angulo E, Rivalan P, Hall RJ, Signoret L, Bull L, Meinard Y (2006) Rarity value and species extinction: the anthropogenic allee effect. PLoS Biol 4(12):e415. https://doi.org/10.1371/journal.pbio.0040415

Crame JA, Clarke A (1997) The historical component of marine taxonomic diversity gradients. In: Ormond RFG, Gage JD, Angel MV (eds) Marine biodiversity: patterns and processes. Cambridge University Press, Cambridge, pp 258–273

Crawford B, Kasmidi M, Korompis F, Pollnac RB (2006) Factors influencing progress in establishing community-based marine protected areas in Indonesia. Coast Manag 34(1):39–64. https://doi.org/10.1080/08920750500379300

Cruz-Motta JJ, Miloslavich P, Palomo G, Iken K, Konar B, Pohle G, Trott T et al (2010) Patterns of spatial variation of assemblages associated with intertidal rocky shores: a global perspective. PloS One 5(12):e14354. https://doi.org/10.1371/journal.pone.0014354

Cubaynes HC, Fretwell PT, Bamford C, Gerrish L, Jackson JA (2019) Whales from space: four mysticete species described using new VHR satellite imagery. Mar Mamm Sci 35(2):466–491. https://doi.org/10.1111/mms.12544

Cullis-Suzuki S, Pauly D (2010) Failing the high seas: a global evaluation of regional fisheries management organizations. Mar Policy 34(5):1036–1042. https://doi.org/10.1016/j.marpol.2010.03.002

Cury P, Bakun A, Crawford RJM, Jarre-Teichmann A, Quiñones RA, Shannon LJ, Verheye HM (2000) Small pelagics in upwelling systems: patterns of interaction and structural changes in 'wasp-waist' ecosystems. ICES J Mar Sci 57(3):603–618. https://doi.org/10.1006/jmsc.2000.0712

Cury PM, Boyd IL, Bonhommeau S, Anker-Nilssen T, Crawford RJM, Furness RW, Mills JA et al (2011) Global seabird responses to forage fish depletion—one-third for the birds. Science 334(6063):1703–1706. https://doi.org/10.1126/science.1212928

Danovaro R, Gambi C, D'll'anno A, Corinaldesi C, Fraschetti S, Vanreusel A, Vincx M, Gooday AJ (2008) Exponential decline of deep-sea ecosystem functioning linked to benthic biodiversity loss. Curr Biol 18(1):1–8. https://doi.org/10.1016/j.cub.2007.11.056

Daskalov GM, Grishin AN, Rodionov S, Mihneva V (2007) Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. Proc Natl Acad Sci U S A 104(25):10518–10523. https://doi.org/10.1073/pnas.0701100104

Davies TD, Baum JK (2012) Extinction risk and overfishing: reconciling conservation and fisheries perspectives on the status of marine fishes. Sci Rep 2(August):561. https://doi.org/10.1038/srep00561

Davies RG, Irlich UM, Chown SL, Gaston KJ (2010) Ambient, productive and wind energy, and ocean extent predict global species richness of procellariiform seabirds. Glob Ecol Biogeogr 19(1):98–110. https://doi.org/10.1111/j.1466-8238.2009.00498.x

de Bruyn P, Murua H, Aranda M (2013) The precautionary approach to fisheries management: how this is taken into account by Tuna Regional Fisheries Management Organisations (RFMOs). Mar Policy 38(March):397–406. https://doi.org/10.1016/j.marpol.2012.06.019

de Groot R, Brander L, van der Ploeg S, Costanza R, Bernard F, Braat L, Christie M et al (2012) Global estimates of the value of ecosystems and their services in monetary units. Ecosyst Serv 1(1):50–61. https://doi.org/10.1016/j.ecoser.2012.07.005

de los Santos CB, Krause-Jensen D, Alcoverro T, Marbà N, Duarte CM, van Katwijk MM, Pérez M et al (2019) Recent trend reversal for declining european seagrass meadows. Nat Commun 10:3356. https://doi.org/10.1038/s41467-019-11340-4

de Vargas C, Audic S, Henry N, Decelle J, Mahé F, Logares R, Lara E et al (2015) Eukaryotic plankton diversity in the sunlit ocean. Science 348(6237):1261605. https://doi.org/10.1126/science.1261605

Di Franco A, Thiriet P, Di Carlo G, Dimitriadis C, Francour P, Gutiérrez NL, de Grissac AJ et al (2016) Five key attributes can increase marine protected areas performance for small-scale fisheries management. Sci Rep 6(December):38135. https://doi.org/10.1038/srep38135

Di Lorenzo M, Claudet J, Guidetti P (2016) Spillover from marine protected areas to adjacent fisheries has an ecological and a fishery component. J Nat Conserv 32(July):62–66. https://doi.org/10.1016/j.jnc.2016.04.004

Dias MP, Martin R, Pearmain EJ, Burfield IJ, Small C, Phillips RA, Yates O et al (2019) Threats to seabirds: a global assessment. Biol Conserv 237(September):525–537. https://doi.org/10.1016/j.biocon.2019.06.033

Diaz-Pulido G, McCook LJ, Dove S, Berkelmans R, Roff G, Kline DI, Weeks S et al (2009) Doom and boom on a resilient reef: climate change, algal overgrowth and coral recovery. PloS One 4(4):e5239. https://doi.org/10.1371/journal.pone.0005239

Diz D, Johnson D, Riddell M, Rees S, Battle J, Gjerde K, Hennige S, Roberts JM (2018) Mainstreaming marine biodiversity into the SDGs: the role of other effective area-based conservation measures (SDG 14.5). Mar Policy 93(July):251–261. https://doi.org/10.1016/j.marpol.2017.08.019

Dornelas M, Gotelli NJ, McGill B, Shimadzu H, Moyes F, Sievers C, Magurran AE (2014) Assemblage time series reveal biodiversity change but not systematic loss. Science 344(6181):296–299. https://doi.org/10.1126/science.1248484

Duarte CM, Losada IJ, Hendriks IE, Mazarrasa I, Marbà N (2013a) The role of coastal plant communities for climate change mitigation and adaptation. Nat Clim Change 3(November):961–968. https://doi.org/10.1038/nclimate1970

Duarte CM, Sintes T, Marbà N (2013b) Assessing the CO2 capture potential of seagrass restoration projects. J Appl Ecol 50(6):1341–1349. https://doi.org/10.1111/1365-2664.12155

Duarte CM, Wu J, Xiao X, Bruhn A, Krause-Jensen D (2017) Can seaweed farming play a role in climate change mitigation and adaptation? Front Mar Sci 4(April):100. https://doi.org/10.3389/fmars.2017.00100

Duarte CM, Agusti S, Barbier E, Britten GL, Castilla JC, Gattuso J-P, Fulweiler RW, Hughes TP et al (2020) Rebuilding marine life. Nature 580:39–51. https://doi.org/10.1038/s41586-020-2146-7

Duffy JE, Lefcheck JS, Stuart-Smith RD, Navarrete SA, Edgar GJ (2016) Biodiversity enhances reef fish biomass and resistance to climate change. Proc Natl Acad Sci U S A 113(22):6230–6235. https://doi.org/10.1073/pnas.1524465113

Dulvy NK, Sadovy Y, Reynolds JD (2003) Extinction vulnerability in marine populations. Fish Fish 4(1):25–64. https://doi.org/10.1046/j.1467-2979.2003.00105.x

Dulvy NK, Fowler SL, Musick JA, Cavanagh RD, Kyne PM, Harrison LR, Carlson JK et al (2014) Extinction risk and conservation of the world's sharks and rays. Elife 3:e00590. https://doi.org/10.7554/eLife.00590

Durden JM, Lallier LE, Murphy K, Jaeckel A, Gjerde K, Jones DOB (2018) Environmental impact assessment process for deep-sea mining in 'the area'. Mar Policy 87(January):194–202. https://doi.org/10.1016/j.marpol.2017.10.013

Dureuil M, Boerder K, Burnett KA, Froese R, Worm B (2018) Elevated trawling inside protected areas undermines conservation outcomes in a global fishing hot spot. Science 362(6421):1403–1407. https://doi.org/10.1126/science.aau0561

Edgar GJ, Langhammer PF, Allen G, Brooks TM, Brodie J, Crosse W, De Silva N et al (2008) Key biodiversity areas as globally significant target sites for the conservation of marine biological diversity. Aquat Conserv Mar Freshw Ecosyst 18(6):969–983. https://doi.org/10.1002/aqc.902

Edgar GJ, Stuart-Smith RD, Willis TJ, Kininmonth S, Baker SC, Banks S, Barrett NS et al (2014) Global conservation outcomes depend on marine protected areas with five key features. Nature 506(February):216–220. https://doi.org/10.1038/nature13022

Edgar GJ, Bates AE, Bird TJ, Jones AH, Kininmonth S, Stuart-Smith RD, Webb TJ (2016) New approaches to marine conservation through scaling up of ecological data. Ann Rev Mar Sci 8(January):435–461. https://doi.org/10.1146/annurev-marine-122414-033921

Edgar GJ, Alexander TJ, Lefcheck JS, Bates AE, Kininmonth SJ, Thomson RJ, Duffy JE, Costello MJ, Stuart-Smith RD (2017) Abundance and local-scale processes contribute to multi-phyla gradients in global marine diversity. Sci Adv 3(10):e1700419. https://doi.org/10.1126/sciadv.1700419

Edgar GJ, Ward TJ, Stuart-Smith RD (2019) Weaknesses in stock assessment modelling and management practices affect fisheries sustainability. Aquat Conserv Mar Freshw Ecosyst 29:2010–2016. https://doi.org/10.1002/aqc.3161

Eigaard OR, Bastardie F, Hintzen NT, Buhl-Mortensen L, Buhl-Mortensen P, Catarino R, Dinesen GE et al (2017) The footprint of bottom trawling in european waters: distribution, intensity, and seabed integrity. ICES J Mar Sci 74(3):847–865. https://doi.org/10.1093/icesjms/fsw194

EIU (Economist Intelligence Unit) (2015) The blue economy: growth, opportunity and a sustainable ocean economy. EIU, New York

Elvidge CD, Ghosh T, Baugh K, Zhizhin M, Hsu FC, Katada NS, Penalosa W, Hung BQ (2018) Rating the effectiveness of fishery closures with visible infrared imaging radiometer suite boat detection data. Front Mar Sci 5(April):132. https://doi.org/10.3389/fmars.2018.00132

Ens EJ, Pert P, Clarke PA, Budden M, Clubb L, Doran B, Douras C et al (2015) Indigenous biocultural knowledge in ecosystem science and management: review and insight from Australia. Biol Conserv 181(January):133–149. https://doi.org/10.1016/j.biocon.2014.11.008

Erisman B, Heyman W, Kobara S, Ezer T, Pittman S, Aburto-Oropeza O, Nemeth RS (2015) Fish spawning aggregations: where well-placed management actions yield big benefits for fisheries and conservation. Fish Fish 18(1):121–144. https://doi.org/10.1111/faf.12132

Estes JA, Terborgh J, Brashares JS, Power ME, Berger J, Bond WJ, Carpenter SR et al (2011) Trophic downgrading of planet earth. Science 333(6040):301–306. https://doi.org/10.1126/science.1205106

FAO (2011) FAO technical guidelines for responsible fisheries: fisheries management—4. Marine protected areas and fisheries. No. 4, Suppl. 4. FAO, Rome

FAO (2018) The State of world fisheries and aquaculture 2018—meeting the sustainable development goals. FAO, Rome. http://www.fao.org/3/i9540en/i9540en.pdf

FAO (Food and Agriculture Organization of the United Nations) (2008) The FAO International guidelines for the management of deep-sea fisheries in the high seas. Fisheries and Aquaculture Department, FAO, Rome

Fautin D, Dalton P, Incze LS, Leong JAC, Pautzke C, Rosenberg A, Sandifer P et al (2010) An overview of marine biodiversity in United States waters. PloS One 5(8):e11914. https://doi.org/10.1371/journal.pone.0011914

Fernandes PG, Cook RM (2013) Reversal of fish stock decline in the Northeast Atlantic. Curr Biol 23(15):1432–1437. https://doi.org/10.1016/j.cub.2013.06.016

Fernandes PG, Ralph GM, Nieto A, Criado MG, Vasilakopoulos P, Maravelias CD, Cook RM et al (2017) Coherent assessments of Europe's marine fishes show regional divergence and megafauna loss. Nat Ecol Evol 1(May):0170. https://doi.org/10.1038/s41559-017-0170

Fischer AG (1960) Latitudinal variations in organic diversity. Evolution 14(1):64–81. https://doi.org/10.1111/j.1558-5646.1960.tb03057.x

Fisher R, O'Leary RA, Low-Choy S, Mengersen K, Knowlton N, Brainard RE, Caley MJ (2015) Species richness on coral reefs and the pursuit of convergent global estimates. Curr Biol 25(4):500–505. https://doi.org/10.1016/j.cub.2014.12.022

Foglini F, Grande V, Marchese F, Bracchi VA, Prampolini M, Angeletti L, Castellan G et al (2019) Application of hyperspectral imaging to underwater habitat mapping, Southern Adriatic Sea. Sensors 19(10):2261. https://doi.org/10.3390/s19102261

Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CS (2004) Regime shifts, resilience and biodiversity in ecosystem management. Annu Rev Ecol Evol Syst 35(December):557–581. https://doi.org/10.1146/annurev.ecolsys.35.021103.105711

Ford AK, Bejarano S, Nugues MM, Visser PM, Albert S, Ferse SCA (2018a) Reefs under siege—the rise, putative drivers, and consequences of benthic cyanobacterial mats. Front Mar Sci 5(February):18. https://doi.org/10.3389/fmars.2018.00018

Ford JH, Bergseth B, Wilcox C (2018b) Chasing the fish oil—do bunker vessels hold the key to fisheries crime networks? Front Mar Sci 5(August):267. https://doi.org/10.3389/fmars.2018.00267

Ford JH, Peel D, Hardesty BD, Rosebrock U, Wilcox C (2018c) Loitering with intent—catching the outlier vessels at sea. PloS One 13(7):e0200189. https://doi.org/10.1371/journal.pone.0200189

Fossheim M, Primicerio R, Johannesen E, Ingvaldsen RB, Aschan MM, Dolgov AV (2015) Recent warming leads to a rapid borealization of fish communities in the Arctic. Nat Clim Change 5(July):673–677. https://doi.org/10.1038/nclimate2647

Frank KT, Petrie B, Choi JS, Leggett WC (2005) Trophic cascades in a formerly cod-dominated ecosystem. Science 308(5728):1621–1623. https://doi.org/10.1126/science.1113075

Freduah G, Fidelman P, Smith TF (2017) The impacts of environmental and socio-economic stressors on small scale fisheries and livelihoods of fishers in Ghana. Appl Geogr 89(December):1–11. https://doi.org/10.1016/j.apgeog.2017.09.009

Freiwald A, Rogers A, Hall-Spencer J, Guinotte JM, Davies AJ, Yesson C, Martin CS, Weatherdon LV (2017) Global distribution of cold-water corals. Version 5.0. Fifth update to the data set in Freiwald et al. (2004). http://data.unep-wcmc.org/datasets/3

Friedman K, Garcia SM, Rice J (2018) Mainstreaming biodiversity in fisheries. Mar Policy 95(September):209–220. https://doi.org/10.1016/j.marpol.2018.03.001

Friess DA, Rogers K, Lovelock CE, Krauss KW, Hamilton SE, Lee SY, Lucas R, Primavera J et al (2019) The State of the World's mangrove forests: past, present, and future. Annu Rev Env Resour 44:89–115. https://doi.org/10.1146/annurev-environ-101718-033302

Gagné TO, Hyrenbach KD, Hagemann ME, Van Houtan KS (2018) Trophic signatures of seabirds suggest shifts in oceanic ecosystems. Sci Adv 4(2):eaao3946. https://doi.org/10.1126/sciadv.aao3946

Gagné TO, Reygondeau G, Jenkins CN, Sexton JO, Bograd SJ, Hazen EL et al (2020) Towards a global understanding of the drivers of marine and terrestrial biodiversity. PloS One 15(2):e0228065. https://doi.org/10.1371/journal.pone.0228065

Gaines SD, Lubchenco J (1982) A unified approach to marine plant-herbivore interactions II. Biogeographic patterns. Annu Rev Ecol Syst 13:111–138. https://www.jstor.org/stable/2097064

Gaines SD, White C, Carr MH, Palumbi SR (2010) Designing marine reserve networks for both conservation and fisheries management. Proc Natl Acad Sci U S A 107:18286–18293. https://doi.org/10.1073/pnas.0906473107

Gamfeldt L, Lefcheck JS, Byrnes JEK, Cardinale BJ, Duffy JE, Griffin JN (2014) Marine biodiversity and ecosystem functioning: what's known and what's next? Oikos 124(3):252–265. https://doi.org/10.1111/oik.01549

Gangloff MM, Edgar GJ, Wilson B (2016) Imperilled species in aquatic ecosystems: emerging threats, management and future prognoses. Aquat Conserv Mar Freshw Ecosyst 26(5):858–871. https://doi.org/10.1002/aqc.2707

Garcia SM (2010) Governance, science, and society: the ecosystem approach to fisheries. In: Grafton RQ, Hilborn R, Squires D, Tait M, Williams M (eds) Handbook of marine fisheries conservation and management. Oxford University Press, Oxford, pp 87–98

Gattuso JP, Magnan A, Billé R, Cheung W, Howes EL, Joos F, Allemand D et al (2015) Contrasting futures for ocean and society from different anthropogenic CO2 emissions scenarios. Science 349(6243):aac4722. https://doi.org/10.1126/science.aac4722

Gedan KB, Silliman BR, Bertness MD (2009) Centuries of human-driven change in salt marsh ecosystems. Ann Rev Mar Sci 1:117–141. https://doi.org/10.1146/annurev.marine.010908.163930

Gell FR, Roberts CM (2003) Benefits beyond boundaries: the fishery effects of marine reserves. Trends Ecol Evol 18(9):448–455. https://doi.org/10.1016/S0169-5347(03)00189-7

Gill DA, Mascia MB, Ahmadia GN, Glew L, Lester SE, Barnes M, Craigie I et al (2017) Capacity shortfalls hinder the performance of marine protected areas globally. Nature 543(March):665–669. https://doi.org/10.1038/nature21708

Gilman E, Kingma E (2013) Standard for assessing transparency in information on compliance with obligations of regional fisheries management organizations: validation through assessment of the Western and Central Pacific Fisheries Commission. Ocean Coast Manag 84(November):31–39. https://doi.org/10.1016/j.ocecoaman.2013.07.006

Gilman E, Passfield K, Nakamura K (2014) Performance of regional fisheries management organizations: ecosystem-based governance of bycatch and discards. Fish Fish 15(2):327–351. https://doi.org/10.1111/faf.12021

Gilman E, Weijerman M, Suuronen P (2017) Ecological data from observer programmes underpin ecosystem-based fisheries management. ICES J Mar Sci 74(6):1481–1495. https://doi.org/10.1093/icesjms/fsx032

Giri C, Ochieng E, Tieszen LL, Zhu Z, Singh A, Loveland T, Masek J, Duke N (2011) Status and distribution of mangrove forests of the world using earth observation satellite data: status and distributions of global mangroves. Glob Ecol Biogeogr 20(1):154–159. https://doi.org/10.1111/j.1466-8238.2010.00584.x

Gjerde KM, Currie D, Wowk K, Sack K (2013) Ocean in Peril: reforming the management of global ocean living resources in areas beyond national jurisdiction. Mar Pollut Bull 74(2):540–551. https://doi.org/10.1016/j.marpolbul.2013.07.037

Gorelick N, Hancher M, Dixon M, Ilyushchenko S, Thau D, Moore R (2017) Google Earth Engine: planetary-scale geospatial analysis for everyone. Remote Sens Environ 202(1):18–27. https://doi.org/10.1016/j.rse.2017.06.031

Grady JM, Maitner BS, Winter AS, Kaschner K, Tittensor DP, Record S, Smith FA et al (2019) Metabolic asymmetry and the global diversity of marine predators. Science 363(6425):eaat4220. https://doi.org/10.1126/science.aat4220

Graham NAJ, Jennings S, MacNeil MA, Mouillot D, Wilson SK (2015) Predicting climate-driven regime shifts versus rebound potential in coral reefs. Nature 518(February):94–97. https://doi.org/10.1038/nature14140

Green EP, Mumby PJ, Edwards AJ, Clark CD (2000) Remote Sensing Handbook Tropical Coastal Management. United Nations Educational, Scientific and Cultural Organization, Paris

Grémillet D, Ponchon A, Paleczny M, Palomares MLD, Karpouzi V, Pauly D (2018) Persisting worldwide seabird-fishery competition despite seabird community decline. Curr Biol 28(24):4009–4013. https://doi.org/10.1016/j.cub.2018.10.051

Guannel G, Arkema K, Ruggiero P, Verutes G (2016) The power of three: coral reefs, seagrasses and mangroves protect coastal regions and increase their resilience. PloS One 11(7):e0158094. https://doi.org/10.1371/journal.pone.0158094

Guillemette M, Grégoire F, Bouillet D, Rail JF, Bolduc F, Caron A, Pelletier D (2018) Breeding failure of seabirds in relation to fish depletion: is there one universal threshold of food abundance? Mar Ecol Prog Ser 587(January):235–245. https://www.int-res.com/articles/meps_oa/m587p235.pdf

Gurney GG, Cinner J, Ban N, Pressey RL, Campbell SJ, Tasidjawa S, Setiwan F (2014) Poverty and protected areas: an evaluation of a marine integrated conservation and development project in Indonesia. Glob Environ Chang 26(May):98–107. https://doi.org/10.1016/j.gloenvcha.2014.04.003

Haas AF, Fairoz MFM, Kelly LW, Nelson CE, Dinsdale EA, Edwards RA, Giles S et al (2016) Global microbialization of coral reefs. Nat Microbiol 1(April):16042. https://doi.org/10.1038/nmicrobiol.2016.42

Haas B, Haward M, McGee J, Fleming A (2019) The influence of performance reviews on regional fisheries management organizations. ICES J Mar Sci 76(7):2082–2089. https://doi.org/10.1093/icesjms/fsz088

Halpern BS (2003) The impact of marine reserves: do reserves work and does reserve size matter? Ecol Appl 13(Sp1):S117–S137. https://doi.org/10.1890/1051-0761(2003)013[0117:TIOMRD]2.0.CO;2

Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C, Bruno JF et al (2008) A global map of human impact on marine ecosystems. Science 319(5865):948–952. https://doi.org/10.1126/science.1149345

Halpern BS, Lester SE, Kellner JB (2010) Spillover from marine reserves and the replenishment of fished stocks. Environ Conserv 36(4):268–276. https://doi.org/10.1017/S0376892910000032

Halpern BS, Frazier M, Potapenko J, Casey KS, Koenig K, Longo C, Lowndes JS et al (2015) Spatial and temporal changes in cumulative human impacts on the world's ocean. Nat Commun 6(July):7615. https://doi.org/10.1038/ncomms8615

Halpern BS, Frazier M, Afflerbach J, Lowndes JS, Micheli F, O'Hara C, Scarborough C, Selkoe KA (2019) Recent pace of change in human impact on the world's ocean. Sci Rep 9(August):11609. https://doi.org/10.1038/s41598-019-47201-9

Hansen RE (2011) Introduction to synthetic aperture sonar. In: Kolev NZ (ed) Sonar systems. InTech, Rijeka, Croatia, pp 3–28. https://doi.org/10.5772/23122

Hare SR, Mantua NJ (2000) Empirical evidence for North Pacific Regime Shifts in 1977 and 1989. Prog Oceanogr 47(2–4):103–145. https://doi.org/10.1016/S0079-6611(00)00033-1

Harris PT (2020) Anthropogenic threats to benthic habitats. In: Harris PT, Baker EK (eds) Seafloor geomorphology as benthic habitat: Geohab atlas of seafloor geomorphic features and benthic habitats. Elsevier, London, pp 35–60. https://doi.org/10.1016/B978-0-12-385140-6.00003-7

Harris PT, Baker EK (2012) Why Map Benthic Habitats? In: Harris PT, Baker EK (eds) Seafloor geomorphology as benthic habitat: Geohab atlas of seafloor geomorphic features and benthic habitats. Elsevier, London, pp 3–22. https://doi.org/10.1016/B978-0-12-385140-6.00001-3

Harris PT, Macmillan-Lawler M, Rupp J, Baker EK (2014) Geomorphology of the oceans. Mar Geol 352(1):4–24. https://doi.org/10.1016/j.margeo.2014.01.011

Harrison PA, Berry PM, Simpson G, Haslett JR, Blicharska M, Bucur M, Dunford R et al (2014) Linkages between biodiversity attributes and ecosystem services: a systematic review. Ecosyst Serv 9(September):191–203. https://doi.org/10.1016/j.ecoser.2014.05.006

Hector A, Bagchi R (2007) Biodiversity and ecosystem multifunctionality. Nature 448(7150):188–190. https://doi.org/10.1038/nature05947

Heery EC, Bishop MJ, Critchley LP, Bugnot AB, Airoldi L, Mayer-Pinto M, Sheehan EV et al (2017) Identifying the consequences of ocean sprawl for sedimentary habitats. J Exp Mar Biol Ecol 492(July):31–48. https://doi.org/10.1016/j.jejmbe.2017.01.020

Heino M, Díaz Pauli B, Dieckmann U (2015) Fisheries-induced evolution. Annu Rev Ecol Evol Syst 46(December):461–480. https://doi.org/10.1146/annurev-ecolsys-112414-054339

Hilborn R (2018) Are MPAs effective? ICES J Mar Sci 75(3):1160–1162. https://doi.org/10.1093/icesjms/fsx068

Hilborn R, Ovando D (2014) Reflections on the success of traditional fisheries management. ICES J Mar Sci 71(5):1040–1046. https://doi.org/10.1093/icesjms/fsu034

Hilborn R, Amoroso RO, Anderson CM, Baum JK, Branch TA, Costello C, de Moor CL et al (2020) Effective fisheries management instrumental in improving fish stock status. Proc Natl Acad Sci U S A 117(4):2218–2224. https://doi.org/10.1073/pnas.1909726116

Hobday AJ, Oliver ECJ, Sen Gupta A, Benthuysen JA, Burrows MT, Donat MG, Holbrook NJ et al (2018) Categorizing and naming marine heatwaves. Oceanography 31(2):162–173. https://doi.org/10.5670/oceanog.2018.205

Hochard JP, Hamilton S, Barbier EB (2019) Mangroves shelter coastal economic activity from cyclones. Proc Natl Acad Sci U S A 116(25):12232–12237. https://doi.org/10.1073/pnas.1820067116

Hoegh-Guldberg O, Mumby PJ, Hooten AJ, Steneck RS, Greenfield P et al (2007) Coral reefs under rapid climate change and ocean acidification. Science 318(5857):1737–1742. https://doi.org/10.1126/science.1152509

Hoegh-Guldberg O, Caldeira K, Chopin T, Gaines S, Haugan P, Hemer M, Howard J et al (2019) The ocean as a solution to climate change: five opportunities for action. World Resources Institute, Washington, DC. http://www.oceanpanel.org/climate

Hooff RC, Peterson WT (2006) Copepod biodiversity as an indicator of changes in ocean and climate conditions of the Northern California current ecosystem. Limnol Oceanogr 51(6):2607–2620. https://doi.org/10.4319/lo.2006.51.6.2607

Hooper T, Börger T, Langmead O, Marcone O, Rees SE, Rendon O, Beaumont N, Attrill MJ, Austen M (2019) Applying the natural capital approach to decision making for the marine environment. Ecosyst Serv 38(August):100947. https://doi.org/10.1016/j.ecoser.2019.100947

Hsieh CH, Reiss CS, Hunter JR, Beddington JR, May RM, Sugihara G (2006) Fishing elevates variability in the abundance of exploited species. Nature 443(October):859–862. https://doi.org/10.1038/nature05232

Hughes RG (2004) Climate change and loss of saltmarshes: consequences for birds. Ibis 146(s1):21–28. https://doi.org/10.1111/j.1474-919X.2004.00324.x

Hughes TP, Rodrigues MJ, Bellwood DR, Ceccarelli D, Hoegh-Guldberg O, McCook L, Moltschaniwsky N et al (2007) Phase shifts, herbivory, and the resilience of coral reefs to climate change. Curr Biol 17(4):360–365. https://doi.org/10.1016/j.cub.2006.12.049

Hughes TP, Barnes ML, Bellwood DR, Cinner JE, Cumming GS, Jackson JBC, Kleypas J et al (2017) Coral reefs in the anthropocene. Nature 546(June):82–90. https://doi.org/10.1038/nature22901

Hughes TP, Anderson KD, Connolly SR, Heron SF, Kerry JT, Lough JM, Baird AH et al (2018a) Spatial and temporal patterns of mass bleaching of corals in the anthropocene. Science 359(6371):80–83. https://doi.org/10.1126/science.aan8048

Hughes TP, Kerry JT, Baird AH, Connolly SR, Dietzel A, Eakin CM, Heron SF et al (2018b) Global warming transforms coral reef assemblages. Nature 556(April):492–496. https://doi.org/10.1038/s41586-018-0041-2

Humphries GRW, Velarde E, Anderson DW, Haasey B, Sydeman WJ (2015) Seabirds as early warning indicators of climate events in the North Pacific. PICES Press 23(1):18–20. http://aquaticcommons.org/21569/1/PPJanuary2015.pdf

Iken K, Konar B (2003) Natural Geography in Nearshore Areas (NaGISA): the nearshore component of the census of marine life. Gayana 67(2):153–160. https://doi.org/10.4067/S0717-65382003000200004

IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services) (2019) In: Díaz S, Brondízio JSES, Ngo HT, Guèze M, Agard J, Arneth A et al (eds) Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES Secretariat, Bonn. https://ipbes.net/sites/default/files/2020-02/ipbes_global_assessment_report_summary_for_policymakers_en.pdf

IPCC (2019) In: Pörtner HO, Roberts DC, Masson-Delmotte V, Zhai P, Tignor M, Poloczanska E, Mintenbeck K et al (eds) IPCC special report on the ocean and cryosphere in a changing climate. IPCC, Geneva

IPCC (Intergovernmental Panel on Climate Change) (2018) Global warming of 1.5°C. In: Masson-Delmotte V, Zhai P, Pörtner HO, Roberts D, Skea J, Shukla PR, Pirani A et al (eds) An IPCC special report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of

climate change, sustainable development, and efforts to eradicate poverty. IPCC, Geneva
IUCN (International Union for Conservation of Nature) (2016) Increasing marine protected area coverage for effective marine biodiversity conservation. In: Prepared for the World Conservation Congress 2016, Resolution 050, Hawaii, September 1–10. https://portals.iucn.org/library/sites/library/files/resrecfiles/WCC_2016_RES_050_EN.pdf
IUCN (2017) Guidelines for using the IUCN Red List categories and criteria. Version 13. IUCN, Standards and Petitions Subcommittee, Gland
IUCN (2019) The IUCN Red List of threatened species. IUCN, Gland. https://www.iucnredlist.org
IUCN-WCPA (World Commission on Protected Areas) (2019) Recognising and reporting other effective area-based conservation measures. IUCN, Gland. https://portals.iucn.org/library/sites/library/files/documents/PATRS-003-En.pdf
Jablonski D, Roy K, Valentine JW (2006) Out of the tropics: evolutionary dynamics of the latitudinal diversity gradient. Science 314(5796):102–106. https://doi.org/10.1126/science.1130880
Jablonski D, Belanger CL, Berke SK, Huang S, Krug AZ, Roy K, Tomasovych A, Valentine JW (2013) Out of the tropics, but how? Fossils, bridge species, and thermal ranges in the dynamics of the marine latitudinal diversity gradient. Proc Natl Acad Sci U S A 110(26):10487–10494. https://doi.org/10.1073/pnas.1308997110
Jackson JBC, Kirby MX, Berger WH, Bjorndal KA, Botsford LW, Bourque BJ, Bradbury RH et al (2001) Historical overfishing and the recent collapse of coastal ecosystems. Science 293(5530):629–638. https://doi.org/10.1126/science.1059199
Jaramillo-Legorreta AM, Cardenas-Hinojosa G, Nieto-Garcia E, Rojas-Bracho L, Thomas L, Ver Hoef JM, Moore J et al (2019) Decline towards extinction of Mexico's Vaquita Porpoise (*Phocoena sinus*). R Soc Open Sci 6(7):190598. https://doi.org/10.1098/rsos.190598
Jenkins CN, Van Houten KS (2016) Global and regional priorities for marine biodiversity protection. Biol Conserv 204(part B (December)):333–339
Jones KR, Klein CJ, Halpern BS, Venter O, Grantham H, Kuempel CD, Shumway N et al (2018) The location and protection status of earth's diminishing marine wilderness. Curr Biol 28(15):2506–2512. https://doi.org/10.1016/j.cub.2018.06.010
Jones KR, Klein CJ, Grantham HS, Possingham HP, Halpern BS, Burgess ND, Butchart SHM, Robinson JG et al (2020) Area requirements to safeguard earth's marine species. One Earth 2(2):188–196
Kaiser MJ (2005) Are marine protected areas a red herring or fisheries panacea? Can J Fish Aquat Sci 62(5):1194–1199. https://doi.org/10.1139/f05-056
Keith SA, Kerswell AP, Connolly SR (2014) Global diversity of marine macroalgae: environmental conditions explain less variation in the tropics. Glob Ecol Biogeogr 23(5):517–529. https://doi.org/10.1111/geb.12132
Kerswell AP (2006) Global biodiversity patterns of benthic marine algae. Ecology 87(10):2479–2488. https://doi.org/10.1890/0012-9658(2006)87[2479:gbpobm]2.0.co;2
Klein CJ, Brown CJ, Halpern BS, Segan DB, McGowan J, Beger M, Watson JEM (2015) Shortfalls in the global protected area network at representing marine biodiversity. Sci Rep 5:17539. https://doi.org/10.1038/srep17539
Kockel A, Ban NC, Costa M, Dearden P (2019) Evaluating approaches for scaling up community-based marine protected areas into socially equitable and ecologically representative networks. Conserv Biol 34(1):137–147. https://doi.org/10.1111/cobi.13368
Krause-Jensen D, Duarte CM (2016) Substantial role of macroalgae in marine carbon sequestration. Nat Geosci 9(October):737–742. https://doi.org/10.1038/ngeo2790
Krause-Jensen D, Lavery P, Serrano O, Marbà N, Masque P, Duarte CM (2018) Sequestration of macroalgal carbon: the elephant in the blue carbon room. Biol Lett 14(6):20180236. https://doi.org/10.1098/rsbl.2018.0236
Kroodsma DA, Mayorga J, Hochberg T, Miller NA, Boerder K, Ferretti F, Wilson A et al (2018) Tracking the global footprint of fisheries. Science 359(6378):904–908. https://doi.org/10.1126/science.aao5646
Krumhansl KA, Okamoto DK, Rassweiler A, Novak M, Bolton JJ, Cavanaugh KC, Connell SD et al (2016) Global patterns of kelp forest change over the past half-century. Proc Natl Acad Sci U S A 113(48):13785–13790. https://doi.org/10.1073/pnas.1606102113
Kuempel CD, Jones KR, Watson JEM, Possingham HP (2019) Quantifying biases in marine-protected-area placement relative to abatable threats. Conserv Biol 33(6):1350–1359. https://doi.org/10.1111/cobi.13340
Kuparinen A, Festa-Bianchet M (2017) Harvest-induced evolution: insights from aquatic and terrestrial systems. Philos Trans R Soc B 372(1712):20160036. https://doi.org/10.1098/rstb.2016.0036
Lamarche G, Orpin AR, Mitchell JS, Pallentin A (2016) Benthic habitat mapping. In: Clark MR, Consalvey M, Rowden AA (eds) Biological sampling in the deep sea. Blackwell, Chichester, pp 80–102. https://doi.org/10.1002/9781118332535.ch5
Langle-Flores A, Ocelík P, Pérez-Maqueo O (2017) "The role of social networks in the sustainability transformation of cabo pulmo: a multiplex perspective," in "Coastal resilience: exploring the many challenges from different viewpoints," edited by M.L. Martinez, A. Taramelli, and R. Silva, special issue. J Coast Res 77(77):134–142. https://doi.org/10.2112/SI77-014.1
Lefcheck JS, Byrnes JEK, Isbell F, Gamfeldt L, Griffin JN, Eisenhauer N, Hensel MJS et al (2015) Biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. Nat Commun 6(April):6936. https://doi.org/10.1038/ncomms7936
Lenton TM (2013) Environmental tipping points. Annu Rev Env Resour 38(October):1–29. https://doi.org/10.1146/annurev-environ-102511-084654
Lenton TM, Rockström J, Gaffney O, Rahmstorf S, Richardson K, Steffen W, Joachim Schellnhuber H (2019) Climate tipping points—too risky to bet against. Nature 575(7784):592–595. https://doi.org/10.1038/d41586-019-03595-0
Leroy A, Morin M (2018) Innovation in the decision-making process of the RFMOs. Mar Policy 97(November):156–162. https://doi.org/10.1016/j.marpol.2018.05.025
Lester SE, Halpern BS (2008) Biological responses in marine no-take reserves versus partially protected areas. Mar Ecol Prog Ser 367(September):49–56. https://doi.org/10.3354/meps07599
Lester SE, Halpern BS, Grorud-Colvert K, Lubchenco J, Ruttenberg BI, Gaines SD, Airamé S, Warner RR (2009) Biological effects within no-take marine reserves: a global synthesis. Mar Ecol Prog Ser 384(May):33–46. https://doi.org/10.3354/meps08029
Lidström S, Johnson AF (2019) Ecosystem-based fisheries management: a perspective on the critique and development of the concept. Fish Fish 21(1):216–222. https://doi.org/10.1111/faf.12418
Lindstrom E, Gunn J, Fischer A, McCurdy A, Glover LK (2012) A Framework for Ocean Observing. Intergovernmental Oceanographic Commission Information Document 1284. United Nations Educational, Scientific and Cultural Organization, Paris. https://unesdoc.unesco.org/ark:/48223/pf0000211260
Lipcius RN, Crowder LB, Morgan LE (2005) Metapopulation structure and marine reserves. In: Norse EA, Crowder LB (eds) Marine conservation biology: the science of maintaining the sea's biodiversity. Island, Washington, DC, pp 328–345
Liu L, Xu W, Yue Q, Teng X, Hu H (2018) Problems and countermeasures of coastline protection and utilization in China. Ocean Coast Manag 153(March):124–130. https://doi.org/10.1016/j.ocecoaman.2017.12.016
Loke LHL, Heery EC, Todd PA (2019) Shoreline defences. In: Sheppard C (ed) World seas: an environmental evaluation. Ecological issues

and environmental impacts, vol 3, 2nd edn. Academic Press, London, pp 491–504

Longépé N, Hajduch G, Ardianto R, de Joux R, Nhunfat B, Marzuki MI, Fablet R et al (2018) Completing fishing monitoring with spaceborne vessel detection system (VDS) and automatic identification system (AIS) to assess illegal fishing in Indonesia. Mar Pollut Bull 131(B):33–39. https://doi.org/10.1016/j.marpolbul.2017.10.016

Lotze HK, Tittensor DP, Bryndum-Buchholz A, Eddy TD, Cheung WWL, Galbraith ED, Barange M et al (2019) Global ensemble projections reveal trophic amplification of ocean biomass declines with climate change. Proc Natl Acad Sci U S A 116(26):12907–12912. https://doi.org/10.1073/pnas.1900194116

Lough JM, Anderson KD, Hughes TP (2018) Increasing thermal stress for tropical coral reefs: 1871–2017. Sci Rep 8(April):6079. https://doi.org/10.1038/s41598-018-24530-9

Lovelock CE, Cahoon DR, Friess DA, Guntenspergen GR, Krauss KW, Reef R, Rogers K et al (2015) The vulnerability of Indo-Pacific mangrove forests to sea-level rise. Nature 526(October):559–563. https://doi.org/10.1038/nature15538

Lucifora LO, García VB, Worm B (2011) Global diversity hot spots and conservation priorities for sharks. PloS One 6(5):e19356. https://doi.org/10.1371/journal.pone.0019356

MA (Millennium Ecosystem Assessment) (2005) Ecosystems and human well-being: synthesis. Island, Washington, DC

Mace GM, Hails RS, Cryle P, Harlow J, Clarke SJ (2015) Towards a risk register for natural capital. J Appl Ecol 52(3):641–653. https://doi.org/10.1111/1365-2664.12431

MacNeil MA, Graham NAJ, Cinner JE, Wilson SK, Williams ID, Maina J, Newman S et al (2015) Recovery potential of the world's coral reef fishes. Nature 520(April):341–344. https://doi.org/10.1038/nature14358

MacNeil MA, Mellin C, Matthews S, Wolff NH, McClanahan TR, Devlin M, Drovandi C, Mengersen K, Graham NAJ (2019) Water quality mediates resilience on the great barrier reef. Nat Ecol Evol 3(April):620–627. https://doi.org/10.1038/s41559-019-0832-3

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–253. https://doi.org/10.1038/35012251

Martin SL, Ince L, Grooves T (2016) An ecosystem services perspective for the oceanic eastern tropical pacific: commercial fisheries, carbon storage, recreational fishing, and biodiversity. Front Mar Sci 3(April):50. https://doi.org/10.3389/fmars.2016.00050

McCauley DJ, Pinsky ML, Palumbi SR, Estes JA, Joyce FH, Warner RR (2015) Marine defaunation: animal loss in the global ocean. Science 347(6219):1255641. https://doi.org/10.1126/science.1255641

McClanahan TR, Cinner JE, Maina J, Graham NAJ, Daw TM, Stead SM, Wamukota A et al (2008) Conservation action in a changing climate. Conserv Lett 1(2):53–59. https://doi.org/10.1111/j.1755-263X.2008.00008_1.x

McGowan JA, Cayan DR, Dorman LM (1998) Climate-ocean variability and ecosystem response in the Northeast Pacific. Science 281(5374):210–217. https://doi.org/10.1126/science.281.5374.210

McKinley DC, Miller-Rushing AJ, Ballard HL, Bonney R, Brown H, Cook-Patton SC, Evans DM et al (2017) Citizen science can improve conservation science, natural resource management, and environmental protection. Biol Conserv 208(April):15–28. https://doi.org/10.1016/j.biocon.2016.05.015

Mcleod E, Chmura GL, Bouillon S, Salm R, Björk M, Duarte CM, Lovelock CE, Schlesinger WH, Silliman BR (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front Ecol Environ 9(10):552–560. https://doi.org/10.1890/110004

McOwen C, Weatherdon LV, Bochove J, Sullivan E, Blyth S, Zockler C, Stanwell-Smith D et al (2017) A global map of saltmarshes. Biodivers Data J 5(March):e11764. https://doi.org/10.3897/BDJ.5.e11764

Mellin C, MacNeil MA, Cheal AJ, Emslie MJ, Caley MJ (2016) Marine protected areas increase resilience among coral reef communities. Ecol Lett 19(6):629–637. https://doi.org/10.1111/ele.12598

Merrie A, Dunn DC, Metian M, Boustany AM, Takei Y, Elferink AO, Ota Y et al (2014) An ocean of surprises—trends in human use, unexpected dynamics and governance challenges in areas beyond national jurisdiction. Glob Environ Chang 27(July):19–31. https://doi.org/10.1016/j.gloenvcha.2014.04.012

Micheli F, Halpern BS, Botsford LW, Warner RR (2004) Trajectories and correlates of community change in no-take marine reserves. Ecol Appl 14(6):1709–1723. https://doi.org/10.1890/03-5260

Micheli F, Saenz-Arroyo A, Greenley A, Vazquez L, Espinoza Montes JA, Rossetto M, De Leo GA (2012) Evidence that marine reserves enhance resilience to climatic impacts. PloS One 7(7):e40832. https://doi.org/10.1371/journal.pone.0040832

Miller KA, Thompson KF, Johnston P, Santillo D (2018) An overview of seabed mining including the current state of development, environmental impacts, and knowledge gaps. Front Mar Sci 4(January):418. https://doi.org/10.3389/fmars.2017.00418

Miloslavich P, Bax NJ, Simmons SE, Klein E, Appeltans W, Aburto-Oropeza O, Shin YJ (2018a) Essential ocean variables for global sustained observations of biodiversity and ecosystem changes. Glob Chang Biol 24(6):2416–2433. https://doi.org/10.1111/gcb.14108

Miloslavich P, Seeyave S, Muller-Karger F, Bax N, Ali E, Delgado C, Evers-King H et al (2018b) Challenges for global ocean observation: the need for increased human capacity. J Oper Oceanogr 12(S2):137–156. https://doi.org/10.1080/1755876X.2018.1526463

Molinos JG, Halpern BS, Schoeman DS, Brown CJ, Kiessling W, Moore PJ, Pandolfi JM et al (2016) Climate velocity and the future global redistribution of marine biodiversity. Nat Clim Change 6(January):83–87. https://doi.org/10.1038/nclimate2769

Molnar JL, Gamboa RL, Revenga C, Spalding MD (2008) Assessing the global threat of invasive species to marine biodiversity. Front Ecol Environ 6(9):485–492

Mora C, Myers RA, Coll M, Libralato S, Pitcher TJ, Sumaila RU, Zeller D et al (2009) Management effectiveness of the world's marine fisheries. PLoS Biol 7(6):e1000131. https://doi.org/10.1371/journal.pbio.1000131

Mora C, Tittensor DP, Adl S, Simpson AGB, Worm B (2011) How many species are there on earth and in the ocean? PLoS Biol 9(8):e1001127. https://doi.org/10.1371/journal.pbio.1001127

Mora C, Spirandelli D, Franklin EC, Lynham J, Kantar MB, Miles W, Smith CZ et al (2018) Broad threat to humanity from cumulative climate hazards intensified by greenhouse gas emissions. Nat Clim Change 8(December):1062–1071. https://doi.org/10.1038/s41558-018-0315-6

Muller-Karger FE, Miloslavich P, Bax NJ, Simmons S, Costello MJ, Sousa Pinto I, Canonico G et al (2018) Advancing marine biological observations and data requirements of the complementary essential ocean variables (EOVs) and essential biodiversity variables (EBVs) frameworks. Front Mar Sci 5(June):211. https://doi.org/10.3389/fmars.2018.00211

Mumby PJ, Harborne AR (2010) Marine reserves enhance the recovery of corals on Caribbean reefs. PloS One 5(1):e8657. https://doi.org/10.1371/journal.pone.0008657

Mumby PJ, Green EP, Edwards AJ, Clark CD (1999) The cost-effectiveness of remote sensing for tropical coastal resources assessment and management. J Environ Manage 55(3):157–166. https://doi.org/10.1006/jema.1998.0255

Mumby PJ, Hedley JD, Chisholm JRM, Clark CD, Ripley H, Jaubert J (2004) The cover of living and dead corals from airborne remote sensing. Coral Reefs 23(July):171–183. https://doi.org/10.1007/s00338-004-0382-1

Mumby PJ, Harborne AR, Williams J, Kappel CV, Brumbaugh DR, Micheli F, Holmes KE et al (2007) Trophic cascade facilitates coral recruitment in a marine reserve. Proc Natl Acad Sci U S A 104(20):8362–8367. https://doi.org/10.1073/pnas.0702602104

Naeem S (2002) Ecosystem consequences of biodiversity loss: evolution of a paradigm. Ecology 83(6):1537–1552. https://doi.org/10.1890/0012-9658(2002)083[1537:ECOBLT]2.0.CO;2

Narayan S, Beck MW, Wilson P, Thomas CJ, Guerrero A, Shepard CC, Reguero BG et al (2017) The value of coastal wetlands for flood damage reduction in the Northeastern USA. Sci Rep 7(August):9463. https://doi.org/10.1038/s41598-017-09269-z

Navarro LM, Fernández N, Guerra C, Guralnick R, Kissling WD, Londoño MC, Muller-Karger F et al (2017) Monitoring biodiversity change through effective global coordination. Curr Opin Environ Sustain 29(December):158–169. https://doi.org/10.1016/j.cosust.2018.02.005

NEAFC/OSPAR (2014) Collective arrangement between competent International Organisations on Cooperation and Coordination regarding selected areas in areas beyond National Jurisdiction in the North-East Atlantic. NEAFC, London

NEAFC/OSPAR (North-East Atlantic Fisheries Commission/Oslo Paris Commission) (2008) Memorandum of Understanding between the North East Atlantic Fisheries Commission (NEAFC) and the OSPAR Commission. NEAFC, London. https://www.ospar.org/site/assets/files/1357/mou_neafc_ospar.pdf

Nellemann C, Corcoran E, Duarte CM, Valdés L, De Young C, Fonseca L, Grimsditch G (2009) Blue carbon: the role of healthy oceans in binding carbon. United Nations Environmental Programme, GRID-Arendal, Arendal, Norway. https://cld.bz/bookdata/WK8FNPt/basic-html/index.html#1

Nijland W, Reshitnyk L, Rubidge E (2019) Satellite remote sensing of canopy-forming kelp on a complex coastline: a novel procedure using the landsat image archive. Rem Sens Environ 220(January):41–50. https://doi.org/10.1016/j.rse.2018.10.032

Niner HJ, Ardron JA, Escobar EG, Gianni M, Jaeckel A, Jones DOB, Levin LA et al (2018) Deep-sea mining with no net loss of biodiversity—an impossible aim. Front Mar Sci 5(March):53. https://doi.org/10.3389/fmars.2018.00053

Norse EA (1993) Global marine biological diversity: a strategy for building conservation into decision making. Island, Washington, DC

Norse EA (2005) Ending the range wars on the last frontier: zoning the sea. In: Norse EA, Crowder LB (eds) Marine conservation biology: the science of maintaining the sea's biodiversity. Island, Washington, DC, pp 422–443

Norse EA, Brooke S, Cheung WWL, Clark MR, Ekeland I, Froese R, Gjerde KM et al (2012) Sustainability of deep-sea fisheries. Mar Policy 36(2):307–320. https://doi.org/10.1016/j.marpol.2011.06.008

O'Leary BC, Brown RL, Johnson DE, von Nordheim H, Ardron J, Packeiser T, Roberts CM (2012) The first network of marine protected areas (mpas) in the high seas: the process, the challenges and where next. Mar Policy 36(3):598–605. https://doi.org/10.1016/j.marpol.2011.11.003

O'Leary BC, Winther-Janson M, Bainbridge JM, Aitken J, Hawkins JP, Roberts CM (2016) Effective coverage targets for ocean protection. Conserv Lett 9(6):398–404. https://doi.org/10.1111/conl.12247

O'Leary BC, Allen HL, Yates KL, Page RW, Tudhope AW, McClean C, Rogers AD, Hawkins JP, Roberts CM (2018) 30 x 30: a blue print for ocean protection—how we can protect 30% of our oceans by 2030. Greenpeace, London. https://storage.googleapis.com/planet4-new-zealand-stateless/2019/04/f136cbfb-5db0f88b-greenpeace-30x30-blueprint-report1.pdf

Oziel L, Neukermans G, Ardyna M, Lancelot C, Tison JL, Wassmann P, Sirven J, Ruiz-Pino D, Gascard JC (2017) Role for Atlantic inflows and sea ice loss on shifting phytoplankton blooms in the Barents sea. J Geophys Res Oceans 122(6):5121–5139. https://doi.org/10.1002/2016JC012582

Paleczny M, Hammill E, Karpouzi V, Pauly D (2015) Population trend of the world's monitored seabirds, 1950-2010. PloS One 10(6):e0129342. https://doi.org/10.1371/journal.pone.0129342

Palmer MW, Earls PG, Hoagland BW, White PS, Wohlgemuth T (2002) Quantitative tools for perfecting species lists. Environmetrics 13(2):121–137. https://doi.org/10.1002/env.516

Pandolfi JM, Bradbury RH, Sala E, Hughes TP, Bjorndal KA, Cooke RG, McArdle D et al (2003) Global trajectories of the long-term decline of coral reef ecosystems. Science 301(5635):955–958. https://doi.org/10.1126/science.1085706

Pappalardo P, Fernández M (2014) Mode of larval development as a key factor to explain contrasting effects of temperature on species richness across oceans. Glob Ecol Biogeogr 23(1):12–23. https://doi.org/10.1111/geb.12115

Parrish JK, Burgess H, Weltzin JF, Fortson L, Wiggins A, Simmons B (2018) Exposing the science in citizen science: fitness to purpose and intentional design. Integr Comparat Biol 58(1):150–160. https://doi.org/10.1093/icb/icy032

Pauly D, Zeller D (2016) Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining. Nat Commun 7(January):10244. https://doi.org/10.1038/ncomms10244

Pendleton LH, Thébaud O, Mongruel RC, Levrel H (2016) Has the value of global marine and coastal ecosystem services changed? Mar Policy 64(February):156–158. https://doi.org/10.1016/j.marpol.2015.11.018

Pentz B, Klenk N, Ogle S, Fisher JAD (2018) Can regional fisheries management organizations (RFMOs) manage resources effectively during climate change? Mar Policy 92(June):13–20. https://doi.org/10.1016/j.marpol.2018.01.011

Pérez-Matus A, Carrasco S, Gelcich S, Fernandez M, Wieters E (2017) Exploring the effects of fishing pressure and upwelling intensity over subtidal kelp forest communities in Central Chile. Ecosphere 8(5):e01808. https://doi.org/10.1002/ecs2.1808

Peters H, O'Leary BC, Hawkins JP, Carpenter KE, Roberts CM (2013) Conus: first comprehensive conservation red list assessment of a marine gastropod mollusc genus. PloS One 8(12):e83353. https://doi.org/10.1371/journal.pone.0083353

Petersen S, Krätschell A, Augustin N, Jamieson J, Hein JR, Hannington MD (2016) News from the seabed—geological characteristics and resource potential of deep-sea mineral resources. Mar Policy 70(August):175–187. https://doi.org/10.1016/j.marpol.2016.03.012

Petrossian GA, Deby RA, Clarke RV (2018) Illegal longline fishing and albatross extinction risk. Oryx 52(2):336–345. https://doi.org/10.1017/S0030605316000818

Pham CK, Ramirez-Llodra E, Alt CHS, Amaro T, Bergmann M, Canals M, Company JB et al (2014) Marine litter distribution and density in European seas, from the shelves to deep basins. PloS One 9(4):e95839. https://doi.org/10.1371/journal.pone.0095839

Phillips RA, Gales R, Baker GB, Double MC, Favero M, Quintana F, Tasker ML et al (2016) The conservation status and priorities for albatrosses and large petrels. Biol Conserv 201(September):169–183. https://doi.org/10.1016/j.biocon.2016.06.017

Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P et al (2004) Ecosystem-based fishery management. Science 305(5682):346–347. https://doi.org/10.1126/science.1098222

Pikitch E, Boersma PD, Boyd IL, Conover DO, Cury P, Essington T, Heppell SS et al (2012) Little fish, big impact: managing a crucial link in ocean food webs. Lenfest Ocean Program, Washington, DC. https://www.oceanconservationscience.org/foragefish/files/Little%20Fish,%20Big%20Impact.pdf

Pine WE, Martell SJD, Walters CJ, Kitchell JF (2009) Counter intuitive responses of fish populations to management actions: some common causes and implications for predictions based on ecosystem modelling. Fisheries 34(4):165–180. https://doi.org/10.1577/1548-8446-34.4.165

Pinsky ML, Palumbi SR (2014) Meta-analysis reveals lower genetic diversity in overfished populations. Mol Ecol 23(1):29–39. https://doi.org/10.1111/mec.12509

Pitcher T, Kalikoski D, Pramod G, Short K (2009) Not honouring the code. Nature 457(February):658–659. https://doi.org/10.1038/457658a

Polacheck T (2012) Politics and independent scientific advice in RFMO processes: a case study of crossing boundaries. Mar Policy 36(1):132–141. https://doi.org/10.1016/j.marpol.2011.04.006

Polidoro BA, Carpenter KE, Collins L, Duke NC, Ellison AM, Ellison JC, Farnsworth EJ et al (2010) The loss of species: mangrove extinction risk and geographic areas of global concern. PloS One 5(4):e10095. https://doi.org/10.1371/journal.pone.0010095

Pollnac RB, Crawford BR, Gorospe MLG (2001) Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. Ocean Coast Manag 44(11–12):683–710. https://doi.org/10.1016/S0964-5691(01)00075-8

Poloczanska ES, Brown CJ, Sydeman WJ, Kiessling W, Schoeman DS, Moore PJ, Brander K et al (2013) Global imprint of climate change on marine life. Nat Clim Change 3(October):919–925. https://doi.org/10.1038/nclimate1958

Poloczanska ES, Burrows MT, Brown CJ, García Molinos J, Halpern BS, Hoegh-Guldberg O, Kappel CV et al (2016) Responses of marine organisms to climate change across oceans. Front Mar Sci 3(May):62. https://doi.org/10.3389/fmars.2016.00062

Pompa S, Ehrlich P, Ceballos G (2011) Global distribution and conservation of marine mammals. Proc Natl Acad Sci U S A 108(33):13600–13605

Pons M, Melnychuk MC, Hilborn R (2018) Management effectiveness of large pelagic fisheries in the high seas. Fish Fish 19(2):260–270. https://doi.org/10.1111/faf.12253

Puig P, Canals M, Company JB, Martín J, Amblas D, Lastras G, Palanques A, Calafat AM (2012) Ploughing the deep sea floor. Nature 489(September):286–289. https://doi.org/10.1038/nature11410

Purcell SW, Polidoro BA, Hamel JF, Gamboa RU, Mercier A (2014) The cost of being valuable: predictors of extinction risk in marine invertebrates exploited as luxury seafood. Proc R Soc B 281(1781):20133296. https://doi.org/10.1098/rspb.2013.3296

Pusceddu A, Gambi C, Corinaldesi C, Scopa M, Danovaro R (2014) Relationships between meiofaunal biodiversity and prokaryotic heterotrophic production in different tropical habitats and oceanic regions. PloS One 9(3):e91056. https://doi.org/10.1371/journal.pone.0091056

Raudsepp-Hearne C, Peterson GD, Bennett EM (2010) Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc Natl Acad Sci U S A 107(11):5242–5247. https://doi.org/10.1073/pnas.0907284107

Rex MA, Etter RJ (2010) Deep-sea biodiversity: pattern and scale. Harvard University Press, Cambridge, MA

Reygondeau G (2019) Current and future biogeography of exploited marine exploited groups under climate change. In: Cisneros-Montemayor AM, Cheung WWL, Ota Y (eds) Predicting future oceans: sustainability of ocean and human systems amidst global environmental change. Elsevier, Oxford, pp 87–101

Reygondeau G, Dunn D (2018) Data science and its relationship to big data and data-driven decision making. Big Data 1(1):51–59. https://doi.org/10.1089/big.2013.1508

Rice JC, Legacè È (2007) When control rules collide: a comparison of fisheries management reference points and IUCN criteria for assessing risk of extinction. ICES J Mar Sci 64(4):718–722. https://doi.org/10.1093/icesjms/fsm011

Rice J, Ridgeway L (2010) Conservation of biodiversity and fisheries management. In: Grafton RQ, Hilborn R, Squires D, Tait M, Williams M (eds) Handbook of marine fisheries conservation and management. Oxford University Press, Oxford, pp 139–149

Richards DR, Friess DA (2016) Rates and drivers of mangrove deforestation in Southeast Asia, 2000–2012. Proc Natl Acad Sci U S A 113(2):344–349. https://doi.org/10.1073/pnas.1510272113

Ritter F, Panigada S (2019) Collisions of vessels with cetaceans—the underestimated threat. In: Sheppard C (ed) World seas: an environmental evaluation. Ecological issues and environmental impacts, vol 3, 2nd edn. Academic Press, London, pp 531–547

Rivadeneira MM, Fernandez M (2005) Shifts in southern endpoints of distribution in rocky intertidal species along the South-Eastern Pacific Coast. J Biogeogr 32(2):203–209. https://doi.org/10.1111/j.1365-2699.2004.01133.x

RLS (Reef Life Survey) (2019) RLS data informing global biodiversity targets. https://reeflifesurvey.com/rls-data-informing-global-biodiversity-targets/. Accessed 25 Jun 2019

Roberts CM, Bohnsack JA, Gell F, Hawkins JP, Goodridge R (2001) Effects of marine reserves on adjacent fisheries. Science 294(5548):1920–1923. https://doi.org/10.1126/science.294.5548.1920

Roberts CM, O'Leary BC, McCauley DJ, Cury PM, Duarte CM, Lubchenco J, Pauly D et al (2017) Marine reserves can mitigate and promote adaptation to climate change. Proc Natl Acad Sci U S A 114(24):6167–6175. https://doi.org/10.1073/pnas.1701262114

Robinson JPW, Wilson SK, Robinson J, Gerry C, Lucas J, Assan C, Govinden R et al (2019) Productive instability of coral reef fisheries after climate-driven regime shifts. Nat Ecol Evol 3(February):183–190. https://doi.org/10.1038/s41559-018-0715-z

Rockström J, Steffen W, Noone K, Persson Å, Chapin FS, Lambin EF, Lenton TM et al (2009) A safe operating space for humanity. Nature 461(September):472–475. https://doi.org/10.1038/461472a

Rogers AD (2015) Environmental change in the deep ocean. Annu Rev Env Resour 40(November):1–38. https://doi.org/10.1146/annurev-environ-102014-021415

Rogers AD (2018) The biology of seamounts: 25 years on. Adv Mar Biol 79:137–224. https://doi.org/10.1016/bs.amb.2018.06.001

Rogers AD, Gianni M (2010) The implementation of UNGA resolutions 61/105 and 64/72 in the management of deep-sea fisheries on the high seas. Report prepared for the deep-sea conservation coalition. International Programme on State of the Ocean, London. http://www.savethehighseas.org/publicdocs/61105-Implemention-finalreport.pdf

Rogers AD, Brierley A, Croot P, Cunha MR, Danovaro R, Devey C, Hoel AH et al (2015) Delving deeper: critical challenges for 21st century deep-sea research. Position Paper 22. European Marine Board, Ostend, Belgium. https://www.marineboard.eu/sites/marineboard.eu/files/public/publication/EMB_PP22_Web_v4.pdf

Rossiter JS, Levine A (2014) What makes a 'successful' marine protected area? The unique context of Hawaii's fish replenishment areas. Mar Policy 44(February):196–203. https://doi.org/10.1016/j.marpol.2013.08.022

Rousseau Y, Watson RA, Blanchard JL, Fulton EA (2019) Evolution of global marine fishing fleets and the response of fished resources. Proc Natl Acad Sci U S A 116(25):12238–12243

Roux JP, van der Lingen C, Gibbons MJ, Moroff NE, Shannon LJ, Smith AD, Cury PM (2013) Jellification of marine ecosystems as a consequence of overfishing small pelagic fish: lessons from the Benguela. Bull Mar Sci 89(1):249–284. https://doi.org/10.5343/bms.2011.1145

Rowlands G, Brown J, Soule B, Boluda PT, Rogers AD (2019) Satellite surveillance of the Ascension Island exclusive economic zone and marine protected area. Mar Policy 101(March):39–50. https://doi.org/10.1016/j.marpol.2018.11.006

Roy K, Jablonski D, Valentine JW, Rosenberg G (1998) Marine latitudinal diversity gradients: tests of causal hypotheses. Proc Natl Acad Sci U S A 95(7):3699–3702. https://doi.org/10.1073/pnas.95.7.3699

Sadovy de Mitcheson Y (2016) Mainstreaming fish spawning aggregations into fishery management calls for truly precautionary

approach. BioScience 66(4):295–306. https://doi.org/10.1093/biosci/biw013
Sadovy de Mitcheson Y, Andersson AA, Hofford A, Law CSW, Hau LCY, Pauly D (2018) Out of control means off the menu: the case for ceasing consumption of luxury products from highly vulnerable species when international trade cannot be adequately controlled; shark fin as a case study. Mar Policy 98(December):115–120. https://doi.org/10.1016/j.marpol.2018.08.012
Sala E, Giakoumi S (2017) No-take marine reserves are the most effective protected areas in the ocean. ICES J Mar Sci 75(3):1166–1168. https://doi.org/10.1093/icesjms/fsx059
Sala E, Lubchenco J, Grorud-Colvert K, Novelli C, Roberts C, Sumaila UR (2018a) Assessing real progress towards effective ocean protection. Mar Policy 91(May):11–13. https://doi.org/10.1016/j.marpol.2018.02.004
Sauermann H, Franzoni C (2015) Crowd science user contribution patterns and their implications. Proc Natl Acad Sci U S A 112(3):679–684. https://doi.org/10.1073/pnas.1408907112
Sciberras M, Jenkins SR, Kaiser MJ, Hawkins SJ, Pullin AS (2013) Evaluating the biological effectiveness of fully and partially protected marine areas. Environ Evid 2(February):4. https://doi.org/10.1186/2047-2382-2-4
Selig ER, Turner WR, Troëng S, Wallace BP, Halpern BS, Kaschner K, Lascelles BG, Carpenter KE, Mittermeier RA (2014) Global priorities for marine biodiversity conservation. PloS One 9(1):e82898
Sherley RB, Ludynia K, Dyer BM, Lamont T, Makhado AB, Roux JP, Scales KL, Underhill LG, Votier SC (2017) Metapopulation tracking juvenile penguins reveals an ecosystem-wide ecological trap. Curr Biol 27(4):563–568. https://doi.org/10.1016/j.cub.2016.12.054
Shlesinger T, Loya Y (2019) Breakdown in spawning synchrony: a silent threat to coral persistence. Science 365(6457):1002–1007. https://doi.org/10.1126/science.aax0110
Short F, Carruthers T, Dennison W, Waycott M (2007) “Global seagrass distribution and diversity: a bioregional model,” in “The biology and ecology of seagrasses,” edited by B.W. Touchette, special issue. J Exp Mar Biol Ecol 350(1–2):3–20. https://doi.org/10.1016/j.jembe.2007.06.012
Sigwart JD, Chen C, Thomas EA, Allcock AL, Böhm M, Seddon M (2019) Red listing can protect deep-sea biodiversity. Nat Ecol Evol 3(August):1134. https://doi.org/10.1038/s41559-019-0930-2
Skroblin A, Carboon T, Bidu G, Chapman N, Miller M, Taylor K, Taylor W, Game ET, Wintle BA (2019) Including indigenous knowledge in species distribution modelling for increased ecological insights. Conserv Biol 35:587. https://doi.org/10.1111/cobi.13373
Smale DA, Wernberg T, Oliver ECJ, Thomsen MS, Harvey BP, Starub SC, Burrows MT et al (2019) Marine heatwaves threaten global biodiversity and the provision of ecosystem services. Nat Clim Change 9(April):306–312. https://doi.org/10.1038/s41558-019-0412-1
Small N, Munday M, Durance I (2017) The challenge of valuing ecosystem services that have no material benefits. Glob Environ Chang 44(May):57–67. https://doi.org/10.1016/j.gloenvcha.2017.03.005
Spalding MD, Ruffo S, Lacambra C, Meliane I, Hale LZ, Shepard CC, Beck WW (2014) The role of ecosystems in coastal protection: adapting to climate change and coastal hazards. Ocean Coast Manag 90(March):50–57. https://doi.org/10.1016/j.ocecoaman.2013.09.007
Stachowicz JJ, Bruno JF, Duffy JE (2007) Understanding the effects of marine biodiversity on communities and ecosystems. Annu Rev Ecol Evol Syst 38(December):739–766. https://doi.org/10.1146/annurev.ecolsys.38.091206.095659
Standing A (2015) Corruption and state-corporate crime in fisheries. U4 Issue Paper 15. U4 Anti-Corruption Resource Centre, Chr. Michelsen Institute, Bergen, Norway. https://www.u4.no/publications/corruption-and-state-corporate-crime-in-fisheries.pdf
Stehli FG, McAlester AL, Helsey CE (1967) Taxonomic diversity of recent bivalves and some implications for geology. Geol Soc Am Bull 78(4):455–466. https://doi.org/10.1130/0016-7606(1967)78[455:TDORBA]2.0.CO;2
Stehli FG, Douglas RG, Kafescioglu A (1972) Models for the evolution of Planktonic Foraminifera. In: Schopf TJM (ed) Models in paleobiology. Freeman, Cooper, San Francisco, pp 116–128
Stewart JS, Field JC, Markaida U, Gilly WF (2013) Behavioral ecology of Jumbo Squid (*Dosidicus gigas*) in relation to oxygen minimum zones. DeepSea Res II Top Stud Oceanogr 95(15):197–208. https://doi.org/10.1016/j.dsr2.2012.06.005
Stramma L, Prince ED, Schmidtko S, Luo J, Hoolihan JP, Visbeck M, Wallace DWR, Brandt P, Körtzinger A (2012) Expansion of oxygen minimum zones may reduce available habitat for tropical pelagic fishes. Nat Clim Change 2(January):33–37. https://doi.org/10.1038/nclimate1304
Strong JA, Andonegi E, Bizsel KC, Danovaro R, Elliott M, Franco A, Garces E et al (2015) Marine biodiversity and ecosystem function relationships: the potential for practical monitoring applications. Estuar Coast Shelf Sci 161(August):46–64. https://doi.org/10.1016/j.ecss.2015.04.008
Stuart-Smith RD, Bates AE, Lefcheck JS, Duffy JE, Baker SC, Thomson RJ, Stuart-Smith JF et al (2013) Integrating abundance and functional traits reveals new global hot spots of fish diversity. Nature 501(September):539–542. https://doi.org/10.1038/nature12529
Stuart-Smith RD, Edgar GJ, Barrett NS, Kininmonth SJ, Bates AE (2015) Thermal biases and vulnerability to warming in the world’s marine fauna. Nature 528(December):88–92. https://doi.org/10.1038/nature16144
Stuart-Smith RD, Edgar GJ, Barrett NS, Bates AE, Baker SC, Bax NJ, Becerro MA et al (2017) Assessing national biodiversity trends for rocky and coral reefs through the integration of citizen science and scientific monitoring programs. BioScience 67(2):134–146. https://doi.org/10.1093/biosci/biw180
Stuart-Smith RD, Brown CJ, Ceccarelli DM, Edgar GJ (2018) Ecosystem restructuring along the great barrier reef following mass coral bleaching. Nature 560(August):92–96. https://doi.org/10.1038/s41586-018-0359-9
Suazo CG, Cabezas LA, Moreno CA, Arata JA, Luna-Jorquera G, Simeone A, Adasme L et al (2014) Seabird bycatch in chile: a synthesis of its impacts, and a review of strategies to contribute to the reduction of a global phenomenon. Pac Seabirds 41(1–2):1–12. http://www.pacificseabirdgroup.org/publications/PacificSeabirds/VOL_41_1-2.pdf
Suding KN, Hobbs RJ (2009) Threshold models in restoration and conservation: a developing framework. Trends Ecol Evol 24(5):271–279. https://doi.org/10.1016/j.tree.2008.11.012
Sullivan JM, Constant V, Lubchenco J (2019) Extinction threats to life in the ocean and opportunities for their amelioration. In: Dasgupta P, Raven P, McIvor A (eds) Biological extinction: new perspectives. Cambridge University Press, Cambridge, pp 113–137. https://doi.org/10.1017/9781108668675
Sumaila UR, Dyck A, Cheung WWL (2013) Fisheries subsidies and potential catch loss in sids exclusive economic zones: food security implications. Environ Dev Econ 18(4):427–439. https://doi.org/10.1017/S1355770X13000156
Sunagawa S, Coelho LP, Chaffron S, Kultima JR, Labadie K, Salazar G, Djahanschiri B et al (2015) Structure and function of the global ocean microbiome. Science 348(6237):1261359. https://doi.org/10.1126/science.1261359
Sweetman AK, Thurber AR, Smith CR, Levin LA, Mora C, Wei CL, Gooday AJ et al (2017) Major impacts of climate change on deep-sea benthic ecosystems. Elementa Sci Anthropocene 5:4. https://doi.org/10.1525/elementa.203/
Tanhua T, McCurdy A, Fischer A, Appelatans W, Bax N, Currie K, DeYoung B et al (2019) What have we learned from the framework for ocean observing: evolution of the global ocean observing system. Front Mar Sci 6(August):471. https://doi.org/10.3389/fmars.2019.00471

Theobald EJ, Ettinger AK, Burgess HK, DeBey LB, Schmidt NR, Froehlich HE, Wagner C et al (2015) Global change and local solutions: tapping the unrealized potential of citizen science for biodiversity research. Biol Conserv 181(January):236–244. https://doi.org/10.1016/j.biocon.2014.10.021

Thomas N, Lucas R, Bunting P, Hardy A, Rosenqvist A, Simard M (2017) Distribution and drivers of global mangrove forest change, 1996–2010. PloS One 12(6):e0179302. https://doi.org/10.1371/journal.pone.0179302

Thomson JA, Burkholder DA, Heithaus MR, Fourqurean JW, Fraser MW, Stratton J, Kendrick GA (2015) Extreme temperatures, foundation species, and abrupt ecosystem change: an example from an iconic seagrass ecosystem. Glob Chang Biol 21(4):1463. https://doi.org/10.1111/gcb.1269474

Thorson G (1952) Zur jetzigen lage der marinen bodentier-okolgie. Zool Anz S16:276–327

Thorson G (1957) Bottom communities (sublittoral or shallow shelf). In: Hedgepeth JW (ed) Treatise on marine ecology and paleoecology. Geological Society of America, Ithaca, NY, pp 461–534. https://doi.org/10.1130/MEM67V1-p535

Thrush SE, Dayton PK (2002) Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. Annu Rev Ecol Syst 33(November):449–473. https://doi.org/10.1146/annurev.ecolsys.33.010802.150515

Tittensor DP, Baco-Taylor AR, Brewin P, Clark MR, Consalvey M, Hall-Spencer J, Rowden AA et al (2009) Predicting global habitat suitability for stony corals on seamounts. J Biogeogr 36(6). https://doi.org/10.1111/j.1365-2699.2008.02062.x

Tittensor DP, Mora C, Jetz W, Lotze HK, Ricard D, Berghe EV, Worm B (2010) Global patterns and predictors of marine biodiversity across taxa. Nature 466(August):1098–1101. https://doi.org/10.1038/nature09329

Tittensor DP, Beger M, Boerder K, Boyce DG, Cavanagh RD, Cosandey-Godin A, Crespo GO et al (2019) Integrating climate adaptation and biodiversity conservation in the global ocean. Sci Adv 5(11):eaay9969. https://doi.org/10.1126/sciadv.aay9969

Traganos D, Aggarwal B, Poursanidis D, Topouzelis K, Chrysoulakis N, Reinartz P (2018) Towards global-scale seagrass mapping and monitoring using sentinel-2 on google earth engine: the case study of the Aegean and Ionian Seas. Remote Sens (Basel) 10(8):1227. https://doi.org/10.3390/rs10081227

Travis J, Coleman FC, Auster PJ, Cury PM, Estes JA, Orensanz J, Peterson CH et al (2014) Integrating the invisible fabric of nature into fisheries management. Proc Natl Acad Sci U S A 111(2):581–584. https://doi.org/10.1073/pnas.1305853111

Trevathan-Tackett SM, Kelleway J, Macreadie PI, Beardall J, Ralph P, Bellgrove A (2015) Comparison of marine macrophytes for their contributions to blue carbon sequestration. Ecology 96(11):3043–3057. https://doi.org/10.1890/15-0149.1

UNEP-WCMC (United Nations Environment Programme World Conservation Monitoring Centre) and IUCN (International Union for Conservation of Nature) (2020) Marine protected planet. UNEP-WCMC and IUCN, Cambridge. www.protectedplanet.net

UNEP-WCMC, Short FT (2018) "Global Distribution of Seagrasses." Version 6.0. Sixth update to the data layer used in Green and Short (2003). UNEP-WCMC, Cambridge. http://data.unep-wcmc.org/datasets/7

UNEP-WCMC, WorldFish Centre, WRI, and TNC (2018) "Global Distribution of Warm-Water Coral Reefs, Compiled from Multiple Sources Including the Millennium Coral Reef Mapping Project." Version 4.0. Includes contributions from IMaRS-USF and IRD (2005), IMaRS-USF (2005), and Spalding et al. (2001). UNEP-WCMC, Cambridge. http://data.unep-wcmc.org/datasets/1

Unsworth RKF, Nordlund LM, Cullen-Unsworth LC (2019) Seagrass meadows support global fisheries production. Conserv Lett 12(1):e12566. https://doi.org/10.1111/conl.12566

van Overzee HMJ, Rijnsdorp AD (2015) Effects of fishing during the spawning period: implications for sustainable management. Rev Fish Biol Fish 25(March):65–83. https://doi.org/10.1007/s11160-014-9370-x

Velarde E, Ezcurra E, Anderson DW (2015a) "Seabird Diet Predicts Following-Season Commercial Catch of Gulf of California Pacific Sardine and Northern Anchovy," in "California Current System—Predators and the Preyscape," edited by D.G. Ainley, P.B. Adams, and J. Jahncke, special issue. J Mar Syst 146(June):82–88. https://doi.org/10.1016/j.jmarsys.2014.08.014

Velarde E, Ezcurra E, Horn MH, Patton RT (2015b) Warm oceanographic anomalies and fishing pressure drive seabirds nesting north. Sci Adv 1(5):e1400210. https://doi.org/10.1126/sciadv.1400210

Vergés A, Steinberg PD, Hay ME, Poore AG, Campbell AH, Ballesteros E, Heck KL Jr et al (2014) The tropicalization of temperate marine ecosystems: climate- mediated changes in herbivory and community phase shifts. Proc R Soc B Biol Sci 281(1789):20140846. https://doi.org/10.1098/rspb.2014.0846

Victorero L, Watling L, Deng Palomares ML, Nouvian C (2018) Out of sight, but within reach: a global history of bottom-trawled deep-sea fisheries from >400m depth. Front Mar Sci 5(April):98. https://doi.org/10.3389/fmars.2018.00098

Vihtakari M, Welcker J, Moe B, Chastel O, Tartu S, Hop H, Bech C, Descamps S, Gabrielsen GW (2018) Black-legged Kittiwakes as messengers of atlantification in the Arctic. Sci Rep 8(January):1178. https://doi.org/10.1038/s41598-017-19118-8

Wanless S, Harris MP, Redman P, Speakman JR (2005) Low energy values of fish as a probable cause of a major seabird breeding failure in the North Sea. Mar Ecol Prog Ser 294(June):1–8. https://doi.org/10.3354/meps294001

Ward P, Myers RA (2005) Shifts in open-ocean fish communities coinciding with the commencement of commercial fishing. Ecology 86(4):835–847. https://doi.org/10.1890/03-0746

Washburn TW, Turner PJ, Durden JM, Jones DOB, Weaver P, Van Dover CL (2019) Ecological risk assessment for deep-sea mining. Ocean Coast Manag 176(June):24–39. https://doi.org/10.1016/j.ocecoaman.2019.04.014

Waycott M, Duarte CM, Carruthers TJB, Orth RJ, Dennison WC, Olyarnik S, Calladine A et al (2009) Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc Natl Acad Sci U S A 106(30):12377–12381. https://doi.org/10.1073/pnas.0905620106

Webb TJ, Mindel BL (2015) Global patterns of extinction risk in marine and non-marine systems. Curr Biol 25(4):506–511. https://doi.org/10.1016/j.cub.2014.12.023

Wernberg T, Filbee-Dexter K (2019) Missing the marine forest for the trees. Mar Ecol Prog Ser 612(March):209–215. https://doi.org/10.3354/meps12867

Wernberg T, Bennett S, Babcock RC, De Bettignies T, Cure K, Depczynski M, Dufois F et al (2016) Climate-driven regime shift of a temperate marine ecosystem. Science 353(6295):169–172. https://doi.org/10.1126/science.aad8745

Wicaksono P, Fauzan MA, Kumara ISW, Yogyantoro RN, Lazuardi W, Zhafarina Z (2019) Analysis of reflectance spectra of tropical seagrass species and their value for mapping using multispectral satellite images. Int J Rem Sens 40(23):8955–8978. https://doi.org/10.1080/01431161.2019.1624866

Widjaja S, Long T, Wirajuda H, Van As H, Bergh PE, Brett A, Copeland D, Fernandez M, Gusman A, Juwana S, Ruchimat T, Trent S, Wilcox C (2020) Illegal, unreported and unregulated fishing and associated drivers. World Resources Institute, Washington, DC. https://oceanpanel.org/blue-papers/iuu-fishing-and-associated-drivers

Wilkinson C, Salvat B, Eakin CM, Brathwaite A, Francini-Filho R, Webster N, Ferreira BP, Harris P (2016) Tropical and sub-tropical coral reefs. In: Iniss L, Simcock A (eds) The first global integrated marine assessment. World Ocean Assessment I, Division for Ocean Affairs and Law of the Sea, New York, p 42

Williams GJ, Graham NAJ, Jouffray JB, Norström AV, Nyström M, Gove JM, Heenan A, Wedding LM (2019) Coral reef ecology in the anthropocene. Funct Ecol 33(6):1014–1022. https://doi.org/10.1111/1365-2435.13290

Williamson M (1997) Marine biodiversity in its global context. In: Ormond RFG, Gage JD, Angel MV (eds) Marine biodiversity: patterns and processes. Cambridge University Press, Cambridge, pp 1–17

Wilson SK, Graham NAJ, Pratchett MS, Jones GP, Polunin NVC (2006) Multiple disturbances and the global degradation of coral reefs: are reef fishes at risk or resilient? Glob Chang Biol 12(11):2220–2234. https://doi.org/10.1111/j.1365-2486.2006.01252.x

Winnard S, Hochberg T, Miller N, Kroodsma D, Small C, Augustyn P (2018) A new method using AIS data to obtain independent compliance data to determine mitigation use at sea. In: Paper prepared for the Thirteenth Meeting of the CCSBT Compliance Committee, October 11–13, Noumea, New Caledonia. https://www.ccsbt.org/en/system/files/CC13_Info03_BLI_AIS_MitigationUse_Rev1.pdf

WMO (World Meteorological Organization) (2016) The global observing system for climate: implementation needs. WMO, Geneva. https://library.wmo.int/opac/doc_num.php?explnum_id=3417

Woodall LC, Robinson LF, Rogers AD, Narayanaswamy BE, Paterson GLJ (2015) Deep-sea litter: a comparison of seamounts, banks and a ridge in the Atlantic and Indian Oceans reveals both environmental and anthropogenic factors impact accumulation and composition. Front Mar Sci 2(February):3. https://doi.org/10.3389/fmars.2015.00003

Woolley SNC, Tittensor DP, Dunstan PK, Guillera-Arroita G, Lahoz-Monfort JJ, Wintle BA, Worm B, O'Hara TD (2016) Deep-sea diversity patterns are shaped by energy availability. Nature 533(7603):393–396. https://doi.org/10.1038/nature17937

World Bank (2017) The Sunken Billions revisited: progress and challenges in global marine fisheries. World Bank, Washington, DC. https://openknowledge.worldbank.org/handle/10986/24056

Worm B, Myers RA (2003) Meta-analysis of Cod–Shrimp interactions reveals top-down control in oceanic food webs. Ecology 84(1):162–173. https://doi.org/10.1890/0012-9658(2003)084[0162:MAOCSI]2.0.CO;2

Worm B, Tittensor DP (2011) Range contraction in large pelagic predators. Proc Natl Acad Sci U S A 108(29):11942–11947. https://doi.org/10.1073/pnas.1102353108

Worm B, Tittensor DP (2018) A theory of global biodiversity. In: Monographs in Population Biology 60. Princeton University Press, Princeton, NJ

Worm B, Barbier EB, Beaumont N, Duffy E, Folke C, Halpern BS, Jackson JBC et al (2006) Impacts of biodiversity loss on ocean ecosystem services. Science 314(5800):787–790. https://doi.org/10.1126/science.1132294

WoRMS (World Register of Marine Species) (n.d.) Statistics. http://www.marinespecies.org/aphia.php?p=stats. Accessed 22 Mar 2020

Wright G, Ardron J, Gjerde K, Currie D, Rochette J (2015) Advancing marine biodiversity protection through regional fisheries management: a review of bottom fisheries closures in areas beyond national jurisdiction. Mar Policy 61(November):134–148. https://doi.org/10.1016/j.marpol.2015.06.030

WWF (World Wildlife Fund) (2015) Reviving the ocean economy: the case for action—2015. WWF International, Gland. https://c402277.ssl.cf1.rackcdn.com/publications/790/files/original/Reviving_Ocean_Economy_REPORT_low_res.pdf?1429717323

Ye Y, Gutierrez NL (2017) Ending fishery overexploitation by expanding from local successes to globalized solutions. Nat Ecol Evol 1(June):0179. https://doi.org/10.1038/s41559-017-0179

Yesson C, Clark MR, Taylor M, Rogers AD (2011) The global distribution of seamounts based on 30-second bathymetry data. Deep-Sea Res I Oceanogr Res Pap 58(4):442–453. https://doi.org/10.1016/j.dsr.2011.02.004

Zaneveld JR, Burkepile DE, Shantz AA, Pritchard CE, McMinds R, Payet JP, Welsh R et al (2016) Overfishing and nutrient pollution interact with temperature to disrupt coral reefs down to microbial scales. Nat Commun 7(June):11833. https://doi.org/10.1038/ncomms11833

Zhang H, Wu F (2017) China's marine fishery and global ocean governance. Global Pol 8(2):216–226. https://doi.org/10.1111/1758-5899.12419

Zoffoli ML, Frouin R, Kampel M (2014) Water column correction for coral reef studies by remote sensing. Sensors 14(9):16881–16931. https://doi.org/10.3390/s140916881