SUSTAINABLE DEVELOPMENT GOAL 14 - LIFE BELOW WATER: TOWARDS A SUSTAINABLE OCEAN

EDITED BY: E. D. Lorenzo, C. Lønborg, J. H. Andersen, E. G. Escobar-Briones, M. J. Devlin, A. Borja, M. N. Müller, C. Robinson, A. Ford, A. M. Zivian, A. M. M. Sequeira, S. Villasante, B. W. Molony, T. Fortibuoni, S. Libralato and P. Failler

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SUSTAINABLE DEVELOPMENT GOAL 14 - LIFE BELOW WATER: TOWARDS A SUSTAINABLE OCEAN

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Tomaso Fortibuoni, Istituto Superiore per la Protezione e la Ricerca Ambientale (ISPRA), Italy
Simone Libralato, Istituto Nazionale di Oceanografia e di Geofisica Sperimentale (Italy), Italy
Pierre Failler, University of Portsmouth, United Kingdom

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Brett W. Molony[1]*, Alex T. Ford[2], Ana M. M. Sequeira[3], Angel Borja[4,5], Anna Milena Zivian[6], Carol Robinson[7], Christian Lønborg[8], Elva G. Escobar-Briones[9], Emanuele Di Lorenzo[10], Jesper H. Andersen[11], Marius N. Müller[12], Michelle J. Devlin[13], Pierre Failler[2], Sebastian Villasante[14], Simone Libralato[15] and Tomaso Fortibuoni[15,16]

[1] Oceans and Atmosphere, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Perth, WA, Australia, [2] Centre for Blue Governance, University of Portsmouth, Portsmouth, United Kingdom, [3] UWA Oceans Institute and School of Biological Sciences, The University of Western Australia, Crawley, WA, Australia, [4] AZTI, Marine Research, Basque Research and Technology Alliance (BRTA), Herrera Kaia, Portualdea z/g, Pasaia, Spain, [5] Faculty of Marine Sciences, King Abdulaziz University, Jeddah, Saudi Arabia, [6] Ocean Conservancy, Washington, DC, United States, [7] School of Environmental Sciences, University of East Anglia, Norwich, United Kingdom, [8] Aarhus University, Aarhus, Denmark, [9] National Autonomous University of Mexico, México City, Mexico, [10] Georgia Institute of Technology, Atlanta, GA, United States, [11] NIVA Denmark Water Research, Copenhagen, Denmark, [12] Department of Oceanography, Federal University of Pernambuco, Recife, Brazil, [13] Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Lowestoft, United Kingdom, [14] Department of Applied Economics, University of Santiago de Compostela, Santiago de Compostela, Spain, [15] Department of Oceanography, National Institute of Oceanography and Applied Geophysics (Istituto Nazionale di Oceanografia e di Geofisica Sperimentale) OGS, Trieste, Italy, [16] Italian Institute for Environmental Protection and Research (Istituto Superiore per la Protezione e la Ricerca Ambientale) - ISPRA, Ozzano dell'Emilia, Italy

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Edited by:
Donald F. Boesch, University of Maryland Center for Environmental Science (UMCES), United States

****Correspondence:***
Brett W. Molony
brett.molony@csiro.au

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Editorial on the Research Topic

Sustainable Development Goal 14 - Life Below Water: Towards a Sustainable Ocean

United Nations (UN) Sustainable Development Goal (SDG) 14 – Life Below Water – is arguably one of the most challenging of the 17 goals (United Nations, 2016) due to the immense scale of the Ocean (almost three-quarters of the planet's surface) and the direct links to many other SDGs. For example, No Poverty (SDG 1), Zero Hunger (SDG2) and Good Health and Well-Being (SDG 3) all rely on sustainable Life Below Water (SDG 14). In turn, Climate Action (SDG 13) is needed to achieve SDG 14, and the Ocean is essential in achieving SDG 13. There is much that we still do not know; indeed, the Ocean represents more than 99% of the space where organisms can live, yet more than 80% of the Ocean remains unexplored, especially the deep-sea.

The launch of the UN Decade of Ocean Science for Sustainable Development (2021–2030) aims at catalyzing a global focus to advance SDG 14 (Borja et al., 2020a). This will enhance the co-design of knowledge and actions for transformative ocean solutions, to address the challenges of a growing human population and climate change. Human pressures on the Ocean are important – 37% of the human population live in the coast from small villages to megacities exceeding 10 million people (e.g., New York, Shanghai, Lagos) and use the Ocean for a huge range of inputs, outputs and services, including amenity, food, transport, cooling water and waste disposal, as well as traditional and cultural uses. Many of these ecosystem services are undervalued, being conservatively estimated at $12.6 Trillion annually more than 20 years ago (Costanza et al., 1997). This is without considering two of the most severely undervalued services provided by the Ocean, as heat and carbon sinks, that have buffered many of the negative impacts of climate change. Many anthropogenic activities are leaving significant, direct and measurable global footprints in the Ocean with high profile examples including fishing[1,2,3], shipping lanes (Liu et al., 2019; Pirotta et al., 2019), dredging[4], plastic pollution (Hardesty et al., 2017; Barrett et al., 2020), noise pollution (Di Franco et al., 2020; Chahouri et al., 2021; Duarte et al., 2021), and changes in Ocean chemistry[5].

Human populations rely directly on the Ocean for food and other commercial activities, but a growing body of research has identified our dependency on the Ocean for health and well-being (Borja et al., 2020b). Other ecosystem services provided by the Ocean are also yet to be properly considered. These include the cultural and spiritual services provided by the Ocean (Brown and Hausner, 2017; de Juan et al., 2021), which have developed over millennia of human relationships with the Ocean and represent knowledge and connections that extend beyond monetary value. Aiming to integrate this knowledge in scientific endeavours, many indigenous peoples are bringing their traditional science and knowledge to partner with western science (Mazzocchi, 2006) and provide a more in-depth and long-term understanding of the Ocean, especially in coastal areas (Mustonen et al., 2021).

While the challenges are clear and sometimes seem overwhelming, approaches and solutions are being actively developed and tested; several of these are explored in this Research Topic.

With more than three billion people who rely on fish for at least 20% of their daily protein, and more than 120 million directly employed in the fishing and aquaculture sectors[6], sustainable fishing (Penca; Fiorentino and Vitale; Jaiteh et al.) and aquaculture (Azra et al.) were a natural focus of several papers. This included a call for reducing effort in mixed species fisheries, and therefore fishing mortality, to take into account the differing and lower productivity of some species and the risk to their sustainability (Newman et al., 2018), and adopt a quota system based on "pretty good yield" (Hilborn, 2010).

Others emphasized the need for better conservation planning and coordination (Katsanevakis et al.; Ceccarelli et al.; Herrera et al.) as well as integration of their cultural and spiritual values into wider society (Baker et al.). This includes the need to improve spatial management, providing specific approaches to minimize human impacts and risks to charismatic megafauna. This management approach could be applied to whale watching activities, to support sustainable non-extractive human activities in the Ocean (Almunia et al.). The article by Adewumi et al., dealing with the Guinea Current Large Marine Ecosystem shared among Benin, Nigeria, and Cameroon, highlighted the challenges of international ocean governance, a result of political characteristics, the relics of colonialism, and increasing ocean use and pressure on marine ecosystems and services. The administrative and political arrangements differ significantly among countries, complicating transnational collaboration. The review of these arrangements revealed varying levels of convergence at international, regional and national levels, and could be a model to assist regional fishery management organizations to support positive steps toward ocean sustainability (Juan-Jordá et al., 2018).

Future risks to the Ocean (Garcia-Soto et al.), including those imposed by climate change (Green et al.), and the tools (Mariani et al.), approaches (e.g., Endrédi et al.; Hsu et al.), and ways to monitor this complex system (Jones et al.), including biodiversity (Herrera et al.), highlighted the extraordinary and diverse values of the Ocean and challenges (**Figure 1**). Embracing modern technologies (Almunia et al.; Green et al.), including the Internet of Things (Mariani et al.), could also promote and support a harmonization of ocean monitoring among all nations, and support international initiatives and cooperation[7], including platforms to involve the wider community[8].

The social dimension (Haward and Haas) will also be critical as a way of valuing and engaging with direct and indirect stakeholders of the Ocean and in developing better policies for governance (Paredes-Coral et al.; Polejack; Adewumi et al.; Kirkfeldt and Frazão Santos; Archana and Baker; Rohmana et al.). This is especially true at the land-sea interface (Singh et al.) where human populations concentrate and the risks from a changing climate are directly evident, with projected sea level rise (Nicholls and Cazenave, 2010; Hooijer and Vernimmen, 2021), and more frequent and intense storms (Pugatch, 2019; Chen et al., 2020). It is also true for the deep ocean (Howell et al.), which remains largely unexplored. The socio-ecological connections described in this Research Topic of *Frontiers in Marine Science* provide frameworks and hope for a sustainable future for the coasts and ocean.

While this *Frontiers in Marine Science* Research Topic does not represent all initiatives underway globally to address SDG 14, it provides a glimpse of some of the diverse approaches and intellectual capital invested in ocean sustainability. While the goal focuses on Life Below Water, these approaches directly support many other SDGs, which arguably cannot be achieved without a healthy and sustainable ocean (Mustonen et al., 2021).

We hope that other initiatives currently underway will assist in not only highlighting the links between SDG 14 and other SDGs but also provide a way for synergies among disparate knowledge domains to support transdisciplinary and multi-sectoral approaches for good policy development. As examples, we note the significant initiatives around the globe in areas of blue carbon and an equitable "blue economy." Blue carbon projects not only protect and restore seagrass, mangrove, salt marsh, and macrophytes, but also support the associated biodiversity and human livelihoods that depend on these critical habitat-forming species. "Working with nature approaches" including in the restoration of corals, seagrasses, seaweeds, and mangroves are underway around the globe, with new methods being developed and tested [e.g., genetic techniques to identify more heat tolerant species of coral (Buerger et al., 2020) and other marine habitat building species (Alsuwaiyan et al., 2021)].

[1]https://globalfishingwatch.org/.
[2]http://www.seaaroundus.org/.
[3]https://www.minderoo.org/global-fishing-index/.
[4]https://wamsi.org.au/wp-content/uploads/bsk-pdf-manager/2019/10/Dredging-Science-Synthesis-Report-A-Synthesis-of-Research-2012-2018-April-2019.pdf.
[5]https://www.science.org.au/curious/earth-environment/ocean-acidification.
[6]https://www.fao.org/in-action/eaf-nansen/news-events/detail-events/en/c/1413988/.
[7]https://www.geoaquawatch.org/.
[8]https://research.csiro.au/eyeonwater/.

FIGURE 1 | Word cloud generated from the key words from the SDG 14 papers contributed toward this *Frontiers in Marine Science* Research Topic [Generated through WordArt.com - Word Cloud Art Creator]. Some key words were truncated to simplify the generation of the word cloud.

The efforts in these areas will be underpinned by new methods of accounting—such as blue carbon, biodiversity, ecosystem services and a framework of ocean accounting which is currently being developed[9]. This approach embraces environmental, social and cultural accounting, in addition to economic accounting, to better assess and value entire marine areas and ecosystems and integrate a wide range of SDGs. Our hope is that this will support and enable clearer and better decisions by ocean and coastal management agencies. These decisions should be based on a number of decision support tools, including: (i) management strategy evaluation approaches, (ii) scenario testing including assessing a range of alternative approaches, and (iii) potentially creating digital twins to test and explore management decisions before ocean activities commence.

We look forward to making the difficult possible and contributing to a vibrant, thriving future throughout the UN Decade of Ocean Science for Sustainable Development and the UN Decade of Restoration (Waltham et al., 2020) based on some of the cutting-edge approaches detailed in this Research Topic of *Frontiers in Marine Science*.

[9]https://www.oceanaccounts.org/.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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Increased Transparency and Resource Prioritization for the Management of Pollutants From Wastewater Treatment Plants: A National Perspective From Australia

Qurratu A'Yunin Rohmana[1,2], Andrew M. Fischer[1], John Cumming[2,3], Boyd D. Blackwell[1,4] and John Gemmill[2]*

[1] Institute of Marine and Antarctic Studies, University of Tasmania, Hobart, TAS, Australia, [2] Clean Ocean Foundation, Wonthaggi, VIC, Australia, [3] Infotech Research, South Yarra, VIC, Australia, [4] AquaEquis Consulting, Armidale, NSW, Australia

With increasing human populations in coastal regions, there is growing concern over the quality of wastewater treatment plant (WTP) discharge and its impacts on coastal biodiversity, recreational amenities, and human health. In Australia, the current system of WTP monitoring and reporting varies across states and jurisdictions leading to a lack of data transparency and accountability, leading to a reduced ability to comprehensively assess regional and national scale biodiversity impacts and health risks. The National Outfall Database (NOD) was developed to provide a centralized spatial data management system for sharing and communicating comprehensive, national-scale WTP pollutant data. This research describes the structure of the NOD and through self-organizing maps and principal component analysis, provides a comprehensive, national-scale analysis of WTP effluent. Such a broad understanding of the constituents and level of pollutants in coastal WTP effluent within a public database provides for improved transparency and accountability and an opportunity to evaluate health risks and develop national water quality standards.

Keywords: effluent, outfalls, pollutants, spatial data management, impacts, human health, environment

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Edited by:
Simone Libralato,
Istituto Nazionale di Oceanografia e di Geofisica Sperimentale (OGS), Italy

Reviewed by:
Rachel Gjelsvik Tiller,
SINTEF, Norway
Christian T. K. -H. Stadtlander,
Independent Researcher, St. Paul, United States

****Correspondence:***
Qurratu A'Yunin Rohmana
qurratu@utas.edu.au

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INTRODUCTION

With increasing human populations in coastal regions and an increase in extreme weather events due to climate change (Meehl et al., 2000), there is growing concern over the quality of wastewater treatment plant (WTP) discharge and the impacts of effluent on coastal biodiversity and human health (Schwarzenbach et al., 2010; Jagai et al., 2015). Land-based pollutants, from sewage and storm water runoff, enter the coastal marine environment through discharge points, typically from WTPs (Carey and Migliaccio, 2009; Mallin et al., 2009). This effluent significantly increases organic and inorganic nutrients and turbidity levels in receiving waters, which can cascade across several levels of ecological organization to change the key properties of benthos and fish communities (Roberts, 1996; Burd et al., 2012; Campbell et al., 2015; Yu et al., 2016; White et al., 2018). An increase in the level of pollutants can have an impact on coastal ecology and biodiversity and affect the health of recreational water users (Reopanichkul et al., 2009; Schwarzenbach et al., 2010; Zhao et al., 2015; Boehm et al., 2017). Often the economic sector, such as aquaculture industries, are also affected due to high levels of bacterial contamination, which decrease production during the harvest season (Campos et al., 2015).

There are many ways to resolve these issues, one of them is cooperation between governments, policy makers, scientists, and civil society through data transparency and information disclosure. The concept of data transparency has played an important role across most, if not all, disciplines (Friesike et al., 2015) and it has been shown that increased transparency leads to improved accountability of industrial (López and Fontaine, 2019), corporate (Auld and Gulbrandsen, 2014) and government agencies (Harrison and Sayogo, 2014). "Openness" and information disclosure has often been associated with not only economic prosperity, but also improvements to social capital and the environment (Koltay, 2016; Lee et al., 2019). While "transparency" has multiple definitions, as well as multiple purposes, targets, and justifications, the most common one is to resolve issues that a lack of information pose (Fung, 2013). The most suitable form of transparency is constitutional transparency through freedom of information (FOI). The FOI gives citizens the right to request and access government information not exempt under the FOI Act (European Parliament, 2001; OAIC, 2013; OIP, 2019). Other initiatives to increase transparency and accountability and ensure public access to information are through e-government programs (e.g., data.gov) (Pina et al., 2007; Lourenço, 2013). Access to information, under these programs is meant to facilitate organizational accountability toward environmental and public health obligations. More importantly, accurate data and transparent methods are needed for governments to make good policy decisions and for the general public to, for example, assess health risks and make informed decisions about sustainable use of the environment (Gupta, 2010; Friess and Webb, 2011). There is further evidence that improved governance through integrated forms of civil society—led meta-governance is related to information disclosure (Schleifer et al., 2019). Therefore, transparency may lead to well-informed environmental policy, which may play a critical role in anticipating the wider impacts water quality (Fezzi et al., 2015).

Many countries have established a legal framework to protect the health of aquatic and marine environments. Australia, for example, obligated to manage resources of national interest (matters of national environmental significance) and as a signatory to the Convention on Biological Diversity, is required to safeguard its biological diversity, as well as manage the impacts of nutrients on ecosystem function and structure [Aichi Biodiversity Targets (8)] (NRMMC, 2010). The state/territory governments have each established Environment Protection Authorities (EPAs). Each EPA acts as an independent environmental protection regulator to prevent and control pollutant impacts to human health and the environments. With regards to wastewater effluent, each state or territory EPA has a role in regulating WTP discharge. Any activity that may produce a discharge of waste that by reason of volume, location or composition adversely affects the quality of any segment of the environment will require a license from the Authority (DECC NSW, 2009). Throughout each state and territory, emission sources are required to monitor their discharges and to be in compliance with the conditions set out in their licenses. Each WTP is required to conduct monitoring within the vicinity of their outfalls, analyze the samples and report the results to the EPA (DECC NSW, 2009; EPA VIC., 2009).

Monitoring of WTP effluent is managed through license conditions determined together by the EPA and the water treatment authorities (WTA), the body that manages a WTP. License conditions ultimately depend on EPA requirements, WTP treatment level, and the condition of the marine environment (EPA NSW, 2003; EPA VIC, 2017). While WTP operators are largely interested in minimizing expense and staying within their license conditions, the EPA has an interest in regulating "developments and activities that may impact on environmental quality and to promote best practice, sustainable environmental management" (EPA NSW, 2013; EPA VIC, 2017). This system of WTP effluent monitoring and reporting varies across states, jurisdictions and regions. Inconsistency in monitoring requirements, confounded by non-binding international standards for assessing pollutant risk, and a lack of national-level standards for data collection, transmission and sharing, results in a lack of transparency and a reduced ability to comprehensively assess regional and national scale biodiversity impacts and health risks (ANZECC and ARMCANZ, 2000; Reichman et al., 2011; Borgman, 2012; Gemmill et al., 2019; Rohmana et al., 2019). This can hamper the ability to adequately assess progress toward biodiversity conservation targets (Bull et al., 2018).

Through the lens of transparency and accountability, a non-government organization, the Clean Ocean Foundation (COF), with the support of National Environmental Science Program (NESP), developed the National Outfall Database (NOD, 2020)[1]. This initiative provides a centralized spatial data management system for sharing and communicating comprehensive, national-scale pollutant data from outfalls. It provides the potential to empower coastal communities to monitor and evaluate health risks of the outfall effluent, and for federal and state government to prioritize outfall infrastructure reform. It promotes and supports transparency as well as openness of pollutant management from WTPs and accountability of these organizations against environmental and human health obligations. The NOD also provides a baseline of information to develop national-scale monitoring and wastewater re-use.

Data-centric, e-government portals, designed with the intention developing transparency and accountability in waste water management have been developed elsewhere. The European Union (EU) built the Water Information System for Europe (WISE) which consists detailed information of the EU water policies, reported dataset for both inland and marine water, modeling, and relevant research (European Environment Agency, 2017). The notion of having a water portal is also applied in the United States. The Water Quality Exchange (WQX) under Clean Water Act was created under the sponsorship of The United States Geological Survey (USGS), Environmental Protection Agency (EPA) and National Water Quality Monitoring Council (NWQMC) to integrate publicly available any water related data, including current and historical water data as well as water quality monitoring data (NWQMC, 2016).

While these are mostly government initiatives, this work describes the development of a data portal under the direction

[1] www.outfalls.info

of a non-governmental organization and outlines how the data can be used to increase transparency, accountability and guide policy development and the attainment of international goals and targets. Therefore, the purpose of this research is (1) to describe the structure of the NOD, (2) provide the first national-scale analysis of WTP effluent into the marine environment, and (3) examine the spatial patterns of water quality variables across Australia and interrelations among them. The description of the NOD and an analysis of its data is further discussed in the context of data transparency and government accountability with regards to outfall monitoring and reporting standards. The importance of this research is that it provides a comprehensive and transparent data platform to guide government accountability at a national scale through overcoming inconsistencies in data reporting methods.

MATERIALS AND METHODS

Data Collection – The National Outfall Database

The NOD, currently, provides a national inventory of Australia's 181 coastal outfalls including the volume of water and the amount of pollutants and nutrients disposed of into coastal receiving waters (**Figure 1**). Water quality data, recorded in the NOD, were collected from 42 WTAs around Australia. Sampling conducted by the WTAs were taken from the sampling points within the WTP premises as described in the licenses (EPA VIC., 2009; EPA NT, 2013; EPA SA, 2016). Data describing water quality parameters (**Table 1**) and outfall characteristics were transcribed into a database. Outfall characteristics consist of outfall name, manager, license number, WTP capacity, population serviced, treatment level, and location description.

In order to display the data spatially, the database was equipped with a location map, pivot table and trend chart for each parameter. The descriptive statistics function was applied to each outfall to calculate the mean and standard deviation of water quality parameters and the summary of discharge volume. To ensure proper storage and use of the data, the data collected from the public agencies are treated in accordance with the objectives of the *Freedom of Information Act 1999 (Cth)* and equivalent legislation in each jurisdiction, which require government agencies to make information publicly available, subject to certain exceptions listed in that legislation.

Sites that monitored a consistent set of water quality parameters (n = 162) were included in the analysis. These parameters include enterococci (ENTCC), fecal coliform (FC),

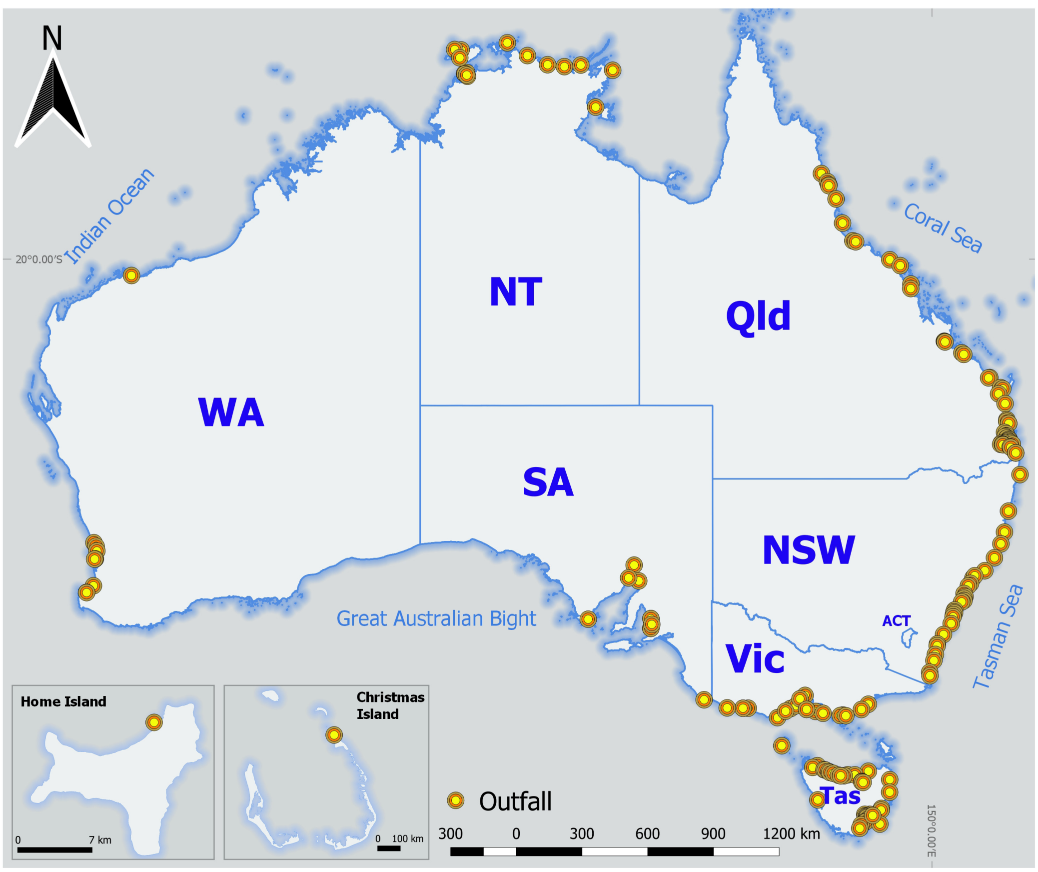

FIGURE 1 | Australian coastal outfalls recorded in the NOD.

TABLE 1 | Water quality parameters recorded in the database with their assigned database ID and units of measurement.

ID	Name	Unit
1	Total suspended solids*	mg/L
2	Total phosphorus*	mg/L
3	Total nitrogen*	mg/L
4	Oil and grease*	mg/L
5	Flow volume	ML
6	Fecal coliform*	org/100 mL
7	pH*	pH
8	BOD_{5-days}	mg/L
9	Ammonia nitrogen	mg/L
10	Enterococci*	org/100 mL
11	Total dissolved solids*	mg/L
13	*E. coli*	org/100 mL
14	Turbidity*	NTU
15	Color	Pt. Co. Units
16	Nitrate nitrogen	mg/L
17	Total kjeldahl nitrogen	mg/L
18	Surfactants (MBAS)	mg/L
19	Total coliforms	org/100 mL
20	Blue green algal bloom	Frequency
21	Chemical oxygen demand	mg/L
22	Total algae count	Cells/mL
23	Total blue-green algae count	Cells/mL
24	Electrical conductivity*	µS/cm
25	Calcium	mg/L
26	Magnesium	mg/L
27	Sodium absorption ratio	SAR
28	Sodium	mg/L

Asterix (*) indicates parameters used for the analysis.

electrical conductivity (EC), turbidity (NTU), oil and grease (OG), alkalinity or acidity (pH), total dissolved solids (TDS), total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS) (**Table 1**). The remaining parameters were not analyzed due to either a lack of samples or inconsistent sampling of parameters across sites.

Data Analysis

Due to the enormity and diversity of the dataset, unsupervised machine learning methods were utilized to explore and observe hidden patterns. Firstly, self-organizing maps (SOM) approach was used because its process allowed the categorization of pollutant characteristics to a set of outfall class structures, identifying similarities and differences in class factors that may influence effluent quality between outfall sites. Secondly, the principal component analysis (PCA) was applied in order to validate the patterns in the SOM plane visualization and to help identify patterns not detected in the SOM planes. Lastly, to determine outfall grouping and support our goal of increased transparency and the establishment of standards, a cluster analysis was conducted using a gap statistics and k-means. All data analyses were conducted using MATLAB 2019a software (MATLAB, 2019). The results were plotted to identify data clusters of, or similarities, between the discharge of WTPs across Australia.

Self-Organizing Maps

The SOM is an unsupervised classification method, in which networks learn to form their own classifications of the training data without external input. The SOM detects patterns or classes in a set of data, preserving the proximity to a set of classes or neighboring relations (Kohonen, 2001). In other words, similar clusters in the multidimensional space are located together on a 2D grid allowing intuitive visualization of classes (Bação and Lobo, 2010). The process includes a self-organizing neighborhood mechanism, the neighboring clusters of the winning reference vector in the 2D lattice space are also adapted toward the sample vector through an iterative training process. Through this approach, an input vector is presented to the multi-class network and the output is compared with the target vector, projecting the topological neighborhood relationships of the high dimensional data space of the class "lattice" (Kelvin, 2006). In order to visualize the neural network map (lattice), a suitable map size was first identified. The size determination is crucial for cluster clarification. If a map is too small, patterns between the nodes are less likely to resolve important detectable differences (Céréghino and Park, 2009; Vatanen et al., 2015). The initialization and training of the input data were performed to calculate the distance of every neuron in the network. Training steps also define the SOM map quality by examining the quantization error (QE) and topograhic error (TE). QE is the average distance between each node and its best matching unit (BMU), while TE measures the wellness of the map structure by calculating the node's first and second BMUs and their position in relation to each other (Villmann et al., 1997; Kohonen, 2001; Breard, 2017). Smaller QE and TE values indicate a better fit of the map itself (Kohonen, 2001; Breard, 2017). Once the SOM has been trained, the data was visualized into a U-matrix (unified distance matrix) along with eight component planes. The U-matrix shows the clustering structure of the SOM data by visualizing the distance between neighbors of the SOM, while the component planes represent the pattern and behavior of one parameter toward others (Kohonen, 2001; Bação and Lobo, 2010). The darker areas in the U-matrix represent shorter distances between nodes, which then forms clusters. Light areas represent longer distances, as well as borders for each cluster. In summary, similarity between clusters is measured as the minimum distance between data vectors and each node on the map (Vesanto et al., 2000). The analysis was conducted using SOM Toolbox version 2 for MATLAB by Vesanto et al. (2000).

Principal Component Analysis (PCA) and Cluster Analysis

Principal component analysis (PCA) is a technique to emphasize variation of patterns in a dataset and has been widely used across a variety of scientific areas (Abdi and Williams, 2010). It is a way of identifying patterns and expressing data to highlight similarities and differences. Since patterns can be hard to find in data of high dimensions, a PCA is a powerful tool for multivariate data analysis. PCA provides useful information to identify the relationship of similar characteristics by representing the spatial and temporal variations of wastewater parameters (Zuur et al., 2007; Abdi and Williams, 2010). Prior to deriving the principal components, Kaiser normalization and varimax rotation were used to weight the water quality samples equally and maximize the loading variances. Principal components, coefficients, scores and variance, were then calculated using the *eigs* function in MATLAB (Mathworks, UAS). The component coefficients represent pollutants loading while the scores represent WTP sites.

Cluster analysis was applied to the PCA score matrix using gap statistics and *k-means* methods. The gap statistics helps determine the optimal number of clusters (k) (Tibshirani et al., 2001), while the k-*means* procedure performs the grouping of water quality data according to the identified optimal *k* values (Hair et al., 2010). The k-*means* method is generally based on a proximity measure, meaning that it measures the distance and location of the mean samples and groups them accordingly (Hair et al., 2010; Jain, 2010; Härdle and Simar, 2015). Each k consists of samples that are close to each other. Due to its simplicity and quick response during the analysis for vast dataset (Hair et al., 2010), k-*means* was chosen as the method to analyze our data. The major challenge of k-*means* is determining the number of clusters (k) needed prior to conducting the analysis. In this case, a gap statistic method was used. The gap statistic evaluates the dataset and provides the highest possible number of clusters suitable for the analysis (Tibshirani et al., 2001; Brodinová et al., 2019). After the gap value was calculated, the accurate *k* value was, then, applied to the k-*means* method. The clustering results will be visualized along with the PCA plot.

RESULTS

Monitoring Data

Of the 181 outfalls, 114 are categorized as ocean outfalls, meaning that effluent is directly discharged into the coastal marine environment. Sixty-seven of them are categorized as estuary or river outfalls. Because effluent is discharged from these WTPs into estuaries or rivers, the dilution and transport of pollutants to the coastal environment is also dependent on other variables such as rainfall, river flow, and tides. The state of Queensland had the highest number of combined (coastal and estuary/river) outfalls at 51, followed by Tasmania (41) and New South Wales (34), Victoria (19), The Northern Territory (14), Western Australia (12), and South Australia (10).

Summary statistics of the assessed parameters are presented in **Table 2**. EC varied between 390 and 6,700 μS/cm, with an average of 1,937 μS/cm. ENTCC and FC tended to have a wide range of values with a mean of 29,945 and 647,153 org/100mL and standard deviation (SD) of 351,951 and 8,095,710 org/100 mL, respectively. OG values ranged from 0 to 312.6 mg/L and reported pH values ranged from 3.22 to 7.8. TDS ranged from 200 to 33,000 mg/L with a mean of 2,737 mg/L. The mean values of TN, TP, and TSS were 15.9, 4.8, and 24.5 mg/L, respectively. The mean concentration value of NTU was 35.7 mg/L. The range and standard deviations values across ENTCC, FC and TDS are extremely high.

SOM, Covariance, PCA, and Clustering

The TE and QE map size for self-organizing maps is critical for the implementation and visualization of the analysis. The QE and TE were computed at different map sizes. A map size of 10 × 7 with 70 nodes was the most appropriate with the QE value of 0.624 and TE 0.049 (**Table 3**).

The SOM component planes of each water quality parameter across the whole dataset is shown in **Figure 2**. It shows that all variables had a clear trend of color gradients for each parameter. The trends also reveal the correlation strength between parameters. The similar patterns of color gradients with the values increasing to the bottom left of the plane were shown on the ENTCC, FC, OG, TN, TP, TSS, and slightly on the EC. On the other hand, pH, TDS, and NTU had almost similar patterns of color gradients with lower values dominating the upper left areas and higher values on the bottom right of the map.

A covariance matrix was computed to corroborate these patterns (**Table 4**). Further, it suggests particularly strong correlations between EC and TP (r = 0.70), ENTCC and FC (r = 0.94), FC and OG (r = 0.62), OG and TSS (r = 0.74), and TN and TSS (r = 0.62). A moderate correlation also can be seen between ENTCC, OG (r = 0.47) and TSS (r = 0.53), FC and TSS (r = 0.57), and OG and TN (r = 0.53).

Prior conducting the PCA and cluster analysis, gap statistics were applied to determine the optimal values for *k-means* clusters

TABLE 2 | Summary statistics of assessed parameters.

Parameter	Unit	N	Min	Max	Mean	*SD*	*SE*
EC	μS/cm	156	390	6,700	1,937.4	1680.9	134.6
ENTCC	org/100 mL	2,228	0	10×10^6	29,945.4	351,950.9	7,471.4
FC	org/100 mL	2,860	0	240×10^6	647,152.8	8×10^6	151,566.8
OG	mg/L	2,492	0	312.6	4.6	16.1	0.3
pH	pH	3,882	3.22	12.9	7.5	0.6	0
TDS	mg/L	628	200	33,000	2,736.9	5,269.9	210.5
TN	mg/L	4,320	0	373	15.9	17.0	0.3
TP	mg/L	4,227	0	86	4.8	5.1	0.1
TSS	mg/L	4,463	0	1,692.5	24.5	60.2	0.9
NTU	NTU	369	0	336	35.7	62.7	3.3

TABLE 3 | QE and TE values for deciding the optimal map size.

Map size	10 × 6	9 × 7	8 × 8	13 × 5	11 × 6	10 × 7
QE	0.668	0.654	0.663	0.631	0.619	**0.624**
TE	0.049	0.043	0.049	0.049	0.062	**0.049**

In bold is the optimal map size.

(Tibshirani et al., 2001). The result shows that a cluster of five is suitable for this research with the gap value of 0.49 (**Figure 3**). Results of the PCA and cluster analysis are shown in **Figure 4**. The first principal component, explaining 65% of the variance on the horizontal axis, has positive coefficients (right) for six parameters and slightly negative (left) for pH. TSS, OG, ENTCC, and FC have a strong influence toward PC 1. The second principal component, explaining 21% of the variance on the vertical axis, has positive coefficients (top) for six parameters, especially EC, and negative coefficients for OG, ENTCC and FC (bottom). The clusters appear to separate outfalls with extreme levels of pollutant concentrations. On the lower right, cluster two and portions of cluster five have high correlation with increased inputs of OG, ENTCC, and FC. The top right quadrant contains the majority of cluster three, four and few from cluster five. These represent high contributions of EC, TP, TN, and TSS. On the top left, pH is the only parameter which seems to be related to some outfall sites in cluster one and three.

Not surprisingly, the cluster analysis results suggest that each outfall site did not group according to its state or territory, instead state and territory representation was spread over five clusters (**Figure 5** and **Table 5**). Tasmania was the most diverse state with sites in four out of five clusters. The second most diverse was New South Wales and Western Australia outfall sites across three clusters. Northern Territory outfall sites were grouped into two clusters only. Cluster 3 is the only group that consists all states/territory (**Table 5**).

Cluster two consists two Tasmanian outfalls (Pardoe and Ulverstone), which discharge some of the highest FC and ENTCC values in the nation. Cluster four represents outfalls (Berrimah, Leanyer Sanderson, Palmerston, Port Pirie and Bolivar WTP) across the Northern Territory and South Australia, which are responsible for contributing higher concentrations of EC, TP, and NTU. Cluster five contains three of the largest outfalls in the nation, located in Sydney (Bondi, Malabar and North Head), Electrona (Tasmania) and Point Peron (Western Australia). These outfalls are responsible for contributing higher concentrations of OG, TN, and TSS.

DISCUSSION

Water quality parameters were collected from 162 outfalls around Australia. The highest level of oil and grease were associated with highly urbanized and industrial areas, as were organic pollutants (nitrogen and phosphorous). Higher levels of N and P were not always associated with agricultural regions (Xia et al., 2016; Tromboni and Dodds, 2017; Rohmana et al., 2019), as other studies have found (Booth, 2015; Clendenon and Atkins, 2016). Clusters identify outfalls that share high fecal coliform and enterococci and discharge high levels of oil and grease and nutrients (e.g., cluster 2 and cluster 5, **Figure 5**). General patterns showed a strong correlation between enterococci

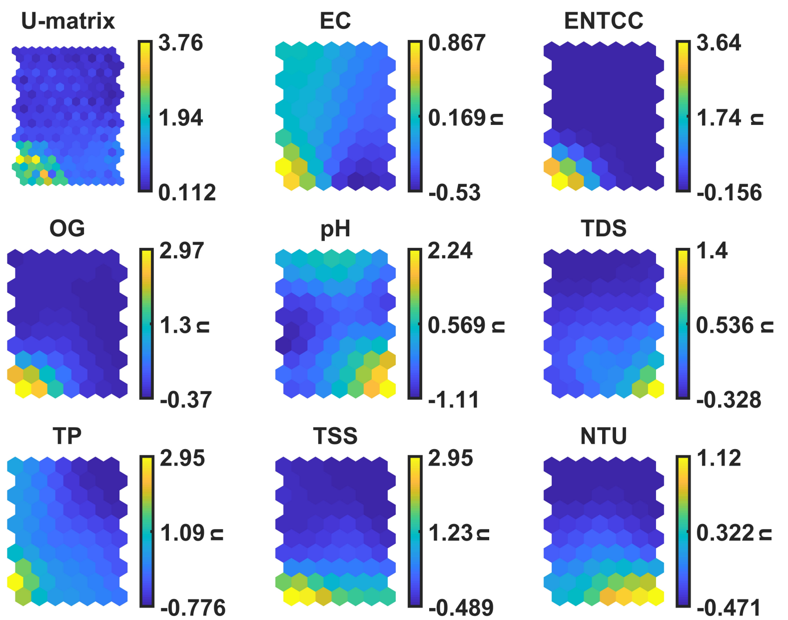

FIGURE 2 | SOM component planes of all parameters, electrical conductivity (EC), enterococci (ENTCC), fecal coliform (FC), oil and grease (OG), pH, total dissolved solids (TDS), total nitrogen (TN), total phosphorus (TP), total suspended solids (TSS), and turbidity (NTU). Dark color indicates low values and light color means high values. n indicates that the data was normalized.

TABLE 4 | Covariance matrix of water quality parameters (r).

	EC	ENTCC	FC	OG	pH	TDS	TN	TP	TSS	NTU
EC	1	−0.02	−0.02	−0.05	0.18	−0.03	0.02	**0.70**	0.04	0.07
ENTCC		1	**0.94**	**0.47**	−0.10	−0.02	0.23	0.08	***0.53***	−0.01
FC			1	**0.62**	−0.13	−0.02	0.22	0.08	***0.57***	−0.02
OG				1	−0.30	−0.05	***0.53***	0.13	***0.74***	−0.05
pH					1	0.24	−0.16	0.08	0.07	0.25
TDS						1	0.00	−0.02	0.05	−0.01
TN							1	0.40	**0.62**	0.10
TP								1	0.26	0.02
TSS									1	0.34
NTU										1

−, negative correlation. Bold indicates a strong correlation. Italic numbers are the matching variable that showed differences between the covariance matrix and the PCA plot.

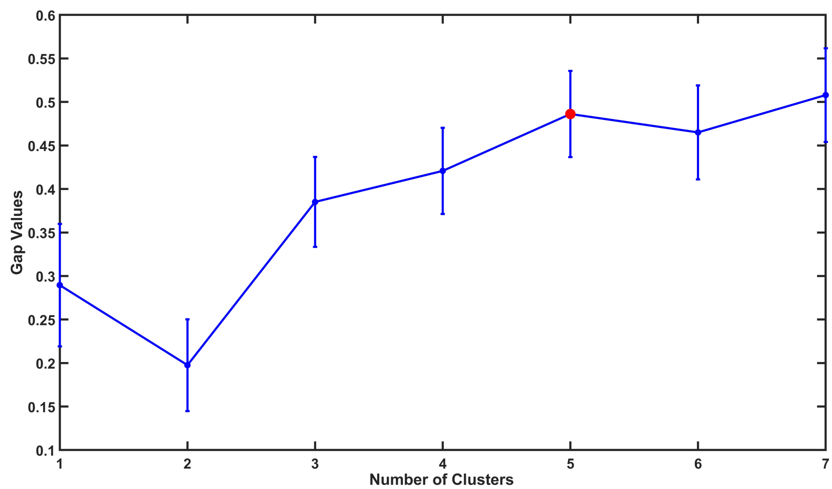

FIGURE 3 | Gap statistics for determining optimal value of clusters. Red dot represents the optimal clusters for *k-means* analysis.

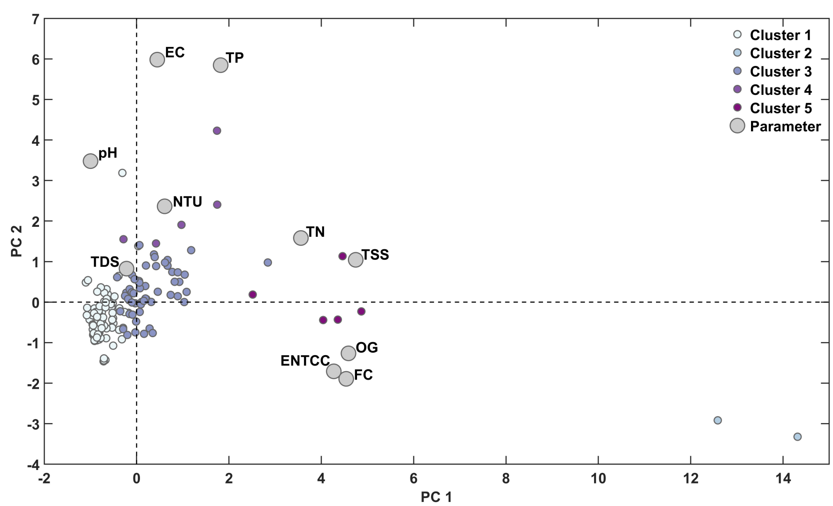

FIGURE 4 | PCA scores in five clusters. PC1 explains 65% and PC2 21% of the variance. Coefficients or parameters illustrate how each pollutant variable contribute to the two principal components.

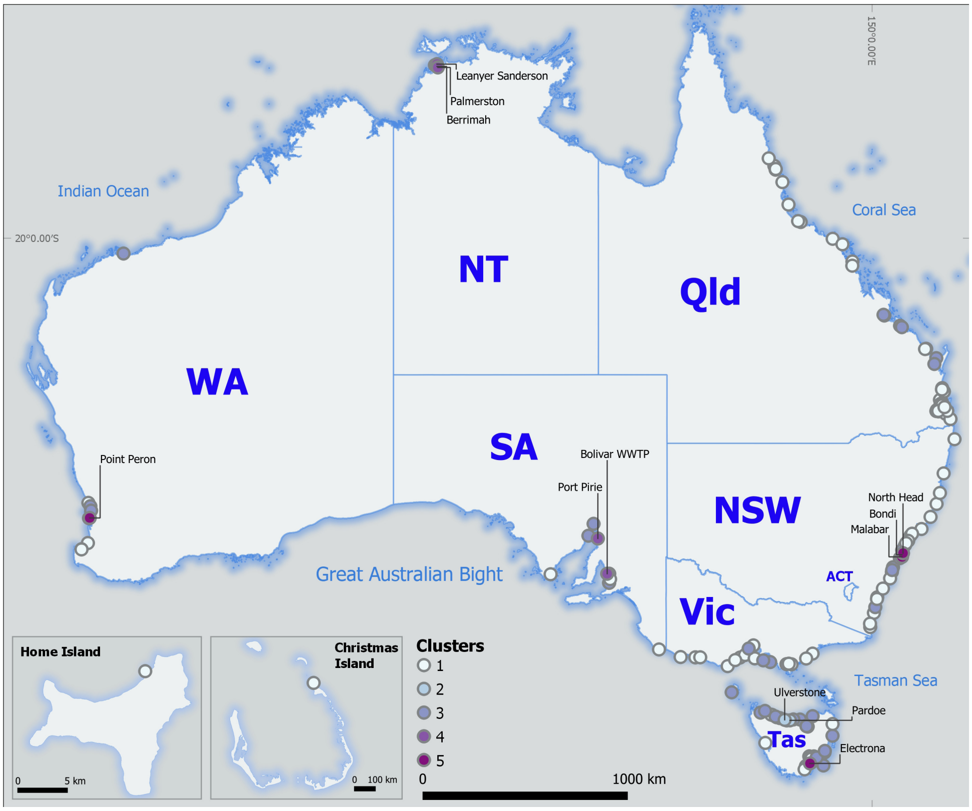

FIGURE 5 | Cluster distribution of Australian coastal outfalls.

TABLE 5 | Australian state and territory distribution over five clusters along with the number of outfalls sites in each group.

Cluster	State/Territory (N)
1	New South Wales (19), Queensland (41), South Australia (5), Tasmania (10), Victoria (14), and Western Australia (7)
2	Tasmania (2)
3	New South Wales (5), Northern Territory (1), Queensland (8), South Australia (3), Tasmania (28), Victoria (5), and Western Australia (4)
4	Northern Territory (3) and South Australia (2)
5	New South Wales (3), Tasmania (1), and Western Australia (1)

and fecal coliform, and total nitrogen, total phosphorus, and total suspended solids (**Figures 2**, **4** and **Table 4**). These correlations were confirmed by the SOM planes, PCA, and the covariance matrix.

Enterococci and fecal coliform are the most common bacterial indicator for monitoring water quality. In order to decrease human health risk, these bacteria have been widely used to detect pathogens in marine and aquatic environments (Havelaar et al., 1986; Efstratiou, 2019). Fecal bacteria, commonly, appear within the wastewater (Ashbolt et al., 2001) and high concentrations of fecal coliform and enterococci occurs when treatment plants have inadequate processes to remove bacteria from effluent (US EPA, 1998; Adams et al., 2008). The most noticeable level of high fecal bacteria counts in cluster two were represented by two Tasmanian outfalls (Pardoe and Ulverstone) (**Figures 4**, **5**). According to the Tasmanian emission limit guidelines for existing sewage treatment plants, fecal coliform must not exceed 1000 org/100 mL (DPIPWE, 2001). However, in this case, these two outfalls exceeded the acceptable limits and have consistently discharged high counts of fecal bacteria since 2015 (NOD, 2019a,b). The permit conditions showed that Pardoe WTP does not have any bacterial limits, although it is required to monitor levels. Ulverstone WTP has a maximum limit of 2000 org/100 mL for fecal coliform in the effluent quality. Currently, the EPA Tasmania and TasWater are in the process of upgrading Pardoe and Ulverstone WTP infrastructures (Infrastructure Tasmania, 2019; OTTER, 2019) to improve the levels of bacterial contamination in these coastal waters.

There are several outfall sites that cluster around the PCA coefficients of total phosphorus, total nitrogen and total suspended solids (**Figure 4**). These sites consist of outfalls in cluster three (Port Sorell, TAS), cluster four (Berrimah, NT) and cluster five (Electrona, TAS and Point Peron, WA). Those two Tasmanian outfalls tend to have slightly high total phosphorus, total nitrogen and total suspended solids, respectively, compared to the Tasmanian acceptable emission limits (DPIPWE, 2001). These elevated levels are likely due to the license conditions of each outfall. Port Sorell STP is not required to monitor the TP, TN, and TSS, while Electrona STP has maximum limit of 12, 65, and 186 mg/L for TP, TN and TSS, respectively. Despite discharging slightly higher concentrations of total nitrogen, Point Peron WTP (WA) did not exceed the 230 mg/L acceptable limit of Western Australian Environmental Quality Guidelines (ANZECC and ARMCANZ, 2000). There is no clear explanation where Berrimah WTP discharge point has a limit on these three parameters on their license, but it has four reported monitoring points limit which all says >30 μg/L (TP), >300 μg/L (TN), and >15 μg/L (TSS) (EPA NT, 2018). However, persistent elevated concentrations of nutrients will eventually affect the marine environment. The cumulative impact of nutrients and suspended solids might alter the original composition of marine environment by increasing eutrophication, algal growth, and reduce the light penetration into the waterbody which may impact marine biodiversity (ANZECC and ARMCANZ, 2000; Thompson and Waite, 2003; Beck and Birch, 2011; Clendenon and Atkins, 2016; Weerasekara et al., 2016).

Previous studies focused mainly on particular cases in certain areas (Burridge and Bidwell, 2002; Thompson and Waite, 2003; Adams et al., 2008; Manning et al., 2019) rather than across a national scale, as in this study. Considering the increasing availability of vast datasets at both national and regional scales, decision makers have the capability to conduct more comprehensive analyses to assist them in allocating limited resources to minimize the impact of wastewater on the environment (Edgar et al., 2016). There are strong relationships between ecological patterns and environmental covariates that emerge clearly when regional-to-global-scale data are considered (Kerr et al., 2007; Webb et al., 2009; Mora et al., 2011). As an unprecedented collection of nationwide water quality data, the NOD acts as a tool to facilitate cross institutional coordination across federal, state, and local authorities to integrate infrastructure planning and decision making of wastewater effluent from coastal outfalls (Marine Biodiversity Hub, 2015).

Currently, each state/territory EPA produces separate regulations and permits for WTPs to monitor their wastewater quality (EPA NSW, 2015; EPA VIC, 2017; OTTER, 2019), prohibiting a comprehensive analysis of national scale impacts. A comprehensive understanding of the constituents and level of pollutants in coastal WTP effluent within a public database provides an opportunity to apply the best possible knowledge to inform decisions in complex transboundary marine ecosystems, such as Australia. The existence of the NOD raises awareness at various scales – local, regional, state, national and international – of the extent of our wastewater effluent and provides essential information to assess the impacts on receiving waters. The ultimate outcome could be an improvement in the management of coastal biodiversity and assist agencies with their obligations to inform citizens of recreational health risks. Important environmental and human health implications are suggested by our research findings. Across Australia, discharge pollution limits appear to be set in a piecemeal and inconsistent manner, with limits being elevated where a plant simply cannot perform. This is not an appropriate way for regulating pollution discharge to limit the risks to environmental and human health and research helps to guide a more consistent and comprehensive approach to ensuring environmental and human health.

Added to this diversity of wastewater levels of quality of treatment and disposal across Australia, it can be seen that the number and type of outfalls vary considerably by jurisdiction and location. For example, Tasmania has one of the highest number of outfalls per capita, while many of the states have poorly performing outfalls. As Blackwell and Gemmill (2019) state, "these can be explained in part by the number of outfalls and size and geographical spread of relevant local populations that will benefit from the upgrades but also by the individual jurisdictional asset condition and their respective histories, evolutions and success with water and wastewater reform." In cases such as Tasmania, where historical legacies prevail and upgrades present a net cost to society, some form of subsidization by the nation will be required to ensure that no single state falls behind others (Blackwell and Gemmill, 2019). Blackwell and Gemmill (2019) found that overall enough states gain a net benefit from upgrades to compensate those states or territories that incur a net loss and remain better-off and this presents a prima face case for some form of subsidization.

National Outfall Standards for Monitoring and Reporting

Currently, state and territory EPAs determine the monitoring parameter standards and reporting requirements. The inconsistency of reporting requirement has resulted in an ambiguous process of governing water quality in Australia's coastal environment. This leads to a lack of clarity when assessing progress against international environmental goals and targets. Transparency equals good governance and elements of good governance include a clear enforceable reporting framework. For this reason, the Clean Ocean Foundation, using the data from the NOD, has started to establish the Standards and Guidelines for National Reporting of Outfall Data. National standards will provide a further legal directive to reduce WTP effluent impacts to the marine environment and improve health outcomes for recreational users and enhance business output (European Commission, 2017; European Commission, 2019; World Bank, 2018). This national standard is designed to redefine parameters, monitoring methods and reporting requirements in an effort to expand Australia's efforts in enhancing biodiversity protection and achieving Sustainable Development Goal 14 and in promoting data transparency and accountability of WTPs.

Many countries have already implemented national wastewater standards in order to effectively govern their aquatic and marine environments. Many of these standards were based on the provision of large amount of water quality data across a variety of environmental conditions. For example, the U.S. Environmental Protection Agency developed a portal (WQX) to store and manage water quality data for freshwater, groundwater and coastal areas (Read et al., 2017). This portal provides a centralized data repository for WTP monitoring allowing for the centralized analysis, reporting and display of water quality across the United States. Similar to the NOD, the portal has a standardized format of data upload, presentation, analysis, and mapping. The WQX portal was established primarily to reduce inconsistency between data sets, reduce workloads, and increase time efficiency associated with data management (Read et al., 2017). While, some data from the WQX are commercialized or available upon request (Read et al., 2017), the NOD enables free access to wastewater quality not only in Australia, but also other countries. This will allow for comparisons across countries and regions in establishing the attainment of targets under SDGs. The NOD water quality data can also be evaluated along with biodiversity surveys allowing for the prioritization of biodiversity conservation, an obligation under the Convention on Biological Diversity.

The European Commission (EC) has developed the Urban Waste Water Treatment Directive (UWWTD) (91/271/EEC) in 1991 (European Commission, 1991, 2019). The Directive is directly related to Water Framework Directive (WFD) 2000/60/EC and Environmental Quality Standards Directive (EQSD) 2013/39/EU, for setting up the water quality parameter concentration limits. It lays down four main obligations, planning, regulation, monitoring and reporting. The UWWTD has helped these countries successfully to reuse the water and maintain the cleanliness of the rivers by having high rates (85%) of recycled water (European Commission, 2019; Pistocchi et al., 2019). The EC invested approximately EUR 25 million each year for the UWWTD framework development, implementation, wastewater infrastructure, drinking water supply, and water conservation (European Commission, 2017). Similar to the United States and European Union, having a standard reporting in Australia may help the relevant stakeholders including citizen science to promote healthy marine environment initiatives and play an active governance role in developing national reporting standards. In terms of accountability, the WTAs would be able to commit to fulfill their obligations toward the general public for improved management of the environment.

The research presented here is anticipated to provide support for the development the Standards and Guidelines for National Reporting of Outfall Data in Australia. Two salient points can be made about these standards. Firstly, the results showed a strong correlation between various parameters (**Figures 2**, **4** and **Table 4**). Understanding these relationships overtime may be useful for predicting the future patterns of water quality, which can also help to redefine and reduce parameter limits (Shakhari et al., 2019). Secondly, the outfall clusters will definitely help water resource managers to discover where pollution problems exist, where to focus pollution control and where water quality improvements have been made (**Figure 5**). Once a national standard is established, redefined monitoring and reporting may help these sites assess water quality and biodiversity impacts in achieving SDG goals. Furthermore, the improved effluent from the WTP might be fully recycled as both potable and non-potable use, which may increase the Australian water supply (NRMMC, and EPHCA, 2006). With the NOD, a centralized database would be a dynamic resource for sharing and communicating comprehensive, national scale pollutant data from outfalls, which may help to reduce the outfall emission into the environment and supporting the sustainability of Australian marine environment biodiversity (Rohmana et al., 2018).

CONCLUSION

The governance and performance of WTPs in Australia sits in stark contrast to the frameworks that have been established in other parts of the world. The NOD provides a comprehensive database for making outfall monitoring data easily accessible and transparent by allowing for the investigation of the general patterns of the effluent quality across Australian coastal outfalls. This research has revealed two key points, which will aid decision makers in prioritizing water pollution governance across Australian waters. Firstly, the correlation patterns of various water quality parameters support the need to redefine and reduce concentrations limit which drive water recycling. Secondly, it helps decision makers to prioritize actions to reduce water pollution and improve environmental and human health outcomes and reduce health risks. The NOD, as with other e-government data initiatives, attempts to provide accessible and transparent data to not only address international environmental obligations, but to also develop sense of transparency and accountability for Australia's WTP stakeholders. The NOD, an NGO led initiative, was developed as a form of social reporting to help not only achieve stakeholder accountability but to guide the development of National Outfall Reporting Standards in a consistent and collaborative fashion across all Australian jurisdictions. The NOD acts as a third party between WTAs and other stakeholders and provides a neutral platform for unbiased decision making, improving governance and promoting accountability. As the National Outfall Reporting Standards is currently being developed, it is recommended that future research focuses on evaluating its implementation and furthering its potential toward advancing public accountability and improved environmental and economic outcomes. Additional research should focus on developing NOD capacity for handling those pollutants that were inconsistently measured across outfall sites and to take into consideration emerging pollutants.

DATA AVAILABILITY STATEMENT

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found below: NOD data resources publication website https://www.outfalls.info/publications and Metadata UTAS https://catalogue.aodn.org.au/geonetwork/srv/eng/metadata.show?uuid=21448123-0170-4aff-9b56-2b6aa21c73.

AUTHOR CONTRIBUTIONS

QR and AF presented the main idea. QR developed the theory, wrote the manuscript, carried out the data collection, and performed the analytical computation. AF directly supervised the findings of this work, and helped and verified the analytical methods. JC verified the numerical results on the summary statistics. BB contributed to the Tasmanian insights in the section "Discussion." JG provided critical feedback of the manuscript. All authors contributed to the section "Discussion," and verified and proofread the manuscript.

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Twelve Recommendations for Advancing Marine Conservation in European and Contiguous Seas

Stelios Katsanevakis[1]*, Marta Coll[2], Simonetta Fraschetti[3,4], Sylvaine Giakoumi[5], David Goldsborough[6], Vesna Mačić[7], Peter Mackelworth[8,9], Gil Rilov[10], Vanessa Stelzenmüller[11], Paolo G. Albano[12], Amanda E. Bates[13], Stanislao Bevilacqua[4,14], Elena Gissi[15], Virgilio Hermoso[16], Antonios D. Mazaris[17], Cristina Pita[18], Valentina Rossi[19], Yael Teff-Seker[20] and Katherine Yates[5,21]

[1] Department of Marine Sciences, University of the Aegean, Mytilene, Greece, [2] Institute of Marine Science, ICM-CSIC, Passeig Marítim de la Barceloneta, Ecopath International Initiative, Barcelona, Spain, [3] Stazione Zoologica Anton Dohrn, Università Federico II di Napoli, Naples, Italy, [4] CoNISMa, Rome, Italy, [5] Centre for Biodiversity and Conservation Science, School of Biological Sciences, The University of Queensland, Brisbane, QLD, Australia, [6] Van Hall Larenstein, University of Applied Sciences, Leeuwarden, Netherlands, [7] Institute of Marine Biology, University of Montenegro, Kotor, Montenegro, [8] Blue World Institute of Marine Research & Conservation, Veli Lošinj, Croatia, [9] Institute for Tourism, Zagreb, Croatia, [10] National Institute of Oceanography, Israel Oceanographic and Limnological Research, Haifa, Israel, [11] Thünen Institute of Sea Fisheries, Bremerhaven, Germany, [12] Department of Paleontology, University of Vienna, Vienna, Austria, [13] Department of Ocean Sciences, Memorial University of Newfoundland, St. John's, NL, Canada, [14] Department of Life Sciences, University of Trieste, Trieste, Italy, [15] IUAV University of Venice, Venice, Italy, [16] Centre de Ciència i Tecnologia Forestal de Catalunya (CTFC), Lleida, Spain, [17] Department of Ecology, School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece, [18] CESAM – Centre for Environmental and Marine Studies, Department of Environment and Planning, University of Aveiro, Aveiro, Portugal, [19] Institute for Research on Innovation and Services for Development, National Research Council, Naples, Italy, [20] Faculty of Architecture and Town Planning, Technion – Israel Institute of Technology, Haifa, Israel, [21] School of Environment and Life Sciences, University of Salford, Manchester, United Kingdom

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***Correspondence:**
Stelios Katsanevakis
stelios@katsanevakis.com

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Like most ocean regions today, the European and contiguous seas experience cumulative impacts from local human activities and global pressures. They are largely in poor environmental condition with deteriorating trends. Despite several success stories, European policies for marine conservation fall short of being effective. Acknowledging the challenges for marine conservation, a 4-year multi-national network, MarCons, supported collaborative marine conservation efforts to bridge the gap between science, management and policy, aiming to contribute in reversing present negative trends. By consolidating a large network of more than 100 scientists from 26 countries, and conducting a series of workshops over 4 years (2016–2020), MarCons analyzed challenges, opportunities and obstacles for advancing marine conservation in the European and contiguous seas. Here, we synthesize the major issues that emerged from this analysis and make 12 key recommendations for policy makers, marine managers, and researchers. To increase the effectiveness of marine conservation planning, we recommend (1) designing coherent networks of marine protected areas (MPAs) in the framework of marine spatial planning (MSP) and applying systematic conservation planning principles, including re-evaluation of existing management zones, (2) designing MPA networks within a broader transboundary planning framework, and (3) implementing integrated land-freshwater-sea approaches. To address inadequate or poorly informed management, we recommend (4) developing and implementing adaptive management plans in all sites of the Natura 2000 European conservation network and revising the Natura 2000 framework, (5) embedding and implementing cumulative effects assessments into a risk management process and making them operational, and (6) promoting actions to reach 'good environmental status' in all European waters. To account for global change in conservation planning and management, we further recommend (7) developing conservation strategies to address the impacts of global change, for example identifying climate-change refugia as high priority conservation areas, and (8) incorporating biological invasions in conservation plans and prioritizing management actions to control invasive species. Finally, to improve current practices that may compromise the effectiveness of conservation actions, we recommend (9) reinforcing the collection of high-quality open-access data, (10) improving mechanisms for public participation in MPA planning and management, (11) prioritizing conservation goals in full collaboration with stakeholders, and (12) addressing gender inequality in marine sciences and conservation.

Keywords: Natura 2000, MPAs, transboundary collaboration, global change, invasive species, cumulative impact assessment, conservation planning, risk management

INTRODUCTION

Marine systems are increasingly threatened by cumulative pressures from multiple human activities (Korpinen et al., 2012; Micheli et al., 2013; Mazaris et al., 2019; Jouffray et al., 2020) (**Figure 1**). In addition, the growing impacts of climate change (Philippart et al., 2011; Marbà et al., 2015; IPCC, 2019) interact in complex and context-dependent ways with local anthropogenic drivers (Ramírez et al., 2018). The European and contiguous seas, i.e., the Mediterranean Sea, the Black Sea, the Baltic Sea, the North Sea, and the North-Eastern Atlantic Ocean, provide iconic examples of the human footprint on marine ecosystems (CIESIN, 2020) and are hotspots of cumulative impacts (Emeis et al., 2015; Halpern et al., 2019). Human population density is very high, especially along the coastline, leading to intense marine uses and generating a number of conflicts over marine space (Katsanevakis et al., 2015; Kafas et al., 2018; Mackelworth et al., 2019).

The latest European Environment Agency report provides a grim picture of the status of European seas (European Environment Agency [EEA], 2015). European seas fall below a "healthy" status, their exploitation is unsustainable, and most ecosystem characteristics are in poor condition with deteriorating trends (Dailianis et al., 2018). In a recent assessment of the vulnerability of marine habitats in the European Union (EU) and adjacent regions (Gubbay et al., 2016), 18% of habitats were Critically Endangered, Endangered or Vulnerable. However, if data-deficient habitats are excluded, this figure rises to 38%, and if (under a precautionary approach) data-deficient habitats are considered threatened, the number rises to 71%. European seas, in particular the Mediterranean Sea, are a hotspot of extinction risk for sharks and rays (Dulvy et al., 2014), with no sign of improvement between the Mediterranean IUCN Red List assessments of 2007 and 2016 (Dulvy et al., 2016). For the majority of species assessments, the conservation status of fish stocks, marine turtles, and marine mammals in European seas is unfavorable (European Environment Agency [EEA], 2015). The frequency of population collapses and local extinctions has also increased especially in land-locked basins impacted by global warming. One such case is the Levantine basin in the Mediterranean Sea (Yeruham et al., 2015, 2019; Rilov, 2016; Corrales et al., 2018; Givan et al., 2018), where native biodiversity is gradually being replaced by alien species (Katsanevakis et al., 2018). Moreover, mass mortalities are increasingly occurring in association with strong and recurrent marine heat waves (Garrabou et al., 2019). Such biodiversity shifts can fundamentally alter ecosystem functions (e.g., Peleg et al., 2020) and compromise the flow of ecosystem services (Díaz et al., 2006; Worm et al., 2006).

As part of the United Nations Environment Programme, four Regional Seas Conventions (**Table 1**) have historically contributed to regionally coordinated conservation efforts in European and contiguous seas (Kirkman and Mackelworth, 2016). Within the EU, several legislative acts (**Table 1**) provide the basis for the development of instruments for the protection of marine biodiversity and ecosystem services, and sustainable use of marine resources (Fraschetti et al., 2018). Among them, the Natura 2000 European network of protected areas forms the cornerstone of EU biodiversity conservation strategy, including ca. 4000 sites, which are marine only or both terrestrial and marine, and cover ca. 12% of EU territorial waters (Mazaris et al., 2018).

Despite several success stories (e.g., Pipitone et al., 2014; WWF, 2017), European policies for marine conservation fall short of being effective (Fraschetti et al., 2018). While the objective of an ecosystem-based approach underpins EU environmental legislation, coupled socio-ecological research to advise on integrated ecosystem approaches are lacking (Visbeck, 2018; Lauerburg et al., 2020). Furthermore, the current attitude to reductionism in marine science hinders the implementation of an ecosystem-based approach (Fraschetti et al., 2008). The marine component of the Natura 2000 network fails to represent the

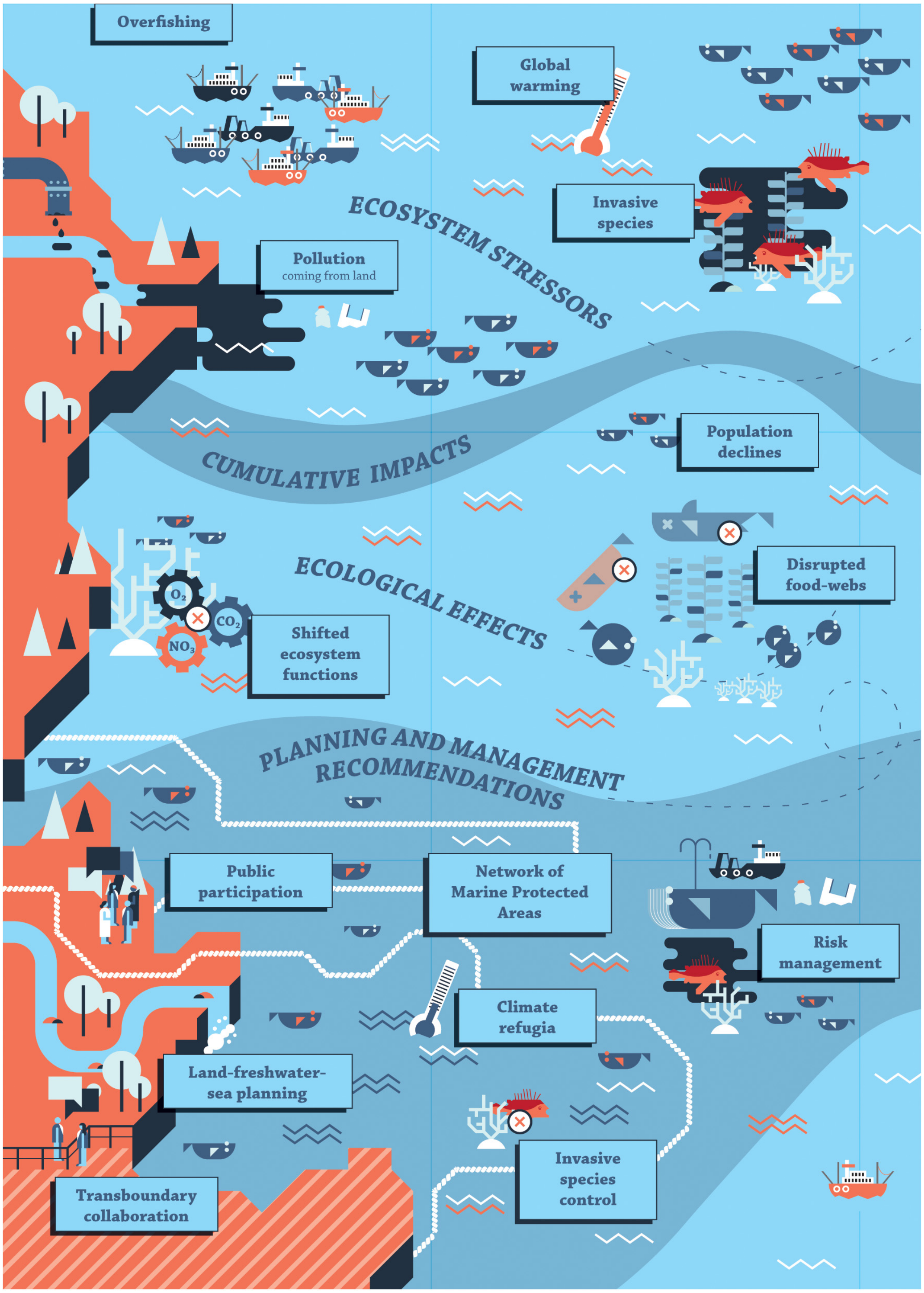

FIGURE 1 | Infographic showing the challenges MarCons aimed to address through planning and management recommendations.

full suite of marine and coastal habitats, largely excluding deep and offshore habitats, and many marine sites are just extensions of terrestrial sites and were not selected on the basis of marine conservation priorities (Mazaris et al., 2018). Indeed, a systematic planning process has not been applied to the design of the Natura 2000 network. Rather, designation has unfolded on a site-by-site basis with spatial configuration and connectivity largely ignored (Giakoumi et al., 2012). Economic interests have often prevailed over conservation objectives in guiding site selection (Olsen et al., 2013; Fraschetti et al., 2018). Furthermore, human activities

TABLE 1 | Conventions and legislative instruments contributing to conservation efforts in Europe and contiguous seas.

Legislative instruments/policies	Short description
OSPAR (https://www.ospar.org/)	Regional Convention for protecting and conserving the North–East Atlantic and its resources.
HELCOM Convention (https://helcom.fi/)	Regional Convention for protecting the Baltic marine environment.
Barcelona Convention (https://web.unep.org/unepmap/)	Regional Convention for the protection of the marine environment and the coastal region of the Mediterranean Sea.
Bucharest Convention (http://www.blacksea-commission.org)	Regional Convention on the protection of the Black Sea against pollution (including protection of biodiversity and marine living resources).
Birds Directive (Directive 79/409/EEC. Amended in 2009 and became Directive 2009/147/EC) (https://ec.europa.eu/environment/nature/legislation/birdsdirective/index_en.htm)	EU Directive aiming to protect all wild bird species naturally occurring in the European Union. It establishes a network of Special Protection Areas (SPAs) including all the most suitable territories for birds. Since 1994, all SPAs are included in the Natura 2000 ecological network, set up under the Habitats Directive 92/43/EEC.
Habitats Directive (Council Directive 92/43/EEC) (https://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm)	EU Directive for the conservation of habitats and a wide range of animal and plant species. It forms the cornerstone of Europe's nature conservation policy with the Birds Directive and establishes the EU-wide Natura 2000 ecological network of protected areas.
Water Framework Directive (Directive 2000/60/EC) (https://ec.europa.eu/environment/water/water-framework/index_en.html)	EU Directive establishing a framework for the Community action in the field of water policy. It sets common EU wide objectives for water (inland surface waters, transitional waters, coastal waters, and groundwater) and introduces an integrated and coordinated approach to water management in Europe.
Marine Strategy Framework Directive (MSFD) (Directive 2008/56/EC) (https://ec.europa.eu/environment/marine/eu-coast-and-marine-policy/marine-strategy-framework-directive/index_en.htm)	The Marine Strategy Framework Directive aims to achieve Good Environmental Status (GES) of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend. It promotes the integration of environmental considerations into all relevant policy areas and delivers the environmental pillar of the future maritime policy for the European Union.
Maritime Spatial Planning Framework Directive (Directive 2014/89/EU) (https://ec.europa.eu/maritimeaffairs/policy/maritime_spatial_planning_en)	EU Directive establishing a framework for maritime spatial planning, aiming to ensure that human activities at sea take place in an efficient, safe and sustainable way.
Common Fisheries Policy (https://ec.europa.eu/fisheries/cfp_en)	The CFP is a set of rules for managing European fishing fleets and for conserving fish stocks. It aims to ensure that fishing and aquaculture are environmentally, economically and socially sustainable and that they provide a source of healthy food for EU citizens.
EU Biodiversity Strategy for 2030 (https://ec.europa.eu/environment/nature/biodiversity/strategy/index_en.htm)	This is the EU plan for protecting nature and reversing the degradation of ecosystems. It contains specific commitments and actions to be delivered by 2030, including establishing a larger EU-wide network of protected areas on land and at sea, building upon existing Natura 2000 areas, with strict protection for areas of very high biodiversity and climate value. The EU Biodiversity Strategy aims to protect at least 30% of the land and 30% of the sea, with at least one third of protected areas strictly protected.

continue to jeopardize conservation efforts within protected sites (Yates et al., 2013; Mazaris et al., 2019), and less than 40% of marine sites have management plans, with many Natura 2000 sites considered just 'paper parks' with no actual conservation measures in place (Beal et al., 2017; Claudet et al., 2020).

Climate change mitigation is rarely addressed by EU marine environmental policies (e.g., in member states Programs of Measures under the MSFD) or national marine spatial plans, and the monitoring of marine protected areas (MPAs) commonly does not depict and clearly distinguish between impacts of local and global stressors (Rilov et al., 2020). Often, European and neighboring countries lack a shared vision, exhibiting remarkable heterogeneity in applying regional or European conservation policies, thus limiting transboundary collaboration and large-scale coherent ecological networks (Fraschetti et al., 2018). In practice, clear guidance and political support for transboundary marine conservation is generally lacking (Mackelworth et al., 2019). Even though there has been a global increase of cumulative effects assessments, member states and neighboring countries have not been effective in guiding management or conservation efforts in a multiple impact context (Stelzenmüller et al., 2018, 2020). For instance, despite the recognition that invasive alien species and neonatives (*sensu* Essl et al., 2019) may substantially compromise conservation efforts (Giakoumi et al., 2019a), biological invasions are rarely taken into account in conservation plans (Giakoumi et al., 2016; Mačić et al., 2018). Connections among realms are also commonly overlooked in conservation efforts, despite the need for integrated cross-realm actions for the protection of many threatened multi-realm species (Giakoumi et al., 2019b).

Acknowledging the challenges for marine conservation in the European and contiguous seas, the 4-year multi-national COST ('European Cooperation in Science and Technology') Action MarCons ('Advancing marine conservation in the European and contiguous seas[1]') aimed to bridge the gap between science, management and policy, and increase knowledge required for halting biodiversity loss (Katsanevakis et al., 2017). By consolidating a network of more than 100 marine scientists from 26 countries, in a series of workshops and meetings spanning from 2016 to 2020[2], MarCons analyzed key challenges, opportunities and obstacles, to build a common vision for research priorities and recommendations for advancing marine conservation (**Figures 1**, **2** and **Supplementary Table 1**). In each of these workshops, experts on the topic were invited and provided their expertise to achieve MarCons objectives (for

[1] http://www.marcons-cost.eu/

[2] http://www.marcons-cost.eu/activities/workshops

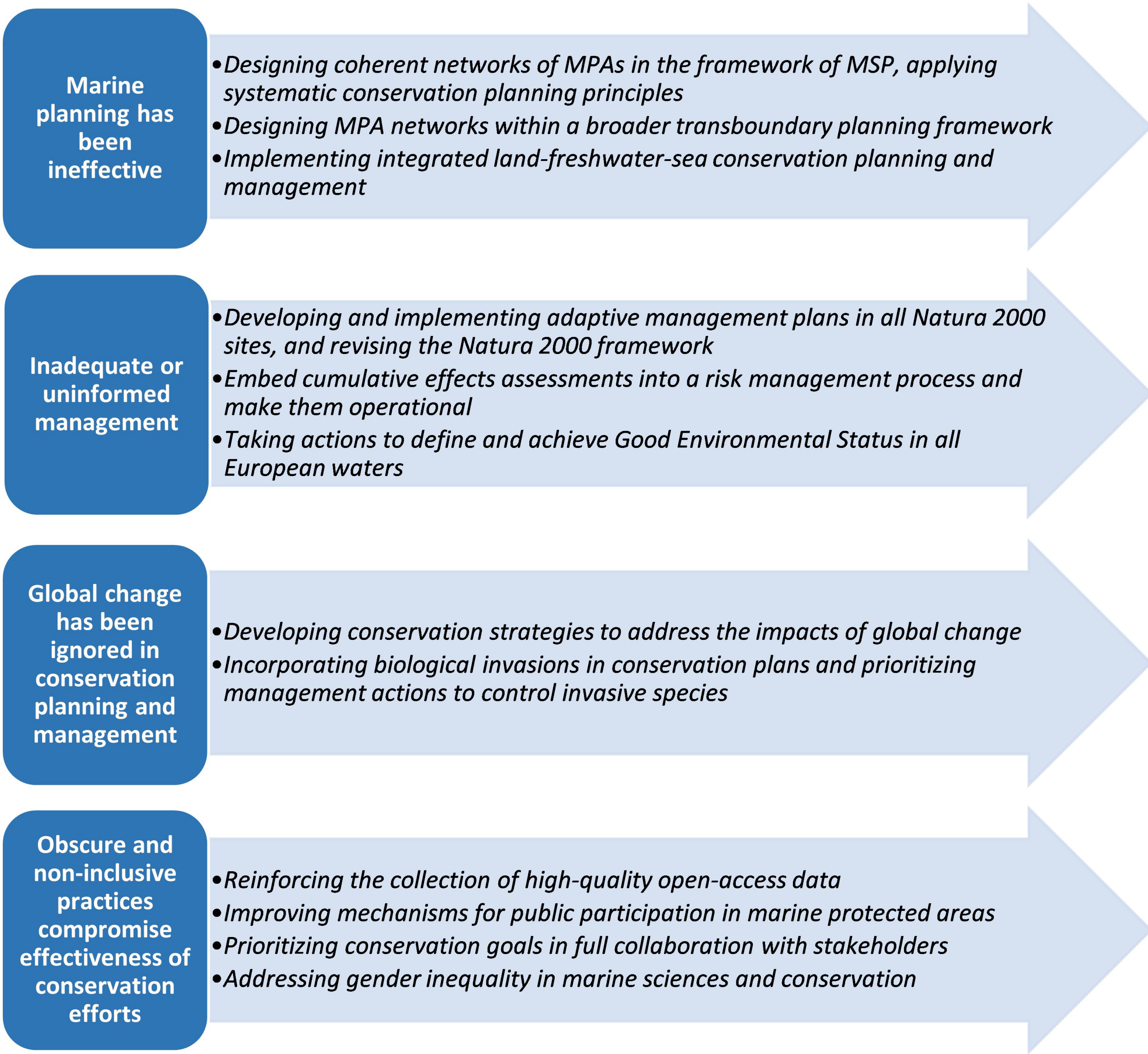

FIGURE 2 | Shortfalls of marine conservation efforts in European and contiguous seas, and main recommendations to advance marine conservation.

details on the working groups and stated MarCons objectives see Katsanevakis et al., 2017). Furthermore, policy makers (e.g., from the European Commission and member states), policy advisors (e.g., members of ICES and IUCN working groups), marine managers (e.g., from MedPAN – Mediterranean association of MPA managers), representatives of transboundary cooperation (e.g., Trilateral Wadden Sea Cooperation), and other stakeholders were invited in MarCons workshops to accommodate their needs and views in MarCons outputs. Various approaches were followed by the MarCons consortium to reach its stated goals. A number of systematic reviews were conducted to critically compile and analyze existing knowledge, current practices, methodological tools, and state-of-the-art in specific topics (e.g., Mačić et al., 2018; Gissi et al., 2019; Corrales et al., 2020). Data from large public databases, such as the Natura 2000 database, the European Red List, and the LIFE program (EU's funding instrument for the environment and climate) database, were retrieved and analyzed to gain insight on conservation outcomes, threats, practices, and efficiency (e.g., Fraschetti et al., 2018; Giakoumi et al., 2019b; Mazaris et al., 2019). Expert knowledge elicitation techniques were applied to evaluate and prioritize management actions (Giakoumi et al., 2019c). Participants offered their knowledge and experience on a national level through a large number of targeted case studies assessing how states have interpreted and utilized different legislative mechanisms over the governance of marine resources or maritime space, evaluating the implementation of conservation tools in Europe and beyond, their effectiveness and regional differences, and testing the operationalization of a risk-based cumulative effects assessment framework (Fraschetti et al., 2018;

Mackelworth et al., 2019; Stelzenmüller et al., 2020). EU Member States Programs of Measures designed for the implementation of EU marine environmental policies and recent European Marine Spatial Plans were critically examined (Rilov et al., 2020). Participating experts offered datasets, whose compilation and analysis provided new insights on the status of the marine environment in European regions (e.g., Bevilacqua et al., 2020). The collective and multi-disciplinary expertise within MarCons, combined with the above-mentioned analyses, was utilized to propose new approaches and tools to advance marine conservation in Europe and beyond (e.g., Bates et al., 2018; Stelzenmüller et al., 2018; Giakoumi et al., 2019b; Rilov et al., 2019). Through all these processes, MarCons working groups provided recommendations to advance marine conservation. These recommendations, published in the peer-reviewed outputs of the working groups, were derived from authors' assessments built upon accumulated knowledge in marine conservation as well as from interactions with different groups of stakeholders and evaluation of their needs.

Here, we synthesize the main findings and key recommendations of MarCons to guide science-based implementation of effective conservation actions in European and contiguous seas beyond 2020, as we step into the UN Decade of Ocean Science for Sustainable Development (Ocean Decade) and for Ecosystem Restoration, and as European research specifically is positioning itself to support the main objectives of the European Green Deal[3]. MarCons results can help to achieve the goals of the new EU Biodiversity Strategy for 2030 (**Table 1**), as setting ambitious targets in biodiversity conservation needs the development of concrete strategies to make their achievement possible. Whilst MarCons focused on European and contiguous seas, the lessons learned apply globally since marine ecosystems are connected and face similar threats.

TWELVE KEY RECOMMENDATIONS

Improved Conservation Planning

Recommendation 1. Designing coherent networks of MPAs in the framework of MSP, applying systematic conservation planning principles.

Decision-making for the management of marine socio-ecological systems is complex, as it must accommodate multiple, often conflicting, objectives/interests. For example, under the Blue Growth initiative the development of economic activities, such as marine tourism and aquaculture, are promoted, which may compromise conservation efforts (Rilov et al., 2020). Disentangling this complex situation requires strategic decision-making that is ideally informed by adequate planning. Marine spatial planning (MSP) initiatives, that explicitly integrate multiple objectives, are expanding worldwide, covering approximately 50% of the Exclusive Economic Zones (Frazao Santos et al., 2019). In European waters, marine spatial plans must be implemented by 2021 (Directive 2014/89/EU). MSP should follow an ecosystem-based approach in allocating maritime uses at sea (Ansong et al., 2017), including priority areas for environmental protection and restoration actions. The way countries will operationalize the ecosystem-based approach in their national MSP initiatives will potentially bring both threats and opportunities to marine conservation and human well-being (Fraschetti et al., 2018; Rilov et al., 2020). Whilst MSP efforts should consider all activities operating in marine space, giving priority to the future allocation of maritime uses that promote blue growth but do not affect ocean health when properly regulated (e.g., diving tourism, ocean energy, and marine biotechnology) will be a win-win strategy.

To ensure that MSP initiatives meet conservation needs and secure the establishment of ecologically coherent networks of MPAs across Europe's seas as requested by EU policies, most notably Article 13(4) of the Marine Strategy Framework Directive (MSFD) and the EU Biodiversity Strategy for 2030 (see **Table 1** for a description of each strategy), the implementation of systematic conservation planning is recommended. The importance of systematic conservation planning for marine spatial prioritization in the European seas has been consistently highlighted by scientists (e.g., Smith et al., 2009; Giakoumi et al., 2012; Metcalfe et al., 2013, 2015; Mazor et al., 2014). Systematic conservation planning provides a transparent, comprehensive framework for guiding the location, configuration, and management of biodiversity conservation areas (Pressey and Bottrill, 2009). The implementation of its core principles – connectivity, adequacy, representativeness, and efficiency – can support the design and management of ecologically coherent networks of MPAs in the European seas (Giakoumi et al., 2012; Fraschetti et al., 2018). For this to happen, systematic conservation planning should be adopted as the selected decision support tool for the future implementation of the key environmental policies, such as the Habitats, Birds, MSFD and MSP directives (see **Table 1**). Beyond species and habitat persistence, to better preserve the functioning of marine ecosystems, networks of MPAs should protect the functionality of marine communities and ecosystems (Bevilacqua and Terlizzi, 2020). To do so, identifying which habitats and species support fundamental ecological roles through space and time is needed. This understanding will provide guidance for the design of coherent networks of MPAs within the framework of MSP.

Marine spatial prioritization approaches with decision support tools, such as Marxan (Ball et al., 2009), have proven to be particularly helpful in integrating systematic conservation planning into MSP, as ecological, economic, and social objectives can be incorporated into the planning process (e.g., Mazor et al., 2014; Yates et al., 2015). Marine spatial prioritization is also useful to make the trade-offs between biodiversity conservation and its influence on economically important sectors more explicit (Gissi et al., 2018a). Given that many MPAs have already been designated within European waters, but so far have little or no conservation actions in place (Beal et al., 2017), systematic conservation planning can be utilized as an effective tool to prioritize actions within existing designations, as well as soliciting the implementation of additional MPAs to achieve

[3] https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en

the 30% conservation target set by the new EU Biodiversity Strategy for 2030.

Recommendation 2. Designing MPA networks to function within a broader transboundary planning framework.

All ecosystems straddle national borders. Often two or more countries share access and responsibility for the same habitats, species and ecosystem services, which is especially true in the highly connected marine system (Mackelworth, 2016). Transboundary cooperation can be highly beneficial, as it can allow the exchange of data and knowledge, synergize conservation and monitoring efforts, increase conservation planning efficiency, reduce overall conservation costs, and allow for joint management of transboundary natural resources (Kark et al., 2009; Mackelworth et al., 2019). In marine environments, where borders are not always as clearly marked or strictly upheld as those on land, transboundary cooperation should be easier. However, the same ambiguous characteristics of the border can also lead to disputes and conflicts over food, materials and space (Katsanevakis et al., 2015; Jouffray et al., 2020).

While the Natura 2000 network is considered a European wide network, in many instances its application is significantly different, even in adjacent states protecting the same resource (Mackelworth et al., 2019). One typical example is the application of the Natura 2000 network within the Wadden Sea World Heritage Site. A coherent network was not the output of the consultations between the three states of Denmark, Germany and the Netherlands, but the consequence of the decisions of the European Court of Justice (Enemark, 2016). In the Dogger Bank, there are ongoing management disputes despite the fact the borders are clearly defined and agreed. Of the four states that share the bank (Denmark, Germany, Netherlands, and United Kingdom), three have declared Natura 2000 sites, and the fourth has not. Even within the three sites declared there are serious incompatibilities in the conservation objectives (Mackelworth et al., 2019). The implementation of transboundary MPA networks becomes even more difficult in regions where severe international conflicts hamper any collaboration, such as in the Levantine Basin (south east Mediterranean) (Teff-Seker et al., 2019).

While there are transboundary areas that are recognized and protected, often the management systems in place differ between the states. Developing a coherent network that enables and encourages states to work together to protect common resources would be a major step forward in transboundary conservation. Approaching systematic conservation at a macro-regional level would help to facilitate transboundary cooperation, as shown in the Adriatic and Ionian Macro-region (Gissi et al., 2018a). The development of macro-regional strategies has the potential to facilitate conservation at the border, and even in the area beyond national jurisdiction.

Recommendation 3. Implementing integrated land-freshwater-sea conservation planning and management.

To achieve the EU's conservation target of halting biodiversity loss, the explicit consideration of connectivity and more effective protection of multi-realm species is required (Giakoumi et al., 2019b; Hermoso et al., 2019a). In particular, we recommend that the integration of conservation efforts across realms incorporates the following two steps:

(i) Recognition of the need for integrated management across realms at a policy level. Management policies and strategies will be much more efficient if they consider a broader array of ecosystems and their connections (Giakoumi et al., 2019b). This is needed to address the challenges associated with managing species with complex biological cycles that span across more than one realm. Conservation actions that only cover partially these complexities will often be ineffective (e.g., management of threats affecting just one of the realms the species relies on) (Tallis et al., 2008). Integrated management does not necessarily translate into large increases in area or other resource requirements, if planned adequately (Beger et al., 2010). For this reason, efficient cross-realm management needs to be accompanied by adequate planning (see below).

(ii) Implementation of integrated land-freshwater-sea conservation planning and management. An integrative approach when designating new Natura 2000 sites across realms could increase conservation outcomes and efficiency (Giakoumi et al., 2019b). Integrated conservation planning allows us to meet conservation needs in multiple realms in a more balanced and efficient way, to account for the needs of multi-realm species more by adequately enhancing connectivity across realms for those species that need it, and to explicitly consider the trade-offs between enhancing connectivity across realms and increases in cost (see also recommendation 1). However, further assessments are needed to evaluate the effectiveness of Natura 2000 as a tool for the integrated management of land-freshwater-sea and the species, communities and ecosystems that rely on these connections, and to identify critical areas for conservation outside currently protected areas.

Informed and More Effective Management

Recommendation 4. Developing and implementing adequate and adaptive management plans in all Natura 2000 sites, and revising the Natura 2000 framework.

All Natura 2000 sites were selected on the basis of the same criteria and procedures as defined in the Birds (2009/147/EC) and Habitats (92/43/EEC) Directives, and are subjected to common monitoring schemes and protocols; regular pan-European seminars and meetings aim to ensure a coherent network (Evans, 2012). These top-down processes resulted in a network characterized by a homogenization in the design, establishment and reporting phases. Still, biological features and processes (e.g., population dynamics, species interactions, and community stability), environmental conditions and fluctuations (e.g., ocean weather, frequency of extreme weather events) and socio-economic factors, which drive human activities differently across sites (Mazaris et al., 2019), make every single site a unique entity deserving site-specific, multidimensional efforts toward the understanding of its inherent complexity before being managed and protected. To improve management efficiency, these site-specific needs should be embedded in and fulfilled by flexible management plans that have been adapted to current conditions and regularly revised based on new knowledge and assessments of the effectiveness of previous decisions (Katsanevakis et al., 2011).

While management plans are vital for effective conservation, in many of the marine sites of the Natura 2000 network neither a management plan nor conservation measures are in place (Buhl-Mortensen et al., 2017; Fraschetti et al., 2018; Mazaris et al., 2018). Even when a management plan exists, there are often substantial time lags between site designation and plan implementation (in some cases of more than a decade), with great delays in their assessment and revision (Álvarez-Fernández et al., 2020), or they are never enforced due to legal challenges or lack of political will (Fraschetti et al., 2018).

As any management plan, the management plans of Natura 2000 sites, to be effective in a dynamic environment, should be periodically reviewed and revised. Adaptive management, as the process that involves the identification and consideration of shortfalls in planning through a monitoring-assessment-revision loop, will be more efficient if embedded within a risk-based framework for the operationalization of cumulative effect assessments (see recommendation 5). Under this context, systematic conservation planning and prioritization of management actions (see recommendation 1) can support in determining priorities and concerns for which plans need adaptive solutions, especially in view of the uncertainties, regime shifts and new challenges imposed by climate change (see recommendation 7) and biological invasions (recommendation 8).

Furthermore, it is time for the entire Natura 2000 framework (28 and 41 years after the adoption of the Habitats Directive and Birds Directive, respectively) to be revised to adapt to new knowledge, state-of-the-art systematic conservation planning approaches, and to better represent threatened biodiversity. It is common knowledge that the Annexes of the Habitats Directive (including species to be protected) inadequately represent marine biodiversity (Fraschetti et al., 2008), and species prioritization for protection is inconsistent with their actual conservation status as reflected by assessments using objective criteria (Maiorano et al., 2015; Habel et al., 2020). These Annexes urgently need revision to improve coverage of threatened species (Hermoso et al., 2019a), and a framework of regular reassessments and revisions of conservation priorities are needed to adapt marine conservation efforts within the Natura 2000 network to the actual changing conservation requirements (see also Cardoso, 2012; Hochkirch et al., 2013). We recommend that species lists in the EU Habitats and Birds Directives that define EU conservation priorities are revised and harmonized with the European Red List. The IUCN Red List Assessment is the most comprehensive global source of information on species extinction risk (Rodrigues et al., 2006) and central to setting conservation priorities (Stuart et al., 2010). Periodic revisions should capture the effectiveness of management actions financed through LIFE-Nature projects or any other funding scheme (Giakoumi et al., 2019b).

Acknowledging the complexities behind revising the Annexes of the Habitats and Birds Directives, alternative strategies should be also reinforced in the future. Among these, opening resource investments to all threatened species through programs like LIFE (Hermoso et al., 2018) or including these threatened species in Prioritized Action Frameworks (i.e., strategic pluriannual tools that review species conservation actions and financing needs across the Natura 2000 network) could provide funding opportunities to high-risk species not adequately covered by current provisions (Hermoso et al., 2019b).

Recommendation 5. Embed cumulative effects assessments (CEA) into a risk management process and make them operational.

Ecosystem-based management requires an assessment of the cumulative effects of human pressures and environmental change. Current decision-making processes do not include operationalization and integration of cumulative effects assessments (CEA), mainly due to their complexity and limitations of knowledge and evidence to allow for the identification of human activities and pressures that should be reduced. To make CEA operational, we suggest applying a comprehensive and transparent framework that embeds CEAs within a risk management process (**Figure 3**; Stelzenmüller et al., 2018). Applying such a risk-based CEA framework can structure the associated complex analyses and facilitate the establishment of direct science-policy links. We recommend a process consisting of the steps of risk identification (finding, recognizing, and describing risks), risk analysis (describing the risk of cumulative effects after accounting for the performance of existing management measures) and risk evaluation (comparing the results of risk analysis with the established risk criteria and benchmarks to determine the significance of the risk) (**Figure 3**). These three steps can help to reveal the likelihood of exceeding accepted risk of ecosystem state changes (Stelzenmüller et al., 2018). Embedding CEA into a management process decreases complexity, allows for the transparent treatment of uncertainty, and streamlines the uptake of scientific outcomes into the science-policy interface. Overall, we propose moving toward standardizing the CEA framework, with common terminology and procedures, and further developing integrative methods.

Cumulative effects assessments need to be well-framed to contribute in integrated planning, and function as tools that bridge different management objectives (Stephenson et al., 2019). Thus, applying the risk-based CEA framework proposed in MarCons (Stelzenmüller et al., 2018) and defining a strategy to communicate uncertainty is key for the operationalization of CEA (Stelzenmüller et al., 2020). This can contribute to overcome imperfect knowledge on the sensitivity of ecosystem components to distinct pressures, and embrace uncertainty around the scientific evidence (Cormier et al., 2017). Differentiating the aim of the CEA to advise policies, marine spatial planning or regulatory processes can facilitate the integration of ecosystem management considerations across multiple sectoral policies. In the process of operationalizing CEAs, and due to the involvement of many stakeholders, describing the roles of scientists and decision-makers well in advance will ensure transparency and clarify expectations. To improve current practices, assessing the effectiveness of management measures and how they can reduce the risk of negative impacts from cumulative effects is essential, but challenging for future research (Borja et al., 2020). It seems difficult to achieve a 'good environmental status' across European seas, without changing governance structures to integrate ecosystem considerations across multiple sectoral

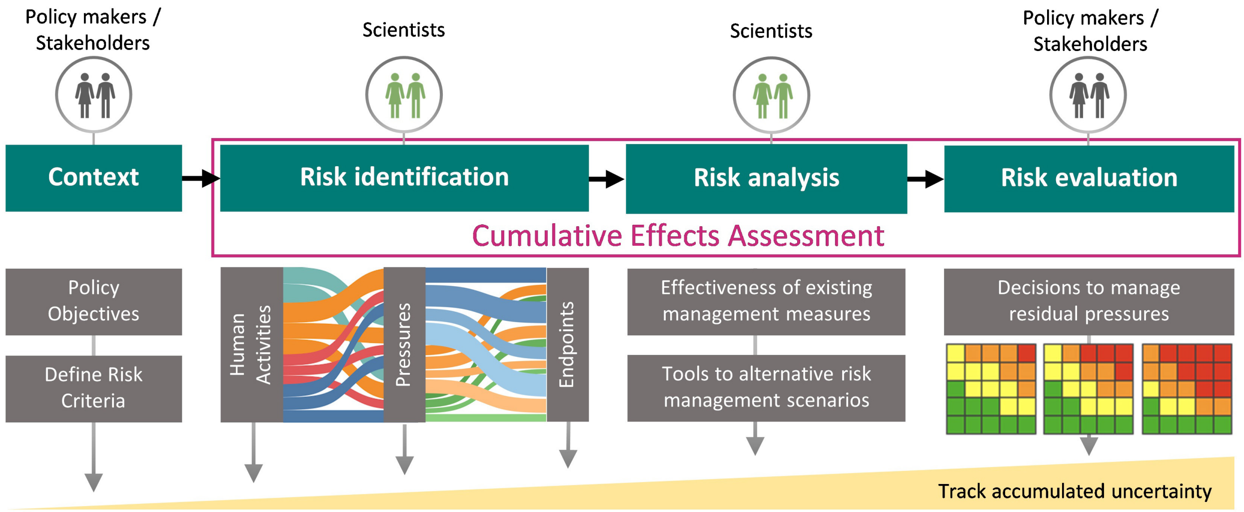

FIGURE 3 | Integrating Cumulative Effects Assessments (CEA) into a risk management process. CEA consists of the three basic steps of risk identification, risk analysis, and risk evaluation. Modified from Stelzenmüller et al. (2020).

policies (Cinnirella et al., 2014; Stelzenmüller et al., 2020). This is a difficult task, but we argue that well-framed and structured CEA can function as a strategic tool in this direction.

Recommendation 6. Taking actions to define and achieve good environmental status in all European waters.

The MSFD has set out a list of descriptors of environmental status. In practice, 'good environmental status' means that the different human activities use marine resources at a sustainable level, ensuring their continuity for future generations. Although MPAs and Natura 2000 sites are focal areas in the Marine Strategy Framework Directive, the condition of 'good environmental status' should be attained across all European waters, not only within areas under conservation regimes. EU seas and oceans are under high levels of human pressures from regional (Coll et al., 2012; Micheli et al., 2013) to local scales (Guarnieri et al., 2016), with inconsistent patterns in the ecological status of systems across entire basins, such as the Mediterranean Sea (Bevilacqua et al., 2020).

A major challenge for broad scale assessments is to define and quantify 'good environmental status.' In this respect, a critical limitation (still to be tackled) is the definition of thresholds to discriminate between different ecological conditions, which requires the knowledge of pressure-state-response relations of marine ecosystems (Borja et al., 2020). A second main problem is that we need spatially continuous data on the ecological condition of different components of marine ecosystems, which is largely unfeasible under current funding.

Cumulative effects assessments could be of crucial help to overcome these hindrances (see recommendation 5), by modeling expected ecological condition over large areas. Reliable predictions, however, should rely on extensive data on the status of ecosystems at varying pressure levels (Bevilacqua et al., 2018). To better define and guide the achievement of 'good environmental status,' future research should (i) capitalize on available spatially explicit data on the ecological status of marine ecosystems and associated pressures, (ii) fill information gaps for poorly studied areas and ecosystems, (iii) provide guidance for applying sound and robust indicators of the ecological status of marine ecosystems across all EU countries, tracking representative pressure-state response relationships to enhance the reliability of CEA, and (iv) define what is ecologically sustainable in a fast-changing ocean when conflicts between protection and the increasing human uses under the growth of the blue economy are rising. By prioritizing these themes, European funding schemes would substantially contribute to the efforts to reach good environmental status in the European seas.

Account for and Be Responsive to Change

Recommendation 7. Developing conservation strategies to address the impacts of global change.

Despite the increasing impact of global climate change on marine biodiversity, Europe and contiguous seas still focus on local and regional anthropogenic pressures. Yet, climate change can cause mass mortalities, reshuffle biodiversity patterns and drive shifts in species distributions, which can strongly affect management efforts. This tendency to consider mostly local and regional human pressures is reflected by the lack of consideration of climate change issues in actual marine management practice, as was exemplified with the implementation plans of the MSFD and MSP European directives by most member states (Rilov et al., 2020). Recently, Johnson and Kenchington (2019) argued that climate-change refugia (areas where climate change impacts are minimal) should become a criterion for the identification of ecologically or biologically significant marine areas as part of the actions proposed by the Convention on Biological Diversity. Under the rapid increase of climate change impacts, it becomes clear that networks of MPAs need to include climate-change refugia as areas of highest priority (Groves et al., 2012), for example, areas of upwelling of cooler waters from depth (Lourenço et al., 2016). Fine-scale data on ocean conditions will help to identify where potential refugia exist (Bates et al., 2018). Long-term ecological monitoring inside and outside properly managed MPAs should be promoted since it offers one of the strongest tools that can distinguish between local and global stressors (mainly climate change), and identify where signals of resilience exist.

The failure to distinguish and quantify climate change impacts means it is difficult to effectively incorporate climate change dynamics into the MSP process through conservation priorities, and prioritize adaptive management actions (Katsanevakis et al., 2011; Gissi et al., 2019). Toward adaptive management, it is critical that stakeholders acknowledge that marine conservation is a fast-moving target because of climate change. Consequently, management actions and policies will have to be able to cope and respond quickly to strong shifts in biodiversity and marine resources, driven by increasingly intense, and many times unpredictable, impacts of climate change (Rilov et al., 2020).

It is widely acknowledged that shifting from single MPAs into coherent networks will benefit conservation objectives (Olsen et al., 2013). However, climate change poses widespread and pervasive threats that may challenge the goal of MPA networks to fully protect biodiversity. Supporting marine conservation under climate change has been acknowledged as one of the grand challenges for the coming decade (Borja et al., 2020). We therefore suggest that in order to mitigate climate change impacts in European seas, we should focus on: (1) having well designed physical, ecological and socio-economic monitoring programs in MPAs and beyond as requested by the MSFD; (2) effectively including climate change risks into CEA; (3) identifying and considering potential climate refugia areas (where safety margins against extreme weather are large) in conservation plans; (4) setting different targets or criteria for the health of the system in climate hotspots (for example, focus on maintaining ecosystem functions instead of protecting specific species where thermally-sensitive native species rapidly collapse due to warming); (5) counting on safety in numbers and habitat diversity by ensuring that protection networks reflect different environmental conditions to allow for climate adaptation and recovery from extreme climatic events through population connectivity; (6) improving our ability to map climate-driven eco-evolutionary changes and identify vulnerable and resistant populations; (7) implementing adaptation and mitigation strategies iteratively, allowing for their evaluation as our knowledge base improves; and (8) adapting environmental policies by taking into account the above issues. We need to be realistic and well informed when attempting to address the challenge of on-going climate change, and we need to define precisely what is ecologically sustainable in the fast-changing ocean we observe today.

Recommendation 8. Incorporating biological invasions in conservation plans and prioritizing management actions to control invasive species.

The process of conservation planning usually focuses on native biodiversity and almost always overlooks alien species, either as a threat or as a benefit (Giakoumi et al., 2016; Mačić et al., 2018). A global cross-realm systematic review estimated that only 3.2% of conservation planning papers considered alien species in shaping their conservation plans (Mačić et al., 2018), although they often threaten native biodiversity and can cause a complete failure to achieve conservation goals (Simberloff et al., 2013; Katsanevakis et al., 2014). Hence, it is vital to carefully consider the ecological and socio-economic impacts of all alien species in conservation plans, with particular attention to invasive ones that exert the greatest impacts, with the aim to mitigate negative effects through specific conservation actions. Such plans should also recognize that some alien species might contribute to the achievement of conservation goals by securing ecosystem functioning and the flow of ecosystem services (Katsanevakis et al., 2014; Corrales et al., 2018), especially in regions suffering from multiple human stressors and global warming (Katsanevakis et al., 2018; Rilov et al., 2019). Even so, the new ecological state may be profoundly different from the pre-impact one (Peleg et al., 2020).

Furthermore, many species have extended their geographic ranges, without any direct human intervention, tracking human-induced environmental changes (Bates et al., 2014). These "neonatives" (as defined in Essl et al., 2019) may differ from alien species in their features of organismic novelty in the new regions (Essl et al., 2019), and there is evidence that they can become invasive with substantial impacts that are often functionally similar to those caused by alien species (Nackley et al., 2017).

In the marine environment, where most species have dispersal larval stages, eradication is extremely difficult, unless at a very initial stage of invasion (Ojaveer et al., 2015). It is also difficult to prevent geographic range expansions in neonatives, as propagules come from large and nearby source populations. Thus, conservation scientists, managers and decision makers should consider these species at all phases of the conservation planning process. Mačić et al. (2018) proposed 11 steps for the incorporation of alien species into conservation planning, building up on the planning design suggested by Pressey and Bottrill (2009). These steps capitalize on the early inclusion of alien and neonative species in the planning process and conservation goal setting, and in the recognition of a flexible and multi-faceted approach that includes avoiding areas too affected by alien and neonative species, or protecting such areas instead, either with the aim to control aliens and neonatives or include them in the protection plan when recognized useful for achieving conservation goals. We recommend incorporating biological invasions (alien and neonative species) in conservation plans through this stepwise approach (**Figure 4**), as ignoring alien and neonative species can change substantially conservation priorities (Giakoumi et al., 2016) and lead to considerable failures in the achievement of conservation goals (Bax et al., 2003).

Controlling marine invasive species is more likely to succeed when the species are detected early and management responses are rapid. Fast management responses require the early prioritization of actions based on their effectiveness, technical feasibility, social acceptance, impact and cost. In Giakoumi et al. (2019c), management actions were prioritized for groups of invasive species that share similar characteristics (differences in dispersion capacity, distribution in the area to be managed, and taxonomic identity). We recommend this

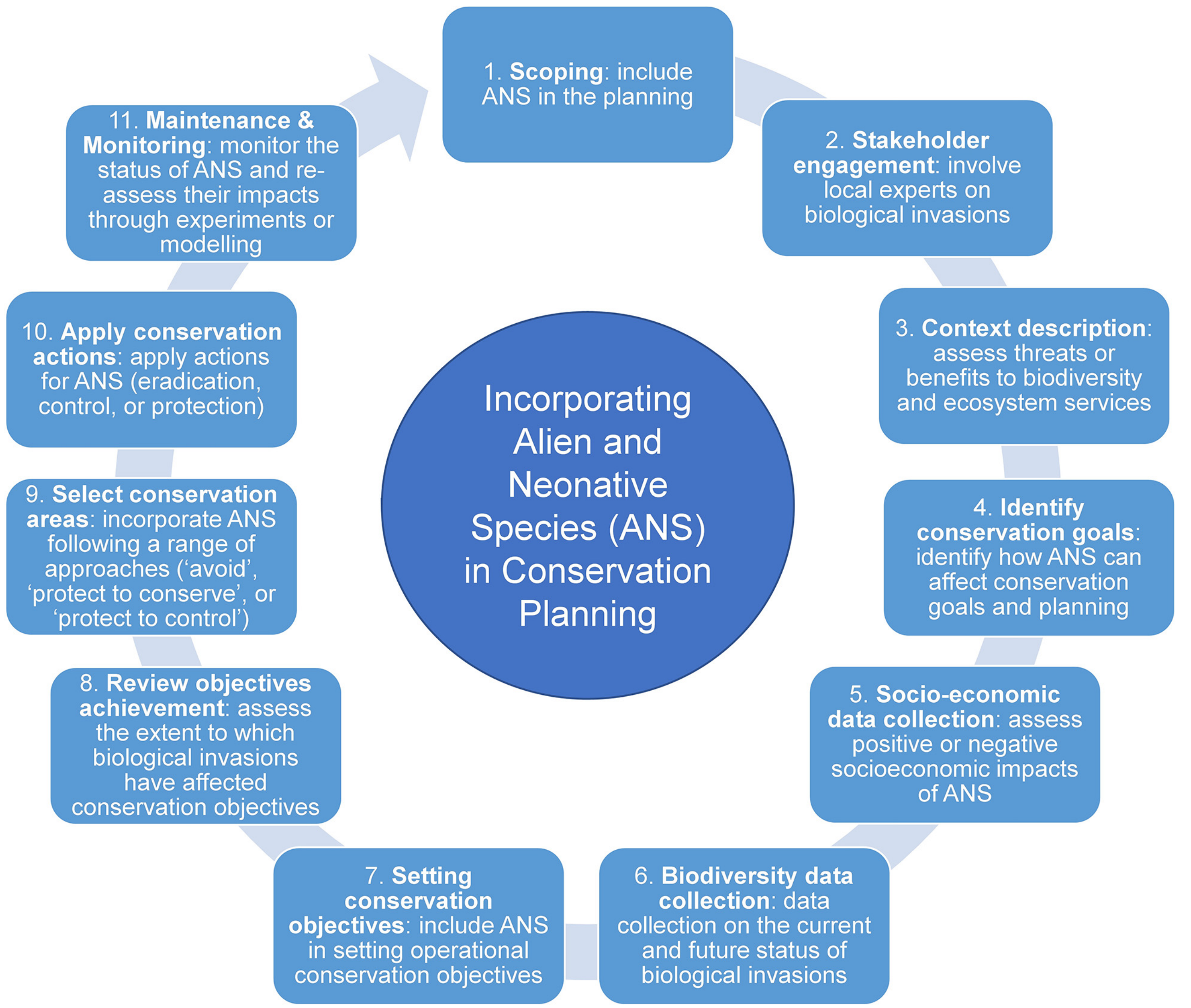

FIGURE 4 | Incorporating biological invasions in the 11-step process proposed by Pressey and Bottrill (2009). Short titles of the original steps are given in bold, followed by our proposed inclusion of Alien and Neonative Species (ANS) at each step. Adapted from Mačić et al. (2018).

approach whereby management actions are prioritized based on species characteristics and current spread, as a way for setting rapid management response priorities, without time-consuming species-specific evaluations. Actions such as raising public awareness and education, and physical removal and encouragement of commercial utilization of marine invaders are fundamental (Giakoumi et al., 2019c) and should be given special attention. Although waiting for invaders to diminish without any action may be considered the easiest and least expensive option, this approach should be discouraged. Spontaneous population decline in invasive species is difficult to predict and may only occur after persistent ecological damage has unfolded.

Transparent and Inclusive Conservation

Recommendation 9. Reinforcing the collection of high-quality open-access data.

Marine ecosystems are subject to a complex interplay of processes acting at different spatial and temporal scales, and are highly dynamic. Long-term monitoring programs are, therefore, essential to understand mechanisms underlying ecological changes and to guide an adaptive management of conservation strategies (see also recommendation 7). Evidence-based feedback through continuous and iterative monitoring, evaluation and reporting is crucial for achieving the objectives of any adaptive management framework (Day, 2008; Katsanevakis et al., 2011). Yet, the extent to which management measures are implemented and their outcomes monitored is poorly known for most European MPAs (Rilov et al., 2020). A limited number of MPAs have a monitoring plan assessing changes in the main species and habitats, and few MPA managers are aware of the current status of their protected areas and the effectiveness of conservation measures (Scianna et al., 2019). Even when monitoring programs exist, it is often unclear whether measures are effective to reach stated conservation targets. Aligned data across MPAs in terms of taxonomic resolution, sampling methods, habitat coverage, and collection at appropriate spatial and temporal scales are missing.

The setup of observing systems provides the data required to evaluate changes in habitats and species following the implementation of MPAs. More investments are needed to map the distribution and status of ecosystems, habitats and species and set observation platforms to improve our knowledge of biodiversity and ecosystem functioning. A fine-scale mapping of human pressures inside and outside of the MPAs is also fundamental to building measures, priorities, and decisions relevant at local and regional scales. This baseline information is mandatory for reserve siting, planning, and zoning in an MSP perspective and is of fundamental importance to make effective cumulative effect assessments.

New data should complement existing information, which is presently too fragmented in a plethora of repositories and digital archives. Specific investment is required to reinforce an exhaustive and homogeneous data collection of marine data at EU scale within a single, easily accessible platform (Vandepitte et al., 2010; Levin et al., 2014). A major impediment to facilitate open data and integration is the tendency of different disciplines involved in fundamental research, conservation and management of marine systems to act as separate compartments. Initiatives such as the COST Action MarCons, and platforms such as EMODNET (European Marine Observation and Data Network[4]; Calewaert et al., 2016), that increase exchanges and data sharing among experts in different, but complementary, disciplines are a prerequisite for future advances in marine conservation and spatial management. This transdisciplinarity will increase our understanding of pressure-state responses, improve the reliability of cumulative effect assessment models and enhance the effectiveness of conservation strategies in the context of MSP. Promoting and enforcing the obligation to release all standardized datasets produced through public funding under an open access license can maximize their use.

Recommendation 10. Improving mechanisms for public participation in marine protected areas.

Public participation in decision-making is an indelible element of environmental governance intended to foster sustainability of policies, promoting economic efficiency, environmental effectiveness, equity, and political legitimacy (Eden, 1996; Bryson et al., 2012; Pita et al., 2012; Yates and Schoeman, 2013). This governance approach is particularly relevant in the context of nature conservation. Biodiversity is a public resource with benefits that transcend society, and management requires instruments and approaches adequate to address the complex distributive and procedural justice implications of biodiversity loss (Rands et al., 2010). A key instrument for public participation is the United Nations Economic Commission for Europe (UNECE) Convention on Access to Information, Public Participation in Decision-Making and Access to Justice in Environmental Matters (Aarhus Convention). The Convention is a legally binding instrument on environmental democracy that puts Principle 10 of the Rio Declaration in practice and sets legal standards for public participation. The three pillars of public participation are: access to information, participation in decision-making processes and access to judicial and administrative proceedings (United Nations Economic Commission for Europe [UNECE], 1998).

A MarCons study analyzed official websites on MPAs in light of internationally agreed legal standards on public participation provided by the Aarhus Convention to investigate how States deal with public participation in the specific context of MPAs in the EU and contiguous seas (Rossi et al., unpublished data). The study evaluated information on 61 MPAs in 14 countries covering 5 EU regional seas. The results highlighted that access to information was typically limited and that making information available to allow the public to evaluate the "performance of public functions" is a target still far from being achieved. Public participation in decision-making processes is scarce: less than half of the MPAs provide information concerning specific decisions to be adopted that affect or are likely to affect the MPA. This, despite the Aarhus Convention specifying that public participation must be 'informed' and effective. Access to justice also raises serious issues in its implementation. Indeed, information concerning review procedures is very rare, and only 19% of the MPAs studied provide information on available means to challenge unlawful acts and omission that may be prejudicial to the objectives of the MPA. In fact, the implementation of the Aarhus Convention in the specific context of MPAs has been widely unsatisfactory. There is a disconnect between what countries say they are doing regarding the Aarhus Convention in general and what is visible regarding MPAs. The 2017 UNECE country reports on the implementation of the Aarhus Convention often only refer to generic participation platforms and mechanisms, and do not report on specific topics such as biodiversity conservation.

It is crucial to enhance public authority's awareness of their obligations but, most of all, public awareness of 'environmental procedural rights'. This 'right-based approach' to environmental protection is, finally, gaining increasing attention in biodiversity conservation (Knox, 2017). The full implementation of the Aarhus Convention can help ensure that biodiversity is truly managed as a public good. The involvement of the public and stakeholders is usually considered as a means to increase the efficiency of MPAs, guarantee buy-in of resource users to support management decisions, and increase compliance with rules and regulations (Gray, 2005; Berghöfer et al., 2008; Leite and Pita, 2016). There is the need for more meaningful public input than the archaic consultation process, which is really only effective at incorporating views of a very small subset of the public (Yates, 2018). There is also need for more transparency in the MPA designation process and on-going management (Saarman et al., 2013; D'Anna et al., 2016), as well as greater promotion of co-management and community stewardship (Alexander et al., 2017).

Recommendation 11. Prioritizing conservation goals in full collaboration with stakeholders.

Various actors involved in the use and protection of marine space rarely interact, with a substantial lack of involvement of MPA managers in the preparation of national programs of

[4]https://www.emodnet.eu/en

measures and marine spatial plans, and little or no collaboration with different national authorities. Although the MSFD requires the national Programs of Measures to go through a formal consultation process, this is only a consultation with no requirement to act based on stakeholder input. MSP can be seen as an instrument to facilitate the realization of blue growth, i.e., finding space for new human activities in marine areas. Yet, the involvement of stakeholders in MSP processes is very much at the discretion of the competent authorities, and MSP is often driven by top-down processes aiming to fulfill specific policy objectives such as renewable energy targets (Ehler, 2018).

Stakeholder participation has been criticized in the past on the grounds that it is often inefficient, that it often does not achieve genuine participation in planning and decision making (Yates, 2018), and that it seldom improves institutional decision-making (Innes and Booher, 2004). And yet, international organizations that advise and support marine planning processes around the world, such as UNESCO's Intergovernmental Oceanographic Commission, posit that it is a vital part of any such processes (Ehler and Douvere, 2009). Stakeholder participation can reduce conflicts among users of marine space (Ehler, 2008; Yates et al., 2013; Yates, 2018) and is critical for marine planning due to the public nature of marine resources and the need for integration in planning and management, including several dimensions at spatial, temporal and governance levels (Smith et al., 2009; Portman, 2014).

We recommend that early stakeholder involvement, in particular those that can influence or be affected by conservation actions, constitutes an important step in marine conservation planning and management (Pressey and Bottrill, 2009; Smith et al., 2009; Giakoumi et al., 2018; Yates, 2018). Such involvement has important benefits for the effectiveness of conservation, such as: eliciting information and valuable data on biodiversity and human activities that would otherwise be unavailable (Yates and Schoeman, 2013; Yates, 2014); better understanding of concerns of people likely to be affected by conservation actions (Gelcich et al., 2009; Pita et al., 2011; Yates, 2014); engendering trust among environmental managers and other key players; empowering people from all levels and areas of society and providing them a chance to impact their future; producing more sustainable policies; engaging with actors who may facilitate conservation actions financially and politically; helping to identify unexpected opportunities; and gaining important support by governmental and non-governmental organizations and the public (Arnstein, 1969; Pierce et al., 2005; Portman, 2009; Pressey and Bottrill, 2009; Smith et al., 2009; Gopnik et al., 2012).

Recommendation 12. Addressing gender inequality in marine sciences and conservation.

Gender equality has been identified as a key component of the health of marine social-ecological systems (Friedman et al., 2020). Gender equality is also key in defining research interests and priorities regarding ocean health; women have raised important, and often neglected, concerns in marine conservation (Gissi et al., 2018b). Within the framework of MarCons, we explored data from the EU (European Commission, 2019) and three EU research institutes and academia: the Spanish National Research Council (CSIC), the French National Centre for Scientific Research (CNRS), and the Academia in Italy. We found a consistent pattern of gender imbalance across institutions and nations. Whereas a relative gender balance was observed in Ph.D. graduates, a gap was formed between women and men representation in latter career stages, with women being most underrepresented in senior positions. The proportion of women in senior positions varied from 13% in CSIC to 24% in the Academia in Italy (Giakoumi et al., unpublished data). Furthermore, we observed the same pattern in publishing, funding (through European Research Council grants), leadership roles in research institutions, with EU women scientists being more underrepresented in latter stages of their scientific career path. This generalized gender bias can have an impact on setting conservation research priorities and communicating results to policy- and decision-makers (Tallis et al., 2014).

Michalena et al. (2020) showed that inclusive management is critical for the effective creation, use and adoption of environmental governance. We also conducted a global survey to explore the perceptions of marine scientists and practitioners on the role of women in marine sciences and conservation, and found that the vast majority (71%) of respondents (n = 768) believe that gender balance in leading scientific roles influences marine conservation outcomes in a positive way (Giakoumi et al., unpublished data). This perception was related to personal experience and/or scientific evidence demonstrating that gender diversity leads to solving problems more efficiently (Nielsen et al., 2017). There is evidence that women exhibit higher levels of social sensitivity and emotional awareness, and teams with a high proportion of women achieve greater equality in participation, boosting the collective intelligence in scientific team-work (Woolley et al., 2010). As women tend to be more likely to recognize the expertise of fellow team members, gender-integrated teams can also be more productive by fully exploiting team expertise (Joshi, 2014; Nielsen et al., 2017). To bridge the gap between science and policy and achieve biodiversity conservation more effectively, one prerequisite should be to close the gap of gender inequality in marine science and social-ecological systems, and thus harness the potential of gender diversity for collective innovation and increased effectiveness in conservation research and marine management.

CONCLUDING REMARKS

Despite the uptake of important EU conservation initiatives during the last decades, marine conservation in Europe is still challenged by knowledge gaps, inefficiencies, methodological limitations, bad practices, and a substantial gap between science and policy making. Systematic prioritization of economic needs often comes above the needs of the environment, in spite of future costs of short-term economic prioritization and the loss of natural capital. As a consequence, the European and contiguous seas face ineffective conservation policies and measures, and cumulative effects of multiple local and global human pressures, resulting in deteriorating trends and failure to halt biodiversity loss. A holistic vision of the conservation and management of marine space that balances conservation and exploitation of the natural capital can contribute in reversing these trends. This means that the business-as-usual scenario for marine conservation and current *ad hoc* reactive and segregated approaches need to drastically change. We need to plan for the future by taking proactive steps in revising current conservation policies, acknowledging the dynamic context of marine ecosystems, make explicit the human value systems underpinning management and conservation strategies, secure transparent, inclusive and collaborative decision-making, and bridge the gap between conservation science and policy making.

As the Natura 2000 network constitutes the backbone of conservation efforts in Europe, failing to address its weaknesses compromises the effectiveness of marine conservation in the European seas. Management plans and conservation actions are missing from most Natura marine sites and urgently need development. Unresolved conflicts among economic sectors or among countries hinder the effectiveness of conservation measures. Moreover, many procedures and rules for the governance of the Natura 2000 network are outdated, and insufficiently address the challenges of shifting policies and global change.

MarCons made 12 recommendations aiming to advance marine conservation by making marine planning more effective, improving management, accounting for global change, and improving current practices in marine conservation. Marine conservation needs to escape from inertia by incorporating the following key components: new risk-based approaches for cumulative effects assessments, regional collaboration, strategies for mitigating global change threats, systematic conservation planning approaches across realms instead of *ad hoc* and non-transparent spatial prioritization, adequate monitoring frameworks, adaptation strategies, data accessibility, and stakeholder engagement.

We have provided several examples of how the 12 recommendations can be implemented in existing and future management efforts as short and medium term strategies. For the present recommendations to find their way to European policy making and not remain just a wish list, further actions are needed. The twelve recommendations should be adopted at high levels by European institutions (i.e., the legislative instruments of the EU and regional conventions) to secure their wide implementation. This set of recommendations is a timely intervention, in view of the targets of the new EU Biodiversity Strategy for 2030, and the need to draft new legislation and implementation acts. A pathway to implementation mainly requires extensive lobbying with EU policy makers utilizing all points of intervention (i.e., Directorates General of the European Commission, Members of the European Parliament, parliamentary committees, working parties of the Council of Ministers, Commission expert groups).

Scientists have long expressed their fears that humanity has been pushing Earth's ecosystems beyond carrying capacities and proclaimed that fundamental changes in environmental policies and management are needed (Ripple et al., 2017). Despite the advances in conservation science and numerous past recommendations for better management of the oceans (e.g., Douvere, 2008; Heller and Zavaleta, 2009; Pressey and Bottrill, 2009; Smith et al., 2009), the gap between science and policy remained, representing one of the limits for making substantial progress in effective marine conservation and in halting biodiversity loss (Johnson et al., 2017; Ripple et al., 2017). Setting new targets, as the new ambitious EU Biodiversity Strategy for 2030 has been announced, is of critical importance to plan urgent conservation initiatives. However, without a change in the vision about the importance of developing a sustainable economy in harmony with healthy ecosystems those targets will never be reached. The valuable marine ecosystems in European seas and beyond need adequate protection before it is too late, and here we strongly advocate for substantial advances toward this overarching goal. The launching of the European Green Deal is an important recognition of the need for rapid action for building resilience of human and natural systems against global stressors, and it could be an important vehicle for the implementation of the list of recommendation provided here.

AUTHOR CONTRIBUTIONS

SK coordinated this work. MC, SF, SG, DG, VM, and PM coordinated the six MarCons working groups, and GR coordinated the *ad hoc* climate change focused working group, the work of which led to the recommendations made herein. SG supervised the creation of **Figure 1**. SK created and adapted **Figures 2–4**. All authors co-developed the 12 recommendations, contributed to the drafting of the manuscript, and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.565968/full#supplementary-material

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A Blueprint for an Inclusive, Global Deep-Sea Ocean Decade Field Program

Kerry L. Howell[1]*, Ana Hilário[2], A. Louise Allcock[3], David M. Bailey[4], Maria Baker[5], Malcolm R. Clark[6], Ana Colaço[7], Jon Copley[5], Erik E. Cordes[8], Roberto Danovaro[9], Awantha Dissanayake[10], Elva Escobar[11], Patricia Esquete[2], Austin J. Gallagher[12], Andrew R. Gates[13], Sylvie M. Gaudron[14,15], Christopher R. German[16], Kristina M. Gjerde[17], Nicholas D. Higgs[18], Nadine Le Bris[19], Lisa A. Levin[20], Elisabetta Manea[21], Craig McClain[22], Lenaick Menot[23], Nelia C. Mestre[24], Anna Metaxas[25], Rosanna J. Milligan[26], Agnes W. N. Muthumbi[27], Bhavani E. Narayanaswamy[28], Sofia P. Ramalho[2], Eva Ramirez-Llodra[29,30], Laura M. Robson[31], Alex D. Rogers[30], Javier Sellanes[32], Julia D. Sigwart[33], Kerry Sink[34,35], Paul V. R. Snelgrove[36], Paris V. Stefanoudis[37,38], Paulo Y. Sumida[39], Michelle L. Taylor[40], Andrew R. Thurber[41], Rui P. Vieira[42], Hiromi K. Watanabe[43], Lucy C. Woodall[37,38] and Joana R. Xavier[44,45]

[1] School of Biological and Marine Sciences, Plymouth University, Plymouth, United Kingdom, [2] CESAM and Biology Department, University of Aveiro, Aveiro, Portugal, [3] Ryan Institute and School of Natural Sciences, National University of Ireland Galway, Galway, Ireland, [4] Institute of Biodiversity, Animal Health and Comparative Medicine, University of Glasgow, Glasgow, United Kingdom, [5] School of Ocean and Earth Science, University of Southampton Waterfront Campus, National Oceanography Centre, Southampton, United Kingdom, [6] National Institute of Water and Atmospheric Research, Wellington, New Zealand, [7] Instituto do Mar, and Instituto de Investigação em Ciências do Mar – Okeanos, Universidade dos Açores, Horta, Portugal, [8] Department of Biology, Temple University, Philadelphia, PA, United States, [9] Department of Life and Environmental Sciences, Polytechnic University of Marche, Ancona and Stazione Zoologica Anton Dohrn, Naples, Italy, [10] University of Gibraltar, Gibraltar, Gibraltar, [11] Universidad Nacional Autónoma de México, Instituto de Ciencias del Mar y Limnología, Ciudad Universitaria, Mexico City, Mexico, [12] Beneath the Waves, Herndon, VA, United States, [13] National Oceanography Centre, European Way, Southampton, United Kingdom, [14] UMR 8187 Laboratoire d'Océanologie et de Géosciences (LOG), Université de Lille, ULCO, CNRS, Lille, France, [15] Sorbonne Université, UFR 927, Paris, France, [16] Department of Geology and Geophysics, Woods Hole Oceanographic Institution, Woods Hole, MA, United States, [17] Global Marine Program, International Union for Conservation of Nature, Cambridge, MA, United States, [18] Cape Eleuthera Institute, Eleuthera, Bahamas, [19] Oceanological Observatory Banyuls, Sorbonne University, Paris, France, [20] Center for Marine Biodiversity and Conservation, Scripps Institution of Oceanography, University of California, San Diego, San Diego, CA, United States, [21] Institute of Marine Science, National Research Council (ISMAR-CNR), Venice, Italy, [22] Louisiana Universities Marine Consortium, Chauvin, LA, United States, [23] DEEP/LEP, Brest, Ifremer, Plouzane, France, [24] CIMA – Centro de Investigação Marinha e Ambiental, Universidade do Algarve, Faro, Portugal, [25] Department of Oceanography, Dalhousie University, Halifax, NS, Canada, [26] Halmos College of Arts and Sciences, Nova Southeastern University, Dania Beach, FL, United States, [27] School of Biological Sciences, University of Nairobi, Nairobi, Kenya, [28] Scottish Association for Marine Science, Oban, United Kingdom, [29] Norwegian Institute for Water Research, Oslo, Norway, [30] REV Ocean, Lysaker, Norway, [31] Joint Nature Conservation Committee, Peterborough, United Kingdom, [32] Departamento de Biologia Marina and Nucleo Milenio ESMOI, Universidad Catolica del Norte, Coquimbo, Chile, [33] Senckenberg Research Institute, Frankfurt, Germany, [34] Centre for Biodiversity Conservation, South African National Biodiversity Institute, Pretoria, South Africa, [35] Institute for Coastal and Marine Research, Nelson Mandela University, Port Elizabeth, South Africa, [36] Department of Ocean Sciences and Department of Biology, Memorial University of Newfoundland, St. John's, NL, Canada, [37] Department of Zoology, University of Oxford, Oxford, United Kingdom, [38] Nekton Foundation, Oxford, United Kingdom, [39] Instituto Oceanográfico, Universidade de São Paulo, São Paulo, Brazil, [40] School of Life Sciences, University of Essex, Colchester, United Kingdom, [41] College of Earth, Ocean, and Atmospheric Sciences and Department of Microbiology, College of Science, Oregon State University, Corvallis, OR, United States, [42] Centre for Environment, Fisheries and Aquaculture Science, Lowestoft Laboratory, Lowestoft, United Kingdom, [43] X-STAR, Japan Agency for Marine-Earth Science and Technology, Yokosuka, Japan,

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Edited by:
Rowan Trebilco,
Centre for Marine Socioecology,
Australia

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Rachel Przeslawski,
Geoscience Australia, Australia
Alix Post,
Geoscience Australia, Australia

***Correspondence:**
Kerry L. Howell
kerry.howell@plymouth.ac.uk

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[44] Interdisciplinary Centre of Marine and Environmental Research (CIIMAR), University of Porto, Matosinhos, Portugal,
[45] Department of Biological Sciences, K.G. Jebsen Centre for Deep Sea Research, University of Bergen, Bergen, Norway

The ocean plays a crucial role in the functioning of the Earth System and in the provision of vital goods and services. The United Nations (UN) declared 2021–2030 as the UN Decade of Ocean Science for Sustainable Development. The Roadmap for the Ocean Decade aims to achieve six critical societal outcomes (SOs) by 2030, through the pursuit of four objectives (Os). It specifically recognizes the scarcity of biological data for deep-sea biomes, and challenges the global scientific community to conduct research to advance understanding of deep-sea ecosystems to inform sustainable management. In this paper, we map four key scientific questions identified by the academic community to the Ocean Decade SOs: (i) What is the diversity of life in the deep ocean? (ii) How are populations and habitats connected? (iii) What is the role of living organisms in ecosystem function and service provision? and (iv) How do species, communities, and ecosystems respond to disturbance? We then consider the design of a global-scale program to address these questions by reviewing key drivers of ecological pattern and process. We recommend using the following criteria to stratify a global survey design: biogeographic region, depth, horizontal distance, substrate type, high and low climate hazard, fished/unfished, near/far from sources of pollution, licensed/protected from industry activities. We consider both spatial and temporal surveys, and emphasize new biological data collection that prioritizes southern and polar latitudes, deeper (> 2000 m) depths, and midwater environments. We provide guidance on observational, experimental, and monitoring needs for different benthic and pelagic ecosystems. We then review recent efforts to standardize biological data and specimen collection and archiving, making "sampling design to knowledge application" recommendations in the context of a new global program. We also review and comment on needs, and recommend actions, to develop capacity in deep-sea research; and the role of inclusivity - from accessing indigenous and local knowledge to the sharing of technologies - as part of such a global program. We discuss the concept of a new global deep-sea biological research program *'Challenger 150,'* highlighting what it could deliver for the Ocean Decade and UN Sustainable Development Goal 14.

Keywords: deep sea, blue economy, Ocean Decade, Biodivercity, essential ocean variables

INTRODUCTION

Researchers have long recognized the ecological, economic and social importance of the natural capital of the global ocean to humanity (Costanza, 1999; Baker et al., 2020). However, ample evidence shows that, over time, the ocean has suffered increased stress from resource extraction, pollution, and climate change (Díaz et al., 2019; IPCC, 2019; Rogers et al., 2020a), including in the deep sea (Glover and Smith, 2003; Ramirez-Llodra et al., 2011; Sweetman et al., 2017). In 2015, the United Nations (UN) General Assembly set out 17 Sustainable Development Goals (SDGs) as a universal call to action to end poverty, protect the planet and ensure that all people enjoy peace and prosperity by 2030. SDG 14 specifically relates to marine biodiversity and its sustainable use, whereas other SDGs, for example SDG 2 on food security, SDG 8 on economic growth, SDG 12 on sustainable consumption, and SDG 13 on climate, amongst others, also apply to ocean health. Sustainable use of the marine environment also features in other UN policy commitments, including the Convention on Biological Diversity's Aichi Targets. Most recently the UN General Assembly proclaimed the UN Decade of Ocean Science for Sustainable Development (A/RES/72/73), hereinafter referred to as the Ocean Decade. The Ocean Decade will span a 10-year period commencing 1 January 2021, coordinated by UNESCO's Intergovernmental Oceanographic Commission (IOC). In preparation, the IOC issued a Roadmap (revised June 2018), that emphasized the need to drastically improve the current conditions of the world's ocean through science-based solutions and increased cooperation. To this end, the Revised Roadmap outlined six critical societal outcomes (SOs) that should be achieved through actions taken under the Decade (**Figure 1**) and identified the links between the strategic objectives of the Ocean Decade and the SDGs (United Nations, 2018). The Ocean Decade Draft Implementation Plan, published in March 2020, and revised in May 2020, presents the Objectives (Os) (**Figure 1**) for the Ocean Decade. It provides a framework within which to develop and deliver Ocean Decade Actions, defined in a scale hierarchy as programs, projects, activities, or contributions.

The revised Roadmap for the Ocean Decade recognizes the deep sea as a frontier environment. In order to deliver the Ocean Decade SOs and Os, it specifically calls on the scientific community to conduct research that advances understanding of deep-sea ecosystems, and their functions and services to human society. The Roadmap identifies an aspiration of the Ocean Decade to expand sustained and systematic ocean observations to all ocean basins and depths, to enable characterization of Essential Ocean Variables (EOVs-physical, biogeochemical and biological) and detect natural and human-induced changes. The deep sea (> 200 m) encompasses the largest living space on Earth, and accounts for more than 95% of the habitable volume (Danovaro et al., 2017). It supports diverse species and habitats, with the continental slope supporting higher diversity than the continental shelf (Rex and Etter, 2010; Muthumbi et al., 2011). According to some estimates, bathyal and abyssal diversity are amongst the highest on the planet (Grassle and Maciolek, 1992; Mora et al., 2011). The combination of geological, physical, and geochemical attributes of the deep seafloor and water column creates a mosaic of complex habitats with unique characteristics (Ramirez-Llodra et al., 2010). While we have only sampled or visually investigated a very small proportion of the deep ocean to date (0.01% with remote instruments Ramirez-Llodra et al., 2010), our current state of knowledge links society's well-being to the health of the deep sea through a wide range of ecosystem services (see Armstrong et al., 2012; Thurber et al., 2014; Baker et al., 2020 for a review). The remoteness of deep-sea ecosystems has historically led to a presumption that they are homogeneous and impervious to human activities; however, these diverse ecosystems increasingly face large-scale and cumulative impacts from multiple human activities with global influence (Ramirez-Llodra et al., 2011). Finding solutions to these risks challenges the scientific community, industry, national and international authorities and organizations to work collaboratively toward sustainable use and conservation of deep-sea ecosystems. Technological development, investment in research by both industry and philanthropic organizations, and an overall recognition of the significance of the deep-sea in broader Earth systems has driven rapid expansion in our investigation and understanding of deep-sea ecosystems over the last 20 years. However, fundamental questions remain in deep-sea biology and ecology that must be addressed in order to achieve the Ocean Decade SOs and Os. These questions have persisted despite almost 150 years of deep-sea research, and will continue to persist in the absence of a coordinated strategically targeted global effort to change the *status quo*.

In this paper, we review research needs of the Ocean Decade in the context of the design of a new 10 year deep-sea biology research program under the Ocean Decade. This paper aims to serve as the blueprint for global deep-sea research efforts for the next 10 years and likely beyond.

KEY SCIENTIFIC QUESTIONS MAPPED TO THE OCEAN DECADE SOCIETAL OBJECTIVES

Various fora and groups have reviewed and documented critical outstanding research questions in deep-sea research, including the European Marine Board (Rogers et al., 2015) and Census of Marine Life deep-sea field projects [e.g., Chemosynthetic ecosystems (German et al., 2011); Seamounts (Clark et al., 2012)]. Most recently, the Deep-Ocean Stewardship Initiative (DOSI) specifically convened the Decade of Deep-Ocean Science working group (DOSI-DDOS WG) to promote global-scale research to understand the role of deep-sea ecosystems in ocean health and resilience. DOSI, a network of over 1400 experts from 77 different countries, seeks to integrate science, technology, policy, law and economics to advise on ecosystem-based management of resource use in the deep ocean. The DOSI-DDOS WG currently consists of 67 people from 21 countries and is open to new members at all times. This group worked with the wider DOSI community during 2 events: (1) DOSI Day 2018 (09/09/2018, Monterey, United States) and (2) a meeting of the working group (October 2018, Aveiro, Portugal), to summarize science priorities and knowledge gaps pertaining to the deep ocean, and place them in the context of the SOs identified in the Revised Roadmap. Here, we report the outcome of those discussions, and identify four broad questions and their links to the Ocean Decade SOs.

What Is the Diversity of Life in the Deep Ocean?

Although knowledge on deep-sea community composition and ecosystem functioning has advanced rapidly in recent decades, we still lack fundamental ecological data for much of the deep sea (Glover et al., 2018). Poor knowledge of what lives there, how it is distributed from global to local scales, over environmental gradients (depth, temperature, oxygen, pH, primary productivity, etc.), and over time (seasonality, event-based phenomena, hydrodynamic cycles), precludes establishing effective baselines; in some cases, we still do not know what species are common or rare. Science has described many species, but many more await discovery and description, with repeated examples where presumed "common" species mask the presence of cryptic species (Vrijenhoek et al., 1994; Etter et al., 1999; Havermans et al., 2013). Baseline ecological data form the input to all biological ecosystem models and maps. Our ability to forecast how marine biodiversity will respond to environmental changes and anthropogenic-related pressures (SO3), depends on good base knowledge such as species identities, distributions, physical and chemical drivers of distribution, abundance, biomass, growth rates, etc. Fundamental ecological knowledge severely limits efforts to model and map present-day species distributions to fill data gaps, and predict future distributions under climate

Objectives

O1: Increase capacity to generate, understand, manage, and use ocean knowledge
O2: Identify and generate required ocean data, information and knowledge
O3: Build comprehensive understanding of the ocean and ocean governance systems
O4: Increase the use of ocean knowledge

Societal Outcomes

SO1: A clean Ocean whereby sources of pollution are identified, quantified and reduced, and pollutants removed from the Ocean
SO2: A healthy and resilient Ocean whereby marine ecosystems are mapped and protected, multiple impacts (including climate change) are measured and reduced, and provision of ocean ecosystem services is maintained
SO3: A predicted Ocean whereby society has the capacity to understand current and future ocean conditions, forecast their change and impact on human wellbeing and livelihoods
SO4: A safe Ocean whereby human communities are protected from ocean hazards and where safety of operations at sea and on the coast is ensured
SO5: A sustainably harvested and productive Ocean ensuring the provision of food supply and alternative livelihoods
SO6: A transparent and accessible Ocean whereby all nations, stakeholders and citizens have access to Ocean data and information, technologies, and have the capacities to inform their decisions.

FIGURE 1 | Decade of Ocean Science for Sustainable Development objectives identified in the recently published revised Draft Implementation Plan, and societal outcomes published in the revised Decade Roadmap.

change (SO2). Existing models and maps are simplistic, and of questionable accuracy due to limited, and/or poor quality input data (Davies and Guinotte, 2011; Howell et al., 2016; Morato et al., 2020).

How Are Populations and Habitats Connected?

Effective ocean management and sustainable use critically depend on identifying linkages among deep-sea ecosystems, communities, and populations. Collectively termed *connectivity* these linkages include: migration routes, ontogenetic or seasonal movement between habitats, spawning sites, larval dispersal pathways and genetic connectivity, or energy flow pathways in the form of trophic links and food webs. For example, maintaining well-connected populations, communities, and ecosystems underpins the design of effective networks of Marine Protected Areas (MPA) (Cowen and Sponaugle, 2009; Jenkins and Stevens, 2018) (SO2). Effective management of fish stocks (SO5) requires knowledge of how fish use their environment (essential fish habitats, spawning areas, migrations, larval and juvenile dispersal, food web interactions, etc.). Increasing evidence demonstrates that numerous commercially valuable fishes (e.g., tunas), marine mammals, and seabirds prey on deep-sea (mesopelagic) fishes, which form a significant component of their diet (Battaglia et al., 2013; Giménez et al., 2018; Watanuki and Thiebot, 2018). Strong connectivity promotes healthy and resilient populations (SO2), and disruptions to these connections, for example through changes in ocean circulation patterns or plumes from mining activities, can impact population persistence and recovery after disturbance, as well as the effectiveness of MPAs and other spatial protection measures. We must identify these connections to (1) help ensure that human activities do not alter them (SO2) and (2) enable us to predict the consequences of their disruption (SO3).

What Is the Role of Living Organisms in Ecosystem Function and Service Provision?

Sustainable development goal 14 widely recognizes the services provided by the ocean. Within the Decade Roadmap, SO5 focuses entirely on the provision of food supply and alternative livelihoods as key services provided by the ocean. We are at an early stage in understanding the role of the deep sea in provision of services (Armstrong et al., 2012; Thurber et al., 2014; Folkersen et al., 2018; Baker et al., 2020), with scant details on mechanisms of delivery. What are the key species/habitats involved in carbon sequestration? Are some groups more important than others? For example, we know sponges may play an important role in global Si cycling (Maldonado et al., 2019), as well as a sink for inorganic nitrogen, surpassing that of marine sediments at equivalent depths (Hoffmann et al., 2009). Does redundancy exist within the system, i.e., do more than one group of organisms perform the same functions associated with service delivery? The answer largely depends on the shape of the relationship between biodiversity and ecosystem functions (the so-called biodiversity-ecosystem functions curve; Danovaro et al., 2008). To ensure the ongoing provision of those services (SO2), and to understand better marine system processes such as biogeochemical cycling, we must identify the functional groups present, their role in ecosystem function, and how that function relates to delivery of services. Quantifying the variability, in space and time, of these processes on a global scale will enable us to predict changes to function and ecosystem service provision as a result of anthropogenic activities (SO3).

How Do Species, Communities, and Ecosystems Respond to Disturbance?

This question addresses both natural and anthropogenic disturbance (e.g., pollution, mining, fisheries, climate change, etc.) and gets to the heart of the knowledge required to manage marine ecosystem use effectively and deliver SO1 (a clean ocean), SO2 (a healthy and resilient ocean), SO3 (a predicted ocean), and SO5 (a sustainably harvested and productive ocean). Sustainable development requires knowledge of baseline environmental data and species tolerance thresholds to disturbance, as well as measurements and predictions of realistic natural disturbance regimes in order to place biological observations in context. Moreover, effective management of deep-ocean use in the future will hinge upon understanding impacts of multiple and cumulative stressors. Limited case studies from past and ongoing disturbances provide some information (e.g., Ashford et al., 2018; Vieira et al., 2020), but even fewer manipulative experiments address the mechanisms behind the responses at different temporal and / or spatial scales (Jones et al., 2017). This gap represents a key area for research development and one of the most important categories of information required for effective management efforts and stewardship of the global ocean. Related to this we must understand the potential for restoration of impacted deep-sea ecosystems with challenges largely associated with observation technologies (Van Dover et al., 2013; Levin et al., 2019), as well as with social, political, and economic interactions with science (Ehrlich and Pringle, 2008) and estimated costs (Van Dover et al., 2013).

These four broad questions lie at the heart of our ability to sustainably manage ocean use. The truly global challenge of addressing them requires a coordinated international and inclusive effort. As we approach the start of the Ocean Decade, the task before the scientific community is clear. We are charged with contributing knowledge to enable the Ocean Decade to achieve the stated SOs and Os. Our ability to plan for sustainable human use of the oceans, and adapt to environmental change lies in our ability to accurately predict possible outcomes and their socio-economic consequences. However, accurate forecasting requires at its base, ecological knowledge of species and habitats that, for the deep sea, is scant, highly spatially biased, with very few temporal data A new, globally coordinated program can address priority research questions that inform the development of a more holistic, non-sectoral, and equitable approach to sustainable use of deep-sea ecosystems. This program requires a coordinated, stepwise, and modular design. Next, we consider the design criteria in light of the identified research questions, and review current understanding of the role of key variables in shaping ecological pattern and process.

KEY ENVIRONMENTAL DRIVERS OF ECOLOGICAL PATTERN AND PROCESS

To address the SOs identified under the Ocean Decade we must quantify biodiversity and characterize species ecological niches, including their relationship to important climate-related variables. Until recently, the Inter-governmental Panel on Climate Change (IPCC) assessments had largely ignored climate change at the deep seafloor. However, the Special Report on Ocean and Cryosphere in a Changing Climate (SROCC) refers to clear regional and depth-related differences in projected temperature, POC flux, pH, and oxygen at the seafloor (Bindoff et al., 2019) under RCP 8.5 and 2.6 projections. To facilitate greater inclusion of the deep sea in IPCC assessment efforts we must stratify our sampling across these Essential Ocean Variables (EOVs) in order to quantify biological responses. Current and projected deep-ocean climate velocities exceed those at the surface, with consequences for the pelagic environment and consequently the benthos (Brito-Morales et al., 2020). Range mismatches among species across depths, could compromise vertical connectivity in the deep ocean (Brito-Morales et al., 2020).

Stratification by Latitude as a Proxy for Climate Related Variables

Ocean temperature above the permanent thermocline decreases with increasing latitude with sea-ice present in polar regions. POC export has a more complex relationship with latitude, but peaks at mid-latitudes (40–60 degrees) north and south of the Equator, with minima at ∼20 degrees (Lutz et al., 2007). POC export relates to POC flux to the seafloor, which in turn shapes benthic community composition (Billett et al., 2010). Both temperature and POC flux are implicated in driving global patterns of species diversity, but relationships between drivers and responses are still uncertain (Worm and Tittensor, 2018). The latitudinal diversity gradient (LDG) in species richness is one of the most well-established ecological paradigms for many terrestrial, freshwater, and coastal systems (Hillebrand, 2004), but with equivocal support for LDGs in the deep sea. Originally described by Humboldt and Bonpland (1807), the pattern predicts highest species diversity at the equator, with decreasing diversity toward the poles. The diverse proposed processes underlying LDGs include wide spatial variation in biological interactions, evolutionary processes, energy availability, climatic variability, physical heterogeneity and patchiness, and neutral processes (Pianka, 1966; Rohde, 1992).

Evidence suggests latitudinal gradients of diversity in benthic deep-sea fauna, although this conclusion rests on much lower sampling effort compared to other ecosystems. Rex et al. (1993, 2000) reported LDGs in the North Atlantic for gastropods, bivalves, and isopods. Patterns in the South Atlantic were weak and only present in some taxa (Brandt et al., 2005). This weaker South Atlantic pattern could relate to either lower sampling effort or strong regional effects on diversity, e.g., terrigenous carbon inputs in the Amazon Basin (Rex et al., 1997). Elevated Antarctic deep-sea biodiversity could also weaken the South Atlantic LDG if deep-sea biodiversity mirrors the high diversity of the Antarctic continental shelf (Clarke, 2008). Gage et al. (2004) found poleward declines in the diversity of deep-sea cumaceans for the entire Atlantic but only on the eastern corridor, supporting earlier observations of regional and basin effects. However, polychaetes in the Arctic Ocean illustrate that LDGs may occur even with a basin (Bodil et al., 2011). Among the benthic meiofauna, deep-sea foraminiferans show latitudinal gradients in the North and South Atlantic (Culver and Buzas, 2000) related to seasonality in pelagic production (Corliss et al., 2009). Nematodes peak in diversity at mid-latitudes in the North Atlantic (Mokievsky and Azovsky, 2002). More recent research demonstrates the complexity and variability of LDGs in the deep sea. Woolley et al. (2016) reported that both the patterns and underlying drivers of ophiuroid diversity clines can vary with depth, transitioning from diversity patterns driven by temperature at shallower depths to productivity at deeper depths. This pattern, along with an earlier body of evidence (Gooday et al., 1990; Lambshead et al., 2002; Rosa et al., 2008; Smith et al., 2008; Corliss et al., 2009; Tittensor et al., 2011; McClain et al., 2012a), points to the importance of organic matter availability in driving large scale patterns of diversity in the deep oceans.

In polar latitudes, where the effects of climate change are being expressed more rapidly, changes in surface primary production as a result of decreasing sea-ice cover will likely influence carbon supply to the deep sea (Rogers et al., 2020b). In the Arctic, declining summer sea-ice cover with additional strong and complex multiyear sea ice effects (CAFF, 2017), has resulted in a 30 percent increase in surface primary production (Arrigo et al., 2008; Arrigo and van Dijken, 2015). Regional patterns of change in the duration of sea ice as well as the collapse of ice shelves characterize Antarctica. In the Atlantic sector, including the Antarctic Peninsula, the present rapid decrease in sea ice coverage and duration follows a 40-year increasing trend

(Parkinson, 2019; Vernet et al., 2019). Collapse of ice shelves in the Weddell Sea has exposed new areas of the ocean for primary production. Changes in surface primary production, including the size (AMAP, 2017; CAFF, 2017) and functional types (Orkney et al., 2020) of phytoplankton cells, will alter pelagic food webs as well as POC export and flux to the seabed, subsequently affecting benthic communities (Gutt et al., 2011).

Although surface pH tends to increase with latitude, at the deep seafloor, Highest pH values occur in the North Atlantic, and lowest values in the North Pacific (Sweetman et al., 2017). Intense research in shallow water marine ecosystems over the last decade has examined biological responses to ocean pH stemming from the predicted decrease in pH under climate change scenarios termed 'ocean acidification.' Ocean acidification results from the absorption of atmospheric CO_2 by the ocean. This facet of global climate change significantly affects calcifying organisms by requiring them to utilize larger proportions of their energy budget to offset calcium carbonate dissolution (Muller and Nisbet, 2014). In surface waters, this change can alter phytoplankton community composition, potentially altering primary productivity (Dutkiewicz et al., 2015). These surface level changes will impact pelagic food webs and POC flux to deep-sea communities. In addition, the saturation state of carbonate decreases with depth, and therefore calcifying organisms in much of the deep ocean already face energetic challenges. Although deep-sea corals show remarkable capacity to maintain calcification in waters undersaturated in carbonate (Thresher et al., 2011; Gómez et al., 2018), further declines in pH will further challenge their calcification and growth, potentially leading to dissolution of existing deep-water coral reefs.

Highest sea surface oxygen concentrations occur in polar regions and lowest concentrations in equatorial regions. At the seafloor, oxygen patterns resemble those for pH, with highest dissolved oxygen concentrations in the North Atlantic, and lowest concentrations in the North Pacific (Sweetman et al., 2017). Oxygen strongly influences benthic fauna density, biodiversity, species distributions, taxonomic composition, food web structure, biogeochemical cycling, body size and species-level population and physiological rates (Levin and Gooday, 2003; Muthumbi et al., 2004; Laffoley and Baxter, 2019; Wishner et al., 2020). In the deep sea, the strongest influence of oxygen occurs at bathyal depths (200–1200 m) within oxygen minimum zones, but particularly in the prevalent extreme OMZs in the North and Eastern Pacific Ocean, Northern Indian Ocean, and off west Africa (Helly and Levin, 2004) as well as the Western Indian Ocean (Muthumbi et al., 2004). Many of these low oxygen areas are expanding under climate change (Stramma et al., 2008; Breitburg et al., 2018; Levin, 2018). The North and East Pacific and Southern Ocean have experienced the greatest oxygen losses over the last half century (Levin, 2018; Bindoff et al., 2019).

Although stratification in relation to latitude captures a range of current environmental variability, it does not capture evolutionary scale processes that have shaped faunal patterns of diversity and distribution. Many studies have considered regionalization of the marine environment into biogeographic zones. Watling et al. (2013) provides a detailed review of the history of deep-sea benthic biogeography as part of their development and refinement of a Global Open Ocean and Deep Sea (GOODS) classification (UNESCO, 2009; Watling et al., 2013). The GOODS classification was initially developed in 2009 in an expert consultation workshop for use in high seas management, initially basing proposed units on regions and provinces recommended by Menzies et al. (1973); Zezina (1973)Zezina (1997), and Vinogradova (1979) for bathyal and abyssal regions. However, boundaries were modified with recent data, published and unpublished observations, or re-analyses of existing data. Watling et al. (2013) further developed the classification, using physical and chemical proxies considered good predictors of benthic population distribution, to delineate 14 lower bathyal and 14 abyssal provinces. The fully classification also incorporated hadal provinces defined by Beliaev (1989).

Although the GOODS classification provides a convenient system by which to stratify benthic survey and monitoring, it does not consider the pelagic environment. Surface ocean properties from the basis of most pelagic biogeographic schemes (e.g., Longhurst, 1998; Spalding et al., 2012). However, in order to characterize the mesopelagic realm (200 — 1000 m), Sutton et al. (2017) collated expert opinion on physical and chemical oceanographic conditions, and biological expertise to define 33 mesopelagic ecoregions within four biomes (polar, westerly winds, trade wind, distant neritic). Their ecoregions reflected broad-scale patterns in the daytime distributions of mesopelagic fauna, with water mass structure, surface productivity, oxygen minimum zones and temperature extremes included as variables of particular importance. These ecoregions closely parallel those identified by Watling et al. (2013) in their classification of deep bathyal and abyssal biogeography. However, Sutton et al. (2017) note that their classification omits temporal variability in conditions, which plays a central role in the fluid and dynamic pelagic realm, and, like much of the deep pelagic biome, contains extensive data gaps (Webb et al., 2010). Using back scatter data, Proud et al. (2018) examined the global distribution of biomass (as backscatter intensity) within the deep scattering layer and identified 22 provinces, that correlated with overlying primary productivity and temperature at the depth of the deep scattering layer. No study to date has attempted to classify the bathypelagic realm, likely reflecting the lack of available data.

We propose a global program uses the revised GOODS classification (Watling et al., 2013) and mesopelagic ecoregions from Sutton et al. (2017), to stratify respectively benthic and pelagic deep-sea survey and monitoring, ensuring overall stratification by latitude.

Stratification at the Regional Level (e.g., Within Biogeographic Class)

Stratification by Depth

While latitude/biogeography can serve as a proxy for many key environmental variables, vertical gradients with depth arguably represent the strongest gradient of environmental change in the global ocean. Physical and chemical oceanographic drivers, and biogeochemical and biological responses all vary with depth. Key variables that correlate with depth (not always monotonically), and play a significant role in determining species distributions and community composition, include temperature (Haedrich et al., 1975), pressure (Somero et al., 1983), oxygen (Gallo and Levin, 2016), sediment type (Day and Pearcy, 1968), water mass structure (Howell et al., 2002), pH and aragonite saturation (Guinotte et al., 2006), and food supply (Rowe and Menzies, 1969), amongst others. Fauna undergo a non-repeating sequential change in composition with depth, with the combination of environmental variables that correlate with depth defining species depth ranges, as well as the fundamental ecological niche occupied by the species. Changes in environmental variables over depth also influence many ecological measures including diversity, abundance, and biomass. Stratifying sampling by each of these key environmental variables would be challenging. As with latitude/biogeography, depth itself serves as an easily measurable, widely accepted proxy for environmental variation. We therefore outline guidance on stratification of sampling by depth.

Although experts generally accept 200 m as the start depth for the deep sea, little evidence supports the existence of a benthic faunal boundary at this depth. Geomorphological rather than biological criteria define the transition between the deep circalittoral and deep sea, which occurs at the shelf edge break, typically taken as 200 m. Coral reef biologists recently described a new rariphotic zone extending from 150 to 300 m with community members predominantly related to shallow-water families (Baldwin et al., 2018). Carney (2005) summarized patterns of deep-sea benthic faunal zonation with depth and concluded that transition zones typically occur at 300 - 500 m (the shelf-slope zone of transition), ~1000 m (upper slope zone of transition), and 2000 —3000 m (lower slope zone of transition). Researchers typically classify the pelagic ocean by downwelling solar light levels where: the epipelagic zone (0 — 200 m), receives sufficient solar light for photosynthesis; the mesopelagic zone (200 — 1000 m), receives sufficient solar light for vision; the bathypelagic zone (> 1000 m) receives light coming only from biological sources (Sutton, 2013).

We propose a global program uses the following indicative depth horizons as a general guide for a target range for all biogeographic regions in order to achieve an unbiased global dataset: 150 — 300, 300 — 500 then every 500 m to the deepest point of the oceans at ~10 km. We chose these horizons to capture the scales of known faunal transitions over the depth gradient. These depth-delineated horizons should serve both pelagic and benthic known faunal zonation patterns. We do not dictate the density of sampling (fine or coarse) at any individual site.

Stratification by Horizontal Distance

The degree of faunal turnover (β-diversity) or its converse (similarity) between pairs of communities links closely to the spatial or environmental differences between them. The presence or absence of a species at one location, and similarity to another location, can reflect the geographic distance between them (i.e., the distance-decay relationship), acknowledging interplay with dispersal ability of the species, ocean currents, and availability of suitable habitat. Animal migrations can occur on scales of 1000s of kilometers. Larval dispersal of brooding invertebrates span scales on the order of meters to 10s of meters, whereas broadcast spawners disperse 10s to 1000s of kilometers depending on ocean currents and planktonic larval duration (McClain and Hardy, 2010; Hilário et al., 2015). Dispersal of adult stages can vary from centimeters to 1000s of kilometers, depending on size, swimming ability (e.g., crawling isopods versus migrating fish) and current patterns. Alternatively, these biogeographic and community patterns can reflect habitat heterogeneity (Cordes et al., 2010), niche-based processes such as environmental filtering, i.e., how species specific niche requirements map out on the environmental landscape, and the long-term consequences of interspecies interactions (e.g., Quattrini et al., 2017; Ashford et al., 2018).

At large geographical scales, the deep sea varies considerably in species diversity over latitude and depth (Rex and Etter, 2010). Few studies address regional to oceanic patterns of deep-sea β-diversity for megafauna. In general, specialist assume that even at larger scales, matching environment and species' niche requirements (i.e., environmental filtering) primarily drives patterns in biogeography and biodiversity (reviewed in McClain and Hardy, 2010). Some studies downplay the importance of dispersal limitation because the planktonic larval phases of many deep-sea invertebrates theoretically allow long-distance dispersal and potentially large biogeographic ranges (reviewed in McClain and Hardy, 2010), although empirical evidence suggests limited realized connectivity of 100s rather than 1000s of kms (Baco et al., 2016). Yet few studies have evaluated these concepts.

Most research on deep-sea β-diversity has focused on benthic macrofauna or meiofauna. McClain et al. (2012b) explored patterns of β-diversity in taxonomic, phylogenetic, and functional diversity across the Atlantic Ocean, using the bivalve data set of Allen (2008). Strong environmental filtering and dispersal limitation both drove turnover in taxonomic, functional, and phylogenetic composition. Blake and Grassle (1994) detected faunal changes both across and along isobaths in the southern region of the ACSAR program, NW Atlantic off North and South Carolina (United States). Depth explained most faunal variation, but with pronounced horizontal variation in the bathyal region. For meiofauna, Danovaro et al. (2009) found significant differences in assemblages across a longitudinal gradient for the entire ocean for abyssal but not bathyal sites in the Mediterranean Sea. Bianchelli et al. (2013) subsequently found evidence that energy availability was an important driver of the structure of deep-sea nematode assemblages. Food quantity drove patterns at larger scales, and the quality and bioavailability of food determined small-to-local scale patterns. However, other studies of nematodes found that productivity played a subordinate role to sedimentary characteristics (Vanreusel et al., 2010). In contrast Leduc et al. (2012) found distances between sites, both horizontally and vertically, explained the greatest proportion of variance in β-diversity of nematodes on the upper New Zealand continental slope.

The high level of coexistence of species in the deep sea represents one of the most intriguing paradoxes in ecology. Species richness in some deep-sea localities can exceed 300 species of macrofauna within a square meter (Grassle, 1989; Etter and Mullineaux, 2001). Sampling often yields species-accumulation curves that rarely reach an asymptote (reviewed in Etter and Mullineaux, 2001; Snelgrove and Smith, 2002), a pattern frequently interpreted as evidence of high turnover of local species. Empirical studies on the role of environmental patchiness in explaining these findings is mixed (Jumars, 1975, 1976; Thistle, 1979; Jumars and Eckman, 1983; Lamont et al., 1995; reviewed in Rex and Etter, 2010; McClain et al., 2011).

Although latitude/biogeography and depth provide useful frameworks for understanding ecological processes at a global scale, comprehensive understanding requires studies that describe and quantify ecological patterns at finer spatial scales. To enable us to begin to identify the scales of faunal turnover and connectivity, we propose that regional modules of a global sampling program stratify by horizontal distance at resolutions of 1 m, 10 m, 100 m, 1 km, 10 km, 100 km. Not all spatial scales will be appropriate for all size fractions or faunal components and this stratification should serve as a guide to interpret as appropriate to the specific study system.

Stratification by Anthropogenic Pressure

SO's 1, 2, 3 and 5 all require an enhanced understanding of human impacts on deep-sea ecosystems, in order to guide marine spatial planning and the sustainable exploitation of resources while safeguarding deep-sea life (Manea et al., 2020). Therefore, a comprehensive global study should stratify sampling by single and multiple anthropogenic pressures.

A design approach that stratifies by latitude will enable us to understand how species might respond to climate change. However, the degree of climate hazard (change relative to natural variability), and the time of emergence of climate change in the deep sea vary spatially (FAO, 2018; Bindoff et al., 2019; Levin et al., 2020). Some sites will experience climate change sooner than others. We propose targeting some early and some late emergence (e.g., potential climate refugia) sites. To effectively distinguish climate impacts requires selection of sites otherwise un-impacted, or minimally impacted by other anthropogenic pressures (e.g., not fished, low levels of pollution). This strategy implies inference from some prior knowledge on the intensity of human impacts at a given site from proxy data.

Global overfishing of coastal fish stocks and the need to feed a growing human population have led capture fisheries to target stocks inhabiting progressively deeper waters (e.g., Watson and Morato, 2013) and Areas Beyond National Jurisdiction (ABNJ) (e.g., Merrie et al., 2014) since the 1950s. In general, life-history traits such as slow growth, late maturation, and low fecundity (e.g., Drazen and Haedrich, 2012) increase the vulnerability of deep-sea fishes to fishing pressures. Long-lived species that form dense, local aggregations, such as orange roughy and oreos, are particularly vulnerable to rapid overexploitation, but approximately equal population reductions occur in all species whose ranges fall within the fished depth range (Bailey et al., 2009). The economic returns from demersal fishing decrease, and ecological costs increase with depth below c. 600 m in the NE Atlantic (Clark et al., 2016a). Restrictive measures to support sustainability of deep-water fisheries and to protect benthic ecosystems are now in place in EU Atlantic waters, including a depth ban on bottom trawl fishing below 800 m (Regulation (EU) 2016/2336). Recent papers suggesting exceptionally high mesopelagic fish biomass (Kaartvedt et al., 2012; Irigoien et al., 2014) have helped drive a resurgence of interest in targeting mesopelagic fishes as a source of protein for fishmeal. Acknowledging that deep-living pelagic fauna represent a largely unexploited marine resource, any future use must carefully balance benefits against both the considerable lack of knowledge about the ecology and ecosystem function of the deep pelagic realm (Webb et al., 2010), the high value of mesopelagic fauna in carbon transport and sequestration and their role in oceanic food webs (Colaço et al., 2013; Trueman et al., 2014). In addition to impacts on fish stocks, fishing, and specifically use of bottom contact gear, can cause significant adverse change or serious harm to benthic habitats and species (Rogers and Gianni, 2010), reducing structural and functional diversity and altering biogeochemical cycles (Puig et al., 2012; Ramalho et al., 2018; Vieira et al., 2019). Little information exists on resistance to, and rates of recovery from, the physical damage associated with bottom contact gear for many species. However, available data indicate limited resilience and thus high vulnerability in some species. At regional levels, Regional Fisheries Management Organizations (RFMOs) and national monitoring programs can provide satellite-derived data on marine vessel movements. These data enable a reasonably accurate assessment of the spatial distribution of fishing pressure (e.g., NAFO data input to benthic studies in the NW Atlantic; Ashford et al., 2018, 2019).

Litter and contaminants from different sources now infiltrate the deep sea, and have been identified in sediments and in biota (e.g., Ramirez-Llodra et al., 2010; Pham et al., 2014; Woodall et al., 2015; Courtene-Jones et al., 2020). These contaminants include debris of different types and sizes, such as plastics (ranging in size from nano- to macro-) or fishing gear, and other particulate or dissolved chemicals such as hydrocarbons, metals, and legacy and emergent persistent organic pollutants (POPs) (e.g., Pham et al., 2014; Woodall et al., 2014; Taylor et al., 2016; Jamieson et al., 2017). Nevertheless, we lack information on how these materials spread, whether a gradient of contamination decreases away from terrestrial sources, or how, and for how long, they move into the deep sea. High resolution ocean circulation models (van Gennip et al., 2019), indicate a ∼2 year transit time of the litter produced in the western coast of South America, including debris produced by the industrial fishery operating in the high seas off Chile and Peru, to the center of the South Pacific Gyre. However, we lack any reliable estimate of the portion of this debris reaching the deep sea and the time it takes. Importantly, how do pollutants accumulate in sediments and biota, and how do they impact fauna? Some initial studies reported microplastics in megafauna from the Rockall Trough dating back to the mid-1970s (Courtene-Jones et al., 2017) and we now know these contaminants spread to the deepest ocean trenches (Jamieson et al., 2019). Environmental risk assessment, for instance, requires knowledge of baseline levels for concentrations of contaminants, in order to develop sediment quality guidelines and to assess baseline levels of biomarkers of stress in organisms, against which to measure effects of disturbance. Stratification of sampling over a gradient of contamination adds significant challenges; ocean circulation and topography can concentrate litter (e.g., canyons, gyres), whereas bioaccumulation can concentrate pollutants. Distance from pollution sources, such as land, river mouths, or major shipping lanes offers one potential proxy.

Other forms of anthropogenic activities are more spatially constrained. Licensing requirements limit deep-seabed mining and oil and gas activities to specific locations best studied through dedicated regional monitoring programs. A global program could provide a baseline against which to monitor, and therefore should ensure inclusion of sampling locations within areas licensed for oil and gas extraction, or contracted for seabed mining exploration, as well as comparable areas (e.g., potential reference sites) that are protected from various forms of anthropogenic impact where possible.

We propose a global program replicates the following treatments in regional designs where possible: high and low climate hazard under early/late time of climate change emergence, fished/un-fished, near/far from pollution sources, licensed/protected from industry activities.

Other Considerations

Substrate type, an essential ocean variable, shapes benthic biological community composition. Historically, we largely based our knowledge of deep-sea benthic ecosystems on data obtained using trawls and sledges from soft-sediment seafloors. However, over the last 40 years, advances in technology coupled with decreases in cost, have supported growth in the use of video and still image-based tools for semi-quantitative sampling of previously inaccessible hard substrate habitats. The resulting new findings challenge the prevailing view of deep-sea ecosystems (Danovaro et al., 2014). The global design must factor in substrate type. However, lack of knowledge on seafloor composition constrains *a priori* stratification by substrate type. Collecting acoustic survey data (multibeam/sidescan sonar) prior to any biological work, undertaking topographic and acoustic backscatter classification, and stratifying biological surveys by remotely sensed bottom type (Brown et al., 2011; Riehl et al., 2020) provides a useful approximation. Alternatively, published models of seafloor lithology may be useful (Diesing, 2020). However, these models likely lack the resolution required for realizing stratification by substrate type in survey design.

Global bathymetry data, such as the General Bathymetric Chart of the Oceans, offers another approach to stratification, based on topography. This simplified form of geomorphological classification analyses terrain derived variables such as slope, rugosity, bathymetric position index, to differentiate terrain types. Bottom slope provides a particularly useful proxy for multiple ecologically relevant variables (McArthur et al., 2010). Existing geomorphological classifications (e.g., Harris et al., 2014) may provide a useful standardized means by which to consider the global stratification of sampling. However, such classification may not produce ecologically meaningful geomorphological classes, and thus should not form the basis of stratification efforts. Nevertheless, a global program should strive to include different ecosystem types, an issue addressed further in section 6.

We propose that a global program should stratify sampling by substrate type and / or topography, including bottom slope, within regional designs.

TEMPORAL SURVEY AND MONITORING NEEDS

Effective assessment of human impacts requires long-term monitoring (time series) of both impacted and control sites. In addition, to determine the functional significance of organisms and their role in the delivery of goods and services to humankind critically requires temporal sampling and experimentation. Although our blueprint focuses on setting spatial design criteria for a global survey program, the design must include sites prioritized for monitoring and temporal surveys.

Within each biogeographic region experts should identify and include potential monitoring sites in the design. Levin et al. (2019) and the DOOS initiative provide an inventory of current sustained deep-ocean observing activities, and propose a series of potential region-specific, interdisciplinary projects to demonstrate the feasibility of sustained deep-ocean observing, relevant technologies, and the impact and utilization of deep-ocean observations. These proposed locations include the Clarion-Clipperton Zone, Azores Archipelago, Northeast Pacific: Cascadia Margin to the Juan de Fuca Ridge, Western Pacific, and Ocean Trenches: Izu-Ogasawara Trench and Mariana Trench. Researchers selected these sites on the basis of strategic advantages and existing infrastructure, and they represent excellent choices for demonstration projects, but we must now identify further sites for all biogeographic regions (e.g., Indian Ocean) and take the first steps toward establishment of a globally comprehensive network of sites for sustained observations. Following Levin et al. (2019), we propose a global program use the following criteria in site selection: access to different strata outlined in the global design, availability of existing observing infrastructure, opportunity and ease of installing and maintaining new infrastructure.

PRIORITY AREAS FOR NEW BIOLOGICAL AND ECOLOGICAL DATA COLLECTION

The strong spatial bias in our knowledge of the marine environment largely drives the need for a globally coordinated and inclusive program. Latitudinally, the most undersampled regions include equatorial and polar areas as well as southerly latitudes (Menegotto and Rangel, 2018). Researchers have prioritized the data poor south Atlantic, south and central Pacific and Indian Oceans for research (Clark et al., 2012; Saeedi et al., 2019). Globally, sampling effort decreases with depth. For example as of 2019 10.4% of the Ocean Biodiversity Information System (OBIS) records were from > 200 m, with only 1.5% of time series data (> 5 years) falling below 200 m. Only 158,000 records fell between 500 and 10,900 m. At the conclusion of the Census of Marine Life, Clark et al. (2010) highlighted the lack of data available on deeper sections of seamounts (> 2000 m depth). Recently, Taylor and Roterman (2017) identified only nine published papers that dealt with population genetics below 3500 m depth. The bathypelagic environment is the least studied, and largest component of the deep oceans by volume (Webb et al., 2010). The BioTIME initiative reports similar spatial bias in time series data with most studies occurring in Europe, North America and Australia (Dornelas et al., 2018), with large data gaps in the Pacific, South Atlantic and Indian Oceans. As ice cover in the Arctic continues to decline, this ocean basin will experience increasing anthropogenic influence and deep-sea research efforts should also prioritize this key region. There is a clear and well documented need to prioritize research effort in southern and polar latitudes, deeper depths, and midwater environments.

SPECIAL CONSIDERATIONS FOR DIFFERENT ECOSYSTEM TYPES

The global design detailed above sets out a strategy that is independent of any perceived ecosystem type, and thus moves away from the traditional silos in which many deep-sea researchers find themselves. However, the global design should preferentially ensure inclusion of different ecosystem types. Here, Harris et al.'s (2014) global geomorphological classification scheme may provide a useful standard against which to classify a given study site. However, from an ecological perspective some of Harris et al. (2014) classes easily group into single ecosystem types that correspond to established research areas with the deep-sea biological research community. Some of these ecosystem types require additional design considerations to facilitate a more complete representation of these specific systems with respect to addressing the Ocean Decade SOs. We identify different ecosystem types, the equivalent Harris et al. (2014) class(es) and further variables by which to stratify individual or regional project level designs (**Table 1**).

THE NEED FOR STANDARDIZATION IN OBSERVATIONS AND METHODOLOGIES

Increasing evidence in recent years illustrate that inconsistencies in sampling of different habitats and regions have challenged efforts to bring datasets together and provide a global picture. Several deep-sea field projects under the Census of Marine Life (2001–2010) noted this issue when collating and collectively analyzing their data. The highly variable array of sampling equipment and survey approaches constrained analyses. Effective broad-scale analyses of ecological patterns and processes, and human impacts requires standardized comparable data (Clark et al., 2016b). The high diversity of life forms, from microscopic bacteria to large cetaceans, require different sampling approaches and methods depending on the composition and abundance of the biological communities and environmental characteristics of their habitat. Although national goals often drive scientific objectives and specific survey design, we identified consistent sampling across national and international programs as a priority to advance our knowledge of deep-sea ecosystems. The guidance on "best-practice" sampling of deep-sea environments (Clark et al., 2016b) complemented other deep-sea texts (e.g., Danovaro et al., 2010, 2020; Eleftheriou, 2013) and initiatives such as the Global Ocean Observing System (GOOS) and the Deep Ocean Observation Strategy (DOOS) in trying to improve global-scale science.

Building on the efforts of GOOS (Miloslavich et al., 2018a), DOOS has proposed essential ocean variables (EOVs) for the deep ocean (Levin et al., 2019). Many of the GOOS variables identified by the Bio/Eco panel occur only in shallow water (e.g., mangrove, seagrass, algal cover, turtles). DOOS has identified a suite of physical, biogeochemical, and biological/ecological variables that are sufficiently mature (technologically ready), suitable for sustained, and, in some cases automated, observing; but this prioritization omits many critical kinds of information. Danovaro et al. (2020) identified, through expert elicitation, a set of essential ecological variables necessary to address (1) biodiversity; (2) ecosystem functions; (3) impacts and risk assessment; (4) climate change, adaptation and evolution; and (5) ecosystem conservation at the deep seafloor.

The need for a consistent approach to data collection and close collaboration between marine scientists from different countries and disciplines to advance knowledge of the ocean also catalyzed the development of the General Ocean Survey and Sampling Iterative Protocol (GOSSIP) (Woodall et al., 2018). Focusing on 20 biological, chemical, physical, and socioeconomic parameters the detailed GOSSIP framework supports consistency in future marine data collections. It highlights standardized collection methods and discusses their relevant limitations and caveats to help researchers apply (or at least understand) best practice techniques for generating globally comparable marine data. Although concerned with mesophotic, deep-pelagic, and bathyal biological communities, application of this protocol more widely offers a good starting point for research efforts under the Ocean Decade. In addition, the International Oceanographic Data and Information Exchange of the IOC has developed the Ocean Best Practices System (OBPS) including an open access, permanent, digital repository of community best practices in ocean-related sciences and applications (Pearlman et al., 2019). This repository contains highly detailed standard operating procedures and operational field manuals for a range of survey and sampling equipment and techniques, and may provide additional detail and points of reference for specific gear types and procedures, including quality assurance and archiving of data.

Although standardized observations and methodologies require further work, we recommend that a global program adapt Woodall et al. (2018); Levin et al. (2019), and Danovaro et al. (2020), relevant archived best practice documents in the OBPS (Pearlman et al., 2019) as a basis for further development of standardized approaches to deep-sea biological survey and monitoring. Danovaro et al.'s (2020) and **Table 1** provides a summary of actions required for deep-sea monitoring of the most important essential ecological variables. Woodall et al.'s (2018) and **Table 1** summarizes key measurements and methods to obtain such measurements in a robust, standardized,

TABLE 1 | Ecosystem-specific considerations.

Ecosystem type	Stratification variable	Importance of stratification variable	Recommendation
Topographic rises including Seamounts, guyots, ridges, abyssal hills, abyssal mountains, mid ocean ridge.	Position on topographic rise, e.g., summit, flanks, base, aspect.	Oceanographic conditions differ substantially between summit, flanks and base related to terrain slope and the wider geomorphological nature of the feature (Rogers, 1994).	Ensure sampling of summit, flanks, and base.
	Aspect	Can reflect different oceanographic conditions.	Aspect (e.g., up-current versus down-current; ridge axis versus flanks either side) should be factored in.
Canyons – shelf incising and blind.	Shelf-incising vs slope-confined (blind) canyons (sensu Huang et al., 2014)	Contrasted organic resource supply in intensity and frequency, hard substrates/soft bottom habitats, both in relation to canyon hydrodynamics (Fernandez-Arcaya et al., 2017)	Sampling of different geomorphological units within canyons. Huang et al. (2014) provide a classification system for canyons that may be a useful means of standardizing definitions of canyon types.
	Active vs inactive canyons (sensu Bernhardt et al., 2015)		
	Large volume vs small volume canyons (Sensu Huang et al., 2014)		
	Canyon head (shelf incision) and mouth (deposition lobes on abyssal plain)		
Trenches	Latitude/Depth	Major influence of productivity and deep water mass influence, on trench biogeography and community structure, connectivity aspects included (Jamieson et al., 2010)	Can be captured with latitude – based stratification guidance and by considering trenches' distance from the continents and from each-others
	Overlying Productivity	Fundamental source of nutrient supply	
	Hard vs soft substrates	Unexpected habitat heterogeneity with diverse topographical features within the trenches (e.g., ridges, cold seeps, sedimentary ponds)	Sampling of different geomorphological units within the trenches, along bathymetric transects, and along trenches axis to ensure capture of all topographic features and associated communities through the sampling effort
	Trench topography (e.g., flanks, bottom)	Different organic resources supply in both intensity and frequency mainly depending on trench hydrodynamics (deep-sea currents), and earthquakes and volcanic eruptions (gravity-driven landslides phenomena). Different level of natural disturbance within the trench system (Jamieson et al., 2010).	
Abyssal Plains	Nodule cover	Nodule mining	Stratify across nodule cover
	Topographic setting	Abyssal plains often feature numerous, but small, abyssal hills and troughs as part of the ecosystem	Sample in both topographically complex versus simple settings. Could use the global design substrate-based stratification guidance in this context.
	Distance to bathyal depths	Will permit testing of bathyal/source - abyssal/sink hypothesis	Use the global design distance-based stratification guidance in this context.
	Productivity in space and in time	Productivity derived from surface, terrestrial inputs, organic falls, etc., determines interannual carbon flow variations	Cover areas with different productivity regimes (e.g., oligotrophic vs. eutrophic). May be considered as part of the global design latitudinal based stratification guidance in this context.
Slopes	Island vs Continental;	Terrestrial influence (manifested on continental margins, and to a lesser extent on islands) modifies slope communities (Levin et al., 2001)	Ensure inclusion of continental and island slopes.
	Upwelling regime	Upwelling regimes affect oxygenation and food supply to the slope, modifying densities, diversity, lifestyles, body size and ecosystem function (Rogers, 2000)	Include Eastern boundary upwelling areas and western boundary current regions in N. and S. hemisphere
Seeps	On vs off seep	Seeps are unique chemosynthetic ecosystems (Levin et al., 2016a)	Stratify by on vs off seep
	Hard vs Soft substrates	Different communities develop in seep sediments than on authigenic carbonates (Levin et al., 2016b)	Sample seeps dominated by hard and soft substrates. Could use the global design substrate-based stratification guidance in this context.

Continued

TABLE 1 | Continued.

Ecosystem type	Stratification variable	Importance of stratification variable	Recommendation
	Depth	Strong depth zonation for seep megafaunal taxa (mussels, tubeworms, clams) (Rodrigues et al., 2013; Levin et al., 2016a)	Follow guidance in main design.
Hydrothermal vents	Vent biogeography	They have their own biogeographic provinces (about 11 after Rogers et al., 2012) linked to spreading rate, ocean basin and connectivity.	Stratify by vent biogeography rather than GOODS biogeography.
	Active / inactive hydrothermal systems	Seafloor Massive Sulfide mining may target inactive hydrothermal deposits, but known deposits so far are in proximity to active vent fields (exploration leases include active areas) (Van Dover et al., 2018; Van Dover, 2019)	Sample the continuum from active vents out to "dormant" sites.
	Mid Ocean Ridges vs Back Arc Basins, volcano flanks, seamount summits and serpentine-hosted systems (including Lost city type)	Geological diversity and natural instability (eruption, seismicity) are predominant drivers of vent biodiversity (Ramirez-Llodra et al., 2007)	Sample on different geological settings with contrasted environmental properties (acidic to alkaline pH, metal-rich or depleted, different energy sources for chemosynthesis H_2/CH_4/H_2S/FeII and others to be discovered)
	Proximity to the nearest vent field	Peripheral non-vent fauna benefit from local primary production has received less attention, despite its importance for regional biogeography and regarding ecosystem services (Van Dover, 2019)	Stratify by distance to nearest vent field
	Axis / off axis vent deposits on Mid Ocean Ridges	Relates to geological age (Beaulieu et al., 2015)	Stratify by on axis/off axis
Ice-covered ocean	Seasonal sea ice, cover by multiyear ice, or ice shelf	Ice cover determines the quantity and quality of primary production reaching the deep-sea floor. Permanent/ ice age old ice shelf may cover very rarely studied /undiscovered ecosystems (Beaulieu et al., 2015; Boetius et al., 2015).	Stratify according to ice cover.

Ecosystem types conform to geomorphological classes presented in Harris et al. (2014) where applicable.

and affordable approach. Levin et al.'s (2019) and **Table 2** provides a list of biological and ecosystems Essential Ocean Variables, including new EOVs proposed by the Deep Ocean Observing Strategy (DOOS). Here, we align these three study recommendations on what to measure and how (**Table 2**) to provide advance understanding of deep-sea ecosystems and form a basis for further discussion on this topic under a global program.

Physical Specimen Sampling Needs

Importantly many of the measures identified in **Table 2** require the collection of physical specimens. Biodiversity measures (KSQi) require physical specimens for biomass measurements, and for morphological and genetic analysis (DNA barcoding) to confirm organism identification and resolve taxonomy and phylogeny unambiguously. Connectivity studies (KSQii) specifically population connectivity require physical specimens for microsatellite, AFLP, or NGS studies, as well as reproductive studies such as fecundity, reproductive mode and timing of spawning. Trophic studies require samples for stomach content, fatty acid, pigment, stable isotopes, and potentially eDNA analyses. Further questions around ecosystem function (KSQiii) require physical specimens to quantify physiological processes and ecosystem services such as carbon sequestration, and to determine biological traits such as growth rates, longevity, age and size at first maturity, population size structure, and length/weight relationships. Earlier efforts to develop a functional traits database for vent species highlighted how few life-history traits could be assembled for all species (Chapman et al., 2019). Impact and risk assessment measures (KSQiv) require physical specimens for analyses of contamination such as microplastics, particulate or dissolved metals, hydrocarbon exposure, and legacy and emergent persistent organic pollutants. We propose that targeted physical specimen sampling form an important part of a global program. Coordinated and targeted physical specimen sampling efforts, and development of processing pipelines that include access to experts and effective archiving require further consideration.

DATA AND SPECIMEN ARCHIVING

In addition to the fundamental need for standardization of measures, collection and processing methods, data accessibility following collection also remains a challenge. O1 and SO6 (**Figure 1**) consider a transformative increase in ocean knowledge that include data and specimen archiving as part of the desire to expand, innovate and integrate knowledge in global systems (O2). We expect that this information will enhance understanding and prediction of the global oceans (O3) as part of the interconnected system and the need to develop a decision support system (O4). Critical elements include data storage, and ensuring it follows both the Findable, Accessible, Interoperable, and Reusable (FAIR) principle and the principles of Collective benefit, Authority to control, Responsibility and Ethics (CARE). These principles

TABLE 2 | Woodall et al. (2018); Levin et al. (2019), and Danovaro et al. (2020) recommendations for what to measure and how, aligned for equivalency, and supplemented by additional considerations.

Scientific Area as defined in Danovaro et al. (2020)	Primary Components as defined in Woodall et al. (2018)	Essential ecological variables as defined in Danovaro et al. (2020)	Detail of priority measurements recommended by all three papers and supplemented with new parameters.	Data acquired inform which Societal Outcomes (SOs) and key scientific questions (KSQ's) (see Section 2).
Biodiversity: Water column components	Biological parameters: Pelagic	Macro and meso zooplankton	Provided in Woodall et al. (2018) under the following headings in **Table 1**. (1) Size structure and species composition of mesozooplankton, pelagic micronekton, and pelagic nekton (fish abundance and distribution – Levin et al., 2019) (2) Acoustic sensing of water column biomass (3) Size structure and abundance of gelatinous zooplankton (4) Microbial community (5) Census of associated biota Levin et al. (2019) add zooplankton biomass and diversity We add genetic/genomic diversity.	SOs: 2, 3 KSQs: i
Biodiversity: Sediments components	Biological parameters: Benthic	Macro- and megafauna	Provided in Woodall et al. (2018) under the following headings in **Table 1**. (6) Deepwater hyperbenthos (7) Mesophotic hyperbenthos (8) Epibenthos [including hard coral cover and composition and benthic invertebrate abundance and distribution (Levin et al., 2019)] (9) Infauna including benthic invertebrate abundance and distribution (Levin et al., 2019) Levin et al. (2019) also add microbial biomass and diversity. We add genetic/genomic diversity.	SOs: 2, 3 KSQs: i
Not covered	Environmental drivers	Not covered	Provided in Woodall et al. (2018) under the following headings in **Table 1** (10) Bathymetry and Seafloor morphology (11) Seafloor composition (substrate type) (12) Current velocity (13) Temperature, salinity, pressure (derived density) (CTD) (14) Nitrate/nitrite (NO3, NO2), silicate (SiO4), and phosphate (PO4) (15) Dissolved oxygen (DO) (16) pH (17) Dissolved inorganic carbon (DIC) and total alkalinity (TA) Although other programs under the Ocean Decade, for example GOOS and Seabed2030 will measure physical properties of the ocean, a deep-sea biology program should make *in situ* physical measurements to accompany biological data.	SOs: 1, 2, 3, 5 KSQs: i, ii, iii, iv
Ecosystem functions	Not covered	Trophic structure	Requires collection of physical specimens for dietary analyses.	SOs: 2, 3, 5 KSQs: ii, iii
Ecosystem functions	Not covered	Benthic faunal biomass	Guidance provided in Danovaro et al. (2020) and **Table 1**. Bio-volume estimates (for example, class size frequencies from individuals' body lengths) We add direct measurements of biomass.	SOs: 2, 3, 5 KSQs: i, iii
Impact/risk assessment	Sociocultural parameters and impacts	Habitat damage	Provided in Woodall et al. (2018) under the following headings in **Table 1**. (19) Records of litter and anthropogenic damage (20) Microplastic abundance and diversity Guidance provided in Danovaro et al. (2020), **Table 1**. The analysis of seascapes changes based on habitat mapping approaches and georeferenced photomosaic compositions Levin et al. (2019) add ocean sound No study considers other particulate or dissolved chemicals such as metals, legacy and emergent persistent organic pollutants (POPs).	SOs: 1, 2, 3, 5 KSQs: iv

Continued

TABLE 2 | Continued.

Scientific Area as defined in Danovaro et al. (2020)	Primary Components as defined in Woodall et al. (2018)	Essential ecological variables as defined in Danovaro et al. (2020)	Detail of priority measurements recommended by all three papers and supplemented with new parameters.	Data acquired inform which Societal Outcomes (SOs) and key scientific questions (KSQ's) (see Section 2).
Impact/risk assessment	Not covered	Recovery rate (as a proxy of resilience)	Time-series data required. Guidance provided in Danovaro et al. (2020), **Table 1**. Multivariate analysis time-series counts for species depicting fluctuations according to concomitant oscillations of key environmental drivers (for example, temperature and oxygen maxima and minima). We add data on growth rates, longevity, fecundity, reproduction, recruitment rates, size at first maturity (puberty), maximum body size, dispersal.	SOs: 2, 3, 5 KSQs: iv (and may draw on i, ii, iii)
Global change, adaptation and evolution	Not covered	Shifts in bathymetric distribution	Time-series biodiversity data required. See Biodiversity / Biological parameters	SOs: 2, 3, 5 KSQs: i, ii, iv
Global change, adaptation and evolution	Not covered	Local extinctions	Time-series biodiversity data required. See Biodiversity / Biological parameters	SOs: 2, 3, 5 KSQs: i, ii, iv

Woodall et al. (2018) provide further detail on how measurements should be made to ensure consistency in collections which we will not repeat here, but direct others to read.

apply not only to digital products, but also to physical specimens, which all form the foundation of repeatability in science.

The IOC, together with the Center for the Fourth Industrial Revolution: Ocean under the Ocean Data Platform[1], are developing an Ocean Data and Information System to improve significantly the availability of ocean data and information, and to enable open source products and services catered to the needs of a broad community of users, including academia and ocean managers. This, together with existing UN supported initiatives such as OBIS, will provide the means to archive data collected under a global program. However, whether this initiative includes provision for open sharing of specimens as well as data remains unclear. Participants in a global program should commit to the open sharing of specimens as well as data, including deposition of specimens with an established museum, an institution with a recognized charter that supports both the permanent storage and care of archive specimens, and access to those specimens by the scientific community. Natural history museums facilitate loans to enable the global community to utilize specimens in their care. Nonetheless, any material collected within the territorial waters of a specific nation falls under the purview of the Convention on Biological Diversity and particularly the Nagoya Protocol and its access and benefit sharing rules. A framework governing access and benefit sharing for marine genetic resources in areas beyond national jurisdiction (ABNJ) is in discussion at the UN (The International Legally Binding Instrument for the Conservation and Sustainable Use of the Biological Diversity of ABNJ; Wright et al., 2018). Both of these legal frameworks require traceability of biological specimens and any digital sequence information from the origin of the samples through subsequent uses, especially if used commercially (Rabone et al., 2019). This traceability means that a unique identifier for samples taken at sea may become an important requirement both for tracing the use of samples, but also for linking specimens or digital sequence information to sample locations and their associated metadata.

Rapid technological change over the coming decade will parallel accelerating species loss. Consequently, a standardized repository approach, to archive or 'bank' frozen specimens, tissue samples, and specimens fixed for morphological visualization, will enable scientists to address future questions not yet envisaged or for which technology does not yet exist (e.g., regarding functions). Museum specimens represent the pinnacle of sustainable science: material collected now will be used by future generations another 150 years in the future, just as we use the original specimens collected on the *Challenger* Expedition, the birth place of deep-sea science, 150 years ago. Guiding principles in both data and specimen archiving should include rapid accession and minimal embargo, and a commitment to collect broadly. Collection and archiving of data and specimens should not only serve the goals of an individual project, but prepare for synergistic and unpredictable future uses.

BUILDING CAPACITY AND PROMOTING KNOWLEDGE EXCHANGE IN DEEP-SEA RESEARCH

Objective 1 of the Ocean Decade (**Figure 1**) focuses on increasing capacity to generate, understand, manage, and use ocean knowledge. This objective has particular relevance for deep-sea research. While more than 70% of countries have a deep-sea environment within their EEZ, economically developed nations (sensu UN Department of Economic and Social Affairs) conduct most deep-sea research. Availability of samples, bias in available

[1] https://www.oceandata.earth/

data, and overall knowledge of deep-sea ecosystems all reflect this bias. Countries with developing economies face significant barriers to participating in deep-sea research, including access to technological capability and infrastructure, and specific expertise. Yet the least studied parts of the deep sea often occur within the EEZs of less economically developed nations. A global assessment of capacity development needs in ocean science was undertaken by the UN through a series of regional workshops between 2011 and 2013. This assessment highlighted particular needs for capacity building in deep-sea research (Ruwa et al., 2016). In 2015, the Intergovernmental Oceanographic Commission Assembly adopted its Capacity Development Strategy for 2015–2021 (IOC-UNESCO, 2016), identifying six high-level outputs to address on a long-term and sustained basis:

(1) Human resources developed
(2) Access to physical infrastructure established or improved
(3) Global, regional and sub-regional mechanisms strengthened
(4) Development of ocean research policies in support of sustainable development objectives promoted
(5) Visibility and awareness increased
(6) Sustained (long-term) resource mobilization reinforced.

A global program should aspire to contribute to the Ocean Decade O1 by committing to core principles of effective research capacity sharing and building (e.g., Hind et al., 2015):

(1) *Co-development and co-creation of contributing regional research projects*. To ensure a truly global and inclusive program, the community should consider their proposed research region, and actively seek early engagement with other region-based collaborators to facilitate co-design and development of research plans and funding applications. Particularly in the case of Small Island Developing States (SIDS), a.k.a. Large Ocean States, local/indigenous methodologies and epistemologies have great potential in ocean observations from under sampled locations as well as knowledge production. Many countries increasingly focus on applying traditional ecological knowledge to coastal and shallow marine research, but less so in deep-sea research. We propose that capacity sharing and building actions actively invest in and support diverse practitioners to pursue deep-sea research. A new generation of deep-sea scientists, from a more diverse geographic pool, would bring new perspectives and approaches to research in the open ocean. In addition, local/indigenous knowledge systems linked to the deep ocean should be given a voice and considered alongside natural science in evolving deep sea exploration targets and management.
(2) *Investment in training for scientists from economically developing countries.* Previous studies have identified the crucial need to develop human capacity for oceanographic research in economically developing countries, as well as examples of activities that can achieve this objective (Morrison et al., 2013; Miloslavich et al., 2018b). Research projects contributing to a global program should, where possible, include a budget for full participation of regional partners in ship-board training activities, as well as knowledge-exchange and networking activities, e.g., conference and meeting attendance (Stefanoudis et al., 2020). Projects should also consider small investments in local research infrastructure that may enable long-term data collection (see new technologies section below), allowing local researchers to continue producing data beyond specific projects (Hind et al., 2015).
(3) *Sharing research products.* Projects should assign time and resources for co-analysis of data and dissemination of research outcomes with regional partners. We also strongly recommend open access publication of research, a small but valuable step in international engagement under a global program.

THE POTENTIAL ROLE OF NEW TECHNOLOGY

Technological development forms the nexus in our ability to identify and generate ocean data, information and knowledge (**Figure 1**, O2). Access to technology is one of the barriers to broadening participation in deep-sea research ((**Figure 1**, O1). The revised Roadmap for the Ocean Decade outlines the potential role for new technologies in helping researchers better measure biodiversity, functions of deep-sea ecosystems and cumulative impacts of ocean stressors, and define the carrying capacity of ocean ecosystems to sustain human impacts and economic development. Anticipated developments over the decade largely grouped into three areas: improved access to the oceans, in terms of exploration (spatial coverage), variability (temporal monitoring) and costs (low cost technology); improved extraction of information from observations including automated data processing; and democratization of the sharing of both data and knowledge obtained.

Data Acquisition

Historically, access to the deep ocean has been both expensive and sparse, reliant upon access to large ocean-going ships and deep submergence research assets (both human occupied and robotic) operated by relatively few nations and often in regions far away from the largest of Earth's ocean basins. Moving beyond satellite-based remote sensing, other fields of ocean research have advanced the use of new technologies that enable larger-scale coverage for global-scale ocean investigations. For example, recently developed biogeochemical (BGC) ARGO floats extend the capabilities of the ARGO array to measure important parameters in the deep ocean including pH, oxygen, chlorophyll, nitrate, suspended particles and downwelling irradiance. Simultaneously, the development of autonomous surface vessels such as the sail drone have enabled completion of demonstration projects that couple persistent presence at the ocean surface with remote sensing satellite data to guide vehicles (equipped with suitable instrument payloads) to ground-truth observations from remote sensing data. Under the Ocean Decade, judicious use of deep gliders and long-range AUVs, reporting back to shore-based scientists via autonomous surface vessels, could begin to conduct first-pass exploration of remote portions of the deep ocean floor (German et al., 2012). Even this modest contribution could immediately begin to improve the efficiency and efficacy with which the science community can deploy the most expensive assets among the international research community (global-class research ships and deep diving submersible assets).

Although recent technological advances leave no part of the world's deep ocean out of reach, the capacity to deploy those assets remains limited worldwide. The extreme expense associated with buying, operating and maintaining large-scale oceanographic infrastructure, including ships and deep - submergence facilities pushes them out of the reach of most developing countries. However, satellite-enabled telepresence has enabled many thousands of individuals across the planet to join in discoveries and investigations in the deep ocean in real time but only if they have internet access. Partnerships between research institutes, universities, museums, and aquaria that can offer free access to the video, annotation, and scientists themselves during an expedition could augment this access. However, broader participation in this discovery field clearly requires the development of low-cost and smaller technologies to apply to deep-sea research (Phillips et al., 2019).

Data Extraction

Extraction of information from acquired data also offers ripe opportunity for technological advancement over the Decade. Miniaturization and increasingly lower power requirements for *in situ* sensors for multiple environmental parameters enable installation on an increasing number of platforms obviating the need for laborious analyses and increasing coverage in the oceans, both in terms of spatial coverage and temporal monitoring. In parallel, non-invasive identification of species through genetic samples via environmental DNA (eDNA) sequencing provides an important, portable, and non-invasive technique enabled by emerging sequencing technology (Mariani et al., 2019). The approach offers an exciting opportunity to quickly analyze the diversity of fauna present within any given environment, although lack of an effective reference library to identify sequences by comparisons constrains application of the approach to the deep-sea biota (Howell et al., 2019). Many deep-sea eukaryotic species have never been sequenced before, or may be species new to science, and eDNA therefore cannot yet offer definitive identification. An emphasis on well-curated physical specimens entrusted to museums and the principles of open data sharing will overcome this bottleneck to identifying many deep-sea species.

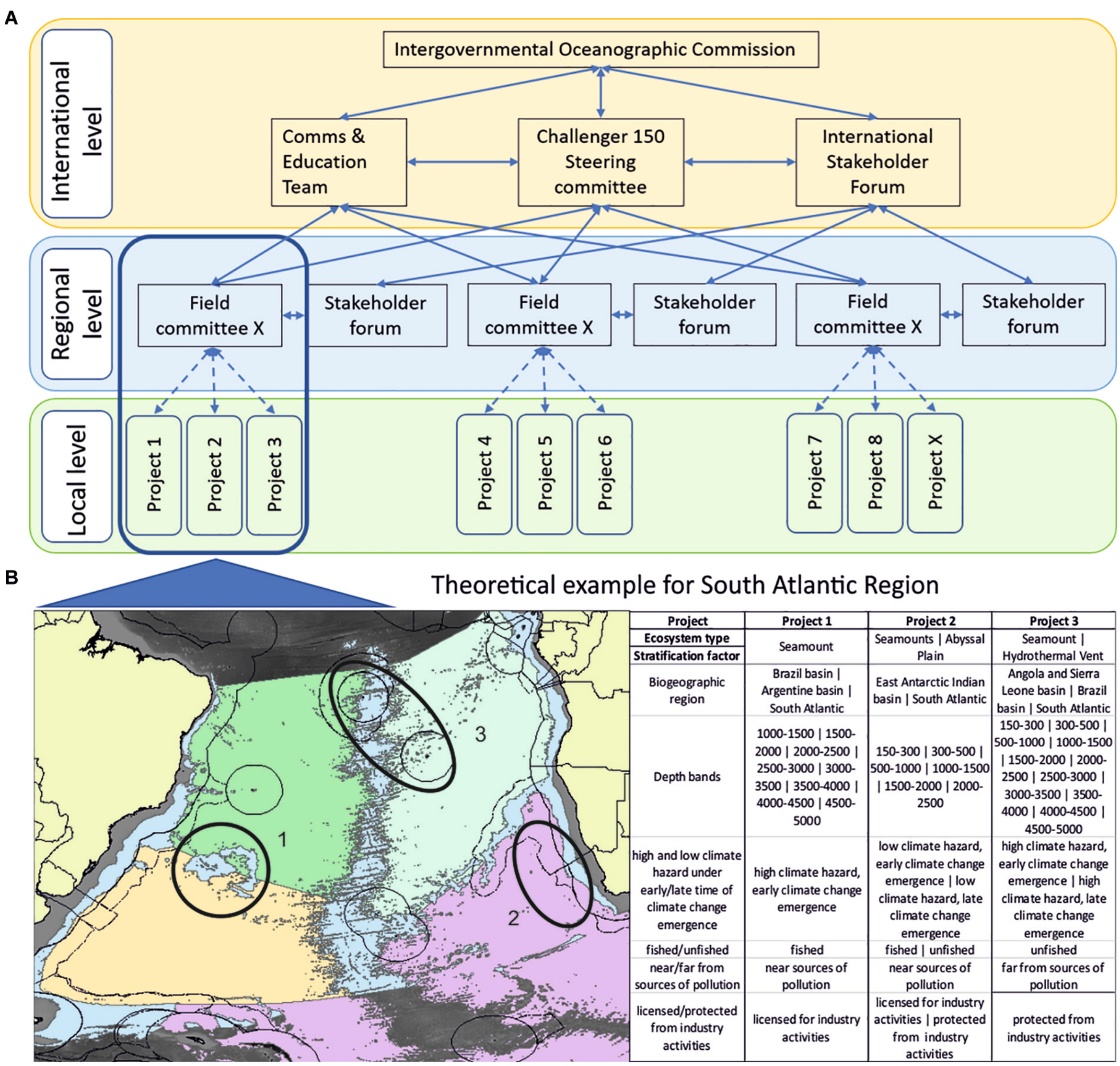
In parallel, the increasing volumes of image and video data we anticipate scientists will acquire over the decade, not just from conventional methods but from expanded use of ship-free deep-ocean robotic assets will require a comparable ramp-up in the throughput of image analysis, and highlight a need for software solutions to improve pipeline processing and automatic analysis. Howell et al. (2019) identify the requirement for manual image analysis as a significant bottleneck in image-based marine ecological survey and monitoring, and that artificial intelligence (AI) and computer vision (CV) offer a potential means by which to both accelerate and standardize the interpretation of ecological image-based data (Piechaud et al., 2019). However, significant barriers to further development of these methods remain, including the lack of a standard morphospecies reference image catalog against which to base identifications and annotations. Such a catalog is in development for deep-sea fauna (Howell et al., 2019), and other potentially useful classification schemes already exist (Althaus et al., 2015). The deep-sea research community should prioritize agreement on a standard approach in order to expedite the use of AI and CV.

Technological advance will form a significant aspect of the Ocean Decade, and a global program should seek to contribute to, and benefit from, these developments in progressing toward the achieving the SOs.

DISCUSSION

In this paper, we have reviewed research needs of the Ocean Decade in the context of the design of a new 10 year deep-sea biology research program. This paper offers a blueprint for the further development of a global program as an official 'Action' of the Ocean Decade that we name here *Challenger 150*. Scientists and the public alike associate the name "Challenger" with exploration of new frontiers. One hundred and fifty years ago '*HMS Challenger*' spent 4 years circumnavigating the globe, mapping the seafloor, recording the global ocean temperature, and providing us with a first panoramic view of life in the deep sea. Rightly scientists now attribute the birth of deep-sea biology and oceanography to *HMS Challenger*. The NASA space program later used the same name, first for the Apollo 17 Lunar Module that landed on the Moon in 1972, and later forthe space shuttle *Challenger* that flew the first American woman, African-American, Dutchman and Canadian into space. Today the name describes the deepest point of the ocean, the bottom of the Mariana Trench where, in January 1960 Jacques Piccard and Don Walsh made the first human descent to the Challenger Deep in the bathyscaphe Trieste. More recently, in March 2012 film director James Cameron made the first solo descent in the deep-submergence vehicle *Deepsea Challenger*. We use the name Challenger here to invoke the same spirit of exploration embodied by the previous Challengers, and to recognize the importance of that first global deep-sea biological dataset that, for some parts of the ocean, remain the only data available. However, we fully acknowledge that past exploration involved colonialism and exclusion. We advocate that in keeping with the Ocean Decade objectives *Challenger 150* should forge a new inclusive, representative and equitable face for an historic name.

We present *Challenger 150* as a concept for an Ocean Decade 'Program-level Action' as defined in the draft Implementation Plan. It would serve as a community-led collaborative endeavor in the stepwise development of a coherent, well-designed, deep-sea global survey and monitoring program. The concept would be realized through individual research projects committing to align with the blueprint presented here, and in so doing, becoming a piece in a larger global jigsaw puzzle. Such a program would require an unprecedented level of communication and coordination between research projects over the course of the Ocean Decade, and thus the program would require an effective management structure established to coordinate with, and support the community to follow the outlined design criteria regionally, annually reviewing progress toward the overall global design and Ocean Decade SOs and Os (**Figure 2**). We envision a process where-by individual projects, supported by a diverse range of funders, will formally align with the Challenger 150 program and the recommendations within this text. Lead PIs of projects from the same region, together with other relevant regional researchers will form a regional field committee (**Figure 2A**) to coordinate and monitor fieldwork efforts at that scale, and to support regional teams to develop new projects to fill survey gaps over the course of the Ocean Decade (**Figure 2B**). They will also interact with a regionally relevant stakeholder pool to ensure field projects complement research occurring within other disciplines, and remain aligned with end-user needs. Development of these regionally relevant stakeholder forums will draw upon existing regionally relevant bodies, for example the Second International Indian Ocean Expedition in the Indian Ocean, or the Benguela Current Commission in the South East Atlantic. Representative membership of each regional field committee will sit on the Challenger 150 steering committee to ensure coordination of field projects at a global scale, and monitor progress against the program aims and global survey design. The steering committee will report on progress to the IOC. They will also interact with a global stakeholder pool specific to the program, or possibly at the



Project	Project 1	Project 2	Project 3
Ecosystem type Stratification factor	Seamount	Seamounts | Abyssal Plain	Seamount | Hydrothermal Vent
Biogeographic region	Brazil basin | Argentine basin | South Atlantic	East Antarctic Indian basin | South Atlantic	Angola and Sierra Leone basin | Brazil basin | South Atlantic
Depth bands	1000-1500 | 1500-2000 | 2000-2500 | 2500-3000 | 3000-3500 | 3500-4000 | 4000-4500 | 4500-5000	150-300 | 300-500 | 500-1000 | 1000-1500 | 1500-2000 | 2000-2500	150-300 | 300-500 | 500-1000 | 1000-1500 | 1500-2000 | 2000-2500 | 2500-3000 | 3000-3500 | 3500-4000 | 4000-4500 | 4500-5000
high and low climate hazard under early/late time of climate change emergence	high climate hazard, early climate change emergence	low climate hazard, early climate change emergence | low climate hazard, late climate change emergence	high climate hazard, early climate change emergence | high climate hazard, late climate change emergence
fished/unfished	fished	fished | unfished	unfished
near/far from sources of pollution	near sources of pollution	near sources of pollution	far from sources of pollution
licensed/protected from industry activities	licensed for industry activities	licensed for industry activities | protected from industry activities	protected from industry activities

FIGURE 2 | A framework for realizing the Challenger 150 Ocean Decade 'Program-level Action' concept, using the South Atlantic as an example region. **(A)** A proposed management structure for the program. **(B)** An example of how different projects operating within a region will be monitored against the global design and gaps in survey coverage identified.

level of the IOC, where a global stakeholder pool may interact with multiple Ocean Decade programs. A communications and education team will coordinate activities between projects, and liaise with the IOC's communications and education team, to ensure joined up efforts between different Ocean Decade programs. Participating research projects would be required to commit to use of new knowledge, as it accumulates, to address the Ocean Decade SOs and Os. By the end of the Ocean Decade the data from all projects could be combined to provide 'one giant decadal leap' for human knowledge of the deep ocean. Through this approach, contributing projects could potentially make a major contribution to both the Ocean Decade (**Table 3**) and SDG14, that simply could not be achieved working in isolation.

The revised Roadmap for the Ocean Decade calls upon the scientific community to think beyond business as usual and aspire for real change in the level of knowledge of the ocean in support of sustainable development. The implementation plan foresees a funding resource base that is multi-actor in nature, broad and flexible, taking a variety of forms. As we stand on the brink of the Ocean Decade, the *Challenger 150* concept seeks to coordinate global research efforts in deep-sea ecology and oceanography for the express purpose of contributing to the delivery of the Ocean Decade SOs. However, success depends

TABLE 3 | What an inclusive global program could deliver against each of the relevant Ocean Decade societal outcomes and by inference SDG14.

Societal Outcome	Deliverable	Comments
1	A comprehensive knowledge of the distribution, abundance and diversity of contaminants in our ocean including litter, microplastics, dissolved metals, hydrocarbons, legacy and emergent persistent organic pollutants.	These data can be used to identify sources and pathways of pollution, and inform action to reduce the level of contaminants entering the Ocean.
2	New mapping data including bathymetry, environmental data, species and habitat distributions, ecosystem functions, and human impacts. Knowledge of the individual and cumulative effects of different anthropogenic pressures including climate change. An established global network of sites where we can monitor climate change. New knowledge of the roles of deep-sea species and habitats in delivering ecosystem goods and services.	These deliverables can be used to inform decision-making, through the integration of updated biological and ecological knowledge within marine spatial planning processes to guide a sustainable blue economy. This could include use of ecosystem-based management approaches, including the establishment and designation of area-based management tools (e.g., Ross and Howell, 2013). Both the spatial footprint and cumulative impacts of anthropogenic and climate-derived pressures on species, habitats, ecosystem functions, services and goods will be better measured. These measurements will (1) inform policy decisions that reduce human impacts and/or ensure ecosystem services are maintained; and (2) support estimates of the natural capital the deep sea provides to societies and their wellbeing, and so help reveal different trade-offs faced in different use scenarios, informing policy decisions on ocean management.
3	New data on species and habitat densities, distributions, ranges, environmental drivers and tolerances. An established global network of sites through which we can measure and monitor change. New data on abundance, diversity, biomass, size structure, fecundity, growth rates, dispersal, longevity, nutrient uptake, respiration rates, and diets of species. New data quantifying human impacts on abundance, biomass, diversity, and community composition. New knowledge of the roles of deep-sea species and habitats in delivering ecosystem goods and services.	These data, together with those collected under other Ocean Decade programs (e.g., Seabed 2030, physical oceanographic programs) will facilitate the use of various modeling approaches to predict current and future conditions, for example current patterns of, and shifts in, the distribution of species and communities (Howell et al., 2016; Brito-Morales et al., 2020; Morato et al., 2020) under climate change, changes in fishing effort, onset of mining activities, etc. The establishment of appropriate monitoring sites will provide temporal data with which to train temporal models, forecast into the future with greater confidence, and disentangle climate-related change from changes caused by other human activities. Coupled with new knowledge of the role of species and habitats in delivering ecosystem goods and services, this will support further predictions of what modeled changes mean for human wellbeing and livelihoods.
5	New data on the abundance, density, biomass, size structure, fecundity, growth rates, dispersal, longevity, connectivity, respiration rates, diets, and habitat preferences of species including commercially harvested species.	As with SO3 these new data will facilitate the use of modeling approaches to predict ecological responses to fishing pressure (Howell et al., 2009; Heymans et al., 2011), as well as cumulative pressures to support policy decisions around sustainable harvesting. Coupled with socio-economic research these data can be used to inform scenario modeling around provision of food supply and livelihoods.
6	New data and specimens appropriately archived and openly accessible A larger number of nations actively engaged in deep-sea research Improved deep-sea ocean literacy	The global nature of the research described herein will take place in a wide variety of state-controlled waters as well as areas beyond national jurisdiction (ABNJ, or The Area). New consortia formed to address each biogeographic region, composed of/or at least including researchers and students from respective regions, will help to train a new generation of deep-sea scientists as part of a global community. The data generated will be included in the UN designated open-access databases as a product of the Ocean Decade, to support Ocean management and decision-making around the world. Physical specimens will be archived and accessible in relevant regional natural history museums.

upon the nations and their research communities mobilizing and obtaining funding to support such efforts. Ship time represents a significant cost, and the only realistic option for some nations will be to seek public-private or philanthropic partnerships such as the REV Ocean, Schmidt Ocean Institute, Ocean Exploration Trust, Nekton Foundation, OceanX, The International Seakeepers Society, and industry to provide access to appropriate platforms. Industry could play a particularly important role to play in less economically developed countries. The IOC and REV Ocean have already agreed on several areas of collaboration under the Ocean Decade, including use of the REV Ocean vessel, offering a real opportunity to advance the *Challenger 150* concept in the identified priority areas for new biological data collection. For other nations who can access large infrastructure, national, regional and bilateral funding mechanisms may be more appropriate or accessible as a means to fund contributing projects.

Although data collection represents a challenge, data processing, interpretation, archiving and storage represent a significant and on-going cost that at present can only be met by multiple applications for funding from national research budgets or philanthropic mechanisms. Regardless of where scientists apply for funding, alignment of projects with the blueprint outlined here as part of a community led, globally coordinated 10 year program brings greater opportunity for both efficiency and impact. Therefore projects aligned with the *Challenger 150* concept may appear more attractive to funders thus offering benefits to the wider community.

The Ocean Decade begins on the 1st January 2021 and already the deep-sea research community has begun to sow the first seeds of coordination between projects to contribute to a global program via the Deep Ocean Stewardship Initiative's (DOSI) Decade of Ocean Science working group and the Scientific Committee on Oceanic Research (SCOR) working group 159. We hope that the blueprint provided in this paper helps the wider deep-sea community to engage with the *Challenger 150* concept as a shared endeavor; to forge regional and inclusive consortia, co-develop research plans and funding bids aligned with the blueprint, and help achieve the SOs and Os of the Ocean Decade.

Summary of Recommendations

We Propose a Global Program

(1) uses the revised GOODS classification (Watling et al., 2013) and Sutton et al. (2017) mesopelagic ecoregions to stratify respectively benthic and pelagic deep-sea survey and monitoring, ensuring an overall stratification by latitude.
(2) uses the following indicative depth horizons as a general guide for a target range for all biogeographic regions to achieve an unbiased dataset: 150 — 300, 300 — 500 then every 500 m to the deepest point of the oceans at 10 km.
(3) uses the following indicative horizontal distances as a general guide for component projects to stratify sampling by: 1 m, 10 m, 100 m, 1 km, 10 km, 100 km.
(4) uses the following replicated treatments in regional designs where possible: high and low climate hazard under early/late time of climate change emergence, fished/unfished, near/far from sources of pollution, licensed/protected from industry activities.
(5) stratifies sampling by substrate type and / or topography, including slope, within regional designs.
(6) uses the following criteria in selection of sites for potential monitoring: access to different strata outlined in the global design, availability of existing observing infrastructure, opportunity for and ease of installing and maintaining new infrastructure.
(7) prioritizes research effort in southern and polar latitudes, deeper depths, and midwater environments.
(8) considers additional ecosystem-specific stratification in addition to those of the main design (**Table 1**).
(9) uses **Table 2**, and the papers cited within, to provide guidance on what to measure and how in order that the data can be used to help deliver the SOs; and visit the OBPS digital repository at oceanbestpractices.org for more specific guidance.
(10) ensures that targeted physical specimen sampling form an important part of the program.
(11) follows the following guiding principles in both data and specimen archiving: rapid accession and minimal embargo, a commitment to collect broadly, FAIR, CARE.
(12) commits to and provide for the deposition of specimens with an established regionally relevant museum.
(13) commits to core principles of effective research capacity sharing and building, including engagement with local and indigenous communities.
(14) seeks to contribute to, and benefit from, technological developments in progressing toward the achieving the SOs.

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KH and AH conceived the idea and convened the working group. KH, AH, AM, LL, ER-L, MB, PVRS, JX, HW, MC, EE, CG, SR, PE, and LM helped to conceive the manuscript and outline structure. All authors contributed ideas, text, and edits.

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Multifunctionality of an Urbanized Coastal Marine Ecosystem

Anand Archana[1] and David Michael Baker[2]*

[1] Department of Civil and Environmental Engineering, Stanford University, Stanford, CA, United States, [2] School of Biological Sciences, University of Hong Kong, Pok Fu Lam, Hong Kong

Coastal marine ecosystems provide numerous invaluable services and benefits to humankind. However, urbanization of coastal areas has homogenized and reduced the biodiversity of the surrounding marine environment and the sustainability of the multiple ecosystem services it provides. Studies have focused on single ecosystem functions despite human populations relying on several functions being delivered at once (known as multifunctionality). This study investigates five ecosystem functions (primary productivity, herbivory, predation, organic matter decomposition and carbon sequestration) and overall multifunctionality in four sites along a gradient of 16 environmental parameters. Ecosystem function varied significantly between sites that were farthest apart. In determining factors that drove ecosystem functioning, we found a positive relationship between coral cover and primary productivity but negative relationships between coral cover and levels of herbivory and predation intensity. Higher temperatures and greater concentrations of chlorophyll-a had a positive impact on predation and herbivory, respectively. Notably, we found a significant negative impact of total inorganic nitrogen and significant positive impact of total Kjeldahl nitrogen on carbon sequestration. Further, individual functions were compared with fish abundance (obtained from seawater eDNA), and benthic community composition (obtained from plate % coverage of autonomous reef monitoring structures). Increasing herbivorous fish abundance had a positive impact on *Ulva* mass loss. Overall, relative abundance of predatory, omnivorous and planktivorous fish exerted overriding influences on primary productivity and predation intensity, implying that fishing pressure and marine protected area status are important factors. Importantly, we found significant effects from environmental parameters indicating that reliably predicting the effects of future anthropogenic impacts will not be straightforward as multiple drivers are likely to have complex effects. Taken together, urbanized coastal ecosystems exhibit varying levels of multifunctionality depending on the extent of human impact, and the functional diversity of the benthic community present.

Keywords: urban, marine, ecosystem, multifunctionality, coral, eutrophication, coastal

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University of Hull, United Kingdom
Andrew Martin Lohrer,
National Institute of Water
and Atmospheric Research (NIWA),
New Zealand

****Correspondence:***
Anand Archana
archanaa@stanford.edu

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INTRODUCTION

Marine environments are significant contributors to both local and global economies, providing ecosystem services and jobs to local people on the one hand, and representing a great bank of undiscovered species, including many of potential importance for human health, on the other. However, of the many changes underway in the world today, one of the most striking is the decline and homogenization of marine biodiversity through overexploitation, climate change, habitat loss, nutrient pollution and species invasions (Meyer et al., 2016). This raises considerable concern, since human society depends on the ocean for its services and the loss of marine biodiversity has detrimental impacts on the environment. For example, Villnas et al. (2013) demonstrated that repetitive anthropogenic stressors result in the gradual degradation of individual functions in sedimentary ecosystems. Additionally, based on (a) experimental systems and (b) individual functions, Gamfeldt et al. (2015) concluded that loss of marine biodiversity likely decreases ecosystem function and impacts habitat resilience. Specific to coral reefs, Lefcheck et al. (2019) demonstrated that tropical fish diversity significantly impacts functioning and Brandl et al. (2019) observed the same relationship between biodiversity and ecosystem functioning. In the last decade, an increasing number of studies have arrived at similar conclusions about the biodiversity-ecosystem function relationship in natural systems (non-experimental, 'real-world' ecosystems; drylands – Maestre et al., 2012; Berdugo et al., 2017, grasslands – Soliveres et al., 2016, forests – Zhang and Wang, 2012).

Our understanding of ecosystem function, however, is not only far from complete but is also biased toward experiments that have been conducted (1) over a small-scale, (2) with a few species (3) across a few lower trophic levels (Duffy et al., 2007; Nelson et al., 2009; Berdugo et al., 2017) and (4) are biased toward single functions (Stachowicz et al., 2007; Cardinale et al., 2011; Byrnes et al., 2014). In the last decade, efforts have been made to change this trend, by expanding ecosystem function studies across spatial scales (Thompson et al., 2018; Gonzalez et al., 2020). A recent meta-analysis by Duffy et al. (2017) synthesizes these biodiversity effects from multiple ecosystem types. Specifically, Ptacnik et al. (2008); Mora et al. (2011), Zimmerman and Cardinale (2013); Duffy et al. (2015)Duffy et al. (2016), and García-Comas et al. (2016) present measured effects of eelgrass, fish species and plankton diversity on reef ecosystem functioning after statistically controlling for environmental covariables.

Ecosystem function components, when considered synergistically, or additively can have different and likely stronger effects on biodiversity (Hector and Bagchi, 2007; Hautier et al., 2017; Meyer et al., 2018). For instance, Maestre et al. (2012) used a natural experiment to test the biodiversity-ecosystem function relationship (BEF; Duffy, 2009; Gamfeldt and Roger, 2017) between plant species richness and multiple functions in semi-arid ecosystems. This proved to be timely because previous work on dryland systems had only assessed ecosystem functions individually. This led to the concept of a habitat's "multifunctionality" which was defined as the "simultaneous provision of multiple functions" (Hector and Bagchi, 2007; Gamfeldt et al., 2008). Recently, in a study by Byrnes et al. (2014), the authors suggested that aspects of the BEF relationship, which continue to remain unresolved in several marine ecosystems, could be tested stringently when multifunctionality is taken into account and quantified. Taking this forward, Lefcheck et al. (2015) highlighted the importance of taking multiple ecosystem functions into consideration to understand the effect of biodiversity on integrated functioning, an aspect that is unclear when only individual functions are analyzed. By presenting evidence of biodiversity's effect on multifunctionality, the authors demonstrated that communities with higher species richness maintain a higher number of ecosystem functions than those with lower species richness. In this manner, the concept of multifunctionality became useful to understand ecosystem functioning in fundamental and applied ecology.

The measurement of multifunctionality, however, has proven extremely challenging as it takes into consideration only a small subset of all possible individual ecosystem functions. To our knowledge, there is no standardized way of defining or carefully accounting for specific functions that will capture an ecosystem's "true" multifunctionality (Manning et al., 2018). For instance, Alsterberg et al. (2014) illustrated that there were no effects of nutrient enrichment, toxicants, sedimentation and warming on marine ecosystem multifunctionality. However, a few years later, Alsterberg et al. (2017) demonstrated that habitat diversity has a direct effect on marine ecosystem functionality. As a result, data on ecosystem functioning of marine ecosystems are especially limited and fragmented, despite the fact that such information could help inform effective management of crucial environments (Manning et al., 2018).

The last decade has seen a surge in the development of methods to quantify ecosystem multifunctionality (Maestre et al., 2012 – the averaging approach; Gamfeldt et al., 2008; Byrnes et al., 2014 – single and multiple threshold approaches). Manning et al. (2018) introduced two additional components - ecosystem function (EF) multifunctionality and ecosystem services (ES) multifunctionality to help environmental managers and economists quantify the value of ecosystems using monetary units and life satisfaction. Recently, Hoölting et al. (2019) proposed a new way of measuring multifunctionality across spatial scales. Although these methods have been reasonably successful in assessing an ecosystem's performance (Hensel and Silliman, 2013 reviews a few of the methods listed), the literature cited have not yet completely addressed how biodiversity when coupled with environmental factors influences ecosystem function, especially for urbanized coastal marine ecosystems. Therefore, it is crucial to understand the resilience of coastal marine ecosystems in response to global change and environmental stressors.

This is especially relevant to a coastal megacity such as Hong Kong, where eutrophication (nutrient-driven marine pollution) has contributed to a marked reduction in water quality (Duprey et al., 2017) and loss of critical habitats such as hard corals, thus reducing the complexity, diversity and function of benthic ecosystems (Scott, 1990; Fabricius and McCorry, 2006). For example, Tolo Harbour, in northeast Hong Kong was once a pristine, coral-fringed bay home to various coral communities and vibrant fisheries (Morton, 1989). However, several sites in the harbor have experienced a dramatic loss of coral cover from >60% to <10%. In 2015, we utilized stable isotope analysis to understand nitrogen source dynamics in both wastewater effluents and receiving seawaters in the Tolo Channel and found sewage effluent to be the dominant source of nitrogen pollution (Archana et al., 2016, 2018). Consistently, Wong et al. (2017) measured δ^{15}N of hard corals from several sites in this region, which also revealed strong human-derived nutrient signals. Today, we see a punctuated gradient in water quality with nutrient concentrations decreasing with distance from coastal populations. Yet we have little understanding of how these eutrophic conditions affect the ecosystem function of such benthic marine communities and their multifunctionality.

Here, we analyzed ecosystem processes – primary productivity, herbivory, predation, organic matter decomposition, carbon sequestration and overall ecosystem multifunctionality along a gradient of environmental parameters in an urbanized coastal marine environment. We used environmental data (nutrient concentrations, dissolved oxygen, chlorophyll A, pH, salinity, turbidity, secchi depth, temperature, and suspended solids) to test the hypothesis that, ecosystem multifunctionality is significantly correlated with geochemical parameters. Next, we used abundance data of functional groups from annotated settlement plate photos of autonomous reef monitoring structures (ARMS) deployed in the same sites to test the hypothesis that increased abundance of benthic taxa is significantly correlated with ecosystem functioning. We used fish abundance data from eDNA of water samples collected from the same sites to test the hypothesis that increased fish abundance significantly alters ecosystem functioning.

FIGURE 1 | Map showing four study sites in Tolo Harbour (Center Island: CI, Che Lei Pai: CLP, Port Island: PI and Tung Ping Chau: TPC) in Hong Kong's marine environment.

MATERIALS AND METHODS

Study Area

The study was undertaken over a period of 1 year across four field sites within Tolo Harbour, Hong Kong: Center Island – CI, Che Lei Pai – CLP, Port Island – PI, and Tung Ping Chau – TPC (**Figure 1**). These sites represent a gradient in coral cover, CI < CLP < PI < TPC (Duprey et al., 2016; Wong et al., 2017) and several environmental parameters. Center Island is the site closest to the harbor, with almost no hard corals, and is also the most polluted compared to the other sites. Tung Ping Chau, on the other hand is a marine reserve with over 60 species of hard corals. The study used a simple toolkit to allow the quantification of key ecosystem functions and overall multifunctionality.

Primary Productivity

Primary productivity is the rate at which energy is stored as organic matter (Fahey and Knapp, 2007). We measured primary productivity as net mass gain (%) of macroalgae (*Ulva*) on substrata protected from grazers. We deployed four replicates at each site in plastic bottles with holes, 4 m from the ocean floor (to standardize light availability). Ropes were attached to bricks and deployed on the substratum at 5–10 m intervals (**Figure 2**). After 48 h, we collected the macroalgae, returned them to the laboratory and assessed their mass loss/gain (ΔM) as follows:

$$\Delta M = (\mathrm{Mf} - \mathrm{Mi})/\mathrm{Mi}$$

Where i and f refer to initial and final wet weights as per Rasher et al. (2013).

Herbivory/Grazing Intensity

Herbivores play a vital role in maintaining a healthy coral-dominated community through intense feeding and grazing of unwanted macroalgae that indirectly interfere with the growth, reproduction and survivorship of corals (Burkepile and Hay, 2008). To estimate herbivory, we exposed samples of macroalgae (referred to as "algae pops") to determine their susceptibility to grazing. The macroalgae were deployed on 90 cm lengths of three-stranded nylon rope, upon which the algae had been grown for 2 weeks prior to use (**Figure 2**). At each site four replicates were deployed and spaced 5–10 m apart on the sea floor. Following exposure for 48 h, loss of biomass (ΔC) was calculated as follows:

$$\Delta C = \{[(1 + \Delta M)^{*}\mathrm{Mi}] - \mathrm{Mf}\}/\mathrm{Mi}$$

Where i and f refer to initial and final wet weights as per Rasher et al. (2013) and ΔM is the % mass gain calculated from the primary productivity assay.

Predation

Fish feeding intensity is often used as a measure of predation in a shallow water ecosystem. A simple assay was used to assess predation – "the squid pop" (Duffy et al., 2015). This refers to a piece of dried squid deployed at the end of a tether tied to a stake as bait for fish. In order to standardize the squid pop assay, we cut the dried squid into disks that all had the same size. Following this, the disks were deployed from the seafloor

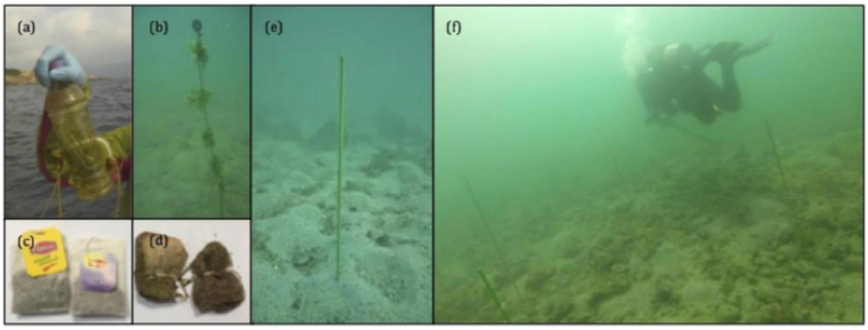

FIGURE 2 | Ecosystem function assays for **(a)** primary productivity using *Ulva* in closed plastic bottles **(b)** herbivory using *Ulva* tied to ropes **(c)** green and rooibos tea bags shown before decomposition and upon retrieval **(d)** after 6 months of deployment **(e)** and **(f)** predation using squid pops.

by tethering them to individual metal stakes using fishing line. Squid pops were deployed at three sites (CI, CLP, PI; 25 replicates per site). Squid pops were not deployed in the fourth site – TPC due to inclement weather forecast and budget restrictions. One hour after deployment, we returned to observe whether or not fish consumed the bait. The degree to which the squid pops were consumed was used as an indicator of the level of predation (or the activity of fish feeding) in the marine ecosystem (Duffy et al., 2015; **Figure 2**).

$$\text{Bait loss} = 1 - [(\text{Si} - \text{Sf})/\text{Si}]$$

Where i and f refer to the initial and final number of squid pops, respectively.

Organic Matter Decomposition and Carbon Sequestration

To measure organic matter decomposition and carbon sequestration potential, we used the tea bag assay as per Keuskamp et al. (2013). The tea bag assay was developed to measure organic matter degradability in a standardized, cost-effective manner across diverse ecosystems globally. Recent studies have demonstrated its usefulness and its applicability across aquatic and terrestrial ecosystems (Al-Maliki and Al-Masoudi, 2018; Mueller et al., 2018; Duddigan et al., 2020). Further, Seelen et al. (2019) demonstrated the applicability of the tea bag method to marine environments by incorporating a leaching factor (40% for green tea and 20% for rooibos tea).

We deployed 10 pairs of commercial rooibos tea bags and green tea bags (rooibos tea – ASIN: B00BAUF9KA, Lipton, Unilever, United Kingdom and green tea – B0173LLHNM, Lipton, Unilever, United Kingdom) in nylon mesh (200 μm) that were held within a porous plastic bottle and buried ~10 cm beneath the sea floor for a period of 6 months in the four study sites (**Figure 2**). Upon retrieval, the tea bags were oven-dried at 60°C for at least 72 h and weighed after removing the mesh bags. Decomposition rate (k, d^{-1}) of rooibos tea and organic matter stabilization (S, %) of green tea were calculated using the following equations:

$$k = -\text{Ln}\ [(\text{final mass} - \text{initial mass})/\text{duration of deployment}]$$
$$s = 1 - [(\text{final mass} - \text{initial mass})/\text{initial mass of tea}] \times 100.$$

Multifunctionality

In order to assess the ability of a given site to perform multiple ecosystem functions, we calculated a multifunctionality index following a geometric mean approach and a multiple threshold based approach (Byrnes et al., 2014). For each of the five functions (primary productivity, herbivory, predation, organic matter decomposition and carbon sequestration), we defined "maximum functioning" as the highest value among the recorded observations. Following this we calculated the standardized percent functioning for each function and took the geometric mean of these values to get the multifunctionality index for every site. Next, we selected a multifunctionality threshold (50% of the calculated maximum functioning values) to evaluate the effect of coral species richness on ecosystem function. The rationale for this is that Gamfeldt et al. (2008) made reasonable observations assuming a 50% functioning threshold to study the effects of species loss on multifunctionality. We recorded 4 values between 0 and 1 for each site, where 1 represented a function was maintained over the selected threshold and 0 represented a function that was maintained below this threshold. We took the sum of these values for each site, which resulted in numbers between 0 and 4, with 4 indicating that the site was able to maintain all ecosystem functions above the specified threshold.

ARMS Plate Photo Annotations

Plate photos from 12 autonomous reef monitoring structures (ARMS), that were deployed in the same four study sites were analyzed and annotated by functional group. The ARMS are similar to mini apartment blocks that are made of 9 stacked PVC plates and are deployed on the sea floor. The structures have cave-like spaces for marine fauna to crawl in, hide and settle. Following deployment for 2 years, the ARMS were retrieved, and everything on the plates was collected and identified (Leray and Knowlton, 2014). During this step, high resolution photographs of individual plates were taken (**Figure 3**). Similar to settlement plates, ARMS plate photos can be used to assess the diversity, coverage and size of sessile taxa. Recently, David et al. (2019) validated ARMS as a promising monitoring tool for hard bottomed communities and to investigate environmental effects. An online software CoralNet, a repository and resource

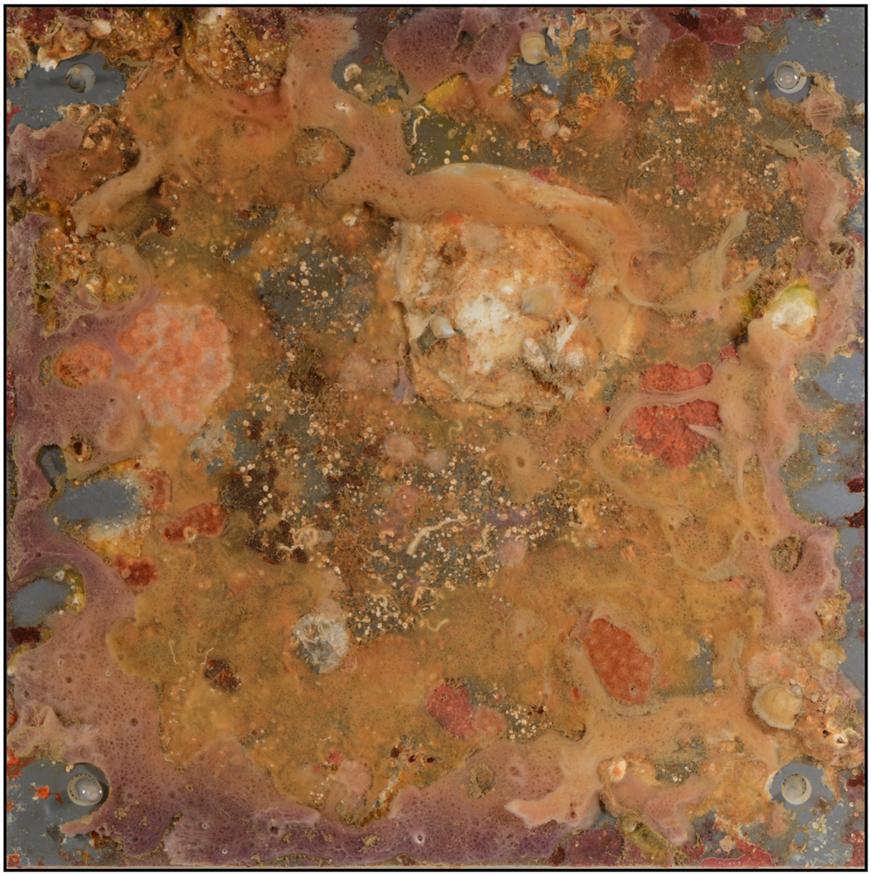

FIGURE 3 | Plate photo of ARMS from Port Island, Hong Kong (October 2017).

for benthic image analysis was used that helped generate 50 random points on each plate photo (Beijbom et al., 2015). The software uses computer algorithms that allows fully and semi automated annotation. Based on already identified taxa, the organisms on the plates were assigned to various taxonomic groups such as sponges, bryozoans, bivalves, etc. The assignments were made based on annotation categories used by the NOAA CRED program that is already available on CoralNet. In total, 204 plate photos were annotated at the functional level. Relationships between the abundance of various taxonomic groups on ARMS plates and individual ecosystem functions were assessed to understand the mechanisms that drive variability in functionality.

eDNA Data on Fish Abundance

Seawater Sampling

Seawater samples were collected in August 2016 from the surface in 1-L Nalgene bottles (3 replicates per site from the four study sites) and filtered through a 47 mm diameter PES filter (nominal pore size, 0.22 μm; Millipore Express). Filters were wrapped in aluminum foil and stored at −20°C for DNA extraction.

DNA Extraction

DNA was extracted in triplicate from every sampling site, using the DNeasy Blood and Tissue Kit (Qiagen) according to the manufacturer protocol and supplemented with a few steps: first, the filter was cut into 4 pieces and put into a UV-sterilized 1.5 mL tube. Next, 180 μL ATL was added to each tube. Following this, 20 μL Proteinase K was added to each tube individually and vortexed. The tubes were vortexed a few more times during the day and incubated at 55°C overnight. To increase the yield, the DNA was eluted with 100 μL Buffer AE twice. Next, following incubation at room temperature for 5 min, the spin column was spun at 8000 RPM for 1 min. For the remaining steps, we followed the manufacturer's protocol. The extracted DNA was stored at −20°C. To monitor contamination, a blank filter was used as negative control. DNA from four fish species (*Nemipterus bathybius, Gambusia affinis, Xiphophorus helleri*, and *Macropodus opercularis*) was extracted and used as the positive control.

Library Preparation

We chose a set of universal PCR primers for metabarcoding eDNA from fishes in subtropical habitats (MiFish-U). The primers amplify an approximately 163–185 bp region of the 12S rRNA gene, which is known to resolve taxonomy of most fishes to the family level (Miya et al., 2015). We used a two-step PCR amplification protocol. In the first round of PCR amplification, 12 samples with the universal fish primer pairs (MiFish-U) were used. The PCR amplification was done in 30 cycles and the total reaction volume was 12 μL, containing 6 μL 2 × KAPA HiFi HotStart ReadyMix (KAPA Biosystems, Wilmington, MA, United States), 0.36 μL of each primer (10 μM), 2.0 μL of template and 3.28 μL of sterile distilled H_2O. The thermal cycle profile was as follows: initial 3 min denaturation step at 95°C followed by denaturation at 98°C for 30 s; annealing at 63°C for 30 s; extension at 72°C for 20 s; and a final extension at 72°C for 5 min. Collected PCR products were purified using the MinElute Kit (Qiagen) with an elution volume of 10 μL. DNA fragments were recovered and ready to use in downstream analysis. The concentration of DNA was measured using a Thermo Scientific^{TM} μDropTM Plate. The second round of PCR followed by library preparation and sequencing were performed at Genewiz (Suzhou, China) following protocols described in Miya et al. (2015). Indexed amplicon libraries were constructed, pooled and multiplexed sequenced using the Illumina MiSeq platform.

Sequencing Data Optimization

For base calling and preliminary analysis of the raw data, we used Bcl2fastq (v2.17.1.14). To optimize the raw sequencing data – (a) two sequences of each read pair were merged according to overlapping sequences, after which undetermined bases were removed (b) Primers and adapter sequences were removed (c) the 5′ and 3′ bases with Qphred score < 20 were trimmed (d) resulting sequences >200 bp in length passed this stage of optimization. Cutadapt (v1.9.6), Qiime (1.9.1), and Vsearch (1.9.6) were used (see **Supplementary Information** for raw data statistics).

Taxonomic Assignment

OTU refers to an operational definition of a classification unit (genus, species, grouping, etc.) used commonly for data analysis. All the sequences in a sample were classified to obtain information on species and genus. By classification, the sequences were grouped according to their similarity, and one group is an OTU. First, unique sequences were extracted from optimized sequences with read count information. Next, OTU clustering of unique sequences (read count > 1) was performed with similarity of 97%, and chimeric sequences were further removed to obtain representative OTU sequences. The RDP classifier was used to select and annotate the representative sequence for each OTU to obtain the community composition of each sample. Community composition was subsequently analyzed and summarized using Silva_128 12S rRNA database for species annotation. The relationship between fish community composition (**Supplementary Appendix Tables A.4, A.5**) and ecosystem function was characterized to understand the factors driving its variability.

Environmental Data

Environmental data was obtained from surface seawater samples collected from the four study sites by the Hong Kong Environmental Protection Department in 2016 (10 replicates per site, every month). The parameters used for the analysis were – turbidity, total phosphorous, total nitrogen, total Kjeldahl nitrogen (TKN), total inorganic nitrogen (TIN), temperature, suspended solids, secchi depth, salinity, pH, orthophosphate, nitrate-nitrogen, ammonia-nitrogen, dissolved oxygen, chlorophyll-*a*, and coral cover (**Supplementary Appendix Table A.1**).

Data Analysis

Data were screened for normality using a Shapiro–Wilk *W* test and for homoscedasticity using Levene's test. One-way ANOVA was used to evaluate the variability of primary productivity (% mass gain), herbivory (% mass consumed), predation (% bait loss), decomposition rate and carbon sequestration by habitat. *Post hoc* comparisons to test for significant differences between sites were conducted using Student's *t*-test and Tukey's test. Generalized linear regression was performed with second-order AICc (Hurvich and Tsai, 1989) model selection to obtain the model that could best explain the relationship between the parameters. All statistical analyses were performed in JMP 15.0. Visualizations were carried out using Datagraph.

RESULTS

We present our analyses on the level of individual functions and joint ecosystem functioning (or multifunctionality) in four habitats that have varying environmental parameters, fish abundance and benthic community composition. We analyzed the effect of urbanization, as reflected in the environmental variables on multiple ecosystem functions – primary productivity, predation, herbivory, decomposition and carbon storage. Next, we use two methods to assess multifunctionality – the average approach and the 0.5 threshold approach. Finally, we evaluate the effects of fish abundance and benthic community composition on individual ecosystem functions and overall multifunctionality.

Individual Ecosystem Functions

Net mass gained by protected *Ulva* (primary productivity), net mass loss by exposed *Ulva* (herbivory), loss of squid pop (bait) to predators (predation intensity), and carbon sequestration varied significantly among the four habitats (see **Supplementary Materials 1–5** for one-way ANOVA results); One-way ANOVA; $n = 47$; $F_{(3,44)} = 8.9$ $p = 0.0001$; One-way ANOVA; $n = 47$; $F_{(3,44)} = 301.2$ $p < 0.001$; One-way ANOVA; $n = 47$; $F_{(3,44)} = 21.1$ $p < 0.001$; $n = 47$; $F_{(3,44)} = 3.4$ $p = 0.03$, respectively; **Figure 4**. *Post hoc Tukey HSD* tests revealed significant differences between the farthest sites (Tung Ping Chau and Center Island) for primary productivity ($p = 0.0001$), herbivory ($p = 0.0001$), and predation ($p = 0.0001$); and adjacent sites for carbon sequestration (Che Lei Pai and Port Island $p = 0.04$), primary productivity (Tung Ping Chau and Che Lei Pai $p = 0.0007$; Tung Ping Chau and Port Island $p = 0.02$), herbivory (all site pair comparisons $p < 0.001$) and predation (Tung Ping Chau and Che Lei Pai $p < 0.0001$; Center Island and Port Island $p = 0.0001$; Tung Ping Chau and Port Island $p = 0.04$).

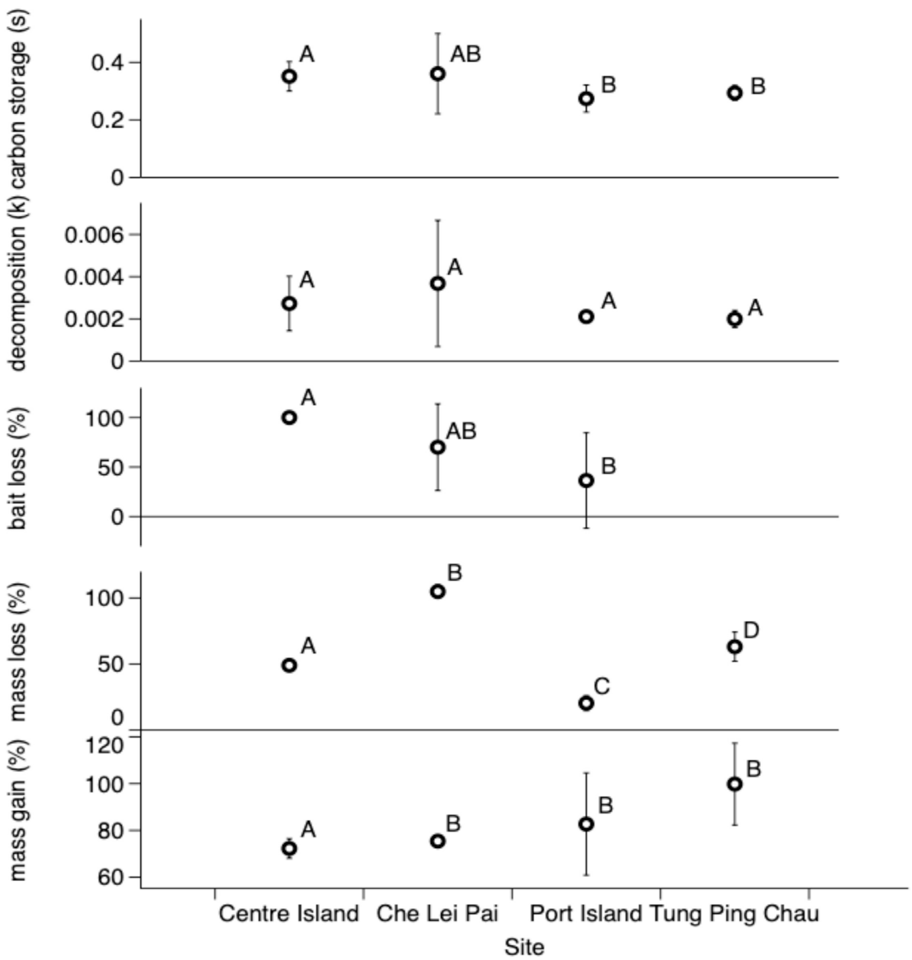

FIGURE 4 | Ecosystem function data for primary productivity, herbivory, predation, organic matter decomposition and carbon sequestration across the four study sites (Center Island, Che Lei Pai, Port Island, Tung Ping Chau). Primary productivity was measured as the % gain in mass of macroalgae over a deployment period of 48 h; herbivory was measured as the % loss in mass of macroalgae over a deployment period of 48 h; predation was measured as the % loss in bait (dried squid) deployed over a period of 1 h – note: data missing from TPC; organic matter decomposition and carbon sequestration were calculated using the tea bag assay (refer to section "Materials and Methods"). Data is represented as the average value ± standard deviation. Connecting letters indicate significant differences between sites revealed by *post hoc Tukey* HSD tests ($\alpha = 0.05$).

To identify the key environmental factors driving the variability, step-wise linear regression with model selection using the second-order Akaike Information Criterion (AICc) was performed (see **Supplementary Material** for detailed step-wise linear regression results with AICc scores: **Supplementary Tables 6A–E**). Chlorophyll-a and coral cover were significant drivers ($p < 0.05$) of primary productivity, on the other hand inorganic nitrogen and dissolved oxygen were closely significant ($p = 0.05$; best-fit model $r^2 = 0.44$). Coral cover was also critical and the most significant ($p < 0.05$) for determining herbivory, however, secchi depth, an indicator of light ($p = 0.05$) could not be eliminated (best-fit model $r^2 = 0.96$). Temperature and coral cover ($p < 0.05$) were important in predicting predation intensity (best-fit model $r^2 = 0.66$). In determining factors that drove the variability of organic matter decomposition, no significant environmental parameters were observed. However, inorganic nitrogen significantly impacted the carbon sequestration potential in the sites ($p < 0.05$; best-fit model $r^2 = 0.44$).

Analysis of Plate Imagery: Abundance of Sessile Fauna on ARMS Plates

ARMS plate photo analysis has been proven to be a powerful way to compare marine benthic communities (David et al., 2019). We observed 11 broad groups of organisms present on all ARMS deployed. This included several types of algae – chlorophytes, rhodophytes, and phaeophytes, tunicates, bivalves, and tube worms. To identify the key groups of benthic organisms driving the observed ecosystem function variability, step-wise linear regression with model selection using the second-order Akaike Information Criterion (AICc) was performed (see **Supplementary Table 7**). We observed no significant effect of the ARMS plates community composition on herbivory, primary productivity, carbon sequestration and organic matter decomposition. On the other hand, the relative abundance of green macroalgae ($p = 0.00001$), red upright macroalgae ($p = 0.00251$), soft tube worms ($p = 1.46e^{-5}$) and tunicates ($p = 0.00044$) had a significant effect on the intensity of predation (best-fit model $r^2 = 0.95$).

eDNA Data on Fish Abundance

Illumina MiSeq analysis yielded an average of 196540 million reads. High quality reads were obtained upon using a Phred score cutoff of 90% on the paired end pre-processed reads (**Supplementary Appendix Tables A.4, A.5**). Overall, ~75% of all reads were annotated to local fish species. Samples from Center Island (the innermost site) did not amplify during the second PCR reaction. Therefore, these samples were excluded from library preparation.

eDNA sequences of a total of 91 taxa were obtained, in 44 families and 14 orders. Over 98% were registered under the Hong Kong Register of Marine Species, a comprehensive database for marine species in Hong Kong waters (Astudillo et al., 2018). Commercially valuable fish species were abundant, such as golden threadfin bream *Nemipterus virgatus*, flathead gray mullet *Mugil cephalus*, and yellow croaker *Larimichthys crocea*. Pollution-tolerant species including *Mugil cephalus*, and *Siganus canaliculatus* were also obtained from the samples. When compared with other conventional fish surveillance methods such as trawling (Leung, 1997), line fishing and netting (Hksar Agricultural, Fisheries and Conservation Department, 2016), as well as underwater visual census surveys (Sadovy and Cornish, 2000), eDNA captured ~75% of taxa previously recorded in the same study sites. Moreover, there were 11 species that were only recorded using the eDNA method, but not observed in the conventional methods (Tsang, 2018).

Relative abundance of all fish varied significantly between sites (One-way ANOVA $n = 11$; $F_{(3,8)} = 103.5$ $p < 0.0001$). *Post hoc* Tukey HSD revealed significant differences between the farthest sites Che Lei Pai and Tung Ping Chau and also all adjacent site pairs except Port Island and Tung Ping Chau (**Figure 5**). It is worth noting that OTU relative abundance is only correlated but does not reflect the relative abundance of the fish species under discussion. This is due to limitations with sequencing methods that yields platform specific data (reads), therefore making relative abundance analysis challenging (Harrison et al., 2020). Nevertheless, acknowledging these constraints imposed by compositional data is vital when analyzing sequencing data.

We classified the observed fish species according to their dietary habits to herbivorous, omnivorous, planktivorous and predatory fish using FishBase (**Figure 6**). Specifically, relative abundance of herbivorous, omnivorous and planktivorous fish varied significantly between sites, however, the relative abundance of predatory fish showed no significant variability.

We performed step-wise linear regression with model selection using the second-order Akaike Information Criterion (AICc), to characterize the relationship between fish abundance, fish diversity, dietary preference and ecosystem function (see **Supplementary Tables 8A–E**). Mass loss significantly depended on the relative abundance of herbivorous fish ($p = 0.00061$; best fit model $r^2 = 0.22$). Mass gain significantly depended on the relative abundance of the following 12 genera of predatory fish ($p = 0.00369$; best fit model $r^2 = 0.46$): *Platycephalus indicus*, *Protonibea diacanthus*, *Trichiurus japonicus*, *Larimichthys polyactis*, *Polynemus dubius*, *Anguilla japonica*, *Nemipterus virgatus*, *Collichthys lucidus*, *Silurus meridionalis*, *Istigobius campbelli*, *Cephalopholis boenak*, and *Larimichthys crocea*. The relative abundance of all planktivorous fish ($p = 0.01198$), omnivorous fish ($p = 0.00179$), and certain genera of predatory fish (*A. japonica, I. campbelli, N. virgatus, C. lucidus, L. crocea, P. indicus, P. dubius, T. japonicus, S. meridionalis, C. boenak, L. polyactis, P. diacanthus*; $p = 0.00431$) were significant drivers of bait loss variability (best fit model $r^2 = 0.52$). The relative abundance of 12 genera of predatory fish – *C. boenak, C. lucidus, I. campbelli, L. polyactis, N. virgatus, P. indicus, P. dubius, S. meridionalis, T. japonicus, A. japonica, L. crocea,* and *P. diacanthus* significantly drove the variability of organic matter decomposition ($p = 0.00126$; best fit model $r^2 = 0.24$). Finally, *N. virgatus, P. indicus, A. japonica, L. polyactis, P. diacanthus, C. boenak, L. crocea, T. japonicus, P. dubius, S. meridionalis, C. lucidus,* and *I. campbelli* significantly determined carbon sequestration variability ($p = 0.00224$; best fit model $r^2 = 0.18$).

Overall Ecosystem Multifunctionality

In this study, overall multifunctionality was calculated using two approaches – average approach and threshold approach. Multifunctionality using the average approach was calculated without bait loss, as there was no data for Tung Ping Chau. Overall multifunctionality index was 0.70 for Center Island, 0.72 for Tung Ping Chau, 0.92 for Che Lei Pai and 0.51 for Port Island. The number of functions that could be maintained above a 50% threshold was 4, 5, 3, 4, respectively for Center Island, Che Lei Pai, Port Island and Tung Ping Chau (**Table 1**).

DISCUSSION

In the last decade, research into the concept of multifunctionality (ability of ecosystems to provide multiple ecosystem functions) has increased owing to tremendous anthropogenic pressure on natural resources. This has resulted in the need to design and manage ecosystems so that they can efficiently provide multiple ecosystem functions simultaneously (Manning et al., 2018). However, not all ecosystem functions exhibit the same response to environmental drivers (Bradford et al., 2014). Given the complexity and diversity of ecosystem functions, the present study discusses five vital functions (primary productivity, herbivory, predation, organic matter decomposition and carbon sequestration) in an urbanized coastal marine environment. Data from each individual ecosystem function has been used to derive the overall multifunctionality for the human-impacted habitat. Linkages between multifunctionality, individual functions, environmental

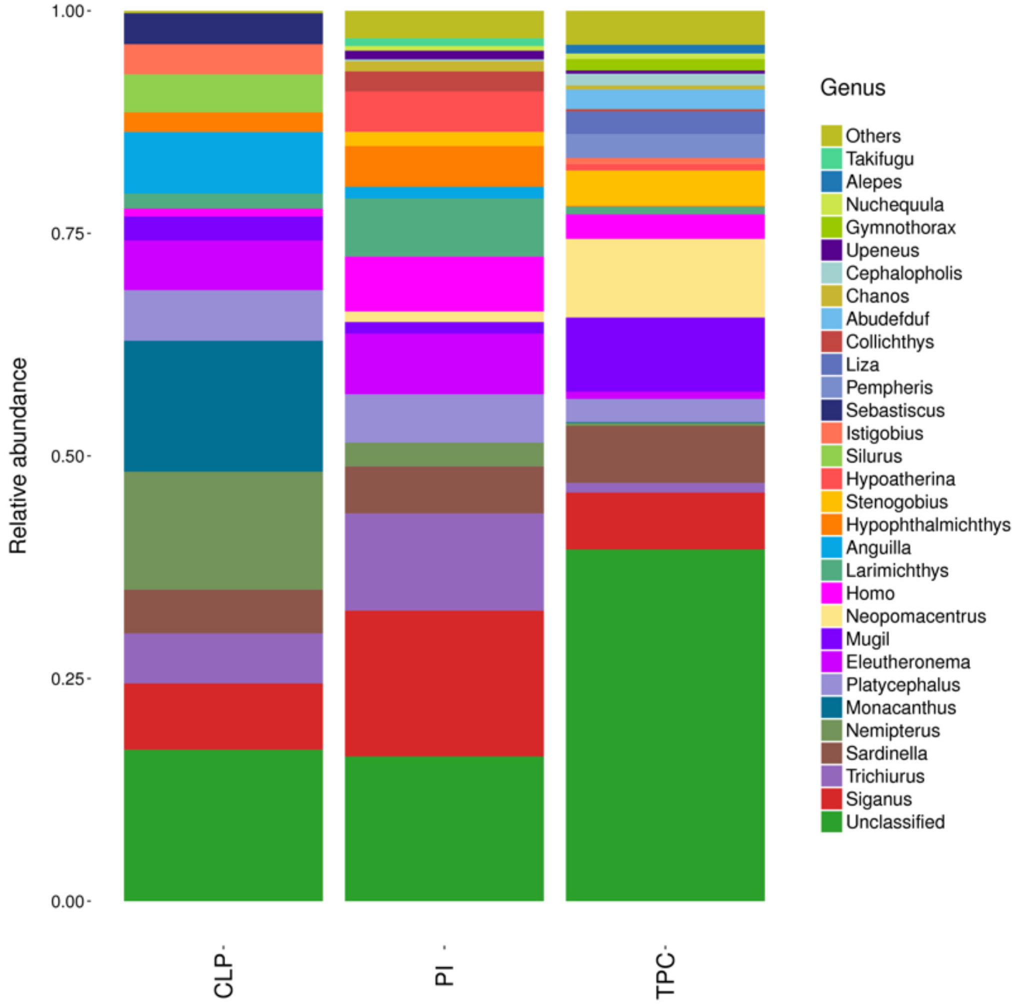

FIGURE 5 | Relative abundance of top 30 genera of fish obtained from eDNA analysis of seawater samples collected from the same study sites. We recorded no fish in Center Island (CLP - Che Lei Pai; PI – Port Island; TPC – Tung Ping Chau).

factors, fish abundance and benthic community composition datasets have been presented here.

Habitat as a Driver of Ecosystem Function

Coral reefs and coral communities are vital components of the Earth's biosphere dominating several coastal habitats (Knowlton et al., 2010). In recent years, however, these habitats have been destroyed and homogenized owing to unsustainable coastal development, increased levels of pollution, overfishing and disease. This decline in coral cover is projected to continue even if the goals of the December 2015 Paris Agreement are implemented (Hoegh-Guldberg et al., 2019). Changes in habitat (coral reef complexity, coral cover, coral species richness, etc.) can have important implications for the functions that coral reefs provide such as productivity, sediment generation and so on (Perry et al., 2018). Our study sites in Hong Kong are along a gradient of coral cover (Center Island has almost no hard coral species to Tung Ping Chau which has ∼65 hard coral species) and serve as natural laboratories to discuss how changes to coral cover can affect multifunctionality in an ecosystem. Specifically, in testing for the effects of habitat on ecosystem function, we observed a positive relationship between coral cover and primary productivity. This is consistent with several literature showing that homogenized reef communities can jeopardize ecosystem functioning and productivity (Alvarez-Filip et al., 2015; Hughes et al., 2018; Richardson et al., 2018).

Functional Diversity as a Driver of Ecosystem Function

Functional diversity has been proven to be an indicator of ecosystem function (Petchey et al., 2004) and has been correlated to productivity (Cadotte et al., 2009). When coral communities shift, due to stressors, some visible changes often occur. One such example is the shift from a coral-dominated community to an algae-dominated one (Brooker et al., 2016). We observed

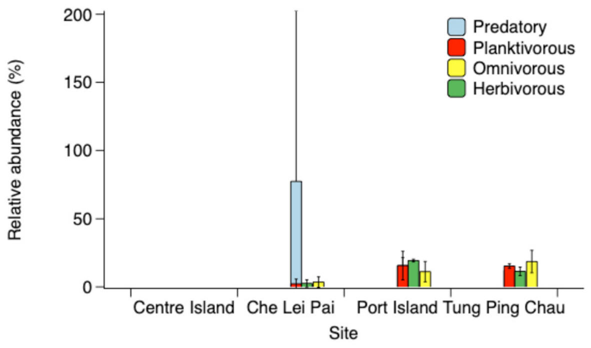

FIGURE 6 | Results of relative abundance of predatory, herbivorous, planktivorous and omnivorous fish obtained from eDNA analysis of seawater samples collected from the same study sites. We recorded no fish in Center Island. Data is represented as the average value ± standard deviation.

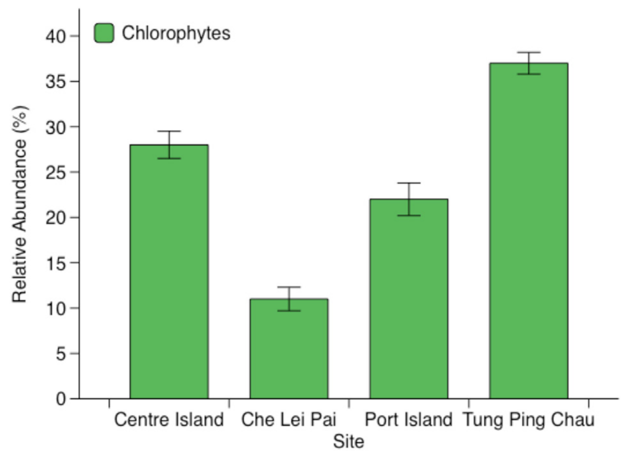

FIGURE 7 | Results of annotations of plate photos revealing average relative abundance of chlorophytes from Autonomous Reef Monitoring Structures (ARMS) deployed in the same study sites.

TABLE 1 | No. of functions maintained above the 50% threshold (1-maintained; 0-not maintained)

	Mass gain	Mass loss	Bait loss	*k*	*s*	No. of functions maintained above the 50% threshold
Center Island	1	0	1	1	1	4
Che Lei Pai	1	1	1	1	1	5
Port Island	1	0	0	1	1	3
Tung Ping Chau	1	1	0	1	1	4

this visually during scuba dives in Center Island and validated it through our analysis on the relative abundance of food sources as captured by the ARMS plates. Center Island recorded high relative abundance of algae, but no hard-coral species. On the other hand, Tung Ping Chau recorded relatively high abundance of both corals and algae (**Figure 7**).

When looking for functional diversity drivers of overall multifunctionality, the relative abundance of benthic filter-feeders emerged significant, outweighing effects from abiotic factors. This is consistent with literature citing bivalves as pivotal players in ecosystem functioning owing to their contribution to nutrient regeneration and productivity (Norkko et al., 2013).

Abiotic Factors as Drivers of Ecosystem Function

Nitrogen is the biological nutrient limiting factor in marine ecosystems, as it is generally less available for ocean plants and animals. In this study, there were significant inorganic nitrogen mediated effects on primary productivity and carbon sequestration in the study sites. Consistent with Zhang et al. (2015) and Mueller et al. (2018) increased inorganic nitrogen significantly reduced organic carbon preservation in coastal marine sediments. The results were also was consistent with other studies such as Bristow et al. (2017) which observed that nitrogen exerts a critical control on primary productivity. On the other hand, we observed the opposite relationship between total Kjeldahl nitrogen (TKN: a measure of organic nitrogen) and carbon sequestration, where carbon storage decreased with decreasing TKN concentration. TKN is directly related to the source of bottom deposition. Wastewater discharges often contain relatively high concentrations of organic nitrogen and Hong Kong discharges over 3 million cubic meters of treated wastewater into its surrounding marine environment every day. Archana et al. (2016) recorded that wastewater was indeed the dominant source of the nitrogen pool using stable isotopes. Therefore, characterizing the effect of bottom sediment TKN is valuable in this highly urbanized coastal system.

We observed no nutrient-driven effects on primary productivity, herbivory, predation and organic matter decomposition. This is consistent with Alsterberg et al. (2014), which summarized findings from some studies that demonstrated no effects of nutrients on ecosystem multifunctionality in a marine habitat. We hypothesize that nutrient-driven effects were masked by effects from other significant abiotic factors such as secchi depth (an indicator of light), chlorophyll-a, dissolved oxygen and temperature.

The Coral-Algae-Herbivore Triangle

The complexity of the coral-algae-herbivore relationship is well known (Holbrook et al., 2016). A popular hypothesis (that is also somewhat controversial) is that herbivore fishes help corals thrive in a reef benthos by managing the distribution and abundance of algae (Hixon, 2015). However, overfishing of herbivorous fishes can degrade this association (Heenan et al., 2016). The data recorded from the herbivory assay showed significant variability between the four sites. We observed relatively more algal consumption in Tung Ping Chau and Che Lei Pai when compared to what was observed in Center Island and Port Island. We hypothesize that this could be due to several reasons – firstly, we propose that the lack of consumption in Center Island and Port Island could be due to the availability and preference for other food sources, such as *Sargassum*, which was also present at some sites. This was confirmed by the ARMS plate photo analysis, which revealed several types of algae (**Figure 7**) in varying abundances at all the four study sites. Secondly, we observed herbivorous rabbitfish and pufferfish at the sites that recorded more herbivory. This was corroborated by fish abundance from eDNA analysis (**Figure 6**), which suggested the presence of herbivorous and omnivorous fish in increasing abundances from Che Lei Pai to Tung Ping Chau. However, it was unusual that we observed the presence of herbivorous and omnivorous fish in the eDNA analysis at Port Island, yet did not observe any algal mass loss in the herbivory assay. As the coral-algal-herbivore interaction does not occur in isolation of the rest of the ecosystem, we suspect the variability in our dataset to be due to a combination of abiotic factors, other organisms and excessive fishing.

Relative Feeding Intensity of Generalist Predators

There continues to be some controversy on whether algal distribution and growth in the marine benthos is controlled by top-down herbivory/predation or bottom-up nutrient cycling (Lapointe et al., 2004; Poore et al., 2012). While bottom-up control by resources is relatively well understood, it is necessary to characterize the geography of top-down control by predators and herbivores through space and time (Meyer et al., 2016). One way to do this is to measure prey vulnerability – however, across geographies and larger scales, it is desirable to have a standardized food type and a simpler tool to measure the relative feeding intensity of generalist predators. In line with this, we utilized the squid pop technique as per Duffy et al. (2015). Feeding intensity (bait loss from squid pops) was higher in the sites with higher hard coral species richness. A recent study that employed the squid pop technique found increased feeding intensity to be directly correlated with fish abundance. This pattern could therefore reflect a higher level of predation in Port Island, a site where visual observations have also recorded more fish compared to inshore sites. Consistently, fish abundance from the same sites revealed a similar pattern, thereby corroborating our findings. However, we did not obtain predation intensity for Tung Ping Chau. Therefore, we did not obtain a holistic picture for the difference across habitats in relative predation intensity. We believe these limitations are compensated by the assay's ease of applicability and standardization, making it possible to replicate the experiment and obtain data fairly quickly.

Intermediate Disturbance Affects Overall Ecosystem Multifunctionality

To determine the overall ecosystem multifunctionality, we followed two standard approaches – the average approach and the threshold approach. Both these approaches include all available measures of ecosystem functions in a given habitat. Consistently across both approaches, Che Lei Pai recorded a relatively higher multifunctionality index and multifunctionality threshold while Port Island recorded a relatively lower index and threshold compared to the other sites. The site with the least disturbance (Tung Ping Chau) did not record the highest multifunctionality index or threshold, neither did the site with the most disturbance (Center Island) record the lowest multifunctionality index or threshold. The site with intermediate disturbance (Che Lei Pai) recorded the highest multifunctionality index and threshold. This was contrary to our hypothesis (multifunctionality increases from inshore – more human impacts to open ocean - less human impacts). The intermediate disturbance hypothesis was presented originally to describe species-mediated effects on corals and trees (Connell, 1978). It predicts that sites with maximum disturbance and least disturbance will have low diversity and that intermediate disturbance will maximize diversity. In line with this, we propose that intermediate disturbance maximizes diversity and correspondingly multifunctionality in Che Lei Pai when compared to the other sites. However, it must be noted that both approaches to characterize ecosystem multifunctionality are not without limitations and multivariate methods for quantification is recommended.

In testing for drivers of multifunctionality, we observed abiotic-mediated (turbidity) and functional diversity-mediated (benthic filter feeder abundance) effects. However, maintaining high levels of multifunctionality in an ecosystem is not likely to be driven solely by these two factors. The maximization of functions favored by the habitat is likely to be largely responsible as per Hensel and Silliman (2013). The findings from our study revealed that when human impact decreases, the consequences to overall ecosystem functioning depend not only on which single functions are taken into consideration. This implies that ecosystem multifunctionality is also driven by the presence or absence of other metazoans and microbial organisms present that drive trophic level interactions and functions such as herbivory and predation. This is consistent with Gamfeldt et al. (2008) who proposed that even in eutrophication impacted coastal marine environments, overall multifunctionality is more susceptible to species loss than are individual ecosystem functions. So, taking these factors into consideration and shifting focus to a variety of functions provided by a diversity of species is likely to provide a holistic picture of ecosystem multifunctionality and will subsequently help in conservation management.

CONCLUSION AND IMPLICATIONS FOR CONSERVATION MANAGEMENT

The results from this study have important implications for understanding processes regulating the structure of human-impacted coastal marine habitats. By characterizing important ecosystem functions (primary productivity, herbivory, predation, decomposition and carbon sequestration) and overall multifunctionality, the findings significantly expand on what we already know about urbanization impacts on key ecosystem processes. The highlight of this study is the finding that urbanization impacts multifunctionality in coastal marine systems. However, these results warrant a more detailed investigation on the links between multifunctionality and community composition (microbial and metazoan). Further research into the role of decomposition in overall ecosystem multifunctionality and associated carbon storage mechanisms can be used to help in global carbon inventory management.

Moreover, research into functional level responses, microbial interactions and additional ecosystem services (Manning et al., 2018) such as shoreline stabilization, biogeochemical cycling and carbon sequestration will help shed light on the holistic properties of the ecosystem.

DATA AVAILABILITY STATEMENT

All data generated for this study has been made publicly available in the **Appendix** and **Supplementary Material** of this article.

AUTHOR CONTRIBUTIONS

AA and DB conceptualized the research. AA conducted the fieldwork and did the analysis. DB reviewed the manuscript and analysis. Both authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.557145/full#supplementary-material

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Mainstreaming Sustainable Consumption of Seafood Through Enhanced Mandatory Food Labeling

*Jerneja Penca**

Euro-Mediterranean University, Piran, Slovenia

To contribute to the debate about sustainable seafood consumption, this article considers the role of mandatory food labeling. The article first flags the rise of a policy paradigm of shared responsibility and policy imperatives at various levels calling for increased integration of the citizen/consumer into public regimes, including in fisheries governance. It then explores the options available to citizen/consumers to engage in the fisheries regime in different stages of the value chain and evaluates their readiness to respond to the expectations. Mandatory food labeling of seafood is introduced as an under-unexplored governance tool, alongside the key enabling technological and policy trends. The rise of transparency and traceability, both as norms and a set of technological capabilities, is highlighted as an opportunity for implementation of mandatory seafood labeling. While recognizing equity challenges and various supplementary actions needed to ensure an effective behavioral and attitudinal shift toward more engaged governance (better education and enforcement and an enabling social setting), the article suggests to further explore mandatory labeling within the governance toolbox. It should be particularly relevant in the context of developed markets with global trade and political influence, and as means of fostering ocean literacy and transparent, participative and deliberative kind of governance.

Keywords: responsible consumption, sustainable seafood, shared responsibility, mandatory labeling, seafood supply chain, food traceability, transparency, ocean literacy

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****Correspondence:***
Jerneja Penca
Jerneja.penca@emuni.si

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INTRODUCTION

The sustainability record of capture fisheries remains insufficient. The Sustainable Development Goals (SDG), in the indicator 14.4, mandate the governments to effectively regulate harvesting and end overfishing, illegal, unreported, and unregulated (IUU) fishing, and destructive fishing. It is estimated that the target is unlikely to be achieved within 2020 and that it will require more time and effort on the part of all stakeholders, including consumers, where progress is needed in transformation of their perceptions and in provision of transparent and timely information to the public (FAO, 2020: 54). A greater role of citizen/consumer in accomplishing sustainability targets is anticipated also elsewhere in SDGs (SDG 12 and 17). Across the board, a fundamental shift can be noted in the expectation of citizen/consumer involvement in the governance model. Sustainable consumption has moved from a voluntary domain and its dependency on the consumer's sensitivity to ethical issues or "willingness to pay" a price premium (Richter et al., 2017; Zander and Feucht, 2018; Hilger et al., 2019), to a more mainstream policy expectation, according to which all consumers should be animated to do good for the public regime. Against this policy expectation, too little clarity exists over *how* to effectively do so. Key questions remain unanswered: Are existing policy tools adequately designed to engage the consumer to the desired extent? If not, how could they be improved?

This article contributes to the scholarship on sustainable consumption as part of sustainable governance in seafood (referring here to wild fishery, while acknowledging a heavy interaction with aquaculture products). The starting point is that consumer is a stakeholder in the "governance concert" of sustainable seafood (Barclay and Miller, 2018), but receiving insufficient attention in their influence on sustainable resource governance (Crona et al., 2015). Research has examined various instruments in that orchestration, ranging from the earliest sustainability campaigns and eco-certification or eco labeling (Iles, 2007; Jacquet and Pauly, 2007; Thrane et al., 2009) and more recent inventions of fishery improvement projects (FIPs), fishery credit systems, sustainability sourcing policies, and traceability schemes (Gutiérrez and Morgan, 2017; Kittinger et al., 2017; Bailey et al., 2018; Bush and Oosterveer, 2019). This body of literature is concerned with solutions to various sustainability challenges of the 21st century, including overfishing, social injustices, and unsustainable consumption patterns. One feature of this research is that it has focused on instruments devised by private actors, although recognizing their close interaction with government structures, and indeed mutual reinforcement between the two, within a model of hybrid governance (Gale and Haward, 2011; Bailey et al., 2016b; Bush et al., 2017). But while government is seen to play a role, that is understood as limited to hard regulation and providing supporting institutions and assurance for private governance to thrive. The tools that the governments can avail of to foster the consumer engagement have been sidelined. Nevertheless, public policy can do more to effectively engage this so-far neglected stakeholder in fisheries governance. Indeed, as trends enhancing the role of the citizen/consumer in global governance are on the rise, such instruments should be explored.

The aim of the article is to explore the potential of a core public policy tool, mandatory food labeling, to contribute to seafood sustainability. It asks: Can mandatory labeling play a role among adequate policy tools to respond to the emerging paradigm about the involvement of the consumer and under what conditions? The article defines the gap that exists between the policy targets related to sustainable consumption and the actual policy tools available to the consumer, but also advances the debate about how to overcome it. The research is based on an integrative review of existing theories and thinking from published literature, contextualized with key policy developments. Selected relevant literature from different traditions is assessed, critiqued, and synthesized in a way that enables the emergence of a new perspective on potential policy tools in a new policy context. Sources of policy developments are legal and political, including published strategies and public statements by policy and business actors. Through these, the piece details the inconsistencies between the policy paradigms and rhetoric, but also the emerging opportunities for progress toward a type of governance that provides workable tools for a responsible and empowered consumer.

The discussion in this paper is biased toward the consumers in developed countries and markets, noting that their governance frameworks, including enforcement and traceability mechanisms, and consumer awareness and organization, differ considerably from those in the developing countries. We examine the current EU mandatory labeling requirements and policy developments. The EU can serve as a case study of governance processes elsewhere, most notably North America and developed markets in the South. Furthermore, due to a high level of economic interdependency in seafood markets, these processes have a leverage to influence markets beyond their own. Large developed markets may also exert political influence, for instance, through bilateral and multilateral negotiations in international political processes. The EU is openly committed to acting as a sustainability leader in international ocean policy, including fisheries (European Commission, 2016), and to integrating sustainability concerns into its trade policy (European Commission, 2015). As such, the discussion about responsible consumption may at present be limited to some countries only, but it is a potential precursor for policy developments in other countries and in multilateral policies.

The article is structured as follows. Section "Governance Goals for Consumer Engagement" outlines the rationale for engaging the consumer in governance and reviews the expected role of the consumer in (capture) fisheries governance. Section "Commonly Examined Governance Tools for Sustainable Consumption and Their Weaknesses" reviews the typically discussed seafood governance tools in the context of their adequacy to respond to the identified policy expectations. Section "Mandatory Food Labeling as a Seafood Governance Tool" zooms onto the current application and the potential of mandatory seafood labeling in the context of recent normative trends and technological solutions. The article concludes by outlining the future of research work on consumer-targeting policy tools in the transition toward more sustainable consumption of fisheries governance.

GOVERNANCE GOALS FOR CONSUMER ENGAGEMENT

Across the policy domains, sustainable consumption is an ongoing governance challenge (Mont, 2019). This is no different in conservation and management of fisheries more specifically, although it might seem exotic in this context as the fisheries regime has been historically particularly insulated from the involvement of citizens and their concerns. The cornerstone of international fisheries law, the 1982 UN Law of the Sea Convention, put governments into center-stage. Since then, the governance focus has been on the cooperation among the governments on the one hand, and between governments and scientists on the other hand. Gradually, also fishermen and local resource users have begun being considered as sources of policy advice (Berkes, 2009) and fisheries management started to be conceptualized in terms of a system of interactions between state, market, and civil society groups (Kooiman and Bavinck, 2013). But even this differed from a wide engagement of consumers as citizens in the conservation and management of fisheries that we are witnessing today.

The rising expectations of the consumers in global challenges have arisen as a result of a number of inter-related trends. At the heart stands an ever-increased material interdependence, fueled by globalization, which requires cooperation, rather than just co-existence. The notions of shared responsibility, concerted action, decentralization, and cooperation are central to global governance, even if they have been approached distinctly by various bodies of literature. For instance, international law has pointed to the changing role of states (Brunnée, 2008; Nollkaemper and Jacobs, 2013), supply chain management has alerted us to the rise of consumers and producers (Lenzen et al., 2007; Jacobs and Subramanian, 2012; Global Economic Forum, 2015), while scholarship in the context of natural resource management has advocated for theories of commons and highlighted the existence of polycentricity, interdependence, and collective action (Ostrom, 1990, 2010; Berkes, 2006). Adding to these developments in the information age is the rise of transparency (Mol, 2015). Jointly, much literature provides justification for integrating the consumers into the global resource exploitation, and advocates for it without necessarily measuring its impact.

A testimony to the relevance of these theories in practice was the adoption, in 2015, of SDGs with a strong focus on the consumers, despite their nature as primarily country-led and country-owned. Indicator 12.2 requests the sustainable management and efficient use of natural resource by all (including citizens) and SDG 12.8 expects that by 2030 people everywhere have the relevant information and awareness for sustainable development and lifestyles in harmony with nature. Additionally, the SDG 17.14 expects policy coherence for sustainable development, and SDGs 17.16 and 17.17 encourage effective multi-stakeholder and multi-resource partnerships to support the achievement of the SDG in all countries. The SDGs are also formally communicated to individuals and couched in terms of advice to citizens (Sustainable Development Goals [SDG], 2019). It is important to recognize the SDGs as not only catering to the interests of the consumer, but also carving out a more visible role for him/her in the responsible management of resources.

A more consumer-focused governance approach started to appear in the fisheries management discourse. Some influence can be attributed to the adoption of the "blue growth" agenda, which loaded fisheries with expectations of technological development, added value to fish, and upgrading fish as commodity (Boonstra et al., 2018). The rise of policy objectives of "ocean literacy" and the understanding of the citizens' scientific and educational potential also played a role (United Nations, 2018). Finally, an influential factor was also the framing of fisheries as part of a food system (Aksnes et al., 2017; Science Advice for Policy by European Academies [SAPEA], 2020), where citizens' perceptions and expectations are seen as key drivers of sustainability (European Commission, 2020). Policy makers invented a more active role for consumers, as typified by the following statement: "Changing fish consumption is vital in helping fishing become more sustainable. As consumers and market actors we have to be aware that what, when, and how we eat, buy, and sell seafood has a huge impact on this precious food source" (European Commission, 2018c). Policy makers want to create a more prominent role for the individual.

The policy makers' effort is certainly also a response to the consumers' interests. Apart from the concern for quality of the product, consumers have increased their awareness and susceptibility to the ethical issues implicated in global food trade. However, their expectations of accountability and stewardship of producers in the seafood have generally not been met. Consumers have been critical of the existing policy tools regarding sustainable seafood consumption (McClenachan et al., 2016). European consumers claim they are ready to make more sustainable food choices, but blame price, lack of information, and knowledge as top barriers (BEUC The European Consumer Organisation, 2020). The outbreak of the Covid-19 epidemic seems to be further reinforcing the sustainability-oriented attitudes by food consumers (Accenture, 2020; Hubbub, 2020). As sustainable consumption is becoming more of a norm, enabling it becomes a priority.

COMMONLY EXAMINED GOVERNANCE TOOLS FOR SUSTAINABLE CONSUMPTION AND THEIR WEAKNESSES

This section provides an overview of the types of mechanisms and approaches that account for the consumer and evaluates their ability to respond to the outlined policy goals. The survey of tools seeks to convey the conceptual frame rather than provide a complete listing of ongoing initiatives. The incredible range of initiatives, codes, and standards for sustainable seafood under constant development is difficult to capture, while their formats are more standard. I distinguish between two fundamental types of policy mechanism that factor the consumer into the resource management: supply-chain interventions (focusing on the business-to-business operation before a product reaches the consumer) and consumer-facing tools (focusing on the consumer's leverage to affect the value chain). The concept of mandatory labeling fits under the latter, but it is singled out in a separate section of the article, to allow a more focused analysis.

Supply Chain Interventions

Supply chain interventions are invisible to the consumers, even if they are triggered by the concern for, and ultimately impact them. Indeed, influences over seafood supply chains take place in the backstage of the consumer's decisions and affect the producers and intermediates in the seafood trade, but they originate from seafood buyers' pressures for more sustainable sourcing. One example of such interventions is FIPs, which are tailored to the nature of the fishery (Cannon et al., 2018; Barr et al., 2019). Another one is more open-ended structural cooperation between fishers, processors, distributors, and retailers, such as Global Dialog on Seafood Traceability (GDST) and The Seafood Business for Ocean Stewardship (SeaBOS), which connect most of the world's largest seafood production companies (SeaBOS and GDST, 2019). The results can be tangible: in February 2020, the GDST issued the first industry-led Standard for Interoperable Seafood Traceability Systems. The standard determines the data elements that need to be documented and transmitted within seafood supply chains, and protocols on how to share that data. The standardization of key elements of data across the industry would significantly facilitate traceability of seafood products and increase their verifiability.

Alongside the industry-led initiatives, governments are increasingly dedicated to citizens' concerns. The key issues are human rights and labor conditions in seafood value chains ranging over modern slavery, hazardous working conditions, lack of safety equipment, forced child labor, human trafficking, and others. In 2016, the governments had entrusted the FAO to develop the Draft Guidance on Social Responsibility in Fish and Aquaculture Value Chains. These were developed through a multiple stakeholder consultation and delivered in 2019, but have subsequently been put on hold in the FAO Committee on Fisheries Sub-Committee on Fish Trade. Allegedly, some countries oppose excessively obliging language, although the voluntary nature of the document is clearly stated (SeafoodSource, 2020). Despite the political reluctance of some countries, a level of commitment to socially responsible value chains by the majority of states should be noted.

Another area of governments' concern is IUU fishing. Governments are increasingly using trade measures to prevent of IUU-sourced fish from entering the international market or importing it. The EU, for instance, has sought to influence producers by establishing a mandatory catch system and advising States to improve the transparency of their markets in order to ensure traceability, although without requiring a full traceability (Van der Marel, 2019: 313). There seems to be little formal interaction between industry and government-led, voluntary and obligatory, activities in accomplishing sustainability targets. More coordination should be a priority given that the success of various seafood sustainability governance is dependent on the extent to which market initiatives interact with the relevant public law (Gutiérrez and Morgan, 2017).

Consumer-Facing Tools

While supply chain interventions only take note or acknowledge the consumers, governance tools that more directly *engage* the consumers are rather scarce. Sources of information that seafood consumers can consult in taking a decision to contribute to sustainability are not abundant. This is true for both the average, more passive, consumer and for the more sensitive and more aware consumer.

Generalized messages to consumers regarding seafood consumption are often a bad proxy. Consumers can be encouraged to rely on "freshness" or "localness" of seafood (European Commission, 2018c), but there are no systemic means of verifying those attributes. Advice given by fishmongers, retailers, or restaurants can be too subjective, or an inadequate simplification of scientific complexity in fish stocks or social complexity in value chains. Similarly, the invitation to consume less-popular, under-utilized species does not necessarily lead to an overall positive outcome (Farmery et al., 2020). Finally, the consumer cannot also be expected to draw on scientific publications and stock assessment analyses, as these are too complex and inaccessible for most consumers.

Seafood campaigns, including consumer guides, seek to strike a balance between accessibility and rigor, but may end up urging for "sustainability" as a general notion and without the comprehensiveness, accuracy, and precision on specific products (Parkes et al., 2010). The majority of seafood guides (Marine Stewardship Council, 2018a; Mr.Goodfish, 2018; Slowfood, 2018; World Wide Fund for Nature, 2018) do also not feature detailed information on both ecological and social aspects of single value chains.

A valuable instrument for communicating the value chain directly to the consumer and a heavily debated governance tool is eco-labeling or third-party certification scheme. The strength of eco-labels lies in the fact that they communicate to the consumer in a simplified manner (through a label) the outcome of a prior rigorous assessment process applied to the value chain. The examples of eco-labels are Marine Stewardship Council (MSC), Dolphin-free label, Iceland Responsible Fisheries, Marine Eco-label Japan Fisheries Certification or Audubon G.U.L.F. (relating to fisheries from the Gulf of Mexico), and many others.[1] Eco-labeling schemes formulate their goal (their definition of sustainability) and then allow third-party entities to independently manage certification and assessment methodology. Third-party certification schemes rely on the power of demand (consumer preferences) on supply (the type of fish being fished and their fishing methods) (Deere, 1999; Roheim, 2008; Ward and Phillips, 2008). The assumption is that whenever a buyer chooses to purchase certified fish, certified fisheries are rewarded for their sustainable practices through that market preference, encouraging in turn more fisheries to undergo certification, and ultimately improving the stewardship of the world's oceans (Marine Stewardship Council, 2011).

The largest eco labeling scheme, MSC, is recognized for having improved the management and production capacities of many fisheries (Agnew et al., 2014), especially in absence of effective governmental regulation (e.g., Gulbrandsen, 2009). However, it has done little for setting an agreeable standard across the fishing industry. It is criticized for its various biases.

First, the acquisition of the MSC label is geographically highly unbalanced across the globe (Marine Stewardship Council, 2018b).[2] Developing countries and small-scale fishing enterprises are lagging behind in certification mainly due to the high fees involved (Pérez-Ramírez et al., 2012; Bush et al., 2013; Sampson et al., 2015; Duggan and Kochen, 2016; Wakamatsu and Wakamatsu, 2017). Second, the market penetration of the MSC labeling scheme is mostly limited to North American and North European countries. Even within the developed world,

[1]The proliferation of eco-labels and campaigns has led to the creation of a form of meta-governance—the Global Sustainable Seafood Initiative (GSSI) Global Benchmark Tool—a reference framework, which benchmarks and provides recognition to reliable certification schemes.

[2]Currently, the two leading Food and Agriculture Organization (FAO) areas (FAO 27—North East Atlantic and FAO 21—North West Atlantic) have more certified fisheries than all the other areas combined (Marine Stewardship Council, 2018b).

certain European fishing industries (especially those from South Europe) with long history and wide variety of marketed seafood products demonstrate little interest for the MSC label (Salladarré et al., 2010). Third, the MSC's understanding of "sustainability" entirely disregards the social aspects and is thus exclusionary and monopolistic (Hadjimichael and Hegland, 2016). Fourth, the MSC has not so much reduced unsustainable consumption as it has implemented a new market of seemingly "sustainable" seafood products along the reinforcement of consumerism (Ponte, 2012; Akenji, 2014; Hadjimichael and Hegland, 2016). Fifth, the MSC's principles are believed to be too lenient and discretionary to be authoritative (Christian et al., 2013). Finally, the MSC is being challenged for its static interpretation of "sustainability" and for lacking incentives for fisheries' improvements once they are certified (Goyert et al., 2010; Bush et al., 2013).

While MSC, being the largest in size, attracts by far the most research, the fundamental lines of critique apply to other eco-labels in fisheries. They relate to legitimacy and credibility, mismatch between the requirements and realities, potential distortions to practices and livelihoods, equity and feasibility, and barriers to trade (Gardiner and Viswanathan, 2004). On the other hand, eco-labeling and voluntary standards are believed to contribute to some positive systemic impact (ISEAL Alliance, 2018). An evaluation, which is still ongoing, has revealed that they create an enabling environment, including the facilitation of a dialog among government, civil society, industry, and producers, as well as raise awareness among the consumers in a particular sector (ibid.).

Apart from the eco-labels, other initiatives exist to provide visibility to certain types of seafood products in the market, specially to differentiate the products by small-scale fishers. Initiatives have emerged in different parts of the world, and include novel approaches to re-organizing the supply chain at stages of branding, marketing, and selling the product, including creating own labels (Witter and Stoll, 2017; Penca, 2019; Duggan et al., 2020). Most of such initiatives are local and territory-embedded, even if capitalized on through an international network, such as Slowfood. These initiatives, however, are all deeply entrenched in transnational governance and production networks (Foley and Havice, 2016) and constitute a legitimate standard-setting practice, which recognizes individual seafood products and the production process behind them, comparable to the more globalized, technical standard-setting (Penca, 2019).

The ability of eco-labels and other market tools to communicate effectively with the consumer and to affect the market patterns are important qualities that could outbalance the weaknesses of any single eco-labeling scheme. A key question emerges: Is there a tool that taps into the strengths, while mainstreaming the choice over sustainable consumption?

MANDATORY FOOD LABELING AS A SEAFOOD GOVERNANCE TOOL

The potential role of mandatory food labeling rules in regulating seafood claims has been identified in the context of EU consumer law (Schebesta, 2016), but this tool is conspicuously absent from the various overviews of governance instruments on sustainable seafood. This section reviews the premise of this governance tool and its application in the EU context as a case in point, and then proposes possible modifications in its design to increase the effectiveness, as well as the necessary subsidiary measures in the policy context.

The Rationale for Mandatory Food Labeling

Mandatory labeling is the visual output of a complex body of food information law that addresses multiple objectives, all of which focus on the consumer. Eco-labeling and mandatory labeling have commonalities and differences in their potential to advance sustainability in the context of seafood. They both focus on incentivization, rather than deterrence; combine prescriptive regulation with the potential of the market and rely on the power of information regarding a product. The difference between them is in the authority making the claim (International Organization for Standardization [IOS], 2012): while eco-labels are awarded to products or producers through an independent certification process conducted by a third-party private entity, mandatory labeling originates from public policy, where the regulator's requirements determine the kind of information to be provided on the product. Because it can be made compulsory, information contained within the food label is also more accessible to consumers and can have a broader outreach than eco-labeling.

Mandatory food labeling allows highlighting certain attributes of the product, without making a quality statement or judgment. This is particularly appropriate in the context of seafood, because sustainability can mean different things to different people (Bailey et al., 2018) and is measured by different indicators (Tlusty et al., 2012; Madin and Macreadie, 2015; McClenachan et al., 2016). It can be measured in terms of ecological impact, such as impact of fishing on related species or on ecosystems, animal welfare at harvest, carbon footprint of the product, as well as the socio-economic aspects, such as child labor, fair pay, and inclusion of women. Additionally, nutritional aspects also play an important role in individual's decisions. This renders seafood consumption an act with many possible combinations (Oken, 2012; Hallström, 2019). In such contexts, sufficient information offered on a product can facilitate individual prioritization of parameters, and allows different varieties for different consumers.

This has consequences for the way in which the consumer takes a decision. While eco-labeling informs or *tells* the consumer in a straightforward manner that a certain seafood product is "dolphin-free," "local," or "sustainable," mandatory labeling can ensure that relevant information is available to the consumer, who then decides on the implications and significance of that information. As such, mandatory labeling requires a higher level of consumer engagement with the information. If eco-labels act as a proxy for the consumer's understanding of the resource ecology or production process without requiring knowledge of it (Eden et al., 2008), mandatory labeling requires more background knowledge. In mandatory labeling, a certain level of knowledge investment is needed to allow each individual to assess the product's compliance or adherence with a selected goal. This allows the consumers to make decisions that go beyond opting for "sustainable," which may lack a clear meaning for the consumer (Cude, 1993; Tlusty and Thorsen, 2016; Richter and Klöckner, 2017). **Supplementary Figure 1** illustrates one of the key differences between eco-labeling and mandatory food labeling.

From the systemic point of view, the primary concern of mandatory labeling is to ensure a level-playing field for the operation of a common market and the right of consumers to make informed choices regarding the product. The information included in a label results from several policy concerns. A food label can provide information on product use (e.g., storage instructions), health and safety (e.g., ingredients, health attributes of the product), provenance (e.g., geographic origin), and quality (e.g., nutritional information). But beyond informing the consumer of the qualities relating to health, a food label also allows the communication of ethical claims on broader policy concerns, such as protection of animal welfare or use of genetically modified products. As a result, mandatory labeling can convey key features of the product and its comparative advantage in relation to others, and enable the operation of an internal market, while encouraging dynamic, efficient, and innovative operators (European Commission, 2006). As a result, mandatory labeling ideally benefits both the consumer and the producer.

The crucial question—of concern in this piece—is over the extent to which mandatory labeling enacts new policy concerns. The inclusion of information on nutrition in many countries in 1990s in order to foster nutritionally appropriate food and healthier diets (Wartella et al., 2010) was one example of its adaptive nature. However, it is a rare one, as the legal mandate of mandatory food labeling to pursue "a high level of protection of consumers' health and interests [...], with particular regard to health, economic, environmental, social and ethical considerations" (European Parliament, 2011), has historically developed almost exclusively as a tool of internal market and consumer policy (Purnhagen, 2013). Sustainability concerns have remained outside the scope of this tool.

In fisheries, sustainability concerns had been explicitly confined to the realm of voluntary instruments. When the EU embarked on the sustainability-driven reform of its Common Fisheries Policy (CFP), there was a strict division between narrow consumer concerns (to be captured by mandatory labeling) and ecological sustainability (to be reserved for voluntary eco-labeling) (European Parliament, 2013a,b). The proposition that a more empowering role of the consumer (European Parliament, 2016) was not needed was partly justified by the CFP's (now missed) target to become entirely sustainable by 2020. Nevertheless, the EU strengthened the requirements for mandatory labeling of seafood, based on the reasoning that for the CFP to be a success "it is essential that consumers are informed, through (...) the importance of understanding the information contained on labels" (European Parliament, 2013a). This approach reflects some incoherency in the underlying logic, where the consumer should be informed on some aspects, but not those regarding sustainability. As the EU is becoming explicit about its commitment to empower consumers to make informed, healthy, and sustainable food choices through mandatory labeling (European Commission, 2020), this provides a test for the political will to deploy mandatory labeling in seafood to that end.

Application of Rules and Their Evaluation

Rules on seafood labeling vary across jurisdictions and typically also contain different requirements for different types of products (fresh, prepared, preserved, processed, cooked, or canned) and for different species. The analysis of the application of this tool is thus inherently selective. We focus on the EU rules, which are believed to be at the forefront of requirements for labeling and have also served as a source of inspiration for non-EU countries (European Commission, 2018b), and the segment of fresh seafood product, which have the most comprehensive requirements.

The label of a fresh product in the EU must include the following elements: commercial designation and scientific names, fishing gear and catch area, information on whether the product has been defrosted, a "best before" date, and allergens. It must also contain information on its provenience; for fish caught in the Northeast Atlantic and Mediterranean and Black Seas, the label must display the name of the sub-area or division, along with a name that is easy for the consumer to understand, or present a map or pictogram; for fish caught in the rest of the world the label only needs to contain the name of the area. Other information can be provided, but it is not mandatory. **Supplementary Figure 2** provides a summary of the requirements.

The existing labeling scheme has some weaknesses from the point of view of the consumer's demand for informed decisions and the objectives of the fishery policy. The EU's internal evaluation found out that while the labeling requirements succeeded in achieving a high level of protection of human health and the functioning of the internal market, there is scope for improving the protection of consumer interests and in addressing the challenge of food sustainability and, in particular, food waste (European Commission, 2018b). Furthermore, the majority of consumers do not consult the label to gain insights into the sustainability of the product (Special Eurobarometer 450, 2017).

A key improvement would be to extend the transparency requirements to many more products than fresh and unprocessed, as currently required. The consumer should be able to get the same information when buying processed or canned products, and also in the processing part of the seafood supply chain, such as restaurants, canteens, schools, hospitals, etc. (D'Amico et al., 2016), especially as these are widespread means of consumption.

Further, a set of recommendations can be made regarding the selection of information. The date of the catch (or slaughter in the case of aquaculture) should be included among the requirements. The consumers have confirmed high relevance of this information (Special Eurobarometer 475, 2018). The date of catch seems more meaningful than the current requirement to state the "use by" date. Currently, the consumer is encouraged to buy "fresh" but lacks an objective tool to verify the product's freshness without having access to this piece of information.

Next, consumers should have, and indeed have an interest (Special Eurobarometer 475, 2018) in knowing the flag and port states of the vessel that caught the product, including the fisherman. Information about this already exists in the supply chain but it is not extended to the consumer. Currently, information about the port of landing is optional, while information about the flag state is not a requirement at all. This could be useful, given that different legal requirements apply for EU and non-EU vessels regarding labor standards, vessel safety, phytosanitary norms, and environmental measures—an issue that is considered as constituting inequality and unfair competition (European Parliament, 2018).

Another issue for the consumer in the current organization of information is its contextualization. The indication of commercial and scientific names (e.g., red mullet/*Mullus barbatus*), geographical sub-area (e.g., FAO 22), and fishing gear (e.g., gill net) mean very little *per se*, if the consumer does not have access to the context of that information. Such context is composed of data on the fish stock, fishing fleet exploiting it, and the applicable governing management approach. How can one know if Pacific cod is a more sustainable option than Atlantic cod (Miller et al., 2012)? Should one be careful about buying fish that is undersized or about buying it in a period that is suboptimal for maintaining viable population sizes; if so, what are references for making the right choice? Only access to an appropriate context allows the consumer to support a sustainable purchasing decision or engage with sustainability questions more broadly. It is true that effort has been made to deliver the technical information to a non-specialist target audience, for instance, in form of seafood databases containing information on populations and habitat impacts (FishWatch, 2018) or the stock assessment exercises available to the public (European Commission, 2018d). However, to the extend such information reaches an average consumer, it is currently provided for considerably fewer species than that found on the market and many of the stocks remain un-assessed.

It becomes clear that the list of potentially useful information becomes extensive and thus challenging for implementation. Even if the EU's requirements in principle require a certain level of market transparency, they fail to make this transparency serve the consumers—both average and educated, which is contrary to the objectives (European Commission, 2018a). A fundamental reform would be needed to accomplish a science-based, but also practical implementation that is informative for the consumer.

Scope for Improved Implementation

An improved mandatory seafood labeling should harness the trends and opportunities for implementation. The key among them are the rise of traceability, both as a norm and a set of technological capabilities facilitating the flow of information and connection among them.

Traceability as a quality is enabled by a system that transmits data in an accurate, timely, complete, and consistent way, and allows verification of the claims once they are made. Developing from its original purpose of responding to food safety and food quality concerns (FAO, 2017a), traceability is gaining traction in preventing, deterring, and eliminating IUU fishing (FAO, 2001; Hosch and Blaha, 2017). Besides that, traceability seeks to respond to conscientious consumers' demand to have access to reliable information about their products through all stages of production and distribution, including verification of seafood fraud (Fiorino et al., 2018). For instance, tracking of the product's route could serve to consider the various intermediaries in seafood trading and the product's carbon footprint—a feature that is increasingly significant for any product, including seafood (Madin and Macreadie, 2015). Ultimately, it could also extent to the post-landing stage, where energy consumption, use of chemicals, waste handling, and wastewater emissions are important (Thrane et al., 2009). A simultaneous demand from both the consumers (for reasons of awareness) and the public authorities (for reasons of IUU and food safety) generates an opportunity for the establishment of comprehensive regulatory structures and systems with applicability in both consumer empowerment and the fisheries policy.

The development of reliable and verifiable traceability systems can be greatly facilitated by the advances in methods for both data collection and data transmission. Various geochemical, biochemical, and molecular methods enable reliable results about provenance of seafood products (Leal et al., 2015; Fiorino et al., 2018). Simultaneously, technologies, such as the internet of things, blockchain, and bar/QR coding are capable of considerably advancing the flow of information along the supply chain (Badia-Melis et al., 2015; Deloitte, 2017; Probst, 2019). Proliferating initiatives, such as *Fishcoin*, *TraSeable Solutions*, *Provanence*, and others, demonstrate the possibilities for cooperation between fishing industry and (blockchain) technology companies in making the journey of fish from "bait to plate" perfectly transparent and traceable (Blaha and Katafono, 2020).

To ensure full and effective traceability, the existence of powerful new technologies needs to go hand in hand with the development of standardized chain-of-custody process, determining data elements and storage protocols. In other words, an agreement is needed regarding what should be observable and how. A full traceability of products requires a substantial change with regard to the way fish trade is currently done, focusing on the ability to document a number of key attributes of the product or unit anywhere in the supply chain (Borit and Olsen, 2020). However, these requirements clash against the industrial actors and their demands for manageability of the entire food supply chain. The existing traceability requirements are already frustrating some fishermen as they reduce the storage space on boats and they prevent mixing fish from the same species caught in different areas (Ploeger, 2014). In that context, previously mentioned GDST Standard for interoperable seafood traceability represents a pioneering attempt in proposing a standard that is acceptable to the industry. It proposes to change the focus from batch identification to unit identification, and thus from using Lot Global Trade Item Number to Serial Global Trade Item Number as the new unit (Borit and Olsen, 2020). This proposal would essentially make each fish a lot more visible in the supply chain than it is now. It is hoped that this standard, rather than confirming an often adversarial nature of the food industry that prefers to operate with voluntary rather than mandatory labels (Kurzer and Cooper, 2012; Mayes, 2014), indicates a promising evolution in traceability across the sector.

The attempt to transform the sector is not without risks for equity. A particular group to pay attention to are small-scale fishers, who are the largest group in terms of employment and economic reliance, but operate with small vessels and very limited digital resources. A similar challenge is posed by data-poor fisheries, which may not have the sort of data required for assessment. This is not to present digital solutions and traceability opportunities as incompatible with small-scale or data-poor fisheries. Rather, it is to call for carefully designed solutions that integrate concerns of traceability with the policy process of empowering small-scale fisheries (Abalobi, 2019; Zelasney et al., 2020) and efforts for manageable but precautionary risk assessments (Dowling et al., 2019), and create synergies between policy goals. This remark is closely related to the need for traceability technologies and standards to be implemented not just in developed countries, but across the world's fisheries and world's markets. The seafood trade is too global to allow gray zones of non-compliance or significantly different standards of compliance (D'Amico et al., 2014; Bailey et al., 2016a). In that context, a level of regulatory approximation among different jurisdiction is certainly desirable. To contribute to progress, the traceability drive needs to close, rather than widen the existing equity-related gaps.

On the other end of the market, also the consumer is yet to benefit from the potential of digitalization. Here, the mode of delivery of information can be significantly enhanced. Rather than storied directly on the food product itself, information on various determinants of seafood sustainability could be accessible to consumer through a reference, such as a QR code or NFC tag attached to seafood products or packaging. Such remote information can facilitate access to the latest scientific findings and allow a more responsive consumer–market relationship. In many cases, a close and persistent engagement between stakeholders is a condition for spreading harvesting effort across a range or marine species and ultimately improving the status of fish stocks (Abalobi, 2019). A graphic representation of the digital possibilities in a labeling scheme is offered in **Supplementary Figure 3**.

One prominent example of a possible deployment of the use of state-of-the-art technology and stakeholder cooperation is the Global Record of Stocks and Fisheries (GRFS) database. It is essentially an inventory of global stocks and fisheries records. Data on fish stocks from multiple national and regional sources (even if guided by different standards) are processed to allow comparability. More precisely, the fishery records are compiled from the Fisheries and Resource Monitoring System (FIRMS, 2018) and FishSource (the program of the Sustainable Fisheries Partnership; FishSource, 2018), while stocks records are compiled from FIRMS, FishSource, and RAM Legacy Stock Assessment Database (RAM Legagy Stock Assessment Database, 2018). Constant updates on the data from independent, reliable, and authoritative sources are foreseen and provisions are made to integrate new information into the system. Stock and fisheries are linked by a unique IT and semantic identifier that allows tracing each product to its fishery and stock. The GRFS database has a great potential for supporting policy efforts to capitalize on traceability, serving both third-party (private) eco-labels and public initiatives that target the consumer. It could be capitalized on by different countries, despite the differences in their implementation of traceability (Charlebois et al., 2014). A public pilot of this database will certainly generate important lessons on its large-scale feasibility. Overall, the technological capabilities seem to be less impeding than policy considerations to the success of seafood traceability.

Supporting Measures for Implementation

The goal of empowered citizen/consumer through enhanced labeling depends on several adjunct actions, none of which are without challenges. One is tackling the recurring and widespread problem of mis labeling or incomplete labeling (including misidentification), which heavily diminish the effectiveness of any labeling (Miller and Mariani, 2010; Miller et al., 2012; Helyar et al., 2014; Oceana, 2016; Esposito and Meloni, 2017). It is true that an increased number of stakeholders have strong interest in consistent respect of the rules that level the playing field: these are consumers, fishermen, retailers, and intermediaries seeking to add value to their products. Technological advancements in traceability play a significant role in allowing them to verify the claims and check the integrity of supply chains. However, they do not replace the continuous need for labor-intensive inspection and sanctioning. Enhanced governmental investment in monitoring and verification systems, and in non-compliance measures is essential (Wessells et al., 2001; Hosch and Blaha, 2017).

The next big challenge is getting the consumers to utilize the information effectively. Two types of challenges are highly relevant: is the consumer's tendency to become overloaded with information and prone to sub-optimal decisions (Mitchell et al., 2005) and a persistent knowledge-action gap (Owens, 2000; O'Brien, 2012). Behavioral sciences point to the fact that even if consumers possess the relevant information, they are subject to different cognitive capacities and behavioral biases. They suggest that regulation puts into center-stage a real-world, or average consumer, and his or her likelihood to perceive and process information, rather than an ideal or entirely rational consumer. Thus, traditional regulation (consisting of rules and information) is nowadays complemented by the measures to nudge the consumers into decisions (Alemanno, 2012; Lehner et al., 2016). Concrete proposals in the domain of seafood might encompass positioning more sustainable options of seafood products vis-à-vis others or suggesting the recommended portion size. A tempting means for simplifying the complexity of information would be to also introduce grade-like labeling system (using colors or letters to rank products). However, experiments with such approaches have offered mixed records at best (Hallstein and Villas-Boas, 2013; Hilger et al., 2019). However, in essence they fail to communicate the level of nuance in seafood that has been advocated for and is found in seafood value chains (various ecological and social factors). Additionally, in other contexts, grading-like schemes have been questioned for their ability to continuously push for progress both on the side of consumers and producers (Arditi et al., 2013).

Moreover, sustainable seafood consumption should increasingly be viewed from a societal point of view. The insights from social practice theory refuse to look at sustainable consumption as an individualized action and highlight the material and social structures of consumption. They argue that consumption is not only a result of individual, isolated consumption choices, but also of societal norms, shared practices, conventions, and institutions (Heiskanen and Laakso, 2019). In the context of seafood, this proposition creates scope for reconfiguring people's expectations around what species to eat (e.g., transforming notions of "high-value species") and the social meaning of eating seafood.

A most obvious recommendation flowing from these findings is to support heavy awareness-raising activities in order to develop people's competencies to have a more active role in fisheries sustainability. Indeed, it has been proven that the consumers' understanding of the purpose of food labeling and the state of global fisheries significantly improves the chances of its success (Cowburn and Stockley, 2005; Uchida et al., 2014). In the case of mandatory labeling, the consumers' knowledge on the meaning of each element of the label, and on the socio-economic and ecological context of fisheries would need to become a priority.

These assignments also constitute opportunities. While effectiveness of mandatory labeling is conditioned upon education and awareness raising, it also fuels societal knowledge. The conceptual suggestion is to recognize the value of information on the label as both informative (providing consumers with the information they seek) and communicative (indicating to consumers that certain information is important). In the context of seafood eco-labels, it was found that consumer familiarity with these labels stimulates more pro-environmental seafood consumption (Jonell, 2016). Well-designed seafood policy tools can, and indeed should, activate the role not only of consumers in dynamic sustainable markets, but also of environmentally-conscious citizens and as concrete means to serve the promotion of ocean literacy (Jacquet et al., 2010; Gutierrez and Thornton, 2014; Tlusty and Thorsen, 2016). Building on the concept of "citizen science" (Irwin, 2002; Bonney et al., 2009; Silvertown, 2009), there is scope to explore how the citizen/consumer's use of information in the seafood markets can respond to enhanced monitoring of a regime and improved implementation tools, as urged for by both the international regime on fisheries (FAO, 2015, 2017b) and decent labor (International Labour Organization [ILO], 2007).

CONCLUSION

This article has argued for an enhanced citizen/consumer perspective in seafood consumption and governance in the context of policy imperatives. The evaluation of the existing instruments reveals that these are largely inadequate, in scope or in depth, for delivering the necessary information to the consumer or benefitting from their involvement. A proposal has been made for strengthening mandatory labeling requirements as a means of mainstreaming sustainability concerns into consumer decisions and food policy and enriching the sustainable seafood governance toolbox. The purpose was to flag this particular policy instrument, rather than present a full-fledged plan for its use across jurisdictions.

Assuming there is sufficient willingness for implementation of the policy commitments on sustainable consumption of seafood, future research in this area could further dedicate to the implementational aspects of the proposal. These are not only technical, but to a large extent also societal and political, encroaching on the issue of benefit sharing of all types of fisheries. From the regulatory point of view, an issue to consider is the interaction among various policy tools that operate alongside mandatory labeling. The proposal on enhancement of the mandatory labeling does not imply the need to replace or reduce other ongoing efforts that address the underlying causes for unsustainable fisheries. It can be complementary with other consumer-focused tools, such as retailers' sourcing policies or eco-labeling, as well as to governments' efforts, such as implementation of sustainable fisheries management plans and conservation measures, enhanced enforcement, or harmonization of trade and fisheries policies in free trade agreements. It is nevertheless important to envisage how they can effectively run in parallel (European Parliament, 2018). Further, it is important to anticipate that their interaction might change over time. In that context, relying on the consumer as a catalyst for the outcome of fisheries sustainability might well be a temporary measure. In the best-case scenario, or in the long-run, management of fisheries may improve to such an extent that it sharply reduces or even eliminates the need to involve consumers in the decisions regarding seafood marketing, and only effects the consumer's right to information.

One aspect that continues to require further attention is to explore systemic benefits of empowered citizen/consumer. This is strongly related also to the value of demonstrating leadership by certain governments in absence of a joint action. How far do educated citizens, capable of processing a certain amount of information, allow for a dynamic development of the markets and policy, in sync with the availability of new scientific information? It is certainly challenging to make ambitious policies, such as improving scientific engagement and investing in citizens' knowledge, succeed in an extreme information era where evidence does not necessarily trump. The study of consumers in the fisheries regime could form part of the broader endeavors to capitalize on a more transparent, participative, and deliberative kind of governance.

AUTHOR CONTRIBUTIONS

JP designed and conducted research, and wrote the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.598682/full#supplementary-material

Supplementary Figure 1 | Differences between mandatory labeling and eco-labeling regarding consumers' knowledge and decision-making.

Supplementary Figure 2 | Example of European Union requirements for labeling of unprocessed and pre-packaged products, Courtesy of Oceana, https://usa.oceana.org/sites/default/files/global_fraud_report_final_low-res.pdf, adapted from European Commission, https://ec.europa.eu/dgs/maritimeaffairs_fisheries/magazine/en/policy/more-transparency-consumers-new-rules-seafood-labelling-come-force.

Supplementary Figure 3 | A possibility for enhanced mandatory labeling for seafood.

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The Importance of Ocean Science Diplomacy for Ocean Affairs, Global Sustainability, and the UN Decade of Ocean Science

Andrei Polejack[1,2]*

[1] Sasakawa Global Ocean Institute, World Maritime University, Malmö, Sweden, [2] Ministry of Science, Technology and Innovations, Brasília, Brazil

The ocean is highly impacted by human activities, and ambitious levels of science are urgently needed to support decision making in order to achieve sustainability. Due to the high cost and risk associated with ocean exploration and monitoring in time and space, vast areas of the oceanic social ecological system remain under-sampled or unknown. Governments have recognized that no single nation can on its own fill these scientific knowledge gaps, and this has led to a number of agreements to support international scientific collaboration and the exchange of information and capacity. This paper reviews current discussions on ocean science diplomacy, i.e., the intersection of science with international ocean affairs. Ocean science is intrinsically connected with diplomacy in supporting negotiations toward a more sustainable future. Diplomacy supports essential aspects of scientific work such as capacity building, technology and information/knowledge exchange, and access and sharing of research platforms. Ocean science diplomacy underlies the work of many intergovernmental organizations that provide scientific guidance, such as the Intergovernmental Oceanographic Commission (IOC), the Intergovernmental Panel on Climate Change (IPCC), and the International Council for the Exploration of the Sea (ICES), and United Nations Convention on the Law of the Sea (UNCLOS). To illustrate how critical science diplomacy is to global ocean affairs, this paper examines examples of the influence of ocean science diplomacy in UNCLOS. Furthermore, this paper discusses the utility of ocean science diplomacy in support of the UN 2030 agenda, and the UN Decade of Ocean Science.

Keywords: science diplomacy, United Nations (UN), sustainability, Decade of Ocean Science, 2030 Agenda and SDGs, Law of the Sea (UNCLOS), transdisciplinary science

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Françoise Gaill,
cnrs Inee, France

Correspondence:
Andrei Polejack
andrei.polejack@gmail.com

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SCIENCE AND INTERNATIONAL RELATIONS

Science is a universal language that through empirical observation and evidence-based testing stands on grounds of replicability, transparency, and merit in search of the truth (Oreskes, 2019, p. 24). Science facilitates communication and cooperation as scientists seek ways to compare results across time and space to understand reality and socio-ecological phenomena (Wagner, 2002).

Science is generally perceived by society as apolitical and free of values, a search for evidence that enlightens our knowledge (Iñiguez et al., 2012). Despite the important debate in Academia on the political basis of science (Jasanoff, 1998; McCain, 2016), this public perception promotes science as a reliable source of knowledge that is widely used by policymakers and diplomats, from advising policy to reinforcing political values (Weiss, 2005; Pielke, 2007; Oliver and Cairney, 2019).

Modern diplomacy can be understood as a statecraft in building non-violent international relations advising, shaping, and implementing foreign policy (Barston, 2019; Boyd et al., 2019), whereby diplomats protect and promote national values and interests abroad (Kaltofen and Acuto, 2018a). In international relations, science can act as a country's soft power, as opposed to the traditional hard powers of force and coercion (Nye, 2017), reinforcing and spreading national views and values (e.g., House of Lords, 2014). Evidence-based negotiations bridge international relations and science (Kaltofen and Acuto, 2018b), posing a necessity to strengthen the participation of national science and technology communities in negotiation processes (Colglazier, 2016).

As the global community increasingly meets Anthropocene challenges, the integration of science and diplomacy is pivotal (Steffen et al., 2011; Kotzé, 2014). One current example involves climate science feeding diplomatic negotiations at the UN level. The Intergovernmental Panel on Climate Change (IPCC) reports have informed diplomatic discussions and resulted in progressive commitments from countries. From Kyoto to Paris, scientific advice has informed more assertive commitments to reduce greenhouse gas emissions (Ruffini, 2018). A new field of study has emerged to understand this interlinkage between science and international relations under *le chapeau* of science diplomacy (Fedoroff, 2009). Science diplomacy, though a new term, is being increasingly used by policymakers as a way of promoting international engagement around evidence-based decision making (e.g., Pandor, 2017; Moedas, 2019).

This paper aims to present current discussions on science diplomacy and its application in the context of ocean affairs. Here, I review different examples of what constitutes ocean science diplomacy by briefly analyzing the work of some key intergovernmental organizations, such as the International Council for the Exploration of the Sea (ICES) and the Intergovernmental Oceanographic Commission (IOC). A more in-depth analysis is presented for the United Nations Convention on the Law of the Sea (UNCLOS) (hereafter the Convention) and its implementing institutions as critical avenues for the application of ocean science diplomacy practices and power play among States in vital matters concerning ocean affairs. In addition, I explore the relationship between the UN 2030 Agenda for Sustainable Development and the upcoming UN Decade of Ocean Science for Sustainable Development (2021–2030), as both processes result from ocean science diplomacy practices and contribute to the implementation of the Convention. Finally, I discuss the current and future importance of ocean science diplomacy in global governance frameworks, in particular with a view to enhancing sustainability and regional ocean science and technology capabilities.

METHODS

The work presented here results from a literature review and a desktop analysis of the Convention and related implementing instruments. I analyzed the current theoretical discussions around science diplomacy and framed these into practical examples of the Convention's implementation. The evolution of the implementation of the provisions in the Convention can also be assessed by analyzing the annual UN General Assembly (UNGA)'s Omnibus resolutions for Oceans and the Law of the Sea, where States Parties agree on mutual issues of concern and calls for action with regard to ocean health, sustainability, and use. Therefore, I reviewed the last 10 years (2009–2019) of the omnibus resolution in search of the terms "science," "scientific," "research," and "knowledge." I extracted and compiled the full text of the agreed paragraphs that addressed ocean science at some level, to look for the main themes that States called for scientific expertise. By doing so, I present the recent updates on the role of science to international ocean affairs after the adoption of the Convention, as a means to illustrate the role of science diplomacy in progressing matters of common concern in the law of the sea and ocean affairs among States.

PROGRESSIVE EVOLUTION OF A NEW CONCEPT: SCIENCE DIPLOMACY

Science diplomacy practices date back to ancient times (e.g., Turekian et al., 2015). Reports from the negotiations of the Treaty of Kadesh, in a conflicted Egypt in 1300 B.C., show letters asking for doctors to be exchanged between the powers in dispute (Turekian, 2018). Contemporary examples of science diplomacy include the SESAME synchrotron light facility in the Middle East. SESAME has allowed researchers to cooperate in a politically tense region, arranging member countries to form a dialogue based on science (Rungius, 2020).

There is much debate on what science diplomacy means. International relation scholarship has traditionally placed science exogenous to theoretical discussions (Mayer et al., 2014), a picture that is slowly changing due to the political power that science can exercise in international negotiations, in face of global environmental uncertainties. Consequently, science diplomacy has emerged as a new field to understand the interplay between science and international relations, in particular where there are global, transborder, and regional issues of common concern or interest (Berkman, 2019; Flink and Rüffin, 2019). Studies in this field include the influence of science in diplomatic relations, the dynamics of science acting as a source of power between nations, and the support that diplomacy can provide to research and innovation (Flink and Schreiterer, 2010; Leite et al., 2020). In this sense, science diplomacy can be framed as a discipline grounded on the fields of international relations, science–policy interface, and Science and Technology Studies (Fähnrich, 2017). Science diplomacy can also be described as a practice, and some have advocated that this is the dominant view in the literature, based on practitioners' perspectives and requiring further empirical basis (Ruffini, 2020). Science diplomacy as a practice involves the collection, synthesis, and presentation of evidence to international decision-making processes, joint research projects acting as a dialog hub between nations, and scientific cooperation calling society to address humanitarian challenges (Rungius et al., 2018).

Discussions in science diplomacy generally frame the results into two distinct taxonomies due to the lack of a generally accepted definition of the concept. One of those taxonomies was provided by the Royal Society and the American Association for the Advancement of Science as a result of an event held in 2010 (The Royal Society, and AAAS, 2010). The concept is categorized as shown and exemplified in **Figure 1**.

Subsequently, Gluckman et al. (2018) proposed another set of categories that highlight the utility of the concept in transnational relations. According to those authors, science diplomacy practices would fall into three categories, namely:

i Actions designed to directly advance a country's national needs;
ii Actions designed to address cross-border interests; and
iii Actions primarily designed to meet global needs and challenges.

Both taxonomies, when confronted, show a progressing evolution of the concept. The Royal Society and AAAS taxonomy disregarded the role played by national interests in advancing science diplomacy, being brought to the discussion by Gluckman and colleagues in 2018. National interests are an essential part of diplomacy, and science is one of the many features considered in the decision-making process (Ruffini, 2020). In this case, science can both influence but also be influenced by diplomacy, grounded in national political agendas (Flink, 2020).

Globalization has provided many pathways for researchers to collaborate in global environmental agendas and engage with international decision makers, without undue regard to national political agendas (Leguey-Feilleux, 2017). Non-State organizations have been particularly active in engaging society and calling attention to environmental concerns grounded in scientific findings. These organizations, which include non-governmental and intergovernmental organizations, provide scientific evidence to international discussions by preparing policy briefs, community white papers, and side events in Convention of the Parties, for independent discussion based on science. This track 2 diplomacy, parallel to State-led diplomacy, has being identified as a more flexible and forthcoming form of international relations by which science can exercise its freedom and best address societal benefits and community interests (Jones, 2015; Gore et al., 2020). One example of such is the ongoing negotiation at the UN on a new legally binding instrument to regulate the access and benefit sharing of the marine biodiversity in areas beyond national jurisdiction (Harden-Davies, 2018). Science diplomacy facilitates how national political agendas can be brought into balance with community interests, with researchers centrally placed to provide evidence and inform future joint decisions (Legrand and Stone, 2018). As a pay-off, researchers are provided with access

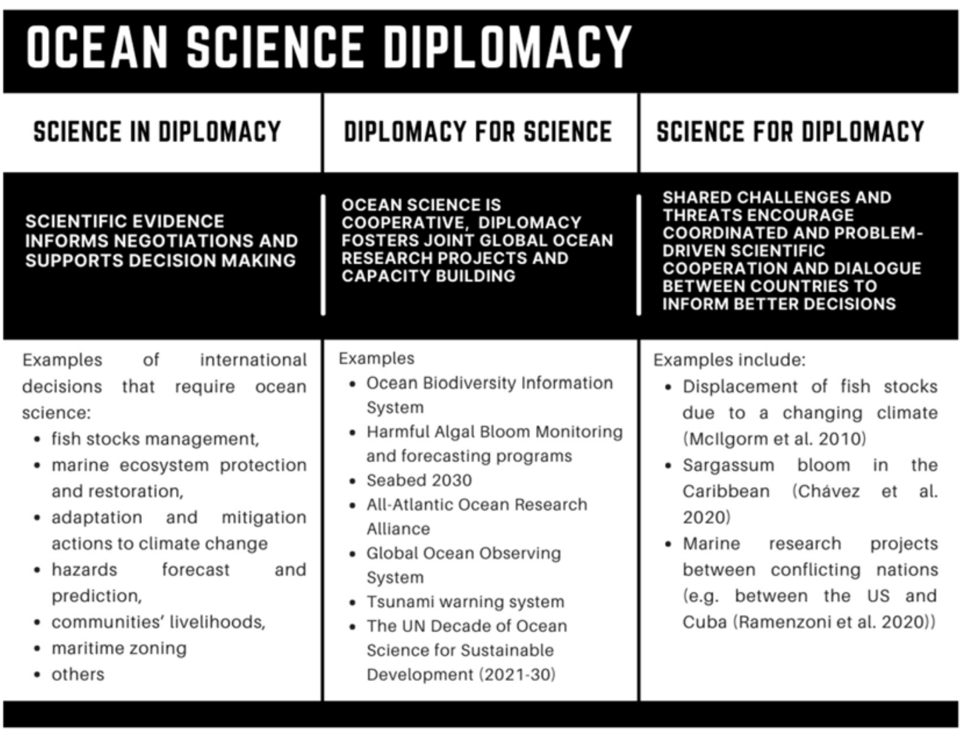

FIGURE 1 | Diagram showing the three categories of Science Diplomacy as informed by The Royal Society, and AAAS (2010), followed by examples of current matters in ocean affairs that illustrate this taxonomy.

to infrastructure and international funding (Berkman, 2019). Consequently, global environmental conundrums are excellent cases for science diplomacy.

THE OCEAN AS A RICH FIELD FOR SCIENCE DIPLOMACY

The ocean supports life on the planet by providing food (Food and Agriculture Organization of the United Nations, 2020), climate regulation (IPCC, 2019), and other essential ecosystem services (Lubchenco and Petes, 2010). Perceived as humankind's last frontier (Gibney, 1978), our relation to the ocean is not only economical (Fleming, 2010), but also social and spiritual (Costanza, 1999). At the same time, the ocean is highly impacted by human activities, including overfishing (Jackson et al., 2001), the loss of biodiversity and ecosystem services (Worm et al., 2006; Hughes et al., 2018), ocean warming (Poloczanska et al., 2013; Cheng et al., 2020), and sea level rise as a direct consequence of climate change (Small and Nicholls, 2003). Ocean ecosystem services are beneficial to humanity in its entirety. Land-locked and geographically disadvantaged States, with low or no proximity to coastal areas, still depend on marine transport systems, as well as food provision, climate regulation, and leisure services from the ocean (Nash et al., 2017).

The marine environment is considered as a global commons, and it is on humanity's best interest to preserve and sustainably use its resources and services (Vogler, 2012; Rudolph et al., 2020). Ocean management relies both on national policies and regulations and on international cooperation (Attard, 2018). Scientists are best placed to identify and comprehend hazardous anthropomorphic phenomena in the ocean, seeking answers to inform policy (Nursey-Bray et al., 2014; Tengö et al., 2014; Sudhakar, 2020). Therefore, ocean science is essential both to assess ocean environmental limits (Baähr, 2017; Nash et al., 2017) and to provide evidence to sustainably limit our efforts on crossing those ocean boundaries (Ingeman et al., 2019).

International non-governmental and intergovernmental organizations play an important role in the international ocean decision-making. For instance, ICES, a North Atlantic intergovernmental scientific body, has been advising policy since 1902, in particular with regard to fisheries management. ICES provides evidence to support regional and national decision making, but also assists countries on crafting their positions in international fora when requested to do so. Advice is delivered by a broad network of scientists who use their peer collaboration to reach out even further and conduct scenario-building, so information is policy-relevant (ICES, 2019). In fact, Robinson (2020a) advocates that ICES has developed subsequent ocean science diplomacy mechanisms, describing ICES critical role in shaping ocean science diplomacy. Historically, ICES is well respected and cooperates closely with other relevant international organizations, such as the IOC of UNESCO.

The IOC is broadly recognized as the international scientific body for ocean affairs at the UN level (Pavliha and Gutiérrez, 2010). It is an institution that has combined science and diplomacy since its inception in 1960. With 150 Member States, IOC has been central in organizing and pushing ocean science under the mandate of the UN General Assembly (UNGA). IOC relies upon at least two definitions of ocean science. First, ocean science includes all disciplines related to the ocean, i.e., the classical fields of oceanography: physical, biological, chemical, and geological, as well as hydrography, health and social sciences, engineering, the humanities, and multidisciplinary research on the relationship between humans and the ocean (IOC-UNESCO, 2017, p. 19). Second, and more recently in the context of the UN Decade of Ocean Science for Sustainable Development, this definition has been expanded to include the supporting infrastructure (observations, data systems, etc.); societal benefits, such as knowledge transfer and applications in regions that are lacking science capacity; science-policy/user interface; and local and indigenous knowledge (IOC-UNESCO, 2020b, p. 2). Although both definitions are debatable, the key message is that ocean science is transdisciplinary in essence and is now being used to fulfill other roles, such as producing goods for social benefit and fostering transfer of technology and capacity development.

THE UNITED NATIONS CONVENTION ON THE LAW OF THE SEA

The Convention on the Law of the Sea sets the rights and obligations of State Parties in relation to the law of the sea and ocean affairs, thereby providing a global ocean governance framework that is almost universally accepted (Koh, 1982). The Convention is a living example of how national interests are balanced with global interests regarding the exploration and conservation of the ocean (Long, 2007). National interests included States claims to extended maritime spaces. Global interests were mainly the expanding threat of unregulated natural resources exploration (Brown and Fabian, 1974). Consequently, the United Nations General Assembly convened the Third United Nations Conference on the Law of the Sea—UNCLOS III in 1973 to discuss ocean matters in plenitude (Koh and Jayakumar, 1977). It was only after 9 years of long and intense negotiations at the UN that the Convention was finally adopted in 1982 and entered into force in 1994. Today, it is the globally recognized regime dealing with all matters relating to the law of the sea, being ratified by 167 States Parties and the EU (United Nations, 2019b).

Science was at the very core of negotiations at UNCLOS III (1973–1982) (Hayes, 2011). Diplomats needed to be supported by scientific information to negotiate Convention matters as well as to rebut evidence presented by other parties. This power of science was very influential to inform the agenda setting as well as the advancement of the negotiations (Brown and Fabian, 1974). For example, during the process of framing the draft provisions of the new treaty, it became evident that countries with better scientific capabilities could drive negotiations by presenting strong evidence that anchored discussions around that information, something called in negotiation theory as the anchoring effect (Furnham and Boo, 2011).

One example of this anchoring effect in ocean negotiations involves the discussions on deep sea mining, which were central to the successful conclusion of UNCLOS III. Evidence on mineral richness and potential commercial value resulted in the creation of the International Seabed Authority (ISA) under the Convention. The ISA is an organization by which States Parties organize, administer, and control activities in the "Area," i.e., the seabed and ocean floor and subsoil thereof, beyond the limits of national jurisdiction [Convention's Art 1 (1)]. The Authority organizes and controls activities guided by the principle that sets the Area as a common heritage of (hu)mankind (Wedding et al., 2015) as adopted by the Convention and later reinforced in the Convention's 1994 Implementation Agreement (Lodge and Verlaan, 2018). Therefore, even States which are not part of the Convention are still bound to the Authority's role in regulating this common heritage as part of customary international law, overseeing equitable opportunities in the Area (Willaert, 2021). ISA's *raison d'être* is basically to apply scientific evidence to regulate both mining and environmental protection, making sure that any resulting benefits are shared among all. The ISA continually develops and enhances codes of conducts and technical guidelines, all based on evidence presented by States Parties. Considering that our knowledge of the deep sea is still inadequate, the lack of sufficient scientific evidence is a common ground, a situation in which the precautionary principle is generally applied (Ardron et al., 2018). However, most Member States to the Convention lack the capacity to produce or evaluate scientific evidence in relation to the deep ocean, leaving those States with higher capabilities to drive the regulatory framework for mining and environment impact assessments of this common heritage of humankind (Wolfrum, 1983).

Historically, disparities in science and technology capacities drove countries to adopt distinct positions in negotiating the Convention. Developing countries recognized their lack of scientific and technological capabilities as a threat, undermining their ability to properly address technical issues as well as progressing on the potential exploration of the marine natural resources and resulting incomes (Hayes, 2011). In addition, sociotechnical imaginaries[1], i.e., technologies that were not yet available or commercially viable, drove developing countries' concerns in relation to sovereignty rights, access, and potential benefit sharing of those explorations (Robinson, 2020b). Developed countries, in turn, were concerned whether the Convention would post obstacles on the conduct of marine research abroad, limiting their access to foreign waters and therefore any potential prospective research on marine resources (Shapley, 1973), in addition that it would require the mandatory exchange of ocean technologies to developing countries. Consequently, the Convention recognized the importance of ocean science in adopting Parts XIII and XIV, addressing Marine Scientific Research and the Development and Transfer of Marine Technology, respectively.

Part XIII calls for international scientific cooperation for peaceful purposes, seeking to diminish the gaps between Member States' technical capacities to implement the Convention. The same applies to Part XIV, in which countries are called to share and transfer marine technologies to less capable nations, so that they can manage their jurisdictional waters and gain the benefits of the resources therein, as well as avail of their rights and discharge their obligations under the Convention. Although essential to the implementation of the Convention, these provisions are among the least implemented (Salpin et al., 2018).

Science in the Convention goes beyond Parts XIII and XIV. For instance, Part XV sets a complex compulsory dispute settlement mechanism for resolving disputes concerning the interpretation and application of the law of the sea (Doelle, 2006). Disputes must be solved peacefully and by negotiation in the first instance, and thereafter by recourse to judicial settlement, such as international arbitration. Resolving disputes are often dependent on the evidence tendered by the parties. For example, if the dispute is about maritime delimitation, countries need to present data on baselines and geological features such as islands, rocks, and low-tide elevations. If it is on natural resources, such as fisheries, evidence on aspects such as fish population dynamics and ecosystems health is needed. In this context, research capacities become a matter of statecraft in international ocean negotiations. Countries with high technical capabilities are best placed to provide stronger arguments that can result in solving disputes in their benefit. Furthermore, scientific experts and their opinions can have a major bearing on the outcomes of judicial settlement (Boyle and Harrison, 2013). Scientific evidence is increasingly decisive in the resolution of international disputes concerning damage to biodiversity and degraded ecosystems (Long, 2019).

CURRENT EXAMPLES OF THE ROLE OF OCEAN SCIENCE IN THE LAW OF THE SEA

There are many examples of how ocean science is essential to implement the Convention, from direct provisions such as Parts XIII and XIV, to provisions indirectly impacted by ocean science, such as dispute settlements and maritime delimitation. We will address a few of these examples regarding how ocean science can be impactful in defining maritime boundaries, setting limits for the exploration of natural resources and regulating access to ocean areas out of national jurisdictions. This non-exhaustive list of examples aims to illustrate the importance of evidence provision to international decision making in ocean affairs.

Boundary Delineation and Delimitation

States Parties to the Convention have the right to define and claim the outer limit of their continental shelf where it exceeds 200 nautical miles. According to Article 76 of the Convention, this right only applies to the seabed and ocean floor and subsoil, not the water column and air space above. This can result in large oceanic areas under States Parties' rights to commercially explore living and non-living resources such as minerals, oil, and gas. As a rule, the establishment of maritime boundaries is within the sovereign powers of countries,

[1] Sociotechnical imaginaries are defined by Jasanoff and Kim (2009) as "collectively imagined forms of social life and social order reflected in the design and fulfillment of nation-specific scientific and/or technological projects." Robinson (2020b) further explores how the ocean imaginaries caused uncertainty in the international community leading to the UNCLOS negotiations.

with the sole exception of establishing the outer limit of the continental shelf beyond 200 nautical miles, which is subject to an important international oversight process and procedural obligations regarding the tendering of scientific evidence to the Commission on the Limits of the Continental Shelf (CLCS). The latter is the body responsible for analyzing States Parties submissions and drawing recommendations on the outer limits of the continental shelf beyond 200 miles. Scientific evidence is all that matters to CLCS, made up of scientific and technical experts, and the outer limit established by the coastal State on the basis of the recommendations of the Commission are final and binding (as per paragraph 8 of article 76 of the Convention). These recommendations can impact demanding States Parties economically, geopolitically, and socially (Suarez, 2013). States Parties had 10 years after the entry into force of the Convention, or until 2004, to submit their claims (as per article 4 of the Annex II of the Convention). Countries with less capabilities to provide such evidence are disadvantaged in exploring their rights over any potential extension of their continental shelf or in meeting the required timeline for making a submission to the CLCS. This shows how technical capacities and scientific evidence are determinant to the Convention's implementation by coastal States. Noteworthy, some countries still proclaim extensions of the continental shelf unilaterally despite the requirements of the Convention (Morales, 2020).

Exploration and Regulation of Living Resources

Another good example of ocean science interaction with the law of the sea is the regulatory framework for the exploration of straddling and migratory fish stocks. This framework was the outcome of its own diplomatic negotiation after the adoption of the Convention and once again ocean science played a central role in its adoption. In 1995, an implementing agreement was adopted under the Convention, with a very long title, namely: the Agreement for the Implementation of the Provisions of the UNCLOS relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, or "Fish Stocks Agreement" (FSA) (United Nations, 1995). The FSA sets the general procedures to manage and conserve fish stocks and is given effect in regional fisheries management organizations (RFMOs), where intense diplomatic negotiations take place, regarding the allocation of fishing entitlements and the setting of conservation and management measures to prevent the collapse of the overall fish stocks. Scientific evidence in the form of stock assessment advice has a bearing on decisions, on the one hand, to close highly lucrative commercial fisheries or, on the other, to facilitate the over exploitation of fish populations. The Agreement provides a solid legal basis for the application of the best available scientific knowledge, the precautionary approach[2], and the ecosystem-based management. Thus, the Agreement is aimed at ensuring that scientific evidence is an intrinsic component of decision making in fisheries with potentially huge economic, social, and environmental consequences (Robinson, 2020a). Ocean science diplomacy has a major bearing on how this evidence is used by RFMOs to address these complex issues and, once again, scientific and technical capacities are of pivotal importance to statecraft and to redressing global conservation concerns (Worm and Branch, 2012).

Unknowns abound in vast parts of the ocean. Many questions remain unanswered by ocean science. Diplomacy walked hand in hand with science even in face of great uncertainties at UNCLOS III and subsequent negotiations on the seabed mining regime in 1994 and the straddling fish stock agreement in 1995. Both science and diplomacy inform all aspects of this engagement. As more evidence becomes available due to progressive availability application of new ocean technologies and research tools, the possibility arises that States and intergovernmental organizations can press ahead in addressing some of the issues left unresolved by UNCLOS III. A case in point relates to the regulation of the access and benefit sharing arising from the exploration of the biodiversity beyond national jurisdiction, or simply BBNJ (Long and Chaves, 2015). The BBNJ negotiating process is currently underway, based on a draft text for this new implementing agreement (United Nations, 2019a). Negotiations are centered in four main themes: marine genetic resources, including questions on the sharing of benefits; measures such as area-based management tools, including marine protected areas, environmental impact assessments and capacity-building, and the transfer of marine technology. The current draft posits the use of the best available scientific knowledge as a guiding principle. Ocean areas beyond national jurisdiction are among the least known by science, so this agreement, if successfully negotiated, can improve the scientific endeavor needed to unveil almost half of the Earth's surface (St. John et al., 2016). Scientific evidence will be determinant to identify the source of living resources and to advance in marine omics. Diplomacy will be essential to foster programs of capacity building and transfer of marine technology. In addition, the governance of international marine protected areas and the conduction of ecosystem impact assessments will rely intensively on the dynamics between science and diplomacy. Thus, BBNJ is a new interesting case of science diplomacy in action, as pointed out by Harden-Davies (2018).

OCEAN SCIENCE IN THE UN GENERAL ASSEMBLY RESOLUTIONS

In the previous section, we presented examples of major aspects of the law of the sea which require science to inform State practice as well as diplomatic processes under the Convention. Since the Convention does not hold regular Conference of the Parties as other UN conventions do (e.g., Climate Change Convention), the evolution of themes that concern States about ocean health can be assessed in the annual omnibus resolution on the ocean and law of the sea adopted by the UNGA. These UNGA resolutions reflect the progress that is being made and the challenges that arise in implementing the Convention, along with emerging issues of States Parties' concern.

[2]Art. 6 (2)—States shall be more cautious when information is uncertain, unreliable, or inadequate. The absence of adequate scientific information shall not be used as a reason for postponing or failing to take conservation and management measures.

Table 1 presents the full extract of the adopted paragraphs in a 10-year timeline (2009–2019), with the corresponding numbering of each paragraph for further reference.

Over the past 10 years, ocean science issues of concern have increased, resulting in UNGA's omnibus resolutions to expand each year in term of the number of paragraphs as well as in terms of themes covered. Three issues have been present for the past 10 years. First, the UNGA has adopted a *chapeau* paragraph stating how important ocean science is to advance knowledge, provide well-being, and contribute to decision making. Second, ocean science was acknowledged as essential to improve risk management tools in conserving and managing vulnerable marine ecosystems. Lastly, ocean science is essential to the establishment of marine protected areas. Another recurring theme since 2010 is the use of ocean science to identify and protect ecologically or biologically significant areas. In brief, science was identified as relevant for social, economic, and cultural benefit as well as more generally to promote marine conservation. More recently, there has been a distinct focus on the issue of pollution in the UNGA's resolution, with marine litter and underwater noise being addressed since 2016 and 2018, respectively. Looking at this 10-year sample, we can identify that once a subject is incorporated into the UNGA resolution, it remains there. Such a feature opens to the possibility of two hypotheses: (i) there is an inefficiency of the adopted measures to solve those issues or (ii) there is a lack of sufficient scientific evidence to support effective conservation measures. These two hypotheses open a series of questions on the efficiency of UN actions toward ocean conservation. Efficiency in this case is of course dependent on States' national policies and regulations, which are very diverse on the use of the available scientific information. Further research on how UNGA's annual resolutions are impacting national policies shall be necessary and the Sustainable Development Goals (SDGs), as we will discuss later, can present a good case. Science diplomacy can be challenged in this sense on how effectively it is producing better policies and public goods. For now, provisions on the importance of ocean science are thus recurring items of the UNGA's resolution. Accordingly, it can be expected that the progressive implementation of the UN Decade of Ocean Science for Sustainable Development (2021–2030) shall be continuously updated in years to come.

THE UN DECADE OF OCEAN SCIENCE FOR SUSTAINABLE DEVELOPMENT

The Decade of Ocean Science shall be an important opportunity for science diplomacy to target global community interests in spite of national interests in the ocean.

The Decade targets seven societal goals, with ambitions to achieve a clean, resilient, productive, safe, well-observed, documented, and predicted ocean (Ryabinin et al., 2019). It also envisages engaging with society and delivering results for an evidence-based decision making, based on sustainability and peace. Ocean scientists are being urged to break the silos and work closely with international affairs and purveyors of traditional knowledge.

Scientists are answering this call and are expecting much from the implementation of this UN Decade (Claudet et al., 2019). The Decade presents itself as "an important opportunity to address gaps in ocean science, increase knowledge, improve synergies, and support the sustainable conservation and management of marine resources" (A/RES/74/19, para. 301, **Table 1**). The Decade's roadmap (IOC-UNESCO, 2018) highlights how critical it is to coordinate and cooperate in ocean sciences to progress sustainable development. Four distinctive aspects of the role of ocean science diplomacy are highlighted below around the thematic areas of inclusivity, sustainability, inequality, and community interests.

Enhancing Inclusivity

Perhaps, a major oversight to date is that official documents from this Decade primarily highlight natural science's evidence, with far limited participation from social sciences. The seven societal goals themselves very much reflect the gaps identified by traditional natural science, such as oceanography and hydrology. These gaps have been already identified in several documents (e.g. Inniss et al., 2017; IOC-UNESCO, 2017, 2019; Miloslavich et al., 2018) which, up to this point, have been largely unsuccessful in producing the desired change through decision maker's actions.

In times when Governments are failing to implement effective solutions to global problems and trust in science is diminished, public engagement becomes essential (Colglazier, 2020). Social sciences can provide evidence in support of actions to improve public engagement and science uptake in decision-making processes (Bennett et al., 2019). Thus, this UN Decade of Ocean Science should be a turning point for a more equitable participation of knowledge producers and users (along with the difficulties in identifying them). In this context, it needs to be transdisciplinary. Transdisciplinary actions in the Decade of Ocean Science need to start by building up research questions and hypotheses among different disciplines and stakeholders (as in Rudd, 2014). As Jahn and colleagues propose:

> *Transdisciplinarity is a critical and self-reflexive research approach that relates societal with scientific problems; it produces new knowledge by integrating different scientific and extra-scientific insights; its aim is to contribute to both societal and scientific progress; integration is the cognitive operation of establishing a novel, hitherto non-existent connection between the distinct epistemic, social–organizational, and communicative entities that make up the given problem context.* (Jahn et al., 2012, p. 8)

Therefore, the UN Decade of Ocean Science for Sustainable Development is an opportunity to change how scientists organize themselves around a common goal, as well as interact with policymakers and society in general (Wisz et al., 2020). In turn, it can represent an avenue for society to better acknowledge science and engage in science making through citizen science (Schrögel and Kolleck, 2019) and be empowered through Ocean Literacy (for further readings on the later, please refer to Santoro et al., 2017; Squarcina and Pecorelli, 2017; Marrero et al., 2019).

TABLE 1 | Exact extracts from the United Nations General Assembly resolutions on oceans and the law of the sea in which references to marine science or scientific are made. Ten years of exerts (2009–19)[a].

Original text in the resolution	Year and corresponding paragraph in the original text										
	2009	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Recalling that marine science is important for eradicating poverty, contributing to food security, conserving the world's marine environment and resources, helping to understand, predict, and respond to natural events, and promoting the sustainable development of the oceans and seas, by improving knowledge, through sustained research efforts and the evaluation of monitoring results, and applying such knowledge to management and decision-making	Preamble	Preamble	Preamble	Preamble	Preamble	Preamble	Preamble	Preamble	Preamble	Preamble	Preamble
Reaffirms the need for States, individually or through competent international organizations, to urgently consider ways to integrate and improve, based on the best available scientific information and the precautionary approach and in accordance with the Convention and related agreements and instruments, the management of risks to the marine biodiversity of seamounts, cold water corals, hydrothermal vents, and certain other underwater features	132	150	173	190	206	221	227	249	252	254	260
Reaffirms the need for States to continue and intensify their efforts, directly and through competent international organizations, to develop and facilitate the use of diverse approaches and tools for conserving and managing vulnerable marine ecosystems, including the possible establishment of marine protected areas, consistent with international law, as reflected in the Convention, and based on the best scientific information available	134	153	176	195	211	226	232	254	259	261	267
Encourages States, in this regard, to further progress toward the establishment of marine protected areas, including representative networks, and calls upon States to further consider options to identify and protect ecologically or biologically significant areas, consistent with international law and on the basis of the best available scientific information	*	156	178	194	210	225	231	252	257	259	265
Recognizes the need for better understanding of the sources, amounts, pathways, distribution, trends, nature, and impacts of marine debris, especially plastics and microplastics, and to examine possible measures and best available techniques and environmental practices to prevent its accumulation and minimize its levels in the marine environment, and welcomes in this regard the work conducted under the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, led by the Intergovernmental Oceanographic Commission, and its report entitled "Sources, fate and effects of microplastics in the marine environment—a global assessment," and the report of the Executive Director of the United Nations Environment Program on marine plastic debris and microplastics, which reviews best-available knowledge and experiences in this regard and gives recommendations for further steps to reduce plastic litter and microplastic in the oceans	*	*	*	*	*	*	*	205	209	210	218
Calls upon States to consider appropriate cost-effective measures and approaches to assess and address the potential socioeconomic and environmental impacts of anthropogenic underwater noise, taking into account the precautionary approach and ecosystem approaches and the best available scientific information, as appropriate	*	*	*	*	*	*	*	*	*	275	281

(Continued)

TABLE 1 | Continued

Original text in the resolution	Year and corresponding paragraph in the original text										
	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Notes the discussions at the twentieth meeting of the Informal Consultative Process, from 10 to 14 June 2019, on the theme of ocean science and the United Nations Decade of Ocean Science for Sustainable Development, during which delegations, inter alia, stressed the importance of marine scientific research, international cooperation and coordination, as well as of a stronger science–policy interface in understanding and effectively addressing the unprecedented pressures on the ocean, provided input to assist in preparing for the Decade and considered that the Decade will be an important opportunity to address gaps in ocean science, increase knowledge, improve synergies, and support the sustainable conservation and management of marine resources, and during which several delegations underlined the important complementary role of traditional knowledge held by indigenous peoples and local communities	*	*	*	*	*	*	*	*	*	*	301

"" means the absence of a paragraph in that year's UNGA resolution.*

[a]Documents included were A/RES/64/71; A/RES/65/37A; A/RES/66/231; A/RES/67/78; A/RES/68/70; A/RES/69/245; A/RES/70/235; A/RES/71/257; A/RES/72/73; A/RES/73/124; and A/RES/74/19. Accessed in June, 2020 at https://www.un.org/Depts/los/general_assembly/general_assembly_resolutions.htm.

Surprisingly, neither the UNGA resolutions nor the Decade's official documents express the importance of science diplomacy as a concept that bring about transformative change in relation to the ocean. All the elements associated with science diplomacy are, however, evident expressly or implicitly in the UNGA resolutions (as discussed above) and the Decade's official documents: science advising policy making, diplomacy relying on evidence, and promoting further research in answer to global challenges, countries overcoming political tensions to address global concerns, and building a science-based dialog. The Decade of Ocean Science is an opportunity to recognize and highlight the importance of science diplomacy in achieving the objectives of the Decade. On this basis, there is a compelling case that ocean science diplomacy should be one of the pillars of this UN Decade for it highlights how multi-stakeholder partnerships are built to deal with global ocean matters, as was done during UNCLOS III negotiations and other international multilateral mechanisms.

Promoting Sustainability

The Decade should be recognized as a science diplomacy process intended to feed into another UN process based on science diplomacy: the 2030 Agenda on the Sustainable Development Goals (SDGs). The Decade's motto "The science we need for the future we want" is a clear reference to the UN document "the Future we Want" that constitutes the basis for the 2030 Agenda (United Nations, 2012), making one effort directed to achieve the other.

The SDGs were established by the UNGA in 2015 as agreed goals negotiated by UN Member States to achieve a more sustainable world. It brings society, economy, environment, policy, and international relations together around 17 goals (Nilsson et al., 2016). The goals deal with social challenges such as poverty, education, equality, as well as environmental concerns related to the ocean, land and atmosphere. They are a result of diplomatic negotiations underpinned by information and knowledge, most of which is scientific, in particular to Earth's capacities to sustain life as we know (Sachs et al., 2019).

Science is particularly important to achieve ocean sustainability, which is addressed by Goal 14—life under water (hereafter, SDG 14) (Visbeck, 2018). SDG 14 has been identified as the most transversal of the 17 (Singh et al., 2018; Nash et al., 2020), although not considered as a priority in almost all political settings in different regions (Custer et al., 2018). When it comes to investment and development, leaders typically choose other priorities which are not environment themed, like education (Goal 4), peace and justice (Goal 16), and decent work (Goal 08) (McDonnell, 2018). Goal 14, however, is the only one that has an explicit call for more investment in science and technology[3], which complements the aims of the UN Decade of Ocean Science.

[3]Objective 14.A—Increase scientific knowledge, develop research capacity, and transfer marine technology, taking into account the Intergovernmental Oceanographic Commission Criteria and Guidelines on the Transfer of Marine Technology, in order to improve ocean health and to enhance the contribution of marine biodiversity to the development of developing countries, in particular small island developing States and least developed countries.

The existence of SDG 14 was made possible through an intense science diplomacy process at the UN. Small Islands Developing States (SIDS, but also known as Large Ocean States), pushed for an ocean related SDG that would bring their concerns forward and were skillful in presenting sufficient evidence on how their livelihoods are affected by a healthy ocean system (Quirk and Hanich, 2016). This diplomatic effort exemplifies how democratic ocean science diplomacy can be. SIDS countries usually have limited research capacities and international cooperation is a useful tool to access foreign research infrastructure. By building these partnerships, SIDS have the potential to access foreign funding and infrastructure and drive research projects to their own needs, generating evidence to feed their domestic policies. As a result, the civil understanding of the importance of a healthy ocean has influenced these countries' external policies in search for more just international relations.

Most developing countries and SIDS need to pool resources to access ocean research infrastructure and undertake projects that will enable them to implement SDG 14. Thus, international cooperation is also an important tool to deliver capacity for the 2030 Agenda. Ocean science diplomacy can present the necessary mechanisms for countries to advance their scientific capacities in exchange of granting foreign access to their waters, in a win–win situation. It is therefore necessary to identify where developing countries and SIDS strengths and weakness lie so as to negotiate directly or through competent international organizations in demanding the "fair and reasonable terms and conditions" in agreements, as predicated by the Convention [Article 266 (1)].

Addressing Global Inequalities

As seen previously, the disparities in ocean science and technology capacities between countries are determinant of their success in implementing the Convention and related instruments. Implementing Goal 14 and the UN Decade of Ocean Science will be particularly challenging for developing countries. Not many countries in the world have access to the necessary technology and human capacity to deliver ocean science, especially due to the high costs associated with marine research infrastructure and the challenges to develop and maintain scientific capacities domestically. UNESCO's Global Ocean Science Report (IOC-UNESCO, 2017) highlights the global disparities in science indicators, particularly the production of ocean science publications and citations (**Figure 2**). These disparities result, *inter alia*, in large sampling and knowledge gaps for immense ocean spaces, in particular the Southern parts of the

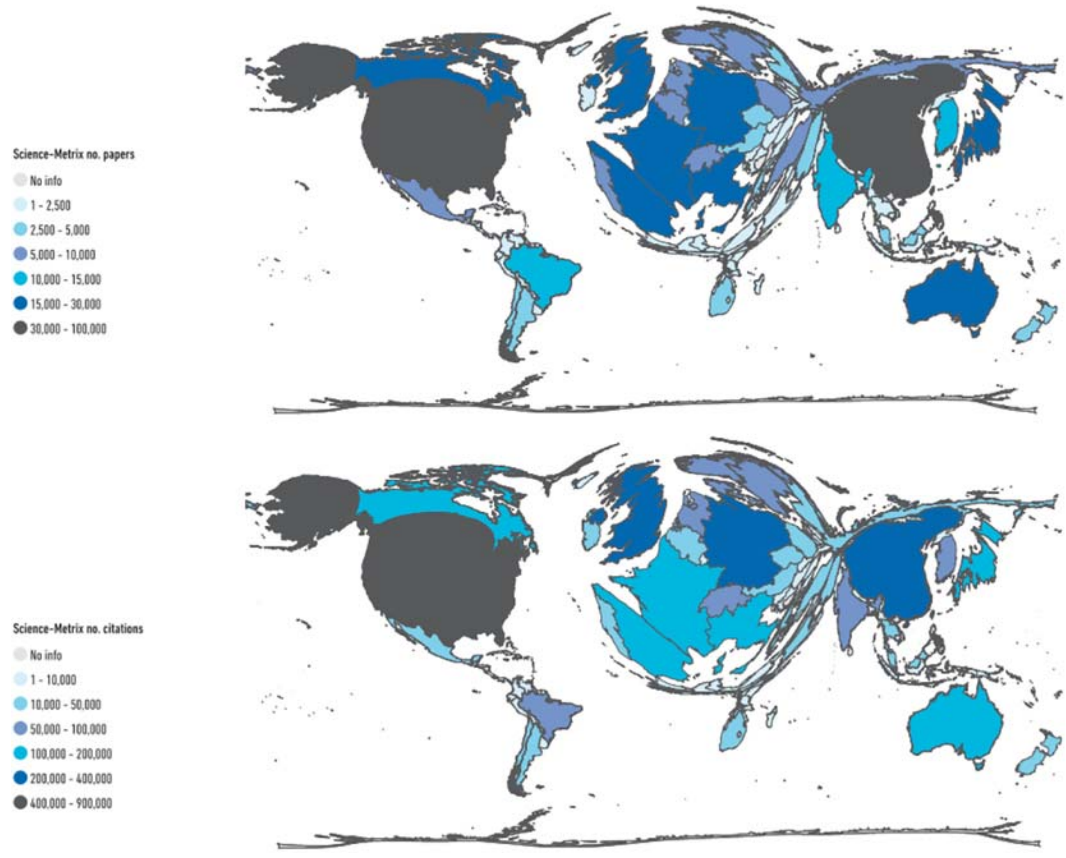

FIGURE 2 | Reproduced from the Global Ocean Science Report (IOC-UNESCO, 2017, p. 28). Original caption: "Publication and citation map of the world. The area of each country is scaled and resized according to the number of ocean science publications (top) or citations received (bottom). Different colors indicate a different number of publications (top) or citations (bottom)."

Atlantic and the Pacific (on the need for a more comprehensive assessment, see Inniss et al., 2017).

While most developing countries depend on foreign research capacities to explore their waters and offshore resources, developed countries gain benefits from accessing other coastal States' waters and exploiting the natural resources therein. Developing countries need to take their geopolitical needs into consideration when negotiating access to infrastructure and scientific capacities with more capable nations. By working together through science diplomacy schemes, they can then enhance their scientific capacities and gain the necessary knowledge to promote better ocean management and sustainability nationally and internationally. In this context, ocean science diplomacy can be a game-changer in finding common grounds of understanding and promoting research capacities worldwide by providing access to research infrastructure and human capacities (Harden-Davies and Snelgrove, 2020). The central issue to be resolved is to understand and apply science diplomacy as an aid to reorganize relevant stakeholders internationally to solve wicked humanitarian puzzles.

Advancing Community Interests

Governments frequently fail to apply the best available scientific knowledge for making decisions, and the ocean science diplomacy framework proposed in this paper shall aid authorities to recognize the benefits in further applying evidence to international policymaking. A force in this regard pertains to organizations that are not under the scrutiny of governments. Non-governmental organizations can have a leading role in presenting updated research evidence and call States to promote change. Non-State actors and international organizations have proven to be effective in promoting the linkage of science and international affairs on urgent ocean matters (Kaltofen and Acuto, 2018a). Experience in international and national decision-making processes over the past three decades demonstrates that NGOs in particular are very effective in gathering experts on certain topics and promoting public concern and engagement around what can be understood as a community interest (Cohen, 2011), communities here being defined as a group of individuals who share common values and concerns (Besson, 2018). Thus, NGOs and other non-State stakeholders promote evidence provision and community interests in international negotiations by organizing the technical debate and assisting delegations with experts and the organization of events. In this regard, these actions should also be considered as science diplomacy practices and a form of Track 2 diplomacy, i.e., diplomacy that happens beyond the formal State channels (Jones, 2015). This para-State form of international relations gives voice to societal concerns and foster community interests that are not necessarily aligned with any country's political view.

As official UN documents call for a stronger participation of knowledge producers and users in both the science and policy making, it will be critical to promote inclusiveness and transparency. Ocean science diplomacy practices in the past and present have broken silos and promoted better communication. It thus represents a tool to assess and foster community interests, by promoting citizenry engagement in both research and decision making. In this regard, the role of indigenous and traditional knowledge has been gaining much attention in ocean affairs and that specific community shares important interests that both scientists and diplomats must consider (Kaiser et al., 2019).

CONCLUSION

Science diplomacy research can promote better coordination and transdisciplinary science in global ocean affairs. Ocean science diplomacy can also ensure the conduct of more effective equitable negotiations and the attainment of fair agreements between States and other entities, including international organizations, by balancing national interests with regional and global shared goals, as prescribed by the Convention. Understanding past negotiations in ocean affairs can help us shape future scenarios where science and international relations leverage expertise and scientific capacity to inform transnational decision making, as exemplified by the success of UNCLOS III and subsequent law of the sea negotiations. Clearly, there is a historical gap in scientific capacities between developed and developing countries (IOC-UNESCO, 2020a). This gap shaped different positions at the UNCLOS III negotiations. However, diplomacy, supported by scientific evidence, was successful in advancing on the adoption of the Convention and establishing mechanisms to address these differences. The necessary diplomacy to overcome those differences involved clustering (e.g., G77 + China, Landlocked, etc.) and trade-offs among States in achieving the compromises and the package of issues codified by the Convention. Capacity building and access to research infrastructure were some of those elements being traded over negotiations, in particular by countries with less capabilities (Nordquist et al., 1990). However, as shown by the Global Ocean Science Report (**Figure 2**), the mechanisms in place to boost research capacity and technology transfer have not yet been effective (Salpin et al., 2018; IOC-UNESCO, 2020a).

With the upcoming UN Decade of Ocean Science for Sustainable Development, there is a chance to look back and to learn from previous lessons in successful law of the sea negotiations. Ocean science diplomacy will be essential in advancing coordination of the necessary elements needed to overcome historical difficulties. The Decade should be an opportunity to understand how ocean science happens in the global south and what is needed to balance these inequalities to deliver the expected results, for instance, in the 2030 Agenda. The Decade not only represents an opportunity to continue long identified but necessary science initiatives, like mapping the entire seafloor (about this ambition, please refer to the Seabed 2030 Project in Mayer et al., 2018) and improving ocean forecast, but also to capture these certainly important actions in a broader framework. This framework will be cognizant of enabling developing countries to thrive in their national ocean scientific capacities in order to contribute over time with the necessary evidence for future decision making. The ocean community needs to leave the assistance provider view and adopt a co-ownership and co-development perspective in relation to transnational processes, so finally “no one is left behind” becomes an imperative for a sustainable future (United Nations, 2016).

Fairness and justice would entail properly addressing intellectual property rights of ocean technologies, discussing benefit sharing mechanisms, investing in local communities, and establishing researchers in key areas so innovation and development would follow. The Decade is a global movement that needs to be dealt with through diplomacy, informed by cross disciplinary ocean science. The invisibility of local researchers that do not have access to ships and equipment, nor are able to calibrate and maintain oceanic instruments, needs to be properly addressed by diplomacy. Business as usual will not solve the problems. The Decade, however, can if it genuinely and successfully encourages partnerships through which change can be made.

Indeed, the effective management of current ocean issues demands broader participation and better communication between sectors, not just scientists and policymakers, but also society, private sector, coastal communities, educators, NGOs, and so on. Since there is still much to be revealed about the functioning of the ocean and science is being called upon to have a stronger societal role, investments need to be made in research infrastructure and human capacities, so our collective will be able to produce the necessary knowledge to feed into public policies and international negotiations.

Our dependency in the ocean is clear: as our life-supporting system or as the basis for many economies, life cannot thrive without healthy oceans. On the other hand, food provision in face of exponential population growth calls for a wise change in the use of marine resources. Science can certainly provide information, but not in the necessary pace. Thus, stakes are high, so are uncertainties, a scenario that fits well within the post-normal science theory (Funtowicz et al., 1991; Funtowicz and Ravetz, 1993). Post-normal science states that if science is to keep producing knowledge in the normal mode, established under the Kuhnian scientific method, it will not be effective enough to address community interests as fast as necessary. Academia needs to break the silos and allow a broader peer review community, encompassing the views from non-Academics into the scientific process (Kønig et al., 2017). By doing so, reorientations can be promoted in accordance with user’s needs and results can be combined with traditional and indigenous knowledge, for example (Nursey-Bray et al., 2014). This approach facilitates better communication and mutual understanding would be triggered around a shared goal, exactly as the UN Decade for Ocean Science and the SDG’s 2030 Agenda are requesting. Further research will be needed to understand the connecting dots on how post-normal science theory can boost science diplomacy mechanisms since both call for a break of silos and stronger interaction.

Society’s participation in science and policymaking should not be undermined (Kahan et al., 2011; Stilgoe et al., 2014; Porter and Dessai, 2017; Squarcina and Pecorelli, 2017). Therefore, further studies on public engagement, public perception of science, and ocean literacy will certainly be key to inform the implementation process of the Decade of Ocean Science. In this context, ocean science diplomacy is one of the possible ways of promoting this post-normal science, allowing inclusive participation of non-experts, and bridging communities. Further research on this aspect should also be promoted.

From a national perspective, countries need to build internal mechanisms to align researchers with policymakers and society to identify gaps and strengths in its science and technology domestic frameworks. This will help enable States to negotiate internationally on fairer grounds. Science diplomacy research can provide good examples of practices that have progressed in this sense, such as the designation of science attachés to Embassies to act together with diplomats in both identifying opportunities for collaboration as well as promoting national’s endeavors abroad (AAAS, 2017). Domestically, appointing science advisors to high Government hierarchies has proven to be an effective way to advance in the science–policy interface that desirably should connect to the country’s external policy in negotiating possible solutions to national challenges (Gluckman, 2014).

Ocean science diplomacy can significantly contribute to global agendas on sustainable uses of the ocean that rely on national policies and international frameworks. It can be a change in balancing ocean research capabilities, allowing a broader participation of scientists and communities in the international decision-making process, and finding some hope for a more sustainable ocean in the future.

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

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Challenges and Recommendations for Equitable Use of Aerial Tools for Mangrove Research

Astrid J. Hsu[1], Sara Pruckner[2], Erin V. Satterthwaite[3,4,5], Lauren V. Weatherdon[2], Keiran Hadley[1] and Elizabeth Thanh Tam Nguyen[1]*

[1] Scripps Institution of Oceanography, La Jolla, CA, United States, [2] United Nations Environment Programme World Conservation Monitoring Centre, Cambridge, United Kingdom, [3] National Center for Ecological Analysis and Synthesis, University of California, Santa Barbara, Santa Barbara, CA, United States, [4] Future Earth, School of Global Environmental Sustainability, Colorado State University, Fort Collins, CO, United States, [5] California Sea Grant, Scripps Institution of Oceanography, University of California, San Diego, La Jolla, CA, United States

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***Correspondence:**
Astrid J. Hsu
ajhsu@ucsd.edu

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As the use of aerial tools such as unmanned aerial vehicles (UAVs) for mangrove monitoring gains in popularity, understanding who leads this research and where is critical for expanding efficient monitoring methods and achieving international commitments to sustainable development, technology transfer and reduced inequality. Between 2000 and 2019, mangrove research using aerial tools was largely conducted in and led by institutions in higher income countries, despite High-income countries accounting for only 9% of global mangrove coverage. Of studies where the country of the lead institution differed from that of the study site, only 38% of the studies included local co-authors. These results echo historical patterns of research conducted by researchers from higher income countries on biodiversity concentrated in lower income countries, frequently with limited involvement of local scientists—known as "helicopter research." The disconnect between where mangroves are located and where aerial research is conducted may result from barriers such as government restrictions, limited financial and technical resources, language barriers hindering UAV deployment, or hampered findability of local research. Our findings suggest that expertise for aerial surveys currently lies in "High-income, Annex II" and "Upper-middle-income, Non-Annex" countries, and both groups could invest time and resources in building local, long-term technological capacity in Upper-middle, Lower-middle and Low-income countries. We identify strategic partnerships to expand aerial tools for mangrove research that also address commitments under the United Nations Framework Convention on Climate Change and potential international collaborations under the framework proposed by the UN Decade of Ocean Science for Sustainable Development.

Keywords: UAVs, ocean observing, capacity development, technology transfer, country income

INTRODUCTION

As the global climate rapidly changes, mangroves have emerged as critical players for adaptation and mitigation, protecting coastlines against storms, and erosion and sequestering atmospheric carbon (Alongi, 2008; Donato et al., 2011). Yet, mangroves have historically high rates of deforestation, and land-use changes continue to threaten the habitat's future (Friess et al., 2019). Mangroves are considered an essential ocean variable to be monitored by the Global Ocean Observing System (GOOS) and are listed as an ecosystem indicator by the Sustainable Development Goals (SDGs) (Malone, 2003; CBD High-Level Panel, 2014; Friess et al., 2019). Long-term observations are necessary to support implementation of country-specific Nationally Determined Contributions (NDCs) of the Paris Agreement and fulfilment of other commitments that foster sustainable mangrove management (Bax et al., 2018). Monitoring mangroves provides baseline data foundational to guiding spatial planning decisions and financial investments that mitigate pressures from climate change, coastal development, pollution, agriculture and other land- and sea-use changes (Schmitt and Duke, 2015; Pham et al., 2019).

Least Developed Countries and Small Island Developing States often lack the resources and technology to carry out sustained observations, which tend to be financially costly and require long-term investment. This is especially true of *in situ* observations, which provide extremely detailed diversity and extent information crucial to local management. Establishing sustained monitoring requires building capacity and transferring technology (and associated technological skills) with these communities (Bax et al., 2018). Lower income countries are especially vulnerable to "helicopter research," the process of researchers from high-income countries conducting field research in lower income countries without local researchers involved in the study or benefiting from the results; a trend spotted in soil science, biology, and genomics (Minasny et al., 2020). Thus, capacity development and technology transfer have been highlighted as priorities by the World Ocean Assessment, UN Convention on the Law of the Sea, Intergovernmental Oceanographic Commission, and the UN Decade of Ocean Science for Sustainable Development (UN Ocean Decade) (Inniss et al., 2016; Bax et al., 2018).

To better observe mangrove ecosystems, researchers have increasingly used remote sensing technologies (Wang et al., 2019). While satellite imagery has been leveraged to provide global estimates of mangrove extent (Bunting et al., 2018), the use of aerial tools is particularly well suited for *in situ* coverage. Aerial tools such as manned aircrafts and unmanned aerial vehicles (UAVs) have captured high-resolution data on many mangrove attributes, including forest extent, species biodiversity, vertical forest structure, and carbon flux (Zulueta et al., 2013; Feliciano et al., 2017; Ruwaimana et al., 2018). Due to their cost-effectiveness, UAVs are revolutionizing conservation management by providing high spatio-temporal observations, especially in small, inaccessible, or highly sensitive areas (Jiménez López and Mulero-Pázmány, 2019). Although the cost is variable, employing UAVs is often cheaper and more efficient than using manned aircrafts or conducting on-the-ground mangrove surveys (Otero et al., 2018; Ruwaimana et al., 2018; Navarro et al., 2020).

This study explores the global distribution of studies using aerial imaging to monitor mangrove forests worldwide in relation to country income and United Nations Framework Convention on Climate Change (UNFCCC) designations. We investigate how equitable access to aerial tools can directly contribute to the societal outcomes of the: (1) UNFCCC, for sustainable partnerships between Developed Countries and Developing/Least Developed Countries (UNFCCC, 1992) and; (2) UN Ocean Decade, for the integration of remote and *in situ* observations to reduce observational costs (A Predicted Ocean) and increase data accessibility to stakeholders (A Transparent and Accessible Ocean) (Ryabinin et al., 2019). Under these frameworks, we detail challenges and recommendations to guide capacity building and technology transfer of aerial tools for mangrove conservation in lower income countries with greater mangrove coverage.

CURRENT GLOBAL DISTRIBUTION OF AERIAL RESEARCH

We investigated differences in the distribution between mangroves and aerial research alongside each country's socioeconomic status and UNFCCC commitment. We referenced the 2016 Global Mangrove Watch extent layer to map the mangrove distribution across countries, cross-referencing these countries with: (i) 2019 income group classifications from the World Bank; and (ii) Annex I, Annex II and Non-Annex designations from the UNFCCC (Bunting et al., 2018; UNFCCC, 2018; World Bank Data, 2019; UNEP-WCMC, 2020). The term "countries" is used interchangeably with economies as defined by the World Bank, and "does not imply political independence but refers to any territory for which authorities report separate social or economic statistics."

To describe where aerial research is occurring and who leads the effort, we performed a manual literature search in English for each country with mangrove coverage on Google Scholar and Web of Science using a formula of key terms: "(country name) + mangroves + (aerial tool)." In place of "aerial tool," the search was repeated for "aerial photography," "drone," "UAV," "unmanned aerial system (UAS)," and "remotely piloted aircraft (RPA)." Search results were reviewed for relevant scientific publications, conference documents, and theses. After three search pages of no additional relevant literature, the search was determined to be exhausted. From each article, we collected where the study was conducted, who led the study, and the country of the lead author's institution. We examined the institutions of all co-authors and noted authors with institutions local to the study site. From our literature review, we analyzed a total of 72 aerial *in situ* studies conducted in 24 countries led by researchers from 19 countries. The majority of the studies used small UAVs to capture imagery (60%), followed by small airplanes (42%) and kites (1%).

We identified three main discrepancies:

1. Gap between mangrove coverage and study site based on income

Based on World Bank income levels, **Figure 1A** shows that global mangrove coverage differs from the countries of study sites. High-income countries published at four times the rate of the amount of coverage they have (mangrove coverage of 11%, but comprise 43% of studies) and Upper-middle-income countries published at 1.5 times their coverage (32 and 44%). Lower-middle-income countries published at 0.25 their coverage (46 and 10%) and Low-income countries published at 0.1 times their coverage (10 and 1%). Thus, mangrove occurrence was greater in lower socioeconomic countries, while mangrove aerial research was conducted in higher income countries.

2. Gap between mangrove coverage and lead country based on Annex designations

Annex II countries led 61% of the studies but accounted for just 9% of mangrove coverage, while Non-Annex countries led just 35% of the studies but represented 88% of mangrove coverage. Study sites were split between Annex II (40% of studies) and Non-Annex countries (56% of studies) (**Figure 1B**). Thus, mangrove occurrence was greater in Non-Annex countries, but research was largely led by Annex II countries.

3. Gap in capacity building

In 21 of the 72 studies (29%), the research site was in a different country than that of the lead institution (**Figure 2A**), and eight of these 21 studies (38%) included a co-author from a local institution. Of these eight studies, seven had lead institutions in High-income/Annex II countries and study sites in two other High-income/Annex II countries, four Upper-middle/Non-Annex countries, and one Lower-middle/Non-Annex country. The last study of the eight had an Upper-middle/Non-Annex country leading a study in a country without a World Bank income.

The gaps presented mirror the broader trend of a mismatch among biodiversity research and areas of high biodiversity. Higher biodiversity tends to be in countries with developing economies, yet biological research tends to be conducted in or by higher income countries (Fazey et al., 2005). The gaps also suggest trends of "helicopter research, and highlight a need to broaden aerial research endeavors and engage local communities in scientific leadership through capacity building and technology transfer.

CURRENT CHALLENGES AND RECOMMENDATIONS FOR UAVS

As UAVs are still an emerging technology, trends of helicopter research could be avoided. The 72 studies identified in this paper are just a sliver of the entirety of mangrove research; however, this problem has been identified across many disciplines and tools (Parsons et al., 2017; Minasny et al., 2020). By focusing on technology transfer, scientific equity, and collaborative processes across regions, the benefits of aerial tools could be harnessed globally and lead to greater collective knowledge generation.

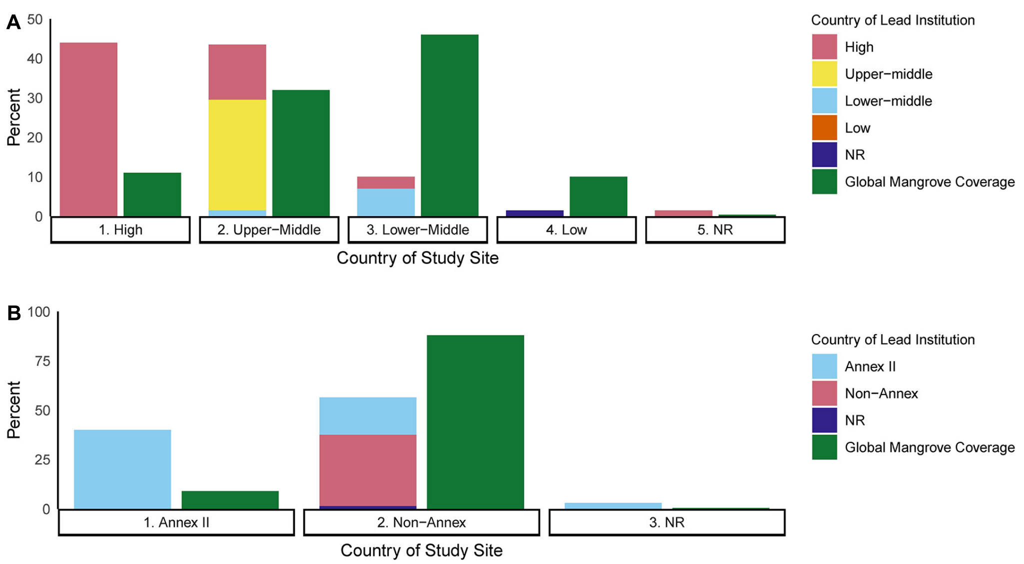

FIGURE 1 | (A) Percentage of studies conducted within each country by income level designation from the World Bank Data (2019). The income designation by the World Bank is not recorded (NR) for the following countries and territories: Mayotte, Anguilla, Guadalupe, Bonaire, Sint-Eustatius, Saba, and Martinique. In green, the percentage global mangrove coverage from Bunting et al., 2018. **(B)** The distribution of studies based on the UNFCCC designations of the country where the study occurred and the country that led the study (lead institution). Percentages are shown in relation to the all studies per UNFCCC party (*n* = 72). Countries or territories that are not listed as a party to the UNFCCC (NR) are French Guiana and Turks and Caicos Island. Percentage of global mangrove coverage by UNFCCC party is also shown.

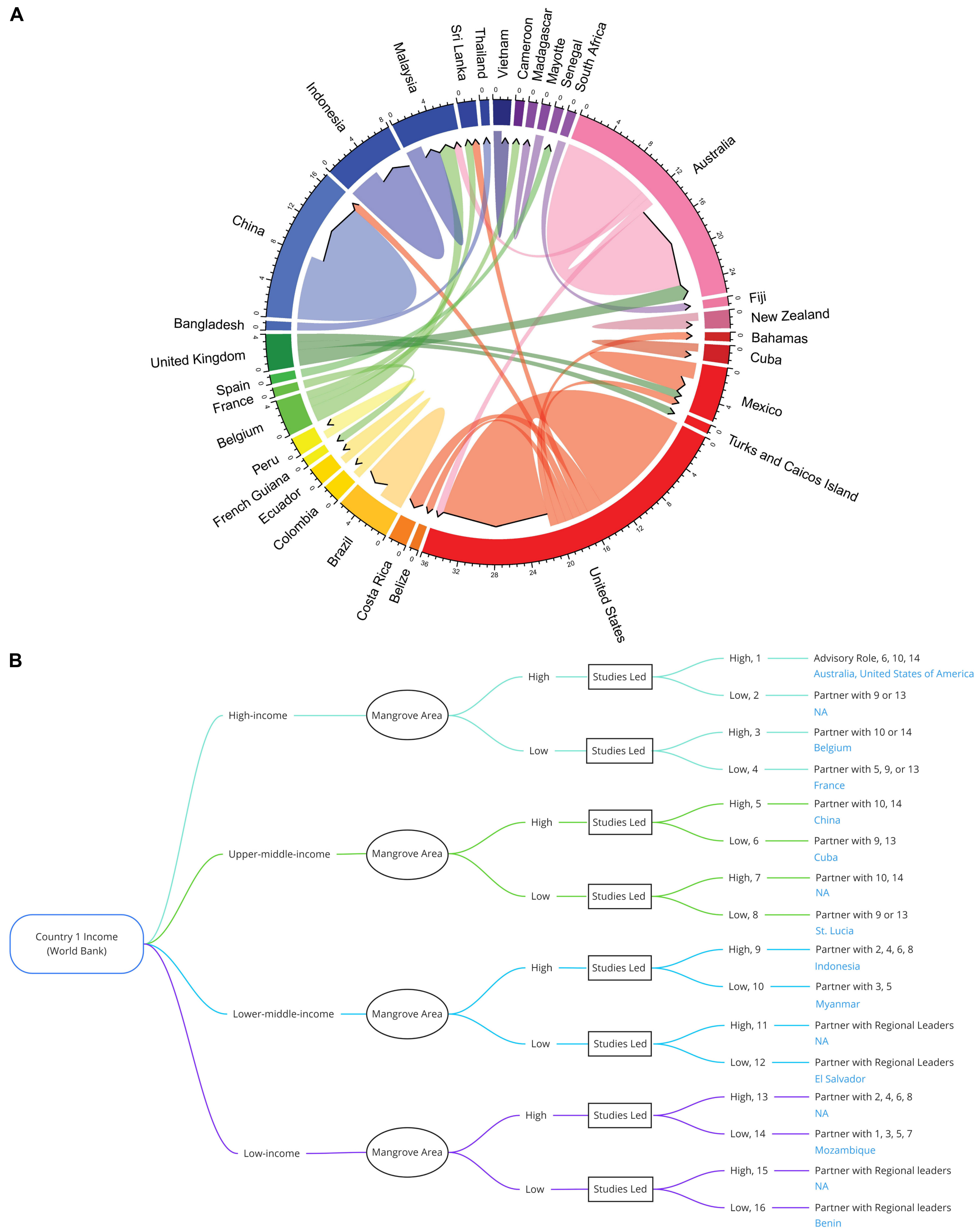

FIGURE 2 | (A) Summary of countries with researchers leading 72 aerial studies, and where those studies are conducted. Continents are represented by the following shades: Asia—blue; Africa—purple; Oceania—pink; North America—red; Central America—yellow; South America—orange; Europe—green. Color of the chord represents the country that is leading the study. Chords leaving the country denote the number of studies led by that country, while black-tipped chords pointing to a country denotes the number of studies that are conducted within that country. **(B)** Decision-tree of suggested partnerships regarding mangrove monitoring using aerial tools. Thresholds of each step is determined as follows: Country income: Levels determined by World Bank; Mangrove Area: High is top 28 countries in global mangrove coverage, which covers up to 90% of the world's mangroves, Low is any country outside of this top 15; Studies led: High is four and above (average of the data set), Low is three and below. Example countries are in blue. NA denotes that no example country was found.

Scaling UAV use for mangroves aligns with the UN Ocean Decade and could be achieved by filling gaps in data generation and resource availability early in the process and following FAIR – findable, accessible, interoperable and reusable – data principles (Wilkinson et al., 2016). This study's results suggest a disparity between countries where mangroves occur and countries currently conducting aerial surveys. This can be explained by two reasons: the data have not been generated and/or were not found in this review. Both are likely true and are addressed below.

Data Generation

Government Restrictions

A lack of aerial data generation in some countries may be associated with restrictive government regulations of the scientific use of UAVs. For example, countries such as Cuba and India have bans against the commercial and scientific use of UAVs while others, including Chile and Colombia, place "effective bans" that technically allow UAV use but with strict requirements and licenses (Jones, 2017; Stöcker et al., 2017). Comparatively, several European organizations (i.e., European Union Aviation Safety Agency) focused on integrating regulated UAV use, allowing greater accessibility for commercial or scientific use (Stöcker et al., 2017). Countries like China, France, and the United States have experimental—and thus more flexible—UAV regulations as well as corporations (DJI, Parrot, Skydio) that manufacture UAVs. To enable emerging technologies to be used and established in low-income countries, regulations should be flexible to permit scientific UAV use while considering the sensitivity of local habitats.

Limited Financial and Technical Resources

Where government restrictions do not inadvertently restrict data generation, limited resources for mangrove research can be another common reason. As shown by several meta-analyses, biodiversity research tends to happen more in higher-income as opposed to lower-income countries, often due to lower research capacities in lower-income countries (Fazey et al., 2005).

To overcome financial difficulties, international collaboration for science and accessible funding sources for low-income country-led studies are key. The UNFCCC provides a framework to facilitate international capacity building and technology transfer in relation to climate change, since Annex II countries have committed financial support and knowledge transfer to developing countries (UNFCCC, 1992). Studying mangroves through aerial research is covered by this, as UAVs are well-suited to estimate local mangrove extent and carbon stocks relevant to NDCs (Ruwaimana et al., 2018). Thus, this study's results can serve as one metric for Annex II countries' mandatory reports on climate finance and technology support, and guide developing countries in their requests for technology and capacity building needs (Ellis and Moarif, 2015).

Outside of Annex II obligations, certain countries may find themselves in the position to lead. Upper-middle-income, Non-Annex countries such as China, Brazil, and Mexico comprised 28% of the studies and 32% of mangrove coverage. These Upper-Income Non-Annex countries may find themselves well-equipped to not only study their local mangroves, but to also act as regional leaders and extend their knowledge to neighboring countries. As such, the expertise for aerial surveys currently lies in High-income, Annex II countries and in Upper-middle-income, Non-Annex countries. These categories of countries can invest time and resources in the transfer of technology and skills to other countries, especially Low-income countries (10% of mangrove coverage) which have only 1% of mangrove studies (**Figure 1**).

Furthermore, international journals could allow researchers and students in low-income countries access to articles free of charge, to promote the development of research capacity. This approach has already been implemented by organizations such as The Royal Society through their Research4Life initiative (Hamilton and Hurst, 2018). Other journals could consider implementing similar initiatives.

Software Language

Language barriers exist during the data acquisition process. With UAVs developed by Western and Chinese companies, languages are often limited to English, Chinese, and a handful of other languages. For example, DJI Ground Station Pro, a mission-planning companion app to DJI UAVs, is currently available in only Chinese, Japanese, and English (DJI, 2020). Likewise, Pix4Dcapture is available in English, German, Japanese, Spanish, Chinese, and Portuguese (Pix4D, 2020). The incompatibility in languages between the software and the end-users can be a challenge for capacity building of aerial technology. Private-public relationships and increasing demand for Unmanned aerial vehicle products could encourage broadening of available languages for associated software.

Data Findability and Accessibility

Our findings may also indicate that existing aerial mangrove observations and research conducted in Non-Annex and lower income countries are often not readily available online or are published in national journals or documented in other languages that hinder their integration into global baseline datasets, which are predominantly curated using the English language.

An additional challenge relates to making data accessible, which is a multi-faceted problem in itself. Often, data are not under a clearly defined data license, and "open access" itself is often insufficiently defined and communicated. Creative Commons licenses can help clarify different levels of access, and more organizations are promoting and implementing open data policies. Specific data contributor agreements often need to be put in place to use small, individual datasets together, which can be time-consuming and costly to implement.

DISCUSSION

To address these barriers, we recommend leveraging current frameworks for best practices and establishing new partnerships to facilitate knowledge sharing and technology transfer.

Leveraging Current Frameworks for Best Practices and Shared Methodologies

To support building capacity and the technology transfer of aerial tools for mangrove conservation, sharing the methodologies, metadata, and current data is crucial. Knowledge sharing can be done through the Global Mangrove Alliance (GMA), an alliance of global mangrove actors and stakeholders. The GMA's extensive network encompasses a wide range of users, and their

Mangrove Knowledge Hub acts as a clearinghouse for accessible mangrove-related information. Adding resources for UAVs to the Mangrove Knowledge Hub can help increase the findability and accessibility of methodologies, and standardize methodologies. Further, through the GMA, encouraging members and non-members to deposit gray literature into the resources library can help elevate these studies, which can further increase the findability of gray literature—especially non-English literature. Findability of gray literature is further supported by platforms such as ePrints in Library and Information Science (E-LIS) which focuses on open science in multiple countries and over 27 languages (De Robbio et al., 2020). Effort on behalf of government and academic institutions to promote FAIR principles would also greatly reduce the "grayness" of literature and promote findability without the need to build new facilities or infrastructure (Schöpfel and Rasuli, 2018).

Partnerships

The low percentage of studies including local co-authors when lead institutions are foreign suggests that aerial research of mangroves may fall prey to helicopter research, and that robust partnerships and capacity building efforts are needed. To further support capacity building and technology transfer under the UNFCCC and UN Ocean Decade, developing effective partnerships to shepherd UAV training is needed. Some countries, such as Indonesia, maintain a legal mandate in which foreign researchers must involve local Indonesian scientists as equal collaborators (Rochmyaningsih, 2019). We identified the following potential partnerships to transfer use of aerial tools for mangrove conservation (**Figure 2B**):

- Upper-middle-income or High-income, Annex II countries could partner with Lower-middle or Low-income countries with high mangrove coverage to support training.
- Higher income countries that lead a low number of studies and have low mangrove coverage could offer investments for automated image analysis, such as improving internet access and supporting remote cloud processing (Miloslavich et al., 2018).
- Lower income countries that lead a low number of studies and have low mangrove coverage could train with respective regional leaders.

Even with these partnerships, effective capacity building and training workshops must be carefully conducted. As mentioned, limited language availability of associated software remains an obstacle, and demonstrates the important role of private institutions in expanding their multilingual support (Beekhuyzen et al., 2005). Further, key considerations include determining the appropriate criteria for trainee candidates, the local stakeholders and scientists involved, teaching styles, local infrastructure limitations, and financial considerations (Miloslavich et al., 2018). A high degree of in-person support is often vital to successful capacity development, especially to avoid 'brain drain' of young locals to high-income countries and to foster technical resilience of local researchers. These partnerships can further promote data repatriation and findability, ensuring that these data are held by in-country institutions or hosted on national data platforms to encourage accessibility and use for local decision-making (e.g., Dias et al., 2017; Asase and Schwinger, 2018). The GOOS and UNFCCC provide international guidance to facilitate capacity development and technology transfer (Bax et al., 2018).

Implications

Between 1996–2010, the world lost 12% of its mangroves, with 50% of this loss occurring in Southeast Asia (Thomas et al., 2017). This region not only maintains 34% of the world's mangroves, but the income designations of these countries are largely Low, Lower-middle and Upper-middle-income. There is often an economic impetus to deforest mangroves: in Southeast Asia, mangroves have mostly been replaced by aquaculture, rice fields, and oil palm plantations, which are exportable commodities often held by large corporations (Richards and Friess, 2016), whereas the ecosystem services provided by healthy mangroves are more equitably accessible (Armitage, 2002). Equipping these countries with the tools needed to effectively monitor their mangroves can therefore be a double-edged sword. Changing the political and economic narrative to include the benefits and alternative livelihoods that local communities and the rural poor gain from mangroves is essential (Armitage, 2002).

Implementing and monitoring sustainable mangrove management is crucial to achieving international and regional commitments related to climate and sustainability. With just 45 NDCs including mangroves and global efforts failing to achieve the Aichi Biodiversity Targets, transferring UAVs and associated technology can boost countries' capacity to integrate mangrove-specific targets in their NDCs and contribute to tracking relevant indicators within the post-2020 global biodiversity framework (Gallo et al., 2017; Secretariat of the Convention on Biological Diversity, 2020).

Mangrove ecosystem services, particularly coastal protection, are especially beneficial to developing countries (Barbier, 2016). Thus, capacity building of aerial tools for effective mangrove management can enhance resilience in the most vulnerable communities. *In situ* aerial studies further provide greater local context for on-the-ground issues while strengthening global datasets, but integration of aerial tools such as UAVs into community-based monitoring efforts requires external assistance and financing to build local capacity (Worthington et al., 2020). By sharing methods and data, promoting effective partnerships and FAIR data standards, and implementing mindful training, equitable access to aerial tools for mangroves can be achieved.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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The Contributions of Shellfish Aquaculture to Global Food Security: Assessing Its Characteristics From a Future Food Perspective

Mohamad Nor Azra[1]†, Victor Tosin Okomoda[1,2†], Meisam Tabatabaei[1], Marina Hassan[1] and Mhd Ikhwanuddin[1,3]**

[1] Higher Institution Centre of Excellence (HICoE), Institute of Tropical Aquaculture and Fisheries, Universiti Malaysia Terengganu, Kuala Terengganu, Malaysia, [2] Department of Fisheries and Aquaculture, College of Forestry and Fisheries, University of Agriculture Makurdi, Makurdi, Nigeria, [3] STU-UMT Joint Shellfish Research Laboratory, Shantou University, Shantou, China

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Sebastian Villasante,
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Spanish Institute of Oceanography, Spain

***Correspondence:**
Mohamad Nor Azra
azramn@umt.edu.my
Mhd Ikhwanuddin
ikhwanuddin@umt.edu.my

† These authors share first authorship

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The United Nation's 2030 development agenda adopted in 2015 outlines 17 Sustainable Development Goals (SDGs), and the organization has continued to put food security in the center of its vocalization. Aquaculture is currently the fastest-growing food production sector globally and a sustainable option for attaining food security. Food as a basic necessity for man's survival is always a timely issue. Hence, owing to aquaculture's unique role, it is expected that the demand for aquatic products (especially seafood) will continue to increase due to geometric population growth. Many seafood products are among the critical protein sources in the world. This is partly because they have micronutrients and essential fatty acids that are not present in land-based protein sources. According to the Food and Agriculture Organization, shellfish is one of the main cultured aquaculture groups in the world. Hence, the development of shellfish aquaculture has an important role in sustainable food supply and food security. In this article, an overview of the current and projected contributions of shellfish aquaculture to global food security is presented. Apparently, shellfish aquaculture in the next few decades will have to be intensified to bridge the gap between demand and supply in a cost-effective manner. Also, food waste would have to be reduced and natural resources should be used more efficiently to minimize the negative impacts on aquaculture on the environment.

Keywords: aquaculture, breeding, food security, future foods, shellfish, sustainability, sustainable development goals

INTRODUCTION: AQUACULTURE INDUSTRY

Per the United Nations' agenda for the 2030 sustainable development goal, the relationship between food production and population growth is a very critical issue of discussion. Future food debates have only recently emerged, with the global population projected at 10 billion by 2050, the bulk of whom will be residents in developing countries (5.6–7.9 billion). Food as a basic necessity for man's survival is one of the most popular topics in the last decades, as reflected by the number of publications recorded in the Web of Science database. As depicted in **Figure 1**, the number of food-related publications continuously increased from 2014 to 2019. This number is projected to increase, reaching 85,000 in 2024 alone, as shown by the time series forecast of the Auto Regressive Integrated Moving Average (ARIMA). Therefore, at the current publication rate, the total publication hits might reach 1.25 million by 2024 (**Figure 1**).

There is currently no "one size fits all" approach to meet the expected increase in the demands for food. However, short-term recommendations have been provided to ensure more sustainable food production for current and future human consumption, one of which is aquaculture. Aquaculture is popularly known as the fastest-growing food production sector globally. Hence, it could be exploited to provide sustainable food production in the future. Global seafood consumption (which includes finfish and shellfish), for instance, is growing faster at a mean annual rate of 3.1% (projection from 1961 to 2017) than the global population growth of 1.6% (within the same period). This growth rate is also higher than the growth rate of other livestock and animal production sectors at 2.1% per year (FAO, 2020). This might be because the fisheries and aquaculture sector creates more economic value through production, trade, and marketing (Cai et al., 2019). In addition to that, seafood of aquaculture and wild origin is a significant source of animal protein; hence, it contributes substantially to the overall health as it contains micronutrients and essential fatty acids that are not found in many land-based protein sources (FAO, 2016). Besides the high nutritive content, shellfish culture are ecologically beneficial systems to the environment as they are involved in nutrient cycling. More so, the simple culture techniques help eliminates the need for energy-intensive processes characterized by other aquatic species. Despite the mentioned advantages, sustainable aquaculture production would require knowledge and skills, environmental requirements for culture, favorable policy framework for aquaculture practices, and availability of a large market to drive the production and supply of the cultured species (Broitman et al., 2017). It was estimated some decades ago that a large percentage of the food consumed by man would originate from the sea (Rothschild, 1981). However, as it stands currently, aquaculture accounts for about half of the world's fish supply, and it is projected to grow even further, becoming a crucial part of high-quality protein supply for the global population (Tacon, 2020). Terrestrial and aquatic animal protein sources account for about 43% of the world's protein supply. Given their nutritional values mentioned earlier, they are critical in mitigating malnutrition, especially in low-income countries (FAO, 2010). A significant proportion of global seafood consumption occurs in East Asia and Pacific countries, which could be attributed to the fact that most aquaculture production comes from this geographical region[1].

According to FAO, shellfish[2] are the aquatic invertebrates possessing a shell or exoskeleton. Shellfish consist of mollusks and/or crustaceans, such as mussel, clam, crab, lobster, shrimp or prawn, etc. FAO also defines seafood[2] as human food derived from the sea or marine aquaculture. Future food was described by Parodi et al. (2018) as food whose production capacity is rapidly developing owing to technological advancements, offering the potential to scale up production level. Such capacities also include the ability to reduce production costs and environmental concerns. On the other hand, Karlsson et al. (2018) described future food as nutritious food accessible to everyone with a less negative impact on the environment. Several authors have also given different indirect definitions, including that of Gebbers and Adamchuk (2010). They defined future food as a product of adequate quantity and quality, obtained through sustainable exploitation of resources, and hence, it is environmentally-safe. A more recent definition by McClements (2020) described future food as an affordable, convenient, safe, nutritious, and sustainable product without posing any harm to the environment. Based on these definitions, future food must possess the following four characteristics: (i) adequacy in supply, (ii) reduced production cost, (iii) being environmentally friendly, and (iv) produced through sustainable exploitation of resources.

This paper focuses on the potential contribution of shellfish to global food production, supplies, and food security in the years ahead. Also, an overview of the main criteria of shellfish selection as a future food is provided herein.

SHELLFISH PRODUCTION

There are currently 73 important global aquaculture species listed by FAO along with their "Species Fact Sheet Information," which details the steps to their production and their various cultural aspects. Most of the shellfish species on the FAO list are marine-based and constitute more than half of the total fish group (52.2%) (**Figure 2A**). Therefore, this emphasizes the importance of shellfish as a potential contributor to global future food production, originating from the saltwater ecosystem. Notable among them is the whiteleg shrimp *Litopenaeus vannamei*, Red swamp crawfish *Procambarus clarkii*, Chinese mitten crab *Eriocheir sinensis*, Giant tiger prawn *Penaeus monodon*, and Mud crab *Scylla* sp. based on the latest aquaculture production value and statistics (Tacon, 2020). This is not to say that other crustacean has less potential as future food. It is opined that as research on other candidates intensifies, their production, consumption, and values are likely to increase hence becoming important future food candidates.

Global aquaculture production projections have shown that shellfish are among the most valuable groups for culture (**Figure 2B**). In this report, we analyze the development of shellfish aquaculture using FAO data on global aquaculture production projections (FAO, 2020). Although the data generated affirms the popularity of finfish as the most cultured group of aquatic species (**Figure 2Bi**), the shellfish groups are also top the list in terms of value when compared to other individual groups of finfish or aquatic plant (**Figures 2Bii,iii**). The growth in global shellfish aquaculture production from 1985 to 2018, shown in **Figure 2C**, reveals that the production of shellfish has increased by 10-folds with a total production capping at 27 mmt in 2018 as compared to 2.76 mmt in 1985. This resulted in an increase

[1]Data taken from: https://ourworldindata.org/

[2]Definition of the term available at: http://www.fao.org/faoterm/en

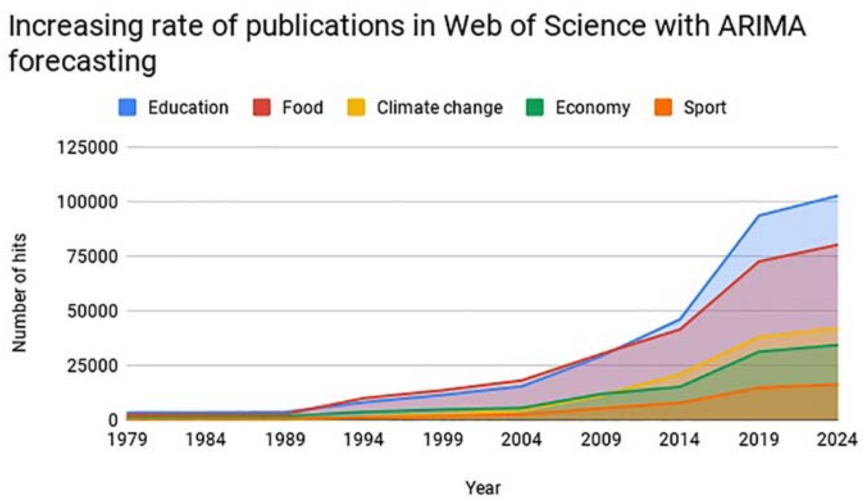

FIGURE 1 | Increasing rate of publications in the Web of Science database between 1979 and 2019 on the topic of food. The red curve represents the cumulative number of documents (left-hand scale). The dark blue column represents the number of documents published per year (right-hand scale). Data taken from https://apps.webofknowledge.com, searched on 15 October 2020. The 2024 data was estimated using Auto Regressive Integrated Moving Average (ARIMA) time series p,d,q: 1,1,1.

in shellfish global revenue from USD 3.56 billion in 1985 to USD 104.55 billion in 2018.

SHELLFISH AS POTENTIAL FUTURE FOOD: INDICATORS

Identifying suitable seafood as future food requires information from a multitude of sources, including research articles, patents, strategic reports by international organizations, and notes from think tanks support groups. This section identifies five indicators shellfish are required to comply with to qualify as potential future food. Three of these indicators are linked to the production phase (i. response and tolerance to biotic and abiotic stress, ii. availability of biological and technical knowledge, and iii. life cycle and broodstock maturation period, while two concern the consumption phase, iv. nutritional value and health benefits, and v. demand, cost, and affordability).

Response and Tolerance to Biotic and Abiotic Stressors

As climate change continues, potential future food (i.e., shellfish) will be those species that can mitigate the climate effects as dictated by the changes in the biotic and abiotic stressors of the environment. The study by Gong et al. (2015) revealed that low temperatures decrease growth and lengthen the intermolt periods of mud crabs, *Scylla paramamosain*, while elevated temperatures stimulate growth and shorten intermolt periods. Azra et al. (2019) also noted that the blue swimming crab, *Portunus pelagicus*, instars had a decreased intermolt period and duration of exuviation when reared at a high temperature of about 32°C. Although many marine shellfish species are more versatile than others in terms of thermal tolerance (Sunday et al., 2012), there are shortcomings to their consideration as primary future food sources. One of which is their low survival rate in captivity (Azra et al., 2019). Consequently, this is an important priority area requiring immediate research attention to enhance shellfish survival through improved breeding technologies (i.e., genetic improvement) and optimization of environmental conditions during culture. Enhanced survival and production characteristics of shellfish in the presence of rapidly changing biotic and abiotic stressors caused by efforts put into maintaining or increasing production efficiency is also another essential domain of research to be considered.

Availability of Biological and Technical Knowledge

Plenty of successes in aquaculture operations can be attributed to a sufficient understanding of the biological and technical aspects of the cultured species. For example, efforts put into perfecting the intensive culture of *Litopenaeus vannamei* since 1973 have led to widespread culture around the world (Briggs et al., 2004). However, the inadequate knowledge about the breeding technology for commercially valuable crabs (e.g., *Scylla olivacea*) has staggered the growth of the industries as its captive culture relies heavily on wild-caught seedlings and gravid females (Ikhwanuddin et al., 2014). Thus, for every potential future seafood candidate, information about their biological and culture techniques must be researched to attain sustainable mass production. Today, sufficient biological and technical information/knowledge are available on some crabs, marine bivalve, shrimps and many other shellfish. This partly justifies their candidature as potential future food.

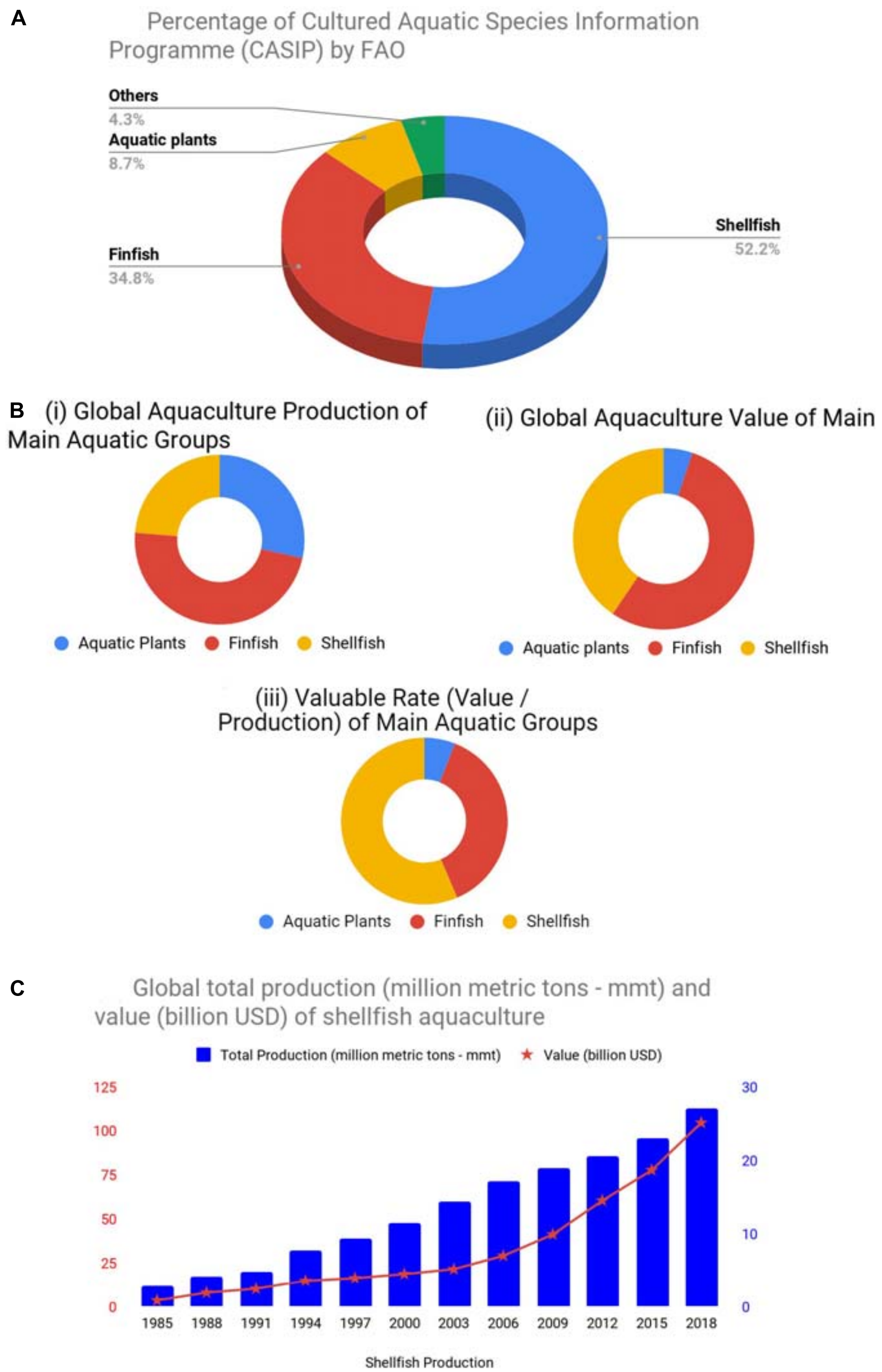

FIGURE 2 | (A) Global species information provided by FAO, excluding the anadromous and catadromous fish species on the original list. **(B)** Comparison of (i) aquaculture production with their (ii) value and (iii) rate for the main aquatic groups of finfish, shellfish, and aquatic plants in 2018 and **(C)** global total production (million metric tons) (bar graph) and its value (billion USD) (line graph) of shellfish aquaculture from 1985 to 2018. Data taken from http://www.fao.org, searched 15 October 2020.

Life Cycle and Broodstock Maturation Period

It is important to note that a cultured species' short life cycle will be translated to fast food production. Hence, to generate enough future food originating from seawater, species with relatively shorter life cycles must be considered. Most marine shellfish fall into this category when compared to the relative life cycles of marine fish species. This includes their embryonic development up onto the market size. Taking mud crab *Scylla serrata* and *S. olivacea*, for instance, the embryonic stage to market sizes ranges between 8–12 months (1-year-old), and at 18–24 months, they are matured to be used as broodstock (Phelan and Grubert, 2007; Alberts-Hubatsch et al., 2016). However, grouper fish would require about 6-year to attain the age of sexual maturity, about twice the time needed for crabs (FAO Fact Sheets[3]). Therefore, many shellfish species have great potential to become sources of future food based on their shorter life cycles.

Nutritional Value and Health Benefits

Shellfish are also among the largest sources of animal protein in the world (FAO, 2016). Moreover, most shellfish contain appreciable quantities of digestible proteins, essential amino acids, bioactive peptides, long-chain polyunsaturated fatty acids, astaxanthin, and other carotenoids, vitamin B12, and minerals (including copper, zinc, inorganic phosphate, sodium, potassium, selenium, iodine) (Venugopal and Gopakumar, 2017). The nutritional components of shellfish and their beneficial health effects have been comprehensively reviewed by Venugopal and Gopakumar (2017). The crude protein contents of green crab *Carpocoris mediterraneus* ranges from 13 to 18.2% depending on the body part enumerated (Cherif et al., 2008). The brown shrimp *Crangon crangon* has high contents of EAAs and non-EAAs (Turan et al., 2011). While the edible portions of Asian hard clam *Meretrix lusoria* contain about 188 mg of EAAs per gram dominated by leucine and lysine (Karnjanapratum et al., 2013). Some shellfish such as marine mussels have also been demonstrated to be promising sources of bioactive compounds that can be exploited for other uses in different industries (Grienke et al., 2014).

Demand, Cost, and Affordability

Consumer demand for shellfish and other seafood is one of the critical drivers of the current expansion in aquaculture activities, with a total production of 73.8 MMT in 2014 and estimated value of US$ 160 billion (FAO, 2016). The increase in demand is partly due to many factors, among which nutritional value and health benefits top. With the current trend of human population and health needs, it can be well-hypothesized that the demands for aquaculture products such as shellfish will only continue to increase in the next few years. However, it is noteworthy that despite the current increasing demands for shellfish from different consumer groups, the full potential of this sector of the aquaculture industry has not yet been fully understood. Moreso, the demand-driven planned production of these aquaculture species should meet the consumers' and farmers' needs at the best and most affordable prices. Although it is expected that the prices of shellfish will increase along with growing demands, efforts must be put in place to make them affordable; or else, households within the low-income class may face economic difficulties in obtaining these products. Therefore, removing the barriers to accessing these products is essential to realize them as future food. It should be highlighted that food security is not only dependent on the availability of produces in adequate quantities but also on their affordability by the large section of the populace (Teneva et al., 2018).

[3] Available information at: http://www.fao.org/fishery/culturedspecies/search/en

ISSUES TO BE ADDRESSED

This section briefly discusses the issues associated with seafood aquaculture-based products designated as "Future Food candidate." As earlier reiterated, the indicators that qualify shellfish as potential future food candidates are related to their production and consumption characteristics. Based on those indicators, there are still a few issues and problems related to shellfish aquaculture that need to be resolved. For example, the cannibalism behavior of most shellfish is a critical source of loss and an aspect that needs more fundamental research to mitigate. Manipulation of husbandry requirements and other abiotic/biotic factors to reduce cannibalism during different life stages is one of those research domains. Cutting down the production cost to increase the affordability of the cultured shellfish is another critical issue to tackle and will largely depend on reducing feeding costs. Therefore, research into various unconventional feeding practices and nutrient optimization is essential in this regard. Genetic manipulation of shellfish for the production of fast-growing strain progenies can also help reduce the life cycle and increase the aquaculture ventures' productivity. In this regard, the perfection of the all-male progeny production through biotechnology techniques is much needed. The application of other green farming techniques in the production of shellfish is also an area of future research that must be exploited in the quest of realizing the future food potential of the sector.

CONCLUDING REMARKS AND PROSPECTS

The present paper has provided a perspective on why many shellfish could be considered potential future food candidates. Since future food by definition has to be produced sustainably, efficiently, sufficiently with less cost, and using minimum natural resources without negative impacts on the environment, necessary measures should be taken by all the stakeholders involved (ranging from academicians and researchers to government officials and supply chain players) to ensure these elements will be effectively implemented by the shellfish industry.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

MA produced the first draft of the manuscript. VO produced the revised version of the document. MT, MH, and MI contributed to the conceptualization and design of the study. All authors read and approved the final version of the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.654897/full#supplementary-material

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Seagrass Structural Traits Drive Fish Assemblages in Small-Scale Fisheries

Benjamin L. Jones[1,2], Lina M. Nordlund[3], Richard K. F. Unsworth[2,4], Narriman S. Jiddawi[5] and Johan S. Eklöf[1]*

[1] Department of Ecology, Environment and Plant Sciences, Stockholm University, Stockholm, Sweden, [2] Project Seagrass, Bridgend, United Kingdom, [3] Department of Earth Sciences, Natural Resources and Sustainable Development, Uppsala University, Uppsala, Sweden, [4] Department of Biosciences, Swansea University, Swansea, United Kingdom, [5] Institute of Fisheries Research, Ministry of Agriculture, Natural Resources, Livestock and Fisheries, Zanzibar, Tanzania

Seagrasses – a group of foundation species in coastal ecosystems – provide key habitat for diverse and abundant faunal assemblages and support numerous ecosystem functions and services. However, whether the habitat role of seagrasses is influenced by seagrass diversity, by dominant species or both, remains unclear. To that end, we sought to investigate the specific seagrass characteristics (e.g., species diversity, seagrass traits) that influence tropical fish assemblages, and place this in the context of small-scale fishery use. We surveyed seagrass variables at 55 plots, nested within 12 sites around Zanzibar (Tanzania) in the Western Indian Ocean, and used Baited Remote Underwater Video (BRUV) systems to assess fish assemblages across plots. Using linear mixed models, we reveal that seagrass structural complexity and depth were the best predictors of fish abundance, with higher abundance occurring in deeper meadows or meadows with high canopy, leaf length and number of leaves per shoot. Moreover, an interaction between seagrass cover and land-use was the best predictor of fish species richness, where sites closer to human impacts were less affected by cover than sites with lower human impact. Overall, models with seagrass species richness or functional diversity as predictors poorly explained fish assemblages. Fish taxa that were important for small-scale fishery sectors (e.g., emperors, snappers, rabbitfish, and parrotfish) were primarily driven by seagrass structural complexity. Our results provide a unique analysis of the relationship between seagrass habitat and its associated fish assemblages in that we show that seagrass species diversity had little effect on seagrass fish assemblages, which instead appear driven by specific seagrass traits and seagrass cover. If conserving high value species that support adjacent fisheries is the priority for protecting seagrass meadows, then seagrass areas should be chosen with high cover and structural complexity that are in deeper waters. Any conservation measures also need to balance the needs of fishers that use the resources supported by seagrasses.

Keywords: seagrass meadows, fish assemblages, species diversity, small-scale fisheries, habitat structure, functional ecology

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****Correspondence:***
Benjamin L. Jones
Benjamin.Jones@su.se

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INTRODUCTION

Foundation species like trees, corals and seagrasses play a pivotal role in driving ecosystem functions and services globally (Angelini et al., 2011), notably by facilitating the creation of habitats. Therefore, loss of biodiversity, particularly habitat biodiversity, as well as the homogenization of ecosystems is a global threat (Hoag, 2010; Oliver, 2016). Two not mutually exclusive ecological hypotheses are proposed to influence the effects of biodiversity on ecosystem function. First, the 'mass ratio' hypothesis proposes that ecosystem functions, like complex habitats favoring high biodiversity, are primarily determined by the functional traits of *dominant species* within the community (Grime, 1998). In contrast, the 'complementarity hypothesis' proposes that the taxonomic and/or functional diversity within the community are instead the key drivers of ecosystem functions (Tilman et al., 1997). In order to manage ecosystems for the services they provide, it is vital to understand how biodiversity drives ecosystem functions (e.g., whether it is species composition or diversity, or both), especially in the context of global change (Benkwitt et al., 2020). While the topic has become a heated debate, especially with regards to terrestrial ecosystems (Picasso, 2018), meta-analyses of hundreds of experiments in terrestrial (Cardinale et al., 2011) and marine ecosystems (Gamfeldt et al., 2015) suggest that both the species composition and diversity can jointly influence ecosystem functions.

In the marine environment, structure-forming foundation species can strongly influence fish assemblage organization (Gorham and Alevizon, 1989; Caley and St John, 1996; Beukers and Jones, 1998; Harding and Mann, 2001). Indeed, many small-scale fishers have their own opinions on where the best place to fish is, be it a complex structure or a specific area defined by certain species traits, which might also shift over time (Katikiro, 2014). Yet in general, we value the role of marine foundation species either in isolation (e.g., kelp) or by grouping species into much broader functional entities (e.g., coral reefs). As a result, we see a focus on protecting flagship and iconic foundation species or habitats, regardless of their state, qualities or characteristics (Caveen et al., 2014). However, this overlooks the fact that most ecosystems are structured by co-occurring foundation species, which can range greatly in their structural and functional attributes (Bruno and Bertness, 2001). One example is seagrasses, a group of habitat-forming marine plants occurring along the world's coastlines (McKenzie et al., 2020). Pooling seagrass assemblages into bioregions reveal the highest species richness in the tropical Indo-Pacific (Short et al., 2007), consistent with the latitudinal diversity gradient (Willig et al., 2003; Jablonski et al., 2006). Up to 14 seagrass species co-occur with each other in the region (Miguel et al., 2018), yet many of these have very different functional attributes (e.g., low-high root biomass, short-long leaf length, low-high nutrient content). With reported seagrass loss across the region being high (Waycott et al., 2009), and some species being more at risk than others (Short et al., 2011), we need to urgently unpack this broad functional entity of seagrass meadows to understand how specific seagrass species losses could influence ecosystem service provision, and target management accordingly.

Given the Indo-Pacific region is one of the most densely populated by humans (Williams, 2013), sustaining the important ecosystem services that biodiversity provides is vital. Human dependence on seafood is substantial (Donner and Potere, 2007), but the foundation habitats that contribute to this provision are being degraded (Burke et al., 2011; Coles et al., 2011; Giri et al., 2011). In response, much research and conservation funding in the region has been directed toward coral reefs (Unsworth et al., 2019a), partly due to a failure to recognize that coastal fishers utilize multiple habitats, like seagrass meadows (Nordlund et al., 2018b), and partly because coral reefs are much more well-known and researched than other coastal habitats (UN Environment et al., 2018). Yet we now know the vital role seagrass plays globally in supporting fisheries and food supply (Unsworth et al., 2019b), which in the Western Indian Ocean can provide economic gains that locally may be four times greater than coral reefs (de la Torre-Castro et al., 2014). Previous research shows that tropical fish assemblages are influenced by variations in the structure of seagrass habitats (Heck and Orth, 1980), by seagrass canopy complexity (Bell and Westoby, 1986a,b; Nakamura and Sano, 2004) and by seagrass landscapes (Salita et al., 2003). Yet, the extent to which this is driven by seagrass diversity and/or by traits of dominant species remains unclear. The UN Decade of Ocean Science for Sustainable Development mandates for solution orientated science and management that balances biodiversity conservation with the needs of local people (von Schuckmann et al., 2020). The aspects of seagrass diversity that are key to sustaining biodiversity, and thus ecosystem services, remain unclear in general and for the (relatively speaking) 'hyper-diverse' seagrass meadows within the Indo-Pacific Ocean in particular (Nordlund et al., 2018a). This is especially pressing when we consider the multiple threats facing seagrass (Unsworth et al., 2018).

The aim of this study was to assess to what extent the fish habitat function of seagrass meadows is explained by the diversity and/or composition of seagrasses, from both a taxonomic and traits perspective. We surveyed seagrass meadows across a gradient of fishing pressure and anthropogenic impact around Zanzibar, Tanzania. We sought to determine (i) which seagrass meadow attributes best influenced seagrass meadow fish assemblages, and (ii) investigate the relationships between seagrass meadow attributes and the presence of fish important to small-scale fisheries. In the context of documented seagrass loss across the Indo-Pacific (Harcourt et al., 2018; Unsworth et al., 2018; Tin et al., 2020), we seek to indicate which attributes are key for managing seagrass for the maintenance of important ecosystem functions.

METHODS

Study Design

We used standard Baited Remote Underwater Video (Cappo et al., 2007) and benthic quadrat assessment methods (McKenzie et al., 2001) to survey seagrass flora and motile fauna at 55 distinct plots nested within 12 sites around Zanzibar (Unguja Island), Tanzania, between October and November 2018 (**Figure 1**).

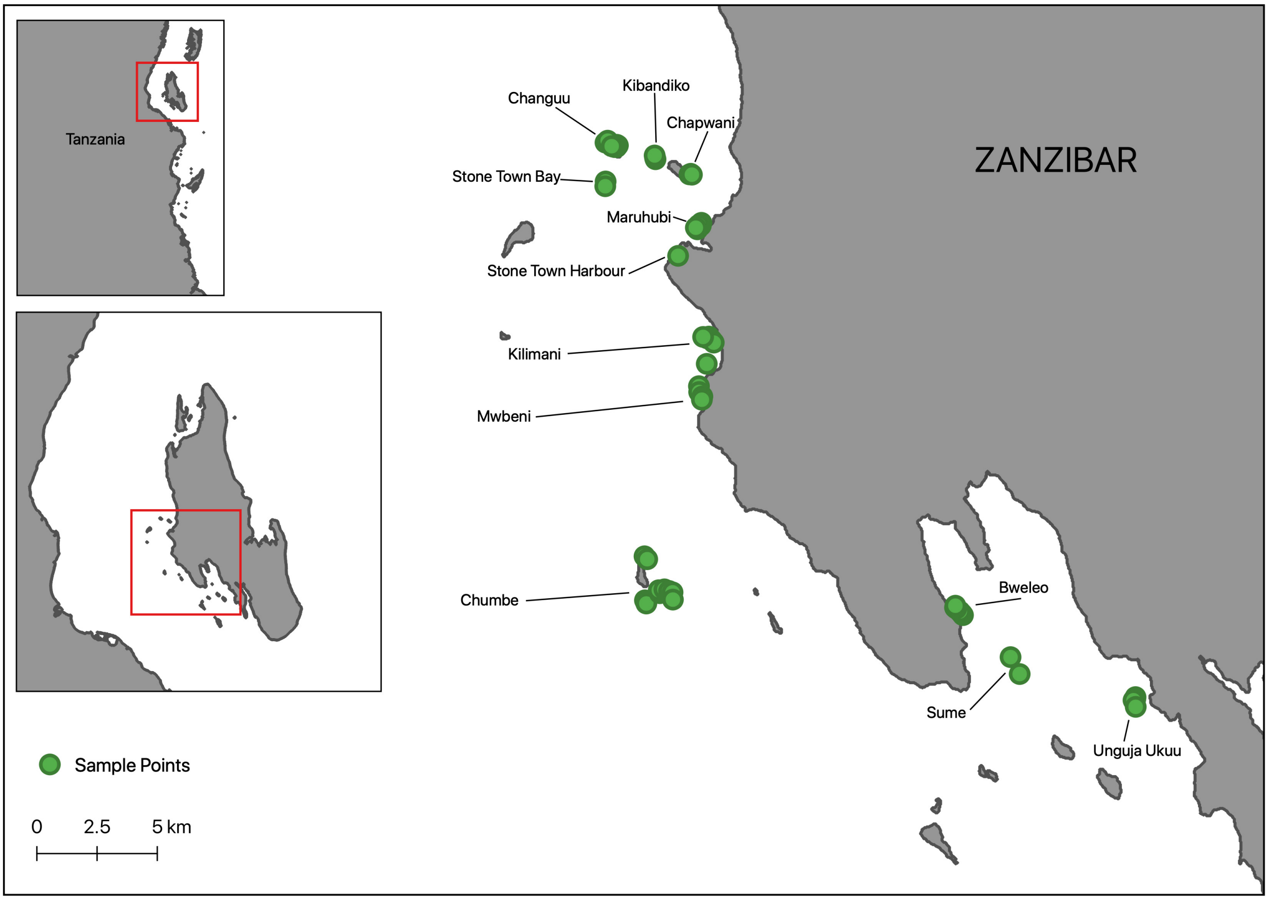

FIGURE 1 | Map of Zanzibar, Tanzania showing the location of 55 plots, nested within 12 distinct sites (identifiable by names), where fish assemblage and seagrass surveys were conducted during October-November 2018.

Seagrass meadows around Zanzibar provide a suitable setting to examine the influence of seagrass species composition, diversity and structure on faunal productivity in that they are complex and comprised of a number of species in varying densities (Gullström et al., 2002). These are characteristically comprised of mixed, or monospecific *Halodule uninervis*, *Halophila ovalis*, and *Cymodocea rotundata* areas toward the upper intertidal limits of the meadow, shifting to both mixed and monospecific *H. uninervis*, *C. rotundata*, *Thalassia hemprichii* and *H. ovalis* areas in the lower intertidal limits of the meadow. The upper subtidal areas are comprised of *C. serrulata*, *T. hemprichii*, *H. ovalis* and *Syringodium isoetifolium*, shifting to *C. serrulata*, *T. hemprichii*, *H. ovalis*, *S. isoetifolium* and *Thalassodendron ciliatum* before being often dominated by *T. ciliatum* and *Enhalus acoroides*, growing in monospecific or mixed strands, in deeper areas. While one site was adjacent to a no-take marine reserve (but not within), all sites were typically fished. Plots were selected non-randomly to encapsulate variability in both seagrass species composition and species richness; we wanted to include mixed and monospecific plots for multiple seagrass genera. However, not all sites had the same seagrass species which ultimately led to the unbalanced nature of number of plots within each site.

Seagrass Community Structure

At each of the 55 plots, two 50 m transects were placed in a cross, centered around the location of the fish sampling point (**Figure 2**). Within twenty-two 0.25 m^2 quadrats, placed at 5 m intervals along transects (0-50 m), we collected data on various seagrass meadow attributes pertinent to our hypothesis (structure, cover, diversity and composition; **Table 1**). We estimated seagrass shoot density (0.0225 m^2), total percentage cover (0-100%) and floral species composition (McKenzie et al., 2001). Canopy height (cm), leaf length (cm), leaf width (cm) and number of leaves per shoot were recorded from three random shoots within in each quadrat, and the mean was later used in the statistical analyses (McKenzie et al., 2001). Seagrass species were sorted into two structural trait categories - leaf growth form (strap, branched, paddle, and cylindrical) and canopy type (low, middle and high) - to calculate community weighted functional trait indices (functional dispersion) and functional richness (see *data analysis*). Prior to data analysis, the seagrass trait characteristics (n = 1210) were averaged for each plot (n = 55) to match the scale of the fish sampling (see site averages in **Supplementary Table 1**).

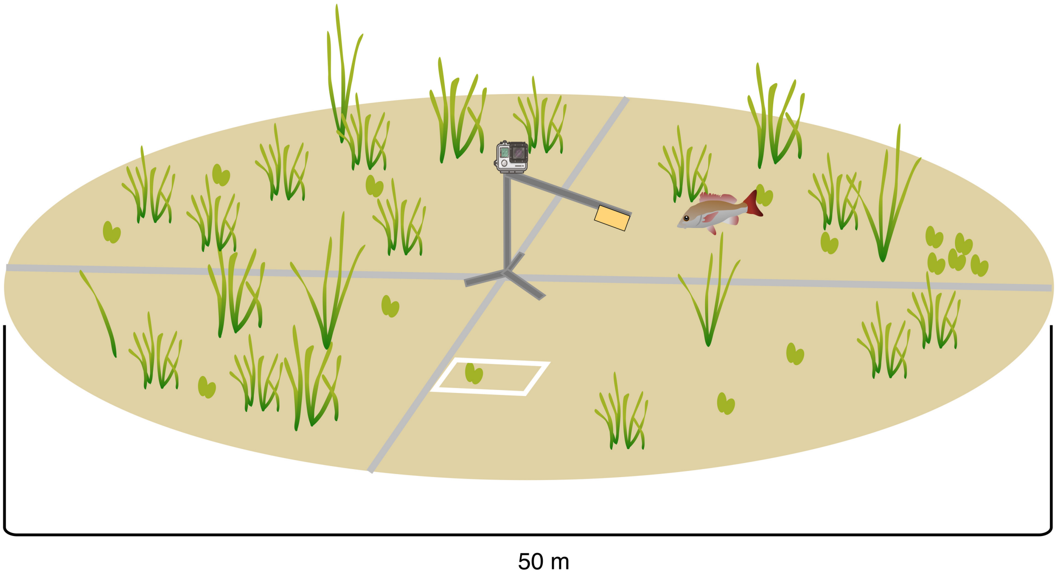

FIGURE 2 | Conceptual diagram of survey setup at each sample location, showing the location of the Baited Remote Underwater Video (BRUV) system toward the center of the sample plot and location of the two 50m transects (not to scale).

Environmental Characteristics and Anthropogenic Impact

For each plot, we then calculated several seascape variables that have been previously shown to structure fish assemblages (see site averages in **Supplementary Table 2**), including distance to nearest reef or area of coral bommies (Campbell et al., 2011) and nearest mangrove area (Dorenbosch et al., 2007). Individual mangrove trees were not considered and only areas with sufficient mangrove cover were included. Water depth, which ranged from 0.6 to 6.3 m, was recorded at time of fish sampling for each plot (Pogoreutz et al., 2012).

Human land-use intensity can strongly affect seagrass state (Quiros et al., 2017). Therefore, we utilized a similar method to that used by Quiros et al. (2017) to assess land-use characteristics at a site level. We estimated human land-use characteristics within a 2 km radius from each site (using Google Earth Pro v. 7.3) and calculated the proportion (km^2) of land utilized in this 2 km radius by six categories. These were bare ground (exposed soil, shrub), human development (houses, villages, industry, and roads), aquaculture, farmland, vegetation (forest, mangrove) and water (ocean, river). We also recorded distance to nearest catchment.

Fishing pressure is also a well-known driver of seagrass fish assemblages, including those around Zanzibar (Alonso Aller et al., 2014). Even though the influence of fishing was not a main factor in our analysis, we sought to account for its

TABLE 1 | Seagrass meadow structure, cover, composition, richness and community-weighted trait values investigated as predictors of fish abundance and species richness in the present study.

Category	Variables	Description	Hypothesis
Meadow structure	Shoot density	Total number of seagrass shoots per 0.25 m^2	Mass ratio
	Leaves per shoot	Average number of leaves of three random shoots	
	Canopy height	Average canopy height for three random shoots	
	Leaf length	Average length of leaves for three random shoots	
	Leaf width	Average width of leaves for three random shoots	
Seagrass cover and composition	Seagrass cover	Total% cover of seagrass per 0.25 m^2	Mass ratio
	Seagrass composition	Abundance of each seagrass species present (% cover) as a proportion of total seagrass cover	
Richness	Seagrass richness	Number of seagrass genera	Diversity
	Functional richness	Number of functional groups based on two seagrass traits (leaf type and canopy type)	
Functional dispersion	Functional dispersion	Trait dispersion based on two seagrass traits (leaf type and canopy type), weighted by seagrass composition	Diversity

Descriptions and hypothesis are provided for each seagrass category included.

influence as a covariate. First, distance to fishery landing site was calculated for each plot based on landing sites reported by the Department of Fisheries Development (2016). Second, fishing effort was approximated using a semi-quantitative two-level scale (low, high). This was developed based on discussions with local fishers and resource users (McCluskey and Lewison, 2008) and further quantified using field observations of ongoing fishery activity during Oct-Dec 2018 (**Table 2**). With observations, we considered both the number of fishers and the fishing gear used. For example, observations of multiple spear and trap fishermen operating from non-motorized boats were considered to be lower intensity than a single > 10 person drag net team operating from a larger motorized boat since drag nets catch more fish (Jiddawi and Öhman, 2002) and are less selective (McClanahan and Mangi, 2004).

Surveys of Seagrass Fish Assemblages

The abundance and diversity of seagrass fish assemblages were evaluated at each of the 55 plots using a single Baited Remote Underwater Video system (BRUVs) (Jones et al., 2018). BRUVs are a widely used and established method for understanding motile fauna in marine environments. Examples include their use in understanding the effects of marine reserves and protected areas (e.g., Whitmarsh et al., 2014; Bornt et al., 2015; Coleman et al., 2015; Gilby et al., 2017), assessing the faunal communities of reef systems (e.g., Lindfield et al., 2016), quantifying shark populations (e.g., MacNeil et al., 2020) and investigating demersal fish populations in different habitat types (e.g., Furness and Unsworth, 2020). Such assessments focus on fish diversity as a response and not only focus on top predators, but numerous demersal fish species. Moreover, the use of different types of video methods is becoming increasingly common for investigations into seagrass habitat and fish interactions (e.g., Smith et al., 2011; Whitmarsh et al., 2014; Díaz-Gil et al., 2017; Henderson et al., 2017; Kiggins et al., 2018). We chose to use a BRUV system for several reasons: they have been shown to be more effective for describing seagrass fish assemblages given the "diver effect" caused by traditional methods (Edgar et al., 2004; Zarco-Perello and Enríquez, 2019) have greater statistical power at low sample numbers and also attract herbivores (e.g., Harvey et al., 2007; Langlois et al., 2010; Watson et al., 2010; Andradi-Brown et al., 2016; Schramm et al., 2020). While this choice may result in fewer cryptic species being recorded (Watson et al., 2005), we considered traditional methods inappropriate for use in such intensely fished areas (Lindfield et al., 2014). The ability to ensure a consistent methodology between observations (Jones et al., 2021) was considered of greater importance than greater species resolution afforded by other methods.

TABLE 2 | Observations of ongoing fishing activity between October and December 2018 at 12 sites in Zanzibar, Tanzania.

Site	Field Observations	Fishing activity
Bweleo	Fishers utilizing basket traps	Low
Changuu	Drag net teams in operation within seagrass areas	High
Chapwani	Drag net teams in operation within seagrass areas	High
Chumbe	Drag net teams in operation within seagrass areas	High
Kibandiko	Drag net teams in operation within seagrass areas	High
Kilimani	No fishing activity observed	Low
Maruhubi	Fishers using fishing rod, high boat traffic	Low
Mwbeni	Fishers utilizing basket traps and spear	Low
Stone Town Bay	No fishing activity observed	Low
Stone Town Harbor	No fishing activity observed, high boat traffic	Low
Sume	Drag net teams in operation within seagrass	High
Unguja Ukuu	Drag net teams in operation within seagrass, fishers utilizing basket traps	High

A semi-qualitative two-point scale was used to classify fishing activity based off observations in leu of publicly available reported data.

The mono-BRUV system was constructed based on designs by Cappo et al. (2004). An aluminium tripod (Smatree X1S; 50 cm high) was used as a mount for a GoPro Hero 5 camera, with 2 kg lead weight placed at the base for stability underwater. A bait arm (20 mm Ø PVC) extending 1 m from the camera supported a plastic bait container (Trappy Betesbox; 112 cm^3), which was filled with standardized oily bait (*Sardinella* sp. purchased locally) before all deployments. Oily fish were used as they are considered the most effective bait for use in BRUVs (Bond et al., 2012; Dorman et al., 2012).

BRUVs were deployed for 35 min, with the first 5 min considered a buffer time to allow fauna to respond to disturbance. The remaining 30 min were used for analysis which is considered adequate time to assess fish assemblages (Wraith et al., 2013). All BRUVs were deployed from a boat during daylight hours on an incoming tide, around one hour after low tide in order to reduce variability in bait plume area caused by current velocity (e.g., Taylor et al., 2013; Piggott et al., 2020). All deployments were conducted within 5 days of a low spring tide and therefore the majority of plots were placed in seagrass areas that were almost always subtidal.

Each of the 55 plot videos was analyzed to determine the MaxN of each fish species and fish species richness; a metric commonly used for the quantification of the relative abundance of fish observed in underwater video (Cappo et al., 2004; Unsworth et al., 2014). MaxN is equal to the maximum number of fish recorded at any one time (single video frame) and therefore removes concerns associated with double counting of individual fish (Priede et al., 1994). MaxN was determined for each species in every video frame throughout the 30 min of footage. The highest MaxN for each species at the end of each 30 min was used in further analysis.

Fish were identified to species level where possible using FishBase (Froese and Pauly, 2015) and numerous identification guides (Richmond, 2011; Smith and Heemstra, 2012; Taquet and Alain, 2012). All fish species identified were then categorized into four categories based on perceived value to small-scale fisheries sectors [based on Thyresson et al. (2013)]; (i) Low Value [species not mentioned by Thyresson et al. (2013)], (ii) Local Household, (iii) Small-Scale Trader and (iv) Town Market (**Table 3**).

Data Analysis

All statistical analyses were conducted using R version 3.6.3 (R Development Core Team, 2020). We first calculated community weighted functional trait indices (*functional dispersion*) and *functional richness* using the *FD* package in R (Laliberté and Legendre, 2010). Using a trait matrix based on leaf growth form and canopy type (see *seagrass community structure*), we calculated functional dispersion by using species trait scores and species abundance (based on species composition). Functional dispersion, a measure of functional diversity, is defined as the mean distance in multidimensional trait space of individual species to the centroid of all species, weighted by abundance (Laliberté and Legendre, 2010); in other words how morphologically different each seagrass species is compared to the average seagrass morphology in the local community. The same approach was used for functional richness, but negated species abundance and instead used species presence to account for richness. Seagrass community variables were grouped into four distinct categories pertinent to the hypotheses: *meadow structure* (shoot density, canopy height, leaf length, leaf width, and number of leaves per shoot), cover and composition (*seagrass cover, seagrass composition),* richness (*species richness*, *functional richness*) and *functional dispersion* (**Table 1**). Given that multiple variables were included within the category *meadow structure,* a Principle Component Analysis (PCA) was performed using the *prcomp()* function in R and the values for PC1 decomposed and exported to create a single variable accounting for the majority of variance (James and McCulloch, 1990; Harrison et al., 2018). We accounted for dispersion in meadow structure (e.g., whether meadows are similar in their structural properties) by computing a PCA on all structural quadrat variables (n = 1210) and calculated the coefficient of variation (mean/standard deviation) for the PC1 values for each of the 55 plots. We called this variable *meadow structure variability*. We also used the PCA approach for *seagrass composition* (% composition of each species present as a proportion of total cover) and for *land-use* (% area of bare ground, human development, aquaculture, farmland, vegetation, ocean within a 2 km radius and distance to catchment).

In the PCA of *meadow structure*, PC1 accounted for 58.02% of the variance. All variables had substantial factor loadings on PC1 which was negatively correlated with canopy height, leaf length, leaf width and number of leaves per shoot and positively correlated with shoot density (**Supplementary Figure 1**). We inverted this axis so that high PC1 values represent high seagrass structural complexity (e.g., higher canopy, longer

TABLE 3 | Fish families recorded in this study and their frequency across samples.

		Small-scale fishery sector			
Fish Family	***Frequency (%)***	***Limited value (LV)***	***Local household (LH)***	***Small-scale trader (SST)***	***Town market (TM)***
Apogonidae	4	✓			
Atherinidae	4	✓			
Aulostomidae	2	✓			
Belonidae	2	✓			
Carangidae	4			✓	✓
Chaetodontidae	2	✓			
Ephippidae	2	✓			
Fistulariidae	2	✓			
Gerridae	13	✓			
Gobiidae	2	✓			
Haemulidae	5		✓	✓	
Labridae	51	✓			
Lethrinidae	75			✓	✓
Lutjanidae	2		✓	✓	✓
Monacanthidae	2	✓			
Mullidae	35			✓	✓
Muraenidae	20			✓	
Nemipteridae	5	✓			
Ophichthidae	2	✓			
Ostraciidae	2	✓			
Pomacentridae	15	✓			
Scaridae	55			✓	
Siganidae	53		✓	✓	
Sphyraenidae	9				✓
Teraponidae	4	✓			
Tetraodontidae	11	✓			

Families are also grouped by their value to small-scale fishery agent categories based on data presented in Thyresson et al. (2013). Limited value represents fish species not mentioned by fishermen in Thyresson et al. (2013). Families were only included in categories if they were reportedly sold to that agent category by over 50% of fishermen.

and wider leaves, greater number of leaves per shoot) and low values represent low seagrass structural complexity. For *seagrass composition*, PC1 accounted for 53.35% of variation and was positively associated with higher abundance of canopy-forming species (e.g., *T. ciliatum*) and negatively associated with shorter, meadow-forming species (e.g., *C. serrulata*, *T. hemprichi*; **Supplementary Figure 2**). In the PCA for *land-use*, PC1 was responsible for 58.14% of variance and negatively associated with area of human development and farmland, and positively associated with distance from catchment and area of ocean (**Supplementary Figure 3**). We also inverted this axis so that low values represent low impact sites, and higher values represent high impact sites.

We then used mixed-effects linear models with maximum likelihood estimation to evaluate the influence of our seagrass community variables on seagrass fish assemblages. We determined the maximum complexity of models by using the square root of N ($\sqrt{55} = 7.4$), and the most complex model included 6 parameters. We specifically examined whether response and predictor variables were normally distributed and whether there was multicollinearity between any of the predictors by calculating the variance inflation factor (VIF). We used log (MaxN, *seagrass species richness*, *community composition*, *functional richness* and *functional dispersion, depth, distance to coral reef*) and cube root transformations (*meadow structure variability*) to transform predictors and response variables, and used Pearson's correlation tests to test for multicollinearity ($r > 0.7$) (Dormann et al., 2013). Based on these tests we excluded *distance to mangrove*, *distance to coral reef*, *distance to landing site* and *fishing pressure* from the final variable set, as all were strongly correlated with *land-use* (**Supplementary Figure 4**). Our final variable set consisted of 9 predictors (seven seagrass community predictors, two environmental predictors), all with VIF values < 3 (Zuur et al., 2010).

Using linear mixed-effects models (Zuur et al., 2009), we explored the relative importance of seven seagrass variables (*meadow structure*, *meadow structure variability*, *seagrass cover*, *seagrass species richness, community composition*, *functional richness* and *functional dispersion*), *depth* and *land-use* on two response variables: fish abundance and fish species richness. We also computed models to evaluate the effects of seagrass variables on the abundance of fish within the four value categories. Linear mixed-effects models were fitted with the *lmer()* function in the *lme4* package for R (Bates et al., 2015). Each of the seagrass predictors was tested independently with an interaction with land-use and depth as fixed predictors, given that land-use influences seagrass condition (Quiros et al., 2017) and depth influences fish abundance (Alonso Aller et al., 2014). Given that the 55 plots were nested within 12 sites, all models included a random effect of site to account for biogeographic differences in environmental conditions and potential differences in fishing pressure (given we excluded it). Based on Akaike's Information Criterion (AIC) scores (Akaike, 1974), we then pruned each model without removing the seagrass variable itself. We also included a null model that included land-use, depth and the random factor of site to compare against seagrass predictors. We report the fit of models using R^2_{GLMM} which was calculated using the *rsquared()* function from the *piecewiseSEM* package for R (Lefcheck, 2016).

RESULTS

General Description of Fish Assemblages

A total of 1,676 individual fish, representing 65 species and 26 families, were recorded. Many of the species were rare (<10 observed individuals) and five fish families (Siganidae, Lethrinidae, Lutjanidae, Scaridae, and Labridae) were responsible for over 85% of the total abundance pooled across sites. In addition, fish families important for human consumption in the region were the most common, with emperors (Lethrinidae) occurring in 75% of samples, followed by parrotfish (Scaridae, 55%), rabbitfish (Siganidae, 53%), wrasses (Labridae, 51%), goatfish (Mullidae, 35%) and moray eels (Muraenidae, 20%). For six of the samples, no fish species were recorded and only two sites recorded greater than 100 individuals.

Seagrass Meadow Variables Driving Fish Abundance and Diversity

We compared seven models testing the influence of seagrass meadow variables on fish abundance (MaxN), along with a null model (*land-use* + *depth*). The best-fitting model (lowest AIC) included the additive effects of *meadow structure* and *depth*, with an AICc weight of 0.99 ($R^2 = 0.73$; **Table 4**). *Meadow structure* had a positive effect on fish abundance; meadows with taller canopies, longer, broader and more numerous leaves and lower shoot density had higher fish abundances (**Figure 3**). All other models had a Δ AIC > 10, and only *seagrass cover*, *functional dispersion* and *meadow structure variability* were better predictors of fish abundance than the null model (**Table 4**).

In terms of fish richness, the best model included an interaction between *seagrass cover* and *land-use*, as well as the effect of *depth*, and had an AICc weight of 0.36 ($R^2 = 0.40$; **Table 4**). *Seagrass cover* positively influenced fish richness at sites with low levels of human development. This influence declined with increasing human development, with no effect at the highest levels of development (e.g., sites close to large human populations; **Figure 4**). However, while seagrass cover had little effect at sites with the highest levels of human development, it was at these sites where we recorded high fish species richness – regardless of whether seagrass cover was low or high. Meanwhile, fishing pressure was negatively associated with human development (see **Supplementary Material Appendix 1** and **Figure 1**). Consequently, sites with higher fish richness were also sites with lower fishing pressure. Finally, *depth* influenced fish assemblages, with generally greater richness observed in deeper seagrass areas.

Meadow structure was also a strong predictor of seagrass fish richness, with an AICc weight of 0.20 ($R^2 = 0.44$), with an additive effect of *depth* (Δ AIC < 2). As with fish abundance, *meadow structure* and *depth* had a positive effect on fish richness; generally deeper seagrass meadows with complex canopy structures (high canopy, long and more numerous leaves, but low shoot density) had greater fish richness (**Figure 5**). All other models had a Δ

TABLE 4 | Candidate models for fish abundance (MaxN) and richness, sorted by AIC corrected for small sample sizes.

Response	Predictors	K	AICc	Δ AICc	AICc Wt	R^2_{GLMM}
Fish Abundance (MaxN)	**Meadow structure + depth**	**5**	**156.8**	**0.0**	**0.99**	**0.73**
	Seagrass cover * land-use + depth	7	166.8	10.0	0.01	0.49
	Functional dispersion + land-use + depth	6	168.1	11.3	0.00	0.57
	Meadow structure variability + depth	5	170.4	13.6	0.00	0.53
	Land-use + depth *(null model)*	5	170.5	13.7	0.00	0.50
	Seagrass composition + depth	5	171.9	15.1	0.00	0.53
	Functional richness + depth	5	172.0	15.2	0.00	0.53
	Seagrass richness + depth	5	172.2	15.4	0.00	0.52
Fish Richness	**Seagrass cover * land use + depth**	**7**	**289.8**	**0.0**	**0.36**	**0.40**
	Meadow structure + depth	**5**	**290.9**	**1.2**	**0.20**	**0.44**
	Meadow structure variability * land-use + depth	6	291.8	2.1	0.14	0.37
	Land-use + depth *(null model)*	5	291.9	2.2	0.12	0.35
	Seagrass composition + land-use + depth	6	293.6	3.8	0.05	0.34
	Functional dispersion + land-use + depth	6	294.0	4.2	0.04	0.37
	Functional richness + land-use + depth	6	294.1	4.3	0.04	0.36
	Seagrass richness + land-use + depth	6	294.3	4.5	0.04	0.35

All models included a random effect of site. Models in bold represent models with a Δ AICc < 2.

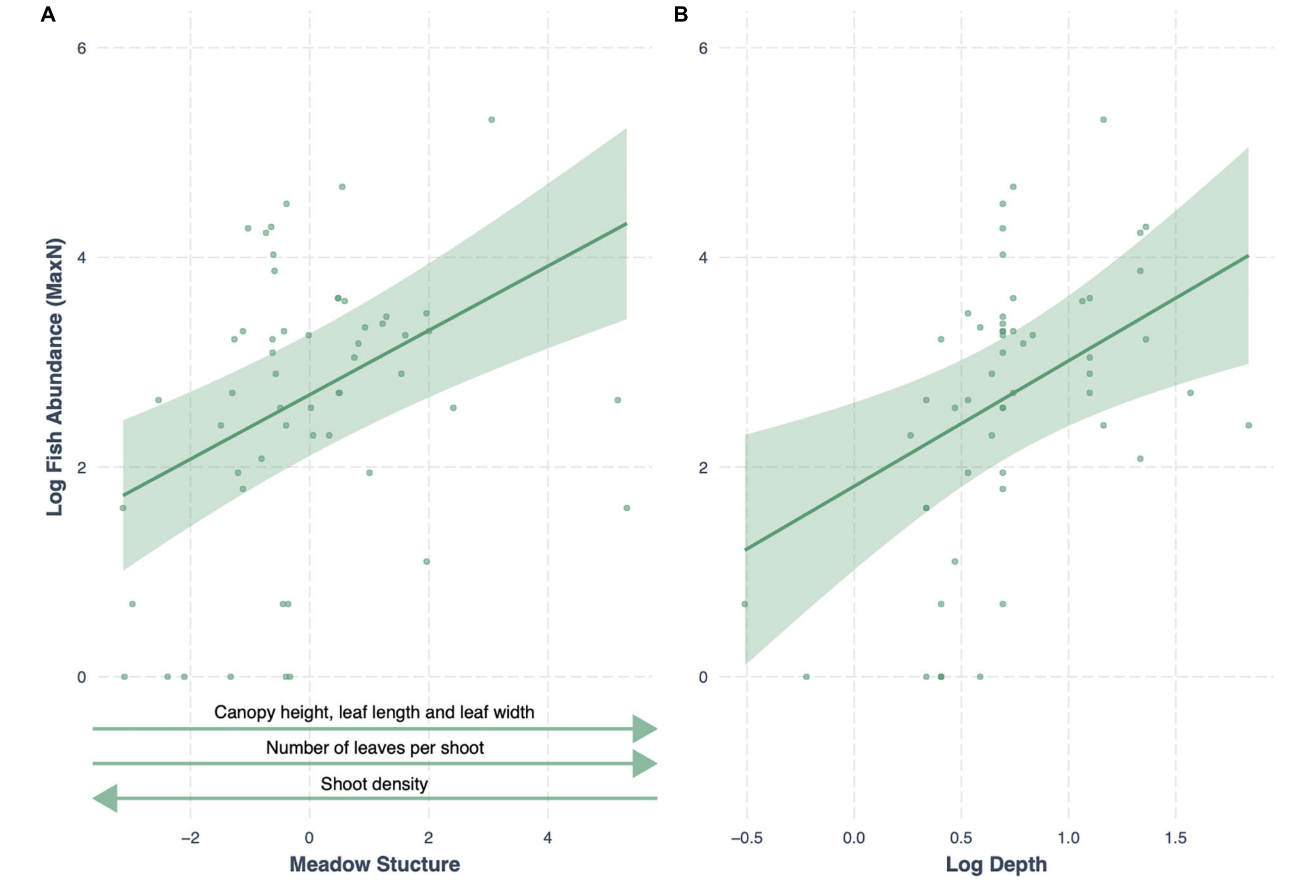

FIGURE 3 | Conditional effect plots for the model best predicting log fish abundance (R^2 = 0.73) in tropical seagrass meadows across 12 sites. Predictor variables include **(A)** meadow structure (inverse of PC1) and **(B)** log depth (m). Shaded areas represent 95% confidence intervals, reflecting the variance of the fixed effects.

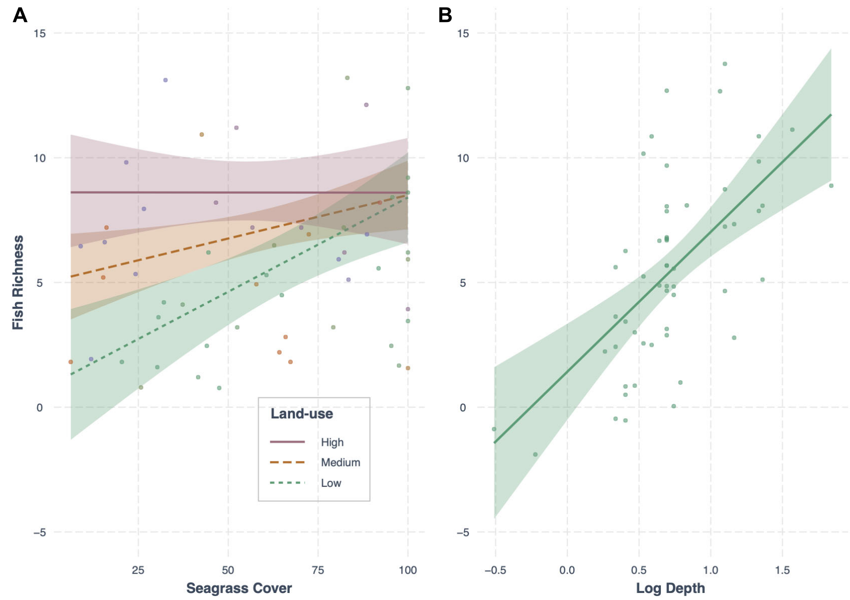

FIGURE 4 | Conditional effect plots for the model best predicting fish richness (R^2 = 0.47) in tropical seagrass meadows across 12 sites. Predictor variables include **(A)** the interaction between land-use (low, medium, high) and seagrass cover (%), and **(B)** log depth (m). Shaded areas represent 95% confidence intervals, reflecting the variance of the fixed effects.

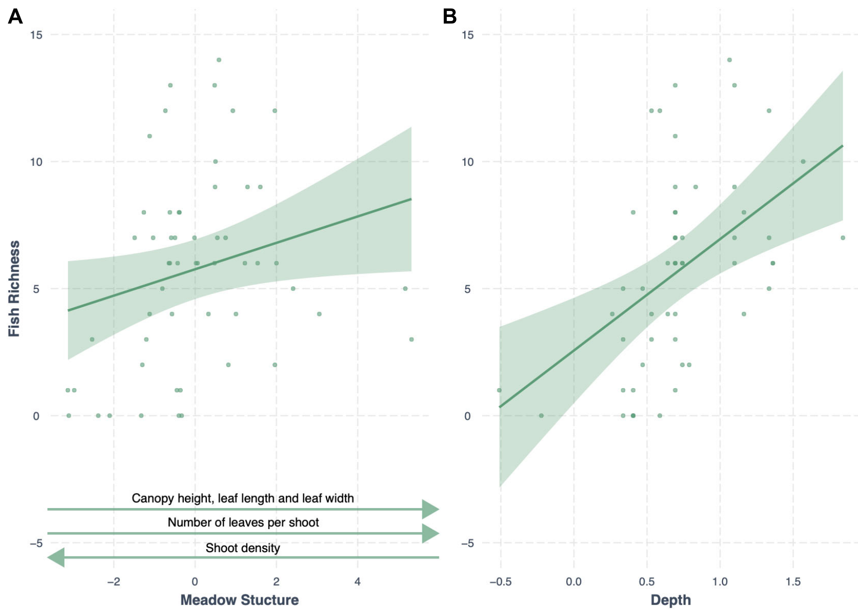

FIGURE 5 | Conditional effect plots for the second-best model predicting fish richness (R^2 = 0.42) in tropical seagrass meadows across 12 sites. Predictor variables include **(A)** meadow structure (inverse of PC1) and **(B)** log depth (m). Shaded areas represent 95% confidence intervals, reflecting the variance of the fixed effects.

AIC score > 2, and only *meadow structure variability* was a better predictor than the null model (**Table 3**).

For both response variables tested (fish abundance and richness), *seagrass richness*, *functional richness* and *seagrass composition* were poor predictors of fish richness, all with AICc weights of less than 0.05 for both response variables (**Table 4**) and performing worse than the null model.

Seagrass Meadow Variables Driving Abundance of Fish Important to Small-Scale Fishery Sectors

Three models with a cumulative AICc weight of over 0.83, were important for driving the abundance of fish important for sale at town markets (e.g., Carangidae, Sphyraenidae, and Mullidae;

Table 5). These were *seagrass meadow structure* (R^2 = 0.47), *functional dispersion* (R^2 = 0.44) and *seagrass cover* (R^2 = 0.34; **Table 5**). *Meadow structure* and *depth* had positive effects on abundance for this fish group (**Figure 6**), as did *seagrass cover*, which included the additive effects of land-use and depth (both of which were positive for abundance). However, *functional dispersion* had a negative effect on abundance, with more fish present when certain functional traits are present.

The abundance of fish important for sales to small-scale traders was primarily explained by seagrass *meadow structure* (**Figure 6**), with an AICc weight of 0.78 (R^2 = 0.63). *Meadow structure* and depth influenced fish abundance (**Table 5**), with deeper and more complex meadows (e.g., high canopy height, leaf length) characteristic of higher abundance of fish from this fishery sector. All other models had a ΔAIC of > 2.

The abundance of fish important to local household consumers was best predicted by two candidate models with a cumulative AICc weight of 0.70 (**Table 5**). *Functional dispersion* was the best predictor (R^2 = 0.47) followed by *seagrass structure* (R^2 = 0.21). In the top model, *functional dispersion* and *land-use* both had a negative effect on fish abundance, suggesting more fish are present when certain functional traits are in high abundance, situated away from areas with high human development. *Seagrass meadow structure* was positively associated to fish abundance.

Finally, the abundance of fish species with a limited or low value to small-scale fisheries sectors was best explained by two models, with a cumulative AICc weight of 0.69. These were *meadow structure variability* (R^2 = 0.25) and *seagrass cover* (R^2 = 0.30; **Table 5**). Both models included an effect of *depth*, and the *seagrass cover* model also included an interaction between *seagrass cover* and *land-use*. As with total fish richness, the models suggested that *seagrass cover* was important for driving fish abundance in areas of low human development but not in high. *Meadow structure variability* had a negative effect on the abundance of fish with limited or low value; sites with higher *more variable structural traits* had lower abundance of fish species.

DISCUSSION

The present study provides a unique analysis of the relationship between different features of seagrass habitats and their associated fish assemblages. We show that specific seagrass traits and seagrass cover drive seagrass fish assemblages, and

TABLE 5 | Candidate models for fish abundance across small-scale fishery sectors, sorted by AIC corrected for small sample sizes.

Response	Predictors	K	AICc	Δ AICc	AICc Wt	R^2_{GLMM}
Town Market	**Meadow structure + depth**	**5**	**166.0**	**0.0**	**0.30**	**0.47**
	Functional dispersion + depth	**6**	**166.1**	**0.0**	**0.30**	**0.44**
	Seagrass cover + land-use + depth	**6**	**166.6**	**0.6**	**0.23**	**0.34**
	Seagrass composition + depth	5	169.4	3.4	0.06	0.41
	Seagrass richness + land-use + depth	6	170.0	4.0	0.04	0.37
	Meadow structure variability + land-use + depth	6	170.4	4.3	0.03	0.35
	Functional richness + land-use + depth	6	170.5	4.4	0.03	0.35
Small-scale trader	**Meadow structure + depth**	**5**	**165.7**	**0.0**	**0.78**	**0.63**
	Functional dispersion + land-use + depth	6	169.1	3.4	0.14	0.56
	Seagrass cover + land-use + depth	6	171.8	6.1	0.04	0.43
	Meadow structure variability + depth	5	173.8	8.1	0.01	0.49
	Seagrass composition + depth	5	174.1	8.4	0.01	0.50
	Seagrass richness + depth	5	174.2	8.5	0.01	0.50
	Functional richness + depth	5	174.3	8.6	0.01	0.50
Local household	**Functional dispersion + land-use + depth**	**6**	**190.2**	**0.0**	**0.51**	**0.47**
	Seagrass cover + land-use + depth	6	192.2	2.0	0.19	0.21
	Meadow structure + land-use + depth	6	192.9	2.7	0.13	0.31
	Functional richness + land-use + depth	6	194.3	4.2	0.06	0.28
	Seagrass richness + land-use + depth	6	195.2	5.0	0.04	0.24
	Meadow structure variability + land-use + depth	6	195.7	5.5	0.03	0.24
	Seagrass composition + land-use + depth	6	195.7	5.5	0.03	0.24
Low value	**Meadow structure variability + depth**	**5**	**142.3**	**0.0**	**0.45**	**0.25**
	Seagrass cover * land-use + depth	**7**	**143.5**	**1.2**	**0.24**	**0.30**
	Meadow structure + depth	5	144.7	2.4	0.13	0.26
	Seagrass richness * land-use + depth	7	146.0	3.7	0.07	0.27
	Seagrass composition * land-use + depth	7	146.9	4.6	0.05	0.26
	Functional richness + depth	5	147.9	5.6	0.03	0.17
	Functional dispersion + depth	5	148.2	5.9	0.02	0.18

All models included a random effect of site. Models in bold represent models with a Δ AICc < 2.

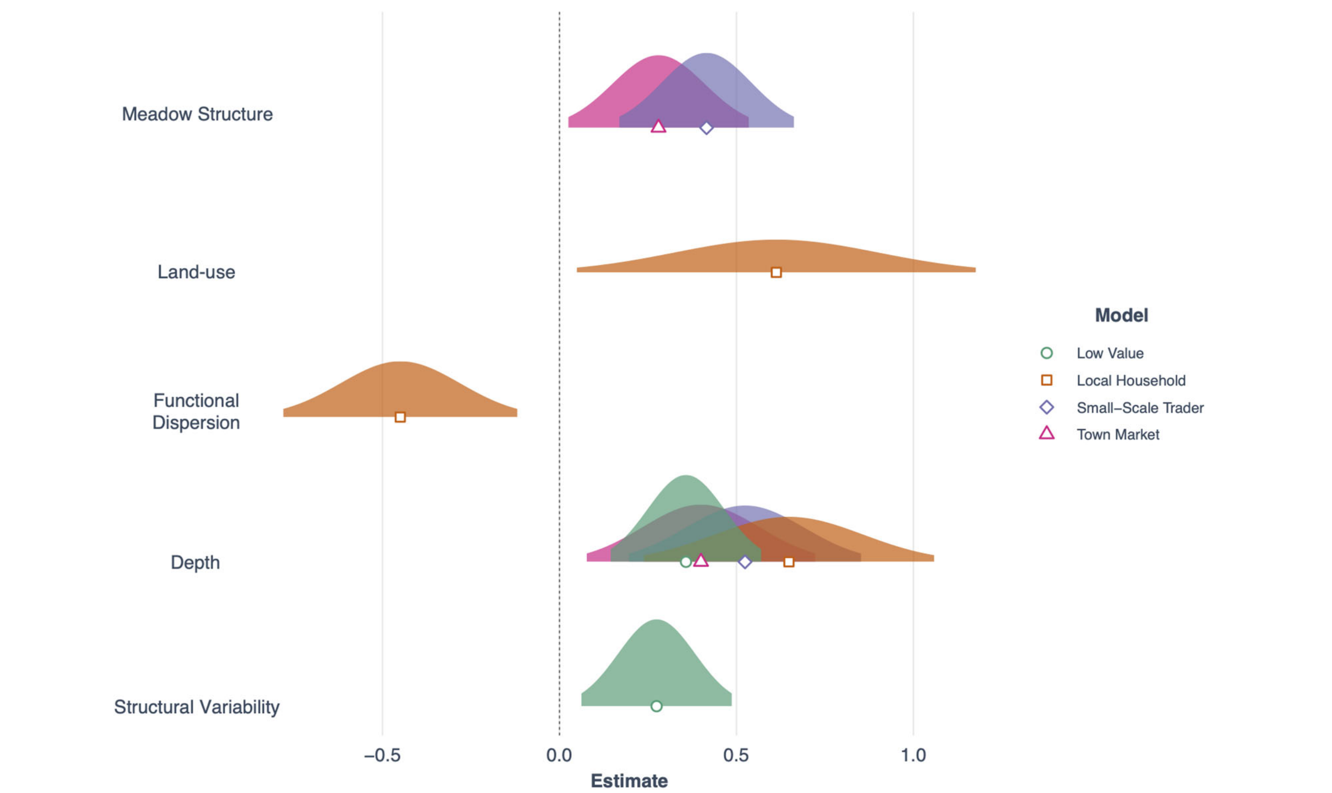

FIGURE 6 | Forest plot showing the effects of seagrass and environmental predictors across the four fish value categories used in this study (Low value, Local household, Small-scale trader and Town market). Plotted are normal distributions of 95% confidence intervals.

that seagrass species diversity (both taxonomic and functional) had little effect. Understanding how biodiversity influences the structure and function of faunal assemblages is key (Morin et al., 2018), and a necessity for more effective marine protection. While trait-based approaches have been utilized to gain a greater understanding of the mechanisms that drive fish assemblages in coral reefs (Darling et al., 2017), this study is one of few to use such an approach for seagrass meadows.

In our study, seagrass meadow structure consistently drove two fish assemblage indicators; abundance (MaxN) and richness. Specifically, seagrass meadows with greater structure (e.g., higher canopy, longer and wider leaves, greater numbers of leaves per shoot, and lower overall shoot density) harbored faunal assemblages with greater abundance and richness (**Figure 7**). We stress here the importance of structural complexity in influencing seagrass fish assemblages (Gullstrom et al., 2008), given that more structurally complex meadows increase habitat complexity and the availability of food resources, and reduce predation pressure (Hovel et al., 2002; Vonk et al., 2010). However, in our analysis, we compressed numerous seagrass trait characteristics into a single variable, meadow structure, which accounted for 58% of variability across sites. Therefore, in our attempt to create an index for seagrass structural complexity, we advertently lost variability which is important to note when interpreting these findings.

Unlike any previous work (which focus on one or two seagrass species), our study tested the relative importance of all seagrass genera found within the tropical seascape and can suggest that the important traits (e.g., long leaves, high canopy) were primarily driven by dominant seagrass species rather than high seagrass species diversity (e.g., mass ratio: Grime, 1998; Díaz et al., 2007). So far we can only speculate on the reason(s) behind the lack of a seagrass diversity (richness, functional dispersion, etc.) effect. One potential reason could be that the diversity gradient is relatively short (1-6 co-occurring species in plots) compared to that in e.g., terrestrial grasslands, or that co-occurring seagrass species may compete more than complement each other (e.g., many are functionally similar in terms of structure). It should also be noted that our results do not exclude the possibility of stronger seagrass diversity effects at larger spatial and/or temporal scales (Bracken et al., 2017) or for other processes/functions or ‘ecosystem multifunctionality’ (Lefcheck et al., 2015).

Depth was also a factor driving fish assemblages, supported by previous studies conducted in Zanzibar (Gullstrom et al., 2008; Alonso Aller et al., 2014), as well as others in the tropics that show that deep seagrass meadows are important for species such as emperors and rabbitfish (Hayes et al., 2020). Given both these fish groups accounted for a large proportion of overall fish abundance, this may explain these findings. Using a novel technique to locate seagrass meadows within the region,

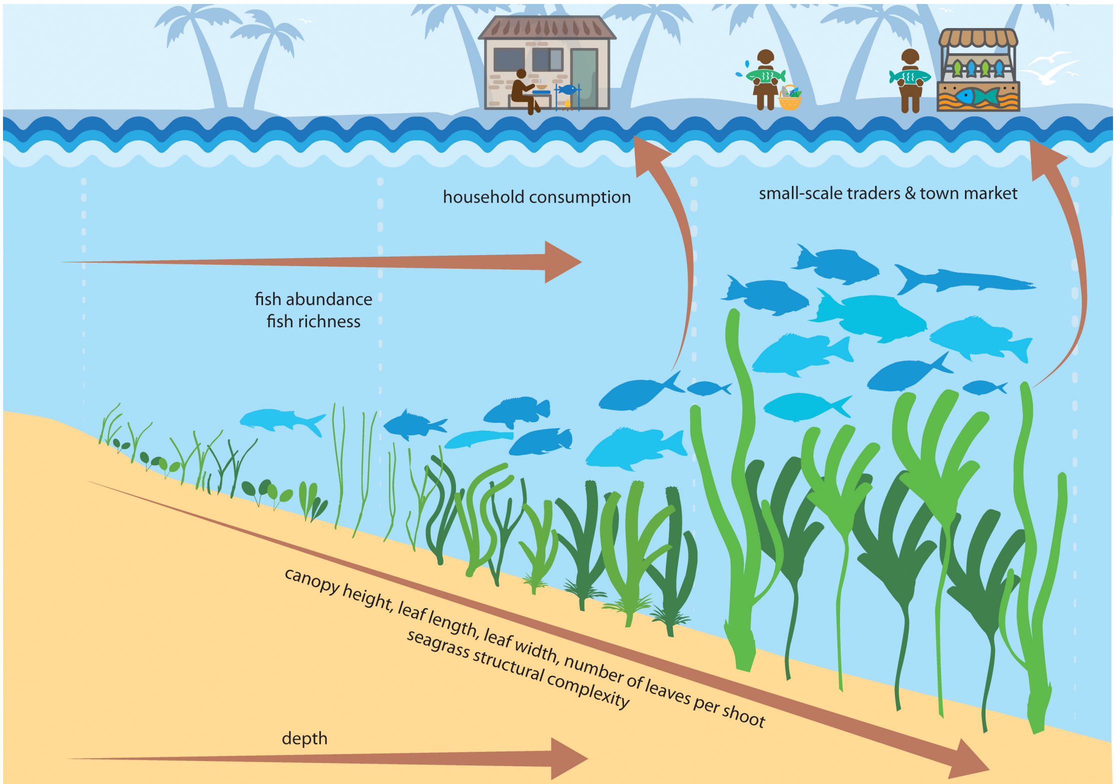

FIGURE 7 | Conceptual diagram showing how seagrass structural complexity influences seagrass social-ecological systems. Seagrass meadows with taller canopies, longer and wider leaves and more numerous leaves per shoot have greater fish abundance and richness, and in particular support species of value to small-scale fisheries.

Esteban et al. (2018) showed that deep seagrass meadows, comprised of a single dominant species (*T. ciliatum*), supported diverse and near-pristine fish assemblages. The broadly positive effects that we see here may be due to several factors such as their likely position within the seascape and the increased likelihood of being in closer proximity to adjacent coral habitat (Gullstrom et al., 2008).

Seagrass fish richness was best explained by an interaction between seagrass cover and land-use intensity; a variable negatively correlated with fishing pressure. Conforming to previous studies in the region (e.g., Alonso Aller et al., 2014), the positive effect of seagrass cover on fish richness decreased with increasing human impact (higher land-use), with no effect where human populations were highest. However, in our study, we surprisingly found that human impacts had no single effect on fish richness. Where there was greatest human development, fish richness remained high regardless of whether seagrass cover was low or high. One possible explanation for this lack of effect may be that high human development in fact acts to buffer local fishing pressure and benefit local fish populations, because of the high frequency of boat traffic but also potential concerns over water quality in these areas. Such sites were Stone Town Harbor and Maruhubi, situated just outside the Port of Zanzibar (**Figure 1**). The area is well-known as being polluted (De Wolf and Rashid, 2008) and highly disturbed (Khamis et al., 2019), but still harbored high abundances of both fish and perceivably healthy seagrass communities. In these areas, seagrass cover (or meadow structure) had no effect on fish richness; the simple presence of seagrass was enough to support high fish richness.

In general, the strong effects of seagrass meadow structure, seagrass cover and depth were also observed when looking at the abundance of fish species valuable to town markets and small-scale traders. The fish species included in these value categories were primarily predators (fish and invertebrate feeders), with exception of herbivorous parrotfish and rabbitfish. For the predators, the positive effects of complexity are likely driven by increased food availability (Hovel et al., 2002; Vonk et al., 2010); more complex structures provide a greater range of habitats for fish and invertebrates. However, these predators are in turn also eaten by larger predators occasionally feeding in seagrass areas (e.g., sharks), so they may also benefit from seagrasses for shelter. The key role of depth here is specifically important for herbivores given commercially valuable rabbitfish species use networked habitats (coral and seagrass), but have a small home range of <200 m (Ebrahim et al., 2020a). Deeper seagrass meadows, with higher structural complexity, are generally closer to adjacent coral structures. Given that rabbitfish can dominate catches across the Western Indian Ocean (Hicks and McClanahan, 2012) and are a source of protein that sustain humans populations (Grandcourt and Cesar, 2003), our findings provide evidence that high structure seagrass meadows are key in supporting this provision. Moreover, meadows of this type support the fish species of value to fish traders (so called 'middlemen'); a stakeholder group instrumental in shaping social-ecological small-scale fishery systems (Crona et al., 2010; Fröcklin et al., 2013).

Three families key in the household consumption category – grunts, snappers and rabbitfish (Thyresson et al., 2013) – are reportedly declining in the region (Benansio and Jiddawi, 2016). Their abundance was negatively influenced by functional dispersion – seagrass meadows with similar traits were more important – but positively influenced by depth and land use. Given that seagrass structure was not a driver, these effects were likely driven by meadows comprised of intermediate genera like *Thalassia* and *Cymodocea* which have similar traits, that were closer to areas of human populations with greater epiphyte coverage. Epiphytic material is an important food source for many small invertebrate grazers in tropical seagrass meadows (Belicka et al., 2012), which can be a primary food source for juvenile grunts and snappers (de la Moriniere et al., 2003). Likewise, epiphytes constitute an important food source for generalist herbivores like rabbitfish (Ebrahim et al., 2020b).

Fish categorized as low value were primarily small bodied omnivores, and while the effect size was low, the local abundance of these species was driven by high variability in seagrass structural traits (i.e., high coefficient of variation for seagrass structure). Omnivores select food based on a range of characteristics including nutrition, shape and texture. Meanwhile, Prado and Heck (2011) found that variability in seagrass structural traits — different seagrass shapes and sizes — were a dominant driver of food choice. Specifically, this is because narrow (e.g., *Halodule*, *Syringodium*) and paddle shaped (e.g., *Halophila*) seagrass leaves are easier to manipulate for smaller bodied fish (Prado and Heck, 2011). This likely explains why the abundance of this fish category increased with trait variability but was not influenced to the same extent by meadow structure (e.g., dominant traits). Finally, it should be noted that while these species were low value in fishery terms, they can contribute to important ecological functions such as regulating epiphyte loads (Gilby et al., 2016).

Conservation is generally underpinned by protecting key species or habitats (Caveen et al., 2014). This has led to the creation of numerous well intentioned yet misplaced marine protected area's (MPA's), which fail to meet the function they were intended for Jantke et al. (2018). In the wake of calls to acknowledge seagrass meadows within the marine conservation agenda (e.g., Unsworth et al., 2019a), we are increasingly seeing seagrass meadows protected or highlighted as priority areas for protection (e.g., as nursery areas or for 'blue carbon'). However, a failure to align policy with science, means these protected seagrass areas could have a limited function for the reasons that they have been protected. For example, Cambodia's first MPA designates seagrass for its nursery function (Boon et al., 2014), yet the meadow in question is dominated by sparse growing *Halodule* and *Halophila* spp. (Phalla et al., 2014), which based on our analysis here could provide very limited function as a fish habitat due to their trait characteristics (e.g., low canopy, small leaves, low number of leaves per shoot). BRUV surveys conducted within the MPA in Cambodia support this, with very low fish abundance (Gourlay, 2017).

It is prudent to acknowledge that traits associated with fish assemblages may not be the only important variables to consider when thinking about ecosystem services more broadly. For example, the aboveground traits that were important for driving fish assemblages in this study, such as high canopy and increased seagrass cover, have little effect on sedimentary carbon trapping, which is instead driven by belowground traits and sediment type (Gullström et al., 2018). Therefore, for effective management of seagrass meadows across the Indo-Pacific, we stress the importance of pairing reason for protection with ecological functions. This is fundamental given calls for community seagrass conservation based on payments for ecosystem services (UNEP, 2020). As seagrass ecosystem provision is non-linear among seagrass taxa it is not economically, ecologically or socially efficient and sustainable to protect seagrass meadows if the seagrass species present do not provide the ecosystem services in question (Nordlund et al., 2016).

In conclusion, we show that fish assemblages in tropical seagrass meadows are driven by characteristics provided by dominant species, such as high structural complexity and cover, as more broadly by depth. Consequently, we show that seagrass diversity (both functional and species) had little effect on fish assemblages. While this study was conducted in Zanzibar Island, Tanzania, the seagrass species studied here are common across the Indo-Pacific in similar densities (e.g., Short et al., 2007; Ambo-Rappe et al., 2013; Jones et al., 2018; Jinks et al., 2019), provide habitat for the same groups of fish families (Unsworth et al., 2008; Pogoreutz et al., 2012; Honda et al., 2013) and are subsequently targeted by small-scale fishers (e.g., Cullen-Unsworth et al., 2014; de la Torre-Castro et al., 2014; Quiros et al., 2018). Our findings are therefore not just useful for Zanzibar, but across the Indo-Pacific and show that different types of seagrass meadows have different values for both ecology and society. Placing the results of this study in a social-ecological systems context, focusing on fisheries, then leaves us in a predicament. From an ecological perspective, priority areas for seagrass conservation are high cover and high canopy, as these are areas with both the highest abundance and richness of fish species. Yet from a social perspective, these areas are likely the best and most favorable fishing grounds. On the one hand, protecting these habitats either as an MPA or no take area would therefore force fishers to increase fishing effort in other areas, with implications for livelihoods and food security. On the other hand, protecting seagrass areas with the most abundant fish assemblages may, in theory, generate enough 'spillover' to enhance fisheries in those nearby areas (McClanahan and Mangi, 2000; Januchowski-Hartley et al., 2013). Our study revealed that seagrass sites in Zanzibar close to large human populations have high fish richness and abundance, and low relative fishing intensity. While the mechanisms underpinning this need further study, qualitative observations suggest that fishers perceive these areas as either too 'dirty' (polluted) or inaccessible due to the high frequency of boat traffic, to utilize as fishing grounds. These findings somewhat flip the so called human 'gravity' hypothesis that exists for coral reefs (Cinner et al., 2018), i.e., that with increasing human population size and accessibility to coral reefs conservation gains are diminished (increased fish biomass and predators). Whether this is specific to Zanzibar or a wider case for seagrass warrants further investigation. If this is indeed the case, then these relatively unfished areas adjacent to areas of high land-use could become a priority, or an easy win, for fisheries management strategies, that could help fuel recruitment for nearby and fished areas.

DATA AVAILABILITY STATEMENT

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found below: All data and R code is available from https://github.com/BoardshortsBen/SeagrassFish.

AUTHOR CONTRIBUTIONS

BJ, JE, and LN conceived the ideas and designed the study with methodological input from RU and site choice input from NJ. BJ collected and analyzed all data and led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.640528/full#supplementary-material

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The Need for Social Considerations in SDG 14

Marcus Haward[1,2]* and *Bianca Haas*[1,2]

[1] Institute for Marine and Antarctic Studies, Hobart, TAS, Australia, [2] Centre for Marine Socioecology, University of Tasmania, Hobart, TAS, Australia

Sustainable Development Goal 14 acknowledges the need for action to achieve a sustainable future for our ocean. Many initiatives are working on ocean-related issues; however, social problems are often overlooked. In this article, we argue that to achieve a sustainable ocean, social aspects need to be considered. We explore the link between SDG 14 and SDG 8 as labor and working conditions on fishing vessels receive increasing attention. Regional Fisheries Management Organizations have the mandate to manage fisheries at the high seas, therefore, we argue that these organizations need to act on, and implement, resolutions and measures, addressing labor standards. Labor conditions related to the fishing sector have not received the level of scholarly attention that they deserve, thus more research is needed.

Keywords: fisheries management, human rights, labor conditions, regional fisheries management organizations, International Labor Organization

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Baris Salihoglu,
Middle East Technical University,
Turkey

***Correspondence:**
Marcus Haward
marcus.haward@utas.edu.au;
bianca.haas@utas.edu.au

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INTRODUCTION

The importance of the oceans is globally acknowledged with important political events happening to discuss the sustainability of our oceans. In 2020 the Decade of Ocean Science for Sustainable Development to run from 2021 to 2030 was announced by the United Nations, aiming to stimulate action and funding for ocean science (UNESCO, 2020). Interdisciplinary and solution-oriented science are imperative to achieve a healthy and sustainable ocean (Visbeck, 2018). This decade is inherently linked with the United Nations Sustainable Development Goals (SDGs), especially SDG 14, *life below water*. This SDG addresses all the major issues related to the ocean, such as marine pollution, overfishing or ecosystem degradation.

The work by Singh et al., 2017, emphasized the linkages between SDG 14 and other goals and the need to link environmental sustainability and social and economic issues. The SDGs do provide a significant opportunity to build on the promise of the Rio + 20 Summit in 2020 for "the future we want" and other international initiatives related to the oceans. The UN Decade of Ocean Science provides great opportunities but may understate social and economic considerations (Fleming et al., 2019) in an emphasis on blue growth or the blue economy. We believe that a sustainable ocean cannot be achieved without taking social considerations into account (Rudolph et al., 2020). Thus, we are interested in what we believe to be a neglected area of fisheries governance, that of working conditions of crew on fishing vessels. This article focuses on the need to explicitly address linkages between SDG14 and SDG8, *decent work and economic growth*.

In this article, we are focusing on Regional Fisheries Management Organization (RFMOs), which have mandates to manage fisheries at the high seas. A study by McDonald et al. (2020) showed that the risk of forced labor is neither solely linked to exclusive economic zones (EEZs) nor high seas, but occurs globally. Moreover, one of the key areas of focus for RFMOs is the problem of illegal, unreported and unregulated (IUU) fishing that has often been linked to forced work and labor abuse on fishing vessels (Marschke and Vandergeest, 2016). While we focus on RFMOs, it is, however, important to note that there are also other international fisheries organizations, such as the Food and Agriculture Organization that also address this issue.

It is important to examine this issue through the lens of the SDGs since social issues and non-compliance with management regulations are linked with each other (Bennett, 2019). While not directly addressed under SDG 14, the issue of labor conditions is gaining increased attention at meetings of RFMOs. Even though some member states consider that labor standards are outside of the mandate of RFMOs, criticism that they are avoiding this topic may also increase the organization's reputational risk. We note, too, that in 2018, the Western and Central Pacific Fisheries Commission (WCPFC) adopted a non-binding resolution on labor standards for crews on fishing vessels (WCPFC, 2018). At its most recent Commission meeting in 2020, the WCPFC agreed to work on a conservation and management measure targeting crew and labor conditions on fishing vessels (WCPFC, 2020).

Generally, it is important to address this issue in a scientific manner as currently much of the information linking IUU fishing and labor conditions has been provided by journalists and non-governmental organizations (Marschke and Vandergeest, 2016). While research on this area is increasing, most peer-reviewed literature has focused on case studies such as the offshore fishery in Thailand (Marschke and Vandergeest, 2016; Vandergeest and Marschke, 2020), Myanmar (Belton et al., 2019), or New Zealand (Stringer et al., 2016). The first section of the article provides an overview of the SDGs and especially key aspects and targets within SDG 8 and SDG 14. The following section addresses labor issues in the fisheries sector, noting the relative salience in contemporary fisheries governance, yet at the same time, we recognize increased attention given to this issue by the International Labor Organization. The final section of the paper provides a synthesis of these key issues and outlines the potential solutions to the current gap in fisheries governance.

THE SUSTAINABLE DEVELOPMENT GOALS

The *2030 Agenda for Sustainable Development* with 17 Sustainable Development Goals (SDGs) was adopted in 2015 (Le Blanc et al., 2017). A number of these goals reiterate, reinforce, and/or consolidate previously agreed actions and link to existing international instruments such as the United Nations Convention on the Law of the Sea and the United Nations Fish Stocks Agreement[1] (UNFSA). The SDGs evolved from the UN Millennium Development Goals of 2000 and the principles contained in the 1992 *Rio Declaration on Environment and Development* (Fukuda-Parr, 2013). While considerable attention has been given to each of the goals as drivers for change, less attention has been shown toward the linkages between the goals as highlighting areas for action. Singh et al. (2017) emphasized that the different goals cannot be achieved in isolation and the separation of social and ecological aspects need to be minimized. It is time that issues of ecological sustainability are linked with social issues, as it is difficult to fully address the former without paying attention to the latter.

This is clear with respect to SDG14 (*Life below water*) that aims to "conserve and sustainably use the oceans, seas and marine resources for sustainable development" (SDG14 – **Supplementary Appendix 1**) and provides a focus for ongoing action by addressing seven targets and three sub-targets many of which have a direct influence in emerging fisheries governance (Haas et al., 2019), see **Supplementary Appendix 1**. For example, SDG14 Target 4: "By 2020 effectively regulate harvesting and end overfishing, illegal, unreported and unregulated fishing and destructive fishing practices and implement science-based management plans"..., reinforces the role of regional fisheries management organizations (United Nations [UN], 2018). This target has a direct link to Article 10 of the UNFSA that outlines the "functions of subregional and regional fisheries management organizations and arrangements" (United Nations [UN], 1995).

Effective regulation and management of natural resources are important for the realization of human rights. The issue of labor standards and decent work conditions is addressed by SDG 8 (*Decent work and economic growth*) and its 12 targets. This SDG aims to "promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all," see **Supplementary Appendix 2**. Fishing involves hard and dangerous work, often called 3Ds work (dirty, dangerous, and difficult) in an unforgiving environment. While the International Labor Organization (ILO) recognize that a majority of fishing vessel operators comply with regulations and avoid "decent work deficits" (ILO, 2016) it is also recognized that the sectors "is notorious for severe decent work deficits and has come under scrutiny over the past years for the use of forced labor and child labor, as well as links to human traffickers and people smugglers" (ILO, 2016: v).

In the context of fisheries, two targets of SDG 8 are especially relevant as they can be directly linked to two targets of SDG 14. Target 8.7 – end modern slavery, trafficking and child labor – is partly dependent on the progress made in Target 14.4 – sustainable fisheries – which calls for an end of all illegal, unreported and unregulated fisheries. As previously noted, illegal, unreported and unregulated fishing has often been associated with forced work and low labor standards (Marschke and Vandergeest, 2016). Target 8.7 is also linked to target 14.6 – end subsidies contributing to overfishing – as subsidies play a notable role in overfishing, which pressurizes fishing companies to save money on labor costs to make a decent income. The second

[1]The United Nations 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 (in force as from 11 December 2001).

target is 8.8 – protect labor rights and promote safe working environments – which is also linked to the targets 14.4 and 14.6.

LABOR CHALLENGES IN CURRENT FISHERIES GOVERNANCE

Fishing is important for the livelihood of millions of people. Approximately 59.5 million people worked in the primary sector of fisheries and aquaculture in 2018 (FAO, 2020a). Most of the workers live in developing countries, where a lack of controls and regulations make them especially vulnerable to labor abuse (OSA, 2020). It has been estimated that around 24.9 million people are victims of forced labor (OSA 2017), with an estimated 1.76 million workers in the fisheries and agriculture sectors (ILO, 2017a).

The International Labor Organization (ILO) has adopted two instruments in the last decade which are central to addressing these matters: the Work in Fishing Convention, 2007 (No. 188) and the Protocol of 2014 to the Forced Labor Convention, 1930 (PO. 29). These instruments are claimed by the ILO to provide a comprehensive framework for regulating working conditions (ILO, 2016). Other key ILO instruments, including the Freedom of Association and Protection of the Right to Organize Convention, 1948 (No. 87), the Right to Organize and Collective Bargaining Convention, 1949 (No. 98), the Labor Inspection Convention, 1947 (No. 81), and the Private Employment Agencies Convention, 1997 (No. 181) are important to the promotion of decent work in fishing (ILO, 2016: v).

The most important instrument for labor issues in the fisheries sector is the ILO Work in Fishing Convention No. 188. This convention sets standards for issues such as occupational safety, health and medical care at sea and ashore, written work agreements and living conditions on board (ILO, 2017b). While this convention entered into force in November 2017, following ratification by 10 states (eight of whom were coastal states), to date it has only received 17 ratifications and so is not considered to be as influential as it could be in driving changes.

Another important instrument is the Cape Town Agreement developed by the International Maritime Organizations. This agreement (concluded in 2012 but yet to enter into force) provides minimum global standards, aims to ensure the safety of fishing vessels and their crew (FAO, 2021). Other important instruments concerning labor standards and crew welfare include for example the IMO Convention on Training and Certification for Fishing Vessel Personnel or the FAO Code of Conduct for Responsible Fisheries, especially article 8, which calls for decent employment and social security (FAO, 2020b).

So far, the Western and Central Pacific Fisheries Commission is the only RFMO that has a non-binding resolution in place concerning this issue (WCPFC, 2018) and at the 17th Commission meeting in 2020, the members agreed to work intersessionally on a conservation and management measure (i.e., binding) on improving crew labor standards. However, the South Pacific Regional Fisheries Management Organizations performance review panel highlighted the need to engage with this topic due to increasing global interest (Ridings et al., 2018).

Besides the issue of working conditions, forced labor and at worst slavery is another serious issue in the fishing industry (Tickler et al., 2018). This issue is addressed by SDG 8.7 and 8.8 which aim to eradicate forced labor, protect labor rights and promote safe and secure working environments (**Supplementary Appendix 2**). While the issue of labor conditions received some attention in RFMOs, the issue of forced labor and slavery has not yet been addressed.

ADDRESSING THE PROBLEM – LINKING THE SDGs TO ACTION

To fully achieve the aspirations of SDG 14 it is important to address social concerns such as labor issues. RFMOs are the main organizations handling fisheries matters in areas beyond national jurisdiction, thus, we argue that these organizations need to acknowledge and address this issue. RFMOs need to follow the example of the Western and Central Pacific Fisheries Commission (WCPFC) and adopt a resolution or binding conservation and management measures concerning labor standards. As a result, there is a need for greater collaboration between RFMOs and the International Labor Organization, to assure that members are enforcing labor standards on their fishing fleets and to encourage members to ratify the ILO Convention No. 188. Even though it can be argued that these matters are outside the traditional mandates of RFMOs, RFMOs are the only organizations that directly address the fishing industry and, therefore, are a key platform from which to discuss labor issues.

It is important that RFMOs establish binding standards and guidelines, as national laws are severely limited in application to international waters. One important aspect is the collection of data. The increasing application of monitoring control and surveillance systems provides an opportunity to collect these data but also to ensure compliance with existing resolutions and measures. Moreover, there is a strong push for member states to enforce the UNFSA and the FAO Port State Measures Agreement. These latter instruments would provide another layer of monitoring and surveillance of working conditions.

The Agreement on Port State Measures to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated Fishing (Port State Measures Agreement) was adopted by FAO members in November 2009. The Agreement entered into force in 2016. A key element is "the threat of the denial of the use of ports and their services is a key enforcement thread that runs throughout the 2009 Port State Measures Agreement" (Witbooi, 2014, p:300).

Port state control is a key tool in the regulation of merchant shipping and is a key tool in addressing ship safety, environmental performance (i.e., control of ship-sourced pollution) and seafarer safety and welfare. Port state measures have been relatively slow to be applied to fisheries, even though there is no doubt that a coastal "state can assert maximum enforcement jurisdiction over their internal waters" (Telesetsky, 2015: 1244). Witbooi comments that "although RFMOs, on the whole, have agreed on adequate strict port state measures, they are frequently at fault for failing to ensure that their members implement these measures consistently and effectively" (Witbooi, 2014, p:302).

Under UNFSA a port state has the right and duty to take certain measures, including to "inspect documents, fishing gear and catch on board vessels when such vessels are voluntarily in its ports" (UNFSA, Article 23, see Serdy, 2016, p: 427). A port state may adopt regulations "prohibiting landings and transshipments where it has been established that the catch has been taken in a manner that undermines the effectiveness of subregional, regional or global conservation and management measures on the high seas" (UNFSA, Article 23, see Serdy, 2016, p: 427).

A further issue relates to at-sea transshipment of fish, which indirectly also impacts labor conditions. For example, the members of the WCPFC proposed a transshipment ban on the high seas. While this proposal may be driven by economic interests gained by port access fees, transshipment is also linked with illegal, unregulated and unreported (IUU) fishing. Furthermore, banning high seas transshipment could reduce human trafficking, forced labor and bad labor conditions due to greater control over the respective vessels (Ewell et al., 2017). However, it is important to note that addressing IUU fishing only marginally addresses the issue of labor abuse, as it is more an add-on than a primary factor affecting working conditions (Marschke and Vandergeest, 2016).

Important drivers for better working conditions are the attitudes of market states. For example, in 2015 the European Union issued Thailand with a "yellow card," which affected Thailand's ability to export fish products to the EU. In 2019, the EU lifted the "yellow card," as Thailand had successfully addressed shortcoming in its fisheries management approach. The EU highlighted work done on human rights abuse and forced labor in the fisheries sector as part of these reforms (European Commission [EC], 2019). Generally, non-state actors such as industries play an important role in addressing the issue of labors and work conditions, and there has been an increasing call for the inclusion of social conditions in fisheries certification schemes and assessments (Fleming et al., 2020). For example, the members of the International Seafood Sustainability Foundation (ISSF) have adopted a conservation measure that requires seafood companies to have policies for social and labor standards in place, throughout the whole supply chain (ISSF, 2020) **(Box 1)**. Another example is the International Pole and Line Foundation (IPNLF) also committed to social sustainability, addressing areas such as decent working conditions and gender equity (IPNLF, 2021). While these are only two examples, it shows that the industry is starting to tack this issue seriously. The fishing industry also influences decisions in RFMOs and might be an important driver to emphasize the importance to address labor issues on RFMO level. The development of third-party assessment and certification in fisheries open further areas of activity. This includes a continuum of processes and approaches from producer-based self-certified and labeled place-, or product-based label, through to rigorous third party independent certification, using processes external to, and separate from, the producer (Potts and Haward, 2007). Third-party non-state actors have long been active in debates over sustainable resource exploitation. One of the most known certification standards in the fisheries sector is the Marine Stewardship Council.

BOX 1 | ISSF.
The ISSF is an non-governmental organization aiming to undertake and facilitate science-based initiatives to ensures long-term sustainable use of tuna resources and minimizing environmental impact. Participating companies are all members of the International Seafood Sustainability Association, which have to comply with the ISSF conservation measures. Members include for example Tri Marine, Bumble Bee, or Thai Union. To achieve its mission the ISSF engages with RFMOs, for example.

The Marine Stewardship Council (MSC), established in 1996, is an example of an approach to governance that steps outside state-based governance and address market and consumers directly through product certificates and ecolabels (Potts and Haward, 2007). The heart of the MSC process is the certification of "sustainable fisheries" under its standard defined by Principles and Criteria, and linking this certification to a logo that influences consumer behavior and provides price signals (Potts, 2006; Lee, 2009). This certification process is independent of the MSC; it does not directly perform certifications but remains a standard-setter that accredits qualified certification organizations and trains them in the methodology to be applied. Control over the certification process, that is auditing certifiers and the application of standards, are the core functions of the MSC. MSC currently accredits organizations, termed Conformity Assessment Bodies (CABs) for MSC certifications. The CABs are also subject to monitoring by a further independent body, the Accreditation Services International (ASI), providing further checks and balances in the process.

Of the three principles that underpin the MSC process, Principle 3 is the most relevant in this case. Principle 3 states:

The fishery is subject to an effective management system that respects local, national and international laws and standards and incorporates institutional and operational frameworks that require the use of the resource to be responsible and sustainable (MSC, 2018).

Principle 3 requires vessel operators not only to comply with local, national, and international law but also with regulations enforced by RFMOs. Companies which are flagged under a country which has ratified the ILO Convention No. 188 have to follow labor standards. This emphasizes the role of RFMOs in considering and promoting labor standards. The MSC has often been criticized for the lack of social consideration (Ponte, 2012; Kourantidou and Kaiser, 2019). However, assessing social considerations requires expertise which might not be covered by the MSC. Partnerships with organizations, which are focused on social accountability, might provide a way forward in addressing issues related to appropriate labor standards and link such standards to sustainable seafood production.

CONCLUSION

In this article, we argue that ocean sustainability cannot be achieved without adequate attention being given to social issues such as safe and humane working conditions. It is a clear failing of current fisheries governance that more attention is placed on the assessment of the conditions pertaining to fish being harvested than the assessment of the conditions of the people who are harvesting the fish. Therefore, we suggest a key way forward to address this failure is to actively explore the connection of SDG 14 and other SDGs, such as SDG 8, that deal, *inter alia*, with decent work conditions. There is an increasing interest and push toward increased working labor conditions on fishing vessels. It is important that RFMOs, which are responsible for international fisheries, act and implement resolutions or binding conservation and management measures. While members of these organizations might argue that labor standards and conditions are not within the RFMO's mandate, we argue that it is not only a social responsibility but also that there is an inherent reputational risk if RFMOs do not address this issue. Thus, RFMOs need, for example, establish greater collaboration with the International Labor Organizations and encourage its members to ratify the ILO Convention No. 188 on labor standards for fishing vessels. There is an increasing push from market states and non-state actors to consider social issues in the fisheries sector. Thus, it is important to further explore the linkages between SDG 14 and other social-related goals.

AUTHOR CONTRIBUTIONS

MH and BH contributed equally to the concept development and writing the manuscript. Both authors contributed to the article and approved the submitted version.

SUPPLEMENTARY MATERIAL

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Satellite Observations Are Needed to Understand Ocean Acidification and Multi-Stressor Impacts on Fish Stocks in a Changing Arctic Ocean

Hannah L. Green[1,2*], *Helen S. Findlay*[1], *Jamie D. Shutler*[2], *Peter E. Land*[1] and *Richard G. J. Bellerby*[3,4]

[1] Plymouth Marine Laboratory, Plymouth, United Kingdom, [2] Centre for Geography, Environment and Society, College of Life and Environmental Sciences, University of Exeter, Penryn, United Kingdom, [3] Norwegian Institute of Water Research, Bergen, Norway, [4] State Key Laboratory for Estuarine and Coastal Research, East China Normal University, Shanghai, China

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United States

***Correspondence:**
Hannah L. Green
hgr@pml.ac.uk

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It is widely projected that under future climate scenarios the economic importance of Arctic Ocean fish stocks will increase. The Arctic Ocean is especially vulnerable to ocean acidification and already experiences low pH levels not projected to occur on a global scale until 2100. This paper outlines how ocean acidification must be considered with other potential stressors to accurately predict movement of fish stocks toward, and within, the Arctic and to inform future fish stock management strategies. First, we review the literature on ocean acidification impacts on fish, next we identify the main obstacles that currently preclude ocean acidification from Arctic fish stock projections. Finally, we provide a roadmap to describe how satellite observations can be used to address these gaps: improve knowledge, inform experimental studies, provide regional assessments of vulnerabilities, and implement appropriate management strategies. This roadmap sets out three inter-linked research priorities: (1) Establish organisms and ecosystem physiochemical baselines by increasing the coverage of Arctic physicochemical observations in both space and time; (2) Understand the variability of all stressors in space and time; (3) Map life histories and fish stocks against satellite-derived observations of stressors.

Keywords: fish stocks, Arctic Ocean, satellite earth observation, multi-stressor, ocean acidification

INTRODUCTION

Models project that anthropogenic warming will increase the importance of the Arctic Ocean for supporting economically valuable fish stocks as marine species exploit new ranges and move northwards to remain in their thermal niches or as stock sizes increase (Lam et al., 2014; Cheung et al., 2015; Wisz et al., 2015). Populations of Atlantic cod and haddock, both of significant commercial value, have already expanded their range northwards (Renaud et al., 2012; Fossheim et al., 2015; Misund et al., 2016). Pacific Cod have also been seen to have a summer northward range shift (Spies et al., 2020). However, the Arctic Ocean is particularly susceptible to Ocean acidification (OA), and it is currently unknown how OA will manifest on these northward moving populations. OA is the change in ocean carbonate chemistry that occurs by the absorption of excess carbon dioxide (CO_2) into the ocean (Doney et al., 2009). Each year the ocean absorbs upwards of 25% of the anthropogenic CO_2 emissions (Friedlingstein et al., 2019; Watson et al., 2020), which has resulted in a 30% increase in hydrogen ion concentration (decrease in pH) since the industrial revolution. Recent assessments suggest regions in the Arctic are already seasonally corrosive to aragonite (a key mineral for some shell-building species) as a result of OA (IPCC, 2019). Warming further increases the Arctic's susceptibility to OA, with continued loss of multi-year ice increasing the surface area available for CO_2 gas exchange (Bates et al., 2006), while lower salinity and total alkalinity reduces the buffering capacity (Woosley and Millero, 2020). However, the remote and often hostile nature of the Arctic Ocean means collecting *in situ* data can be costly and challenging; this results in most data sets having a high seasonal bias toward the summer with little data collected under or around multiyear sea ice and/or during winter (Steiner et al., 2014).

The Intergovernmental Panel on Climate Change Special Report on the Ocean and Cryosphere in a Changing Climate acknowledged OA as a risk to shellfish fisheries but overlooked any risk to fin fisheries (IPCC, 2019). Whilst there is debate about the impact of OA on finfish (Kroeker et al., 2013; Haug et al., 2017), there is reasonable evidence from laboratory studies to suggest enough cause for concern (Frommel et al., 2012; Stiasny et al., 2016, 2018, 2019; Dahlke et al., 2017; **Supplementary Table 1**) and that the changing carbonate chemistry needs to be considered when assessing future fish stocks (Voss et al., 2019). Atlantic cod is the highest landed and most economically valued wild captured species in the Arctic (Pauly et al., 2020) and recent work predicts that although near-future conditions will at first be advantageous to the Atlantic cod (*Gadus morchua*) fishery in the Northeast Arctic due to reaching the optimal temperature for the spawning stock, once that optimal temperature is reached, further temperature rise combined with OA will lead to a steep decline in stock levels, and by year 2100 the fishery will be at risk of collapse (Hänsel et al., 2020).

Here our aim is to identify the current level of knowledge of the impact of OA combined with other climate change stressors on the most commercially important species in the Arctic. We do this by conducting a literature review, the results of which are primarily single species laboratory studies investigating the impact of OA and other combined stressors. These laboratory studies have their own limitations, and there are still large uncertainties regarding how to scale-up from a single species to ecosystem level (Hänsel et al., 2020). Here we discuss the major knowledge gaps, including some of the caveats of laboratory studies, before putting forward a roadmap of how to close those gaps using satellite observations as an additional tool.

ASSESSING KNOWLEDGE AND GAPS OF IMPACTS OF OA AND MULTIPLE STRESSORS ON FISH STOCKS

In order to conduct a literature search and assess the current knowledge, the most recently published annual dataset of finfish and shellfish landings in the Arctic Ocean was accessed: the year 2014 data from the "Sea Around Us" database (Pauly et al., 2020). The data was categorized into five biogeochemical regions based on previous work by Carmack and Wassmann (2006) and following Findlay et al. (2015): the Atlantic influenced seas (AiS) and Pacific influenced seas (PiS); the river influenced seas (RiS); the central Arctic Ocean (CAO); and the outflow shelves (OFS) (**Figure 1**). Further definition of these regions can be found in **Supplementary Table 2**. Forty-seven species were used in the literature search on OA and multiple stressors impacts. Details of the literature search are in **Supplementary Information** (section 1.0).

Knowledge Gap 1: Lack of Studies on OA Impacts on Fish, and Their Supporting Food Webs

The review identified that only ten of the top forty-seven landed species (by tonnage) in the Arctic have been studied for OA impacts, and not all these studies were performed on Arctic populations (**Supplementary Table 1** and **Figure 1B**). Therefore, 37 species remain untested. Responses were found to vary with species and life stage (**Figure 1B**), though eight of the ten species studied showed a negative response to OA in one or several of their life stages, particularly larvae and juveniles (Dupont et al., 2014; Dahlke et al., 2017).

Atlantic cod is the most well studied species, with 18 studies identified, results show a complex response to OA (**Supplementary Table 1**) with many unknowns: e.g., some studies show insignificant (Frommel et al., 2013) and significant (Dahlke et al., 2017) affects from OA on hatching, survival and development of Atlantic cod in the Baltic Sea, the processes behind these different responses are not yet known but it could be that different populations have local adaptations. The evidence for this is not clear, for example populations from the Barents Sea and Western Baltic Sea were both found to have their daily mortality rate approximately double under OA treatment, thus both populations showed similar responses to OA conditions (Stiasny et al., 2016). In other studies, complex responses were found. For example, Mittermayer et al. (2019) found a limited cellular response to OA in larvae yet at the same stage post hatch larvae were also found to have high mortality. The authors suggested that not enough is known about the mechanism that produces a response to OA in Atlantic Cod. Several studies have found responses to OA that are much more difficult to quantify in terms of ecological consequences such as changes in swimming turn angle and reduced stop duration (Maneja et al., 2013). It is clear further investigation into the mechanisms behind these response to OA is still needed.

Some species, including the Atlantic Cod (Stiasny et al., 2019), Atlantic Herring (Sswat et al., 2018), and Norway lobster (Wood et al., 2015) have shown greater tolerance to OA stress when food is plentiful, highlighting that a change in food supply is likely to have interactive effects with other stressors like OA. The food web perspective must be considered. Indeed, these links between stressors are complex: Sswat et al. (2018), for example, used an ecosystem perspective approach and found that increased primary production from

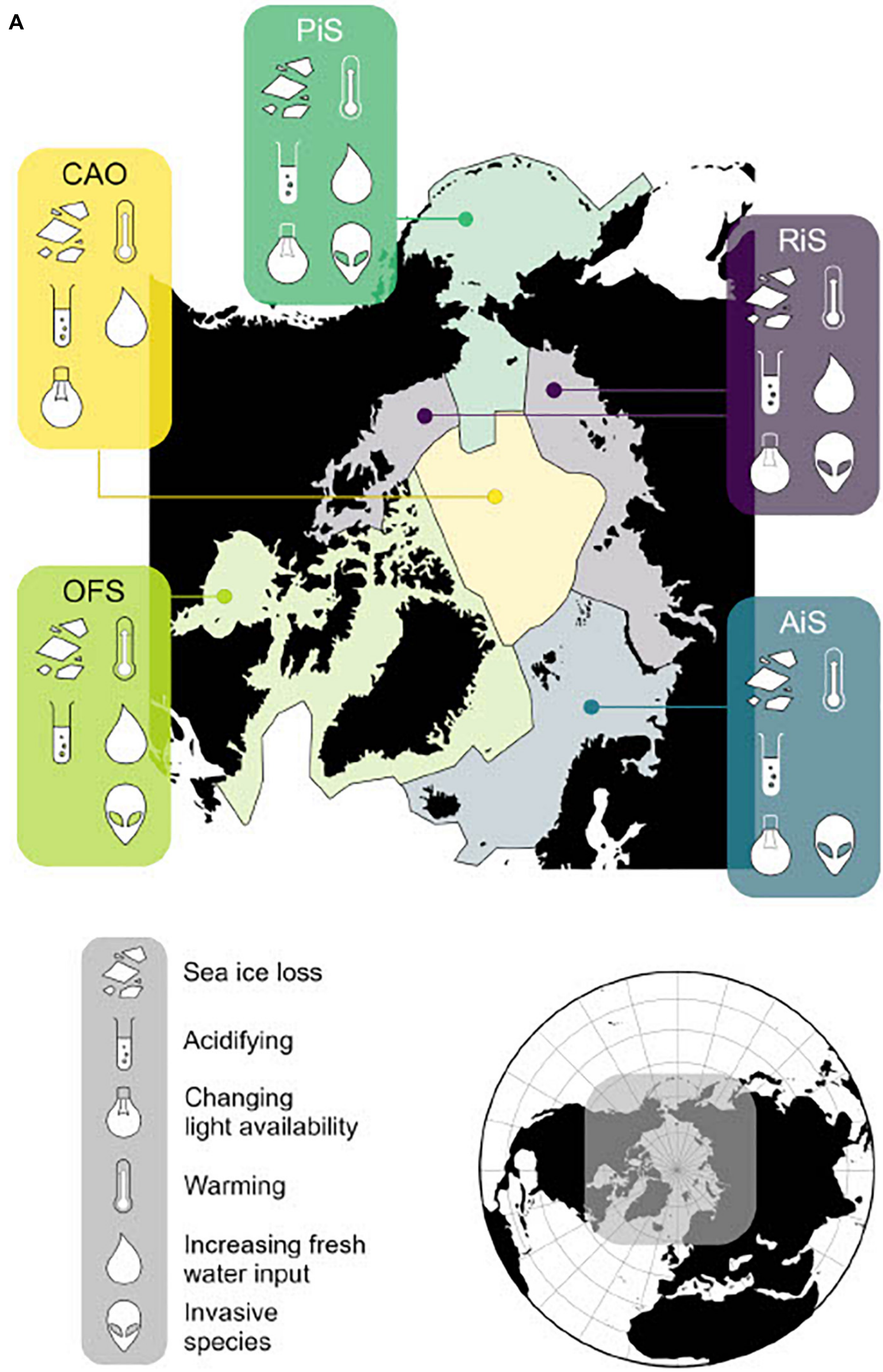

FIGURE 1 | Continued

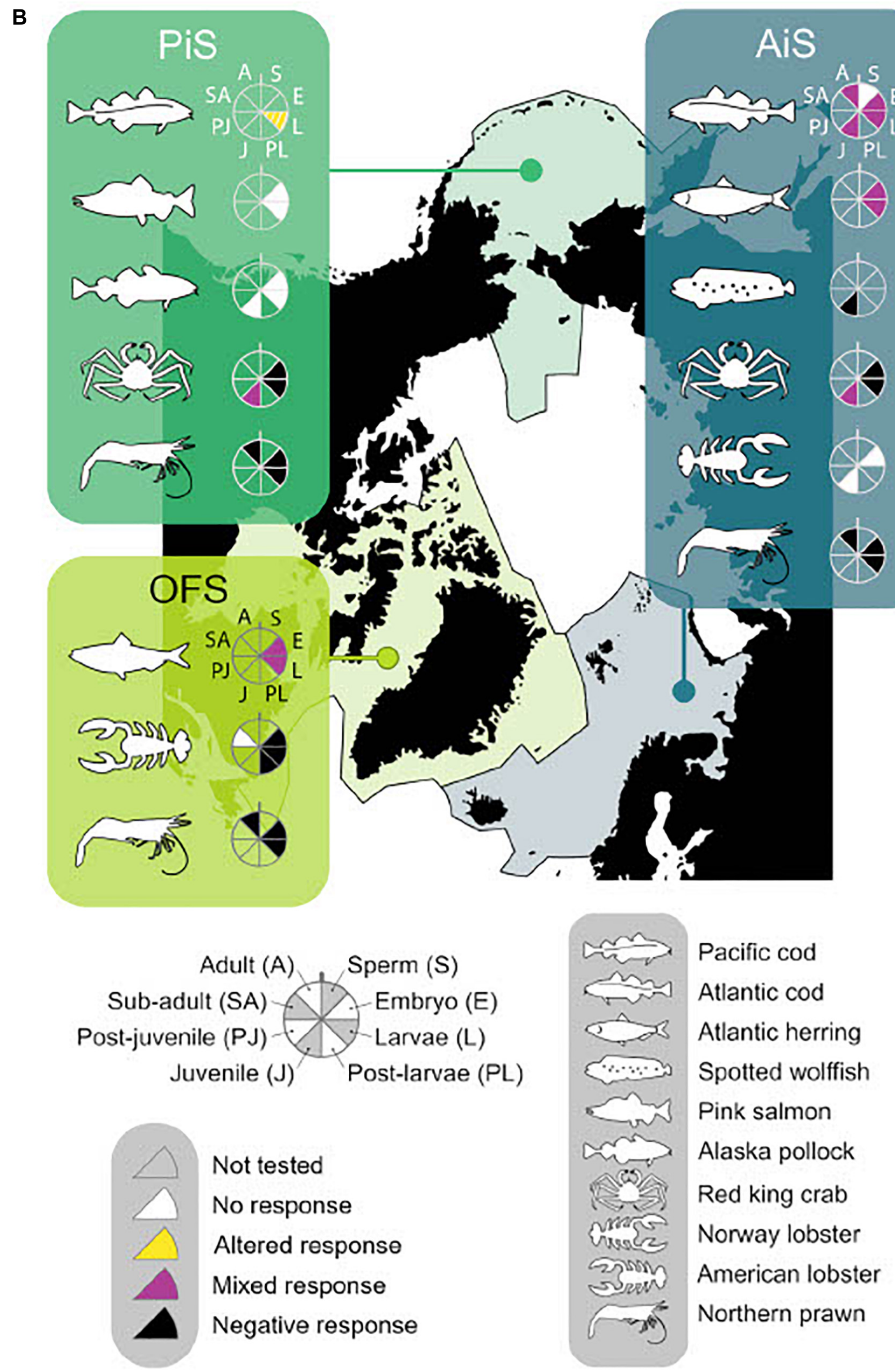

FIGURE 1 | (A) Arctic regional map showing different stressors for each region. AiS, Atlantic influenced seas; PiS, Pacific influenced seas; RiS, river influenced seas; CAO, central Arctic Ocean; OFS, outflow shelves (more detail refer to **Supplementary Table 4**). **(B)** Species icons represent each commercial species where response OA has been studied. Black = negative response, white = no response, blank = not tested, yellow = significantly altered response, purple = mixed response (this includes both positive, negative and significant altered response for more detail refer to **Supplementary Table 1**). Each life stage tested is represented by the slice of the wheel next to icon of species.

OA increased the survival of Atlantic herring larvae by 19%. This connection between increased primary productivity and alleviation of OA is particularly interesting in the context of the Arctic as the net primary production of the Arctic Ocean estimtated to have increased by 30% from 1998 to 2012 (Arrigo and van Dijken, 2015). However, primary production is also predicted to be limited by stratification, which itself is predicted to increase with ice reduction and higher temperatures (Slagstad et al., 2015). It is still unknown if other commercially important species will show the same alleviation from OA when there is more food available, and further work must be carried out to include this food web perspective.

Knowledge Gap 2: Lack of Understanding of Multiple Stressors

Understanding climate change interactions on a pan-Arctic scale is difficult as changes are not happening uniformly throughout the Arctic due to regional heterogeneity (**Supplementary Table 3** and **Figure 1A**). The river influenced seas, for example, experience a pH range of 7.6–8.3 (total scale) and salinity range of 0–34.4, compared to the Atlantic influenced seas, which have a pH range of 8.1–8.3 and salinity range of 13.6–35.4. Furthermore, tolerance capacity of organisms to OA has been shown to differ with additional stressors (Wood et al., 2015; Harrington and Hamlin, 2019; Stiasny et al., 2019). Organisms in the Arctic Ocean are experiencing changes in multiple oceanic conditions, including salinity, light, (Nicolaus et al., 2012; Langbehn and Varpe, 2017) and food or nutrient supply, predominantly arising from the impacts of sea ice loss (Barber et al., 2015; Polyakov et al., 2017; Dai et al., 2019) and increased river run-off (Peterson et al., 2002; Carmack et al., 2016; Woosley and Millero, 2020: **Supplementary Table 4**). The changes in salinity, light and food supply can act as stressors impacting on organism tolerance and sensitivity (Jørgensen et al., 2019; **Figure 1**). Indeed, only nineteen of the OA impact studies found in the literature search had been combined with another stressor (**Supplementary Table 1**). The main additional stressor tested was temperature, but food was also used as a secondary stressor in several of the studies. It is not yet understood the interactions of multiple climate change factors on organisms more generally, and how this might impact the food web, and therefore indirectly impact fish stocks (Faalenberyg et al., 2018). Only one study was found to have considered any stressors other than temperature and food in conjunction with OA: Hernroth et al. (2015) investigated the impacts of hypoxia and metal contaminant alongside OA. To the best of our knowledge, there are no studies on Arctic-relevant fish species that combine OA with changing light and/or salinity. It is therefore not yet possible to understand the impact of OA combined with additional climate change stressors on present and future Arctic fish stocks.

Knowledge Gap 3: What Environmental Conditions do Arctic Organisms Experience

Regional heterogeneity and natural variability have largely been neglected from experimental studies, and the Arctic is no exception. The literature review highlights that experimental OA impact studies used pH levels in their "control" treatments ranging from 8.08 to 7.97 (**Supplementary Table 1**), with most studies being toward the top end of that range. Given that *in situ* data shows that organisms in the Arctic may already periodically experience pH_T as low as 7.6 (**Supplementary Table 3**), it seems unlikely that these experimental levels reflect true ambient conditions. Furthermore, the Arctic Ocean has large seasonal variability and organisms are unlikely to experience the stable pH, pCO_2, salinity, oxygen and temperature conditions usually used in laboratory experiments (**Supplementary Table 1**) and are likely to experience different conditions at different life development stages. Seasonal variability can be as much as ~30 pss salinity, > 10°C temperature, > 200 μmol kg^{-1} oxygen (**Supplementary Table 3**). Indeed, some of the RiS already experience oxygen concentrations at or below 200 μmol kg^{-1}. Therefore, the lack of environmental specific treatment levels in experimental studies could be producing misleading results. It is not yet known if Arctic organisms may be pre-conditioned to high levels of pH variability and exposure to low pH and may consequently have some tolerance to long-term OA or have some level of adaptation (e.g., Vargas et al., 2017). The studies reviewed here were largely single generation, except for Stiasny et al. (2018) who exposed parents to OA conditions 6 weeks before spawning and found some evidence of transgenerational alleviation in Atlantic Cod when food was plentiful.

THE PATH AHEAD

Here we present a framework using satellite earth observation to address several of these key knowledge gaps and challenges to determine how present-day conditions and future projected changes will impact Arctic fish stocks. This approach has three inter-linked research aims: (1) Establish organisms and ecosystem physiochemical baselines by increasing the coverage of Arctic physicochemical observations in both space and time; (2) Understand the degree of variability of all stressors in space and time; (3) Map life histories and fish stocks against satellite observation data of stressors. In each of these sub-sections we discuss what is currently feasible with the present knowledge and technology and what additional developments are needed to achieve these aims.

Establish Baselines

The oceanographic and geographic characteristics of each region in the Arctic Ocean affect how each oceanic region is currently responding to climate change pressures and how they will respond in the future. Automated measurements from satellite observation underpinned by remotely operated vehicles, autonomous vehicles, and buoys (such as data collected from the International Arctic Buoy Programme) offers the only currently available solution to providing the necessary synoptic measurements of multiple oceanographic parameters to characterize surface environmental heterogeneity (Shutler et al., 2019). Satellite observation can be used to study environmental conditions important in polar waters (Shutler et al., 2019) including: freshwater fluxes (e.g., Nichols and Subrahmanyam, 2019); surface water temperature (e.g., Vincent, 2019); Chlorophyll-a concentration, primary production and net community production (e.g., Babin et al., 2015), and sea ice type and depth (e.g., Kwok, 2018). Recent developments have shown that satellite observation measurements of temperature and salinity can provide observational-based estimates of surface carbonate system conditions (Land et al., 2019). Although satellite measurements have additional challenges in the Arctic such as land and sea ice contaminations and radio frequency interference, there are now Arctic specific satellite reprocessed datasets to reduce biases (Olmedo et al., 2018). Ongoing development of algorithm work, together with improvement in uncertainties, could provide the ability to remotely observe and characterize multi-year Arctic-wide surface carbonate chemistry and its heterogeneity, and to identify longer-term variations in surface conditions. Therefore, satellite observation data has the potential to fill the gaps in the knowledge required to understand how OA combined with other climate change factors might affect fish stocks in the Arctic.

Understand and Monitor Temporal Variability and Exposure

Using satellite observation as a synoptic monitoring tool could provide additional data that can capture the variability on both a pan-Arctic and regional scale. Understanding the degree of variability in carbonate chemistry and other climate change stressors (**Figure 1A**) in space and time will be vital for improving model predictions. In addition, this will also provide the data for laboratory studies to be more representative of the environmental conditions organisms naturally experience, as well as providing data on a temporal scale for informing studies on multiple generations. This information is also valuable for the biological monitoring community, for understanding how changes in biological communities or processes are related to environmental change. Specifically here, monitoring the longer-term trends together with the variability in carbonate chemistry and other climate change stressors is relevant for understanding which regions are changing fastest and therefore which regions may need fishing and stock management strategies to be implemented.

Map Vulnerabilities

An aim for future research should be for satellite observation products to be used together with biological datasets to assess what environmental conditions species are experiencing during different life cycle stages. This is important, as many species spend time in different regions, including refugia, depending on their life stage. Mapping the physicochemical conditions alongside biological distribution data allows the identification of species or populations that presently live in more variable environments. This knowledge could be used to test the hypothesis that organisms already exposed to higher variability may have higher tolerance to future environmental change (Vargas et al., 2017). Mapping stressors for a specific organism, ecosystem or region can provide relatively quick assessments of the key stressors, as well as extreme events, which may combine to increase the risk to species and ecosystems. While these mapping activities are potentially very valuable for management and planning, the biological datasets required to do this style of mapping currently do not exist for all regions, species, and life stage. A pan-Arctic scale collaboration for monitoring and mapping, such as an extension to projects like the Arctic Marine Biodiversity Monitoring Network or through the Arctic Council working groups (e.g., Conservation of Arctic Flora and Fauna), would be needed to fill these knowledge gaps.

SOCIETAL RELEVANCE

Integrated ocean management has been identified as the future methodology to achieving sustainable and resilient marine ecosystems by managing the progress of economic development whilst minimizing environmental impact (Winther et al., 2020). The research priorities proposed here would strengthen the data system needed for ocean policy and progress in governance of marine areas. There are two reasons why now is a crucial time for these ideas to be actioned: (1) the Arctic nation states signed the "Agreement to prevent unregulated high seas fisheries in the Central Arctic ocean" in 2018, providing an opportunity for information and knowledge to be collated before fishing grounds made newly available by ice loss are exploited; and (2) the United Nations Decade of Ocean Science for Sustainable Development from 2021 to 2030 aims to bring together scientists, policy makers, managers, and service users to ensure that ocean science delivers greater benefits for both the ocean ecosystem and society. The regional diversity of the Arctic Ocean does not fit any political boundaries, and adaptation strategies based on an Arctic wide collaboration are more likely to be successful than those based on individual country management. The United Nations Ocean Decade of Ocean Science for Sustainable Development provides an unrivaled opportunity for collaboration, action, and progress among Arctic countries toward ensuring sustainable and healthy fish stocks. Retrieving surface carbonate chemistry data derived from satellite products, together with other, more readily available satellite products provides a wholistic tool for helping to fill the knowledge gaps on the spatial and temporal scales necessary for end-user stakeholders. To implement climate-smart management and adaptation practices, including, for example, establishing marine protected areas, no-take zones, or catch-limits, OA needs to be taken into consideration alongside the other climate stressors.

CONCLUSION AND RECOMMENDATIONS

Here we highlighted three key points that the current literature is lacking: (1) studies on Arctic fish species; (2) understanding of the impact of multiple stressors; and (3) environmental data on appropriate temporal and spatial resolution to understand what environmental conditions species already experience. We propose that satellite and remotely sensed data can play a key role in filling these gaps as new technologies and developments take shape. To that end we propose a series of recommendations for moving these technologies forward alongside field, laboratory, and modeling research.

- Use Arctic specific satellite reprocessed datasets to develop Arctic-specific algorithms to monitor carbonate chemistry
- Produce synoptic scale datasets for carbonate chemistry, alongside temperature, ice cover, and ocean color products from satellites, to provide a multi-stressor view
- Improve biological observations of key fish species
- Encourage research into all stages of the life cycle of key fish species
- Continue to develop real-life variability and multi-stressors into experiments.

AUTHOR CONTRIBUTIONS

HG conducted the literature search, performed the analysis, and wrote the manuscript. HG, HF, JS, PL, and RB guided the work and contributed to writing. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.635797/full#supplementary-material

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Relationships Matter: Assessing the Impacts of a Marine Protected Area on Human Wellbeing and Relational Values in Southern Tanzania

Dana Baker[1], Grant Murray[1], Jackson Kaijage[2], Arielle Levine[3], David Gill[1] and Enock Makupa[4]*

[1] Duke University Marine Laboratory, Nicholas School of the Environment, Duke University, Beaufort, NC, United States, [2] Department of Social Studies, Stella Maris Mtwara University College, Mtwara, Tanzania, [3] Department of Geography, San Diego State University, San Diego, CA, United States, [4] Department of Geography and Environmental Studies, University of Dodoma, Dodoma, Tanzania

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Department for Environment, Food and Rural Affairs, United Kingdom

***Correspondence:**
Dana Baker
Dana.m.Baker@Duke.edu

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The push to meet global marine conservation targets has significantly increased the scope and scale of marine protected areas (MPAs) worldwide. While the benefits derived from MPA establishment are often optimistically framed as a "win-win" for both marine biodiversity and for the wellbeing of coastal peoples, this assumption is challenged for several reasons, including the fact that current science and practice frequently fails to account for the full impact of MPAs on human wellbeing. This context poses a danger that the context specific, place based aspects of wellbeing, like relations to others and the marine environment, will not be accounted for, examined, or reported in evaluation and decision-making processes. To address this challenge, this research investigates how MPA implementation can change and challenge the relational wellbeing and relational values of small-scale fishers (SSFs) living in Mnazi Bay-Ruvuma Estuary Marine Park, Tanzania. Fieldwork occurred over 2019–2020 and used qualitative data collection methods, including: 140 semi-structured interviews, document analysis, and observation. Results highlight a dynamic interaction between the MPA and SSFs relational wellbeing, including how relational values inform everyday fishing practices, cultural and place identities, as well as interactions with others and connections to the marine environment. Top-down approaches used in MPA development worked against key relational values, including social cohesion, reciprocity, place, agency and self-determination to dismantle and disrupt the practices SSFs viewed as fundamental to their livelihood and collective wellbeing. Our findings serve as a starting point to better recognize the context specific factors that underlie relational wellbeing and give insight into how relational values shape social-ecological complexity within coastal communities. The paper highlights how the international marine conservation community can better account for and foster relational wellbeing and relational values to achieve the goals of both human wellbeing and marine biodiversity conservation.

Keywords: human wellbeing, relational values, marine protected area, small-scale fishers, conservation

INTRODUCTION

The push to meet global marine conservation targets has significantly increased the scope and scale of marine protected areas (MPAs) worldwide (Jones et al., 2013; UNEP-WCMC et al., 2018; Ban et al., 2019). While the benefits derived from MPA establishment are often optimistically framed as a "win-win" for both marine ecosystem health and for the wellbeing of coastal peoples, this assumption is challenged for a number of reasons, including the fact that current assessments of MPA outcomes frequently fail to account for the full impact of MPAs on human wellbeing (Spalding et al., 2016; Agrawal et al., 2020; Waldron et al., 2020). Instead, researchers often focus on a few easily quantifiable indicators in the economic and material domains, such as household income or catch per unit effort (Ban et al., 2019; Rasheed, 2020). This situation poses a danger that context-specific, place-based aspects of wellbeing, such as social relations and connections to the marine environment, will remain unaccounted for within decision-making processes because they are neither examined, nor reported (Sterling et al., 2020).

A rich literature exists across the social sciences on how to measure and understand human wellbeing using an array of approaches and frameworks deployed at different scales (Gasper, 2007; Gough and Mcgregor, 2007; White, 2010; Breslow et al., 2016; Johnson and Acott, 2018). While there is no unified definition of human wellbeing, it is generally agreed that it consists of at least three mutually reinforcing and co-constituted material, subjective, and relational dimensions (Ransome, 2010; Coulthard, 2012; Leisher et al., 2013; Beauchamp et al., 2018). In this article, we ascribe to McGregor's definition of wellbeing that describes it as "a state of being with others and the natural environment where human needs are met, where one can act meaningfully to pursue one's goals and where one enjoys a satisfactory quality of life" (McGregor, 2008, p. 1). Within this view, wellbeing is described as a state, or condition that is fundamentally tied to (among other things) healthy and productive relationships with the human and non-human components of the social-ecological system and that is constructed through socially and culturally dynamic processes (Sen and Anand, 1997; Deneulin and McGregor, 2010; McGregor and Summer, 2010; White, 2010; Atkinson and Joyce, 2011; Chan et al., 2016). Accordingly, in this article, we argue that the relational dimension of wellbeing can be defined as a dynamic condition that emerges from relationships themselves, the qualities of those relationships, as well as the (held) values associated with each relationship.

In thinking about the continuous construction of one's relational wellbeing, we also draw on insights from the emerging literature on relational values to express the nature and qualities of key relationships that are constitutive of "the good life" (Jax et al., 2018). The concept of relational values encompasses a range of values fundamental to relational wellbeing and can be described as the "preferences, principles, and virtues associated with relationships both interpersonal and as articulated by policies and social norms" (Chan et al., 2016; Himes and Muraca, 2018; Jax et al., 2018; Stålhammar and Thorén, 2019). When the concept is tied to process-oriented, context specific approaches to understanding relational wellbeing, relational values become rooted in place and can be employed to describe the diversity and qualities of relationships that underlie one's wellbeing (Caillon et al., 2017; Muradian and Pascual, 2018; Stenseke, 2018; Skubel and Shriver-Rice, 2019). Relational values can, at least in part, be seen through the practices and actions taken to construct, secure and reinforce one's state of wellbeing. We refer to these practices and actions as "expressions." This includes how people and collectives make choices, behave, relate, and interact with others and the environment (De Vos et al., 2018; Stenseke, 2018; West et al., 2018; Gould and Pai, 2019). In this article, we primarily focus on the contribution and dynamics of social relationships to human wellbeing.

Social science research in fisheries has recognized a diversity of relationship types and qualities can influence a person's wellbeing and fishing behavior, for example, relationships of obligation, support, dependency, reciprocity, or exploitation (Coulthard et al., 2011; Coulthard, 2012; Chan et al., 2016; Klain et al., 2017; Johnson and Acott, 2018). Within the context of marine and coastal communities, one's relational wellbeing is also influenced by the interactions among individuals and families, fish buyers, boat crews, relevant government authorities, and other international actors. These social relationships are shaped by other factors such as age, wealth, gear type, ownership structures, patron-client ties, and fishing capacity (Walley, 2010; Jadhav, 2018). As such, the range relational values and the ways they are expressed varies across stakeholders, scales, and through time (De Vos et al., 2018).

For example, fishing is frequently valued for the sense of belonging and social cohesion it encourages – two factors seen as fundamental to the construction of one's wellbeing (Fearon et al., 2009; Leisher et al., 2013; Ishihara, 2018). Practices that help reinforce social cohesion may include, for example, teaching children to fish using the same techniques used by their ancestors. In turn, this can foster processes of learning and knowledge exchange, intergenerational interactions, as well as the transmission of local ecological knowledge, which contribute to social cohesion and to one's place identity. Similarly, fishing can be valued by a community by promoting conformity to social norms associated with maintaining key relationships, such as the norms expressed through reciprocal practices, such as non-monetary exchanges, like gift-giving and the sharing of (sea)food (Song et al., 2013). Fishers often follow rules based on reciprocity, an important social response in contexts of uncertainty, to gain access to fishing grounds and the benefits associated with participating in the fishery. In turn, this can strengthen social relations, social cohesion and kinship bonds– fundamental aspects of relational wellbeing (Crona et al., 2010; Poe et al., 2014; Idrobo, 2018).

Similarly, both agency and the right to self-determination have long been shown to be central to the construction of human wellbeing and are particularly important to relational wellbeing (Sen, 2007; Deneulin and McGregor, 2010; Breslow et al., 2017; Quimby and Levine, 2018; Sheremata, 2018). The importance of self-determination can be expressed as a relational value through the lens of governance and decision-making (Sheremata, 2018). In small-scale fishing communities, creating and maintaining the opportunities for fishers to speak and responsive governance systems that listen, learn, and respond to these voices support and reflect the value of self-determination by promoting feelings of agency in decision-making processes (Ribot and Peluso, 2003). In turn, meaningful participation contributes to one's relational wellbeing by enhancing perceptions of empowerment.

MPAs, like other conservation interventions, can change and challenge social relations and connections to the marine environment by applying new decision-making processes and rules of access, through the distribution of costs and benefits, or by prioritizing scientific knowledge over local ecological knowledge (Woodhouse et al., 2015). Such processes can interfere with the practices and activities (expressions) that support relational values and ultimately work to undermine relational wellbeing. This context can foster negative feelings toward marine conservation and can ultimately lead to the failure of the MPA. Despite the importance of relationships to human wellbeing, however, they are rarely accounted for in marine conservation interventions (Breslow et al., 2016; Hicks et al., 2016).

To begin to illustrate the importance of this gap, this research examines SSF communities living in Mnazi Bay-Ruvuma Estuary Marine Park (MBREMP), in southern Tanzania. Tanzania is home to nearly 55 million people and offers a good location to study the relationships between wellbeing and MPAs for several reasons. While the link between biodiversity conservation and poverty are complex and debated, it is widely agreed that poor and marginalized groups are highly dependent on the quality of, and access to, their surrounding environment to secure key aspects of their wellbeing and livelihood (Roe et al., 2013; Brockington and Wilkie, 2015). The World Bank estimates that roughly half of Tanzanians live at, or below, the poverty line of $1.90 USD per person per day (World Bank, 2017). Additionally, the specific study site of MBREMP is located in the region of Mtwara, which is historically one of the more marginalized and impoverished regions of Tanzania, with rural communities having a high dependency on their surrounding natural resources (Liebenow, 1971; Malleret and Simbula, 2004; Mangora et al., 2014; Raycraft, 2016).

Tanzania also offers a good location to study the impacts of MPAs on relational wellbeing because of its lengthy history of conservation and development interventions. Tanzania has approximately 41% of its terrestrial and marine environments under some form of protection (UNEP-WCMC, 2021). **Figure 1** illustrates the geographic extent of protected areas in Tanzania, as well as MBREMP's location in the southern region of Mtwara. It also has an extensive coastal and marine environment recognized as one of the most biodiverse and "pristine" regions in the Western Indian Ocean, which has long been targeted by international actors for marine conservation programs (Mangora et al., 2012). Likewise, fishing in Tanzania is an essential livelihood activity, generating food and income, and plays an important role in social relations and cultural identity of coastal communities (Katikiro et al., 2013).

In this context, this paper draws on the concept of relational values to examine the impacts of MPA implementation on SSFs relational wellbeing. The research objectives of this paper are to: (1) Identify and describe the expression of key relational values and how each relates to the construction of SSF's relational wellbeing; and (2) Identify and describe the primary interactions between, and impacts of, MBREMP on SSFs' relational wellbeing and relational values. Our results highlight the importance of social relations to human wellbeing, the primary drivers of fishing behavior, and the contextual factors that influence the acceptance of, or resistance toward the MPA.

MATERIALS AND METHODS

Study Site Description

MBREMP is located along the border with Mozambique in the southern district of Mtwara (Liebenow, 1971; Raycraft, 2016). Interest in forming the MPA arose from a series of meetings moderated by natural resource managers, conservation scientists, and development practitioners, that took place in the regional capital of Mtwara between 1999 and 2004. In 2000, after a year of discussion, an agreement known as the "Mtwara Declaration" was approved between the regional and national level governments to formally established MBREMP (Katikiro et al., 2015). Participatory approaches to conservation planning were followed, as mandated by Tanzania's Marine Parks and Reserves Act of 1994 (United Republic of Tanzania, 1994). The project was implemented by the International Union for the Conservation of Nature (IUCN) Eastern Africa Program, with joint funding from the Global Environment Facility (GEF)/UNDP and the Fonds Francais pour l'Environnement Mondial (GEF, 2000). The IUCN ran the project for 54 months, after which MPA management was handed over to Tanzania's Marine Parks and Reserves Unit (Gawler and Muhando, 2004; Tortell and Ngatunga, 2007). Shortly after park establishment, a socio-economic baseline and an assessment of the occupational structure of MBREMP communities was completed by the IUCN Eastern Africa Program (Malleret, 2004; Malleret and Simbula, 2004). Together, the assessments found that MPA communities depended heavily on coastal resources and identified poverty as a primary threat to the biodiversity and productivity of marine resources (GEF, 2000; Gawler and Muhando, 2004; Malleret and Simbula, 2004; MPRU, 2011; Mwansasu, 2016).

MBREMP covers 650 square kilometers and consists of marine, coastal, and terrestrial habitats. The area was identified as a priority area for global marine biodiversity in 1995 due to its unique location between the South Equatorial and the Mozambique Currents, an area that has produced some of the highest diversity of hard and soft corals in the Western Indian Ocean (Kelleher et al., 1995; Ngowo, 2003). The park is registered with BirdLife International (2021) as an Important Bird Area (no. 15) and is zoned as a multiple-use marine park (MPRU, 2011). In Tanzania, MPAs encompass expansive terrestrial areas that surround coastal villages. The area inside the boundaries is sub-divided into various user zones including: core zones, within which all extractive activities are prohibited; specified-use zones where extractive activities are regulated at an intermediate level; and general-use zones where marine park residents are given priority to access resources (MPRU, 2011, p. 49). Additionally, an MPA buffer zone extends 1 kilometer from the park boundary, except along the border with

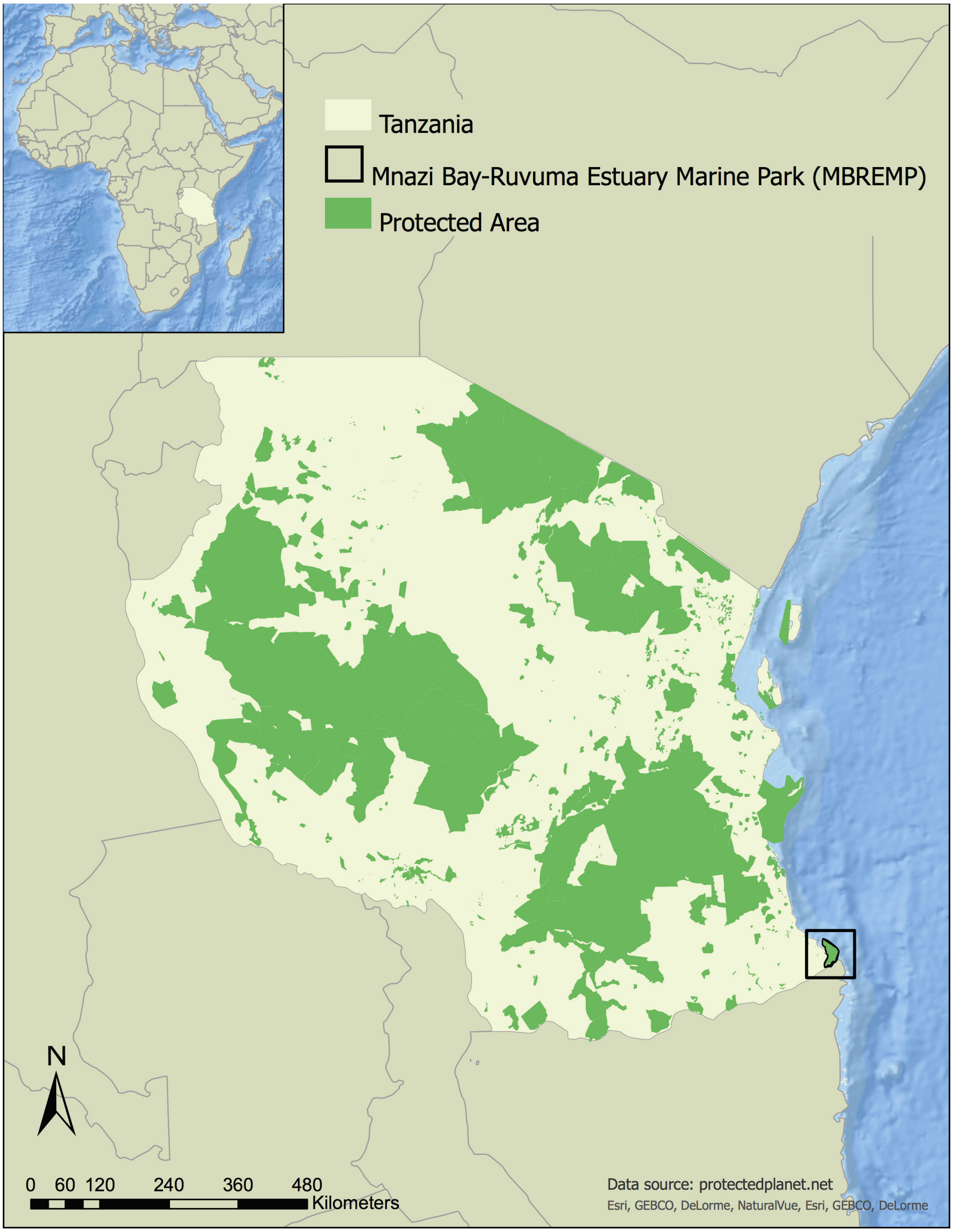

FIGURE 1 | Map of Tanzania, located in East Africa, illustrating geographic extent of protected area coverage across the country. MBREMP is located along the southern border with Mozambique. Data obtained from the World Database on Protected Areas (UNEP-WCMC et al., 2018).

Mozambique. The original aims of the zoning scheme were to provide a clear framework for monitoring and enforcement, a geographical basis for evaluation, and a means of safeguarding traditional fishing grounds (MPRU, 2011).

In initial stages of MPA development, an estimated 30,000 people, in 11 villages, were identified to be living in the catchment area (Malleret and Simbula, 2004). Currently, MBREMP has over 40,000 people living in 17 villages inside its boundaries

(Katikiro et al., 2015). The vast majority of villagers are Muslim and identify as Makonde, a tribal affiliation specific to southeastern Tanzania and northern Mozambique (Raycraft, 2019c). Most speak both Kiswahili and their tribal language of Kimakonde and engage in livelihoods that involve combinations of subsistence farming, cashew farming, fishing of coral reef fish, and the shoreline harvesting of crustaceans (Malleret and Simbula, 2004; Mangora et al., 2014). In MBREMP, fishing is largely artisanal and cash-oriented, taking place in nearshore waters at depths of less than 40 meters, from dug-out canoes (*mitumbwi)* and other dhow-type sailing boats (Jacquet et al., 2010). Several park villages also have access to larger dhow boats, outfitted with outboard motors, and are thus able to target larger pelagic species outside Mnazi Bay (Katikiro et al., 2013). The vast majority of SSFs use a variety of gear types including handlines, different sized seine nets, basket traps, spears, long-lines, cast nets, and scoop nets. However, despite their livelihood importance to coastal communities, inshore fisheries are found to be overexploited (Guard and Mgaya, 2002; Tobey and Torell, 2006; Silas et al., 2020). This context makes the sustainable utilization of marine fisheries, and the successful implementation of MPAs, critical to reduce the vulnerability of coastal communities.

Data Collection

Two three-week scoping activities were conducted in January and August of 2018 to engage with key stakeholders involved in MPA management and to better understand priority issues facing MBREMP. Prior to starting data collection, 5 village wide community meetings were held in select village sites, and several round-table discussions with park authorities, village leaders, and district and regional officials facilitated the development of this research and determined final interview locations. Village sites were selected based on the presence of a fish landing site, the determination of fishing as primary livelihood for SSFs within the village, and sites located in a variety of park habitats (i.e., beach, mangrove, and riverine), and other logistical factors. This preparatory phase served to build strong personal relationships with key actors and allowed for a deeper understanding of the contextual factors of MPA implementation and SSF wellbeing.

Data collection occurred over two 3-month field seasons in 2019–2020 spanning both the North East Monsoon (*kasikazi)* and the South East Monsoon (*kusi)* seasons to coincide with periods of higher and lower fishing activities and the arrival and departure of migrant fishers. The primary data collection method involved semi-structured interviews using protocols adapted to the specific coastal livelihoods of MBREMP residents. Protocols were also influenced by existing wellbeing approaches including the sustainable livelihoods approach (Chambers and Conway, 1992; Scoones, 1998), the WeD/3-D Wellbeing framework (Gasper, 2007; McGregor and Summer, 2010), and the methods handbook for the "Social Assessment for Protected and Conserved Areas" (Franks and Booker, 2018; Franks et al., 2018). Interview questions focused on defining one's wellbeing and the "good life," social-environmental relations, and understanding SSFs perspectives toward conservation programming and the marine park and perceptions of MPA impacts.

Interview participants were purposefully selected (Maxwell, 2013) to collect perspectives from a range of individuals within selected fishing communities. Respondents were identified in direct collaboration with village leaders, with selection based on factors such as fishing gear-type used, livelihood dependence on fishing and marine resources, gender, and age. This process worked to ensure local-level permission to speak to individuals was granted and to find participants who primarily identified SSFs that fish inside marine park boundaries, using a variety of fishing gear-types. Although the act of fishing is culturally constructed as a male activity and women do not self-identify as fishers, women were included in SSF interviews due to the importance of female dominated gleaning practices across MBREMP's intertidal zone. The village leader of each village, as well as key members of the Village Liaison Committee (VLC) and the District Fisheries Officer (DFO) were also interviewed. VLCs are comprised of village members who, in theory, serve as the primary liaison between each park-associated village and MPA management (Katikiro et al., 2017). DFOs are government officers employed at the district level to register fishing vessels, issue fishing licenses, collect revenue, and to record fish landing data. All interviews were conducted in Kiswahili and lasted from 30 to 150 min. Interviews were conducted by the lead author, who is proficient in Kiswahili, and a Tanzanian research assistant, who was hired from a local university and has significant training in social science data collection techniques. A total of 140 semi-structured interviews were conducted with SSFs located in 5 MBREMP villages, including 115 male and 25 female fishers, aged approximately 20–90 years old. To protect the anonymity of respondents, we have withheld specific interview locations.

A key realization from early fieldwork was the need to translate the notion of wellbeing into the local cultural and language context, highlighting how aspects of one's identity and socio-environmental relations are deeply embedded within language (Coscieme et al., 2020). When translating both language and across cultural contexts important nuances can be lost and distorted. For example, the direct translation of the term wellbeing in Kiswahili is "*ustawi*," which has a slightly different connotation in Kiswahili as compared to the English understanding of "wellbeing." In Kiswahili "*ustawi*" is often used in the context of state welfare programs, such as food, aid distribution, education, and infrastructure, and connotes a narrow, more formal view of wellbeing focused on material qualities. The terms "*maisha mazuri*" (the good life) and "*maisha magumu*" (the hard life), on the other hand, were found to suggest a more holistic and balanced conception of one's life and core values beyond material assets. To understand the differentiated impacts of the MPA on SSF wellbeing we therefore focused on understanding how fishers construct and imagine "*maisha mazuri*" and "*maisha magumu.*" To accurately reflect the insight and nuance language provides, we frequently draw on Kiswahili terms to articulate SSF's worldview as close as possible to their perspective.

Data analysis included transcribing, translating, and coding each interview in QRS NVivo 12, a qualitative analysis software. Interview transcription and translation were completed by a professional transcription service and verified by the first author.

Coding used a combination of emergent codes, as well as categories drawn from the relevant literature on wellbeing, relational values, and conservation and development studies. This process organized data into key categories by identifying context specific attributes of wellbeing, associated relational values, and examples of how different relational values were expressed. Key categories were next organized based on how they interacted with and were impacted by the MPA.

RESULTS

In the following two sections, we describe five key relational value categories that emerged as important for SSFs in MBREMP, detailing how each can be expressed and related to the construction of one's relational wellbeing. Next, we describe the primary interactions between, and impacts of, MPA policy and actions on SSFs' relational wellbeing and relational values. Our results highlight the importance of social relations to human wellbeing, the primary drivers of fishing behavior, and the contextual factors that influence the acceptance of, or resistance toward the MPA.

Understanding SSF's Relational Wellbeing

There is a common cultural identity among Makonde fishers, rooted in a shared dependence on ocean resources and a desire to maintain autonomy in the everyday choices they make regarding natural resource utilization. SSFs often described daily life using the term *uwezo*, which translates to one's ability, strength, and capacity. SSFs, however, also use it in a broader sense to describe their community as having the capacity to resist when they believe their autonomy is being interfered with. The desire for autonomy and agency becomes apparent and is expressed as a relational value when they narrate the region's collective history, extensive relational networks, and their ongoing struggle to maintain the customary right to resource access and occupancy. Elders, for example, often described this history by using the idiom "*hii ni bahari yetu na uwanja wetu*" [this is our sea and our fishing grounds]. This particular phrase alludes to how people define, and legitimize, resource access rights through historical experience, including their long occupation of the area and their continued use of ocean resources. This phrase was often followed with detailed accounts of complex trading and marriage networks that connected Makonde fishers to inland areas as far as the Democratic Republic of the Congo and Malawi and to areas reached through ocean routes leading to Madagascar and Oman. The retelling of this history suggests how MBREMP's fishing communities have never existed in isolation and that many fishers intimately understand the importance of building, and maintaining, productive and diverse social relations across contexts and scales.

Enmeshed within these historical narratives, SSF often discussed the impact of conservation programs in other parts of Tanzania, frequently referring to the experience of fishers in Mafia Island Marine Park, where the ocean now exists to benefit the "*wazungu tu*" [tourists and/or foreigners only]. Yet, SSFs made clear they did not necessarily, or inherently, reject state intervention, or the idea of an MPA in and of itself. Rather, they objected the processes used to make decisions on their behalf, which had direct consequences for how they maintained their livelihoods, transforming the practices and expressions of key relational values they believed were fundamental to their survival and a desire to retain a sense of autonomy in how social relationships are arranged and the processes of decision-making. Elders often described inclusive and collaborative decision-making processes that included lengthy discussions where each community member was given the opportunity to express their opinion. In turn, these processes reinforced the relational value of social cohesion by directly shaping social relationships within the community.

Coastal communities in MBREMP have a multi-generational interactions with marine resources, where fishing gear, fishing grounds, and local knowledge are passed down within and among family clans and communities. The transmission of knowledge often takes place in everyday lived experience in close relation with others. For example, in the intertidal zone, women glean a variety of small fish and other invertebrates to sell, eat, or to dry and store for later use. They often glean with their children, friends, and family and referred to these activities as a way of life and as a way to learn about themselves and others. As one female gleaner expressed,

> *I glean because it is what my mother and grandmother taught me. When I was a child, my mother would send my grandmother and I to the ocean to catch crabs, small fish, and sea cucumber. I remember I was afraid of the ocean back then, but working alongside my grandmother, I stopped crying and gained the strength to quickly fill our pots for fish stew! As my grandmother grew weaker, she no longer went with me to gather in the ocean, but I'd go along with friends and show them what my grandmother taught me, so they too could learn how to provide for themselves, their brothers and their sisters. Even today, I take my children when I go gather, in this way they will learn to not be afraid and will build the strength to survive.*

In this context, the intertidal zone served as a key space to reinforce shared cultural identities and practices important to maintain one's relational wellbeing. The woman's grandmother taught her important life lessons through the practice of gleaning, including how to be self-sufficient and resilient when faced with challenges and changing circumstances. The expression of relational values within the intertidal zone included various forms of learning and knowledge exchange, environmental stewardship, place identity, and kinship.

Clearly, many daily practices undertaken by SSFs maintain basic qualities of life, such as food, shelter, medicine, and access to education. As one fisher explained,

> *Fishing is what drives my life. So, for me, the good life is to own the right fishing equipment... Money is scarce and you only make enough for today. So, we must go back to the ocean to eat tomorrow. And because of this, I have not reached the good life.*

In one sense, this fisher is noting that his household's survival is tightly tied to his fishing gear, which represents some of the most important material assets for his household. However, it also became clear that the good life is also accomplished by SSFs pursuing livelihood strategies that included culturally embedded forms of sharing and reciprocal exchange. Fishers often asserted "*wavuvi hawanyimani*." This phrase translates to "fishers do not deny one another" alluding to another wide-spread belief that the ocean and marine resources are to be shared by all and used for collective economic development. SSFs believe it is unjust to deny another the right to access and to benefit from marine resources. To deny this right, as one village leader explained, is to be "complicit in the oppression of their own people." This widespread claim to ocean space illustrates the importance of maintaining the collective right to access marine resources through practices that maintain social cohesion and reduce conflict.

Reciprocal and cooperative relations for SSFs in MBREMP create a safety-net beyond village boundaries and many view expressions of reciprocity as fundamental to their survival. This fisher explained,

> *If you create conflict with other fishers, you'll only be killing yourself. We all fish in the same ocean and we all need help now and again. We cannot be successful every day, so we must share our fuel, and sometimes our catch, so others can get home for dinner. I do this even if it is our first-time meeting. But, I know if I need help tomorrow, I can call on my fellow fishers to help me.*

For this fisher, reciprocal relations extend into ocean spaces and to other fishers he does not personally know, illustrating the importance of building robust social networks for SSFs in MBREMP. He shares his fuel because he knows that 1 day in the future he will likely need the help of others. The importance of reciprocal relations for SSFs in MBREMP is also demonstrated in practices (expressions) like the redistribution of food, livelihood resources, and labor based on need. If someone is unable to fish for reasons such as age, sickness, or other household issues, *sadaka* [gifts of fish and food] are given under the premise that the giver will one day be in a similar position of need. Literally translating to religious offering or alms, *sadaka* symbolically represents a generalized form of reciprocity that fosters respect and trust within and among fishing communities. For example, bringing *sadaka* to an elder's home serves as a means of ensuring they have food, to check on vulnerable individuals, and nurtures relationships between different generations within a community. Likewise, *sadaka* is also used as a form of hospitality and is often extended to guests and newcomers in a village, such as migrant fishers. Migrant fishers are often provided with basic food staples, like *ugali* [a thick cornmeal porridge], and are often introduced to the rest of the village, extending social relations to other coastal fishing villages, which can be tapped in times of need.

MPA Impact on SSF Relational Wellbeing

When discussing the impact of the MPA on their wellbeing, many SSFs simply express "*wameishatudhulumu na wanatudhulumu*" [what they have done is unjust and they have wronged us]. Over time, residents have come to see the MPA and, more generally, the notion of conservation, as a direct threat to their wellbeing and to their way of life. In 2013, this frustration culminated in a handful of residents using dynamite to destroy a newly built MPA gatehouse, that had been funded by the World Wildlife Fund (see Raycraft, 2019a, p. 12).

On the other hand, many described hopeful memories when the marine park was first introduced, that over time turned into frustration and resentment. As this fisher described:

> *Initially, when the marine park came their intentions pleased us. They said, 'We will improve your lives. We will give you working tools. We will educate you.' It was apparent that the Marine Park was supposed to co-work with the community, but after they arrived, they wanted to make decisions without discussion, so we refused to be involved any more. Since then, they have come just once to give us education, but our education is different from theirs. Theirs comes from books, ours comes from a point of knowing each other.*

While participatory and socially inclusive approaches to marine park development were used in theory, such processes did not facilitate positive social relations, nor did they facilitate the long-term participation of residents in the co-management of the MPA. While there are clear budgetary limitations that explain why MBREMP staff have come "just once" to provide education, the fact that many SSFs felt decisions were made without discussion reflects the exclusionary nature of decision-making used in initial stages of MPA development. The fundamental importance of developing good social relations to SSFs is clear when this fisher described his education as coming "from a point of knowing each other," as opposed to the implied education of MPA authorities, which comes "from books."

When residents discuss the original conditions of participation they agreed to at the outset, they often described the MPA as a form of social contract and in reciprocal terms. Residents agreed to MPA implementation in exchange for the provision of social services and local-level development. As this village chairman elaborated,

> *We agreed to something we should not have. Prohibiting traditional fishing without an alternative is a very serious issue. Today, we see no reason of helping them in their work, because they have not helped us, they have not kept their word and we have disengaged, so they can operate by themselves. They should have shown us respect.*

The reference to the fact that "they have not helped us" refers to the initial claim that park residents would share in the economic and other benefits of the MPA. Despite early emphases on eco-tourism related development, poverty alleviation and benefit sharing, infrastructure inside the park remains virtually non-existent. This quote further reflects the importance of reciprocal exchange as a shared value for park communities—they will follow the restrictions and adhere to the rules of conservation— if marine park management fulfill their promises of direct benefits in return.

Many fishers discussed how they felt forced and/or were paid to collaborate in planning discussions and remained critical that open meetings were not held in their villages—a key process used to maintain social cohesion and to reduce conflict at the village level. Instead, meetings only included a few select individuals from each village. Despite the intentions of donor agencies involved in project development, the methods used to facilitate participation were perceived to have circumvented local norms of decision-making and participation. This fueled feelings of mistrust and disrespect between park communities and MPA authorities.

Likewise, the MPA implementation process circumvented local norms of decision-making with the creation and execution of Village Liaison Committees (VLCs). In theory, VLCs were to serve as the primary liaison for communication between marine park authorities and every MPA village. VLCs were intended to be the lowest level of the governance structure and are comprised of select villagers, who are directed to undertake MPA patrols, resource permitting, and to help with the overall protection of marine resources. In practice, VLCs parallel and overlap village-level elected authorities, which created confusion about who reports to whom. This fisher elaborated on the situation,

> *What they [the marine park managers] want is vastly different from what we want. They [the marine park] only make life more difficult, so if someone does something that we feel is against the rules, we just deal with it on our own. Our village has elected officials who were born here and they will go to the person who acted wrongly and ask their reason. They will talk to them to understand their circumstance and personal conditions. Afterwards, there will be a village meeting to discuss the situation, so we can offer help and find a solution. In this way, we know they will not break the rules again.*

This situation illustrates a clear gap between the ideal of collaborative management and reality, as well as how the MPA challenges local level power dynamics and the right to self-determination. When discussing VLCs, residents expressed a shared sentiment that they do not report illegal resource use and non-compliance because, from their perspective, the punishments given by the MPA are harsh and morally unjustified.

The increasing number of conservation regulations has further impacted SSF's right to self-determination and agency in decision-making processes. For example, when the marine park was formed, all of the land inside its boundary was reclassified from "village land" to "reserve land" in accordance with Tanzania's Marine Parks and Reserves Act of 1994 (Section 16). As such, all new development and land allocations within the park must be reported to park officials in writing 30 days prior (MPRU, 2011). The reallocation of property rights, however, has challenged and undermined customary occupancy rights. This elder explains the emotional impact of the reallocation of land,

> *We are told this area now belongs to the marine park. I do not have much except for a small piece of land and my canoe. If I want to do anything on my land, or even if I wanted to sell it, I have to ask permission. If you fail to ask permission, park authorities will claim you are invading the marine park. They let us live here but only under certain conditions. . .*

This quote illustrates how the reallocation of land rights produced new rules of authority and control over people's lives, undoing one's sense of power and autonomy. The impact of these new rules was exemplified when SSFs discussed their ongoing frustration with the MPA's zoning system. Most of Mnazi Bay is zoned as a specified use zone, with key fishing grounds designated as no-take core zones (see **Figure 2**). This made legally fishing inside the bay difficult and, without any form of demarcation between zones, has created lasting confusion and anger. This fisher explained,

> *If you look at the type of fishing we practice, we do not have the capacity to go into the deep sea. When the marine park tells us to fish in the deep water, it is similar to asking us to choose between life and death.*

Many SSFs feel the zoning scheme does not consider the everyday constraints of poverty, nor how difficult it is to adopt and learn to fish with new gear, in new fishing grounds, reflecting a disconnect between MPA design and everyday livelihood needs. Many SSFs cannot afford to lease, rent, or buy the bigger boats and engines needed to fish in the deeper waters outside the park's specified use zones. This context reveals another widespread sentiment held by many SSFs whereby the marine park values marine biodiversity over human life.

The impact of banning fine-net fishing on SSFs relational wellbeing is particularly evident when examining the role of female fishers in MBREMP and highlights the fundamental misalignment between MPA design and reality. When the marine park was formed, the use of fine-mesh nets was banned in the park's intertidal areas, as well as in core and specified use zones. Yet, the banning of fine-mesh nets also prohibited a customary female practice known locally as *kutanda*, whereby women use a fine mesh pull-net called a *tandillo* to harvest from shallow, intertidal environments. The practice was banned because it was viewed as destructive by conservation scientists who argued the practice captures juvenile fish. Yet, the decision to ban the practice reflects the top-down nature of decision-making used to develop MBREMP and worked against the villagers' right to self-determination to make decisions about resource use and access. The restrictions minimized women's economic mobility, their ability to contribute to their family's material needs, and interfered with their sense of self and identity as a provider of their household—all key components to relational wellbeing. However, the women interviewed for this research remained adamant they know the difference between a juvenile fish and a small adult fish. As such, they continue to widely, and often very visibly, engage in the practices of *kutanda* out of what they describe as both economic necessity and a moral right. As this woman explained,

> *They told us that our nets were banned and took them. How were we to feed our children? The marine park made this decision without involving us. This offended us. So, we came together, made new nets and we will continue as our mothers and grandmothers did.*

The intertidal zone is one of the only marine spaces women can access and thus the ban on fine-net fishing had cascading impacts on a woman's relational, material, and subjective wellbeing. Similarly, women no longer had reason, or access to a shared location in the intertidal zone to gather and reconnect, transforming how social relations and kinship bonds were constructed and maintained. Yet, this context also reflects a moral statement about SSFs' collective right to benefit from marine resources.

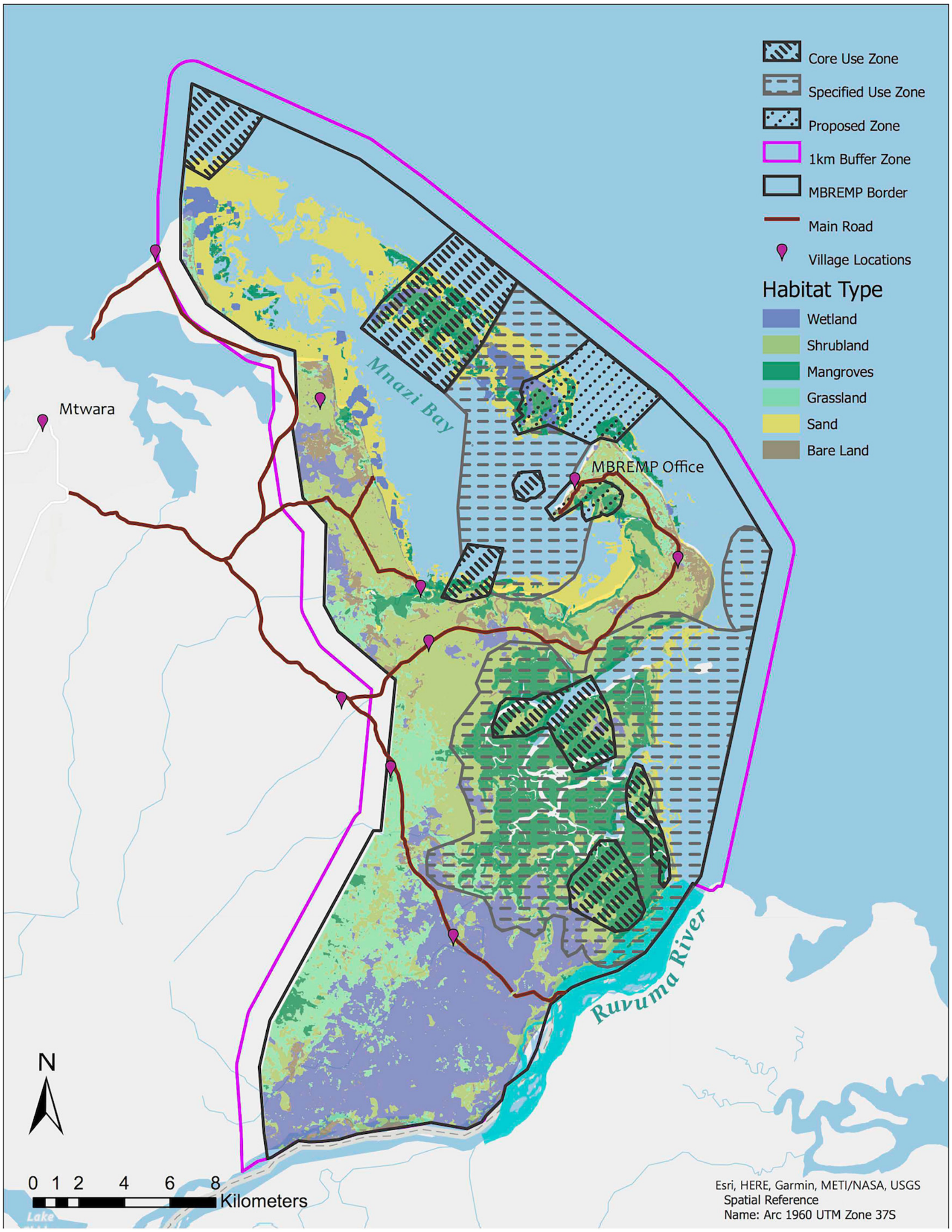

FIGURE 2 | Mnazi Bay Ruvuma-Estuary Marine Park study area map illustrating primary habitat types, primary village locations, location of the marine park offices, the regional capital of Mtwara, as well as the marine park's user zones. User zones were adapted from available park management plans (MPRU, 2011).

DISCUSSION

These findings resonate with other work showing how social and environmental relationships are critical to human wellbeing in the rural small-scale fisheries sector (Crona et al., 2010; Chan et al., 2016; Masterson et al., 2017; Sterling et al., 2020). In MBREMP, the multiple values SSFs associated with others and with marine spaces were described through rich narrative discussions that highlight the social history of the Makonde tribe. **Table 1** summarizes the key relational value categories

TABLE 1 | Key relational value categories that emerged as important to SSFs in MBREMP with associated practices and expressions.

Relational value category	Examples of expression
Freedom, agency and self-determination	Meaningful participation in decision-making practices; building responsive governance systems; expressed desire to maintain individual and collective autonomy.
Identity	Promotion of cultural practices, rules, norms, beliefs, and ceremonies; protection of important sites, monuments, or environments; learning and knowledge exchange.
Social cohesion	Practices that foster multi-generational interactions and social memory; acts of cooperation, like labor exchange and community workdays; participation, civic engagement and collaborative decision making to foster collective action and shared values; expressing shared visions of the future; avoiding and mitigating conflict through collaborative decision-making.
Place	Restricting, or promoting access to key spaces and/or resources; rules and norms regarding resource access, tenure, and occupancy; maintaining cultural resources and practices tied to specific environments, or species through engagement in rituals, or ceremonies; maintaining and promoting traditional, customary or informal resource management systems through continued use of marine environment.
Reciprocity	Practices of sharing and exchange of key livelihood resources, such as fish, (sea)food, and fuel; gift giving; labor exchange; community workdays; stewardship practices and care.

that emerged as important for fishers across MBREMP and examples of how each can be expressed. Social relationships and cultural identities associated with marine space spanned both generations and geographies and SSFs valued the marine environment for the relationships it produced and reinforced. Overwhelmingly, fishers indicated that the long-term benefits promised by the MPA for restricting access, such as fish spillover, alternative livelihood programs, and development, did not offset the cost of displacement or prohibition of fishing, replacing confiscated gear, or the social impacts associated with creating conflict within and between park communities. It was clear that the creation of MBREMP disrupted some of the practices and behaviors (expressions) associated with valued relationships, thus impacting the ways relational wellbeing was constructed and maintained. **Table 2** summarizes the primary interactions between specific MPA policies, or action and SSF's relational wellbeing and relational values. Failure to account for relational dimensions of wellbeing, including key relational values, risks exhasterbating poverty and hardship across park communities (Grantham et al., 2020) and need to be considered relation to other sociopolitical and biophysical variables (Mascia et al., 2010).

Findings also highlight the linkages between the literatures on relational values, relational wellbeing, and the wider literature on MPA governance. In MBREMP, SSF's relational wellbeing is built on a desire to retain a sense of autonomy in how social relationships are arranged and maintained. While MBREMP is theoretically founded on a decentralized, participatory model of conservation programming, there remains a clear gap between the rhetoric of participatory institutional design, often dictated by international agendas, and the realities of everyday implementation (Cooke and Kothari, 2001; Kamat, 2018; Raycraft, 2019b). The methods used in MBREMP development and the creation of VLCs directly challenged SSFs' right to

TABLE 2 | Primary interactions between, and impacts of, the MPA policy/action and small-scale fisher's relational wellbeing and relational values.

Policy of MPA	Community level impact	Disruption to relational wellbeing	Freedom and agency	Identity	Social cohesion	Place	Reciprocity
Top-down decision making and exclusionary processes of participation	Circumvention of local norms of participation and decision making	Fostered negative relations with MPA authorities	x		x		x
Village Liaison Committees (VLCs)	Introduction of alternative community governance structures	Took away agency in decision-making and local-level authority	x		x		
Reallocation of property rights: "village land" to "reserve land"	New rules of authority and loss of autonomy in land-use decision making	Transformed land and resource inheritance patterns and occupancy/tenure rights	x	x		x	
User zones, gear and area-based restrictions	Women no longer (legally) allowed to fish in inter-tidal zone	Disruption of multi-generational interactions and exchange of local ecological knowledge (LEK)	x	x	x	x	
		Disruption to self-reliance, identity, and sense of self	x	x			
	Men required to fish in deeper water	Fishing now requires larger crew, new gear and different market relations	x	x	x	x	x
	SSFs less independent and work for hire on boats	Reduced agency in decision-making and livelihoods	x				
		Loss of intergenerational interactions and transmission of LEK		x	x	x	
		Fishery transformed: communal/artisanal to cash-oriented		x	x	x	x

define their own needs and wants, which are central components to human wellbeing (Sen and Anand, 1997; Deneulin and McGregor, 2010). Likewise, many SSFs felt they were not able to meaningfully, or effectively participate in MPA processes and decision making, fueling feelings of disrespect. This context resulted in a long-standing conflict, resistance, and widespread non-compliance to conservation regulations that continue to have serious implications for MBREMP's success (Raycraft, 2020).

For SSFs in MBREMP, processes of knowledge exchange and learning were central to their relational wellbeing. Everyday fishing and gleaning practices enabled fishers to connect with others and their environment, fostered the transmission of local ecological knowledge, social cohesion, place and cultural identities. Yet, gear and area restrictions required SSFs to fish in new environments and to use new techniques, which required larger boats, larger crews, and new market relations. The processes of relearning new environments and new fishing techniques devalued SSFs lived, everyday experience and local ecological knowledge directly challenging their relational wellbeing (Brueckner-Irwin et al., 2019). The shared struggle of SSFs to maintain and reassert their customary right to resource access and occupancy has united fishers in a common cause against the MPA, as seen elsewhere (Sowman, 2011). Strong social cohesion has deterred SSFs from enforcing MPA rules, or reporting illegal resource use, so they can effectively minimize conflict within and among their communities. The widespread preference of SSFs to remain silent about illegal resource use and poaching in MBREMP is rooted in the fact that it's not socially beneficial to come forward to report illegal activities. Likewise, community members often showed empathy to those who were caught, fined, or punished by the MPA, often pooling financial and material resources to help an individual pay their fine and/or replace confiscated gear. In this sense, the relational lens helps illuminate why non-compliance and resistance can persist.

These findings also emphasize how SSFs' relational wellbeing is reinforced by and connected to the wellbeing of others—what we call collective wellbeing. The importance of collective wellbeing was also expressed through long-standing community practices where non-monetary benefits of reciprocal human and environmental relations outweigh financial and material incentives (Winthrop, 2014). Sharing of resources is common practice across sub-Saharan Africa and this supports other work on how relational values can strengthen social norms and informal institutions for mutual and collective benefit (Jones and Tobin, 2018). In MBREMP, expressions of reciprocity and reciprocal exchange weave through all aspects of life, extending across the seascape to include other fishers, middlemen, migrants, friends, and family. Reciprocal relationships are rooted in a number of values such as solidarity, trust and social cohesion, which are often valued over the individual accumulation of material wealth. In this context, fishers were motivated to maintain expansive relational networks because it secured robust safety-nets that could be utilized in times of hardship, or resource scarcity (Sterling et al., 2020). In a region of the world where the safety-net typically provided by the state is unreliable, social relations and relational networks increase livelihood security and the social resilience of coastal communities.

CONCLUSION

In MBREMP, SSFs' wellbeing is driven by more than the need to secure material resources—it is also driven by a need to fulfill one's obligations to others. Using a relational lens to characterize the impacts of MBREMP on relational wellbeing highlights the ways SSFs connect with others within their environment. It illuminates how social relationships are shaped by relational values, associated norms, and codes of conduct and how these in turn shape behaviors and perceptions of the MPA. In the case of MBREMP, the disruption of multiple relational values that SSF communities view as important has worked against the goals of both marine conservation and human wellbeing (Jentoft and Chuenpagdee, 2015). SSFs were not physically displaced by the MPA but their ability to maintain and pursue valued relationships and to access and benefit from key livelihood resources was critically undermined. SSFs have effectively been "displaced in place" by conservation policies, which have left many unable to meet their basic material needs (Cernea, 2006; Lubkemann, 2008; Raycraft, 2019a).

The particular relational values that emerged as important to SSFs in MBREMP may not be applicable in all contexts, to all MPA communities, or even to all SSFs in MBREMP. They do serve, however, as an important starting point to better recognize how contextual, place-based factors and relational values underlie human wellbeing, as well as how each dimension of wellbeing is co-constituted and inseparable. Likewise, our findings show the importance of using perceptions and lived experience to gain valuable insight into the social impacts, acceptance, and the legitimacy of the MPA (Bennett, 2016). Using a relational lens provided valuable insight into the importance of social relations to human wellbeing, the primary drivers of fishing behavior, and factors influencing perceptions of, and resistance to, the MPA.

The mainstream conceptualization of an MPA is that they exist to improve ecosystem health and services, thereby providing social benefits and driving support for the overarching goals of conservation (De Vos et al., 2018). Similarly, the marine conservation community is increasingly concerned with the use of monitoring and evaluation to support evidence based conservation and to improve conservation outcomes (Bennett, 2016). However, as our case shows, the failure to recognize the multiple values and lived experience that fishing communities hold can work against the goals of both marine conservation and human wellbeing. As such, we argue employing the concepts of relational wellbeing and relational values can guide international policy makers and MPA managers to meaningfully engage with local, place-based values and to better understand the diversity and valued qualities of social-environmental relations in marine environments (Sheremata, 2018; Stenseke, 2018; Gould and Pai, 2019). This conceptual bridging could be relevant in addressing a persistent tension between obtaining international targets for marine conservation and securing the rights of coastal communities (Armitage et al., 2012; Woodhouse et al., 2015). Attaining global biodiversity conservation will only be successful if MPAs support, and not compromise, the multiple aspects of human wellbeing of coastal communities (Brueckner-Irwin et al., 2019).

DATA AVAILABILITY STATEMENT

Derived data supporting the findings of this study are available upon request from the corresponding author. However, each feature layer produced for **Figure 1** and **Figure 2** are freely available for download here: https://dukeuniv.maps.arcgis.com/home/item.html?id=57cf6b3688fb447ba36d9e5499283750.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Duke University Institutional Review Board (IRB Protocol Number: 2020-0035). Interview participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

DB led all parts of this research including project conceptualization, data collection, data analysis and synthesis, and framing and writing of the manuscript. JK helped with the organization of fieldwork, data collection, as well as with the writing and editing of the final manuscript. All other authors including GM, AL, DG, and EM contributed to the project's development, as well as with the framing, writing, and editing of this manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.673045/full#supplementary-material

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How Can We Reduce the Overexploitation of the Mediterranean Resources?

*Fabio Fiorentino and Sergio Vitale**

Institute for Marine Biological Resources and Biotechnologies (IRBIM), National Research Council (CNR), Mazara del Vallo, Italy

Based on the characteristics of the fishing sector (multispecies and multi-gears) and the stock status of main resources (overfishing and overexploitation), some suggestions to improve the sustainability of demersal and small pelagic fisheries in the Mediterranean are proposed. In fisheries exploiting single or few species, such as small pelagics and deep-water red shrimps, the adoption of a management system based on catch quota approaches is suggested. In the case of mixed fisheries exploiting species with very different biological traits, it is proposed to reduce the fishing effort to a level corresponding to the lower range of the "pretty good yield" of the main target species while improving the status of the most sensitive associated species, adopting technical measures to mitigate fishing mortality. The feasibility of the proposed approaches is briefly discussed, taking into account the different levels of development of the Mediterranean countries.

Keywords: overfishing, demersal stocks, fisheries management, pelagic stocks, sustainable yield

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Athanassios C. Tsikliras, Aristotle University of Thessaloniki, Greece
Gorka Merino, Technology Center Expert in Marine and Food Innovation (AZTI), Spain
Stefanos G. Kavadas, Hellenic Centre for Marine Research (HCMR), Greece

Correspondence:
Sergio Vitale
sergio.vitale@cnr.it

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DESCRIPTION OF MEDITERRANEAN FISHERIES

Mediterranean fisheries are known for their strong multi-specificity and multi-gear features. Overall, fleets work mostly close to home ports, except for a few components of the fleet (trawlers, purse seiners, and surface longliners) fishing in distant waters for single target species (e.g., deep-water shrimps, tunas, and swordfish) (Caddy, 1990; Papaconstantinou and Farrugio, 2000; Stergiou et al., 2016).

The official fishing fleet operating in the Mediterranean in 2018 comprises about 76,000 vessels (Food and Agriculture Organization (FAO), 2020). They are unequally distributed, with the Eastern Mediterranean (EM) showing the largest fraction (35.1%), followed by the Central Mediterranean (CM; 26.7%), the Western Mediterranean (WM; 23.8%), and the Adriatic Sea (AS; 14.45%). Multi-gear vessels constitute the dominant group, being 77.8% of all boats. Small-scale fisheries (SSF) predominate along the southern coasts and in EM, while trawling in the WM and the AS (Colloca et al., 2017). From the economic standpoint, trawlers and purse seiners represent 64% of the total revenue, although they provide only 34% of employment (Food and Agriculture Organization (FAO), 2020). Conversely, SSF represent 26% of the total revenue, but provides 59% of jobs. However, SSF remuneration is approximately 50% lower that of trawlers and purse seiners.

Total landings increased from 1970, peaking at about 1,100,000 ton in 1994. In the last two decades, a clear decrease to 790,000 ton in 2018 was observed in the whole Mediterranean Sea, although the yield in some non-European Union (EU) countries is still growing

(Food and Agriculture Organization (FAO), 2020). Considering the main basins and using the 2016–2018 mean yield, the WM (**Figure 1**) dominates (259,000 ton), followed by the AS and the EM (179,000 ton for each), with the CM having the lowest catches (173,000 ton). Although a large variety of species contributes to the total yield, the small pelagics belonging to three species—"incomes" sardine (*Sardina pilchardus*), European anchovy (*Engraulis encrasicolus*), and round sardinella (*Sardinella aurita*)—produce about 44% of the total landing. Despite not representing the largest portion of landings, the multispecies catches of the demersal fisheries provide the highest incomes (Food and Agriculture Organization (FAO), 2020). Among the demersal species, the European hake (*Merluccius merluccius*), deep-water rose shrimp (*Parapenaeus longirostris*), and red mullet (*Mullus barbatus*) amounted to about 7% of the landings.

THE STATUS OF THE MAIN DEMERSAL AND SMALL PELAGIC STOCKS

Most of the Mediterranean fisheries are characterized by a combination of high fishing effort and high level of undersized catch and discards (Colloca et al., 2013).

From 1970 to 2010, developing fisheries and fully exploited stocks were declining at rates ranging from 18% (WM) to 24% (CM), whereas the overexploited and collapsed stocks were increasing at rates between 14% (WM) and 18% (CM) per decade (Stergiou et al., 2016).

Froese et al. (2018), assessing 181 stocks in the Mediterranean and Black Sea by using a Bayesian state-space Schaefer surplus production model, reported that less than 20% of these stocks are exploited at maximum sustainable yield (MSY), while about 60% was depleted (biomass at sea lower than the 50% of the B_{MSY}) in 2014. Simulating their dynamics under different scenarios, depleted stocks would decrease to just 46% in 2030 with 0.95 fishing mortality at MSY (F_{MSY}), while this percentage decreases to 6% with more drastic reduction of fishing effort (no fishing takes place when the stock is depleted and fishing occurs with 0.5 F_{MSY} when biomass is equal to or larger than half the B_{MSY}). A current fishing pressure exceeding several times the MSY was more recently confirmed by Hilborn et al. (2020) and Piroddi et al. (2020). In the last years, however, there has been a decrease in the percentage of stocks in overfishing (from 88% in 2012 to 75% in 2018), as well as in the average exploitation ratio (F/F_{MSY}), which has decreased from 2.9 to 2.4 times the F_{MSY} over the same period (Food and Agriculture Organization (FAO), 2020).

Regarding the main demersal species (76 stocks assessed), *M. merluccius* showed the highest *F*, with the exploitation ratio ranging between 1.7 in CM [geographical sub-areas (GSAs) 12–16)] and 8.5 in WM (GSA 3). *M. barbatus* showed lower values, from 0.3 in CM (GSA 20) to 6.3 in WM (GSA 1). *P. longirostris* showed values between 0.9 in WM (GSAs 9–11) and 3.3 in AS (GSAs 17 and 18). As the small pelagic concerns (seven stocks assessed), *S. pilchardus* resulted in overfishing, with the F_c/F_{MSY} ranging from 1.2 in the WM (GSA 6) to 3.2 in the AS (GSAs 17 and 18). *E. encrasicolus* resulted uncertain in most of the areas, being overfished (1.7) in the AS (GSAs 17 and 18) (Food and Agriculture Organization (FAO), 2020).

Although a situation of overfishing is clearly outlined both for demersals and small pelagics, available information on biomass at sea seems to depict signs of a recovering process. Coupling food web modeling with a hydrodynamical–biogeochemical model (Piroddi et al., 2020) has found increases in the biomass level of elasmobranchs, large pelagics, small and medium demersals, and meso- and bathypelagic fishes when comparing the middle of the 2010s to the late 1990s. Conversely, decreases of large demersal fishes, small pelagics, and commercial and non-commercial cephalopods and crustaceans were reported. Moreover, based on the most recent updated assessment reported by Food and Agriculture Organization (FAO) (2020), the biomass levels in 2018 showed a remarkable improvement compared to that in 2016, with only 36% of the stocks at low biomass (an 11% decrease), 19% at intermediate biomass (a 12% decrease), and 46% with high biomass (a 23% increase). The worst situation is occurring in the WM and in small pelagics. Although resources are still far from MSY, the patterns of both *F* and biomass show that demersal resources seem to react slowly, but positively, to the reduction of fishing effort implemented by the EU countries in the last decade (Maynou, 2020).

THE OBJECTIVES FOR MANAGING THE MEDITERRANEAN FISHERIES

The European Common Fisheries Policy (CFP) [reg. (EU) no. 1380/2013], the Marine Strategy Framework Directive (Directive 2008/56/EC), and the General Fisheries Commission for the Mediterranean (GFCM) mid-term strategy (General Fisheries Commission for the Mediterranean (GFCM), 2016) have adopted the MSY as the main target for fisheries together with a progressive improvement of practices able to reduce the discards of unwanted fish. Furthermore, both policies recognize the protection of essential fish habitats (EFHs) as a tool for improving the sustainability of fisheries while protecting the functioning of ecosystems, in line with the ecosystem approach to fishery management (EAFM). The main tools introduced for improving fishery sustainability in the Mediterranean include the multiannual plans (MAPs) (**Figure 1**).

To balance the fishing fleet to the productivity of stocks, the number of EU fishing units active in the Mediterranean has declined by 30% in the period 1995–2016 (Maynou, 2020). However, it is worth remembering that, while the EU countries' fleet capacity is decreasing, an increase in fishing capacity cannot be excluded in other Mediterranean countries (Colloca et al., 2017).

THE CURRENT MANAGEMENT SYSTEM

Management of Mediterranean fisheries is mostly based on effort control, limiting the number of boats or the time at sea, and some technical measures, such as minimum conservation reference size and minimum mesh sizes (Stergiou et al., 2016;

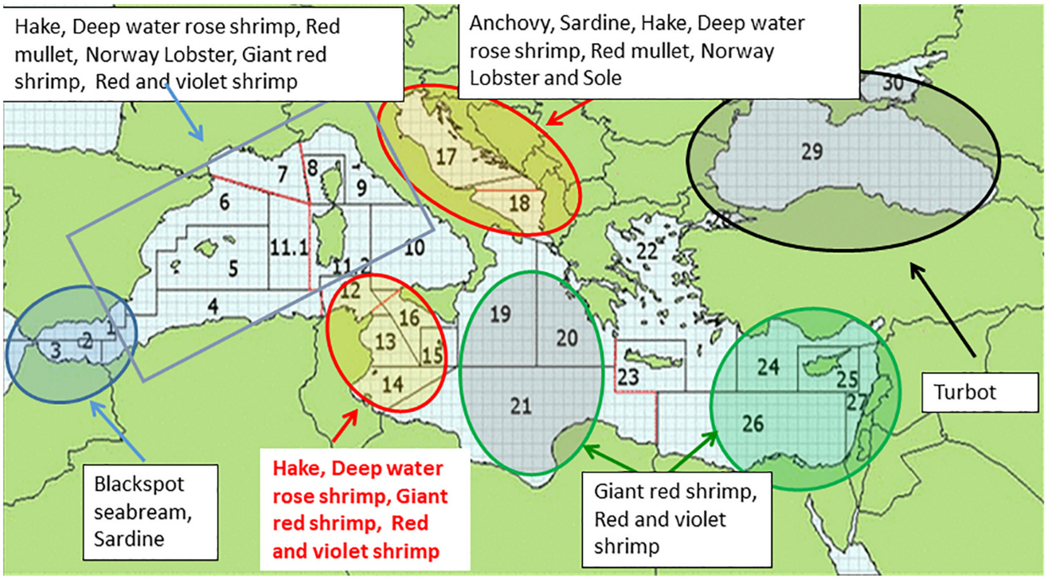

FIGURE 1 | The Mediterranean, the Black Sea, and the geographical sub-areas (GSAs) adopted by the General Fishery Commission for the Mediterranean (GFCM). The *marked areas* show where demersal and small pelagic stocks are shared. Multiannual plans (MAPs) are adopted or being prepared by the EU (*rectangle*) or the GFCM (*ellipses*).

Bellido et al., 2017). However, these approaches were unable to impede the stocks from being overfished (too high fishing mortalities) and overexploited (too low biomass at sea), up to now, without effective common implementation at the scale of the whole basin. One of the main barriers to the effective management of Mediterranean fisheries is the difficulty of less developed countries to implement an effective monitoring, control, and surveillance (MCS) system to contrast illegal, unreported, and unmanaged fisheries. According to Cardinale et al. (2017), the major causes of the critical state of the Mediterranean stocks could be found in the ineffectiveness of the current system to control *F*, the continuous non-adherence to the scientific advice, and the overall inadequacies of the existing management measure. The authors have suggested adopting alternative management measures, such as a catch quota system, currently in force only for bluefin tuna and swordfish in the Mediterranean. Although the multispecies nature of most Mediterranean fisheries and some difficulties in monitoring catches make the widespread adoption of the catch quota system difficult, it could be properly applied for a single or a few species fisheries, such as those targeted to *E. encrasicolus* and *S. pilchardus* or to deep-water red shrimps (Pope, 2009).

THE HIGH FISHING MORTALITIES

Some researchers have proposed very drastic solutions, such as a reduction in fishing effort between 50 and 80% of the present levels, to reverse the current overfishing (Vasilakopoulos et al., 2014; Merino et al., 2015; Froese et al., 2018; Demirel et al., 2020). Although rebuilding overexploited stocks is a priority to guarantee sustainable fisheries in the long term, such a drastic solution does not adequately consider the high expected socioeconomic costs that would require such impressive transformation. To improve fishery sustainability in the Mediterranean, the question cannot be realistically solved by halving the capacity of fleets or their activity, but should be declined in a more composite way.

Attention should be paid to the difficulties in targeting MSY in multispecies fisheries, which is typical of the Mediterranean coastal trawling. When several species with different biological features (first maturity, longevity, and maximum size) are fished together, the F_{MSY} of one leads to the overfishing or underfishing of the other (Sissenwine, 1978). Assessing the sustainable yield curves of mixed trawling in the Ligurian Sea for eight species, with similar weight in landing, and for the entire assemblage by a Schaefer model, Abella et al. (2010) reported that the optimal level of fishing effort in terms of MSY for the assemblage corresponds to that of *M. barbatus*, a small- to medium-sized bony fish, that of *M. merluccius* being lower and that of the horned octopus (*Eledone cirrhosa*) being higher.

Due to the frequent "flat top curves" in the relationship between fishing mortality and yield, Hilborn (2010) suggested using the *F* range delivering 80% of the MSY to provide the so-called pretty good yield. This approach seems to be promising in mixed fisheries, where maximizing the long-term yield could be pursued by choosing target fishing mortalities as the best compromise within a "pretty good yield" range of different species. However, this "pretty" approach is difficult to be applied when the F_{MSY} for species that are caught together is very different (**Figure 2**). For example, in the Strait of Sicily (CM) *P. longirostris* and giant red shrimps (*Aristaeomorpha foliacea*) are the main targets of the Italian trawlers, with more than 50% by weight and 61% by value of demersal yield in 2016,

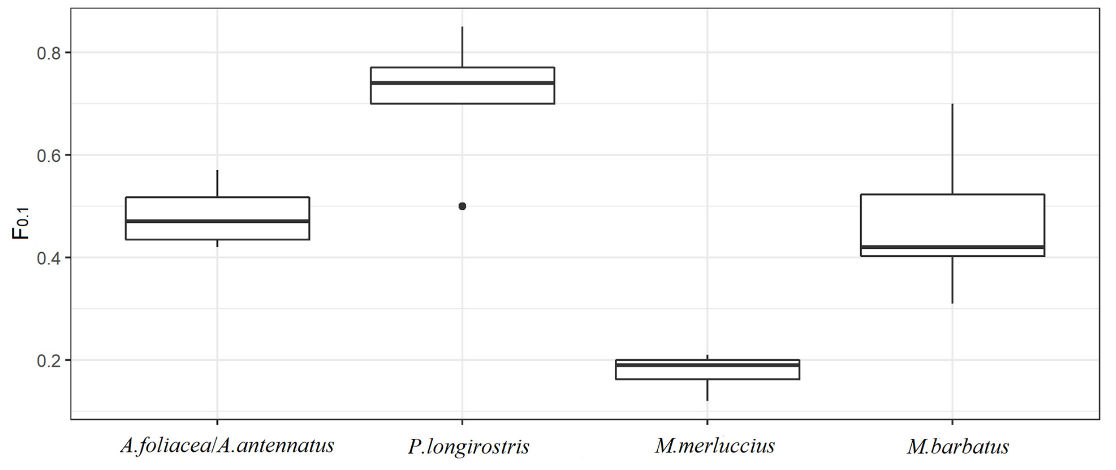

FIGURE 2 | Box plots showing the distribution of $F_{0.1}$, as a precautionary proxy of F_{MSY}, for the main target species of the Mediterranean bottom trawling. With the current exploitation pattern, achieving the F_{MSY} of hake (*Merluccius merluccius*, nine stocks) implies a strong loss of a sustainable yield of red mullet (*Mullus barbatus*, 10 stocks), deep-water rose shrimp (*Parapenaeus longirostris*, five stocks), and red shrimps (*Aristaeomorpha foliacea/Aristeus antennatus*, four stocks) [data from Food and Agriculture Organization (FAO) (2019)].

M. merluccius being the main associated commercial bycatch, with catches amounting to about 10% of the landings and 9% in value (Maiorano et al., 2019).

According to the assessment done to support the Italian MAPs for demersal fisheries in the Strait of Sicily (Ministero delle Politiche Agricole Alimentari e Forestali (MIPAAF)., 2018), the reduction in *F* to reach the *M. merluccius* MSY in 2020 should have been around 80% of the value of 2017. Conversely, to achieve the F_{MSY} for *P. longirostris*, a reduction in *F* of about 30% should be required. However, at MSY of *P. longirostris*, the *M. merluccius* spawning stock biomass would increase by 25% while catches would remain stable, whereas the achievement of the *M. merluccius* MSY would halve the *P. longirostris* yield. Furthermore, from a socioeconomic viewpoint, pursuing the *P. longirostris* MSY would provide, in the medium term, better profitability, economic sustainability, labor cost, and employment indicators compared to the *M. merluccius* MSY strategy. These analyses confirm that reaching the *M. merluccius* MSY implies a deep change in the Mediterranean fisheries with a sharp reduction of trawlers and the development of the longlines fleet targeted exclusively to the adult fraction of the stocks (Aldebert et al., 1993; Lleonart et al., 2003).

THE POOR EXPLOITATION PATTERN

To improve the exploitation patterns of the main target species should be a good objective for two main reasons: the first is that the larger the size of the individuals caught, the higher the level of optimal fishing effort and yield (Beverton and Holt, 1956); the second is that a better exploitation pattern mitigates the problems related to the landing obligation of the CFP (Bellido et al., 2017; Maynou et al., 2018).

Since undersized fishes of many large-sized species, such as *M. merluccius*, are highly vulnerable to the minimum mesh size enforced in the Mediterranean (40 mm square or 50 mm diamond) (Brčić et al., 2018; Mytilineou et al., 2018) and a further increase of the mesh size would lose shrimps, cephalopods, and medium-sized fish, the exploitation pattern can be enhanced through: (i) increasing the trawl net selectivity by adopting grids and separators that allow the undersized fish to escape (Coll et al., 2008; Massutí et al., 2009; Aydın and Tosunoğlu, 2011; Vitale et al., 2018b); (ii) delaying the size/age of the first capture of juveniles through spatial and/or temporal closures to fisheries when and where the juveniles aggregate in order to improve the fraction of fish reaching sexual maturity (Caddy, 1999, 2009; Fiorentino et al., 2003; Garofalo et al., 2011; Colloca et al., 2015; Despoti et al., 2020; Mytilineou et al., 2020; Milisenda et al., 2021); or (iii) a combination of the two approaches.

Empirical evidence of the positive effects of the closure of coastal nurseries to trawling in rebuilding the biomass of *M. barbatus* were provided by Relini et al. (1996) for the Ligurian Sea (WM) and by Fiorentino et al. (2008) for the Gulf of Castellammare (North Sicily—CM). The positive effects of seasonal closure were provided by Mion et al. (2014) for the AS and by Samy-Kamal et al. (2015) for the Catalan Sea (WM).

Furthermore, population dynamics models have highlighted the positive effects of both sorting grid/separator adoption or nursery protection. Vitale et al. (2018a), simulating the effects of a sorting grid mounted on the net of trawlers targeted to *P. longirostris*, showed a benefit for both *P. longirostris* and *M. merluccius* stocks in terms of increasing in biomass and for the fleets in terms of improving the quantity and quality of landings. Fouzai et al. (2012), modeling alternative management scenarios by ECOSPACE in the AS, suggested that protecting EFHs could rebuild the biomass of commercial fish, reporting also benefits for several commercial resources by adopting 3-month closures. Evaluating different management scenarios for demersals in the Strait of Sicily, Russo et al. (2019) showed that both temporal and spatial closures are expected to move to MSY *P. longirostris*, *A. foliacea*, and *M. barbatus*. Despite both closures leading to an improvement in the spawning stock biomass of *M. merluccius* too, the results confirmed that it is not possible to achieve MSY for *M. merluccius* without a very strong reduction of *F*.

HOW TO IMPROVE SUSTAINABILITY OF THE CAPTURE PROCESSES IN THE MEDITERRANEAN

Based on the discussed literature, improving sustainability in the mixed Mediterranean demersal fisheries without causing major social upheaval could be pursued, choosing as a target the optimal *F* of the small- to medium-sized species forming most of the catch of trawling (crustaceans, cephalopods, and fish), considering the concept of the "pretty good yield." Meanwhile, to mitigate the impact of this approach on large-sized fishes, such as *M. merluccius*, skate, sharks, and angler fish, *ad hoc* technical measures should be adopted. Improvement of the current poor exploitation patterns will be best attained by closing trawling areas where undersized fishes are concentrated or adopting sorting devices rather than further increasing the mesh size in the net. This approach, preconized by Caddy (1999) in the late 1990s, is now possible due to the availability of tools for the remote positioning of fishing vessels [vessel monitoring system (VMS), automatic identification system (AIS), and others] (Russo et al., 2016). Although North African countries have extremely few vessels using AIS or VMS technology (Taconet et al., 2019), there are growing initiatives to improve MCS in non-EU countries (Pramod, in press).

Since management based on effort regulation assumes a strong relationship between fishing effort and catch through fishing mortality, this approach should be weak in small pelagics due to the well-known hyperstability of schooling resources' catch per unit effort (CPUE) (Pope, 2009). The small pelagic fisheries in the Mediterranean being based just on two target species and two fishing systems, the adoption of an individual catch quota system should be explored to trigger capture to the productivity of the stock leaving at sea a stock size enough to not impede its renewability.

As climate changes affect strongly the productivity of stocks through changes in recruitment and other demographic parameters, causing a change in the sustainable yield of stocks (Kell et al., 2005; Travers-Trolet et al., 2020), evaluation and management should consider not only fishing effort but also climate and environmental change (Moullec et al., 2019). Consequently, the EU Data Collection Framework and the GFCM Data Collection Reference Framework should be adapted accordingly, improving real-time monitoring of commercial stocks, exploited communities, related environmental drivers, and fishing activities to move toward adaptive management.

To support a spatial-based and adaptive approach to fishery management, scientists are called to improve knowledge on the dynamics of resources and fisheries in space and time, considering climate change and taking into account socioeconomic aspects.

While it should be easier to adopt the suggested control of the spatial pattern of fishing effort or an individual catch quota for the EU vessels, it would be more difficult in those areas where the resources are shared by EU and non-EU fleets, such as the Alboran Sea, the Strait of Sicily, the Adriatic Sea, and the Aegean Sea. Hilborn et al. (2020), reviewing a lot of fisheries including the Mediterranean ones, reported a clear relationship between fishing pressure and management intensity. Although all Mediterranean countries have formally adopted the precautionary approach, the MSY and the EAFM, the different socioeconomic developments in the area suggest that less developed countries pursue reaching the high employment of low-skilled labor with low management costs (Beddington et al., 2007). Since the ecological, economic, and social sustainability of fisheries is not only a technical question but also a cultural and capability-building challenge, the FAO Regional Projects (Copemed II, Medsudmed, Adriamed, and Eastmed) and the GFCM have the main role in constructing a common vision on how to reach more sustainable exploitation of the fishery resources of the Mediterranean, taking into account the complex ecological, social, economic, and political framework.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

Both authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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Mapping Global Research on Ocean Literacy: Implications for Science, Policy, and the Blue Economy

Evelyn Paredes-Coral[1], Melita Mokos[2], Ann Vanreusel[1] and Tim Deprez[1]*

[1] Marine Biology Research Group, Ghent University, Ghent, Belgium, [2] Department of Ecology, Agronomy and Aquaculture, University of Zadar, Zadar, Croatia

In recent years, ocean literacy has become a global movement that connects the human dimension to the ocean and intends to be an incentive for positive change in people's behavior. As multiple initiatives on ocean literacy have arisen, a comprehensive understanding of this topic is required to better engage the broader society. In the present study, we applied a combination of bibliometric analysis and science mapping to a dataset of scientific publications on ocean literacy between 2005 and 2019, obtained from Web of Science and Scopus databases. In order to represent the development of the field, analyze the level of collaborations and uncover its thematic areas, we first used bibliometric analyses to describe the field's main features, including indicators of growth and research collaboration. We then used science mapping techniques to build collaboration networks among countries and institutions, and to identify research communities. Lastly, we performed co-word analysis to reveal the underlying thematic areas and their evolution. Our results reveal a slow-growing number of publications and a promising trend for collaboration among authors, countries and institutions. Education and science were identified as the two major thematic areas on ocean literacy showing that, over time, issues related to these themes have gained more attention among researchers. These findings confirm that ocean literacy is gaining more acknowledgment within the scientific community but still faces considerable limitations to its dissemination in sectors like the blue economy and in regions such as Latin America and Africa. Promoting cross-institutional and cross-disciplinary cooperation among research institutions, marine education networks and the industry is critical to support this purposeful movement and represents an urgent challenge.

Keywords: ocean literacy, science mapping, bibliometrics, blue economy, Sustainable Development Goal 14, Ocean Decade

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Angel Borja, Technological Center Expert in Marine and Food Innovation (AZTI), Spain

Reviewed by:
Ulisses Miranda Azeiteiro, University of Aveiro, Portugal
Baris Salihoglu, Middle East Technical University, Turkey

***Correspondence:**
Evelyn Paredes-Coral
evelyn.paredescoral@ugent.be

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INTRODUCTION

Maintaining a healthy ocean and moving to a more sustainable use of its resources and services is one of the main challenges of the next decade. The ocean is a critical driver of global climate and maintains life providing many vital functions for our planet. It represents a source of food, raw materials, energy and provides the space for many economic activities (Visbeck, 2018; Jouffray et al., 2020). These rapidly evolving human activities have led to unprecedented pressures such as overfishing, pollution, habitat degradation and ocean acidification. Yet, the level of public understanding of basic concepts related to the ocean and the threats associated to human activities remains low to moderate (Gelcich et al., 2014; Fauville, 2019).

Ocean literacy (OL) is a relatively new term that connects the human dimension to the ocean and that intends to be an incentive for positive change in people's behavior. It is defined as the understanding of the ocean's influence on us and our influence on the ocean. An ocean-literate person understands the importance of the ocean to humankind, can communicate about the ocean in a meaningful way, and, is able to make informed and responsible decisions regarding the ocean and its resources (Cava et al., 2005).

The campaign for defining and establishing a framework for OL began in the United States of America (United States) as an initiative to identify key ocean concepts that were missing in the American school curricula. After a series of meetings and workshops that began in 2002, participants from ocean science and education communities together with policy makers, came to a consensus on the definition of OL in 2004 (Cava et al., 2005). As a result, a roadmap for marine educators was published, containing the essential principles (**Table 1**) and fundamental concepts as well as the scope and sequence for each grade at school (Schoedinger et al., 2010).

Few years later, the OL concept reached Europe with the establishment of the European Marine Science Education Association (EMSEA) and the First Conference on Ocean literacy in Europe in 2012 (Copejans and Seys, 2012). Similarly, Canada advanced on its efforts to build an ocean literate society by establishing the Canadian Network for Ocean Education[1] (CaNOE). In a joint effort to promote OL initiatives and to encourage its use when communicating about policy, the European Union (EU), Canada and the United States signed the Galway Statement on Atlantic Ocean Cooperation in 2013 (European Commission, 2013). The Galway Statement stands as an example showing that the OL concept and principles are embedded in the European marine policies. These policies include the Blue Growth Strategy, the Marine Strategy Framework Directive, the Marine Spatial Planning Directive, the Common Fisheries Policy, the Birds Directive, the Habitats Directive and most recently, the European Green Deal (French et al., 2015; European Commission, 2019b).

[1]http://oceanliteracy.ca

TABLE 1 | The seven essential principles of Ocean literacy.

1.	Earth has one big ocean with many features.
2.	The ocean and life in the ocean shape the features of Earth.
3.	The ocean is a major influence on weather and climate.
4.	The ocean makes Earth habitable.
5.	The ocean supports a great diversity of life and ecosystems.
6.	The ocean and humans are inextricably interconnected.
7.	The ocean is largely unexplored.

Cava et al. (2005).

In 2018, the United Nations Educational, Scientific and Cultural Organization (UNESCO) launched the Ocean Literacy Portal[2] as part of the actions to progress on the Sustainable Development Goal 14. The portal provides a free-access compilation of OL resources for students, educators, scientists, policy makers and relevant stakeholders from all over the world. Two years later, in 2020, the European Commission launched the European Ocean Literacy Coalition[3] (EU4Ocean) as a platform to connect organizations, projects and people that contribute to OL and the sustainable management of the ocean. The same year, the Global Ocean Literacy Strategy, supported by the United Nations Decade of Ocean Sciences for Sustainable Development (2021–2030; hereafter referred to as the Ocean Decade), was being drafted.

OL has evolved from a national (United States initiative) to a global scale movement. This dynamic has caught the attention of researchers from several disciplines. As an interdisciplinary field, OL integrates knowledge, techniques and tools from marine sciences (e.g., ecology, oceanography, ecosystem modeling), education sciences, social and behavioral sciences (e.g., sociology and psychology), public health, geography, marine policy, science communication, arts and digital technologies (Dupont, 2017; Fauville, 2017; Costa and Caldeira, 2018; European Marine Board, 2020; Kelly et al., 2021). This diversity of research backgrounds has been accompanied by a broad range of approaches and methods that were included in several scientific publications. However, since this information remains sparse, it is necessary to have an updated outlook to investigate how research advancements are developing in structure and what is the relationship between research communities.

Scientific publications are good indicators of the development of a research field. The quantitative study of scientific publications, citations and journals, is called Bibliometrics (Pritchard, 1969; Broadus, 1987). This technique has been extensively used in a variety of fields ranging from medical sciences (Thompson and Walker, 2015) and cultural evolution (Youngblood and Lahti, 2018) to drug discovery (Agarwal and Searls, 2009) and climate change (Haunschild et al., 2016). In the 1970s and 1980s, bibliometric research was mostly focused on citation analysis to assess the structure of several scientific fields, journal interrelationships, as well as research performance in the humanities and social sciences, citation behavior and interdisciplinary research. In 1990s, powered by the advancements in information technology, international organizations began systematically collecting data to measure and analyze the development of science and technology by means of bibliometrics. Work in the 1990s was focused on the combination of co-citation and word analysis, journal impact measures and the interface of science and technology (van Raan, 2019).

The first decade of the new century was influenced by technological advancements in computer science and the global availability of large bibliographic databases (e.g., Web of Science, previously known as ISI Web of Knowledge) (Chernyi, 2009). Work on bibliometrics addressed new methods for identifying

[2]https://oceanliteracy.unesco.org

[3]www.eu-oceanliteracy.eu

emerging topics, improvements in the visualization of science maps and measures of journal interdisciplinarity, the triple helix model of government–industry–academy interaction, patent citation analysis and the identification of industrially relevant science and text mining. In the last decade, the bibliometric community focused on new indicators of performance and advanced network methods to improve science mapping, university rankings and the comparison between publication-level and journal-level field classifications (van Raan, 2019).

Bibliometrics has undergone a sharp rise since the late 1960s, evolving from a tool to cover library e information center needs, to a powerful field of science with a set of indicators and analytical methods. Over time, this evolution drew the attention of policymakers. Bibliometric research has supported strategic decision making and research funding allocation (Waltman and Noyons, 2018) and has helped to identify the connections between scientific growth and policy changes (Machado et al., 2016). Bibliometric techniques are useful to provide a structured analysis of large datasets, to infer trends over time, identify research themes and shifts in the boundaries of the disciplines. It also enables to detect the most prolific authors and institutions, and to present the "big picture" of a given research area (Aria and Cuccurullo, 2017). In bibliometrics, the two main methods for analyzing a research field are performance analysis and science mapping. While the first method is focused on evaluating the production and impact of publications, science mapping intends to display the conceptual, social and intellectual structure of scientific research, as well as its evolution and dynamical aspects (Gutiérrez-Salcedo et al., 2018).

While OL has captured the attention of diverse research disciplines, previous research has shown that most of the research efforts were focused on educational approaches, particularly at school level (Costa and Caldeira, 2018). Yet, less attention was given to disciplines related to the economic activities happening in the ocean. As the intensity and diversity of these activities continue to grow, the blue economy concept emerges as an approach seeking to promote the sustainable use of ocean resources for economic growth, improved livelihoods, and jobs while preserving the health of the ocean (World Bank and United Nations Department of Economic and Social Affairs, 2017). That being said, it becomes essential to understand the implications of OL as a global movement not only for the scientific community but also in the implementation of sustainable ocean practices and marine policy strategies.

Here, we assess the development of global research on OL with relevance to science, policy and the blue economy. We provide a detailed analysis of what happened and what was published during the last 15 years of research on OL from 2005 (the time when the term OL was first used in a publication) to 2019. To this end, we applied bibliometric techniques aiming (a) to identify the main features of OL research, including indicators of growth, most prolific countries, authors, institutions and publishing outlets; (b) to assess the collaborative structure of OL research at the international and inter-institutional levels; (c) to identify the research coupling OL and blue economy; and (d) to uncover the major thematic areas of research and their progressive evolution.

MATERIALS AND METHODS

Data Collection

Publications related to OL were obtained from Web of Science (WoS) and Scopus databases during August 2020. With the aim to analyze OL as a concept, the search criteria was restricted to publications written in English and the keywords used included "ocean literacy," "ocean literate," "ocean and literacy" and "coast* literacy" as search criteria. Publications were retrieved from the databases' custom data from 1950 and 1960 (WoS and Scopus, respectively) to 2019. The documents where search criteria appeared in the title, keywords, and/or abstract were included in the study. Only documents published in peer-reviewed journals such as article, review and conference paper categories were used. Publications retrieved from WoS and Scopus were merged and duplicates were removed. **Supplementary File 1** includes all keywords and steps used to retrieve publications on OL.

Data Analysis

Bibliometric analysis were carried out using *Bibliometrix* R package (version 3.0.2). *Bibliometrix* is an open-source tool that enables a descriptive and quantitative analysis of the bibliographic data as well as data visualization (Aria and Cuccurullo, 2017). The analysis included the identification of the main features, including indicators of growth, such as number of publications per year, number of authors, institutions and publishing outlets. The most prolific authors, institutions and publishing outlets were also identified. We used the collaboration index (CI) as an indicator of research collaboration. The CI was calculated as the average number of authors on multi-authored papers per year (Elango and Rajendran, 2012). In order to identify the most productive countries, each publication was assigned to its corresponding author's country. For a better visualization of the international collaboration among countries, a collaboration world map was plotted. Afterward, publications were categorized as Single Country Publications, to designate records with authors from the same country, and Multiple Country Publications for records with authors from multiple countries.

In order to complement the macro perspective provided by the collaboration world map, a network analysis was performed using the authors' affiliations (hereafter: institutions) as the units of analysis. The institution collaboration network shows how institutions relate to others in OL research and enables to uncover relevant institutions in a specific research theme. In its graphical representation, the network is made up of several clusters. In each cluster, the institutions are represented by nodes (which size is proportional to its occurrence) and the links represent the collaborations (Aria and Cuccurullo, 2017). Subsequently, we selected the largest network and identified its clusters. A label was assigned to each cluster based on the content of the collaborative publication, to be used as a conceptual guide only. With the aim to identify the publications coupling research on both, OL and blue economy, we extracted the publications in which the title, abstract and keywords were related to the blue economy. We then classified them into categories based on the current sectors of the blue economy.

In order to identify and visualize the major themes on OL research, we performed co-word analysis using the publication's keywords. This technique enables to illustrate associations between keywords by constructing multiple networks based on their similarities (Krsul, 2002). For this specific analysis, we used *KeyWords Plus*, which are the words that frequently appear in the titles of an article's references, but do not appear in the title of the article itself. *KeyWords Plus* is available for WoS publications only. By applying a clustering algorithm on the keywords network, we obtained a two-dimensional diagram, or thematic map, that highlights the different themes present in scientific publications related to OL. Each theme can be analyzed according to the quadrant in which it is placed. The upper-right quadrant indicates the themes that are well-developed also known as motor themes, the lower-right quadrant indicates the basic themes; the lower-left quadrant indicates the emerging or disappearing themes and the upper-left quadrant indicates the very specialized/niche themes (Cobo et al., 2011; Aria and Cuccurullo, 2017). Each sphere represents a network cluster and the cluster names are the words with the higher occurrence values. The sphere volume is proportional to the cluster word occurrences and its position is set according to the cluster's centrality and density. The cluster's centrality measures the strength of the links from one research theme to other research themes, and is an indicator of the significance of a theme in the development of an entire field. The cluster's density measures the internal strength of the network that make up a theme and provides a good representation of the cluster's development (Muñoz-Leiva et al., 2012). To better understand the conceptual evolution of the most recurring themes, we divided the study period in three smaller periods (2005–2011, 2012–2016, and 2017–2019) following the methodology proposed by Cobo et al. (2011). We set a first period of 7 years (2005–2011) given that during the first years of OL research there were few publications and consequently, low number of keywords.

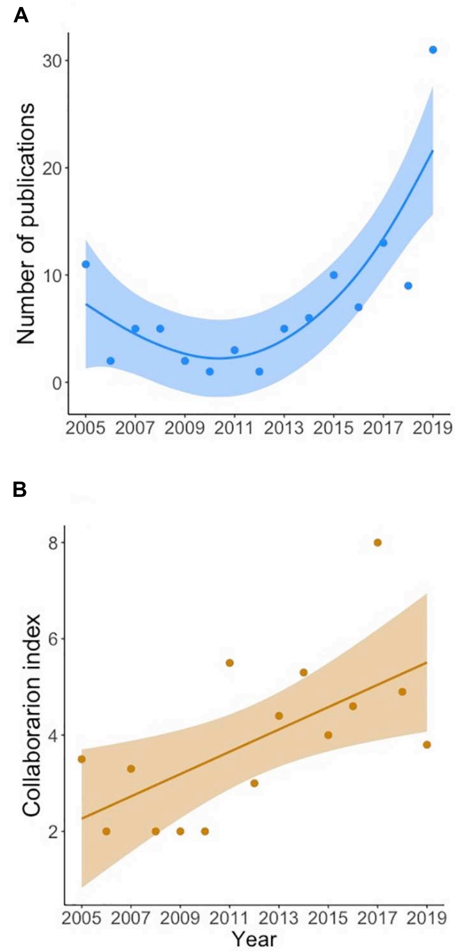

FIGURE 1 | (A) Scatterplot showing the number of publications on Ocean literacy from 2005 to 2019. GAM fitting of the data revealed a positive trend in the number of publications and the appearance of a potential turning point for 2019. **(B)** Scatterplot showing the collaboration index per year. Linear fitting of the data revealed an increase in collaboration among authors.

RESULTS

Development of Global Research on Ocean Literacy

In total, 111 publications were identified suitable for further analysis including 75 articles (67.6%), 30 conference papers (27%) and 6 reviews (5.4%). The development of OL between 2005 and 2019 is shown in the top panel of **Figure 1**. Since 2005, soon after the term OL was formally adopted in the United States, the number of publications has fluctuated over the years, growing by 7.7% on average per year. The overall collaboration index (CI) was 3.8. GAM fitting of the data revealed an increase in the number of publications as from 2012. Linear fitting of CI revealed a positive relationship in the collaborations between 2005 and 2019 (bottom panel of **Figure 1**). In the following years until 2009, publications were dominated by the conference type. The years with less publications were 2010 and 2012 with one article and one conference paper published, respectively. The publication category "review" only appeared in 2017.

A steep noticeable rise in the number of publications was observed in 2019 (n = 31). The number of publishing outlets and authors followed a similar pattern. A total of 368 authors affiliated to 188 institutions have published on OL. Paula Keener-Chavis was identified as the most prolific author with 8 publications (7.2%), other authors included Theodora Boubonari, Mary Carla Curran, Geraldine Fauville and Athanasios Mogias with 4 publications each (3.6%). The majority of authors had an affiliation in the United States (47.7%). The most prolific institutions publishing on OL were led by the National Oceanographic and Atmospheric Administration (NOAA) (14.4%), followed by University of Gothenburg (6.7%), Democritus University of Thrace (5.6%) and National University of Ireland (5.6%). In overall, 57 publishing outlets were identified for the article and review categories (68.4%), and 18 for the conference paper category (31.6%). The most popular journal for publishing on OL was Frontiers in Marine Science (Front.

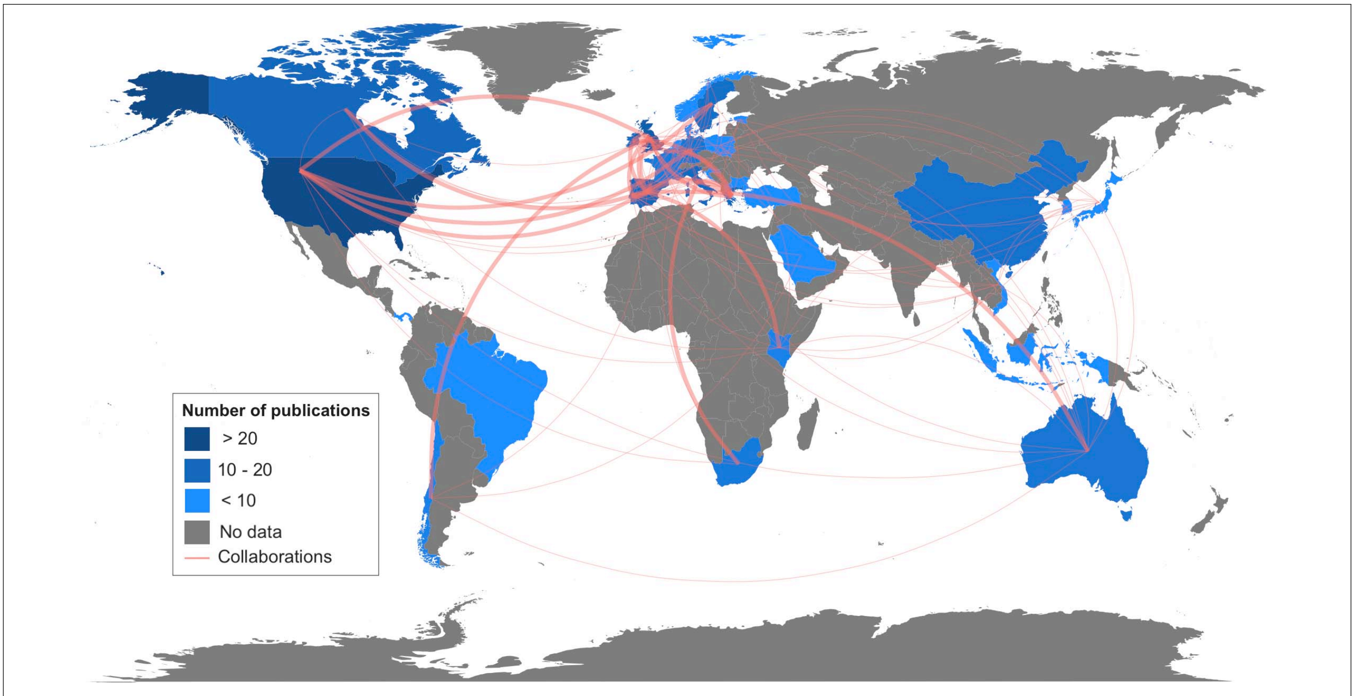

FIGURE 2 | Global collaboration on Ocean literacy research from 2005 to 2019. The blue gradient is proportional to the number of publications by country. Gray color indicates no data and red lines represent collaborations among countries. Line width is proportional to the number of collaborations. World map was created using Biblioshiny app for Bibliometrix R package (version 3.0.2).

Mar. Sci.) with 15 publications, followed by Marine Policy and Sea Technology with 6 and 5 publications, respectively (**Supplementary Figure 1**). Conference papers were published mostly in the Proceedings of OCEANS 05' MTS/IEEE Conference (**Supplementary Figure 2**).

Collaboration Networks

Country Collaboration

A total of 33 countries from five continents have contributed to publishing on OL (**Figure 2**). From the total publications, 20 (18%) were Single Author Publications (SAP) and 91 (82%) were Multiple Author Publications (MAP). The majority of the publications were Single Country Publications (SCP, n = 81; 73%) and a smaller proportion was made by authors affiliated to institutions from different countries (MCP, n = 30; 27%). The United States was identified as the most active country publishing on OL leading with the highest proportion of publications (n = 53; 47.7%) followed distantly by the United Kingdom (n = 10; 9%) and Canada (n = 7; 6.3%) (**Figure 3**). Detailed information regarding country collaboration is shown in **Supplementary Table 1**.

Institution Collaboration

The network analysis of institutions yielded a total of 46 clusters. As most of the clusters were scattered, we extracted the largest network of institutions linked by research on OL resulting in the five clusters portrayed in **Figure 4**. The first group included institutions from the United States and Europe, such as NOAA, College of Exploration, University of California Berkeley, Centro Tecnológico del Mar (CETMAR) and Indigo Med. Based on the content of the collaborative publications, we have chosen to label this group (1) as "ocean exploration and blue economy." Core institutions from group 2 included only European institutions, represented by University of Gothenburg, Democritus University of Thrace, National University of Ireland and the Hellenic Center

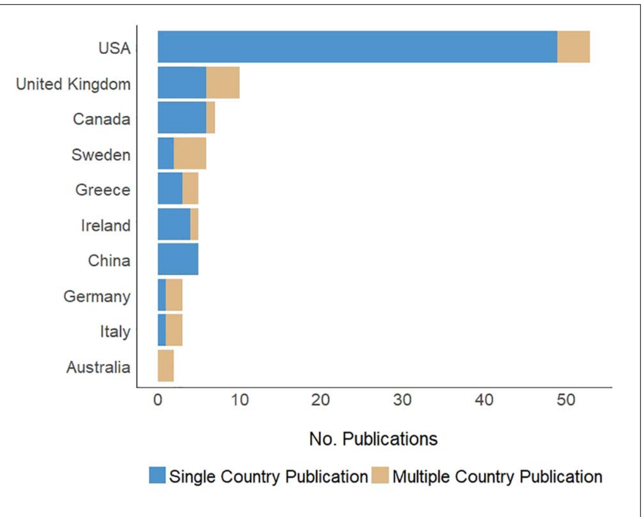

FIGURE 3 | Top 10 publishing countries on Ocean literacy from 2005 to 2019. Multiple Country Publication indicates the number of publications in which there is at least one co-author from a different country. This classification considered the correspondence author's country as the publication's country.

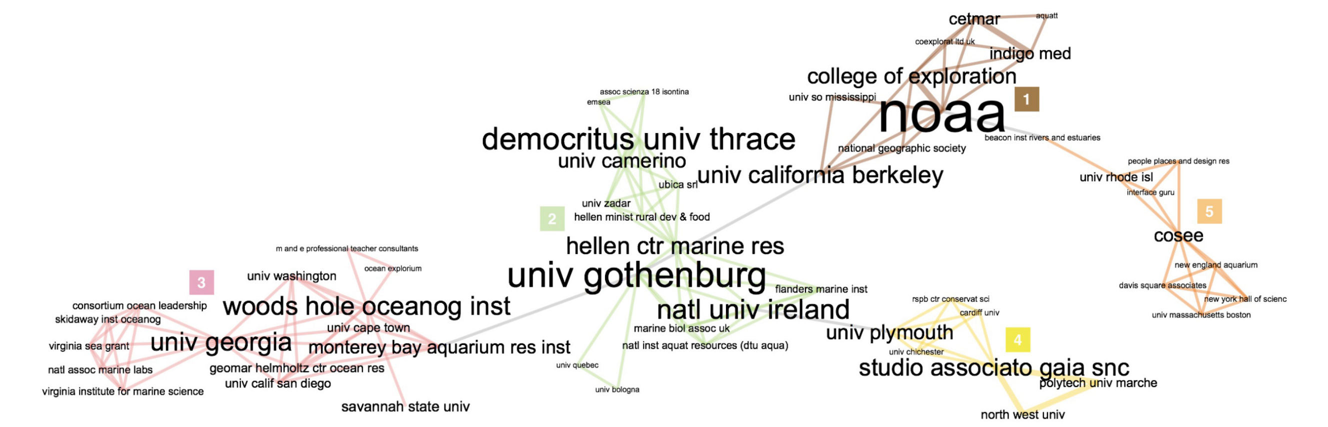

FIGURE 4 | The largest connected institution network in Ocean literacy research between 2005 and 2019 analyzed using collaboration network techniques. Brown corresponds to group 1 ("ocean exploration and blue economy"), green corresponds to group 2 ("marine education and learning technologies"), pink corresponds to group 3 ("oceanography and geosciences"), yellow corresponds to group 4 ("conservation") and orange corresponds to group 5 ("public outreach"). Name size is proportional to the number of publications.

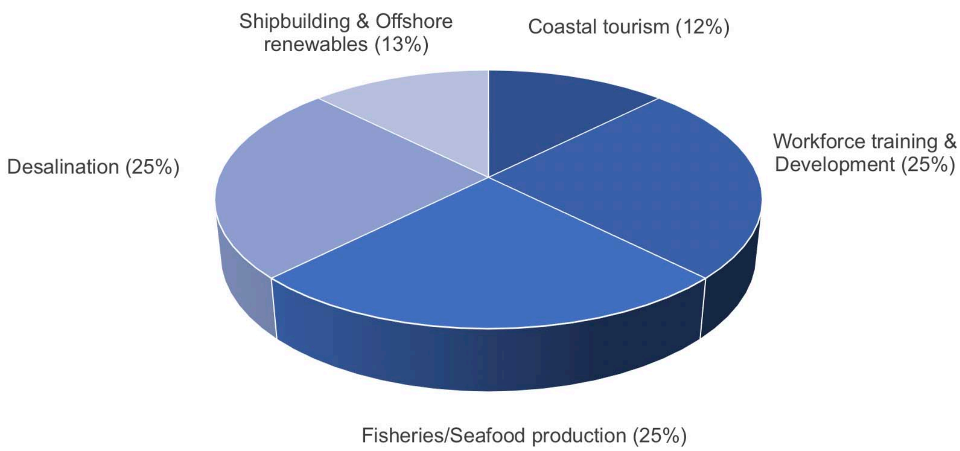

FIGURE 5 | Blue economy topics identified from publications on Ocean literacy for the period 2005–2019 (*n* = 8).

for Marine Research. The label chosen for this group (2) was "marine education and learning technologies." Core institutions in group 3 where mostly from the United States, including University of Georgia, Woods Hole Oceanography Institute and Monterey Bay Aquarium Research Institute. We have chosen to label this group (3) as "oceanography and geosciences." Core institutions in group 4 included the European institutions Studio Associate Gaia SNC and University of Plymouth. The label chosen for this group (4) was "conservation." Core institutions in group 5 belonged to the United States and included the Center for Ocean Sciences Education Excellence (COSEE) and University of Rhode Island. We have chosen to label this group (5) as "public outreach." The aforementioned group labels should be taken as subjective and only used as indicators of research communities rather than referential thematic definitions.

From the five groups, only group (1) consisted of institutions publishing on blue economy, while only 8 publications (7.2%) from our dataset had a focus on the blue economy. The majority of these publications belonged to the article category (75%), followed by conference papers (25%). A total of 28 authors were identified, belonging to 11 institutions from six countries (United States, United Kingdom, Ireland, Spain, Greece and Turkey). All publications had more than one author ranging from 2 to 8 authors, with 3.5 authors per publication on average. Overall, the publications focused on topics related to workforce development and training as well as industrial sectors such as shipbuilding, offshore renewables, coastal tourism, desalination, fisheries and seafood production (**Figure 5**).

Research Themes

According to their location in the thematic map (upper-right quadrant), the themes Education and Science were identified as motor themes on OL research. The themes Management, Attitudes, Knowledge and Climate Change were the most general or basic themes (lower-right quadrant). The themes Hydrothermal Vent, Decision Making and North Atlantic were

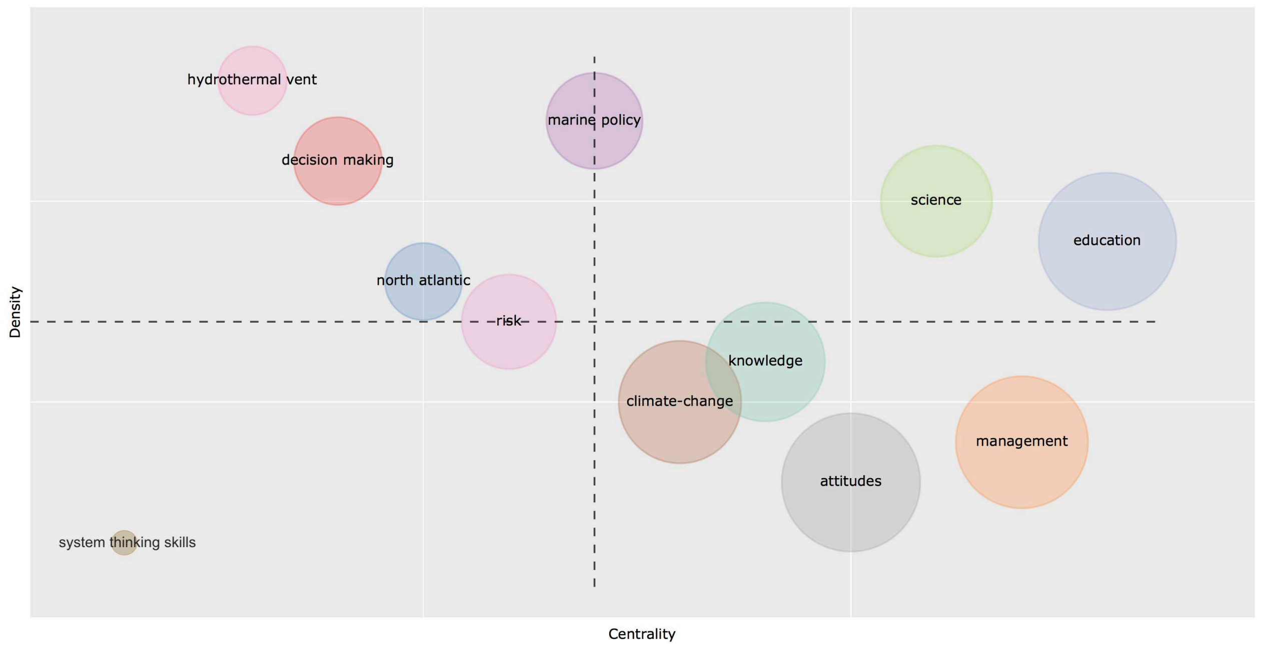

FIGURE 6 | Thematic map on Ocean literacy research for the period 2005–2019 obtained from co-word analysis. The upper-right quadrant indicates the motor themes, the lower-right quadrant indicates the basic themes; the lower-left quadrant indicates the emerging or disappearing themes and the upper-left quadrant indicates the very specialized/niche themes. The volume of the spheres is proportional to the number of publications corresponding to each keyword.

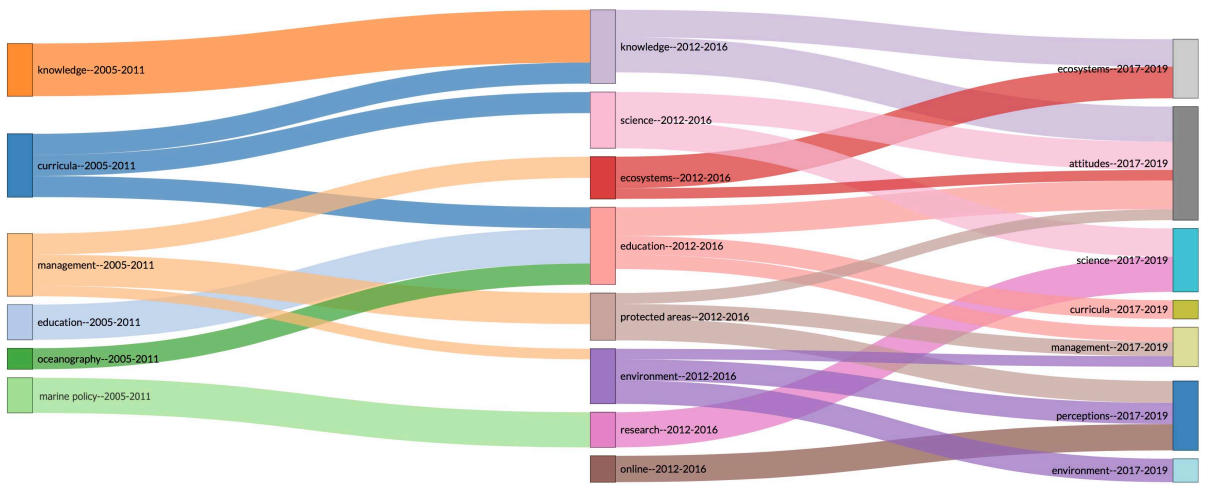

FIGURE 7 | Evolution of Ocean literacy research themes for the periods 2005–2011, 2012–2016, and 2017–2019. Each color represents a research theme and the width of the branches is proportional to the number of publications corresponding to each theme.

three very specialized themes and peripheral in character (upper-left quadrant). The theme System Thinking Skills was presumed to be an emerging theme (lower-left quadrant). The theme Marine Policy was in the transition from motor theme to specialized theme and the theme Risk was in the transition from emerging to specialized theme (**Figure 6**).

Figure 7 illustrates the evolution of the most recurring themes during the periods 2005–2011, 2012–2016, and 2017–2019. Through the analyzed time span, the basic theme Knowledge has unified with Curricula to later become part of the themes Ecosystems and Attitudes. The theme Curricula has diverged into three themes to later reappear in the period 2017–2019. The basic theme Management has diverged into three themes and then has reappeared for the period 2017–2019. The motor theme Education has diverged into three themes, namely Attitudes, Curricula and Management. The theme Oceanography has integrated into the theme Education. Over time, the theme Marine Policy has integrated into Research, which was later integrated into Science. The theme Online has emerged in the period 2012–2016 to be later integrated into the theme

Perceptions. Notably for the period 2017–2019, the theme Attitudes has integrated the motor and basic themes Knowledge, Science and Education.

DISCUSSION

When mapping global research on OL, two main observations regarding its development were identified. First, while the number of publications covering OL showed a slow-growing pattern especially over the first years, the collaboration among researchers seemed more rapidly growing with more authors, countries and institutions involved in publishing. The second observation refers to the identification of research themes underlying in this multidisciplinary topic. Despite its increasing acceptance, the low number of publications on OL indicates that this term has not been widely used in scientific publishing. Previous studies have suggested that research on "environmental literacy" has been more successful than OL research in terms of number of publications (Uyarra and Borja, 2016). Environmental literacy research has produced 292 publications indexed in WoS until 2019, more than twice the amount of publications from OL research. This difference in productivity is understandable given the fact that the OL concept emerged 33 years after the first incorporation of environmental literacy in the scientific literature (Anonymous, 1971). Likewise, Uyarra and Borja (2016) suggested that the interdisciplinary field of "citizen science" was more successful than both OL and environmental literacy. Since 2006 until 2019, citizen science's output has reached 3962 peer-reviewed publications, exceeding by far the other two fields' production (Bautista-Puig et al., 2019). However, "climate literacy," a concept that was adopted in 2006 and that is analogous in structure to the OL concept (USGCRP, 2009), seems to be less successful than OL with only 81 publications indexed in Scopus for the same time span. Additionally, a search of OL in Google gave 155,000 results, suggesting that the term OL is mostly used beyond the scientific domain. We suggest that further research should analyze OL data on websites (web scraping).

Notably, the use of two databases enabled to conduct a comprehensive interdisciplinary search and broaden the field of investigation, minimizing the risk of not capturing the full extent of research on OL. However, the search term "ocean literacy" excludes work by researchers that use different terminology or do not explicitly mention OL. Whereas including other terms in our query such as "marine education" and "ocean awareness" would have expanded our results, we chose to limit our search to one term to avoid over-representing particular themes.

In particular for 2019, the rapid increase reported, with almost five times the average publication rate, may mark a turning point in the OL development with a positive trend that may follow. This increment was, in part, a result of *Frontiers in Marine Sciences* special issue on OL[4]. Considering the new and ongoing initiatives with focus on OL, we should expect them to boost OL publications in the near future. By the time our analysis were done for 2015–2019, there were already 12 publications indexed in WoS and Scopus for 2020, and the Mediterranean Marine Science journal[5] announced a special issue on OL for 2022.

OL research is published in an irregularly distributed manner across publishing outlets. According to our data, the journal with the highest use by OL researchers (Front. Mar. Sci.) accounts for only 13.5% of the publications. Since this topic brings together researchers and ideas from a broad spectrum of academic fields, the journals' scopes are very diverse ranging from computational intelligence and tourism geographies to education and marine policy. Our results indicate the absence of a dedicated journal for OL research, which could be mainly due to the recent origin of the term.

Science mapping enables to reveal hidden patterns in the social structure of a given field, that is, how authors, institutions and countries interact with each other (Aria and Cuccurullo, 2017). Our analysis revealed that authorship is collaborative, with most authors publishing in association with other authors. The positive trend for the collaboration index is particularly promising, suggesting the increase of larger teams and interdisciplinary research that may translate into higher scientific impact (Wu et al., 2019) and productivity (Parish et al., 2018; Murić et al., 2019). In our study, the average level of scientific collaboration on OL research stays aligned to other topics such as biodiversity (Liu et al., 2011), marine sciences (Elango and Rajendran, 2012) and coastal flooding (Gao and Ruan, 2018).

The large number of clusters obtained from the network analysis of institutions, in relation to the total number of institutions, suggests that a cohesive research team has not yet formed. Our results indicate that the international cooperation teams on OL research are gathering but the majority of them are still scattered, with limited cooperation among different institutions. The five research groups represented in **Figure 4** differ in their activity, topics of study and connectivity. The group 2, labeled as "marine education and learning technologies," has the greatest connectivity to other research groups and closeness to the center of the network, suggesting that it is one of the most influential and central research community. Similarly, members of group 1 "ocean exploration and blue economy" and group 3 "oceanography and geosciences" also have high connectivity. This is unsurprising given the fact that several of these institutions have played crucial roles in setting the basis for the foundation of the field and its further dissemination. Institutions from group 4 "conservation" and group 5 "public outreach" are the only non-adjacent groups identified and the furthest from the center of the network. This approach seems very useful to assess the interactions among research communities and has been applied to other interdisciplinary fields such as circular economy (Alnajem et al., 2020) and cultural evolution (Youngblood and Lahti, 2018). However, this approach raises the challenge of labeling the resulting groups in a subjective manner. Hence, we suggest to use our proposed group labels as indicators of research communities rather than thematic areas.

[4]https://oceanliteracy.unesco.org/special-issue-in-frontiers-in-marine-science-on-ocean-literacy/

[5]https://ejournals.epublishing.ekt.gr/index.php/hcmr-med-mar-sc/announcement/view/223

Publications with focus on the blue economy represent a small proportion of the global research on OL (7.2%), indicating that the coupling of these two fields is still developing. OL research has been predominantly pursued within the educational domain, particularly at school level (Williams, 2017; Fauville et al., 2019; Fernández Otero et al., 2019; Mogias et al., 2019), despite its potential to reach citizens in their professional careers and industrial activities across different sectors, including the blue economy (Fernández Otero et al., 2019). As an example of an initiative to reach the maritime sector, the EU-funded MATES[6] project capitalizes the synergies between its partnership integrated by the industry, academia and OL practitioners, to integrate OL as a transversal component in its overall strategy to foster the European shipbuilding and offshore renewable energy sectors (Fernández Otero et al., 2019). Promoting OL research with focus on the blue economy is necessary and of special relevance given that maritime stakeholders, decision-makers and the workforce in general, are not sufficiently aware of the full extent of the environmental, economic, social and political importance of the ocean for their daily lives (Uyarra and Borja, 2016).

International collaboration on OL research is promising. This can be partially attributed to the efforts done by the marine education networks such as the USA-based National Marine Educators Association (NMEA), the Canadian Network for Ocean Education (CaNOE), the International Pacific Marine Educators Network (IPMEN), the European Marine Science Education Association (EMSEA) and the Australian Association for Environmental Education (AAEE) (Marrero et al., 2019). Particularly, the collaboration among European countries is very dense, reflecting high publication activity in a collaborative basis. An example of this are the EU-funded projects Sea Change and ResponSEAble, which have gathered several EU countries and non-EU external experts into partnerships to collectively work in three main societal groups: the general public, formal educators, and policy makers (European Commission, 2018, 2019a). Likewise, the Marine CoLABoration initiative (CoLAB) used a multi-sectorial and values based approach to connect people to the ocean in the United Kingdom (Chambers et al., 2019). Conversely, no research collaborations were found in our dataset within Latin America and Africa. Both regions seem to lack a larger cross-national network to promote OL initiatives in a consistent and culturally relevant way. Nevertheless, the recently created Latin American Marine Educators Association[7] (RELATO) seeks to promote OL in Latin America and the Caribbean, by connecting local initiatives, improving practices and sharing educational material. Additionally, there are several local initiatives on marine education in African countries (SAAMBR, 2019; University of Namibia, 2019; Open Ocean Project, 2020), however, to the best of our knowledge, there is no African network as such. Facilitating the synergies among marine education networks is of particular interest, as this can accelerate the sharing and dissemination of knowledge and attract more attention to OL research, especially in low and middle income countries. Particularly, programs fostering international collaboration for Latin American and African research communities might help to level the playing field.

Education and Science were identified as the most heavily studied themes on OL research, being both well-developed and important for the structuring of this field. This is consistent with previous work that highlighted the emphasis placed on educational approaches on OL research (Costa and Caldeira, 2018). These themes were strategically located in the upper-right quadrant of the thematic map, indicating that they were also related externally to concepts applicable to other themes, such as management and climate change. Our results are well aligned to the current trends in OL research and are supported by previous work done in education (school and higher education) (Schaffner et al., 2016; Mogias et al., 2019) and marine science (Cava et al., 2005; Schoedinger et al., 2005; Visbeck, 2018). Other important themes were management, climate change, attitudes and knowledge, which notably, include strong social aspects and public perceptions (Potts et al., 2016; Ashley et al., 2019; Stoll-Kleemann, 2019). **Figure 7** revealed that most thematic areas evolve in a discontinuous but compact way from their beginning. This suggest that over the time, they attract the interest of the research community, characterized by a progressive growth in the publications on these themes. These findings support the potential advantages of using bibliometric analysis to uncover the intellectual structure and evolution of research themes. Overall, this approach has shown to be effective to analyze the evolution of fields such as climate change (Sharifi et al., 2020), sustainable tourism (Della Corte et al., 2019) and circular economy (Alnajem et al., 2020). One of the limitations of this analysis is that the use of *KeyWords Plus* excludes the publications indexed by Scopus, which does not provide this metadata. However, *KeyWords Plus* was chosen based on its suitability as the best content field for performing analysis on thematic areas and in order to avoid the lack of standardization reported for author's keywords (Ugolini et al., 2001).

Implications for Science, Policy, and the Blue Economy

Our results suggesting that OL is an emerging field of science are not just bibliometric indicators but also powerful evaluation tools for science policy-makers, research managers, and individual researchers. It provides a strategic overview that synthesizes 15 years of research and validates the inclusion of OL as one of the priority areas of research and technology development of the Ocean Decade (R&D 7; Ryabinin et al., 2019). As such, OL should be recognized as a research field and should be allocated adequate funding support for long-term projects and placement in organizational work programs (Eparkhina et al., 2021).

Effective strategies to eliminate the reported disparities in OL research between the Global North and the Global South, are likely to require joint efforts by researchers, practitioners, policy-makers and the industry, with a rapid exchange of knowledge among them (Eparkhina et al., 2021). While research capacity on OL needs to grow globally, particular attention should be given to regions and groups from Small Island Developing

[6]https://www.projectmates.eu
[7]https://relatoceano.org

States, Least Developed Countries and Landlocked Developing Countries (Ryabinin et al., 2019). In addition, training aimed to improve research capacity should be powered by web-based tools, such as MOOCs and virtual reality, to increase information flow and knowledge exchange (Waite et al., 2017; Fauville et al., 2021; Jacobs et al., 2021). To be most effective, OL research will need a solid foundation across the science-policy interface and international cooperation within and across ocean basins, as stated in the Ocean Decade's mission (UNESCO/IOC, 2020a).

OL research with focus on the blue economy seems to be scarce and sector-specific, and will increasingly need to follow an interdisciplinary approach across the marine, maritime, education, social and economic sciences (Bavinck and Verrips, 2020; ten Brink et al., 2020). Managing the blue economy requires managing people, which calls for efforts to better understand their knowledge, attitudes, behavior and needs (Ashley et al., 2019; Cavallo et al., 2020). Such efforts require strategies across multiple sectors, from high-level policy-makers to individual-level behavioral changes (Cisneros-Montemayor et al., 2021). Benchmarking and continued monitoring of OL levels are necessary to evaluate the effectiveness of programs and initiatives (Eparkhina et al., 2021), not only for students but for all actors of society (Kelly et al., 2021), like those directly linked to the ocean, such as maritime workers. This need is well-aligned with the ultimate goal of the Ocean Decade, aiming to connect ocean science with the needs of society and effectively support sustainable development (Claudet et al., 2020; UNESCO/IOC, 2020b).

Overall, this study provides a global perspective on OL research. Our findings evidence the development of the field between 2005 and 2019 using the information contained in scientific publications. Based on our findings, we point out the need to foster coordinated and interdisciplinary collaboration by integrating the scientific community, decision-makers, the industry and relevant practitioners, which can result in stronger and more consistent partnerships. We hope that experts and decision makers could use the results provided by this study to gain a better understanding of the current state of the art in OL research and to orient future research.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

EP-C and TD contributed to conception and design of the study. MM and AV discussed initial ideas. EP-C organized the database, performed the statistical analysis, and wrote the first draft of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.648492/full#supplementary-material

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How to Meet New Global Targets in the Offshore Realms: Biophysical Guidelines for Offshore Networks of No-Take Marine Protected Areas

Daniela M. Ceccarelli[1,2*], *Kate Davey*[3], *Geoffrey P. Jones*[2,4], *Peter T. Harris*[5], *Sione V. Matoto*[6], *Jason Raubani*[7] *and Leanne Fernandes*[3,4]

[1] Marine Ecology Consultant, Nelly Bay, QLD, Australia, [2] ARC Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD, Australia, [3] Oceania Regional Office, International Union for the Conservation of Nature, Suva, Fiji, [4] College of Science and Engineering, James Cook University, Townsville, QLD, Australia, [5] GRID-Arendal, Arendal, Norway, [6] Fisheries Consultant, Nuku'alofa, Tonga, [7] Secretariat of the Pacific Community, Noumea, New Caledonia

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***Correspondence:**
Daniela M. Ceccarelli
dmcecca@gmail.com

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Networks of no-take marine protected areas (MPAs), where all extractive activities are prohibited, are the most effective tool to directly protect marine ecosystems from destructive and unsustainable human activities. No-take MPAs and MPA networks have been globally implemented in coastal seas, and their success has been significantly enhanced where science-based biophysical guidelines have informed their design. Increasingly, as human pressure on marine ecosystems is expanding further offshore, governments are establishing offshore MPAs—some very large—or MPA networks. Globally, there are growing calls from scientists, non-government organisations, and national governments to set global conservation targets upwards of 30%. Given that most of the ocean is found either in the high seas or offshore within national Exclusive Economic Zones, large offshore MPAs or networks of MPAs must be a major component of these global targets for ocean protection. However, without adequate design, these offshore MPAs risk being placed to minimise conflict with economic interests, rather than to maximise biodiversity protection. This paper describes detailed biophysical guidelines that managers can use to design effective networks of no-take MPAs in offshore environments. We conducted a systematic review of existing biophysical design guidelines for networks of MPAs in coastal seas, and found consistent elements relating to size, shape, connectivity, timeframes, and representation of biophysical features. However, few of the guidelines are tailored to offshore environments, and few of the large offshore MPAs currently in place were designed systematically. We discuss how the common inshore design guidelines should be revised to be responsive to the characteristics of offshore ecosystems, including giving consideration of issues of scale, data availability, and uncertainty. We propose 10 biophysical guidelines that can be used to systematically design offshore networks of MPAs which will also contribute to the global goal of at least 30% protection globally. Finally, we offer three priority guidelines that reflect the unique conservation needs of offshore ecosystems: emphasising the need for larger MPAs; maximising the inclusion of special features that are known and mapped; and representing minimum percentages of habitats, or, where mapped, bioregions. Ultimately, MPA guidelines need to be embedded within an adaptive management framework, and have the flexibility to respond to emerging knowledge and new challenges.

Keywords: marine reserves, oceanic, pelagic, marine conservation, ecological principles, marine protected areas, design, guidelines

INTRODUCTION

Our oceans are immensely valuable, both intrinsically and to our economies, societies, and cultures. However, human pressures are causing significant and, in some cases, catastrophic declines in marine species (Duarte et al., 2020). Marine protected areas (MPAs), especially no-take MPAs that prohibit extractive use (Sala and Giakoumi, 2017), are considered among the best tools available to protect marine species and habitats from exploitation and damage, and to conserve marine biodiversity (Graham et al., 2011; Costello, 2014; Roberts et al., 2019). Common biophysical goals of MPAs are to maintain or restore native species diversity, habitat diversity and heterogeneity, keystone species, connectivity, and important ecological processes (McCook et al., 2010; Green et al., 2013, 2014). Usually, achievement of these biophysical and ecological goals allows the consequent achievement of socio-economic and cultural objectives, including, for example, the protection or restoration of fisheries, food security, and cultural landscapes (Gilman et al., 2011).

Whilst MPAs and MPA networks have been broadly established in the world's coastal seas, the application of spatial protection to offshore environments is much newer (Ban et al., 2014a). For the purposes of this paper, offshore waters (also referred to as the open ocean or deep sea) are defined as all marine areas (benthic and pelagic) beyond the seaward edge of the geomorphic continental shelf, which is often at a depth of ∼200 m. Where there is no continental shelf (e.g., oceanic islands and atolls), offshore waters are understood to be marine areas beyond the 80 m depth contour, which is a generally accepted depth limit of light-dependent habitat-building organisms (Bongaerts et al., 2011; Bridge et al., 2011; Althaus et al., 2017; Lesser et al., 2019; Beger et al., 2020). We use the word "offshore" as an umbrella term to encompass benthic, demersal, and pelagic habitats both within the exclusive economic zones (EEZs) of nations and in areas beyond national jurisdiction (ABNJ), as long as they are beyond marine areas that are under the jurisdiction of local communities (i.e., beyond the scope of community-managed marine areas), beyond the continental shelf break or deeper than 80 m around oceanic islands. The legislative, economic, and practical requirements of establishing MPA networks by individual States within their EEZs are different from those of the international community when protecting ABNJ (Merrie et al., 2014). However, whilst important, those considerations are beyond the scope of this paper. This paper focuses on biophysical design guidelines only. These guidelines are not intended to replace existing design principles applied in coastal seas (e.g., Green et al., 2014), and they will most likely be tempered by socio-economic and cultural considerations, national legislation and international agreements.

The open ocean contains a wide variety of ecosystems and species assemblages, from the pelagic habitats at the surface to the deepest realms of the seabed. The view that the deep sea is physically and biologically homogeneous has been dispelled (Herring, 2002; Benoit-Bird et al., 2016), and the deep sea is now known to host levels of biodiversity that rival those of shallow-water coral reefs (Van den Hove et al., 2007).

Far from being resilient, the open ocean and the deep sea are home to some of the most long-lived and vulnerable marine animals, habitats and ecosystems on earth (Verity et al., 2002; Glover and Smith, 2003; Roberts et al., 2019). The open ocean is under increasing pressure from human impacts, especially overfishing, bycatch of non-target species, destructive fishing methods, noise, pollution and litter from land (including plastic), shipping (including cruise shipping), derelict fishing gear, deep sea mining for non-renewable resources and climate change (Verity et al., 2002; Halpern et al., 2008; Ramirez-Llodra et al., 2011; UN, 2015; UN Environment, 2017; Harris, 2020).

As coastal fisheries become depleted and technological improvements allow fishing vessels to venture further offshore, pelagic fish stocks and deepwater seabeds are more at risk of overexploitation than ever (Baum et al., 2003). Numerous heavily exploited offshore species are now of conservation concern, including some tuna, billfish, and sharks (Ferretti et al., 2010; Collette et al., 2011). In the open ocean, overfishing affects not just targeted stocks but also by-catch species, community composition, habitats, trophic functioning, and ecological linkages, in both the horizontal and vertical dimensions (Roberts, 2002; Worm and Tittensor, 2011; Ortuño Crespo and Dunn, 2017). The relatively low productivity, weaker governance, and data deficiency of the open ocean make it difficult to determine what level of fishing activity targeting pelagic and deep-sea species is sustainable (Collette et al., 2011; Norse et al., 2012; Ortuño Crespo and Dunn, 2017; Palomares et al., 2020; Pauly et al., 2020). Furthermore, the two-way coupling between offshore benthic and pelagic systems means that impacts in the upper parts of the open ocean, which are more commonly fished, cascade through the entire vertical span of offshore assemblages (Grober-Dunsmore et al., 2008).

While there are large gaps in knowledge (Palumbi, 2004; Claudet et al., 2010; Dunne et al., 2014), increasing evidence shows that no-take offshore MPAs can offer effective protection against human exploitation and damage (Mills and Carlton, 1998; Koldewey et al., 2010; Davies et al., 2012). Large offshore MPAs and MPA networks can protect pelagic ecosystems along with deep-sea benthic and demersal ecosystems that are highly fragile and closely inter-linked (Norse, 2005; Davies et al., 2007; Williams et al., 2010b; Huvenne et al., 2016). In fact, recent research suggests that offshore no-take MPAs can not only promote the recovery of highly mobile species (e.g., tuna) and protect large swathes of habitat, but also enhance fish stocks and help to stabilise catches outside MPA boundaries (Boerder et al., 2017). There is an increasing body of scientific research devoted to understanding the offshore environment (e.g., Schmidt Ocean Institute, 2020); much of this research identifies the need to define design guidelines for offshore networks of MPAs to achieve conservation and other management goals (Leathwick et al., 2008; Ban et al., 2011; Berglund et al., 2012; Chaniotis et al., 2020).

Currently, 2.7% of the global ocean is fully and/or highly protected within no-take MPAs; the proportion of countries' EEZs under MPA protection is higher (5.7%) than ABNJ (<1%; Marine Conservation Institute, 2020). In recent years, partly due to increased knowledge, the number of large-scale offshore MPAs has grown (Lewis et al., 2017; Duarte et al., 2020), and, worldwide, there are now over 30 no-take MPAs larger than 150,000 km^2. Existing very large (>150,000 km^2) offshore MPAs were shown to encompass at least 10% of the range of 26.9% of all species assessed worldwide; the remaining 73.1% of species fall short of a target of 10% coverage within these MPAs (Davies et al., 2017). The failure to meet species conservation targets is thought to be because, so far, very large MPAs have been opportunistic and placed mostly in remote areas to avoid interfering with commercial interests, rather than systematically designed to adequately protect the full range of habitats and species found within a given area (Leenhardt et al., 2013; OSCA, 2016; Devillers et al., 2020). The need to design offshore MPA networks according to robust biophysical guidelines is clear (Ban et al., 2014b; Davies et al., 2017; Lewis et al., 2017; IUCN-WCPA, 2018).

In 2011, the Convention for Biological Diversity (CBD) formulated the Aichi targets, of which Target 11 states that "*By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes*." (CBD, 2011). This target has been reiterated, in 2015, by all United Nation members in the Sustainable Development Goals (specifically SDG14; UNDP, 2021). In 2016, members of the International Union for the Conservation of Nature (IUCN) at the World Conservation Congress approved new global target for MPAs, calling for 30% of each marine habitat to be set aside in highly protected MPAs and other effective area-based conservation measures by 2030 (IUCN, 2016a). Since then, this call has been echoed by various scientists, non-government organisations and national governments, including the UK Government who recently celebrated over 40 countries joining the UK-led "30 by 30" Global Ocean Alliance Initiative (UK Government, 2021), an international commitment to protect at least 30% of the global ocean in MPAs by 2030, through the UN Convention on Biodiversity in 2021 (O'Leary et al., 2019). In parallel, United Nations representatives are in the process of negotiating a treaty that would, among other things, create a mechanism to establish marine protected areas on the high seas (Gjerde, 2007). This mechanism includes an increasing expectation that global targets of 30% must be met in order to safeguard biodiversity, avoid fishery collapse and build ocean resistance to climate change (Partridge, 2009; O'Leary et al., 2019; Visalli et al., 2020). With most of the ocean found either in the high seas or offshore within national Exclusive Economic Zones, large offshore MPAs or networks of MPAs are integral parts of reaching global targets for ocean protection.

Generally, establishing an MPA or a network of MPAs consists of a series of steps that include defining objectives, planning, design, consultation, declaration, and management (Kelleher and Kenchington, 1992). This paper focuses upon the "design" step in the context of the global objectives referenced above. We describe detailed biophysical guidelines that managers can use to design effective networks of no-take MPAs in offshore waters. The systematic review that led to the definition of these guidelines sought to answer the following questions:

(1) What are the existing design guidelines for no-take MPA networks, largely applied to shallow coastal ecosystems?

(2) How do the differences between coastal and offshore ecosystems and species inform tailoring of those guidelines to offshore environments?

MATERIALS AND METHODS

Existing literature that contained design principles or guidelines for MPAs were collated using online search engines (Web of Science Core Collection, Scopus), Google Scholar and the internal search functions of conservation organisation websites. The search term ("marine protected area*" OR "marine reserve*" OR "no-take") AND (guideline* OR principle* OR criteria) was initially tested on 10 key documents (5 peer-reviewed and 5 "grey literature" reports), to ensure it was capable of detecting the relevant literature. Equal weight was given to grey literature in the literature search, in recognition that MPA principles or guidelines often appear in documents designed for use by management agencies, rather than for academic purposes. All results were uploaded to the online software Cadima (www.cadima.info), through which we specified the research question in terms of Population/Outcome, and which automatically detects duplicates and assists with screening and data extraction (O'Leary et al., 2016a). After the initial literature search and duplicate exclusion, the resulting 795 documents were screened for relevance first by title, then by abstract and lastly by full-text articles, resulting, initially, in 264 articles included for data extraction (**Table 1**, **Supplementary Information 1**). Of these, 177 documents contained information about design principles or guidelines for MPA design. The data extracted from

TABLE 1 | Systematic review results of literature search per search string and database (see also **Supplementary Information 1**).

Search string	Source	Results	Date
("marine protected area*" OR "marine reserve*" OR "no-take") AND (guideline* OR principle* OR criteria)	Web of Science Core Collection	667	2020-08-22
("marine protected area*" OR "marine reserve*" OR "no-take") AND ("design guideline*" OR "design principle*")	Web of Science Core Collection	30	2020-08-22
("marine protected area*" OR "marine reserve*" OR "no-take") AND ("design guideline*" OR "design principle*")	Google Scholar	128	2020-08-23
marine protected area guidelines	Various government and organisation websites	12	2020-08-26
Total records		**839**	
Records after duplicate removal		794	
Records screened at title level		478	
Records screened at abstract level		342	
Full-text articles assessed for eligibility		275	
Included articles		263	
Articles with MPA design guidelines		176	

these documents included the author(s), year, title, geographic location, specific MPAs, individual guidelines or principles, relevance to networks of MPAs (as opposed to individual MPAs) and relevance to offshore environments.

Each guideline or principle relevant to MPA network design was then assessed as to its applicability to an open ocean context and the guidelines tailored accordingly to be offshore-specific.

MPA DESIGN GUIDELINES – RESULTS OF THE SYSTEMATIC REVIEW

One hundred and seventy seven articles, published between 1992 and 2020, contained information about MPA design (**Table 1**). Of the 177 articles:

- 52 included a clear list of guidelines or principles, although 28 were more general in nature and did not provide explicit recommendations on, for example, the magnitude or percentage (of habitats, bioregions, etc.) to include in the MPA design;
- 23 included quantitative guidelines such as sizes, distances, and/or % protection targets;
- 49 were about MPAs in general and discussed or presented guidelines in a theoretical sense. The other 128 documents were about specific regions of the world, or specific MPAs;
- 129 (73%) were relevant to networks, rather than just individual MPAs;
- 89 (50%) had some direct relevance to offshore environments;
- 53 (30%) only considered one guideline or principle; adding those papers that considered only two principles brought the number to 77 (44%);
- the most commonly cited guidelines were representation of habitats (103 documents, 58%), connectivity (91 documents, 51%), and size (70 documents, 40%);
- Of the 52 studies (of the 177) that had a list of guidelines/principles, 25 possible guidelines or principles were presented in various combinations. Not all were biophysical, and this paper does not consider these non-biophysical guidelines further.
- 16 of the 23 papers that provided quantitative design guidelines referred to overarching % targets for no-take protection without specific targets for particular attributes of the environment (e.g., habitats, bioregions). Most others either referred to the Convention on Biological Diversity (CBD) 10% target (Arcos et al., 2012; Balbar and Metaxas, 2019), or used MARXAN (a conservation planning software tool to guide systematic MPA design) or a similar tool to explore ways in which to target a range of area percentages for conservation (e.g., Proudfoot et al., 2020). We note that papers referring to overarching percentage MPA targets recommend a range from 10 to 50% (e.g., Thomas and Shears, 2013; Dunn et al., 2018) and that, largely, the per habitat- or per bioregion-specific targets mentioned in other work, if implemented, would sum to these overarching targets.

TAILORING THE GUIDELINES TO OFFSHORE ECOSYSTEMS

The applicability of guidelines found throughout the literature to offshore waters is detailed in the sections below. Guidelines are listed in order of priority (**Table 2**).

Offshore Guideline 1: Make MPAs Larger

Size is one of the most important design considerations when implementing MPAs, especially in data-poor areas (Halpern, 2003; Gilman et al., 2011). In coastal seas, the representation of habitats and/or bioregions tends to be prioritised over size. However, the ethos of "bigger is better" is one of the five characteristics that has led to the greatest realised benefits of no-take MPAs globally (Edgar et al., 2014), and often incidentally enhances connectivity (Álvarez-Romero et al., 2018; see Guideline 5). Very small, permanent, no-take MPAs can be effective in coastal seas, and especially when designed for

TABLE 2 | Summary of biophysical guidelines for the design of offshore networks of no-take MPAs.

Design guideline	Conditions	Rationale - summary	References
1. Make no-take MPAs 50–200 km in diameter.		Tagging studies show that large pelagic predators (tunas, billfish, blue and shortfin mako sharks, dolphinfish, wahoo, penguins) can move 1,000s of kms, but that the majority of the populations remain within 250 to 1,000 km of their release location. Modelling studies show that protecting 50% of the range of wide-ranging species, especially if critical habitat is included, can benefit the entire population. Additionally, these species can act as "umbrella species"; protecting enough area for them will automatically benefit a large diversity of more sedentary pelagic species and the seafloor below.	Clark, 1996; Hampton and Gunn, 1998; Lauck et al., 1998; Kingsford and Defries, 1999; Sedberry and Loefer, 2001; Kohler et al., 2002; Sibert and Hampton, 2002, 2003; Worm et al., 2003; Bromhead et al., 2004; Micheli et al., 2004; Clear et al., 2005; Alpine and Hobday, 2007; Theisen et al., 2008; Holdsworth et al., 2009; Cosgrove et al., 2010; McClain and Hardy, 2010; Sepulveda et al., 2010; Read et al., 2013; Schaefer et al., 2014; Howey et al., 2016; Huvenne et al., 2016; Robinson et al., 2016; Della Pella et al., 2017; Gary et al., 2020
2. Ensure that no-take MPAs include critical habitats and biologically or physically special and/or unique sites and species. This may include, for example, unique geomorphologic or hydrodynamic features (see **Table 5**), areas important for aggregation, nurseries, spawning, foraging, offshore nesting sites, migratory staging points, mammal calving areas, areas with high biodiversity, endemism, productivity or with threatened, isolated or rare species or habitats.	Features are mapped	For an MPA network to comprehensively and adequately protect biodiversity, known special or unique areas must be included in no-take MPAs. Productive areas are important due to their contribution to ecosystem functioning and potential for high biodiversity; they are usually "hotspots" for multiple species. Areas that are critical to large species are often automatically important for a large variety of other, smaller, more sedentary pelagic or benthic species. It is important to note that for threatened or endangered species, protecting 30% of their habitat niche may be insufficient to prevent extinction. Thus, some habitats may require 100% protection while others can endure with less.	Glover and Smith, 2003; De Santo and Jones, 2007; Hobday et al., 2011; Hooker et al., 2011; Ban et al., 2014a; Clark et al., 2014; Maxwell et al., 2014; Asaad et al., 2017; Ceccarelli et al., 2017, 2018c,d; Lundquist et al., 2017; Rigby et al., 2019
3a. Protect 35% of each habitat type or feature listed in **Table 5** within no-take MPAs.	No bioregions defined	When there is no definition of bioregional boundaries, there is often still at least an approximate understanding of habitats present. When Guideline 4 cannot be applied, capturing a larger proportion of each habitat enhances the likelihood of capturing unknown and therefore unmapped within-habitat variability.	Hyrenbach et al., 2000; Belkin et al., 2009; McClain and Hardy, 2010; Harris et al., 2014; Miller and Christodoulou, 2014; O'Leary et al., 2016b; Chaniotis et al., 2020
3b. Include a percentage of each habitat type or feature as indicated by **Table 5**, within no-take MPAs. Include adjacent habitats as buffer zones.	Bioregions defined at an appropriate scale so Guideline 4 also applies	Mappable features of the open ocean are known areas of high productivity, diversity, or significant ecological processes. To ensure future sustainability of offshore marine environments, examples of the full range of known and mapped biophysical habitats should be included in no-take MPAs.	Hyrenbach et al., 2000; Sibert and Hampton, 2002; Alpine and Hobday, 2007; Williams et al., 2010a
4a. Represent at least 20–30% of marine bioregions in no-take MPAs	Bioregions defined at an appropriate scale	Protection of all habitats, flora and fauna, ecosystem function, integrity and resilience requires that adequate examples of every bioregion are included in no-take MPAs. The best available science informs that at least 20–30% of each marine bioregion should be included in no-take areas, especially if aiming to protect species with lower reproductive output or delayed maturation (e.g., many large offshore and deep-water species), or in areas that host diverse, unassessed, or poorly regulated fisheries, as is common offshore.	Worm et al., 2006; Proud et al., 2017; Beger et al., 2020
4b. If 4a can be implemented, represent at least 20–30% of marine bioregional transition boundaries in no-take MPAs	Bioregions defined at an appropriate scale	Boundaries and transition zones between bioregions in the open ocean tend to aggregate a high diversity and density of open ocean species. Bioregions in the open ocean are often much more extensive than in coastal marine habitats.	Hyrenbach et al., 2000; UNESCO, 2009; Block et al., 2011; Clark et al., 2011; Reygondeau et al., 2012; Kanaji et al., 2017

(Continued)

TABLE 2 | Continued

Design guideline	Conditions	Rationale - summary	References
5. Distance between no-take MPAs should be between 20 and 200 km.		Because of the wide-ranging or widely distributed nature of offshore populations, genetic connectivity is possible across very large areas. However, as the bulk of the population is usually less mobile, MPAs to ensure demographic connectivity will need to take into account the mean or median distances found in tagging studies (see also Guidelines 1 and 7).	Clark, 1996; Hampton and Gunn, 1998; Lauck et al., 1998; Kingsford and Defries, 1999; Sedberry and Loefer, 2001; Kohler et al., 2002; Sibert and Hampton, 2002, 2003; Worm et al., 2003; Bromhead et al., 2004; Micheli et al., 2004; Clear et al., 2005; Alpine and Hobday, 2007; Green and Mous, 2007; Theisen et al., 2008; Holdsworth et al., 2009; Cosgrove et al., 2010; Kahng et al., 2010; Sepulveda et al., 2010; Green et al., 2014; Maxwell et al., 2014; Schaefer et al., 2014; Hilário et al., 2015; Hillman et al., 2018; Gary et al., 2020
6. Include whole features within no-take MPAs.	Features are mapped	Mapped features of the open ocean are often areas of high productivity, diversity or significant ecological processes, and need to be protected in their entirety to allow for the full range of ecological processes to take place.	Hyrenbach et al., 2000; Sibert et al., 2000; Sibert and Hampton, 2002; Alpine and Hobday, 2007; Grober-Dunsmore et al., 2008; Sutton et al., 2008; Long et al., 2013; Garrigue et al., 2015; Rigby et al., 2019; Lecours et al., 2020
7a. Have at least three replicate no-take MPAs: within bioregions; of very large features (e.g., topographic or hydrodynamic features); and of known habitats and ecological processes.	Features are mapped	Replication of protection minimises the risk of losing all examples of a habitat, population or assemblage in the case of disturbance. Areas that remain intact or healthy may act as a refuge, and a source of larvae for the recovery of damaged areas. Replication also helps enhance representation of biological heterogeneity within poorly known habitats, as is commonly the case in the open ocean.	Maxwell et al., 2014; Rigby et al., 2019
7b. Include no-take MPAs at, at least, three points (ideally aggregation sites) along the migration path of migratory species or within the range of other highly mobile species.		Where it is not possible to protect an entire migration pathway, placing several replicate no-take MPAs at critical points along the migration route can disproportionately benefit the whole population. Replication of protection minimises the risk of encountering damaging agents (e.g., purse seiners, longliners) along the entire route.	Gell and Roberts, 2002; Roberts and Sargant, 2002; Block et al., 2011; Briscoe et al., 2017
8. Choose simple shapes.		Simple shapes such as squares or "squat" rectangles maximise the area protected, reduce edge effects and make compliance easier.	Halpern, 2003; Halpern and Warner, 2003; Roberts et al., 2010; Fernandes et al., 2012; White et al., 2012; Rodríguez-Rodríguez et al., 2016
9. Choose permanent protection over temporary protection.		Permanent protection enhances the likelihood of recovery of populations and habitats, even if they are very long-lived, slow-growing or heavily damaged. However, MPAs should be subject to review over time.	IUCN-WCPA, 2008; Williams et al., 2010b; Fernandes et al., 2012; Abesamis et al., 2014
10. Reduce or eliminate threats across the entire MPA network area, e.g., by applying other types of marine managed areas.		Reducing threats to other categories of MPAs and to surrounding areas will enhance the effectiveness of no-take MPAs and the area as a whole. Given the data-poor nature of the open ocean, threat reduction in general can protect areas, features or species not yet identified as requiring protection.	Dunn et al., 2011; Jessen et al., 2011; Brock et al., 2012; Maxwell et al., 2014; Lewis et al., 2017

the replenishment of fisheries target species through "spillover" (Russ, 2002; Jones et al., 2007; Fernandes et al., 2012; Harrison et al., 2012). However, larger areas can hold larger parts of (or entire) populations, and have a greater chance of including unknown habitats and species, bioregions, or special features, and tend to have a degree of biological integrity. Larger areas are more likely to be self-sustaining and therefore will persist over time (Gaines et al., 2010). Larger MPAs also reduce the edge effect, where human activities at the edges of an MPA, including illegal entry and take within MPA boundaries, can be intensive enough to undermine the MPA's overall effectiveness (Lester et al., 2009). The size of an MPA needs to be determined according to the extent and location of the species, features, bioregions, and ecological processes it is intended to protect (Green et al., 2014). Recent research has provided design guidelines for no-take MPAs based on known home ranges or distributions of shallow-water species of interest (Green et al., 2014). For instance, a no-take MPA designed to protect coral reef invertebrates and site-attached fishes could be as small as 400 to 1,000 m across, while an MPA of more than 20 km would be required for offshore pelagic species such as silvertip sharks (*Carcharhinus albimarginatus*) or trevallies (Carangidae; Jones et al., 2007; Green et al., 2014).

Coastal assemblages and sedentary oceanic species can benefit from smaller MPAs, but larger, more mobile and migratory species (as more often found offshore) require larger MPAs. In offshore environments, there is less information about habitats and bioregions. Additionally, habitats tend to be larger (e.g., deep-sea plains and plateaux compared to shallow reef systems) and many offshore species have greater home ranges and larval dispersal patterns (Herring, 2002). The larger information gaps and scale of habitats means that size becomes even more important for habitats protected within MPAs to have sufficient integrity (Shanks, 2009; UN, 2015; Lewis et al., 2017; Weeks et al., 2017). With ongoing and escalating discoveries of important new species in offshore environments, larger MPAs also provide greater insurance with regard to protecting that which remains to be discovered (Bridge et al., 2016). Huvenne et al. (2016) found that a deep-water (∼1,000 m) no-take MPA of at least 30–40 km in diameter adequately protected deep-water coral communities, but where these corals were damaged, even these protected areas could not mediate recovery. Roberts et al. (2010) suggested that in English EEZ continental shelf waters beyond 12 nm, MPAs that are intended to protect commercial species should be at least 30 to 60 km in their minimum dimension. MPAs of >100–1000,000 km^2 have been recommended for the protection of large sharks and rays whose home ranges extend beyond coastal areas (Rigby et al., 2019). In offshore pelagic and benthic habitats, the distributions of many soft-sediment (e.g., bivalves, elasipod holothurians) and pelagic taxa (e.g., tuna, lanternfishes) cover entire ocean basins, and many species are widely dispersed (McClain and Hardy, 2010; Reygondeau et al., 2012), even species with a sedentary adult phase and restricted habitat preferences, such as the mussel *Bathymodiolus thermophilus* at hydrothermal vents (Maas et al., 1999). Another benefit of larger MPAs is that in protecting the range, or part thereof, of a migratory or highly mobile species, they automatically also protect a large array of other species and features (Wilhelm et al., 2014).

The movement distance of marine organisms poses one of the greatest challenges to MPA design. The dispersive larval stage and sometimes far-ranging movements or migrations of juveniles or adults, differences in larval duration and metapopulation dynamics mean that it is highly unlikely for individual offshore MPAs to protect all life history stages of any one species, let alone all species (Gruss et al., 2011). In offshore environments, and especially in the deep sea, the difficulty of capturing species' ranges is compounded by the almost complete lack of data on larval duration and behaviour traits (Hilário et al., 2015). The dispersal of deep-sea organisms presents the added complexity of vertical swimming behaviour (Afonso et al., 2014), which can influence modelled dispersal distances by up to an order of magnitude (Gary et al., 2020; see also Guideline 5). If it is impossible to contain a species' entire range within one MPA, MPA networks that comply with connectivity guidelines (Guideline 5), replication (Guideline 7) and minimum percentage guidelines (Guidelines 3 and 4) can be combined to protect as many of the species' critical areas as possible, thereby achieving the best possible outcome for a species or population.

The potential mobility of species may conflict with their tendency for residency within a geographic location; many highly mobile species with the ability to travel 100s or 1000s of kms have smaller home ranges (10s of kms) once they settle. A tagging study of several pelagic species (tuna, billfishes, sharks) showed that most of them remained within the boundaries of the 450 km-radius British Indian Ocean Territory MPA (Carlisle et al., 2019). The evolutionary selection for behavioural polymorphism (Kaplan et al., 2014) is highlighted in the work of Mee et al. (2017). This genetic modelling research has shown an evolution of increased residency for highly mobile tuna species after the establishment of MPAs, as individuals that choose more sedentary behaviour pass on their genes to successive generations more frequently than those that move beyond MPA boundaries into fishing grounds (Mee et al., 2017). The model remains to be tested, but in a practical sense, this means that the benefits of offshore MPAs will grow over time, including over generations of the target species of interest.

Where documents provided MPA design guidelines with minimum size recommendations, these were highly variable, both for coastal and offshore environments (**Table 3**). In Edgar et al. (2014), the largest benefits were found in MPAs that were at least 100 km^2. In coastal areas, the most common minimum size, and also the upper limit, was 20 km in diameter; no minimum size was found for offshore MPAs, except in Dunn et al. (2018), where the authors recommended 200 km. In practise, almost all existing offshore MPAs are larger than 2,500 km^2 (Marine Conservation Institute, 2020), suggesting that a minimum diameter of 50 km is feasible.

In summary, the existing literature recommends that offshore MPAs (as part of a network of MPAs) be 50–200 km in minimum diameter.

Offshore Guideline 2: Include Special, Unique, Rare Features and/or Species

Sites may be selected for inclusion within an MPA according to criteria such as uniqueness, rarity, or special characteristics. These attributes include areas that are important for particular life stages of species, the presence of threatened, endangered or declining species or habitats, keystone species, distinctive habitat types, oceanographic or geological features, or places of especially high biological productivity or diversity (Salomon et al., 2006; Brock et al., 2012; Clark et al., 2014; Secretariat of the Convention on Biological Diversity, 2014; **Table 4**). For example, a site may be unique because there is a single population of an endemic species not found anywhere else. Special characteristics can be attributed to sites where key processes take place (e.g., spawning and feeding grounds, nurseries, migratory corridors, hotspots, etc.; Rigby et al., 2019). Sites can also be selected on the basis of hosting higher productivity than the surrounding areas; these "hotspots" can support high biodiversity, which is often also used as a criterion for selecting sites for inclusion into MPAs or MPA networks (Possingham and Wilson, 2005; Sydeman et al., 2006; Briscoe et al., 2016). Areas that host a large variety of species are important for the maintenance of resilience, evolutionary potential and ecosystem services (Worm et al., 2006).

The inclusion of critical habitats and special or unique areas as a design guideline for MPAs stems from the biophysical

TABLE 3 | Summary of specific and quantitative MPA network design guidelines.

Size	Overall % target	Bioregions	Habitats	Special, unique areas	Connectivity	Replication	Duration of protection	Shape	Inshore/ offshore	References
≥10 km in length (10 x 10 km or 100 km^2)	–	–	30% of each habitat	–	50–200 km apart	≥ 3 per habitat	> 25 years	–	Inshore	Munguia-Vega et al., 2018
≥2 km in diameter	20% of fishable waters in northern Honduras	–	20% of each habitat per ecoregion	Protect all target species		≥ 3 per habitat	Permanent	Compact	Both	Chollett, 2017
≥10–20 km in diameter	–	–	20–30% of each habitat	–	15–20 km apart	≥3 per habitat	–	Simple	Inshore	McLeod et al., 2009
Mixture of small (40ha) and large (4–20 km across)	~35% of a given area	–	20–30% of each habitat	–	1–20 km apart	≥3 per habitat	20-40 years, preferably permanent	Simple	Inshore	Fernandes et al., 2012
10–60 km in diameter	20–35% of fishing area	–	20-30% of each habitat	–	20–200 km apart	2–5 per habitat	–	Regular shape, minimise edge	Both	Burt et al., 2018
≥20 km diameter except in coastal bioregions	33% of the Great Barrier Reef Marine Park	20% of each bioregion	Habitat-specific % targets	As much as possible	–	3–4 per bioregion	–	–	Inshore	Fernandes et al., 2009
≥5–20 km in diameter	10–50% of marine and coastal areas	–	–	–	50–100 km apart	1–5 examples	–	–	Both	Lundquist et al., 2015
≥20 km diameter except in coastal bioregions	33% of the Great Barrier Reef Marine Park	20% of each bioregion	Habitat-specific % targets	As much as possible	–	3–4 per bioregion	–	–	Inshore	Fernandes et al., 2005
≥100–200 km^2	–	–	–	–	100 km apart	–	–	–	Inshore	Rachor et al., 2001
–	–	–	Proportional to the % of an area to be included in MPA		–	–	–	–	Inshore	Roberts et al., 2003
–	20% (Honduras), 10% (Mexico) of territorial waters	–	20-30% of each habitat	Yes	–	≥ 3 per habitat	> 20-40 years or permanent	-	Inshore	Green et al., 2017
>150,000 km^2	10% globally	–							Offshore	Lewis et al., 2017
–	10% of territorial waters	30% of each bioregion	–	–	Spacing of up to 200km	≥3 per bioregion	–	–	Inshore	The Ecology Centre, 2009
Variable minimum sizes, 0.5–20 km across	–	–	20–40% of each habitat	Yes	Spacing of 1–15 km	≥3 per habitat	>20–40 years or permanent	–	Inshore	Green et al., 2014
23–100 km^2	Test of 10, 20, and 30% conservation target	–	10–30% of each habitat	–	Spacing of 50–100 km	≥2 per habitat	–	–	Inshore	Arafeh-Dalmau et al., 2017
23–100 km^2	–	–	10–30% of each habitat	–	Spacing of 50–100 km	≥2 per habitat	–	–	Inshore	Saarman et al., 2013
–	–	–	–	–	–	–	–	Squares or compact rectangles	Inshore	Meester et al., 2004

(Continued)

TABLE 3 | Continued

Size	Overall % target	Bioregions	Habitats	Special, unique areas	Connectivity	Replication	Duration of protection	Shape	Inshore/ offshore	References
Various, 0.5 km in diameter to 550 km^2	20% of Canadian Northern Shelf bioregion	30% of each bioregion	Various, range 10–40%	–	Various, from 50 to 200 km	1–5 per habitat	–	–	Inshore	Ardron et al., 2015
Minimum length of 200 km along the ridge line	30–50% of total management area	–	30–50% of habitats	100%	–	5 per habitat	–	–	Offshore	Dunn et al., 2018
Variable minimum sizes, 0.5–20 km diameter	–	–	20-40% of each habitat	Yes	Spacing of 1–15 km	≥3 per habitat	>20–40 years or permanent	–	Inshore	Green et al., 2014
Minimum length of 5–10 km diameter, preferrable 10–20 km	10% of New Zealand waters	–	–	–	Spacing of 50–100 km	≥3	–	–	Both	Thomas and Shears, 2013
–	33% of management area	30–40% of each bioregion	20% of each habitat	–	Spacing of 10–20 km, 30 km at most	3–4 per bioregion	–	–	Inshore	McCook et al., 2009
–	Globally 10% by 2020, 30% by 2037 and 50% by 2044	–	–	–	–	–	>21 years	–	Both	Duarte et al., 2020
–	–	–	20% of each habitat	Yes	–	≥3 per habitat	>20–40 years or permanent	Compact shapes	Both	Rigby et al., 2019

Quantitative targets are provided with as much specificity as they appear in the literature. Size usually refers to the minimum diameter of any individual MPA.

TABLE 4 | Some potential topographically or hydrographically unique, special or rare features of the open ocean.

Type	Feature	Characteristics	Key sources
Topographic	Seamounts, knolls, hills, guyots, ridges	Seamounts are "large isolated elevation(s), greater than 1,000 m in relief above the sea floor, characteristically of conical form"; knolls, hills, and guyots are slightly lower elevations of different shapes. Ridges are defined as "elongated narrow elevation(s) of varying complexity having steep sides, often separating basin features." Seamounts and ridges have steep slopes which can cause the upward movement of nutrients from the deep ocean (upwellings) and create hotspots of pelagic productivity and biodiversity, attracting deepwater and pelagic species such as tuna, deep-water snapper, sharks, whales, and dolphins.	Morato and Clark, 2007; IHO, 2008; Harris et al., 2014
	Canyons, trenches	Submarine canyons are steep-walled valleys with V-shaped cross sections. A trench is a long, narrow, usually very deep and asymmetrical depression of the sea floor, with relatively steep sides. Ocean trenches are the deepest parts of the ocean, commonly 6 to 10 km in depth. The steep walls of these features tend to create upwellings that support high productivity and biodiversity. Deep-diving pelagic species tend to congregate in the waters above these depressions to feed.	Shephard, 1964; IHO, 2008; Harris and Whiteway, 2011
	Shelf breaks	The shelf break is "the line along which there is a marked increase of slope at the seaward margin of a shelf." Shelf breaks can form fronts in the waters above them, and tend to be highly productive pelagic habitats.	Belkin et al., 2009; Harris et al., 2014
	Reefs, islands	Oceanic reefs and isolated islands can form as rises and pinnacles from the deep seabed and break the ocean surface. In their wake, there are often turbulent areas and eddies that entrain plankton and attract larger pelagic species. The deep slopes off the islands and reefs support rich benthic communities that are often habitat for feeding and breeding.	Rissik and Suthers, 2000
Hydrographic	Eddies	Eddies are vortex-like circulations of water, usually spinning off major currents, and can occur at various scales. Mesoscale eddies (typically less than 100 km across) tend to be predictable, and can revolve in cyclonic or anti-cyclonic directions, depending on hemisphere. Anticyclonic eddies accumulate organic matter within their cores and exhibit elevated microbial respiration and heterotrophic production. Cyclonic eddies enhance nutrient inputs to the surface ocean increasing new production and chlorophyll concentration. Current estimates suggest that ∼50% of the global new primary production may be caused by eddy-induced nutrient fluxes.	Baltar et al., 2010
	Fronts	A front is a narrow zone of abrupt change in water properties (salinity, temperature, nutrients, etc.) that separates broader areas with different water masses or different vertical structure. They can be a few metres or many thousands of km long. Most fronts are almost stationary and seasonally persistent. The vertical extent varies from a few metres to more than 1 km, with major fronts reaching depths exceeding 4 km. Major thermohaline fronts are associated with fronts in other properties, such as nutrients, ocean colour, chlorophyll, and turbidity. Convergences of surface waters toward fronts contribute to elevated primary production known as "hot spots" of marine life, from phytoplankton to apex predators, and serve as spawning, nursing, and feeding areas for fish, sea birds, and marine mammals, with high biodiversity. The surface convergence can also lead to concentrations of pollutants, thus endangering species frequenting the fronts.	Belkin et al., 2009
	Upwellings and downwellings	Upwelling is a process in which deep, cold water rises toward the surface, usually bringing nutrients from deeper pelagic layers and from the benthos to the upper layers. Downwelling is sinking of accumulated high-density material beneath lower density material, such as colder or saline water beneath warmer or fresher water. Downwelling occurs when warm surface water spins clockwise, creating surface convergence and pushing surface water downwards.	Saldivar-Lucio et al., 2016

operational principles for the Great Barrier Reef Marine Park Authority (2002) and the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) List (OSPAR Commission, 2008), and was included to ensure that special, unique areas were protected even if they were not captured by the protection of percentages of habitats or bioregions (see Guidelines 3 and 4; Great Barrier Reef Marine Park Authority, 2002). It has since been adopted throughout the literature for designing coastal MPAs (Fernandes et al., 2009; Green et al., 2014; **Table 3**); seven of the 23 documents (30.4%) that listed MPA design guidelines included this principle. Currently, it is listed as one of the steps for marine spatial planning processes adopted by multiple Pacific Island countries (Ceccarelli et al., 2018a).

Criteria for selecting Ecologically and Biologically Significant Areas (EBSAs) in offshore environments have already been developed for some regions, such as the Azores and the Southwest Pacific (CBD, 2009, 2014; Clark et al., 2014). Other initiatives that have established criteria for protecting marine environments specifically in offshore areas include the FAO's Vulnerable Marine Ecosystems, which seeks to identify and protect marine areas in the high seas that are vulnerable to deep-sea fisheries (FAO, 2019), and the IMO's Particularly Sensitive Sea Areas, which seeks to identify ecologically, socioeconomically or scientifically valuable areas vulnerable to damage by international shipping activities (IMO, 2006). Some countries have also decided to describe special, unique marine areas within their national boundaries using systematic criteria for their identification and definition (Ceccarelli et al., 2018d). The protection of special features was the first principle used in the creation of the UK's offshore MPA network (Chaniotis et al., 2020).

Prioritising special features provides some insurance against the declaration of very large offshore MPAs in areas that are of little value to commercial interests (Devillers et al., 2020). In fact, to safeguard against the declaration of large MPAs in areas of little value, this guideline could potentially be adopted first—the scale of many offshore features and migratory or mobile species ranges makes it highly likely that MPAs designed around Guideline 2 would automatically also be large. In offshore waters, uniqueness, rarity or special characteristics typically include current systems and fronts, upwellings, seamounts, trenches, deepwater coral or sponge assemblages, hydrothermal vents and fluid seeps (Hyrenbach et al., 2000; Graham et al., 2011; Hooker et al., 2011; Ban et al., 2014b; Lundquist et al., 2017; Barrie et al., 2020; **Table 4**). The Darwin Mounds in UK offshore waters, for example, is an area rich in the deep-water coral *Lophelia pertusa* which, once discovered, was deemed of special importance and protected from trawling, becoming the UK's first offshore MPA (De Santo and Jones, 2007). These features are usually unique to a certain area and isolated from other similar features or populations by sheer distance. The value of unique and/or special features or areas stems from the fact that they are not usually replicated elsewhere and therefore not replaceable (Salomon et al., 2006), and they contribute disproportionately to marine biodiversity and ecosystem function (Lundquist et al., 2017). Their loss results in a reduction in overall biodiversity or abundance of important species (Halpern et al., 2007; Palumbi et al., 2008). For special and/or unique sites or features that may be subject to particular stressors, it is important to understand the spatial distribution of potential stressors or impacts (Halpern et al., 2007; Brock et al., 2012). Any destructive activities taking place within the area should be prohibited (see also Guideline 10). The larger the spatial scale at which special or unique features typically occur, the greater the effect of their loss.

Unique or special species and populations in the open ocean have life histories and adaptations specific to the pelagic or deep benthic habitats they inhabit. Some large pelagic species may range very widely, while deep-dwelling species may have populations that are endemic or genetically disjointed due to the distance between suitable benthic habitats (e.g., seamounts or hydrothermal vents separated by large expanses of seafloor) (Richer de Forges et al., 2000). Despite the wide-ranging nature of many individuals within populations, large pelagic species of conservation interest regularly use particular sites and migration corridors that can be mapped, monitored, or predicted (Ceccarelli et al., 2017, 2018c,d).

Geomorphic features that are known to aggregate life could all be seen as special; mid-ocean ridges, seamounts, and submarine canyons cover only four percent of the seafloor, making them rare biodiversity hotspots within the vast extent of abyssal plains, hills, plateaus, basins, terraces, troughs, valleys, escarpments, and sedimented slopes that, according to current knowledge, tend to be more sparsely populated (Glover and Smith, 2003; **Table 5**). Many of these features are considered individual habitats or habitat types, and may be seen as covered by Guideline 3 (representation of habitats), which is useful when very little or nothing is known about a particular feature or habitat. For example, if a series of ridges are known to exist within an offshore area, with little or no information about their particular attributes, they would be protected under Guideline 3.

*In sum, where knowledge exists about areas that contain special, unique, rare features and/or species in offshore environments, these areas should be included, in their entirety, in the MPA network (see **Table 4**).*

Offshore Guideline 3: Representation of Habitats

Any network of no-take MPAs, inshore or offshore, should include representation of every known habitat type and bioregion (see Guideline 4) to ensure that as many species as possible are protected (Gilman et al., 2011; Day et al., 2012; Fernandes et al., 2012). For habitats, adequate representation requires that they are mapped and that habitat-specific "minimum amounts" of protection can be defined (see, for example, Great Barrier Reef Marine Park Authority, 2002). The concept of "representing" habitats (as opposed to bioregions or other ways of classifying the environment—see Guideline 4) is the most common specific principle or guideline in the MPA design literature, appearing in 44 of 52 papers (88%). Per-habitat protection percentage levels are often suggested to be 10–30% for coastal seas, and higher in the following areas: (1) areas with less existing management of activities outside the no-take MPA; (2) areas with more destructive activities; or (3) areas where marine bioregions are not defined (see Guideline 4).

Most documents that had percentage targets for coastal habitat protection agreed that including 20–30% of each habitat in no-take MPAs would be sufficient for biodiversity conservation and the protection of fisheries stocks (**Table 3**). Support for the 20–30% target was originally gleaned from reproductive theory, knowledge about the vulnerability of coral reef species to exploitation, analysis of fishery failures, empirical, and modelling studies of reserves and the precautionary principle (Bohnsack et al., 2000). These targets were then used and sometimes modified (down to 10% or up to 50%) for designing MPA networks in the Great Barrier Reef Marine Park (Fernandes et al., 2009), the California Channel Islands (Airamé et al., 2003), Honduras (Chollett, 2017), across the Mesoamerican Reef (Green et al., 2017), and in the Coral Triangle (Fernandes et al., 2012).

Global adherence to Guidelines 3 and 4 in the context of MPA design has been assessed by Fischer et al. (2019), who found that only 18 of 66 Large Marine Ecosystems (LMEs) contained greater than 10% of the marine geomorphic features and benthic habitats (listed in **Table 5**) within existing MPAs; hence MPAs in 48 out of 66 LMEs do not comply with the guidelines, even using this 10% requirement, which is at the lower end of the representation range. The OSPAR Convention Guidelines, used to designate deep-water MPAs in the UK's EEZ, include the principle of representation of habitats, and this led to the protection of representative examples of seamounts, canyons, deep-water coral mounds, and other features (Chaniotis et al., 2020). Representation of habitats was also one of three principles (along with comprehensiveness and adequacy) used for the designation of offshore MPAs in the Australian EEZ under the National Representative System of Marine Protected Areas Program (ANZECC, 1996; Commonwealth of

TABLE 5 | Major habitats of open ocean environments and suggested minimum proportions for inclusion in no-take MPAs if Guideline 4 applies (representation of bioregions).

Habitat	Definition	Suggested minimum % for no-take MPAs
Shelf valleys	Valleys incised more than 10 m into the continental shelf, greater than 10 km in length.	10%
Coral reefs beyond the continental shelf	(Oceanic context) A ridge of calcium carbonate rock in the sea formed by the growth and deposit of coral, surmounted by a living coral reef and rising directly from deep water.	25%
Oceanic islands beyond the continental shelf	(Oceanic context) A ridge of rock in the sea, rising directly from deep water, usually at the apex of a seamount or pinnacle.	25%
Basins (of various sizes, of seas and oceans, perched on the continental shelf, plateau or slope)	A depression in the sea floor of variable extent.	10%
Shelf, slope, abyssal and hadal sills	A sea floor barrier restricting water movement between basins.	20%
Slope terraces	An isolated (or group of) relatively flat horizontal or gently inclined surface(s), sometimes long and narrow, which is (are) bounded by a steeper ascending slope on one side and by a steeper descending slope on the opposite side.	10%
Slope, abyssal and hadal escarpments	An elongated, characteristically linear, steep slope separating horizontal or gently sloping sectors of the sea floor in non-shelf areas.	10%
Seamounts (of various types, rising from all depths)*	A discrete (or group of) large isolated elevation(s), greater than 1,000 m in relief above the sea floor, characteristically of conical form.	20% of each seamount type*
Canyons (shelf incising, connected to river systems)	Steep-walled, sinuous valleys with V-shaped cross sections, axes sloping outwards as continuously as river-cut land canyons and relief comparable to even the largest of land canyons. Shelf incising canyons have heads that cut across the shelf break, and in which there are landward-deflected isobaths on the continental shelf, and there is a clear bathymetric connexion to a major river system.	10%
Canyons (shelf incising)	Steep-walled, sinuous valleys with V-shaped cross sections, axes sloping outwards as continuously as river-cut land canyons and relief comparable to even the largest of land canyons. Shelf incising canyons have heads that cut across the shelf break, and in which there are landward-deflected isobaths on the continental shelf, without a bathymetric connexion to a major river system.	10%
Canyons (blind)	Steep-walled, sinuous valleys with V-shaped cross sections, axes sloping outwards as continuously as river-cut land canyons and relief comparable to even the largest of land canyons. Blind canyons are those which have heads that are wholly confined to the slope, below the depth of the shelf break.	10%
Ridges	An isolated (or group of) elongated narrow elevation(s) of varying complexity having steep sides, often separating basin features.	10%
Troughs	A long depression of the sea floor characteristically flat bottomed and steep sided and normally shallower than a trench.	10%
Trenches	A long narrow, characteristically very deep and asymmetrical depression of the sea floor, with relatively steep sides.	15%
Bridges	A geomorphic “bridge” across troughs or trenches; they may partially infill trenches and troughs.	10%
Fans	A relatively smooth, fan-like, depositional feature normally sloping away from the outer termination of a canyon or canyon system	10%
Plateaus	Flat or nearly flat elevations of considerable areal extent, dropping off abruptly on one or more sides.	15%
Epipelagic zone	The first 200 m of open ocean, where planktonic primary producers receive enough light for photosynthesis, and therefore form the basis of the food web.	20–30%
Mesopelagic zone	From 200 to 1,000 m, primary production is replaced by sinking organic matter (marine snow), including plankton, as the primary food source.	20–30%
Bathypelagic zone	Between 1,000 and 4,000 m there is no sunlight penetration, and conditions in any one location are relatively stable and uniform.	20–30%
Abyssopelagic zone	From 4,000 to 6,000 m is an area of immense pressure and very low temperature.	20–30%
Hadopelagic zone	This habitat occurs in ocean trenches, below 6,000 m, to a maximum depth of ∼11,000 m in the deepest parts of the ocean, the Marianas and Tonga Trenches.	20–30%
Any other habitats		20–30%

*Habitat names and definitions adapted from Harris et al. (2014), definitions from Harris et al. (2014) and IHO (2008). Updated from Ceccarelli et al. (2018b), and based on biophysical operational principles from the literature (see **Table 3** and Guideline 3). *Seamount types further classified as per Macmillan-Lawler and Harris (2016).*

Australia, 2003). There is therefore a strong precedent for the use of this guideline in offshore environments (see also **Table 3**).

Offshore environments have a multitude of static, recurring and ephemeral habitats, both benthic and pelagic, that can be mapped and used for spatial planning (Hyrenbach et al., 2000;

Roberts et al., 2003; Belkin et al., 2009; Harris et al., 2014; Miller and Christodoulou, 2014). These habitats occur at a variety of scales and harbour different levels of diversity; for example, expanses of relatively homogeneous and low diversity basins or plains are very different from much smaller features, such as seamounts, which may nevertheless host higher concentrations of life. To a large extent, we still lack the knowledge to differentiate similar-looking open ocean habitats from one another. But we do know, for example, that not all seamounts are equally productive and diverse (Samadi et al., 2006). Identifying the location and mapping the extent of offshore habitats still largely relies on proxies; habitats may be identified by analysing the foraging distribution of higher predators (Hyrenbach et al., 2000; Patterson et al., 2016; Hobday et al., 2017; Queiroz et al., 2017), by making use of sophisticated real-time satellite imagery (Game et al., 2009), by using maps of seabed geomorphology (Harris and Baker, 2012; Harris et al., 2014; Beaman et al., 2016), oceanographic attributes or some combination of the above and other methods. In the context of spatial planning, lessons learned from general design guidelines are more difficult to apply to offshore waters, given the biophysically dynamic nature of pelagic seascapes (Kavanaugh et al., 2016); static geological habitats are more straightforward for MPA design (**Table 5**).

Given the relatively data-poor status of most offshore habitats, and because marine bioregions are not usually defined at a useful scale, (and therefore Guideline 4, below, cannot be applied), research suggests that representing 30–40% of each habitat in offshore no-take MPAs enhances the likelihood of capturing unknown, and therefore unmapped, within-habitat variability, and even unknown features (O'Leary et al., 2016b; **Table 3**). Where there is some knowledge about offshore marine bioregions at a scale useful within countries' EEZs and in ABNJ, Guideline 4 could be applied first, and subsequently 10–30% of each offshore habitat can additionally be represented in no-take MPAs as per **Table 5**.

In sum, the literature recommends that (a) where bioregions are not defined, 30% of each habitat should be included in no-take MPAs; and (b) where bioregions are defined, ensure that 10–30% of each offshore habitat is represented in no-take MPAs.

Offshore Guideline 4: Representation of Bioregions

Using surrogates for patterns of biodiversity during spatial planning allows for MPAs to capture close to 100% of the diversity of marine life within a given area, despite imperfect knowledge, and while requiring much less than 100% coverage of the geographic area (Foley et al., 2010; Bridge et al., 2016). Bioregions are commonly used surrogates that define areas with relatively similar assemblages of biological and physical characteristics, without requiring complete data on all species, habitats and processes (Spalding et al., 2007; Costello et al., 2017). Protecting an adequate proportion of bioregions within no-take MPAs helps to manage for the uncertainty associated with habitat and species distributions, and thus reduces the risk of overexploitation of marine populations in areas that remain open to extraction (Botsford et al., 2003; Gaines et al., 2010; Wilson et al., 2011; Day et al., 2012; Fernandes et al., 2012; Green et al., 2013; Ballantine, 2014).

Of the 52 studies that listed MPA design guidelines or principles, 15 (29%) included the representation of bioregions, indicating that habitats (included in 88% of studies) are more commonly understood than bioregions, even in coastal seas. Among studies that provided numeric guidance, only five included proportions of bioregions, and only in coastal areas, whilst 22 gave percentages of habitats to be included in no-take MPAs (**Table 3**). In coastal ecosystems, the best available evidence advises that at least 20–40% of each bioregion should be included in no-take MPAs or MPA networks to ensure that representative examples of marine biodiversity are captured (see also Guideline 3; **Table 3**). The percentage of each bioregion to be included in MPAs should be increased in areas experiencing less management generally (e.g., poor or absent fisheries management), or subject to more destructive activities.

The logic pertaining to bioregion guidelines is equally applicable to offshore environments. Management of the ocean from a biodiversity protection point of view is usually undertaken within the EEZ of individual countries, but most current bioregionalisations span many countries and are too coarse to undertake planning at a national level (UNESCO, 2009; Clark et al., 2011; O'Hara et al., 2011; Reygondeau et al., 2012; Watling et al., 2013; Proud et al., 2017; Sayre et al., 2017; Sutton et al., 2017). Finer-scale marine bioregions need to be described to support national planning processes (Etnoyer et al., 2004; Reygondeau et al., 2012; Mannocci et al., 2015; Proud et al., 2017). Recently, offshore marine bioregions have been defined at an appropriate scale in some parts of the global ocean; that is, they are described at a scale useful to the area being managed. For example, multiple offshore marine bioregions have been rigorously described within and beyond national jurisdictions within Southwestern Pacific Island countries (Wendt et al., 2018; Beger et al., 2020), Canada (e.g., Arafeh-Dalmau et al., 2017), and Australia (Fernandes et al., 2005; Department of the Environment and Heritage, 2006). Delineating bioregions at an appropriate scale allows for their use in ensuring representation of the range of offshore biodiversity in national-scale MPA design. Guideline 4 therefore applies only to jurisdictions or ABNJ where marine bioregions have been described at such an appropriate scale (Gilman et al., 2011); for other jurisdictions or ABNJ, see Guideline 3.

In sum, the literature recommends (a) the protection of 20–40% of each bioregion within no-take MPAs, or (b) where areas outside the MPA are subject to destructive activities or a lack of management, the percentage should increase.

Offshore Guideline 5: Space MPAs for Maximum Connectivity

Connectivity within a network of MPAs is important because it ensures that if a population vanishes or a habitat is damaged in one MPA, it can be restored through the movement of larvae or adults from another MPA, or an undamaged habitat upstream (Jones et al., 2007; Hilário et al., 2015). Genetic connectivity (genetic exchange among individuals within and between populations) depends on the absolute number of dispersers among populations, whereas demographic connectivity (exchange of individuals between spatially separate populations) depends on the relative contributions to population growth rates of dispersal vs. local recruitment (i.e., survival and reproduction of residents) (Lowe and Allendorf, 2010). Demographic connectivity, which influences recruitment levels, occurs over smaller scales than genetic connectivity. From a genetic standpoint, connectivity ensures genetic diversity within populations, which in turn ensures population persistence and evolutionary potential (Jones et al., 2007).

Connectivity and spacing of MPAs in a network are included in 28 (54%) of the 52 studies that explicitly discuss MPA design guidelines or principles. In a functioning marine ecosystem, populations or patches of similar habitat that are geographically separate are linked through the movement of organic and inorganic matter, nutrients, energy, larvae, juveniles and adults (Cowen et al., 2007; Brock et al., 2012; Worboys et al., 2016; Hillman et al., 2018). Larval connectivity within an MPA network can occur between MPAs that are from 1 to 200 km apart (**Table 3**), depending on the species, with inshore species generally connected over smaller scales than offshore species (Jones et al., 2007; Shanks, 2009; Gilman et al., 2011; Harrison et al., 2012; Green et al., 2014). Larval connectivity research on coastal coral reef fishes suggests that dispersal is a declining function with distance, with many larvae settling in or close to their natal reefs, and fewer travelling 10s to even 100s of kilometres away (Harrison et al., 2012; Almany et al., 2013, 2017; Williamson et al., 2016; Abesamis et al., 2017; Bode et al., 2019). On the Great Barrier Reef, reserves are commonly less than 15km apart, which is clearly well within the dispersal range for most coral reef organisms (Almany et al., 2009). A wide range of reserve spacings have been recommended, including < 100 km apart (Sala et al., 2002), < 20 km (Shanks et al., 2003), 10–200 km (Palumbi, 2004), 40–80 km (Roberts et al., 2010), 1–50 km (Jones et al., 2009), 1–15 km (Green et al., 2014), and 50–200 km (Munguia-Vega et al., 2018). The accumulating empirical research suggests that connectivity levels in MPA networks will be robust to variation in reserve spacings in the ranges advocated, largely because most species appear to have a long tail to their dispersal kernels (Jones et al., 2007). While there is also clear evidence that some marine larvae disperse distances in excess of 1,000 km (Manel et al., 2019), it is questionable whether reserves spaced this distance apart would offer any demographically significant connectivity.

In offshore waters, larger distances between populations or habitats make connectivity more diffuse, but fewer barriers to dispersal means that some populations are more widely distributed than inshore (Maas et al., 1999). Migratory and wider-ranging species have populations that are connected over small scales as well as over 100s, and sometimes 1,000s of kilometres (Lam et al., 2016). It has been shown that designing MPAs with a focus on connectivity, rather than just for species or habitats on their own, is especially important and has a greater chance of success in pelagic ecosystems (Moffitt et al., 2011). The scales of dispersal and connectivity for MPA design in the deep sea are larger than those in shallow water, as suitable habitats tend to be more isolated (Baco et al., 2016). As for inshore and nearshore MPAs, offshore MPAs are likely to benefit from placement that takes into account adjacent inshore or nearshore MPAs, or areas with existing protection, such as areas in which tuna fishing or the killing of sharks is already banned (Jones et al., 2007). Furthermore, in offshore waters vertical connectivity is as important as horizontal connectivity, and occurs through the downward drift of organic matter (marine snow), deep-diving ocean predators, and the vertical migration of deep-dwelling species that move toward the surface to feed at night (Sutton, 2013; Afonso et al., 2014). MPA design needs to take into account potential connectivity pathways along benthic and demersal depth gradients (Papastamatiou et al., 2015). When designing MPAs in offshore waters, it may be necessary to include MPAs that serve as "stepping stones," that play key roles in dispersal or migration, by providing resting or feeding points (e.g., the staging areas known in bird migrations). These may be otherwise unremarkable habitats, but crucial to the persistence of species of interest.

Movement occurs either passively with currents or actively, through active dispersal, movement and migration. Within networks of MPAs, movement ideally occurs between protected areas (Roberts et al., 2010), and also between protected and unprotected areas (Gaines et al., 2010). A study of larval dispersal across a number of different habitat types found that species in soft-bottom subtidal habitats have the greatest potential for extensive larval dispersal (Grantham et al., 2003). However, pelagic larval duration has been estimated for only 93 taxa that reside in depths over 200 m; deep-dwelling taxa have a range of larval durations from 2 to over 200 days (Hilário et al., 2015). The lack of knowledge about larval traits such as swimming ability (both horizontal and vertical) and larval duration is a serious impediment to predicting connectivity in offshore species (Gary et al., 2020).

Dispersal in deep-sea larvae has the added complexity of vertical swimming ability, which has a strong influence on horizontal dispersal because of the vertical layering of different currents (Gary et al., 2020). Dispersal strategies are also important; deep-sea sessile organisms such as corals can have either a dispersive larval stage or reproduce asexually, resulting in either highly connected or isolated populations, similarly to coastal species (Miller and Gunasekera, 2017; Strömberg and Larsson, 2017). However, in the deep sea isolated habitats, such as hydrothermal vents or deep-sea biogenic mussel reefs, are much more widely dispersed than inshore habitats, and are often largely self-seeding (Elsäßer et al., 2013). These discrete habitats can be captured through the application of Guidelines 2 and 3, and MPAs can therefore be sized to allow for self-replenishment and spaced at variable distances to allow for significant levels of connectivity. Greater benefits to the broader marine ecosystem are expected from MPAs that are self-replenishing, interconnected and/or important source areas for larvae (Krueck et al., 2017; Ross et al., 2017). The movement of larvae, juveniles and adults across MPA boundaries can be seen as negative because it implies a lower level of protection for individuals that move into areas where they can be exploited (e.g., Gruss et al., 2011). However, this "spillover" restores populations and target species and can therefore benefit fisheries and the broader ecosystem alike (Gell and Roberts, 2002; Harrison et al., 2012; Kerwath et al., 2013).

The maximum spacing recommended between MPAs in coastal networks is 200 km, with a wide range of distances depending on the geographic characteristics (**Table 3**). For offshore MPAs or in documents that included both coastal and offshore environments, the most common spacing recommendations were 20–200 or 50–100 km (Thomas and Shears, 2013; Lundquist et al., 2015; Burt et al., 2018). Based on these existing offshore guidelines and the current understanding of offshore connectivity, this guideline adopts the entire range (20–200 km) of existing spacing recommendations.

In summary, the existing literature recommends that offshore MPA spacing should be in the range of 20–200 km. This distance adequately encompasses the known range of dispersal distances for offshore marine species (Green et al., 2014) and acknowledges that network designs should be robust to a wide range of reserve spacings.

Offshore Guideline 6: Represent Whole Features

Some habitat areas and features (e.g., seamounts, submarine canyons, etc.) tend to function as complete entities and have a level of ecological integrity. The functioning of a habitat or feature depends on linked processes that may occur in different areas (e.g., the seamount summit vs. the slope), but are connected across the entire habitat or feature. It is therefore important to represent entire habitats or features within the same level of protection and avoid "split zoning" (Day et al., 2012; Fernandes et al., 2012; Rigby et al., 2019; Lecours et al., 2020). The concept of split zoning is not often encountered in the coastal MPA literature, as it is likely to reduce the ecological integrity of an MPA and lead to problems of public understanding and compliance (Day, 2002).

Representing whole features is equally important in coastal and offshore ecosystems. Using a seamount example, primary production and nutrient cycling that occur near the surface produce food which is then distributed to deeper areas; organisms from deeper areas may migrate vertically to feed at night (Clark et al., 2014). Therefore, protecting only part of a habitat or feature (such as a seamount) means that human impacts would still be affecting ecological communities adjacent to the no-take MPA, subjecting it to potential flow-on or indirect effects such as changes in the abundance or behaviour of organisms. Similarly, deeper parts of canyons are strongly dependent on processes from shallower areas, and vice versa. In the open ocean, habitats and features can be isolated by large expanses of deep open water (e.g., seamounts, canyons, ridges) or areas with hydrologically different characteristics (e.g., upwelling, fronts), and protecting them in their entirety becomes even more important than in inshore habitats for safeguarding ecological functions and processes.

Vertical zoning (applying different management rules to benthic and pelagic habitats of the same area) is also not recommended (Grober-Dunsmore et al., 2008; Lausche, 2011). Despite knowledge gaps around benthic-pelagic coupling (Day et al., 2012), emerging evidence suggests that it is stronger than previously thought (Grober-Dunsmore et al., 2008). Benthic communities, especially around prominent undersea features, provide food, shelter, and meeting points for pelagic species (Morato et al., 2010; Garrigue et al., 2015), which, in turn, also directly or indirectly regulate benthic communities. Passfield and Gilman (2010) show that the feeding of predators around seamounts affects seamount benthic ecology; vertical zoning would disturb this coupling. Some tuna aggregations may be present at an individual seamount for up to a period of weeks or months, resulting in a significant contribution to biological and ecological processes (Sibert et al., 2000). Similarly, bathypelagic fish assemblages have been found directly associated with ridge systems, where trophic linkages are likely to be bi-directional (Sutton et al., 2008).

The trophic influence of pelagic species on demersal and benthic communities may be largely indirect, such as large, mobile pelagic species preying on the predators of benthic prey, or preying on bentho-pelagic species (Allain et al., 2006). There is also an ontogenetic link between pelagic and benthic seamount habitats: most seamount benthic species have a pelagic stage, usually as larvae (Allain et al., 2006). Depletion of pelagic predators may therefore indirectly affect benthic communities through release from predation of certain functional groups, increasing prey species abundance and subsequently affecting their interactions with benthic species, such as occurs in trophic cascades (Estes et al., 2011). It could be argued that benthic communities become ever more dependent on pelagic species with increasing depth, as organisms in deeper waters become almost entirely dependent on marine snow and sinking carcasses of larger pelagic animals for food (Bochdansky et al., 2017).

Therefore, where possible, no-take MPAs should protect offshore features in their entirety, both horizontally and vertically.

Offshore Guideline 7: Replicate Protection of Bioregions or Habitats

The concept of replication in MPA design refers to representing each feature, bioregion, or habitat more than once, or placing multiple MPAs within a bioregion, geographic area or other feature of interest, at the scale of the area for which the MPA or MPA network is being designed. In the face of climate change, replication across environmental gradients increases the probability of survival, movement, regeneration, range shifts, or even adaptation of community assemblages and the species within them; this is just as relevant to the deep sea as to shallow-water habitats (Danovaro et al., 2017). Furthermore, representation of latitudinal or longitudinal gradients is important for capturing the range of habitat types and species compositions (Ministry of Fisheries and Department of Conservation, 2008), which are not usually organised into discrete areas, but blend into each other along such gradients.

The replication guideline is common throughout the literature (18 out of 23 documents with a list of guidelines, or 78.3%), and usually recommends protecting three of more examples of a habitat within no-take MPAs (**Table 3**). The primary goal is risk-spreading, to provide some redundancy to protect against unexpected disturbances or population collapse (e.g., Burt et al., 2018). Protecting several spatially separated examples of similar features (e.g., sites important for a population of a threatened species, patches of similar habitat, breeding sites), reduces the risk of losing the entire feature(s) of interest to disturbance, poaching or even random temporal variability (e.g., recruitment failure; cyclones; Gilman et al., 2011). Most destructive events are spatially patchy, allowing some areas or individuals to escape damage and provide a source of regeneration for damaged areas or depleted populations (Salm et al., 2006). However, while a number of papers provide modelled or empirical tests of the effectiveness of size and spacing guidelines (e.g., Edgar et al., 2014; Robb et al., 2015; Hargreaves-Allen et al., 2017), the replication guideline is yet to be explicitly tested, even in coastal MPAs.

Representing multiple examples of features or habitats in MPAs can be both easier and more problematic in the open ocean. On the one hand, larger MPAs are more feasible, which in turn increases the likelihood of encompassing multiple examples of a feature (e.g., multiple seamounts, canyons, hydrothermal vents, etc.). Also, bioregions tend to be large (e.g., O'Hara et al., 2011; Reygondeau et al., 2012), making it easier to include replicate no-take MPAs within a bioregion. On the other hand, and depending on scale, many features of interest in the open ocean are very large and some are unique (e.g., the Tonga Trench); in such cases, there are, effectively, no other features with exactly the same attributes in existence (Richer de Forges et al., 2000).

The replication guideline can also be used to protect populations of protected species along movement and migratory pathways. Migration pathways can cover entire ocean regions, making replication of the whole migration pathway impossible, but replicated sections of an individual pathway can be protected. Migratory and wide-ranging species may focus their routes over areas of high productivity, or they may rest or aggregate at particular locations (Block et al., 2011); these types of locations can also be replicated. Many populations of migratory species have only one main migration pathway (e.g., migratory seabirds, turtles that move between the western and eastern Pacific). MPA networks can therefore be designed to protect several points along each population's known migration route. Where the literature makes quantitative recommendations about replication in offshore MPAs, the numbers range from one to five replicates, but a minimum of three is the most common design guideline (**Table 3**).

In summary, where possible, including 3–5 examples of each feature, habitat or bioregion within the no-take MPA network is recommended.

Offshore Guideline 8: Use Simple Shapes

The boundaries of an MPA need to be determined according to the extent and location of the species, features, bioregions, and ecological processes they are intended to protect. Additionally, to maximise the ease of compliance, the boundaries of both inshore and offshore no-take MPAs are best placed according to parallel or perpendicular coordinates. Edges of MPAs can be subject to intense fishing pressure and fishing incursions, and therefore offer a weaker refuge than the core interior (Halpern, 2003; Halpern and Warner, 2003). Therefore, the ideal MPA shape is simple (**Table 3**) and minimises the edge effect by maximising the protected area to boundary ratio (Roberts et al., 2010; Rodríguez-Rodríguez et al., 2016). Squares or circles are considered to be the most favourable shapes to protect biodiversity; the former, or relatively "squat" rectangles, are preferable from a compliance point of view (Fernandes et al., 2012; White et al., 2012).

Simple, squarish shapes both minimise edge effects and simplify compliance.

Offshore Guideline 9: Choose Permanent Over Temporary Protection

The duration of no-take protection depends on the objectives of the MPA, but for biodiversity conservation objectives, permanent protection is recommended (Dudley, 2008), as the benefits of MPAs are known to increase measurably with age (Edgar et al., 2014). In addition, permanent protection provides time for the entire marine community to recover from human impacts as well as ensuring permanent fisheries benefit from "spillover" effects to be realised (IUCN-WCPA, 2008). Depending on the life cycle of protected species, it can take many years for populations to recover from exploitation (Russ, 2002); the re-establishment of balance and stability within a whole ecosystem can take 10 years or more even for shallow habitats (Johns et al., 2014). While seasonal, rotational or temporary closures may be beneficial for no-take areas designed for fisheries (Cinner, 2005; Kaplan et al., 2010; Sadovy et al., 2011), those benefits are quickly eroded or lost upon opening the area to fishing (Russell et al., 1998; Friedlander and DeMartini, 2002).

In the deep sea, recovery can take between three times and orders of magnitude longer (Huvenne et al., 2016; Fariñas-Franco et al., 2018; Girard et al., 2018). Large pelagic species of conservation interest and deep-water species tend to be long-lived, slow-growing and late-reproducing (K-selected life histories) compared to many of their coastal counterparts; therefore, these populations, once exploited, take longer to recover (Alcala et al., 2005; Hart, 2006). For example, the orange roughy (*Hoplostethus atlanticus*) is highly sought after by commercial deep-trawl fisheries, but its extraordinary lifespan (up to 150 years) makes it extremely vulnerable to overexploitation (Doonan et al., 2015). In the open ocean, recovery may also occur over the scale of decades, as seen, for example, in the case of the humpback whale populations after the cessation of widespread whaling (Pavanato et al., 2017). The rates of population increase of deep-sea elasmobranchs are less than half those of shelf and pelagic species; once a stock has been depleted, recovery is in the order of decades to centuries (Simpfendorfer and Kyne, 2009). Therefore, offshore ecosystems would especially benefit from permanent protection (Huvenne et al., 2016; Mee et al., 2017).

In sum, implement permanent protection of offshore networks of no-take MPAs.

Offshore Guideline 10: Minimise Threats Outside No-Take MPAs

The minimisation of threats to the marine environment as a component of MPA design was included in 12 of the 52 documents (23%) with a list of guidelines; in many cases, one of the tools proposed for threat minimisation was multiple-use zoning. Both stand-alone MPAs and networks of MPAs can allow for multiple-use zoning (Fraschetti et al., 2009), and for a proportion of each MPA to be designated as no-take (Bohnsack et al., 2004). This paper focuses on guidelines for no-take MPAs because of the conservation and compliance advantages they provide, but also due to the fact that most of the available science focusses on no-take MPAs (Edgar et al., 2014). Recognising, however, that other types of MPAs may also be useful for political, cultural or socio-economic reasons, some guidance is also given here for MPAs that allow some degree of human use. Definitions for a range of types of MPAs exist; the IUCN sets out categories for MPAs with different levels and types of permitted use (Dudley, 2008; Day et al., 2012). Zoning for different levels of use allows for the minimisation or exclusion of individual threats from a wider area (Day, 2002; Grantham and Possingham, 2011; Wilson et al., 2011).

Understanding the spatial distribution of potential stressors or impacts can provide additional guidance for the placement of other categories of MPAs (Halpern et al., 2007). The severity and extent of the stressors may also inform the percentage of an area to be included within MPAs of all categories, including no-take. MPA zoning should also be based on an understanding of cumulative impacts, which relies on the availability of both spatial and temporal data. This understanding can also help to assess the potential threats to future and existing MPAs, as well as the threats to unprotected areas. Any highly destructive activities should be prohibited within the area being managed or considered for inclusion within an MPA, regardless of zoning (Fernandes et al., 2012).

For the design of offshore MPA networks, a simplified version of the IUCN categories will be less confusing for stakeholders and easier for compliance monitoring and enforcement (Day et al., 2012). The rationale and guidelines applied to no-take MPAs should also apply, as much as possible, to other categories (Day et al., 2012). Reducing threats by the application of other categories of MPAs and other management to areas surrounding MPAs will enhance the effectiveness of no-take areas and enhance the ecological health of the management area as a whole. These threats may include shipping, fishing, mining, and other potentially destructive human uses (Halpern et al., 2007). Given the relatively data-poor nature of offshore waters, threat reduction in general can help secure areas, features, or species not yet identified as requiring spatial protection (Jessen et al., 2011).

In sum, apply management to minimise threats overall, and use globally accepted zoning categories that are recognisable to stakeholders.

DISCUSSION AND CONCLUSIONS

Many of the same design guidelines used to protect coastal regions apply in the open ocean (e.g., size, shape, distance, replication, percentages), with specific tailoring and prioritisation for the characteristics of oceanic ecosystems and species. However, designing MPAs and MPA networks in the open ocean requires a broader perspective than in coastal seas. The main differences between protecting inshore and oceanic areas are related to scale and distance, and are based on a lower level of knowledge and larger uncertainties associated with the open ocean (**Table 2**).

Next Steps: Applying the Design Guidelines

The guidelines developed in this paper are adapted for offshore environments from existing guidelines for the design and placement of inshore or coastal MPAs (**Table 2**). Whilst detailed, specific offshore MPA biophysical design guidelines have never been proposed before, the science about offshore marine environments and the effects of offshore MPAs has advanced enough for this initial set of guidelines to be developed. Additionally, some offshore MPAs have already been designed and implemented according to guidelines adapted from coastal MPA design, setting a precedent that indicates the need for a globally applicable set of offshore guidelines. These are guidelines based on current knowledge, however, and should not be interpreted as fixed targets (Agardy et al., 2003). Lessons learned from the first offshore MPAs in the UK reveal three key considerations: (1) offshore MPAs require a strong regulatory basis with integration of fisheries and conservation and a clear financial commitment to enforcement and monitoring; (2) uncertainty and the need for precautionary approaches increase with increasing distance offshore; and (3) transparency tends to be reduced, calling for greater stakeholder engagement (De Santo, 2013). The offshore MPA biophysical design guidelines in this paper have been prepared for the use of practitioners, and it is our hope that this paper may stimulate interest in further adaptations and refinements. We suggest that the next steps for operationalising the guidelines on an international level are (1) setting them into a systematic marine spatial planning framework; (2) prioritising guidelines for ease of application; (3) considering uncertainty; (4) emphasising the need for adaptive management; and (5) special considerations for monitoring in offshore environments.

Systematic Planning

Systematic marine spatial planning refers to a multi-step process that can be used to implement any network of MPAs, including offshore MPAs, and includes stakeholder consultation, application of design guidelines, strategies to incorporate uncertainty and adaptive management systems (Kelleher and Kenchington, 1992; Ehler, 2008; Ehler and Douvere, 2009; Ceccarelli et al., 2018a). This paper acknowledges that biophysical design principles for MPAs form only a small, albeit important, part of the overall marine spatial planning process. For example, the biophysical design guidelines presented here will need to be applied in concert with cultural, social, and economic considerations (Ehler and Douvere, 2009; Lewis et al., 2017). Introducing them into a broader marine spatial planning context will allow for their integrations into an existing regulatory context (De Santo, 2013). Ultimately, compliance with any MPA's restrictions will be the most important contributor to MPA success and an effective planning process will stimulate a higher degree of voluntary compliance (Edgar et al., 2014; Arias et al., 2016).

Systematic spatial planning for offshore waters has the same framework as in coastal ecosystems, but with larger areas, different scales of human operations and ecosystem functioning, higher uncertainty, and, in the case of ABNJ, international collaboration (O'Leary et al., 2012). In the absence of comprehensive information, it may be pragmatic to select some sites for MPAs based on fragmented knowledge, or scientific inference based on similar sites (O'Leary et al., 2012). Comprehensive MPA network design guidelines, such as those presented here, help counter and complement information limitations and high uncertainty by incorporating design features (e.g., minimum requirements, replication) that are robust to potential knowledge failures (Langford et al., 2009).

The first consideration identified from the UK experience suggests that the proposed biophysical guidelines need to be applied within a larger process of marine resource management, which may include other tools to managing large pelagic species, ecosystems and fisheries (Dulvy, 2013; Duarte et al., 2020). Part of broader toolbox may be multiple types of MPAs within a network within which different levels of activity can continue taking into account existing threats and endeavour to minimise them across the entire network (Day et al., 2012). Ultimately, no-take MPAs can only stop extractive uses, and must be used in conjunction with other sectoral resource management tools, pollution controls and actions to reduce greenhouse gas emissions (Hilborn, 2016; Duarte et al., 2020).

Prioritising the Guidelines

Depending on the information available, the guidelines may be more difficult to apply in different parts of the world, and especially in ABNJ where reaching international agreements can be a lengthy process. The challenge of wholly applying these guidelines in poorly understood offshore environments under pressure from unpredictable impacts (e.g., climate change) can be more easily met by prioritising at least some guidelines (Fernandes et al., 2009; Gilman et al., 2011; IUCN-WCPA, 2018). We prioritise the "top three" guidelines as follows:

1) Guideline 1 (size), because maximising the size of MPAs increases the likelihood of other guidelines being applied automatically. In data-poor systems, it also provides insurance against missing important, but as yet unknown, features (Rodrigues et al., 2004).
2) Guideline 2 (special areas), because this ensures that large-scale MPAs will not be placed in areas that are simply of less commercial interest, but actually include features or species that require protection (Devillers et al., 2020).
3) Guideline 3 (% representation of habitats) where bioregions are not defined or Guideline 4 (% representation of bioregions) where there are defined bioregions at an appropriate scale (Fernandes et al., 2009). Representativeness is prioritised because maximising the potential for representativeness also maximises the biodiversity and ecological processes that can be captured within no-take MPAs (Harris, 2007).

Guidelines 1 and 2 together offer the best precautionary approach to maximise the inclusion of offshore biodiversity in larger offshore MPAs, while capturing special features/areas and thus avoiding the protection of large areas of little commercial interest. When data are absent or limited for Guideline 2, Guidelines 3 or 4 become more important.

Dealing With Uncertainty

Uncertainty is a pervasive problem in both inshore and offshore marine resource management, because most marine areas are still data-poor. Observed patterns are often governed by multiple interacting factors at various spatial and temporal scales, many of which are poorly understood. Overcoming uncertainty and data challenges in offshore MPA design can include the use of remote measurements of environmental conditions as biological proxies, non-comprehensive data collected at different spatial scales, surrogate species, marine community classifications such as bioregionalisations, expert and stakeholder participatory decision-making, regional-scale remote sensing studies or a combination of these (Harris and Whiteway, 2009; Beger et al., 2020). Furthermore, to achieve most marine resource management goals in data-poor systems, it is prudent to be more reliant on the precautionary principle, where the burden of proof is shifted toward ecosystem protection first, followed by the proof of no environmental damage by human activities (Clark, 1996; Hooker et al., 2011).

A major source of uncertainty is the changing climate and its future impacts on the ocean. Natural disturbance regimes are a component of ecosystems that should also be considered in MPA design (Harris, 2014). Resistance and resilience to disturbance, or the ability to either absorb disturbance without change or to return to pre-disturbance conditions, are becoming more important as large-scale environmental impacts become more pervasive (Game et al., 2008; Palumbi et al., 2008). In fact, 12 (24%) of the documents identified during the systematic review that provided a list of design guidelines specified resilience or adaptation to climate change as a criterion for selecting MPAs. Identification of such areas can be difficult in data-poor systems, because ascertaining these qualities typically requires time-series data. Therefore, we have not included resilience as a specific guideline for offshore MPAs or MPA networks. It is possible, however, that the combination of guidelines, as presented, will contribute to building ecosystem resilience. Katsanevakis et al. (2020) suggest a risk assessment framework when implementing MPAs and MPA networks, which is an effective way to deal with uncertainty and is applicable in offshore ecosystems.

Adaptive Management

Other ways in which limited information and uncertainty can be acknowledged and accounted for is with an adaptive management approach (whereby management is altered if emerging information deems it necessary; Gormley et al., 2015; Weinert et al., 2021). Aside from knowledge gaps, management will need to occur in an uncertain future governed influenced by climate change, shifting distributions, home ranges or migration pathways; for these reasons also, MPA boundaries may require revision over time (Gruss et al., 2011; Brock et al., 2012; Nickols et al., 2019). The adaptive management cycle allows for flexibility and responsiveness to new and improved information as monitoring of the ecosystem reveals more information about an MPA's effectiveness. To allow for adaptive management offshore, MPA zoning could combine permanent protection with flexible approaches. Permanent protection is preferred (Guideline 9), but boundaries could define zones within which certain known destructive activities (e.g., industrial fishing, bottom trawling, deep-sea mining) are always prohibited, while the effects of other activities are monitored and their regulation tailored to new information.

Monitoring

Although monitoring is an integral part of MPA management, many offshore environments are lacking in even the most basic baseline data. However, to meet global conservation targets, it is impractical to wait until these data are collected before proceeding with marine spatial planning and MPA establishment. The guidelines set out in this paper are designed to optimise the placement of offshore MPAs and MPA networks using existing data, and allowing for the incorporation of data collected in the future.

To continue to improve the effectiveness of offshore networks of MPAs, especially within an adaptive management framework, effort must go into gathering and collating baseline data, followed by performance monitoring. This will include the use of remotely collected and centrally compiled biological and socio-economic data. Global datasets on fisheries and oceanic habitats are being compiled with ever increasing levels of spatial accuracy (Harris et al., 2014; Pauly et al., 2020; Global Fishing Watch, 2021). Regional or national datasets may be more available and appropriate to particular planning efforts and may inform baselines for monitoring; data from ABNJ may be less readily available. Information from monitoring can then feed into an adaptive management cycle for existing MPAs (e.g., Dunn et al., 2018) and broader offshore marine resource management efforts as well as help refine and improve the design guidelines listed here for new offshore MPAs.

Monitoring in the open ocean may rely more heavily on proxies or surrogates than in inshore areas, since data collection can be logistically challenging and expensive. Monitoring populations of some of the more wide-ranging species of interest in offshore MPAs will require a combination of methods, such as satellite technology, drifting baited stereo-videography, spotter planes, drones, horizontal acoustics, and vessel-based sampling (Jaine et al., 2014; Bouchet and Meeuwig, 2015; Letessier et al., 2017). Physical and chemical data can be easier to obtain, and, when available, can be a good predictor for the distribution of some open ocean species (Trebilco et al., 2011; Reygondeau et al., 2012; Hewitt et al., 2015; Stephenson et al., 2020). For example, Harris and Baker (2020) concluded that "Sediment grain size/composition was found to be the most useful surrogate for benthic communities in the most studies, followed by acoustic backscatter, water depth, slope, wave–current exposure, substrate type, seabed rugosity, and geomorphology/Topographic Position Index."

Regions of the world where data have been or are being collected, and uncertainty is lower, can serve as testing grounds for MPA design. For example, a large portion of the southwest Pacific has defined bioregions, identified with a combination of empirical and modelled data and verified through participatory planning (Beger et al., 2020; Ceccarelli et al., 2021). Bioregions have also been defined in offshore Australian (Department of the Environment and Heritage, 2006), UK (Chaniotis et al., 2020), South African (Livingstone et al., 2018) and Canadian waters (Burt et al., 2018). Multiple national and international seabed mapping projects are underway, including efforts to map the entire seafloor by 2030 (Wölfl et al., 2019); technology innovations promise to deliver increasingly accurate biophysical data (e.g., Zhang et al., 2021). This ongoing global acquisition of data, together with sophisticated theoretical and modelling approaches, invites both a flexible and adaptive management approach that can incorporate new information within a systematic planning framework.

Contributing to a Global Effort

There is a global willingness to move toward effective ocean conservation, as indicated by the increasing number of large and very large MPAs (IUCN, 2016b; Lewis et al., 2017) and national Marine Spatial Planning efforts (Beger et al., 2020). The global target of "30 by 30" (protecting 30% of the global ocean within MPAs by 2030) is achievable, especially in the light of recent additions to the global MPA estate (Duarte et al., 2020). However, it will rely heavily on the protection of offshore waters, which make up over 90% of the global ocean (Harris et al., 2014; Inniss et al., 2016; Sala et al., 2018). Offshore guidelines, as presented in this paper, are an essential tool to further assist progress toward this global target. The biophysical MPA network guidelines developed here are equally applicable, in principle, within national jurisdictions, within ABNJ and, with international coordination and cooperation, across national boundaries or even across national and ABNJ boundaries. The United Nations Convention on Biological Diversity will convene in October 2021. One target that will be decided upon is a global commitment to 30% marine (and terrestrial) protected areas by 2030 (CBD, 2021). This paper provides a significant input to being able to make such a commitment real.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Information 2**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

LF provided the original concept, overall guidance, contributed intellectual input to the development of the substance and substantial revisions of the paper. DC was the primary writer of the paper, conducted the systematic review, contributed intellectual input to the substance and revision of the paper, and editing. PH, KD, and GJ reviewed versions of the paper, contributed intellectual input to the content and contributed additional references. JR and SM reviewed and contributed to the substance of the original report upon which this paper is based and reviewed this manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.634574/full#supplementary-material

Supplementary Information 1 | PRISMA 2020 flow diagram for systematic reviews of Marine Protected Area guidelines. Adapted from Page et al. (2021).

Supplementary Information 2 | Literature search and data extraction results.

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Using Automatic Identification System (AIS) Data to Estimate Whale Watching Effort

Javier Almunia[1]*, *Patricia Delponti*[2] *and Fernando Rosa*[3]

[1] Loro Parque Fundación, Puerto de la Cruz, Spain, [2] Aula de Comunicación, Departamento de Ciencias de la Comunicación y Trabajo Social, Facultad de Ciencias Sociales y de la Comunicación, Universidad de La Laguna, La Laguna, Spain, [3] Laboratorio de Bioacústica Física y multi Sensores Distribuidos, Departamento de Ingeniería Industrial, Universidad de La Laguna, La Laguna, Spain

The growing concerns about the negative effects caused by whale watching on wild cetacean populations are evincing the need to measure whale watching effort more precisely. The current alternatives do not provide sufficient information or imply time-consuming and staff-intensive tasks that limit their effectiveness to establish the maximum carrying capacity for this tourist activity. A methodology based on big data analysis, using Automatic Identification System (AIS) messages can provide valuable vessel activity information, which is necessary to estimate whale watching effort in areas with cetacean populations. We used AIS data to automatically detect whale watching operations and quantify whale watching effort with high spatial and temporal resolution in the Canary Islands off the west African coast. The results obtained in this study are very encouraging, proving that the methodology can estimate seasonal and annual trends in the whale watching effort. The methodology has also proved to be effective in providing detailed spatial information about the whale watching effort, which makes an interesting tool to manage spatial regulations and enforce exclusion zones. The widespread use of AIS devices in maritime navigation provides an enormous potential to easily extend this methodology to other regions worldwide. Any public strategy aimed at the sustainable use of marine resources should enhance the use of this kind of information technologies, collecting and archiving detailed information on the activity of all the vessels, especially in marine protected areas.

Keywords: automatic identification system, cetacean, whale watching, carrying capacity, sustainability

Edited by:
Tomaso Fortibuoni,
Istituto Superiore per la Protezione e la Ricerca Ambientale (ISPRA), Italy

Reviewed by:
Carmen Ferrá,
National Research Council (CNR), Italy
Blake E. Feist,
National Oceanic and Atmospheric Administration (NOAA), United States

***Correspondence:**
Javier Almunia
dir@loroparque-fundacion.org

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1. INTRODUCTION

There is an increasing number of people who are demanding whale watching boat trips worldwide, fueling a fast-growing industry that already accounted for 3,300 operators by the end of the previous decade (O'Connor et al., 2009). As this activity is not based on lethal or consumptive use of the cetaceans, whale watching has been often labeled as "green," "eco-friendly," or "sustainable" tourism (Schuler et al., 2019); however, early in this century, the first evidence about short-term behavioral changes provoked by vessel density appeared (Allen and Read, 2000) and since then many authors have reported negative impacts of whale watching activities in different cetacean species (Erbe, 2002; Constantine et al., 2004; Schaffar et al., 2009; Christiansen et al., 2014). These short-term behavioral changes included the following: surfacing/diving, agonistic behavior, antipredator behavior, acoustic, group size or cohesion, swimming speed, swimming direction, altered feeding or resting, and altered respiratory frequency [refer to Parsons (2012) for a complete review]. Shortly, after the first evidence of the long-term negative impacts produced by whale watching appeared in one of the best-studied dolphin populations (Bejder et al., 2006), confirming the concern of the International Whaling Commission (IWC), which in 1997 created a working group to monitor whale-watching sustainability (International Whaling Commision, 2004). In that sense, whale watching, such as most other human activities, can be considered an evolutionary selection force, which alters the life of the targeted population (Lusseau et al., 2006). The fact that cetacean (whale, dolphin, and porpoise) watching is the greatest business reliant upon cetaceans worldwide (Parsons, 2012), targeting at least 56 (including endangered and threatened) species in all oceans so far (Bejder et al., 2006), urges sustainable ways to be found to perform these activities.

To accomplish this goal, it is essential to determine the carrying capacity, or maximum whale watching effort, that any cetacean population can bear in the least impacting way. The need to evaluate carrying capacity in whale watching activities has been identified early on in the scientific literature (Curtin, 2003; Higham et al., 2008; Andreu et al., 2009) but the intensity or effort of the activity has been rarely considered as a factor in the impact studies. Traditionally, the whale watching effort has been assumed to be proportional to the number of vessels operating in a certain area. But this is a deficient measurement, as it does not consider the different activity budgets of each vessel, its physical characteristics, or the seasonal and geographical variations in the whale watching events. More recently, some studies have used land-based visual observations (with binoculars or theodolite) and also acoustic data in order to measure whale watching intensity, determining the concurrent number of vessels, or the total time spent in the proximity of the animals (Pirotta et al., 2015; Schuler et al., 2019). This methodology is much more precise and appropriate to establish the effect of different whale watching intensities on the short-term behavioral disturbances produced in the cetaceans. But, it is also geographically limited, enormously time-consuming and staff intensive, which makes its application in regional monitoring programs quite unrealistic. Similarly, the need to obtain precise effort measures will be necessary to feed the mathematical models proposed to address the long-term sustainability of tourist interactions with cetaceans (Higham et al., 2008; Lusseau et al., 2009; New et al., 2020), as the quality of the model projections will heavily depend on the amount and quality of the whale watching effort data available.

The Canary Islands (**Figure 1**) are one of the top whale-watching destinations worldwide. In 1998 Spain, was considered among the three countries that could claim to have taken over one million people whale watching in 1 year (O'Connor et al., 2009), mainly thanks to the visitors registered in the Canaries. Ten years later, despite a visitor reduction due to regulatory measures and weather issues, the Canary Islands were considered the fourth whale watching destination worldwide with 611,500 whale watchers per year (O'Connor et al., 2009). Whale watching in the Canaries is strongly focused on Tenerife Island, which accounts for an estimated 85% of total whale watchers (O'Connor et al., 2009) (and 76 licensed vessels), around a resident population of some 350–450 short-finned pilot whales (*Globicephala macrorhynchus*), which, along with transient visitors, can be found off the south-west coast of Tenerife (Canary Islands) (Servidio et al., 2019), mainly in water depths from 800 to 2,000 m (Heimlich-Boran and Heimlich-Boran, 1990; Aguilar Soto et al., 2008). The short-finned pilot whales (classified as least concern in the IUCN's red list of endangered species) is the most frequently seen species in the Canary Islands, and it was found during every day survey conducted off Tenerife in a recent study, becoming the main target species for the whale watching vessels off the island (Servidio et al., 2019). The short-finned pilot whale is listed as vulnerable in the Spanish and Canarian catalogues of endangered species, and the marine area off south-west Tenerife has been designated by the European Union as a Site of Community Importance (SCI) and included in the Natura 2000 network (European Council, 1992). Gran Canaria accounts for a smaller percentage of whale watchers, and consequently a smaller number of licensed vessels (15). The ample island platform which extends off south Gran Canaria favors the presence of other cetacean species and, as a result, the sightings of short-finned pilot whales by whale watching vessels are rare (Javier Zaera Comm. pers.). The fact that the Canary Islands is one of the leading whale watching destinations worldwide, and the concentration of this activity is in a well-defined resident species, constitutes an ideal laboratory to study whale watching efforts.

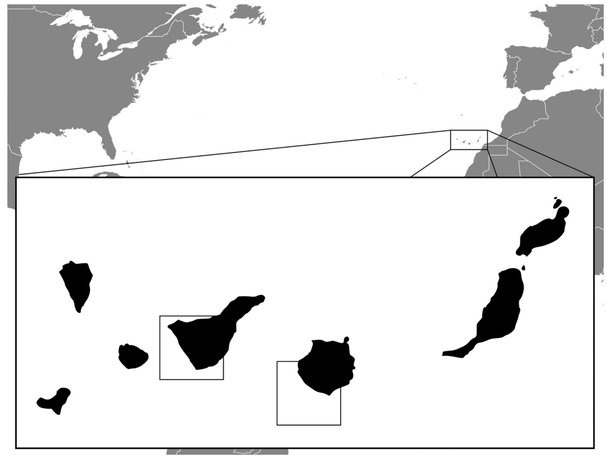

FIGURE 1 | Canary Islands and study sites off south Tenerife and Gran Canaria (squares).

The (AIS) is a location reporting system based on automatic radio messages that were developed for collision avoidance. The AIS transponder automatically broadcasts messages containing information of name, position, course, speed, etc., of the vessel at regular intervals which can be received by AIS stations in the area (Lapinski and Isenor, 2011). The International Maritime Organization mandates the use of AIS in vessels larger than 300 gross tonnes that travel internationally, cargo ships of 500 gross tonnage or more sailing in local waters, and all passenger ships irrespective of size (International Maritime Organization (IMO), 1974). The system was originally designed to extend the radar coverage and vessel traffic services (VTS), but it can be easily used to obtain information about marine traffic in a region. AIS provides basic information with position updates at sample rates varying from 3 s to 3 min dependent on the manoeuvre situation of an individual vessel (Aarsæther and Moan, 2009), but it is not limited to that, and is continuously updated to provide further aids to navigation (Balduzzi et al., 2014). Apart from its original goal, the enormous quantity of AIS data available has proved to be a valuable source of information on human use of marine areas. As a consequence, it has been used for different purposes: from monitoring fishing activity and protected area regulation compliance (Natale et al., 2015; de Souza et al., 2016; Rowlands et al., 2019), or evaluating cetacean-vessel collision risks (Greig et al., 2020; Redfern et al., 2020) to specific risk evaluation associated with different kinds of vessels (McWhinnie et al., 2021). Also, recently two projects, MARCET and WAVES have been explored for its potential to evaluate whale watching effort (Canessa, 2019; Universidad de las Palmas de Gran Canaria, 2020). This study aims to evaluate the potential use of AIS data, combined with an open-source digital terrain model (DTM) to measure automatically the whale watching effort in a specific region.

2. METHODOLOGY

2.1. AIS Data

Two specific geographical areas were selected to characterize the whale watching activities off southern Tenerife Island (27.9–28.4°N; 16.5–17.0°W) and off southern Gran Canaria Island (27.5–28.0°N; 15.5–16.0°W). The vessels included in this study were selected searching, at the MarineTraffic™ database, for the names of the ships authorized by the Canary Islands Government to perform whale watching activities in the region. The Maritime Mobile Service Identity (MMSI) numbers of all the authorized ships that were equipped with an AIS transponder, and consequently appeared in the MarineTraffic™ database searches, were included in the study. The search resulted in a total of 23 vessels (out of 120 authorized for whale watching activities in the region) that produced AIS messages between 2016 and 2020.

AIS data were collected by Exmile Solutions Ltd. (proprietary of MarineTraffic™, London, UK) in the calendar years from 2016 to 2019, and part of 2020, and received either from terrestrial stations or satellite. To ensure effective management of the incoming information in the database, MarineTraffic™ uses proprietary down-sampling techniques not to archive consecutive positions within minutes. This results in a maximum resolution of 1 min for the archived data. All the available archived AIS messages, received from vessels selected for the study, were obtained from the database MarineTraffic™. The available messages were previously filtered by timestamp between "2016-01-01 00:00" and "2020-03-14 00:00," and further filtered to select just the operational hours of the whale watching vessels (between 09:30 a.m. and 17:30 p.m.).

2.2. EMODnet DTM

As the original AIS messages do not include information about the depth of the location of the vessel, it was extracted from a DTM. The"EMODnet Digital Bathymetry (DTM)" is a multilayer bathymetric product for sea basins of Europe. The DTM is based upon more than 7,700 bathymetric survey data sets and composite DTMs that have been gathered from 27 data providers from 18 European countries and involving 169 data originators (Consortium, 2016; Thierry et al., 2019). The data grid available for the Canary Islands region had a resolution of 7.5 X 7.5 arcseconds, and was obtained through the common data index (CDI).

2.3. Data Processing

The data were processed using scripts written by the authors utilizing the programming language Python (Van Rossum and Drake, 2009) running in Anaconda Spyder. A preliminary data quality control was performed removing all the positions out of the geographical range of the study. The depth associated with each AIS message was calculated by obtaining the value associated with the DTM grid cell corresponding to the longitude and latitude of the vessel position according to the AIS. As the goal of this study was to characterize whale watching events, all the messages that were broadcasted from positions shallower than 100 m were filtered to eliminate those messages originating from harbors, moorings, proximity to fish farms, and coastal navigation.

A first analysis of the data was aimed to identify the messages that could be produced while the vessels were on a whale sighting, taking into consideration that the ship speed should be reduced to follow the whales during the whale watching event (less than 4 knots according to the local regulations) (Gobierno de Canarias, 2000), and that the species most frequently sighted in the study areas are commonly distributed in water depths from 800 to 2,000 m (Heimlich-Boran, 1993). A density plot of the AIS messages (**Figure 2**) was produced to evaluate the frequency of messages related to depth and boat speed, and frequency distribution of the AIS messages by depth was calculated for both study areas (**Figure 3**). The geographical distribution of the AIS messages in both study areas was plotted (**Figures 4**, **5**), and density maps were produced for the locations where the messages indicated that the speed of the vessel was under 2.5 knots and depth more than 100 m, using Seaborn (Waskom et al., 2017). As the main goal of this study was to prove the potential use of AIS to identify whale-watching operations, and due to the low number of data from vessels operating off Gran Canaria, the remaining of the analysis focused only on the data obtained from the vessels operating off Tenerife.

To infer the duration of the whale watching events from the AIS messages, the common behavior observed on the tourist boats visiting the resident population of pilot whales off Tenerife was used as a model. The vessels head to the areas where the whales are frequently resting (with a characteristic depth of 800–2,000) at medium-high speed, reducing speed to approach the animals, and head back to the coast at medium-high speed again after a 20–30 min observation (personal observation). This whale watching pattern is the most frequent off Tenerife Island, despite

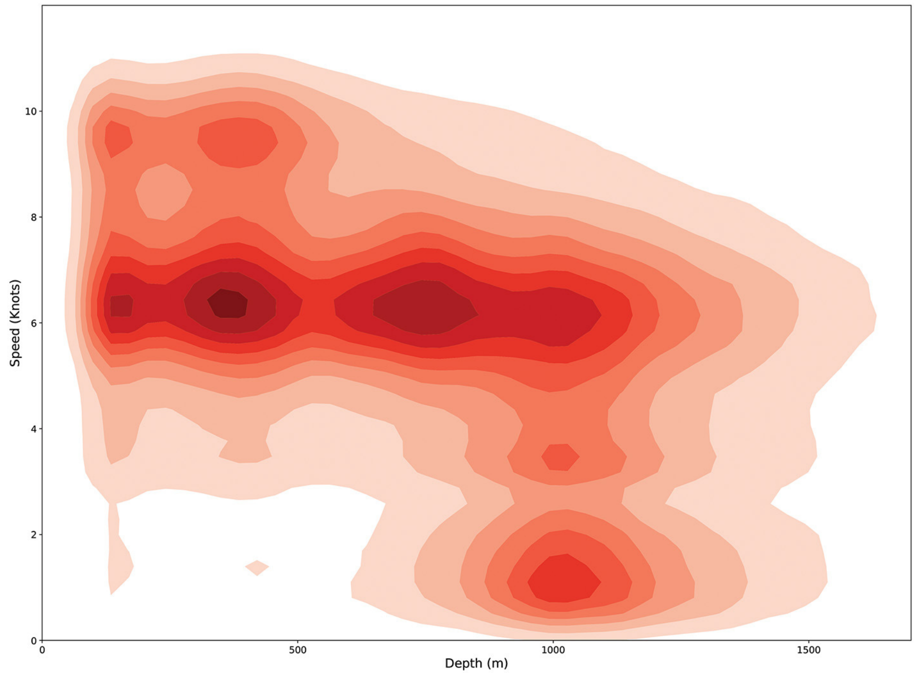

FIGURE 2 | Density plot of the Automatic Identification System (AIS) messages in the data set by speed (knots) and depth (meters).

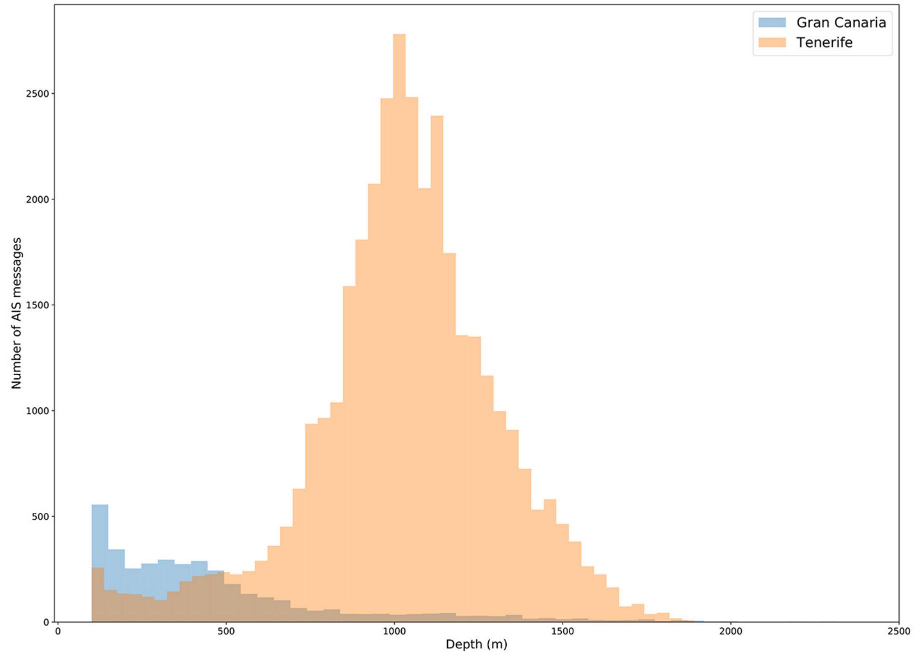

FIGURE 3 | Frequency distribution of the AIS messages broadcasted at different depths in the study areas (Blue tiles Gran Canaria and red tiles Tenerife). All the messages that were broadcasted from positions shallower than 100 m were filtered.

that it can vary when resident bottle-nose dolphins are found on the way to observe the pilot whales, or other transient whale or dolphin species are present in the area. A Python script was created to detect sequences in the AIS messages that could fit this simple model of pilot whale observation. A data sequence was defined to start when the speed of the boat was under 2.5

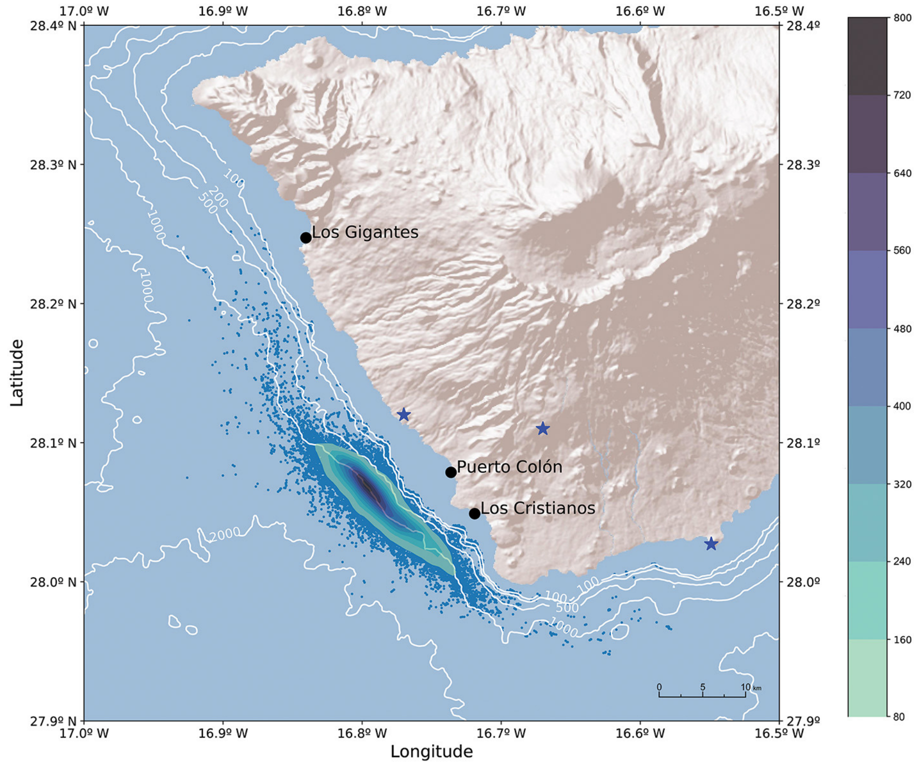

FIGURE 4 | AIS messages broadcasted from potential whale watching events (blue dots) and density map of the potential whale watching events off Tenerife Island. Location of the main harbors (black dots) and the AIS terrestrial stations (blue stars) in the region.White lines represent isobaths in meters.

knots in areas deeper than 100 m. The sequence was subsequently ended when the boat reached a speed over 5 knots regardless of the depth, assuming that it had left the whales, and was cruising again. These criteria were planned to include, in the same sequence, several short-term ship movements intended to approach separated animals within the same group or to re-position the vessel while the pod was moving. The difference between the timestamp of the first and last messages of the sequence was used to calculate the event duration. The mean depth of the whale watching events was calculated as the mean of the estimated depth in each of the messages of a sequence.

Based on the inference of the duration for each whale watching event, it is possible to estimate the integrated monthly duration of whale watching performed by the vessels included in the study. The integrated monthly duration of the whale watching activities was calculated as the sum of the individual duration of each sequence identified within a month, and it was normalized dividing by the mean active vessels. To exclude the ships stranded for maintenance operations, operating seasonally or out of business, any vessel that did at least one whale watching event per week was considered active and to calculate indexes over the study period, active vessels were averaged monthly (mean active vessels) and every year (yearly average active vessel). The normalized duration of whale watching activities is an estimation of the time that the whale watching vessels were in the proximity of whales and, as a consequence, it could be used as an indication of whale watching effort.

Finally, to find out if the methodology could be useful to elucidate some long-term behavioral effects in the whales, such as signs of avoidance of the whale watching vessels, the AIS information was analyzed to evaluate any spatial trend in the location of the whale watching events. As the distance of the whale watching events to shore could be more biased by the low number of boats (and the fact that, usually, they share the daily positions by radio or simply head to the closest whale watching boats slowly sailing in the area) than the mean water depth at the whale watching events, and the geographical positions, these latter were used instead. The daily mean depth, latitude, and longitude of the whale watching events were calculated, and time series were constructed to analyze temporal, directional or stationary aspects of the data. The augmented Dickey Fuller test (ADFT) (Cheung and Lai, 1995) was used to evaluate stationarity of the time series, and the Durbin-Watson test (White, 1992) was used to detect the presence of auto-correlation. Finally, the auto correlation function (ACF) plot was used to reveal how the correlation between any two depth values changed as the time lag increased (Seabold and Perktold, 2010).

3. RESULTS

The original AIS data received had 729,951 messages (10,558 of them from satellite and the rest from terrestrial stations). The data were not evenly distributed over the years (**Table 1**), but showed an increasing trend related to the growing number of

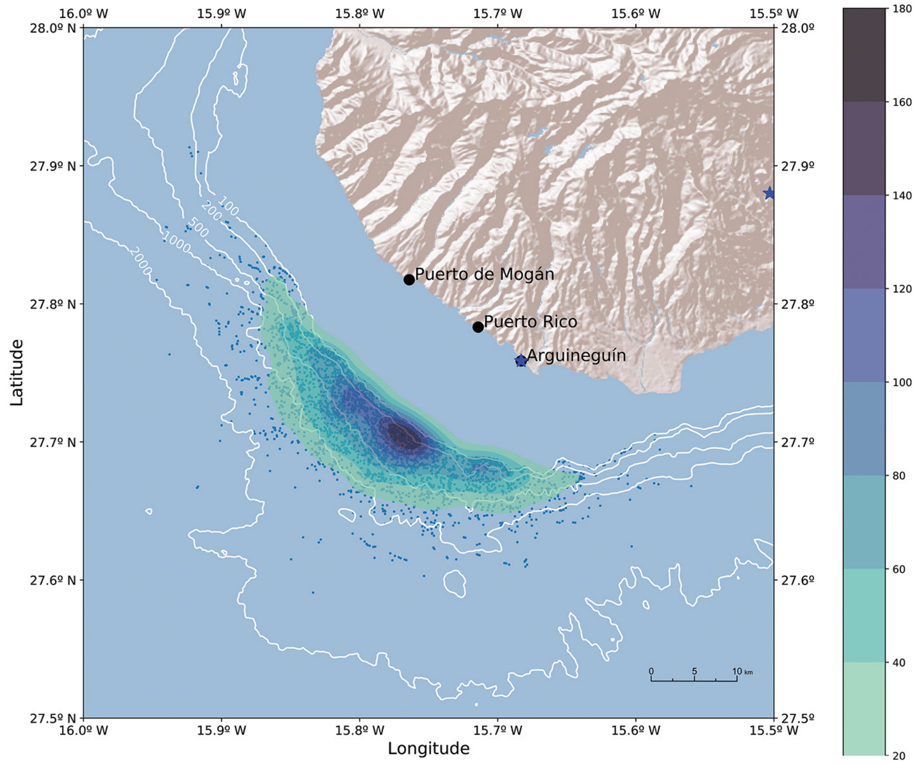

FIGURE 5 | AIS messages broadcasted from potential whale watching events (blue dots) and density map of the potential whale watching events off Gran Canaria Island. Location of the main harbors (black dots) and the AIS terrestrial stations (blue stars) in the region. White lines represent isobaths in meters.

active vessels equipped with AIS transponders in the region: 2016 (72,751), 2017 (135,158), 2018 (217,457), 2019 (261,619), and 2020 (42,966 just in two and a half months). After the 100 m depth filter was applied, 265,909 (36%) AIS messages remained for subsequent analyses.

A density plot of the AIS messages in the data set by speed and depth (**Figure 2**) showed a particular region between 700 and 1,500 m in depth where the vessel speed was consistently under 2.5 knots. This can be considered as an indication of whale watching operations, especially because the figure also illustrates how the vessels typically cruise at 6 knots regardless of the depth. It is also noticeable that AIS messages with speeds lower than 2.5 knots were almost absent in shallower waters, except for two small spots (around 200 and 450 m in depth) that can be seen in the figure.

The histogram of the AIS messages emitted at speeds lower than 2.5 knots (**Figure 3**) shows a clear peak around 800–1,200 m in depth for the messages from the region off Tenerife Island, while messages emitted off Gran Canaria Island were more frequent in shallower waters. This supports the idea that the AIS messages from the boats operating in Tenerife were emitted while the vessels were performing whale watching operations, as this depth range matches the one described for the most common species in the region (Heimlich-Boran, 1993). Furthermore, the whale-watching operations in Gran Canaria target a broader range of species, as reported by the captains (Javier Zaera pers. comm.).

The geographical distribution (**Figure 4**) for the whale watching events off south-western Tenerife (blue dots) show a narrow area (approx. 10 km wide) that extends over 50 km along the island slope. This area resembles the published distribution maps of the most frequently seen whale species in south-west Tenerife determined by dedicated surveys (Carrillo et al., 2010). On the other hand, the density map of the whale watching distribution off the south-west coast of Gran Canaria (**Figure 5**) shows a shallower distribution of the potential whale watching events which is consistent with the fact that whale watching operations off Gran Canaria are focused on multiple species. In both cases, the density maps indicate that whale watching events are more frequent (dark blue in the density map) in a very small space (approximately 1 km^2) compared with the total area where the whales were present.

The analysis of the AIS messages broadcasted off Tenerife Island identified a total of 8,745 sequences matching the model proposed for potential whale watching events, with a duration ranging from 1.3 to 95 min. The frequency distribution of the event duration (**Figure 6**) illustrates that the vast majority of events lasted less than 30 min, with few of them going over 40 min. This distribution is consistent with the fact that most of the whale watching excursions in south-west Tenerife last for 2 h and the time devoted to watching the whales used to be around 20

TABLE 1 | Automatic identification system (AIS) messages and yearly average active vessels equipped with AIS transponder off Tenerife Island during the study.

	2016	2017	2018	2019	2020 (Jan-Mar 15th)
Total AIS messages received	72,751	135,158	217,457	261,619	42,966
Monthly average number of messages per active vessel	1,332	1,805	2,178	2,422	1,931
Total number of active vessels (Yearly average active vessels)	10 *(4.55)*	16 *(6.24)*	16 *(8.32)*	15 *(9.00)*	13 *(8.90)*
Number of AIS messages deeper than 100 m	24,846	51,175	83,389	89,958	15,722

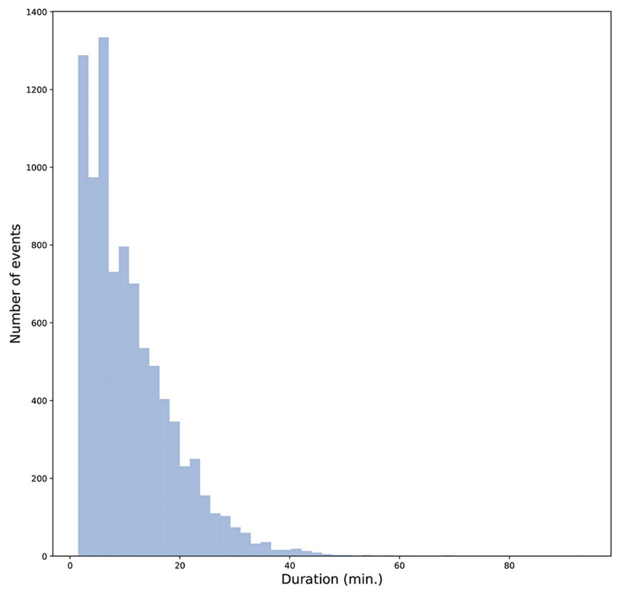

FIGURE 6 | Frequency distribution of the estimated duration for the whale watching events.

min (Pers. obs.) and falls within the maximum time allowance of 30 min established in the local whale watching regulations (Gobierno de Canarias, 2000).

The evolution of the integrated monthly duration of whale watching events, normalized by the mean active vessels, varies through the period covered by the study (**Figure 7**). The graph shows an increase in the whale watching activity in the first half of the study period and also clear seasonal variations with maximum values in summer and minimum in winter. The number of operational whale watching vessels also increased, especially during the first half of the study, but remained fairly constant during its second half (when on average it was a maximum of 9 yearly average active vessels off southwest Tenerife Island, out of 23 equipped with AIS transponders). Consequently, the method seems to be able to detect variations in the intensity of whale watching activities, even when the yearly average active vessels in the area remains fairly constant.

The ADFT (Cheung and Lai, 1995) of the daily mean depth for the whale watching events indicates that the time series is stationary (p-value = 0.007). This would suggest that the mean depth of the whale watching events in the area do not present a long-term trend. The monthly mean depth of the whale watching events (**Figure 8**) seems to show a seasonal trend confirmed by the Durbin-Watson (White, 1992) test (value = 0.026), and the auto-correlation function plot shows a positive auto-correlation around a 350 days lag. No other significant auto-correlation lags (lunar cycles or multi-annual trends) could be found in the auto-correlation function plot. Similarly, the longitude and latitude time series were also stationary (ADFT p-values 0.003 and 0.007, respectively) and their auto-correlation function plots did not show significant auto-correlations, suggesting that the seasonal depth change of the whale watching events were the result of very subtle or inconsistent changes in the position (refer to **Supplementary Materials**).

4. DISCUSSION

The basic model established to identify the whale watching events from the AIS information has proven to be promising, as the results seem to fit the distribution of the population (Carrillo et al., 2010; Servidio et al., 2019) and the fidelity of the species to a certain bathymetric range (Heimlich-Boran and Heimlich-Boran, 1990; Aguilar Soto et al., 2008; Carrillo et al., 2010) as described in the scientific literature for the species most frequently seen in the area, the short-finned pilot whale (Servidio et al., 2019). Notwithstanding the evidence that the criteria seem to be valid to determine the whale-watching operations with short-finned pilot whales in the Canary Islands, its effectiveness and precision should be confirmed by future studies that simultaneously collect AIS information and on board information on the whales sighted. Furthermore, the proposed basic model for short-finned pilot whales should be carefully adapted to the specific characteristics of whale watching operations focusing on different species in other regions. Precise whale watching event measurements are scarce in the scientific literature, and usually related to direct measures (theodolite) (Schaffar et al., 2009; Cecchetti et al., 2018; Schuler et al., 2019), data collected on-board (Robbins and Frost, 2009) or indirect estimations (ship noise) (Houghton et al., 2015). In most cases, the whale watching effort is estimated just by the number of licensed boats that can operate in the area, but this approach lacks information about the amount of time that the animals are perturbed by the vessels, and also

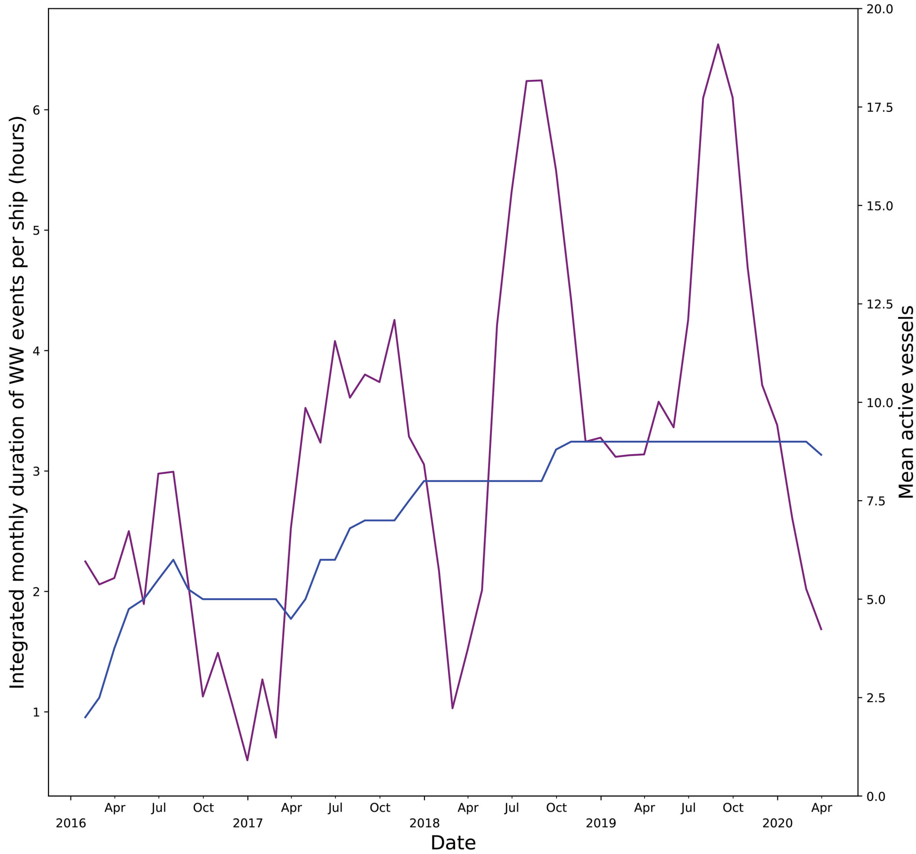

FIGURE 7 | Total integrated monthly duration of whale watching events off south Tenerife normalized by mean active vessels (red line). Mean number of active vessels (blue line).

the spatial distribution of this disturbance. Obtaining detailed information about whale watching events using the theodolite method implies several observation teams of at least three people each (theodolite operator, computer operator, and 1–2 spotters) (Schuler et al., 2019). In addition, the observations from land can be also influenced by adverse meteorological conditions (fog, dust, swell, etc.), and they are very difficult to perform in extensive or convoluted shores. In this situation, the proposed methodology is clearly advantageous as it could cover vast regions, the data collection could be fully automated, and requires no staff. Actually, the use of big data widens the reach of research possibilities in the information society. In this sense, the importance of big data, as one of the disruptive technologies in the public digital landscape, has been gradually growing, as well as the number of private organizations that in recent years have begun to store and process data to meet the demand of a market that uses and analyzes the data to generate knowledge and create business (Salvador et al., 2017); however, how to deal with information management, how to store it and its accessibility in the big data era are challenges for public endeavors, which should ensure not only that data collection is available but also should ensure storage, interoperability, and accessibility. Consequently, ensuring that AIS data from whale watching and other tourist activities are open and accessible would imply a positive impact on sustainability.

The high site fidelity and bathymetric dependence of *G. macrorhynchus* (Heimlich-Boran and Heimlich-Boran, 1990; Aguilar Soto et al., 2008; Carrillo et al., 2010) has direct implications on the available optimum habitat for the species, and it can vary dramatically in two islands within the same archipelago. The particular bathymetry of Tenerife Island configures a small distribution area (approx. 150 km^2) which supports 350–450 short finned-pilot whales (Heimlich-Boran and Heimlich-Boran, 1990; Aguilar Soto et al., 2008). The relatively small size of the distribution area also implies a higher risk of impact on the population by an oversized whale watching industry. And, at the same time, the potential high presence of animals can also impose a higher collision risk to the ships navigating in the area (Carrillo et al., 2010). On the other hand, the fine detail of whale watching activity obtained using AIS methodology, could be useful to define and enforce more precise low-speed areas to reduce collisions (Silveira et al., 2013; Greig et al., 2020), which has been identified as one of the measures to reduce the ship strikes in the region (Carrillo et al., 2010). In that sense, it is important to consider that the very nature of the whale sighting operations implies that each whale watching

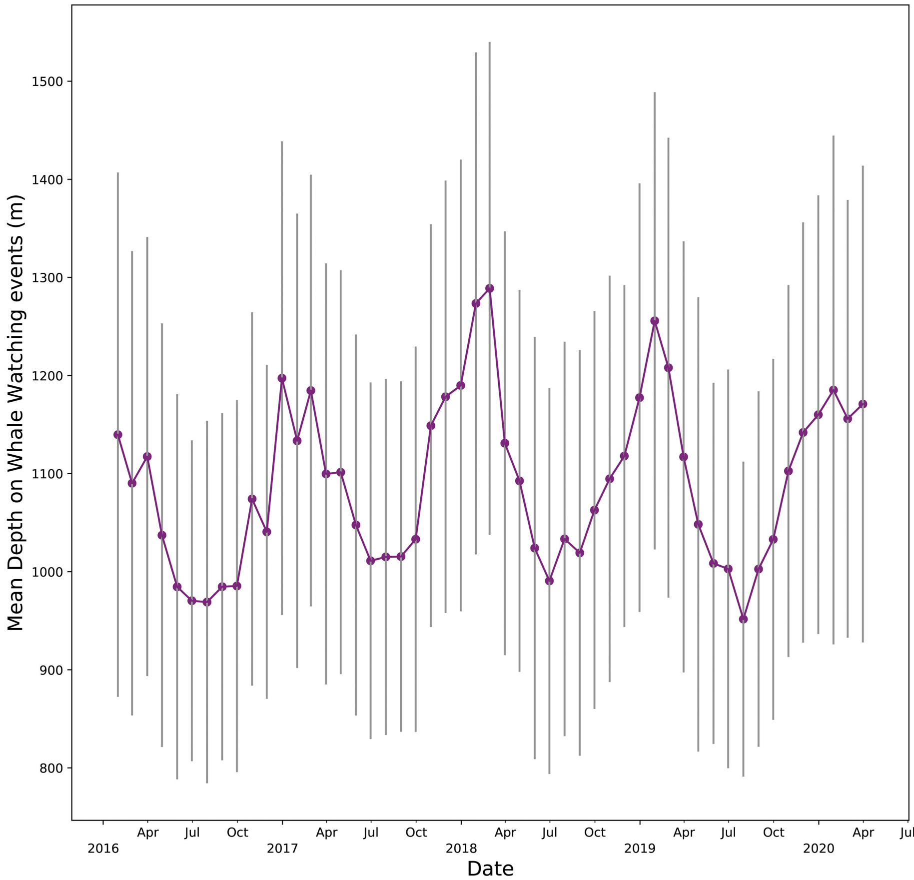

FIGURE 8 | Mean monthly depth of the whale watching activities off south Tenerife during the study period (red line). Gray lines indicate SD.

ship will preferably observe the animals closest to the shore for purely economic reasons. Similarly, the presence of whales in range areas far from tourist harbors could be underestimated for the very same reason. On the other hand, the recent application of AIS information to evaluate specific risks associated with the presence of vessels of different types (McWhinnie et al., 2021) gives a new perspective to this methodology as a risk assessment tool itself. The enormous amount of detailed information, not only on the presence but also on the daily activity of the whale watching boats provided by the analysis of the AIS messages, could serve as an effective management tool that would allow the managers of a protected area to analyze the cumulative effort on different groups of the population, and even intervene by redirecting the effort to other areas in certain situations.

An appropriate field of vision of the AIS stations and their continuous operation is necessary to establish a system that automatically estimates the whale watching effort, using the proposed methodology. Both study areas have a good AIS coverage, thanks to the number and position of terrestrial stations in the south of Tenerife and Gran Canaria, as proven by the low number of satellite messages in the dataset (1,4%), that were even less prevalent in the final sequences (0,3%). The implementation of a high spatial and temporal resolution methodology based on AIS should be based on terrestrial AIS stations that provide a good coverage of the whale watching areas and ensure a high rate of message reception. Since this navigation system is used worldwide to ensure the safety of life at sea (Wieslaw, 2012), it is very likely that AIS coverage is already available in many regions, where whale watching operations take place, and even historically could be available for retrospective studies; however, the fact that a good AIS coverage is needed to detect the short-time behavior of the vessels does not diminish the importance and potential of the satellite messages to analyze whale watching effort, as they allow the recovery of information from enormous areas out of reach of land-based stations.

The estimation of duration of the whale watching event based on AIS messages seems to be quite accurate, judging by the obtained distribution, which is mainly under the maximum time allowance for a sighting established by the local regulations (Gobierno de Canarias, 2000). It has to be taken into account that the highest possible precision for a whale watching-sequence using MarineTraffic™ archived data is 2 min, but this could be improved to 8 s if a dedicated reception network is used, as all the messages could be stored. This is due to the fact that AIS transducers broadcast a message every 12 s when the ship is sailing at 0–14 knots, or every 4 s if it is changing course (Aarsæther and Moan, 2009). The higher potential to detect and store messages of a dedicated reception network would also allow the detection and storage of detailed course change events, that could be useful to estimate the duration of the whale watching events more precisely but also to infer evasive behaviors on the whales (Schaffar et al., 2009; Christiansen et al., 2014). Although the estimation of the sequences uses a conservative threshold to identify the end of the sighting (speed over 5 knots), it does not seem to produce a significant number of overestimated event lengths. A much more detailed study, collecting and archiving all the available AIS messages from the terrestrial stations, and comparing duration estimated by AIS with the sighting time measured by observers on board (or with theodolite from a land-based station), would be necessary to evaluate the precision of the present method; however, even though an exact duration cannot be calculated for single events, it is reasonable to assume that the errors will cancel when the values are integrated monthly. Hence, monthly integrated values could provide a good indication of the temporal trends in the whale watching intensity.

It has been proven that behavioral disturbance in cetaceans is not only related to the presence/absence or the number of vessels in the vicinity, but also to the amount of time spent in the presence of vessels (Schuler et al., 2019). Multiple vessels simultaneously tracking a whale will accentuate this effect (Holt et al., 2009). Vessel characteristics (e.g., size and engine type) and vessel approach (e.g., angle and speed) are also likely to elicit different responses in whales (Schuler et al., 2019). Consequently, measurement and analysis of the time that the whale watching vessels spend observing cetaceans are essential to understand its long-term consequences on the populations. The integrated duration of the monthly whale watching events calculated in this study has captured the trend of the whale watching effort better than the previous estimations. The number of licensed boats alone is unable to reveal seasonal differences in the whale watching activities, while those are clearly captured using the AIS messages. The maximum intensity detected in the summer season by this study matches the peak in the activity due to the higher frequency of days with better weather conditions. On the other hand, the apparent trend observed in the whale watching effort during the study could be biased by the small number of ships in the analysis and its heterogeneous activity during the whole period. To improve the accuracy of this measure, an increase in the number of whale watching vessels equipped with an AIS transponder should be necessary.

The results also show a clear seasonal trend in the average depth where the sightings were made, which is confirmed by the impressions of some whale watching pilots, who refer to the whales tending to be closer to shore in summer, depending on weather conditions. The simple fact that the analysis of the AIS data has detected this subtle annual cycle gives an insight on the potential sensitivity of this methodology.

In addition to noise, the physical presence of boats may disrupt cetacean activity patterns, particularly when boats seek direct interactions (e.g., whale watching). In these cases, theoretical studies suggest that individuals often perceive boats as a risk, and therefore respond through avoidance and other anti-predatory tactics (Pirotta et al., 2015). Cetaceans may begin to avoid particular areas if the disturbance reaches a certain threshold or if there is little cost to abandoning that location (Wright et al., 2011). It has also been observed that marine mammals may temporarily move away during periods of heavy vessel activity but re-inhabit the same area when traffic is reduced (Bejder et al., 2006). Given the fact that the highest intensity in the whale watching activities in the Canaries happens during summer, one could expect that the whales would move far from the island (deeper waters) to avoid the vessels during this season. But, the monthly mean depth of the whale watching events suggests a clear yearly cycle, where the animals slightly approach to the coast in summer and move to deeper waters in winter. This result, and the fact that the time series is stationary, could be indicating a lack of avoidance in the long-term behavior of whales in the area. however, this observation does not exclude more subtle avoidance effects, such as the displacement of the more sensitive animals from the area of disturbance (Bejder et al., 2006). Similarly, the whale watching effort is not homogeneously distributed across the optimal habitat of the whale most frequently sighted in the area. Hence, there could also be some habitat shift over the distribution area, changing the location of the animals, but not the depth. Although this was not observed in the auto-correlation data of the whale watching positions registered in the study, subtle displacements at a constant depth could be addressed through a much more detailed analysis of the positions. Consequently, to accredit the presence or absence of avoidance effects, more comprehensive studies with individual identification of the cetaceans and their movements within the area of distribution would be necessary. If these avoidance reactions could be found, the concurrent determination of the whale watching effort using the proposed AIS methodology would allow the establishment of sustainable whale watching thresholds where avoidance does not occur.

Finally, the fine spatial resolution of whale watching effort obtained by this methodology is very promising as a component to estimate carrying capacity, not only as a tool to analyze the effects of different whale sighting intensities in future studies but also to enforce the spatial regulation of the activities in a region. The existence of guidelines, regulations, or laws in an area is no guarantee of compliance with these guidelines (Parsons, 2012); the best guidelines can become inefficient if there is a chronic lack of enforcement. The most widespread method for effort regulation is to limit the number of licences but this does not take into account the variable effort of each vessel either in time and space; neither the size nor characteristic propeller noise. The actual scientific methods to measure effort and behavioral effects, such as theodolite observations, could be useful to enforce regulations but are either expensive or time-consuming to cover big areas (Bejder et al., 2006; Schuler et al., 2019). The enforcement of exclusion (or limited effort) areas during sensitive seasons using the AIS-based methodology will be very easy to track, making the identification of any vessel breaching the regulations a fully automated process. The methodology proposed in this study would be able to distinguish automatically when a vessel is just sailing through an exclusion zone, when it is performing a whale watching activity in a prohibited or regulated area, or even send an alert when an unlicensed vessel is performing whale watching in a regulated area. The possibility to determine the position also allows the identification of evidence concurrent vessels with the same group of whales, which could also be useful for the enforcement. To achieve that all the licensed ships in the region should be equipped with an AIS transponder. On the other hand, since this methodology can be used to record whale watching operations regardless of the vessel, it could be also used to detect and identify non-authorized ships performing whale sighting. To detect these illegal whale watching operations, not only should the authorized vessels have an AIS transponder, but also all the tourist vessels operating in the area.

The fact that the vessels were not randomly chosen, and that they were not operating homogeneously during the study, does not allow generalization of the effort indices and trends found in the results of this study. In addition, the size of the sample (19% of the whale watching licensed ships in the region) and the smaller number of active ships during the study (mean active vessels between 2 and 9 in Tenerife) could be introducing some bias in the results. It is also important to consider the bias introduced due to the fact that the sample of vessels with AIS transponders will underestimate small cetacean watching vessels. On the island of Tenerife, the mean length of AIS-equipped whale watching vessels (17.63 m, SD 5) is greater than the mean of all licensed vessels (14.36 m, SD 6.4), mainly because none of the vessels less than 12 m in length were equipped with AIS (refer to **Supplementary Materials**). A public network of terrestrial stations would be essential to receive and archive high precision data from a large amount of vessels, as the ships sailing at 0–14 knots transmit AIS messages every 12 s, or every 4 s if they are changing course (Aarsæther and Moan, 2009). This high transmission rate, and the possibility to archive all the messages as open data, would allow more accurate calculation of the individual whale watching events that could be used for effort and carrying capacity estimation, but also to regulate enforcement. This is, especially, interesting when the whale watching activities are performed in remote or difficult to access areas (Parsons, 2012). Considering that some of the fastest growing whale-watching industries are in developing countries, and that there is still an enormous potential for considerable growth in whale-watching operations in other developing nations (Parsons, 2012), the possibility to develop an automatic system to assist the enforcement of regulations would be of great help in the future.

5. CONCLUSIONS

The proposed methodology to automatically estimate whale watching effort using AIS messages and bathymetry data from EMODNET DTM has proven to be effective off south west Tenerife, in the Canary Islands. The results obtained in this preliminary study are very encouraging, allowing the estimation of seasonal and annual trends in the total amount of exposure placed upon cetaceans by whale watching activities. The results also provide detailed geographical distribution of the whale watching effort, which when coupled with onboard observations of the presence and abundance of whale could be used to analyze subtle movements of the pods within their local distribution range.

As the proposed methodology relies heavily on the percentage of vessels equipped with AIS transponders, to achieve an optimum evaluation of the whale watching effort in a region, the competent authorities should promote its installation at least in all the ships authorized to perform whale watching activities. In addition, to detect illegal operations, any vessel capable of performing tourist activities in the area ideally should have an AIS transponder installed.

To survey effectively the whale watching area, a comprehensive study has to be performed to install enough AIS terrestrial stations to attain complete coverage and ensure the maximum reception of broadcasted messages. The system should be dimensioned by considering the number of vessels to manage the simultaneous incoming messages. Once the system is operational, some level of open data policy should be established to grant transparent access to researchers and other stakeholders.

The methodology based on AIS messages has also proved to be successful in providing detailed spatial information about the whale watching effort. This characteristic is very promising to manage spatial whale watching regulations, especially to verify the enforcement of exclusion zones and areas with limited activity.

Having enough open data sets and making them available to science will contribute to the generation of knowledge and the creation of innovative products and services that have an impact not only on social well-being, but also on sustainability. This is the challenge for the appropriate authorities, which must ensure the relevance of this open data: more quality in the diversity of data and greater reflection to facilitate correlations with each other.

DATA AVAILABILITY STATEMENT

The data analyzed in this study is subject to the following licenses/restrictions: Not for commercial use. Requests to access these datasets should be directed to dir@loroparque-fundacion.org.

AUTHOR CONTRIBUTIONS

JA conducted the analysis and led the project. FR conducted specific data analysis. All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.635568/full#supplementary-material

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METHODS
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Governing the Land-Sea Interface to Achieve Sustainable Coastal Development

Gerald G. Singh[1,2], Richard S. Cottrell[3,4,5], Tyler D. Eddy[6] and Andrés Miguel Cisneros-Montemayor[2,7]***

[1] Department of Geography, Memorial University of Newfoundland, St. John's, NL, Canada, [2] Institute for the Oceans and Fisheries, University of British Columbia, Vancouver, BC, Canada, [3] National Center for Ecological Analysis and Synthesis, University of California, Santa Barbara, Santa Barbara, CA, United States, [4] Centre for Marine Socioecology, University of Tasmania, Hobart, TAS, Australia, [5] School of Earth and Environmental Sciences, University of Queensland, St. Lucia, QLD, Australia, [6] Centre for Fisheries Ecosystems Research, Fisheries & Marine Institute, Memorial University of Newfoundland, St. John's, NL, Canada, [7] Resource and Environmental Management, Simon Fraser University, Burnaby, BC, Canada

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Minneapolis, MN, United States

***Correspondence:**
Gerald G. Singh
geralds@mun.ca

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Coastal regions are essential to achieving the Sustainable Development Goals (SDGs) given their importance for human habitation, resource provisioning, employment, and cultural practice. They are also regions where different ecological, disciplinary, and jurisdictional boundaries both overlap and are obscured. We thus propose the land-sea interface as areas where governance systems are most in need of frameworks for systems analysis to meet the SDGs—which are inherently interconnected— and integrate complex interdependencies between human livelihoods, energy, transport, food production, and nutrient flows (among others). We propose a strategic land-sea governance framework built on the sustainable transitions literature to plan for governance to achieve sustainable development across the land-sea interface. To illustrate our proposal, we compare governance planning processes across four case-based scenarios: an industrialized coastal country, a least developed coastal country, a developing coastal country with local dependencies on ocean resources, and a small island developing state primarily dependent on tourism. Through the lens of aligning governance actors and actions vertically (subnational to national), horizontally (across sectors), and programmatically (from goals to implementation), we propose scales at which governance systems may be misaligned, such as where different agencies that affect marine systems have conflicting visions and goals, leading to stalled progress or counterproductive actions. Where possible, we also highlight strategies to align across scales of high level strategic policy, tactical scale institutional mandates and cooperation, and on the ground activities and operations, such as aligning actors based on an analysis of interdependencies of goals.

Keywords: land-sea interface, transition management, sustainable development goals, governance, policy alignment, coastal systems

INTRODUCTION

Coastal systems are home to a large proportion of the world's population, directly support hundreds of millions of livelihoods, and are the direct link between marine resources and seafood supply chains, especially in coastal countries and island states (Singh et al., 2018; Selig et al., 2019; Lam et al., 2020). The land-sea interface that defines coastal systems faces a broad array of impacts from climate change (including stressors from mean temperature rise, ocean acidification, and extreme weather events) across all dimensions of the Sustainable Development Goals (SDGs) (Singh et al., 2019). Importantly from a systems perspective, coasts are also directly impacted by land-based pressures and human activities including increased erosion and sedimentation, nutrient loading, and many forms of pollution stemming from agriculture, urbanization and energy production (Lotze et al., 2006; Halpern et al., 2015; Singh et al., 2017, 2020; Nordhaus et al., 2018). Many of these pressures and industrial sectors do not account for, and may not be aware of, (sometimes literal) downstream impacts on the oceans (Halpern et al., 2008). Governance and decision-making to promote sustainable development for the land-sea interface must therefore be integrative across diverse dimensions of social-ecological systems.

Because coastal systems are so important to people and are so social-ecologically complex, sustainable coastal development is essential for achieving the SDGs. Here, we define coastal sustainable development as human activities and planning processes that contribute across the SDGs and minimized trade-offs between SDG objectives. We are explicitly concerned with development outcomes across multiple SDG outcomes as sustainable development is a multi-criteria problem, and we focus on the SDGs since they are the most widely accepted definition of sustainable development. While a comprehensive and wide-spanning systems approach is clearly necessary to address coastal sustainability issues, this can be a very complex task. Achieving this integrated policy requires a transition away from current institutional regimes, and navigating this transition is often not intuitive (Blythe et al., 2018; Bennett et al., 2019). Frameworks to help structure governance systems to achieve sustainability initiatives have been developed in political science as a planning and research framework for transitioning from current governance systems to integrated policy systems in order to achieve sustainable development objectives (Kemp et al., 2007; Loorbach, 2007; Rotmans and Loorbach, 2009; Loorbach, 2010; Broman and Robèrt, 2017). However, frameworks for structuring governance systems around sustainability goals have not had wide uptake in SDG planning or for environmental governance planning in general (but see, Singh, 2020; Singh et al., 2021).

Recent research focused on interlinkages between UN SDG targets—the most comprehensive contemporary set of multi-disciplinary development objectives—has highlighted the fact that there are both direct and more complex tradeoffs and co-benefits across different policy objectives (Nilsson et al., 2018; Singh et al., 2018, 2021). In some cases, making progress on coastal sustainability can directly contribute to SDG areas such as food security (SDG 2), longer term economic and employment opportunities (SDG 8), and improved ecosystem states (SDGs 14 and 15) (Blanchard et al., 2017; Lotze et al., 2019). In other cases, however, progress can be highly dependent on actions taken on other SDGs, such as how the revenues generated from sustainable coastal development can promote poverty reduction and habitat restoration depending on how these revenues are distributed and invested (Singh et al., 2018).

Beyond determining which SDG topic areas are needed to promote a given policy goal (and which SDG topic areas can be detrimental for a given goal), governing the land-sea interface will require an understanding of *what* management activities to conduct and *how* to best achieve these activities. Aligning management activities in the context of interlinked SDG topic areas requires coordination in a governance system (Singh, 2020; Singh et al., 2021). Coastal systems are often governed by multiple institutions siloed across the multiple sectors of coastal systems (e.g., fisheries, forestry, agriculture) (Halpern et al., 2008). Siloed management can lead to counterproductive outcomes when institutional missions and activities do not align, or when side-effects from one sector affect another (Cottrell et al., 2018, 2019). Though a substantial literature has been developed addressing how siloed management can lead to counterproductive and uncoordinated results, what is missing is a systematic framework to determine how to align institutions to achieve coordinated action toward desired goals (Singh, 2020). Here, we offer a strategic land-sea interface governance framework based on the sustainable transitions and policy coherence literatures, and provide case studies viewed through the lens of this framework.

ALIGNING GOVERNANCE IN LAND-SEA INTERFACE FOR SDGs

Coastal settings have the potential for complex dynamics across all social, economic, and biophysical dimensions of the SDGs, as they include both marine and terrestrial ecosystems with dense human population, and a diverse set of resource users. Determining how SDGs interlink in these regions is therefore very important given the numerous potential interactions available to explore.

The SDGs are listed as 17 discrete goals, each with a set of more specific targets. Interlinkages between the goals are recognized and the SDGs were written to be "indivisible," even if these linkages are not explicitly included in the SDG Agenda (UN, 2015). Identifying and exploring interlinkages is thus vital for understanding how pursuing specific SDGs can affect others and such assessments have been conducted for the oceans (Singh et al., 2018), energy systems (Nerini et al., 2018), eliminating hunger (Rasul, 2016; ICSU, 2017), increasing human health (Bekker et al., 2018), and more general SDG areas of interest (Pradhan et al., 2017). Importantly, however, general knowledge on linkages is not enough to guide a transition to sustainability without deeper information on the scale of change needed to achieve particular or multiple targets (Singh et al., 2021).

Besides the diversity of sustainable development dimensions, governing coastal regions has to contend with existing governance systems that are built on quasi-non-overlapping jurisdictions. Governments and industries are highly siloed, where different sectors of the economy are regulated and acted on by distinct organizations (Halpern et al., 2008). For example, most governments have distinct regulatory organizations that deal with oceans versus terrestrial lands, and between fisheries and farming, even though these different sectors are highly related (Cottrell et al., 2018). Beyond the fragmentation of governance along lines of economic sectors, there are often jurisdictional distinctions between national government and subnational government agencies. For example, to address issues of marine pollution in British Columbia, Canada, a successful initiative would likely need to work between Fisheries and Oceans Canada (a federal department regulating fisheries) Transport Canada (a federal department regulating shipping), Agriculture Canada (a federal department regulating agricultural production), the Ministry of Agriculture (a provincial ministry regulating agricultural lands and production), local government planning organizations, and others.

We propose a framework built on the theoretical perspectives of policy coherence and sustainable transitions. In so doing, we have created a framework that operates across three dimensions; horizontal policy coherence; vertical policy coherence, and programmatic alignment. Policy coherence is theoretically an attribute of policy that systematically reduces conflict and promotes synergies between and within different policy actors and institutions to achieve the outcomes associated with agreed policy objectives (Nilsson et al., 2012). Specifically, working across agencies and organizations that operate at the same scale (e.g., national) is often called "horizontal policy coherence" whereas working across agencies that operate across different scales (e.g., between national and sub-national) is often referred to as "vertical policy coherence" (Nilsson et al., 2012).

Horizontal and vertical policy coherence across agencies needs to consider the programmatic alignment from vision to implementation. To address programmatic alignment, we relied on theoretical framing of sustainability transitions, specifically transition management theory. The literature on societal change and governance systems to promote sustainability identify three governance levels to consider: (1) the strategic level of vision development and goal setting; (2) the tactical level of institutional interactions; (3) the operational level of implementation (Loorbach, 2010; Singh, 2020). Where organizations have disjoint governance actions across these three levels, any sustainability initiatives my fail. For example, if an environmental NGO and a community-based organization share broad goals of ocean conservation, but the local group is not included in decisions and responsibilities of setting up an MPA, the MPA may suffer from a lack of local-buy-in and enforcement, especially if the local group supports alternative conservation actions (Christie, 2004). This governance approach – alignment across sectoral (horizontal), policy resolution (vertical) and policy actors (from goals to institutions and operations) – can be a useful approach to integrate systems analysis into planning (**Figure 1**).

The relationship among these three scales can help determine appropriate policy strategies to achieve sustainable development (Kemp et al., 2007; Loorbach, 2010), as understanding how various dimensions of sustainable development are related to each other (strategic actions), can inform how to structure governance institutions, and the way that governance institutions are structured (tactical actions) can realize which relationships among sustainable development dimensions are achievable and which ones are not. The types of institutions and their relationships to each other also regulates the policy interventions that can be undertaken (operational actions), while identifying effective interventions can determine new potential collaborations between institutions. This model is structured to align governance coherence both from top-down and bottom-up perspectives. Top down processes would help structure and steer activities that occur below, while bottom up processes would instruct higher levels about the effectiveness of projects and policies. This kind of reflexive feedback allows for self-correction in governance structure and treats the process of achieving sustainable development as a complex adaptive system (Kemp et al., 2007; Loorbach, 2010). Below, we provide four case studies of land-sea governance problems that explore these situations. We detail case studies across a range of countries – including small island states, a developing coastal country, and a developed coastal country – to document the diversity of settings that can benefit from the approach outlined here.

CASE STUDIES

Case Study 1: Planning Institutional Network to Support Sustainability Goals in Aruba – Using the Strategic Scale to Inform the Tactical Scale

Problem Context

Aruba is a small island state in the southern Caribbean, with 90% of annual GDP is derived from coastal tourism (WTTC, 2019). A large proportion of Aruba's island surface has been transformed for tourism infrastructure (Barendsen et al., 2008). Aruba's coastal development to date has led to marine pollution problems as well as coastal habitat loss, such as through mangrove removal (Oduber et al., 2015). Though marine tourism has such high economic value, it is not necessarily sustainable and it does not focus on a healthy marine ecosystem but rather having warm, clean, sand beaches (Singh et al., 2021). Aruban institutions responsible for managing the land-sea interface within Aruba operate in a siloed fashion, and initiatives from some may counteract the goals of others (Singh et al., 2021). For example, much of the pollution problems come from coastal and community development, which are regulated by the Aruba Tourism Agency and Department of Economic Affairs and Infrastructure, who promote coastal tourism and development. Yet, tourism is also dependent on clean waters, so regulating marine pollution is beneficial, and requires alignment among agencies that can help regulate pollution.

The Sustainable Development Objective

Aruba has a SDG commission which indicated that SDG 14 (Life Below Water – the Ocean Goal) is a priority for the island state, and hosted a workshop to determine policy priorities

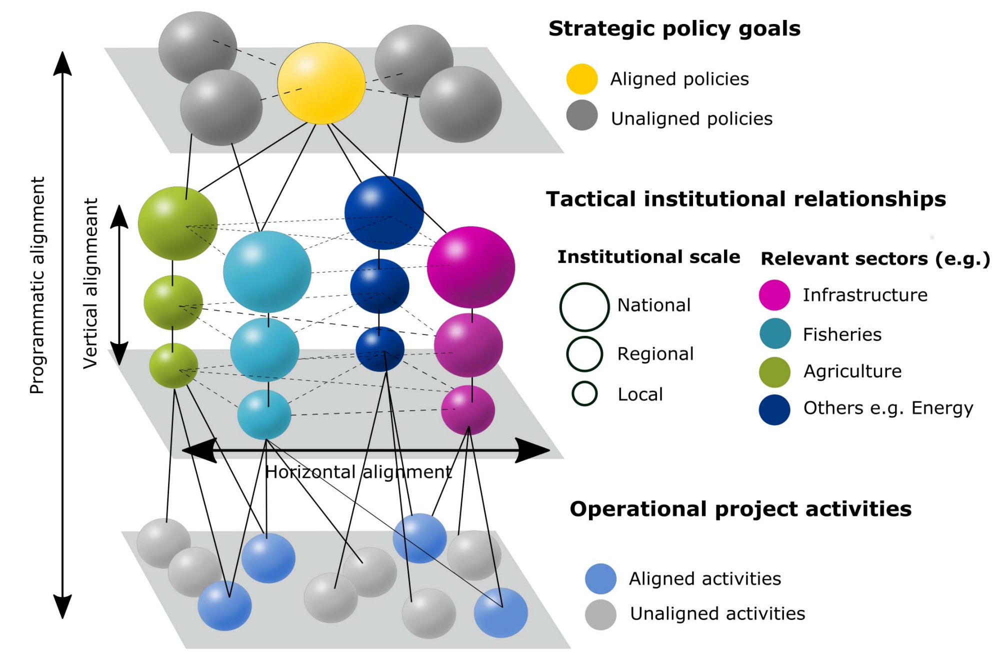

FIGURE 1 | A framework for aligning governance actors and activities for sustainability in land-sea interfaces. This framework considers programmatic alignment (connections between strategic, tactical, and operational components), vertical alignment (connections between tactical institutions acting at different scales), and horizontal alignment (connections within components at the same governance level). Unaligned activities are those which are not connected across all three types of alignment (programmatic, vertical, horizontal).

to achieve sustainable oceans (Singh et al., 2021). Through an SDG interrelationship exercise, SDG 14.1, the target to reduce marine pollution, was determined to be the SDG target that was a pre-requisite across the largest number of SDG ocean targets. Consequently, it was found to be the most important pre-requisite for achieving the largest number of other SDG targets across ocean targets. Determining how to achieve the target of reducing marine pollution, and what actors are needed to work together to achieve it, can be seen as a priority for the small island nation.

Planning Vertical and Horizontal Coherence at the Tactical Scale to Meet Priorities at the Strategic Scale

With a priority target determined, workshop participants conducted another SDG interrelationship exercise, this time to look at what SDG targets promote or detract from achieving SDG 14.1: reducing marine pollution. This exercise was done to explore the multiple policy options and determine the policy requirements needed to effectively manage marine pollution. In effect, this exercise explored the Strategic scale of the transition management framework. Results for this exercise are presented in **Table 1**.

With the interlinkages supporting SDG 14.1 determined across the land-sea interface, workshop participants could make informed recommendations of how Aruban institutions should be structured in order to take advantage of the identified co-beneficial relationships (exploring the tactical scale of transition management framework). First, participants created a scenario where only direct institutional regulation for SDG achievement is considered (SDG interactions do not shape the structure of institutions). Second, participants created a scenario whereby the collaborative structure of institutions was guided by SDG interlinkages that support the achievement of SDG 14.1 (as well as the SDG target that posed a potential trade-off with SDG 14.1). In the first scenario, participants determined six Aruban agencies that collaborate to work toward SDG 14.1, including the Directorate of Nature and Environment (DNE), and all six equally collaborate (determined by the number of links with other institutions, **Figure 2**). However, when SDG interlinkages were considered to support SDG 14.1, a more complex institutional network was produced (**Figure 2**). In this scenario, the three most important Aruban agencies (in order, according to centrality measures) were the Social and Economic Council (SEC), the Department of Economic Affairs (ECO), and the Aruba Tourism Authority (ATA, **Figure 2**), while the DNE was connected to fewer institutions and so might be less influential in coordinating actions across institutions.

Case Study 2: Land-Sea Co-benefits of Climate-Smart Agriculture – Using the Operational Scale to Inform the Tactical Scale

Problem Context

Dominica is a small Caribbean island state that has historically relied heavily on agricultural production for its economy – agriculture has represented 12–16% of total GDP since 2010 (Worldbank, 2021) – and over 60% of the population live in the coastal zone. As the northernmost of the eastern windward islands, Dominica's location exposes it to a range of natural hazards, particularly hurricanes and tropical storms (Barclay et al., 2019). Extreme weather has had a huge influence on natural resource use on the island and has shown capacity for shifting livelihood activities from farming to fishing when agricultural shocks occur (Ramdeen et al., 2014; Cottrell et al., 2019). Banana production has been the dominant crop in Dominica throughout the 1900s (Barclay et al., 2019) but the vulnerability of monocrop dependence has been highlighted by two notable events – Hurricane David in 1979 which led to sudden and widespread crop damage, and the dissolution of historical trade deals with the EU in the 1990s which resulted in a steady decline of banana production (Cottrell et al., 2019). On both occasions, rapid increases in fisheries landings occurred following agricultural collapse, and after Hurricane David these fishing surges were followed by sudden declines in catch thought to be linked to overfishing in nearshore waters (Cottrell et al., 2019). Dominica has committed to protecting "Life below water" (SDG14) through reducing overcapacity, bycatch and discards, and unregulated fishing (SDG 14.2 and 14.4) and increasing marine protected areas (SDG14.5) through its partnership in the Western Central Atlantic Fisheries Commission. However, continuing to meet these targets will require strengthening the resilience of the agricultural systems to guard against such unpredictable shifts between sectors under a future of projected increasing volatility.

TABLE 1 | The SDG targets determined to contribute to (or detract from) the achievement of SDG 14.1 in Aruba.

SDG target	Description	Interrelationship type
6.3	Wastewater management	Prerequisite co-benefit
12.5	Reduction in waste generation	Prerequisite co-benefit
11.4	Protect cultural and natural heritage	Prerequisite co-benefit
12.4	Environmentally sound management of chemicals and waste	Prerequisite co-benefit
9.4	Retrofit industry infrastructure for sustainability	Prerequisite co-benefit
11.6	Reduce per-capita impact of cities	Prerequisite co-benefit
17.14	Assist developing countries in attaining long term debt sustainability	Prerequisite co-benefit
8.4	Improve resource efficiency in economic growth	Prerequisite co-benefit
13.2	Integrate climate change measures into national planning	Prerequisite co-benefit
17.17	Transfer of environmentally sound technologies to developing countries	Potential co-benefit
13.1	Strengthen adaptive capacity to climate-related hazards	Potential co-benefit
8.2	Economic diversification and technological upgrading and innovation	Potential co-benefit
16.4	Combat organized crime	Potential co-benefit
16.10.	Public access to information	Potential co-benefit
13.3	Improve education on climate change mitigation	Potential co-benefit
10.1	Sustain income growth of bottom 40%	Potential trade-off

The targets are shown in descending order of certainty among the workshop participants who determined the linkages from the SDGs to the SDG 14 targets.

The Sustainable Development Objective

Dominica is already in an extraordinary position for transition in its agricultural sector. Following the damage of Hurricane Maria in 2017, the Dominican government published the Emergency Agricultural Livelihoods and Climate Resilience Project [Government of the Commonwealth of Dominica (GCD), 2018]. The government has committed US $16.5 million toward the DEALCRP to restore a productive base for crop- and livestock-based livelihoods and business. However, executing the DEALCRP successfully requires coherence between government and non-governmental actors, which our framework can help with.

Planning Vertical and Horizontal Coherence at the Tactical Scale to Carry Out a Project at the Operational Scale

Referencing key environmental and social challenges for agricultural resilience documented in the DEALCRP as well as peer reviewed literature, we outline how agroforestry (the co-cultivation of crops with shade trees) can work toward mitigating these challenges (planning on the operational scale), and link these elements of an agroforestry program to the governance institutions that are needed to work together to effectively carry out this program (the tactical scale). We also outline anticipated SDG co-benefits of successfully implementing agroforestry in Dominica (**Figure 3**).

Food resource productivity and livelihood vulnerabilities on Dominica are driven by numerous factors. High dependence on a single crop is reinforced by the rapid recovery time bananas can provide after disaster combined with economic incentives for regrowth from the windwards island insurance scheme and the productivity of the crop itself (Mohan, 2017b). Banana crops are known to be more susceptible than many other crops to wind damage, with root dislocation and moisture stress possible even in weak tropical storms (Mohan, 2017a). Dominica's mountainous terrain is also challenge for cultivation in places, with soil erosion during times of heavy rainfall leading to landslides and flooding, and there is recognition of the need for greater soil stabilization than current management practices provide [Government of the Commonwealth of Dominica (GCD), 2018]. These factors are all in addition to Dominica's vulnerability from its physical position in the Caribbean.

Yet integrating bananas into an agroforestry setting could reduce many of these vulnerabilities while delivering multiple co-benefits. Banana agroforestry with fig, mango and Albizia species (for timber) have shown great promise for increasing soil fertility in Uganda, for example (Ssebulime et al., 2019). Shade trees provide sources of income from timber (even after storm damage) and fruits throughout the year, and leaf litter for compost reducing the need for agrochemicals. Similar benefits from livelihood diversification have been demonstrated when growing bananas alongside coffee too (Reay, 2019). If combined with silvopastoral practices (livestock integrated into fruit and timber trees), livestock provide another income stream and

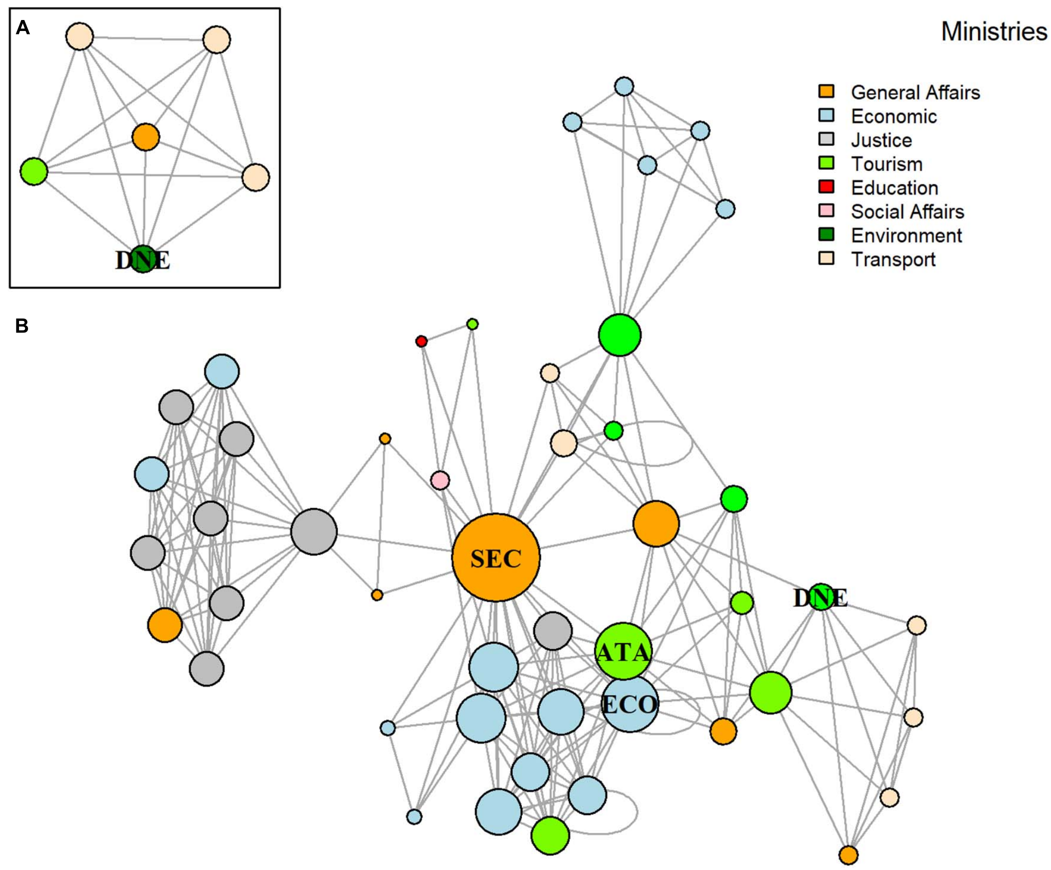

FIGURE 2 | Network diagrams of the institutional structures needed to manage SDG 14.1, considering only direct regulation management **(A)**, and considering all SDG targets that contribute to SDG 14.1, with the institutions that manage these supportive SDG targets **(B)** for Aruba. When only direct regulation is considered, few institutions have an equal role in achieving the SDG target, including the Directorate of Nature and the Environment (DNE). When interlinkages are recognized between SDG targets then more institutions are involved in achieving the SDG target and some have a more central role in coordinating policy and action. In the case of achieving SDG 14.1 for Aruba, this includes the Social and Economic Council (SEC), the Department of Economic Affairs (ECO), and the Aruban Tourism Authority (ATA). Institutions are clustered based on their association among other institutions, and links vary in length based on the distance between institutions they are connecting.

a source of manure (Waldron et al., 2017). Agroforestry can increase above and below ground biomass, reducing surface run-off and binding soils together while buffering the standing crops' exposure to high winds during a storm (Waldron et al., 2017). Forested areas are already recognized for their importance in erosion control in Dominica [Government of the Commonwealth of Dominica (GCD), 2018], so spreading these benefits into food production systems suffering from soil erosion problems is a logical step. In making agricultural systems more resilient in the face of meteorological shocks, Dominica can prevent unpredictable shifts in resource use seen in recent years that threaten marine sustainability targets (SDG 14). But in doing so also generates co-benefits among multiple goals for poverty and hunger reduction (SDG1 and 2), economic development (SDG 8), responsible production and consumption (SDG12) and reduces terrestrial habitat fragmentation with numerous benefits for wildlife (SDG 15) (**Figure 3**).

Successfully realizing these benefits will require effective collaboration among divisions of the Ministry of Blue & Green Economy, Agriculture and National Food Security (MEAF), and the Ministry of Environment Climate Resilience, Disaster Management and Urban Renewal (MECDU), as well as the many private small-scale landowners who engage in agriculture. For the Division of Agriculture in the MEAF, a shift toward agroforestry aligns strongly with its Coffee and Cocoa program which is currently rehabilitating existing plantations, and expanding production over the island to meet objectives of increasing exports, income, and employment (Division of Agriculture, 2021). Close communication with the Forestry, Parks, and Wildlife Division within MEDU would be needed at a number of levels. Firstly, to ensure that suitable companion crops could be grown alongside bananas and that timber resources were able to be optimally utilized within State and private lands. Indeed, current operations to thin State forests provide an opportunity to enrich existing plantations with diverse and profitable fruit crops (Division of Forestry, Parks, and Wildlife, 2021). Second, to ensure agroforestry expansion was attractive, profitable, and feasible for private landowners in parallel with

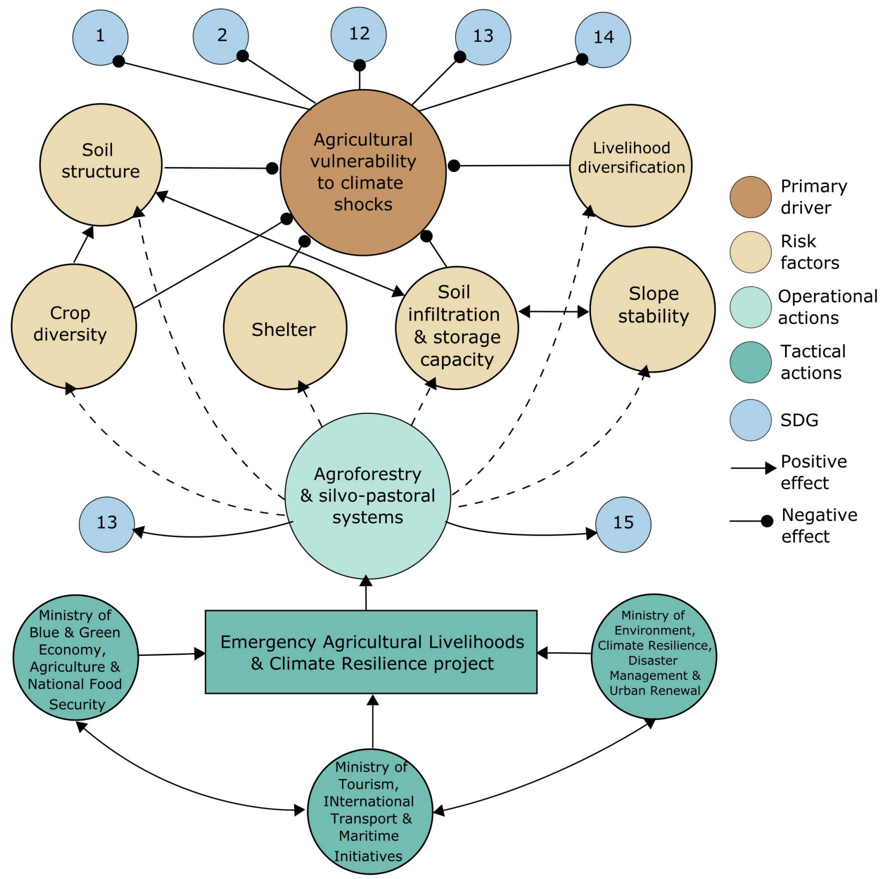

FIGURE 3 | The role of agroforestry for addressing land-sea switches and sustainability in Dominica. With agricultural vulnerability to climate shocks identified as the primary driver of challenges to sustainable development goals, we show how agroforestry at the operational scale can directly and indirectly address sustainability challenges and inform tactical institutional collaboration.

existing responsibilities of the FWPD's silviculture unit. Third, FWPD's aims to minimize soil erosion and maximize the value of forestry units for wildlife refugia could be tracked alongside monitoring agroforestry productivity.

Further, such integrative farming practices can be a feature of agro- and ecotourism programs rather than seen as a source of conflict, enhancing their economic potential (Hakim et al., 2019) and highlighting the need for collaboration with the Ministry of Tourism, International Transport, and Maritime activities[1]. Finally, effective temporal tracking of livelihood mobility between agriculture and fisheries during new fisher registration and agricultural surveys will be necessary for empirical evidence of changes in agricultural resilience through time, and will require efficient data sharing among agriculture and fisheries divisions of the MEAF. By addressing the major challenges that face agriculture and identifying a solution that strengthens and aligns current programs to meet environmental and social objectives – promoting widespread agroforestry as a key operational activity can inform necessary tactical design for effective land-sea governance in Dominica.

[1]https://tourism.gov.dm/

Case Study 3: Developing a Common Strategy for the Great Barrier Reef From Diverse Management Agencies – Using the Tactical Scale to Inform the Strategic Scale

Problem Context

Australia's Great Barrier Reef is managed by agencies at federal and state levels, whose strategic goals for the reef do not always align. Some agencies have a clear preservationist conservation mandate while others are interested in promoting development opportunities (**Table 2**). While management agencies can potentially find an acceptable balance between these two goals, in practice, conflicting management and trade-offs occur. The Great Barrier Reef Marine Park Authority (GBRMPA), is the federal agency primarily responsible for managing, zoning, and permitting activities related to the reef since 1975 (Day and Dobbs, 2013). The Great Barrier Reef was designated as a UNESCO World Heritage Area in 1981 and the federal marine park covers 99% of the Great Barrier Reef Region, while the remaining 1% is under the jurisdiction of The State of Queensland (Day and Dobbs, 2013).

Beyond the boundaries of the GBR, including the larger land-sea interface, growth in mining and industry have led to an increase in development of ports and shipping, managed by the Department of Infrastructure, Transport, Cities, and Regional Development (**Table 2**). Recent proposals for development of coal mines and adjacent ports within the Great Barrier Reef Marine Park (GBRMP) have been met with opposition by scientists who suggest that such development would lead to an increase in both locally derived water quality issues as well as contributing to climate change by further development of fossil fuels (Hughes et al., 2017). The biggest local threat to the inshore reef is water quality (MacNeil et al., 2019), while the greatest overall threats are related to climate change – causing increased water temperatures and bleaching events – which are global in nature and require high level action and international cooperation to address them (Hughes et al., 2017). Much of the water pollution is related to catchment runoff from adjacent sugar cane farms which lead to increased sediment, nutrient, and pesticide loads to the GBRMP (MacNeil et al., 2019). The State of Queensland manages water quality that flows to the Great Barrier Reef, and has targets to reduce sediment and nutrient loads in their draft water quality improvement plan for 2017–2022 (Queensland, 2017).

The Sustainable Development Objective

The conflicting priorities among agencies managing the GBR are a direct result of the conflicting strategic directions of unaligned institutions. Activities in the Great Barrier Reef are regulated by complimentary legislation and joint field management, and permits between federal and state governments (Day and Dobbs, 2013; **Table 2**). The GBRMPA employs a multiple-use marine spatial zone to separate conflicting activities.

In order for the Great Barrier Reef to persist into the future (SDG 14) and keep some development and conservation opportunities available (SDG 8), better alignment among regulatory bodies will be needed. In other words, for the strategic goals to be achievable and not contradictory, the tactical systems that support it need to be complementary.

Arriving at a Cohesive Overall Goal at the Strategic Scale Through Shared Planning at the Tactical Scale

A major conservation challenge identified by the GBRMPA and affiliated institutions concerns the synergistic impacts among ocean warming, the subsequent increased frequency of bleaching events, and the disproportionate impacts these events have on reefs with poor local water quality. While addressing climate change impacts of ocean warming are beyond the sole capacity of federal and state agencies, addressing water quality issues will require cooperation between The State of Queensland and the GBRMPA as well as discussion about the types of land-based industries and activities that are compatible with minimizing impacts on the Great Barrier Reef (**Table 2**). Concessions by the agricultural and mining industries will undoubtedly need to be made to mitigate impacts on the Great Barrier Reef and the associated tourism industry, requiring high level vision at the strategic scale to steer the development of these industries. At the same time, mining and agriculture cannot be expected to end in the region. Instead, shared planning processes between the GBRMPA, state agencies, and mining and agriculture agencies can determine priority areas and activities for different land-and-sea uses (**Table 2**).

Given the often competing interests of the regulatory bodies, it might be helpful to identify a common shared vision that all agencies can contribute to. Using a structured decision-making process, all relevant agencies and stakeholders can develop a common understanding of how the system operates, propose a series of alternative development trajectories (and associated consequences), and evaluate trade-offs of various scenarios (Gregory et al., 2012). Though the likelihood that any resulting plan will fully satisfy all stakeholders is minute, research indicates that stakeholders who participate in planning processes generally consider the resulting decisions as more legitimate as those who do not (Jentoft, 2000).

Case Study 4: Planning a Way to Address Illegal Fishing for Mexican Small Scale Fisheries – Using Operational Challenges to Inform the Tactical and Strategic Scales

Problem Context

Santa Cruz de Miramar, Mexico, is a community of around 1500 people and is economically dependent on a variety of coastal industries, including coastal tourism and artisanal fishing. It is the largest producer of oysters in the state, and a co-management scheme with a local cooperative of around 70 licensed fishers is responsible for much of the fishery. The cooperative was set up in the 1920s, and though it was weakened during a strong neoliberal push in the 1990s (Basurto et al., 2013), it is being strengthened again, aided by local researchers and NGOs. However, despite the recent gains in local management capacity, the fishery has faced a number of challenges that local institutions cannot respond to, namely overharvesting, poaching, and sales of illegally fished product.

The Sustainable Development Objective

The problems with particular fisheries management programs (operational scale) – namely the enforcement of illegal fishing – was evaluated to look for ways in which institutional roles and collaboration (the tactical scale) and changes to broad policy along the land-sea interface (the strategic scale) could provide solutions (De la Cruz-González et al., 2018).

Organizing Institutional Actors in the Tactical Scale and Re-evaluating the Goals of the Strategic Scale to Address Programs at the Operational Scale

To understand the causes and potential solutions around this problem, the cooperative partnered with the National Fisheries Institute (INAPESCA, the science branch of the federal fisheries management in Mexico) to undertake research to inform management strategy and coordination. This included mapping local oyster beds and analyzing population structures and market dynamics, which led to the implementation of individual daily allowable catches, minimum size limits, bed rotations and seasonal closures. This is all implemented, monitored, and enforced by the cooperative itself, including setting punishments for members who break rules, and evidence to date shows significant increases in catch and in value due to larger sizes and harvest timed to coincide with higher seasonal prices (De la Cruz-González et al., 2018).

As part of a SWOT (strengths, weaknesses, opportunities, threats) analysis of the oyster fishery (De la Cruz-González et al., 2018), local fishers identified "unclear institutional mandates and obligations" as a major weakness of the fishery. Cooperative fishers perceive federal institutions as responsible for regulatory services, including researching the status of local stocks and issuing fishing licenses. State agencies are perceived as operational agents, financing projects and monitoring quality controls. Local authorities are perceived as monitoring and responding to illegal fishing and preventing sales of illegally caught seafood, with a narrow scope but essential tactical actions. Local authorities, therefore, are perceived to be responsible for factors they have little capacity to resolve, and which state and federal agencies are mandated to address (i.e., issues of enforcement and organized crime). There are similar examples from around the world that show this type of interplay, where tactical and strategic levels of management operate (or are perceived to operate) almost independently of each other despite obvious overlaps in general goals. An active role of fishers and community leaders is crucial for propelling local sustainability actions but can be challenged by a lack of support or at least tacit approval of higher-level governance institutions. There is an increasingly strong and cross-scale movement to strengthen governance in support of artisanal fishers [Food and Agriculture Organization (FAO), 2015], and a key component is a greater willingness of governments and institutions to share and devolve management power to communities, recognizing unique contexts that require unique knowledge and solutions even within broader national goals (Lozano et al., 2019).

While most current attention for sustainable fisheries is focused on SDG 14 at a strategic scale (ensuring suitable conditions to promote life below water and manage extraction), it is clear that fisheries-related issues often have ultimate causes well beyond the purview of fisheries managers. In the case study presented here, two key additional strategic topics were recognized as important to address fishery sustainability (De la Cruz-González et al., 2018). First, increasing coastal development and pollution from increasing tourism and urbanization are posing a risk to fishery productivity. Second, the lack of employment alternatives and lack of access to wider seafood markets leads to greater pressure on local fish stocks. In the specific context of the SDGs, continued fishery sustainability (SDG 14) would benefit from a greater integrated strategy designed to promote the co-benefits and avoid trade-offs with coastal development (SDG 9), sewage treatment (SDG 6), urban design (SDG 11), and economic opportunities (SDG 8). Because none of these issues are within the purview of fisheries management institutions, interfacing across institutions is evidently critical for success and this can indeed build on the SDGs themselves (Singh et al., 2021).

TABLE 2 | Agencies, their scale of operation, and stated priorities relevant to the management of the Great Barrier Reef (GBR).

Agency	Scale of operation	Stated priorities relevant to GBR
Great Barrier Reef Marine Park Authority	Federal	Care and protection of the Great Barrier Reef Marine Park – issues permits for various forms of use of the marine park, and monitors usage in the park to ensure compliance with rules and regulations
State of Queensland – Economic Development	State	Specialist land use planning and property development unit – works with local governments, industry, and the community to identify growth opportunities
Department of Sustainability, Environment, Water, Population, and Communities	Federal	Regulation of activities including world heritage values
Queensland Parks and Wildlife Service	State	Protect and manage Queensland's parks, forests and the Great Barrier Reef
United Nations Educational, Scientific, and Cultural Organization World Heritage Site Status	International	Legally protected by international treaties and labelled as a protected zone
Australian Fisheries Management Authority	Federal	Management and sustainable use of fisheries resources
Department of Infrastructure, Transport, Cities, and Regional Development	Federal	Regulatory framework for shipping and environmental and safety regulation

Stated priorities were obtained from relevant organizational websites.

CONCLUSION

Promoting sustainable development at the land-sea interface requires a coordinated governance structure that can effectively regulate and act within complex social-ecological systems. Achieving this coordination requires a systematic framework to align strategic priorities, tactical organization, and operational programming. Such a framework provides opportunities for both researcher and policymakers to engage in the process of sustainable development: for researchers it sets out particular research questions around particular planning scales (such as determining how goals fit together at the strategic scale, or evaluating the feasibility of promised activities given the institutional network supporting it at the operational and tactical scales). This research can build on innovative methods used to track relationships between sustainability goals, such as the Sustainable Development Goals. For policymakers, the benefit of the framework is structuring decisions at key governance levels and designing policy and programs that will minimize counterproductive activities and maximize chances of success. Despite the potential of this framework, it has not been formally tested. Though we explore four case studies using the framework in this study, this study is limited by retroactively interpreting cases through the lens of the framework. Future studies to develop this work should use this framework in active governance planning processes. Here, we propose the use of this framework for complex governance problems such as in the Great Barrier Reef – this case may benefit from a process guided by this approach, which would be timely given the multiple issues the region faces. Beyond this case, explicitly focusing on the alignment of various levels of governance scales can be applied across contexts, including in strategic planning and program development in Small Island Developing States, iconic marine areas in the world's most developed countries, and fishing communities in coastal developing nations. Research and policy developed with such a governance framework can be particular important for coastal systems, which are arguably the most complex social-ecological systems on earth, and which are so important to achieve the Sustainable Development Goals.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

GS conceived of the manuscript. GS and RC created the figures. All authors wrote the manuscript, and each author contributed one case study.

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Distribution of Cetaceans in the Canary Islands (Northeast Atlantic Ocean): Implications for the Natura 2000 Network and Future Conservation Measures

Inma Herrera[1,2], Manuel Carrillo[3†], Marcial Cosme de Esteban[1] and Ricardo Haroun[1]*

[1] BIOCON, IU-ECOAQUA, Universidad de Las Palmas de Gran Canaria (ULPGC), Parque Científico y Tecnológico Marino, Telde, Spain, [2] Marine Biology Station of Funchal, Faculty of Life Sciences, University of Madeira, Funchal, Portugal, [3] Red Canaria de Cetáceos Varados, Canarias Conservación - Gabinete Técnico de Medio Ambiente, La Laguna, Santa Cruz de Tenerife, Spain

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Taiki Adachi, University of St Andrews, United Kingdom

***Correspondence:**
Inma Herrera
inma.herrera@fpct.ulpgc.es; inma_herri@hotmail.com

† Deceased

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The waters of the Canary Islands are considered a hotspot for marine biodiversity, especially regarding cetacean species. Based on this fact, this study pays attention to the spatial distribution pattern of cetacean species and the conservation role of the Natura 2000 Network, a set of Special Areas of Conservation (SACs), which were defined mainly based on data compiled in 1996, under the framework of the European Habitats Directive. In recent years, the declaration of conservation areas for cetaceans between the Tenerife—La Gomera Islands by two global conservation programs, Mission Blue ("Hope Spots") and Whale Heritage Site ("Whale Sanctuary") sent clear signals of scientific and social interest to promote better protection of the cetacean species in the Canary Islands. The main aim of the designated SACs is the conservation of its biological and ecological diversity, ensuring the long-term survival of the target species in the waters around islands. In this case, the enactment of the SACs was based only on the sparse data available for the common bottlenose dolphin, *Tursiops truncatus*. This study shows that the spatial distribution of cetaceans in the Canary archipelago generated from a large database of cetacean sightings, from 2007 to 2018. The results obtained show the main marine areas where the different cetacean species are distributed around the different islands of the archipelago. The spatial distribution maps of the cetacean species, when compared with the existing SACs of the Natura 2000, show the need to extend these SACs into the open sea to include more cetacean species and a larger number of individuals for better conservation of the endangered marine mammals. As a consequence, some suggestions were proposed to improve and update the role of SACs in European Northeast Atlantic waters as a key environmental tool for cetacean conservation. The data supporting the recent declarations of these two new milestones the "Hope Spot" and the "Whale Sanctuary" enhance more keystone information to promote a large marine protected area in the Eastern Atlantic Ocean, such as the "Macaronesian Biodiversity and Ecological Migration Corridor for Cetaceans," a conservation figure that has been already proposed in the scientific literature as a deserving candidate of governmental regulations and policies by Portugal and Spain; it would also require joint cross-border cooperation efforts for marine spatial planning.

Keywords: dolphins, whales, East Atlantic Ocean, oceanographic features, abundance, conservation corridor, marine spatial planning

INTRODUCTION

The variations in the oceanographic, hydrological, and topographic features of the oceans create a wide heterogeneity of habitats, favoring the high diversity of cetaceans that is observed in the Northeast Subtropical Atlantic Ocean, especially in the waters around Canary Islands, an area considered a hotspot for marine biodiversity. At present, 30 species of cetaceans, out of the 90 described worldwide (Jefferson et al., 2015), have been recorded in these oceanic waters (23 odontocetes and 7 mysticetes; **Supplementary Table 1**). These species make the area one of the most diverse places for cetaceans; it is also the largest in Europe. The Canary Islands are an oceanic volcanic archipelago in the Northeast Subtropical Atlantic Ocean, comprising eight islands with a total surface area of 7,273 km^2, a coastline of approximately 1,581 km, and an exclusive economic zone (EEZ) of approximately 494,192 km^2.

The complex oceanographic characteristics of the Canary Islands are determined by a combination of factors, for example, the filaments—nutrient-rich waters—which originated in the upwelling system of the Northwestern African shores (Cape Juby, Cape Ghir, and Cape Bojador) that reach the Canary Islands. These filaments have an essential biological function, as they transport fish, cephalopods, and crustacean larvae—food for marine mammals—from the African coast to the coastal waters of the Canary Islands (Rodríguez et al., 1999, 2004; Bécognée et al., 2009; Landeira et al., 2017).

Although the waters of the Canary Islands are considered a hotspot for marine biodiversity, for cetaceans in particular, this does not exempt marine mammals from being subjected to pressure and threats. Some of these threats are due to natural causes, such as predation, but, for the most part, they are consequences of direct or indirect anthropogenic activities (Parsons, 2012), including by-catch, competition with fisheries, habitat degradation (Ruíz de la Rosa et al., 2015), marine pollution (Baulch and Perry, 2014; García-Álvarez et al., 2014, 2015; Puig-Lozano et al., 2018), acoustic/ noise disturbance (Aguilar de Soto, 2006; OSPAR, 2009), stranding (Tejedor and Carrillo, 2018; Puig-Lozano et al., 2020), and maritime traffic, including high-speed ferries—nearly 60% of sperm whale deaths are due to ship collisions in the Canaries (Arregui et al., 2019). These marine mammals are a highly mobile species; their distribution areas cover extensive oceanic areas, which pose a major challenge for their conservation. All cetaceans found in European Union waters receive protection under the Habitats Directive (Council Directive 92/43/EEC of 21 May 1992) and the Marine Strategy Framework Directive (Directive 2008/56/EC). These directives mandate both updating the conservation status and the monitoring of cetacean populations (e.g., distribution, abundance) as well as the adoption of conservation measures if the population status is considered unfavorable (Santos and Pierce, 2015).

Based on cetacean conservation and protection as per the Habitats Directive, this study pays attention to the Natura 2000 Network, a European network of natural areas whose aim is the conservation of the biological and ecological diversity of Europe, taking into account the economic, social, and cultural requirements of its different regions. Additionally, the main goal is to ensure the long-term survival of different species and habitat types in Europe, preventing the loss of biodiversity. The Natura 2000 Network is the main nature conservation instrument used by the European Union.

The Natura 2000 Network involves the natural habitats and species listed in Annexes I and II of the Habitats Directive (Council Directive 92/43/EEC of 21 May 1992), where only two species of cetaceans, the common bottlenose dolphin (*Tursiops truncatus*) and the harbor porpoise (*Phocoena phocoena*), as animals of community interest for whose conservation it is necessary to designate Special Areas of Conservation (SACs), and all other cetaceans as animals of community interest require strict protection. In December 2001, the European Commission approved the designation of 174 Sites of Community Importance (SCI) proposed by the Canary Islands Autonomous Community through the Spanish State. In 2011, as per the Order ARM/2417/2011 of 30 August, the 24 sites of marine community importance in the Macaronesian biogeographic region of the Natura 2000 Network were declared as SACs, and the corresponding management plans were approved, which included the conservation measures and regulation of uses and activities. Subsequently, in 2015 (BOE-A-2015-2329), with the results obtained through the studies carried out within the framework of the LIFE+ INDEMARES project (inventory and designation of the Natura 2000 Network in marine areas of the Spanish State), two new SCIs of the Natura 2000 Network were approved—the Conception Bank (ESZZ15001) and the marine area of the east and south of Lanzarote-Fuerteventura (ESZZ15002) (MITECO, 2019). The species of community interest considered to be declared as SACs are: a marine turtle, the loggerhead sea turtle (*Caretta caretta*), and only one cetacean, the common bottlenose dolphin (*T. truncatus*), following the indications described in Annex II of the Habitats Directive that was previously mentioned.

The field data for the distribution pattern of the common bottlenose dolphin populations were collected before 1996; at that time, it was known that other cetacean species were present in these areas such as the short-finned pilot whale (*Globicephala macrorhynchus*), the Risso's dolphin (*Grampus griseus*), the sperm whale (*Physeter macrocephalus*), the striped dolphin (*Stenella coeruleoalba*), and the spotted dolphin (*Stenella frontalis*). In the official enactment documents of those SACs, no records or distributional patterns were provided for the latter mentioned cetacean species; there was only a note stating that attention and consideration should also be given to their protection in these SACs. It is to be noted that the boundaries, the perimeter of the already approved SACs, have not taken into consideration the spatial distribution of these other cetacean species, with the exception of the common bottlenose dolphin; therefore, the delimited area of these SACs may not be sufficient to protect these animals. La Manna et al. (2020) presented similar results for Mediterranean waters, arguing that the extension of SACs was ineffective for the conservation efforts of these animals and, therefore, proposing to enlarge the borders of SACs for the effective protection of cetaceans in this sea basin.

In recent years, new milestones have contributed toward the protection of cetaceans in the Canary Islands waters. In November 2019, the waters near the Tenerife and La Gomera Islands were declared a "Hope Spot" site by the Mission Blue initiative[1] due to its diversity of cetaceans. More recently, in January 2021, almost the same exact marine area has been nominated as a "Whale Sanctuary" by the global program Whale Heritage Sites[2] as recognition of its outstanding cetacean species richness and the ecosystem services (ES) it may provide for local communities. Furthermore, in 2007 (Carrillo, 2007), considering the wide spatial distribution of cetaceans not only in the Canary Islands but also in the European Macaronesia (Azores, Madeira), the creation of a *Macaronesian Biodiversity, Ecological and Cetacean migration corridor* was proposed; it would greatly contribute to the conservation of these cetaceans, which are so important for the marine ecosystem. This would not be the first time, as there is already a "Mediterranean Cetacean Migration Corridor" (BOE-A-2018-9034).

Any advances are made in research about cetaceans every day, underlining the richness of these marine mammals in the waters of the Canary Islands (Aguilar de Soto et al., 2001; Pérez-Vallazza and Haroun, 2005; Pérez-Vallazza et al., 2008; Fernández et al., 2009; Carrillo and Ritter, 2010; Carrillo et al., 2010; Fais et al., 2016; Puig-Lozano et al., 2020) and in the Macaronesian region (Carrillo, 2007; Alves et al., 2015; Correia et al., 2019). Unfortunately, only a few research articles have been published on the spatial and temporal distribution of these animals, that too only for some specific areas in the waters of the Canary Islands; this study is the first in our knowledge to show the general spatial distribution of cetaceans in the waters of the whole Canary archipelago. This baseline knowledge is fundamental for further assessing the conservation status of the cetaceans with regards to their distribution and to manage the status of cetaceans in Canary Islands waters efficiently.

The aim of this work is to present the spatial distribution of cetaceans in the Canary archipelago, based on an extensive database of marine mammal sightings in recent years, from 2007 to 2018. This study also aims to identify the areas where the greatest number of individual cetaceans can be found, highlighting their relationship with the extant SACs from the Natura 2000 Network.

MATERIALS AND METHODS

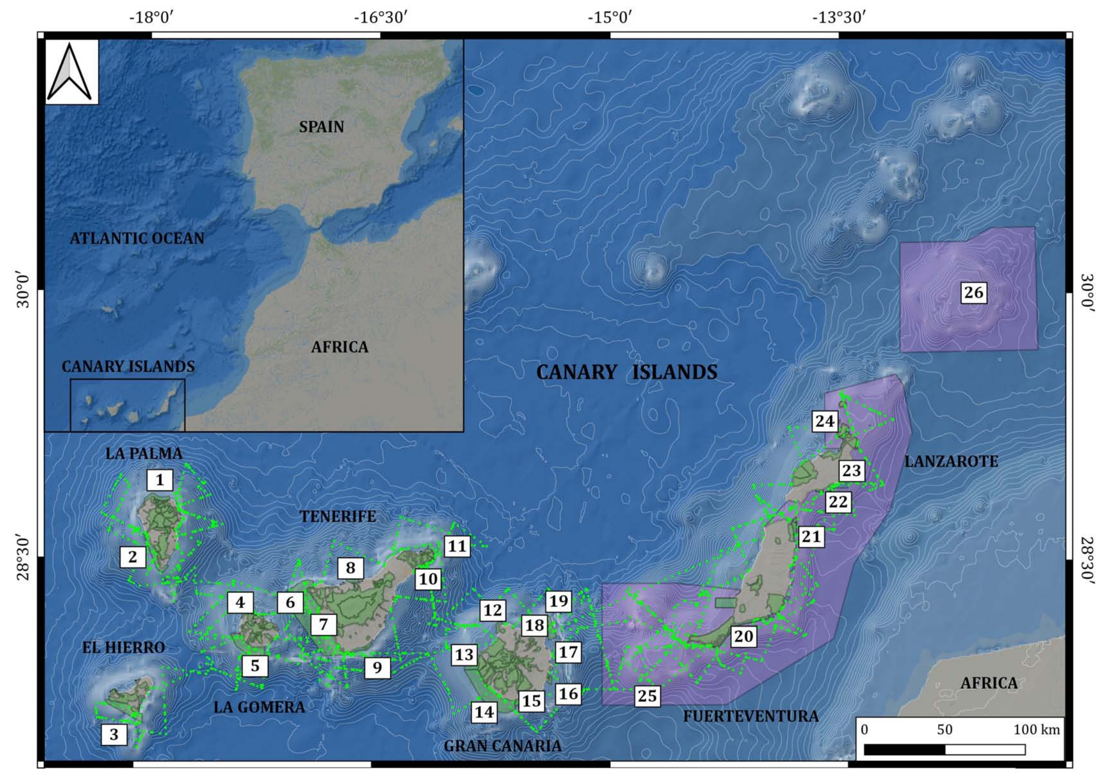
Survey Area and Data Collection

This study focuses on the waters around the Canary Islands, which is a Spanish autonomous region in the Northeast Subtropical Atlantic Ocean (from 29° 24′ 40″ N to 27° 38′ 16″ N and from 13° 19′ 54″ W to 18° 09′ 38″ W). This volcanic archipelago consists of eight islands (**Figure 1**) and is geographically split up into the eastern (La Graciosa, Lanzarote, and Fuerteventura), the central (Gran Canaria and Tenerife), and the western islands (La Gomera, La Palma, and El Hierro).

The cetacean sightings datasets used to conduct this study are the result of enormous human efforts by various data sources, primarily the following ones: (1) Canarias Conservación: 2007–2018 in Fuerteventura, Gran Canaria, and Tenerife islands; (2) MISTIC SEAS II project: September–November 2017 in the Canary archipelago; and (3) Programa POSEIDON: 2013–2015 in Gran Canaria, Tenerife, and La Palma. A total of 1,945 cetacean sightings were observed between 2007 and 2018 (**Supplementary Table 2**), representing 12 years of robust scientific information on marine mammals in the waters of the Canary archipelago. The cetacean sightings were carried out in accordance with the code of conduct of the Government of the Canary Islands (Decree 178/2000; Decree 1727/2007). The data collected for each dataset included time, position, species identity, group size, the presence of calves, the coastal distance, and the spatial trajectory (tracks lines; **Figure 1**), among others parameters. The species were identified to the lowest taxonomic level possible from descriptions in field guides and scientific literature (Carwardine, 1995). Each dataset was obtained with the following singularities in its methodology:

(1) The Canarias Conservación dataset: Most of the data was collected by the marine environmental consulting company "Canarias Conservación"[3] through visual identification, using programmed transects and opportunistic platforms. Cetacean surveys were carried out on three Canary Islands, namely, Fuerteventura, Gran Canaria, and Tenerife, with varying effort, from 2007 to 2018. If the weather conditions allowed—Beaufort scale 3—cruises were conducted along the edge of the island shelf in a 13-m speedboat with a flying bridge located 7 m above sea level, at an average survey speed of 6 knots

[1] https://mission-blue.org/hope-spots/
[2] https://whaleheritagesites.org/
[3] https://www.canariasconservacion.org/



Tracks lines Special Areas of Conservation (SACs) Sites of Community Importance (SCIs)

Nº	Type	Code	Name
1	SAC	ES7020124	Costa de Garafía
2	SAC	ES7020122	Franja marina de Fuencaliente
3	SAC	ES7020057	Mar de las Calmas
4	SAC	ES7020125	Costa de los Órganos
5	SAC	ES7020123	Franja marina Santiago-Valle del Gran Rey
6	SAC	ES7020017	Franja marina de Teno-Rasca
7	SAC	ES7020117	Cueva marina de San Juan
8	SAC	ES7020126	Costa de San Juan de la Rambla
9	SAC	ES7020116	Sebadales del Sur de Tenerife
10	SAC	ES7020120	Sebadal de San Andrés
11	SAC	ES7020128	Sebadales de Antequera
12	SAC	ES7010066	Costa de Sardina del Norte
13	SAC	ES7011005	Sebadales de Güigüí
14	SAC	ES7010017	Franja marina de Mogán
15	SAC	ES7010056	Sebadales de la Playa del Inglés
16	SAC	ES7010053	Playa del Cabrón
17	SAC	ES7010048	Bahía de Gando
18	SAC	ES7010037	Bahía del Confital
19	SAC	ES7010016	Área marina de La Isleta
20	SAC	ES7010035	Playa de Sotavento de Jandía
21	SAC	ES7010022	Sebadales de Corralejo
22	SAC	ES7011002	Cagafrecho
23	SAC	ES7010021	Sebadales de Guasimeta
24	SAC	ES7010020	Sebadales de La Graciosa
25	SCI	ESZZ15002	Espacio marino del Oriente y Sur de Lanzarote y Fuerteventura
26	SCI	ESZZ15001	Banco de la Concepción

FIGURE 1 | Map of the survey area location, Canary Islands Archipelago, including the Special Areas of Conservation (SACs; green color), the Sites of Community Importance (SCIs; pink color) and the surveyed spatial trajectory (- - - tracks lines).

(Carrillo et al., 2010). To establish standard and repeatable protocols for these site conditions, sighting data was collected following the standard method of line transects survey (Buckland et al., 1993; Heimlich-Boran, 1993; Dudzinski, 1999; Schwarz and Seber, 1999), together with the model previously designed by Carrillo et al. (2002) for Tenerife. In each survey, four expert observers were on-board the speedboat. Two of them observed with the naked eye from a platform that was 4.20 m above sea level, while a third person observed from the flight deck with 7 × 50 binoculars and the fourth person worked as a data recorder.

(2) The MISTIC SEAS II project dataset: This data is part of a program named the "Pilot Monitoring Project of the Subprogram Oceanic," carried out during the MISTIC SEAS II project[4] to respond to the requirements of the Marine Strategy Framework Directive (MSFD; 2008/56/EC) in the Macaronesia Northeast Atlantic sub-region. Data was provided by Fundación Biodiversidad, Ministry for Ecological Transition and Demographic Challenge.[5] The data of the sightings in the Canary archipelago were collected on-board the 15-m-long motor vessel "Mariatxi," with a 450-HP Scania engine, a 2.95-m acute observation platform, and a capacity of seven people. The research team comprised four observers and a data

[4]https://misticseas3.com
[5]https://www.fundacion-biodiversidad.es/

recorder. The survey design consisted of going around the entire archipelago, covering approximately 12 nautical miles of the coast of each island and the inter-island channels. The best survey design was the equal-spaced zig-zag (ESZ), adjusting the length and angle of the different blocks to maximize the probability of block coverage. The area was subdivided into 26 different blocks, with 2 replicates in most blocks (see description of the sighting area in MISTIC SEAS II, 2019a,b).

(3) The Programa POSEIDON dataset: The data for Gran Canaria, Tenerife, and La Palma were obtained through a citizen science tool, Programa POSEIDON,[6] which was initiated by the Universidad de Las Palmas de Gran Canaria (ULPGC) to monitor marine biodiversity in the Canary Islands. This database was obtained through the sightings carried out by the whale-watching company "Spirit of the Sea"[7] and Fancy II,[8] both of which have good experience and expertise in cetacean species identification. In each survey, cetaceans were observed from the flight deck with 7×50 binoculars at a height of about 7 m during daylight hours, weather permitting (Beaufort scale 3). Each day, the sighting boats crossed the harbor by making random perpendicular transects to the coastline, depending on the route and the weather situation. Although the search pattern was not systematic, mainly regular zigzag transects were followed. The average speed was 6 knots, followed the methodology used by Carrillo et al. (2010).

[6] https://www.programaposeidon.eu/

[7] https://www.dolphinwhale.es/

[8] https://fancy2.com/

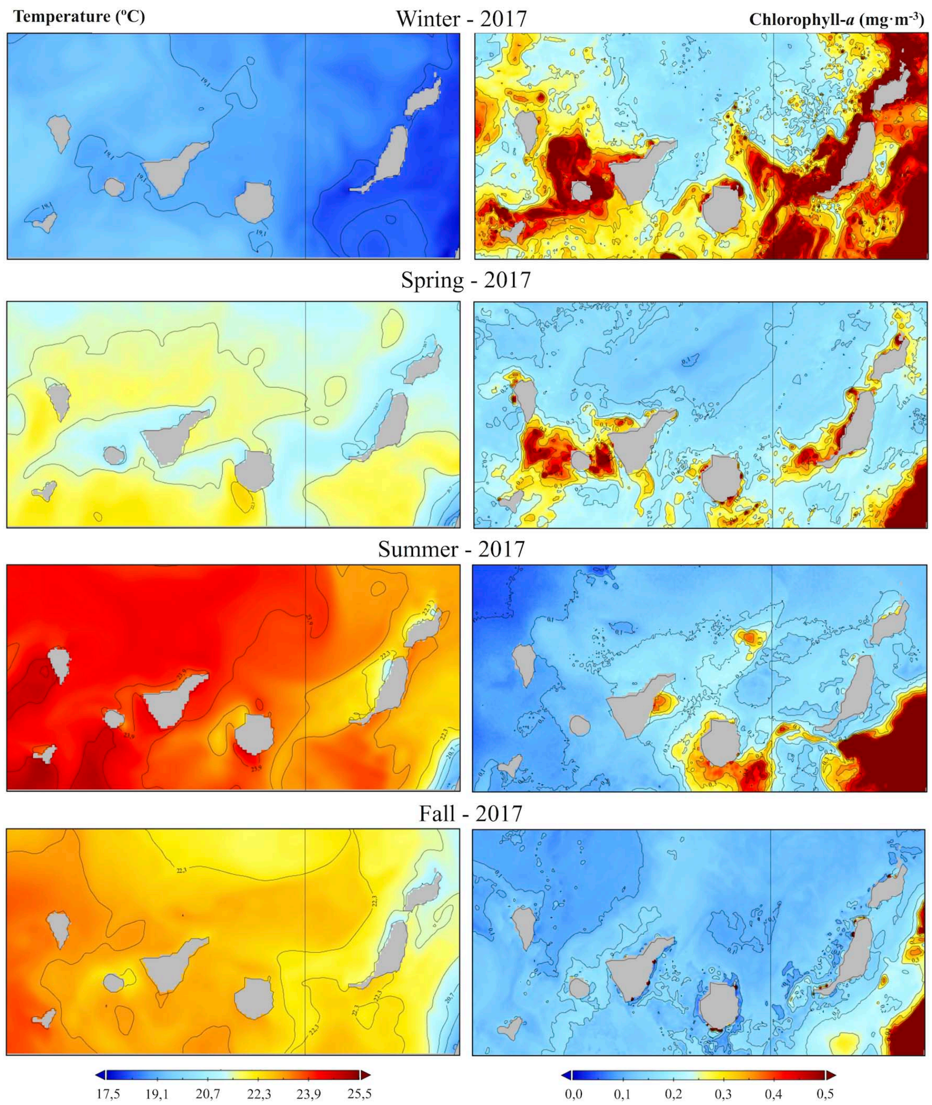


FIGURE 2 | Environmental and biological variables. Left panels: sea surface potential temperature (SST, °C); right panels: chlorophyll-*a* (Chl-*a*, mg·m^{-3}; an example from 2017 (http://marine.copernicus.eu/).

Environmental and Biological Variables

The environmental information was mainly obtained from the Copernicus Marine Environment Monitoring Service (CMEMS[9]). The sea surface potential temperature (SST; °C) and chlorophyll-*a* (Chl-*a*; mg·m^{-3}) were selected from among all the potential parameters, as they were the most relevant variables representing the oceanographic features of the Canarian waters (**Figure 2**). SST (daily L4 product IBI_MULTIYEAR_PHY_005_002) and Chl-*a* data (daily L4 product IBI_MULTIYEAR_BGC_005_003) was compiled by CMEMS. Daily data files, from January 2007 to December 2018 covering the Canary archipelago, were used with a spatial resolution of 0.083° × 0.083°. Data from 2017 was used as an example of SST and Chl-*a* values distribution because of the major effort put into conducting sightings during that year.

Data Processing

Once the information was collected, the sightings data was processed to characterize the different cetacean species identified in the Canary archipelago. To obtain the temporal distribution of each species, distinguishing between Odontoceti and Mysticeti, the frequency was estimated by directly counting the number of sightings. Then, this number was analyzed monthly, using the number of surveyed days. Therefore, the monthly sighting per unit effort (SPUE) was calculated by dividing the number of monthly sightings by the number of total surveyed days per month. The spatial distribution of each species was also presented on a map using the QGIS (3.16 Hannover) software. Sightings of the four most frequent species were represented individually on a map of the island, where SACs and SCIs were also included, to view their presence in waters under figures of conservation and protected areas.

RESULTS

Survey Effort

From a total of 1,945 sightings, 18 species of cetaceans were recorded: 14 odontocetes and 4 mysticetes (**Supplementary Table 2**). The most frequently sighted species for the sub-order Odontoceti were *G. macrorhynchus*, *T. truncatus*, *and S. frontalis*, with sightings of 605, 549, and 316, respectively, forming 31.11, 28.23, and 16.25% of the total, respectively. The most frequently sighted species for the sub-order Mysticeti was *Balaenoptera edeni*, which was sighted on 95 occasions, representing 4.88% of the total number of cetacean sightings in the Canary Islands (**Supplementary Table 2**).

The temporal variation of the total cetacean sightings per unit effort (SPUE; sighting/surveyed days) is presented in **Figure 3**. The dataset allows us to determine that Odontoceti are the dominant cetacean species in Canarian waters throughout the period studied, presenting higher SPUE than Mysticeti. It is also evident that there is a constant increase in cetacean SPUE in the temporal variation for total cetaceans.

[9]http://marine.copernicus.eu/

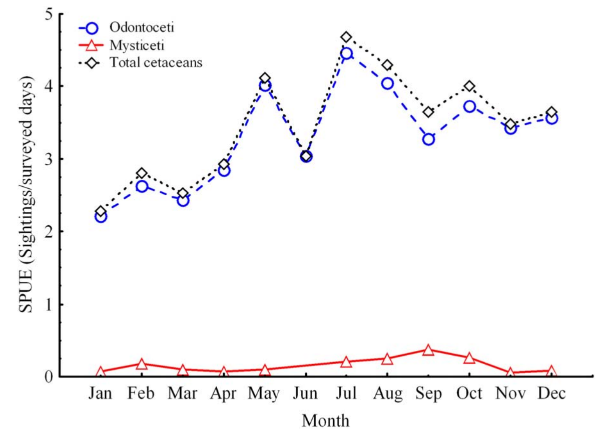


FIGURE 3 | Temporal variation of total cetacean sightings per unit effort (SPUE; sighting/surveyed days) in the Canary Islands during the study period (Jan. 2007–Dec. 2018). Black line: Total cetacean sightings; blue line: Odontoceti; red line: Mysticeti.

The contribution of the different Odontoceti families to the total cetacean SPUE in the Canary Islands is presented in **Figure 4**. The Delphinidae (*G. macrorhynchus, T. truncatus*, and *S. frontalis*) is the most frequently sighted cetacean family, with a minor presence for species belonging to other Odontoceti families (**Figure 4**).

Regarding the SPUE of the sub-order Mysticeti in Canarian waters, only one family, Balaenopteridae, has been observed. In addition, very few species have been observed throughout the different months (**Supplementary Table 3**), *Balaenoptera edeni* being the main species observed with a high SPUE during the summer months (**Figure 5** and **Supplementary Table 3**).

Spatial Distribution and Environmental and Biological Variables

In terms of spatial distribution, 18 species of cetaceans are seen around the Canary Islands (**Figure 6**). These animals can be sighted frequently in areas close to the coast, mainly on the leeward side of the islands, coinciding with warmer temperatures (SST; °C) and high chlorophyll-*a* values (Chl-*a*; mg·m^{-3}; **Figure 2**; an example from 2017). The SST and Chl-*a* time series from 2007 to 2018, for the Canary archipelago, is presented in **Supplementary Figure 1**.

Species Distribution Pattern

The four most frequently sighted cetacean species were three odontocetes (*G. macrorhynchus, T. truncatus,* and *S. frontalis)* and one Mysticeti (*Balaenoptera edeni*). Sighting information for each of these species individually is shown below, noting their spatial distribution with the SACs and SCIs defined in the Natura 2000 Network (**Figure 1**).

Globicephala macrorhynchus

In the case of the short-finned pilot whale (*G. macrorhynchus*), it was observed that this species is mainly distributed outside the Franja Marina of Mogán, in the existing SACs in the La Palma and

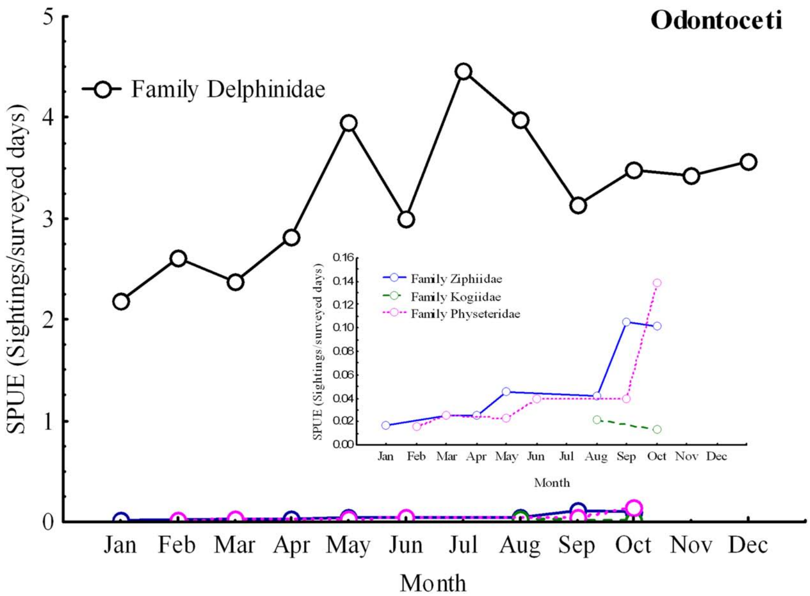


FIGURE 4 | Temporal variation of total Odontoceti sightings per unit effort (SPUE; sighting/surveyed days) in the Canary Islands during the study period (Jan. 2007–Dec. 2018). Blue line: Family Zipphiidae; green line: Family Kogiidae; pink line: Family Physeteridae; black line: Family Delphinidae.

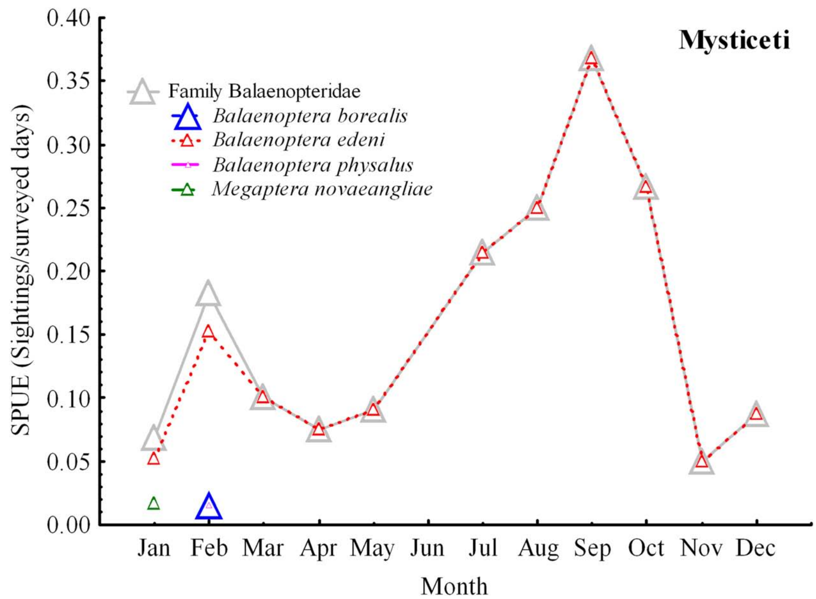


FIGURE 5 | Temporal variation of total cetacean sightings per unit effort (SPUE; sighting/surveyed days) in the Canary Islands during the study period (Jan. 2007–Dec. 2018). Gray triangle: Family Balaenopteridae; blue triangle: *Balaenoptera borealis*; red triangle: *Balaenoptera edeni*; pink triangle: *Balaenoptera physalus*; green triangle: *Megaptera novaeangliae*.

La Gomera islands. In the SAC called Franja Marina de Teno-Rasca (southwest of Tenerife), most individuals were observed inside that protected area (**Figure 7**).

Tursiops truncatus

Tursiops truncatus (common bottlenose dolphin) sightings are concentrated in or around the SACs of Franja marina of Mogán and Sebadales de Güigüi to the southwest of Gran Canaria, in the Franja marina of Teno-Rasca (southwest Tenerife), and the Franja Marina of Fuencaliente to the west of La Palma (**Figure 8**).

Stenella frontalis

The presence of the Atlantic spotted dolphin (*S. frontalis*) was observed practically around the waters of all the Canary Islands, with more sightings to the southwest of the island of Gran Canaria, inside and especially outside the SAC of Franja Marina

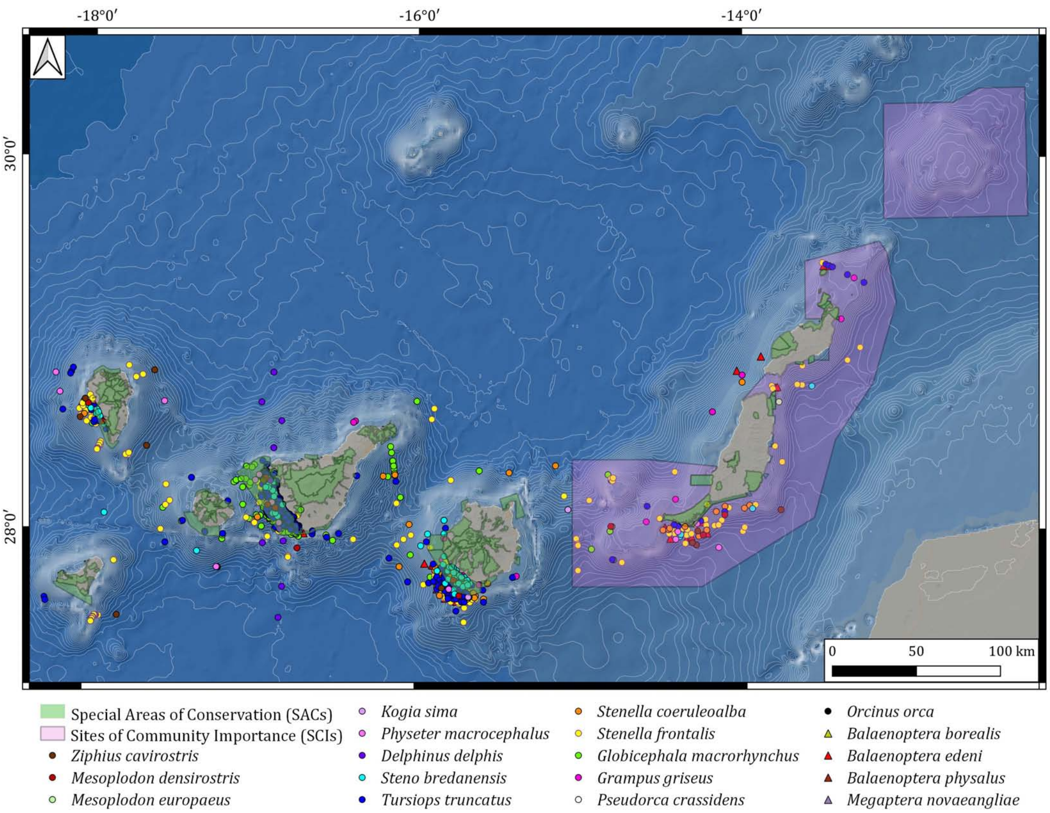


FIGURE 6 | Cetacean spatial distribution around the Canary archipelago (Odontoceti: circles; Mysticeti: triangles), including the Special Areas of Conservation (SACs; green color) and the Sites of Community Importance (SCIs; pink color).

of Mogán; they were also abundantly observed to the west of La Palma in and outside of the SAC of Franja Marina of Fuencaliente (**Figure 9**).

Balaenoptera edeni

Balaenoptera edeni (Bryde's whale) had a spatial distribution throughout the Canary archipelago. It was observed near the coast of different islands, namely, La Graciosa, Lanzarote, Fuerteventura, Tenerife, La Palma, and Gran Canaria, with a major presence in the last island mentioned. It was seen mainly outside the limits of the SAC of Franja Marina of Mogán (**Figure 10**).

DISCUSSION

Throughout the 12 years of marine mammals monitoring in different surveys, 18 species of cetaceans (14 odontocetes and 4 mysticetes) were recorded for the Canary Islands waters. Of these species, ten are labeled as "Least Concern," four as "Data Deficient," two as "Vulnerable," one as "Near Threatened," and one as "Endangered" in the International Union for Conservation of Nature (IUCN) red list (**Supplementary Table 1**; IUCN, 2020). Looking at the four most frequently sighted species in the Canary Islands waters, the odontocetes *G. macrorhynchus*, *T. truncatus,* and *S. frontalis,* and the mysticete *Balaenoptera edeni* are labeled as "Least Concern (LC), unlikely to become extinct in the near future" in the IUCN; however, the species *T. truncatus* and *G. macrorhynchus,* at the national level (Spain) and at the regional level (Canary Islands), are labeled as "Vulnerable." Furthermore, *S. frontalis* and *Balaenoptera edeni* fall under the "special protection regime" (National Government: Royal Decree 139/2011, BOE N° 46, 23/02/2011; Regional Government: Decree 151/2001; BOC N° 097, 01/08/2001). The underlying scientific implication of these figures is the importance and relevance of these animals as key elements of ocean health and their role as major top-down regulators of marine ecosystem oceanic trophic chains (Reynolds et al., 2009; Giralt Paradell et al., 2019). Therefore, there is a clear need for further conservation efforts directed at the populations of these marine mammals in the Canary Islands. Additional studies related to better management and knowledge (e.g., distribution, abundance, feeding, threats, environmental variables) of these cetaceans in the Northeastern Atlantic Ocean, the Canary Islands, and nearby archipelagos such as Cape Verde, Madeira, and Azores are the need of the hour.

Spatial Distribution and Environmental and Biological Variables

The results showed that all species of cetaceans concentrated in areas close to the coast had clear preferences for the leeward areas of the islands, where temperatures and concentrations

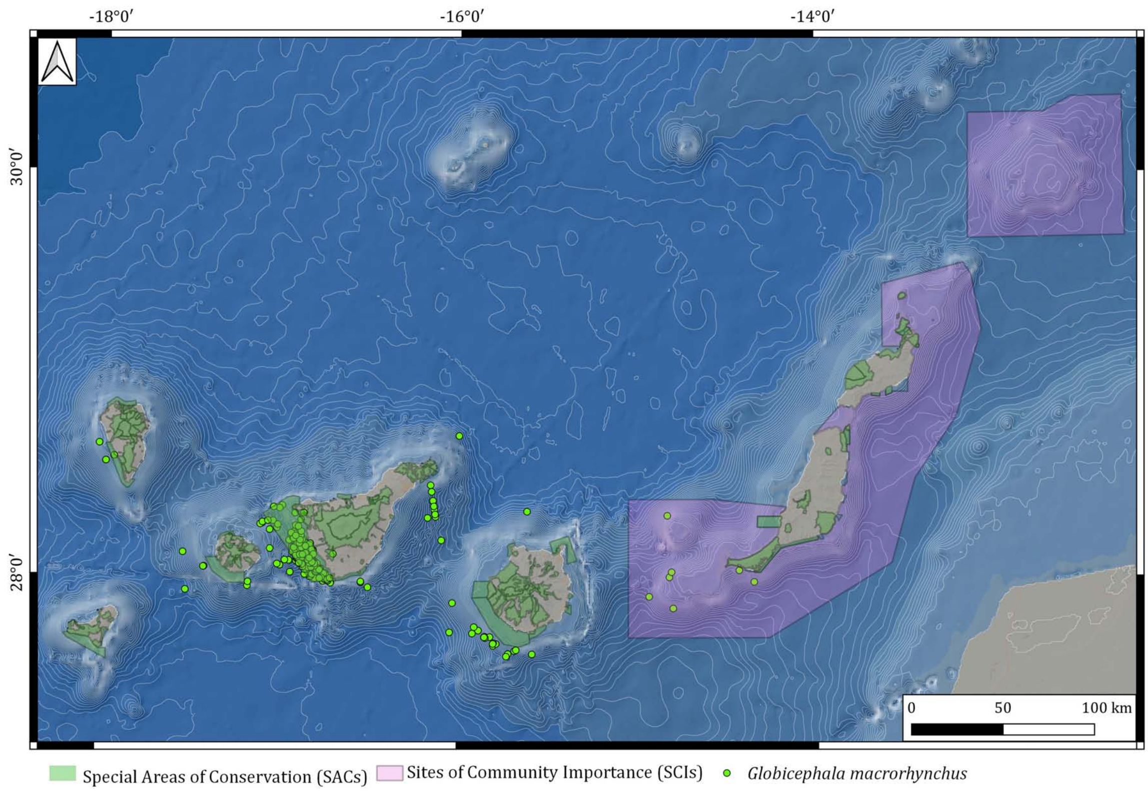


FIGURE 7 | *Globicephala macrorhynchus* spatial distribution around the Canary archipelago (green circles), including the Special Areas of Conservation (SACs; green color) and the Sites of Community Importance (SCIs; pink color).

of chlorophyll-*a* showed high values according to the environmental parameters obtained from the CMEMS. Other authors have observed that a large concentration of cetaceans seems to be connected to the environmental and biological characteristics (Perrin et al., 1994; Pérez-Vallazza et al., 2008). For instance, in the case of *G. macrorhynchus,* there is already a correlation between sea surface temperature and its distribution in the marine waters of Tenerife Island (Montero and Arechavaleta, 1996). Additionally, the high productivity in the Canary Islands waters, as described by various authors (Arístegui et al., 1997; Barton et al., 1998; Arístegui and Montero, 2005; Alonso-González et al., 2013), not only favors cetaceans but also their food/prey, such as cephalopods (Escánez et al., 2018), with large catches being observed in these areas, where the greatest influx of cetaceans (ICES, 2019) and small–medium pelagic fishes are found (Arístegui et al., 2009). Therefore, these areas, sheltered from the trade winds and rich in prey resources, seem to provide a suitable habitat for the cetaceans.

Moreover, of the 14 odontocetes observed during the 12 years of studies, three of them dominated the sightings (75.58% of the total sightings), namely, *G. macrorhynchus, T. truncatus,* and *S. frontalis*. The first two species have been well-studied in the Canary Islands (Morales-Herrera, 2015; Servidio et al., 2019), especially in the waters surrounding Tenerife (Carrillo and Peña, 2002; Carrillo et al., 2006, 2010; Pérez-Vallazza et al., 2008; Tobeña et al., 2014), showing similar spatial distribution in all years of the study. This suggests that environmental fluctuations do not affect the presence of those resident cetaceans. In the case of *S. frontalis,* unfortunately, this species has been less studied individually (Perrin et al., 1994). However, as shown by both our results and previous studies (Pérez-Vallazza et al., 2008; Carrillo et al., 2010; Morales-Herrera, 2015), it is an abundant species in the waters of the Canary Islands throughout the year.

The most frequently sighted mysticete species (and the sixth most frequently sighted cetacean species) was *Balaenoptera edeni.* Despite being a migratory species, its presence in the Canary Islands is quite marked (Aguilar de Soto, 2006; Carrillo et al., 2010; Morales-Herrera, 2015; Lado-Pedreda, 2018); it is found not only in these waters but also in nearby Macaronesian archipelagos, such as in Madeira (Alves et al., 2010) and the Azores (Steiner et al., 2008). Although robust information on the spatial and temporal distribution regarding the family Balaenopteridae and its habitats and/or feeding preferences is scarce, it is observed that all Balaenopteridae species found in the North Atlantic pass through the Canary Islands during certain months of the year. These animals have been monitored through observations coming from opportunity platforms (ferries navigating the inter-island waters), under the umbrella of the Red CetAvist project, where a wide spatial distribution of cetaceans of the family Balaenopteridae was observed in the waters of the Canary Islands (Lado-Pedreda, 2018). As in the case with other odontocetes species, it is assumed that their spatial

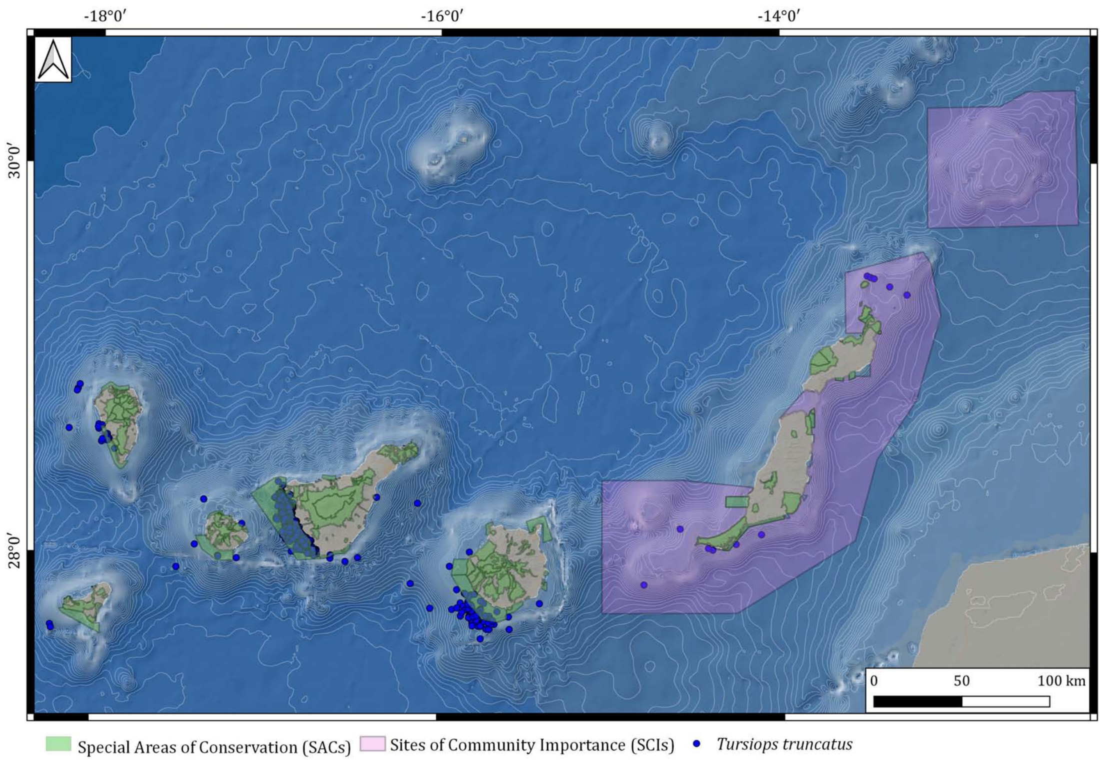


FIGURE 8 | *Tursiops truncatus* spatial distribution around the Canary archipelago (blue circles), including the Special Areas of Conservation (SACs; green color) and the Sites of Community Importance (SCIs; pink color).

distribution near the coast in the Canary Islands is also due to the high productivity in these waters (Arístegui et al., 1997) and may be related to their food preferences, in particular their affinity for schools of small pelagic fish, squid, and plankton (Tershy et al., 1993; Baines and Reichelt, 2014). In addition, the distance to the coast and the slope of the ocean floor (Tardin et al., 2017) may favor their feeding behavior; thus, it is common to find them near the Canary Islands' western coasts.

As a consequence of rapid and mainly unplanned growth in the 80s and 90s, the Canary Islands are under constant threat of habitat degradation due to coastal land-use change, pollution by plastics and waste, maritime traffic, and, more recently, the negative effects of climate change, including rising sea levels and increasing seawater temperatures (IPCC, 2014; EEA, 2016). Climate change is having various negatives effects on the good marine environmental status worldwide. In recent years, a rising trend of an increase in temperature of 0.28°C per decade, from 1982 to 2013, was observed in the waters around the Canary Islands (Vélez-Belchí et al., 2015).

These increases in seawater temperature result in changes in the distribution of the cetacean preys in different trophic levels of the marine ecosystem (e.g., plankton, fish, cephalopods) (Evans et al., 2008). This directly affects the habits, diets, and behavior of whales and dolphins. Nowadays, scarce information is available on the effect of climate change on cetaceans in the Canary Islands and adjacent archipelagos (Madeira and Azores). This is due to the gaps in knowledge of geographical distribution, migration patterns, and diets in the Macaronesian biogeographic sub-region. However, the possible effects of climate change on cetaceans have already been described in other areas, with changes observed in the distribution patterns of these animals, mainly due to variations in the abundance or distribution of prey (Learmonth et al., 2006; Simmonds, 2016) and in the duration and timing of migration, as well as reproductive success rates (Leaper et al., 2006; Ramp et al., 2015). There is no doubt that studies of marine mammals are difficult, and trends, in their abundance and distribution, are inconclusive concerning the causal role of climate change (ICES, 2008). Therefore, it is necessary that decision-makers monitor and evaluate marine ecosystems, and include conservation plans adjusted to the current information on the ecosystems.

Implications for the Special Areas of Conservation of the Natura 2000 Network

Since the conservation of cetaceans is of importance for regulatory bodies, the European Union has recognized the need for distribution maps of basin scales on a monthly basis (Habitats Directive 92/43/EEC, Marine Strategy Framework Directive: 2008/56/EC). Of the 18 species sighted during this study (2007–2018), we wanted to pay more attention to the three species already mentioned (*G. macrorhynchus*, *T. truncatus*, and *S. frontalis*), which reside and/or are present in the waters of the

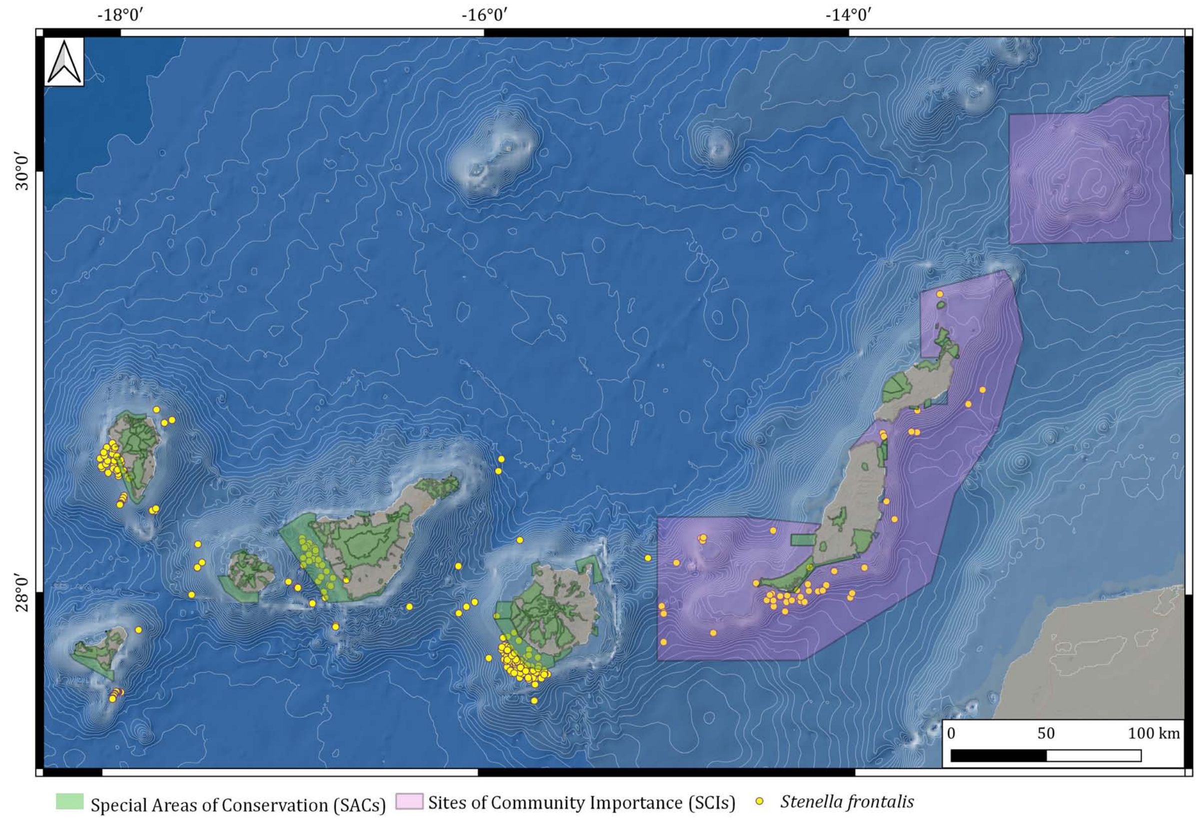


FIGURE 9 | *Stenella frontalis* spatial distribution around the Canary archipelago (yellow circles), including the Special Areas of Conservation (SACs; green color) and the Sites of Community Importance (SCIs; pink color).

Canary Islands all year round (Morales-Herrera, 2015; Servidio et al., 2019). Regarding their spatial distribution, it is essential to highlight that most sightings for these species are found outside the external limits of the SACs included in the Natura 2000 Network. This spatial distribution has already been observed previously for Gran Canaria (SAC Franja Marina de Mogán) (Pérez-Vallazza and Haroun, 2005), Tenerife (SAC Franja Marina de Rasca-Teno) (Aguilar de Soto et al., 2001; Carrillo et al., 2010), La Palma (SAC Franja Marina de Fuencaliente) (Pérez-Vallazza et al., 2008), and El Hierro (Arranz et al., 2008; Amengual et al., 2015). In the marine waters of El Hierro, in addition to hosting the three species already mentioned, it should be noted that it is one of the few places in the world where the resident populations of Ziphiidae (deep-diving cetacean species) (Arranz et al., 2008) are known to exist. During the monitoring conducted between 2003 and 2007 on this island, more than 1,600 individuals of Ziphiidae were sighted in its waters (**Figure 11**, this study; Arranz et al., 2008; Amengual et al., 2015). Their local presence reinforces the role of the Canary Islands as a conservation hotspot for cetaceans, thus emphasizing the need to effectively protect populations of different species from any anthropogenic pressures.

After 12 years of scientific monitoring and sightings of cetacean populations around the Canary Islands, the present study shows that the current SACs are not large enough to effectively protect these endangered marine mammals; these keystone cetacean populations require large marine protected areas that include parts of their oceanic (offshore) habitat (Game et al., 2009). Currently, there are 24 SACs in the Canary Islands that have been designated for the conservation of two species of community interest, namely, the marine turtle *C. caretta* and the cetacean *T. truncatus*. However, there are other cetacean species in the marine waters surrounding the Canary Islands: the short-finned pilot whale (*G. macrorhynchus*), Risso's dolphin (*G. griseus*), the sperm whale (*P. macrocephalus*), the striped dolphin (*S. coeruleoalba*), and the spotted dolphin (*S. frontalis*). These species also deserve to be included in the list for conservation and preservation in the already allocated SACs and SCIs, with special attention to their distribution, areas occupied, and the extents of these marine areas for future updates of management actions. This issue of the Canary Islands is not an isolated event; the same matter is observed in other areas such as the Mediterranean. A recent study by La Manna et al. (2020), which examined the relationships between oceanographic variables and the spatial distribution of the common bottlenose dolphin, showed the importance of updating and implementing current management and conservation instruments to extend the limits of SCIs while working on the reduction of the anthropogenic pressures that impact these marine mammals. The published results and

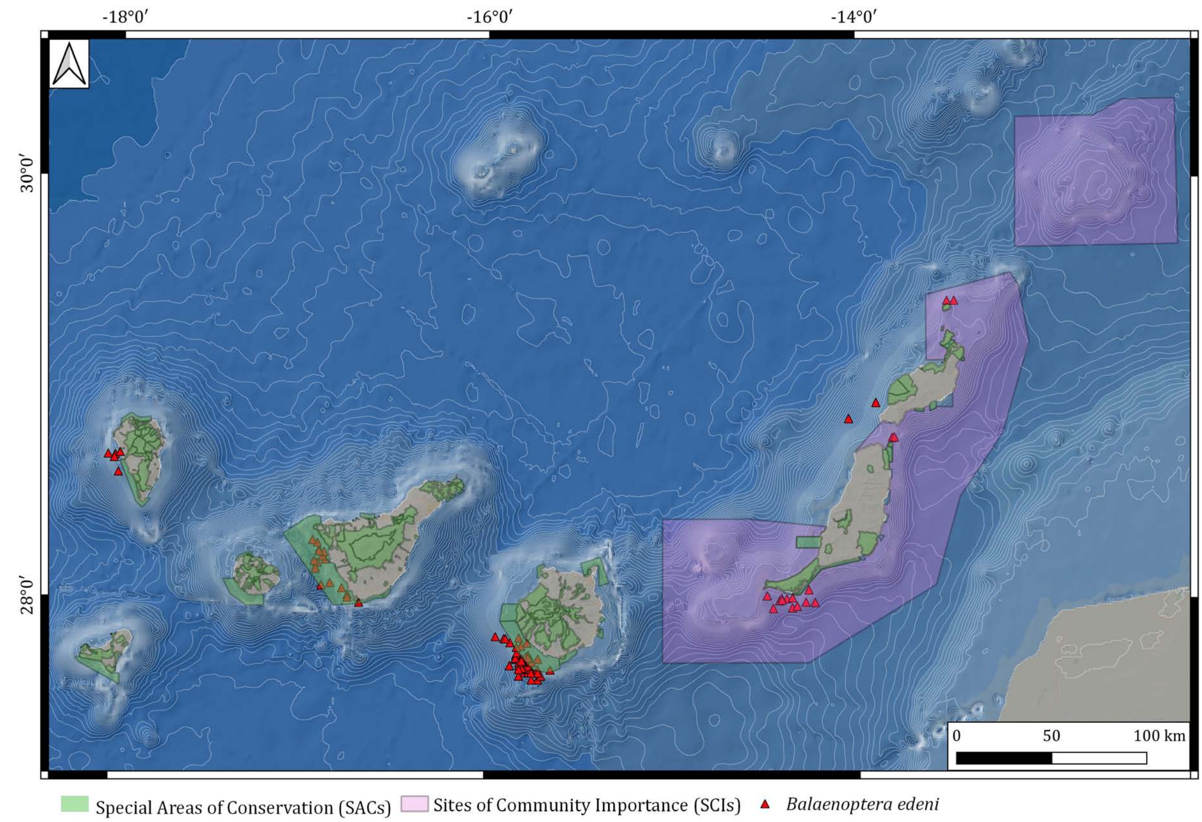


FIGURE 10 | *Balaenoptera edeni* spatial distribution around the Canary archipelago (red triangles), including the Special Areas of Conservation (SACs; green color) and the Sites of Community Importance (SCIs; pink color).

those compiled in the present study show that the borders of several SACs and SCIs must be extended to effectively preserve the ecologically significant features sustaining the population of cetaceans found in the Canarian waters. These results should be considered in future conservation measures for the amendment and updating of management instruments, as well as the revisions of the limits of the Natura 2000 SACs. Thus, jurisdictional and administrative borders should be altered to better consider and preserve key ecological processes and functions that are ultimately responsible for the conservation of cetacean populations. To this end, the spatial distribution maps generated in the present study facilitate the consideration of the conservation of cetacean populations in the maritime spatial planning process.

New milestones in cetacean conservation also need to be considered for future measures. During the last (25th) United Nations Climate Change Conference (December 2019), Mission Blue had declared the Canary Islands as a "Hope Spot" for marine ecosystems, mainly for the cetacean populations, in the area between the islands of Tenerife and La Gomera. The objective was to support further protection of the area, in the spirit of the IUCN target of having 30% of the ocean officially protected by 2030, and to encourage the Spanish and Canarian governments to declare a large marine protected area in the waters of the islands, where humans and nature can thrive together. These Canarian waters are precisely where the greatest influx of cetaceans has been found during the study period of this work. In addition, between these two islands, there are several protected areas, SACs, that belong to the Natura 2000 Network. Considering all these features along with the ES that are generated by those animals (e.g., whale watching, scientific research, bequest, and spiritual value), currently, these waters have also been nominated as a "Whale Sanctuary" by the global Whale Heritage Sites program because of the great diversity and abundance of cetaceans found in these waters surrounding the islands of Tenerife and La Gomera. Cook et al. (2020) further describe the major ES linked to these protected cetaceans as inducing reasoned, rational compromises in decision-making, considering the various ES threats and trade-offs.

There is no doubt regarding the role played by cetaceans in the good environmental status (GES) of the ecosystems of the Canary Islands' waters. They are considered valid indicators of the well-being status of marine ecosystems. Therefore, creating a network of marine protected areas, ecologically coherent with the biogeographical sub-region of Macaronesia, would favor the conservation of vulnerable ecological habitats and species of socio-economic interest. Thus, a wide marine protected area could be proposed for the European Macaronesia (Madeira, Azores, and Canary Islands), such as the creation of a *Macaronesian Biodiversity, Ecological and Cetacean migration*

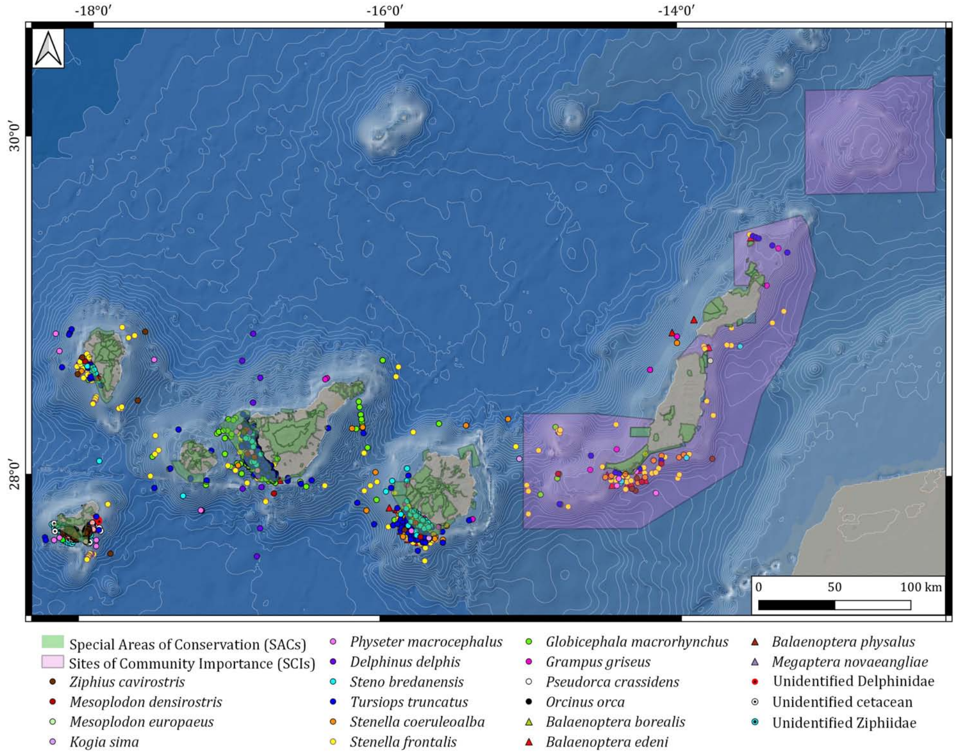


FIGURE 11 | Cetacean spatial distribution around the Canary archipelago (Odontoceti: circles; Mysticeti: triangles), including the Special Areas of Conservation (SACs; green color) and the Sites of Community Importance (SCIs; pink color). Including sightings from El Hierro island.

corridor (Carrillo, 2007). The focus of this proposal for marine protected species is on the design of an area for the conservation of vulnerable migratory species. A similar corridor has already been defined in the Mediterranean, the "Mediterranean Cetacean Migration Corridor," as an important area for cetacean species with high ecological value (BOE-A-2018-9034). This marine protected area is included in the Natura 2000 – LIC-ESZZ16001 network and in the Important Marine Mammal Areas initiative (IMMAs[10]); managing and monitoring this area is a major activity of the Marine Mammal Protected Areas Task Force. The IMMAs are defined as discrete portions of habitat, important to marine mammal species, that have the potential to be delineated and managed for conservation and are identified to prioritize their consideration for conservation measures by governments, intergovernmental organizations, conservation groups, and the general public. Another example of an important conservation area is the "Alborán corridor IMMA"; this area represents a migratory corridor for vulnerable fin whales in the Northern Alborán Sea and Strait of Gibraltar (IUCN-MMPATF, 2017). In the European Macaronesia region, in an area located between Madeira and Desertas islands, there is already an established IMMA area for another marine mammal, the monk seal (*Monachus monachus*), which shares marine space with the common bottlenose dolphin (*T. truncatus)* (IUCN-MMPATF, 2018). Therefore, it is essential that monitoring and novel initiatives are undertaken to protect our waters and marine ecosystems, by identifying and enacting a "Macaronesian Biodiversity, Ecological and Cetacean Migration Corridor," to enhance the protection of cetacean species along with their large distributional range in the Northeast Atlantic Ocean.

CONCLUSION

This study demonstrates that long-term monitoring can provide key information to identify areas of high marine mammal abundance as well as key data about their resident or migratory status in the Canarian waters. The areas with high abundance of cetaceans in many cases coincide with the designated Natura 2000 areas, SACs that were designated only to protect bottlenose dolphin populations and not for the conservation of the other cetacean species that are also present in the Canary Islands. In addition, the extent of the areas where cetacean species are distributed and inhabit extends beyond the outer limits of the designated SACs. These results suggest that improvements must be made to the current conservation measures, enabling the enlargement of the extant SACs limits to promote the efficient management of cetaceans, ecosystems, and GES in the waters of the Canary Islands. The recent declarations of two international conservation figures such as Tenerife-La Gomera "Hope Spot" by the Mission Blue Initiative or the Canary Islands "Whale

[10]https://www.marinemammalhabitat.org/imma-eatlas/

Sanctuary" declaration by the program Whale Heritage Sites also pointed out the presence of cetacean species outside the extant SACs in Tenerife and La Gomera and further underpin the need of larger marine conservation areas in the Canarian archipelago. Moreover, the declaration of the Tenerife–La Gomera "Hope Spot" was considered as a starting point to inspire the Spanish and Canary governments for the enactment of a large marine protected area in the Spanish waters. In this sense, it seems essential to provide more extensive monitoring of oceanic waters around the Canary Islands and nearby archipelagos as well as promote novel initiatives to protect migratory routes and marine ecosystems, by identifying a "Macaronesian Biodiversity and Ecological Migration Corridor for Cetaceans," a conservation figure that has been already proposed in the scientific literature but has not yet been accomplished through the properly governmental measures.

Our findings are also relevant as a contribution to population abundance estimates of cetaceans for the Canarian archipelago and its relationships with other nearby geographic areas in the Northeast Atlantic Ocean, such is the case of the archipelagos of Azores, Madeira, and Cape Verde, where some frequently sighted cetaceans may migrate at different times of the year. The connectivity or the genetic imprint of their populations is an interesting area of research for future funding and scientific efforts.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

Ethical review and approval was not required for the animal study because this work presents information on the distribution of marine mammals collected through controlled observations in accordance with the code of conduct of the Government of the Canary Islands (Decree 178/2000; Decree 1727/2007).

AUTHOR CONTRIBUTIONS

IH and RH conceived the ideas. MC collected most of the data. MCdE did most of the graphics and maps. IH, MC, and RH processed the data. IH led the writing of the manuscript. RH did the fine-tuning of the final document. All authors contributed critically to the drafts and gave final approval for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.669790/full#supplementary-material

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Collaborative Automation and IoT Technologies for Coastal Ocean Observing Systems

Patrizio Mariani[1]*, Ralf Bachmayer[2], Sokol Kosta[3], Ermanno Pietrosemoli[4], Murat V. Ardelan[5], Douglas P. Connelly[6], Eric Delory[7], Jay S. Pearlman[8], George Petihakis[9], Fletcher Thompson[1] and Alessandro Crise[10]

[1] Institute for Aquatic Resources, Technical University of Denmark, (DTU Aqua), Lyngby, Denmark, [2] MARUM, University of Bremen, Bremen, Germany, [3] Department of Electronic Systems, Aalborg University, Aalborg, Denmark, [4] Abdus Salam International Center for Theoretical Physics (ICTP), Trieste, Italy, [5] Department of Chemistry, Norwegian University of Science and Technology (NTNU), Trondheim, Norway, [6] National Oceanography Center, Southampton, United Kingdom, [7] Oceanic Platform of the Canary Islands (PLOCAN), Telde, Spain, [8] Institute of Electrical and Electronics Engineers (IEEE), Paris, France, [9] European Global Ocean Observing System (EURO GOOS), Brussels, Belgium, [10] National Institute of Oceanography and Experimental Geophysics (OGS), Trieste, Italy

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Tomaso Fortibuoni, Istituto Superiore per la Protezione e la Ricerca Ambientale (ISPRA), Italy

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Annalisa Griffa, National Research Council (CNR), Italy

***Correspondence:**
Patrizio Mariani
pmar@dtu.dk

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Coastal observing systems are typically nationally funded and built around national priorities. As a result, there are presently significant differences between countries in terms of sustainability, observing capacity and technologies, as well as methods and research priorities. Ocean observing systems in coastal areas must now move toward an integrated, multidisciplinary and multiscale system of systems, where heterogeneity should be exploited to deliver fit-for-purpose products that answer the diversity and complexity of the requirements from stakeholders and end-users. Essential elements of such distributed observation systems are the use of machine-to-machine communication, data fusion and processing applying recent technological developments for the Internet of Things (IoT) toward a common cyberinfrastructure. This perspective paper illustrates some of the challenges for sustained coastal observations and provides details on how to address present gaps. We discuss the role of collaborative robotics between unmanned platforms in coastal areas and the methods to benefit from IoT technologies. Given present trends in cost-effective solutions in ocean sensors and electronics, and methods for marine automation and communication, we consider that a distributed observation system can effectively provide timely information in coastal regions around the world, including those areas that are today poorly observed (e.g., developing countries). Adaptation in space and time of the sensing nodes, and the flexibility in handling different sensing platforms can provide to the system the ability to quickly respond to the rapid changes in oceanic and climatic processes, as well as to promptly respond to evolving stakeholder and end-user requirements.

Keywords: marine automation, oceanography, robotics, internet of things, communication systems

INTRODUCTION

Coastal areas are the most dynamic and productive parts of the oceans, which makes them a significant source of resources and services for mankind. Coastal waters are located immediately in contact with human populations and exposed to anthropogenic disturbances, placing these resources and services under threat (e.g., Lynch et al., 2014). These concerns explain why, in several coastal regions, a rapidly increasing number of observing systems have been implemented in the last decade (Moltmann et al., 2019). Expansion of coherent and sustained coastal observations has been fragmented and driven by national and regional policies and is often undertaken through short-term research projects (Farcy et al., 2019). This results in significant differences between countries both in terms of sustainability and observing technologies, methods and research priorities.

Unlike the open ocean, where challenges are rather well-defined and stakeholders are fewer and well-identified, coastal processes are complex, acting on several spatial and temporal scales, with numerous and diversified users and stakeholders, often with conflicting interests. To adapt to such complexity coastal ocean observing system must be an integrated, multidisciplinary and multiscale system of systems (GOOS, 2012). But the diversification in data acquisition, handling and storage can inevitably create problems in data management and delivery, hampering interoperability and limiting opportunities to advance our knowledge on coastal processes and resource management.

Looking at the future, system's diversification will likely increase as new sensors and platforms become available, activities in new sectors of the ocean economy are developed, and an increasing number of users will provide heterogeneous societal demands for specific observations. In this perspective we present some of the challenges for sustained coastal observations and illustrate methods to address them. We suggest an approach to exploit current and future system heterogeneity while serving the needs of a sustainable and robust coastal observation. The specific aim of such a system should be harmonized and autonomous acquisition, use of best-practices for data handling and storage and to provide information to enable ecosystem based management of the ocean while responding to data requirements of different blue-economy sectors.

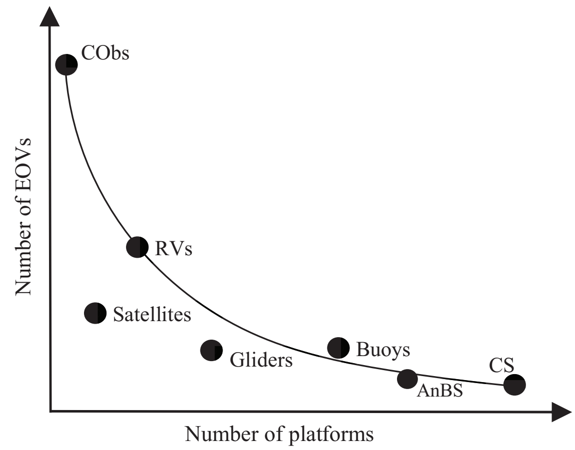

FIGURE 1 | Schematic of the number of observed essential ocean variables (EOVs) from a specific platform vs. the available number of these platforms. A distributed system for observations will use all its nodes to deliver fit- for-purpose information, products and services. Cobs, cabled observatories; RVs, research vessels; AnBS, animal borne sensors; CS, citizen science. Note: additional dimensions could be included in the plot, e.g., observation range, cost of the platforms, etc.

ELEMENTS OF A NETWORK OF DISTRIBUTED OBSERVING PLATFORMS FOR COASTAL AREAS

Coastal systems demands the use of diverse observing platforms for the collection of relevant oceanographic variables, i.e., Essential Ocean Variables (EOVs, Lindstrom et al., 2012; Bax et al., 2018). Present solutions are based on different approaches which include a limited number of cabled observatories, collecting high frequency data on a larger number of EOVs, supplemented by a variety of other platforms typically observing a lower number of variables (**Figure 1**). Similarly, the expected expansion of citizen science programs (Kelly et al., 2020) into the ocean domain can enable in the future many more devices observing selected variables. This intrinsic heterogeneity in observation systems is a challenge for centralized management. But, when managed in a coordinated way, it could offer more flexibility and the opportunity to optimally respond to future ocean data needs. Indeed, a network of heterogeneous systems can better adapt and promptly react to diverse and complex coastal processes and the diverse user demands and priorities, hence delivering fit-for-purpose data and products.

In this perspective we present methods to move toward a distributed system for coastal ocean observations composed of a network of fixed and mobile heterogeneous nodes, which can coordinate data acquisition tasks and data management. Diversity and heterogeneity are two key characteristics which can increase system resilience, as evidenced during the COVID-19 pandemic when the overall observation system performance was rather unaffected although some platforms were impacted. Nodes of the network are stand-alone platforms (e.g., buoys, moorings), cabled observatories, research vessels, FerryBoxes, autonomous underwater and surface vehicles, gliders, bio-logging, satellites and different low-cost, low-power sensors systems, e.g., from citizen science initiatives. Communications in the network are enabled exploiting Internet of Things (IoT) technologies. Data management is performed over a web-infrastructure with near-real-time (nRT) communications, long term storage, secure data retrieval and services for data processing that are offered online by exploiting cloud technologies and services.

An important feature is machine-to-machine (m2m) communication among the nodes to enable adaptive sampling. When specific signals are recorded by one node (e.g., harmful algal bloom, oil spills, ocean heatwaves, etc.) other nodes can refine the event's data acquisition strategy (e.g., extend geographical coverage, provide higher sampling rate, etc.) using collaborative robotic solutions. The distributed structure will also ensure a better tolerance to individual node failures and enable the flexibility to satisfy specific tasks (e.g., aquaculture monitoring, support to tourism, etc.). The complexity of the network structure and functioning is generally hidden to the final users (Carpenter and Cannady, 2004; Crise et al., 2018; Montella et al., 2018) and this process of abstraction is relevant to deliver a portfolio of products. Special interest groups may have specific needs that can be met by these *ad hoc* products and the requirements can change in time and space (e.g., the constant evolution of marine directives) along with the technological advances. Services on the cyberinfrastructure are customizable by the end-users according to the requirements of the specific use (i.e., precision, accuracy, sampling frequency, etc.). Thus, both data and products have to follow FAIR principles (Findable, Accessible, Interoperable, Reusable; Wilkinson et al., 2016), although regional economic, legal and security aspects could in same cases limit data access.

In this context IoT solutions could be exploited for the interconnection at different scales: locally (e.g., sensors), regionally (e.g., platforms) and globally (e.g., observatories). In particular, they include the coordinated (common objective—yet separate execution) and cooperative (common objective—frequent interaction) operations of the components including domain specific high computing facilities for numerical ocean modeling tasks. In the future deployment bottlenecks in the area of interoperability (i.e., common calibration and communication standards) are to be expected.

An extended review of the challenges for global ocean observing system is presented elsewhere (e.g., Moltmann et al., 2019). The use of best practices and recent technology for sensors and data interoperability (Buck et al., 2019), unmanned marine platforms (Testor et al., 2019), cabled observatories (Howe et al., 2019), and marine observatories (Crise et al., 2018) have all been identified to contribute toward implementation of sustainable ocean observations. Here we focus on the role of m2m coordination and IoT techniques as essential elements for the operationalization of a sustained distributed ocean sensing system in coastal regions.

Automation and Collaborative Robotics

The nodes of the distributed system are sensing platforms that have standardized metadata information accessible *via* a common cyberinfrastructure (e.g., a dashboard), including geographical as well as technical specifications on sensors and platforms. Sensor-web architectures (see del Río et al., 2017) are used to achieve process automation, sensor interoperation, and service synergy. Autonomy in the system is expanded relying on a range of unmanned platforms for surface- and underwater-operations (Domingo, 2012; Whitt et al., 2020). Cooperative and collaborative robotics approaches have been developed for autonomous marine vehicles, using different m2m communication and decision-making paradigms (Thompson and Guihen, 2019). Those approaches are not mutually exclusive although generally optimized for different data gathering missions. Specifically:

- ***Cooperative solutions*** work on a single or a small set of similar tasks to accelerate or optimize aspects of the mission (e.g., minimizing completion time; maximizing the coverage area). In these cases, decision-making and m2m communication focus on enforcing a control system that governs each participating platform. Ocean survey missions have particularly benefited from cooperative control methods resulting in solutions that are effective for monitoring over extended periods (Leonard, 2016; Ocean Infinity, 2020[1]; Simetti et al., 2020).
- ***Collaborative solutions*** focus on complex missions that have a "deep" sequence of dependent and interdependent tasks. The m2m communication is shaped to achieve machine-consensus on task allocation and sequencing between platforms based on their metadata profiles and constraints. Collaborative robotics and m2m communications have enabled adaptive sampling (Branch et al., 2019) and extended operations (Lima et al., 2019) in coastal areas, provided an abstract mission planning paradigm.

Different multi-marine platform mission planning tools are available to specify the above tasks that can be executed by individual marine platforms (e.g., Neptus, Pereira et al., 2006; LSTS Toolchain, Pinto et al., 2013; MOOS-IvP, Benjamin et al., 2010; JANUS, Petroccia et al., 2017).

Improved onboard machine intelligence (e.g., nRT data processing, mission planning and optimization, fault response and risk management) is possible with present development in miniaturized electronics and power-efficient algorithms providing direct feedback to the control system of the sensing platform (Zhou et al., 2019). The developments of technology for underwater communication (Song et al., 2019), autonomous-docking, -calibration and -power supply (Yazdani et al., 2020) as well as bio-inspired algorithms for collective behavior and optimal search (Tholen and Nolle, 2017), can open interesting perspectives for the operations of a large fleet of autonomous platforms in coastal areas with little or no human supervision (Schmidt et al., 2016).

Nonetheless, significant technical, operational, economical and legislative challenges must be solved before conducting unmanned coastal observations as a sustainable and long-term program. The technical issues are typically related to limitations in the available power, suitable navigation solutions (for mobile systems), and sensor stability. Existing communication solutions are often expensive and power hungry. Operational challenges are often related to fouling, such as bio-fouling but also fouling due to floating or submerged debris (e.g., ghostnets). Coastal observations often suffer from effects of surface traffic, leading to collisions, unwanted recoveries or a lack thereof, thus leading to limitations in communications, maintenance and recovery. Additionally, existing legislation in coastal regions might limit the operations with unmanned vehicles. The extremely dynamic coastal domain requires significant sensor density to establish an adequate observation capability. Despite recent developments in

[1]Ocean Infinity (2020). Discover Our Projects, available online at https://oceaninfinity.com/projects/.

cost-effective ocean sensors and platforms (Wang et al., 2019), often the cost of sensors alone poses a severe economic challenge to the establishment of a sustainable and comprehensive coastal observation system.

IoT Communication Technologies

In open ocean environments satellites are the only viable communications means, but they are often expensive solutions and can present high latency. Recently, several new initiatives have promised cheaper and faster satellite-based solutions for data collection and continuous monitoring in the ocean. Nevertheless, in coastal areas other technologies are available that offer significant advantages in terms of cost, throughput and latency. For applications in which a limited number of small messages are adequate, long-range low-power wide-area networks (LPWAN) are an inexpensive solution that can reach considerable distances from the coastline.

Among the several LPWAN implementations, the most prominent are Sigfox, LoRaWAN, NB-IoT, and LTE-M (Mekki et al., 2018). Sigfox is currently deployed in more than 70 countries in all five continents, able to cover 1.3 B people[2], while LoRaWAN is currently deployed in 162 countries[3]. When choosing among the different options, the main factors to be considered are: communication range, data rate, spectrum usage, number and size of messages, energy consumption, and cost (Mekki et al., 2019). Sigfox is a proprietary long-range low-power solution widely used for IoT, but it is quite limited in the number of messages per day and the amount of data that can be transferred. LoRa is also based on a proprietary technology, but only at the physical layer, while the data transmission protocol, namely LoRaWAN, is open and free to use and deploy. As for cellular communications, they have been traditionally leveraged for IoT, but their high-power consumption limits their use in battery-operated devices. Newer cellular technologies (e.g., NB-IoT, LTE-M) consume less power, offer longer ranges, and are protected from interference thanks to their use of licensed frequencies. Therefore, where these cellular technologies are available, they can offer a viable solution, but currently they are not yet widely deployed.

Given the characteristics of these long-range technologies (reviewed in Mekki et al., 2019; Parri et al., 2019; Cecílio et al., 2020; and Park et al., 2020) it appears that LoRa implementations are the most effective solutions for IoT-based coastal monitoring systems. A considerable advantage of LoRa is the use of unlicensed spectrum at frequencies below 1 GHz, which are available in most countries and may remove a significant factor in the cost of the communications services. This allows even small organizations to install, operate and maintain the network, independently of commercial communication providers. This can be particularly appealing for building sustainable observing systems with common protocols worldwide, including developing countries where commercial communication services can be unaffordable, or even lacking in certain areas. The use of unlicensed spectrum poses, however, limits to the maximum allowable transmission power and channel occupancy time, to allow sharing the resource among concurrent users. These limitations are country-specific but as exemplified by the great success of WiFi do not constitute a major obstacle for its widespread usage.

The open LoRaWAN protocol offers variable transmission speeds from about 300 bps up to 50 kbps and paves the way for a complete solution from the sensing ocean platform to the network server and from there to a number of application-specific web accessible servers (**Figure 2**). Experiments on coastal areas have already shown that LoRaWAN-based systems yield promising results (Petajajarvi et al., 2015; Parri et al., 2019). Recently, the impact of the height of the nodes when deployed in the water have been investigated (Cecílio et al., 2020), demonstrating that tides have an impact on the communication distance and reliability. A factor that needs to be considered when building the system, by for example positioning the end-nodes as high as possible.

The LoRaWAN gateways and servers can be installed and managed directly by the interested party, but their functionalities can also be obtained from commercial providers or leveraging crowdsourcing initiatives such as the *TheThingsNetwork*[4], a global endeavor that provides LoRaWAN services at no cost. Several end-devices installed on the sensing platform send messages to one or more LoRaWAN gateways, which will then forward them to the network server using IP-based connections (Seid et al., 2020). Nevertheless, when there is a need for higher data rates (e.g., transmission of videos) modified WiFi for long-distance offers a low-cost solution that can be directly installed by the interested organization (Pietrosemoli et al., 2014). Finally delay-tolerant transmission protocols (Msaad et al., 2020) should be considered since some devices, due to their mobile nature, might be connected with the on-land infrastructure only intermittently.

The IoT enabled devices will be crucial in the framework of Digital Ocean initiatives as they will provide the means to assimilate ocean measurements including the potential ability to modify the sampling behavior, on temporal and spatial scales, according to the needs of the digital ocean initiatives (such as the Digital Ocean Twin). This is also a prime example of a cooperative vs. coordinated system.

DISCUSSION

Essential elements toward the implementation of a distributed and autonomous architecture for ocean monitoring and observation are described to achieve the ability to measure physical, chemical, and biological variables across a range of spatial and temporal scales in coastal areas. Cooperative autonomous devices with on-board system management and data processing should be combined with low-power long-range communication technologies, to coordinate data acquisition and management and enable machine-machine interactions, to deliver fit-for-purpose information to a range of end-users with

[2]https://www.sigfox.com/en/coverage

[3]https://lora-alliance.org/

[4]https://thethingsnetwork.org

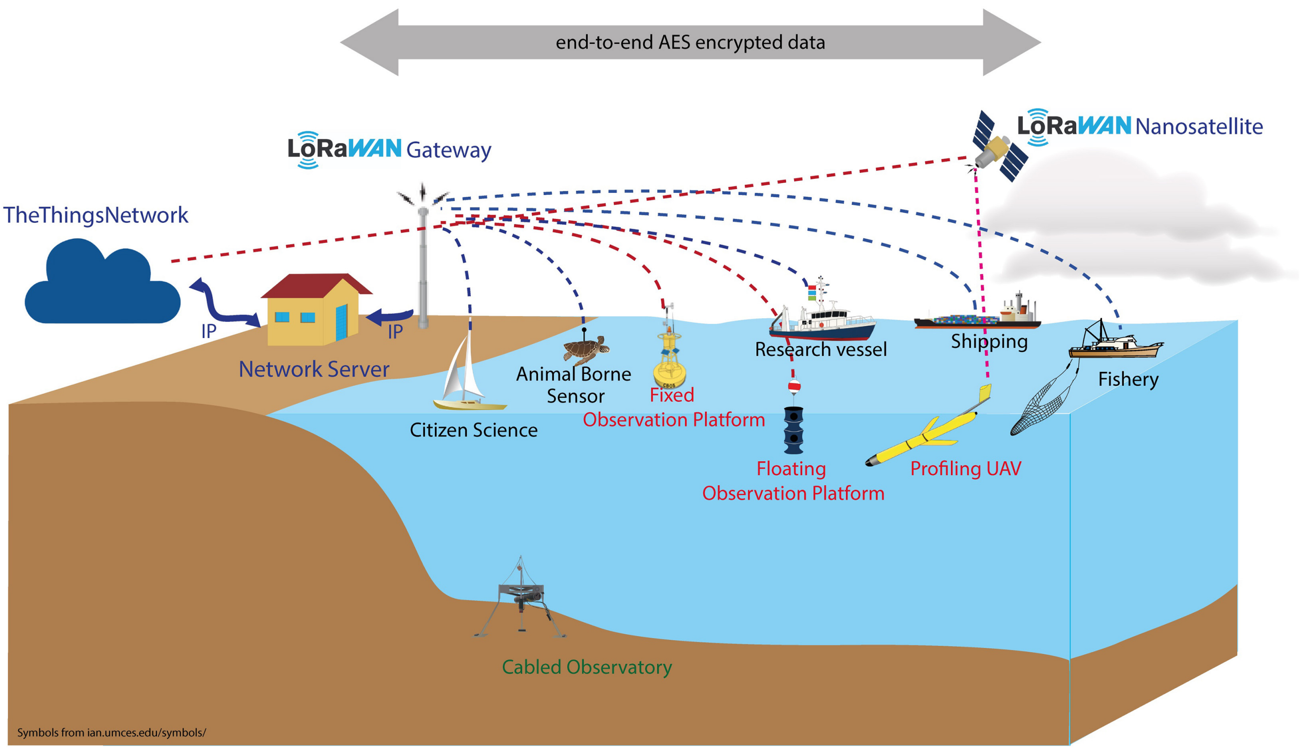

FIGURE 2 | Schematics of the IoT communication network in the ocean. The names in red identify dedicated platforms, in blue cooperative platforms, in green cabled platforms. The dashed lines represent the communication channels connecting platforms to the gateway using a specific wireless communication protocol depending on local availability and cost. The gateway is connected to the network server by any IP communication technology available. The network server connects to a number of specific application servers which make the data accessible through a web browser. The transmission channel employs an end-to-end AES encryption protocol. The schematics includes gliders that in the future may be connected to the system *via* LoRaWAN satellite communication as the other open-ocean platforms.

complex and diverse requirements. These transformative changes in the use of marine robotics, communication technology and autonomous data handling, can strengthen modern coastal ocean observing systems by supporting their economic viability while addressing overlapping interests of end-user groups from science, technology, industry, and policymakers.

The global ARGO program is a prime example of how such step-change can be achieved, demonstrating the successful implementation and operation of a global blue ocean observation system, technologically and scientifically, over the past 20 years with major perspectives for future developments (Roemmich et al., 2019). Albeit being a homogeneous observation system it illustrates the realization of key functionalities such as simple m2m communications, on-board data preprocessing, data quality and sensor standards as well as model integrations and bespoke data products for the open ocean.

Coastal observatories have different economic challenges compared to the open ocean, as they are nationally funded and serve the requirements of several stakeholders; e.g., water authorities, legislative bodies, aquaculture companies, etc. Hence a modular and flexible structure in the observational system is needed to adapt to local requirements while ensuring a coherent data access and interoperability. The Balearic Islands Coastal Ocean Observing and Forecasting System (SOCIB, Tintoré et al., 2013) provides operational solutions for a multi-platform, integrated and multidisciplinary observing system which is able to leverage system's diversity and automation to better respond to end-user needs (Heslop et al., 2019). To achieve integration in such heterogenous system communication and coordination among nodes is paramount (Leonard, 2016; Thompson and Guihen, 2019) and semi-autonomous planning and monitoring tools should evolve to integrate numerical ocean and atmospheric models as well as *in situ* data in order to coordinate and optimize usage of the individual nodes. Underlying all of these attributes should be the adoption of standard methods and best practices to create a foundation for the desired interoperability (Pearlman et al., 2019).

The capacity of a network of heterogeneous system to transfer data is a critical issue, since the quantity of data produced by the platforms is expected to be large. Delay tolerant methods and protocols allowing local data storage and retrieval of the platform could be needed to secure collection of large data. Moreover, modified WiFi could enable near-real time long-distance information transfer of broadband data (Pietrosemoli et al., 2014). Additionally, communication of data or information from mobile systems can rely on surface radio-frequency communications or underwater communication to cabled seafloor communication nodes. While acoustic underwater communication will improve in terms of reliability and in effective bandwidth, high-bandwidth, low-energy, long-range communication (>10 km) will most likely not be achieved (Song et al., 2019). Low-power on-board data processing, fusion and data compression algorithms can alleviate some of the shortcomings imposed by the physics of the acoustic communication channel. Similarly, underwater optical communications will play in the future a more prominent role providing short range point-to-point high bandwidth (>10 Mbps) line-of-sight connectivity. It is foreseen that combined (optical, acoustic) communication systems could provide the means for reliable underwater communication for a range of environmental conditions such as those found in coastal regions.

The ability to benefit from a wide range of diverse sensing platforms to monitor EOVs, could greatly expand our ability to achieve sustained coastal observations in the global ocean. Indeed, the potential low cost of the distributed observational framework can enable data collection, handling and storage in areas that are currently poorly observed (e.g., in developing countries). The arctic region is an important example of where advanced in communication and automation can contribute to design and implementation of sustained observing system (Lee et al., 2019). A network of moored, wirelessly connected platforms, complemented by autonomous vehicles with advanced control systems and a range of low-cost sensors could provide the baseline for sustained coastal observations in many regions. Those systems might initially not have all the EOVs but could be expanded upon availability of reliable sensors to fill the EOV gaps.

The need for sensors in the coastal zone is essentially the same as for the offshore environment, however, the concentrations of the compounds of interest, nutrients, pollutants, etc., are generally higher in the shelf areas. The coastal observatories may then require more frequent servicing due to the faster degradation of sensor performances as a result of biofouling, adverse impacts of oil slicks, marine litter, etc. It has been demonstrated that regular servicing of the observatory and sensors could be performed with unmanned autonomous vehicles (Barceló-Llull et al., 2019; Scoulding et al., 2020) which could also collect samples for ground-truthing to be analyzed using standard techniques to ensure that the data collected are reliable. The availability of reliable and cost-effective ocean sensors is central to the implementation of large distributed observation networks. Some of the required sensors to monitor EOVs are not yet available, but considering the fast evolving technology, many of the variables not covered now, will be likely covered in the near future (Wang et al., 2019).

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

PM and AC provided the main concept with contributions from RB, SK, and EP. PM drafted the first version of the manuscript. AC and PM primarily introduced the network of distributed observations for coastal areas. RB, ED, and FT primarily contributed to the automation and collaborative robotics parts. EP and SK primarily contributed to IoT communication technologies. All authors provided collaborative effort for this work. All authors contributed to the text for the introduction, discussions and in providing relevant concepts and references for the different sections.

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A Review of Sustainability Concepts in Marine Spatial Planning and the Potential to Supporting the UN Sustainable Development Goal 14

Trine Skovgaard Kirkfeldt[1] and Catarina Frazão Santos[2,3]*

[1] Department of Planning, Aalborg University, Copenhagen, Denmark, [2] Marine and Environmental Sciences Center, Faculdade de Ciências, University of Lisbon, Lisbon, Portugal, [3] Environmental Economics Knowledge Center, Nova School of Business and Economics, New University of Lisbon, Lisbon, Portugal

Ecosystems all over the world are under increasing pressure from human uses. The UN Sustainable Development Goal 14 (UN SDG 14) seeks to ensure sustainability below water by 2020; however, the ongoing biodiversity loss and habitat deterioration challenge the achievement of this goal. Marine Spatial Planning (MSP) is a developing practice with a similar objective to the UN SDG 14, albeit research shows that most MSP cases prioritize economic objectives above environmental objectives. This paper presents an assessment of how MSP can contribute to achieving the UN SDG 14. Results are presented in three steps. First, a representative definition of MSP is presented. Secondly, activities that can be addressed through MSP are laid out. Lastly, results are used to assess how MSP can contribute to the achievement of the UN SDG 14 targets and indicators. This assessment shows great potential for MSP to play a role in the achievement of the UN SDG 14.

Keywords: maritime spatial planning, ocean planning, ocean governance, sustainable ocean use, marine conservation, ocean sustainability

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****Correspondence:***
Trine Skovgaard Kirkfeldt
tsk@plan.aau.dk

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INTRODUCTION

The increasing level of interest in the marine space has put severe and diverse pressures on marine ecosystems. For this reason, the United Nations Sustainable Development Goal 14 (UN SDG 14), Life Below Water, was formulated with the objective to "*Conserve and sustainably use the oceans, seas and marine resources for sustainable development*" (UN, 2015). To achieve this purpose, the UN SDG 14 addresses a variety of topics, from marine pollution to ocean acidification, conservation of marine ecosystems, and fishing regulations, among others (see UN, 2021). Still, the 2019 status report on progress toward the SDGs concluded that the level of protection globally is inadequate and incapable of combating the major threats of ocean acidification, overfishing, and eutrophication—even if the number of marine protected areas (MPAs) is growing worldwide. Indeed, it states that "(. . .) *increased efforts and interventions are needed to conserve and sustainably use ocean resources at all levels*" (UN ECOSOC, 2019).

One way of increasing such effort is through marine spatial planning (MSP). MSP has been globally recognized as a way to foster sustainable use of marine ecosystems and to promote ocean conservation (Ehler and Douvere, 2009). As laid out by the European Union Directive on MSP (MSPD), Directive 2014/89/EU, the objective of MSP is to "(. . .) *promote the sustainable growth of* *maritime economies, the sustainable development of marine areas and the sustainable use of marine resources*" (European Commission, 2014). For this reason, the purposes of MSP largely mirror the ones of the UN SDG 14. Indeed, they are both focused on sustainable development of maritime activities and economies while at the same time conserving and ensuring sustainable use of marine areas. By concept, MSP should therefore be able to contribute to the achievement of the SDG 14 (Ntona and Morgera, 2018; Frazão Santos et al., 2020; Calado et al., 2021).

However, research has found ambiguities regarding how MSP should balance objectives for environmental protection and economic development (Douvere and Ehler, 2008; Gilliland and Laffoley, 2008; Maes, 2008; Katsanevakis et al., 2011; Trouillet, 2020). One of the main contributors to such ambiguity is the dichotomous role of MSP in ensuring both environmental and economic objectives at the same time. This ambiguity has resulted in MSP cases predominantly focused on achieving economic objectives before planning for environmental objectives (Jones et al., 2016; Trouillet, 2020). This prioritization supports what is also referred to as *weak sustainability*, as it relies on a fragile foundation if the health of marine ecosystems is not secured. Weak sustainability comes from an economic perception that all capitals are replaceable, i.e., natural capital can be replaced with the right financial or societal capital (Bateman and Mace, 2020). In contrast, planning that ensures environmental sustainability before addressing objectives for economic activities builds a strong and sustainable foundation for marine ecosystems and depending maritime economies, thus aiming for *strong sustainability* (Mee et al., 2008; Frazão Santos et al., 2014). Jones et al. (2016) found vast differences between MSP in theory and MSP in practice, with MSP cases focused on blue growth and economic development being much more prevalent than ecosystem-based MSP focused on a strong sustainability approach (Jones et al., 2016).

This paper aims to further explore and clarify the potential contribution of MSP to achieving SDG 14 and related targets. While doing so, it also aims to decrease the ambiguity regarding the dual role of MSP in supporting both ecosystems protection and human development. These objectives are attained by conducting an in-depth analysis of key literature on MSP, assessing key MSP definitions, and offering examples for concrete action.

MATERIALS AND METHODS

The present study is composed of three main methodological phases, all of them based on the revision of the most cited documents (Scopus database) on both marine and maritime spatial planning. These are: (1) the development of a representative MSP definition; (2) the analysis of the main human uses incorporated or managed in MSP initiatives; (3) the investigation of the contribution of MSP to each target of the SDG 14. Specificities on each phase are provided in the following sub-sections.

First, in order to identify the most applied MSP definitions in scientific literature, the Scopus database was used to search documents that included the terms "marine spatial planning" or "maritime spatial planning" in their title, abstract, or keywords. After reviewing the 50 most cited documents (see **Supplementary Material A**), a pattern in definitions was clear (e.g., literature sources, wording). Most of these 50 documents used secondary sources to defining MSP, in many cases the same ones. These amounted to a total of 30 "defining" documents (see **Figure 1** and **Supplementary Material B**). The 30 defining documents were carefully examined for explicit MSP definitions, which were then extracted for further analysis using Nvivo (2020), and coded based on two overall questions: (1) What is MSP? (2) What is the purpose of MSP? Each of these coding processes led to a list of answers. The most applied elements were then sought combined into one representable definition of MSP. This required some creativity in how to bind all the elements together into one formulation, for which the wording of the coded definitions was used as guidance. In order to test the representativeness of the formulated definition, the latter was compared with a word frequency test (of all definitions from the 30 defining sources) using Nvivo (see **Figure 2** and **Supplementary Material C**). This comparison made it possible to see if any central terms or aspects of MSP were missing from the formulation of the combined MSP definition.

Second, the set of 30 defining documents were manually reviewed for an examination of the human uses and activities that take place in ocean space, and which can be generally addressed and managed through MSP processes (see **Supplementary Material B**). This analysis allows for a comparison of the type of ocean uses and activities that MSP can plan for, and the uses and activities addressed in the SDG 14 targets. Based on the identified human uses and activities, a list of search words (see **Supplementary Material D**) was then established and used to perform a word count for the 50 most cited MSP-related documents, in order to assess which ocean uses gathered the most attention.

Finally, by using the results of the first two stages, a qualitative analysis was developed to unravel the potential contribution of MSP to achieving SDG 14 (see **Figure 1**). This analysis used a list of search words related to each of the 10 SDG 14 targets (see **Supplementary Material E**) and focused on a manual review of the 50 most cited MSP documents—which were investigated regarding how MSP could contribute to achieving each of the targets. Additional relevant sources were also consulted for guidance about which specific actions could be undertaken by MSP initiatives, especially when considering the set of ocean uses MSP can plan for.

RESULTS

Defining Marine Spatial Planning

The in-depth analysis of the 30 defining MSP documents resulted in a list of terms commonly used to describe "what MSP is," some of which being more often referred (**Figure 3**). The most common terminology—mentioned in 11 defining documents—described MSP as being some type of "process" (either in general terms, or specifically as a planning or public process). In addition,

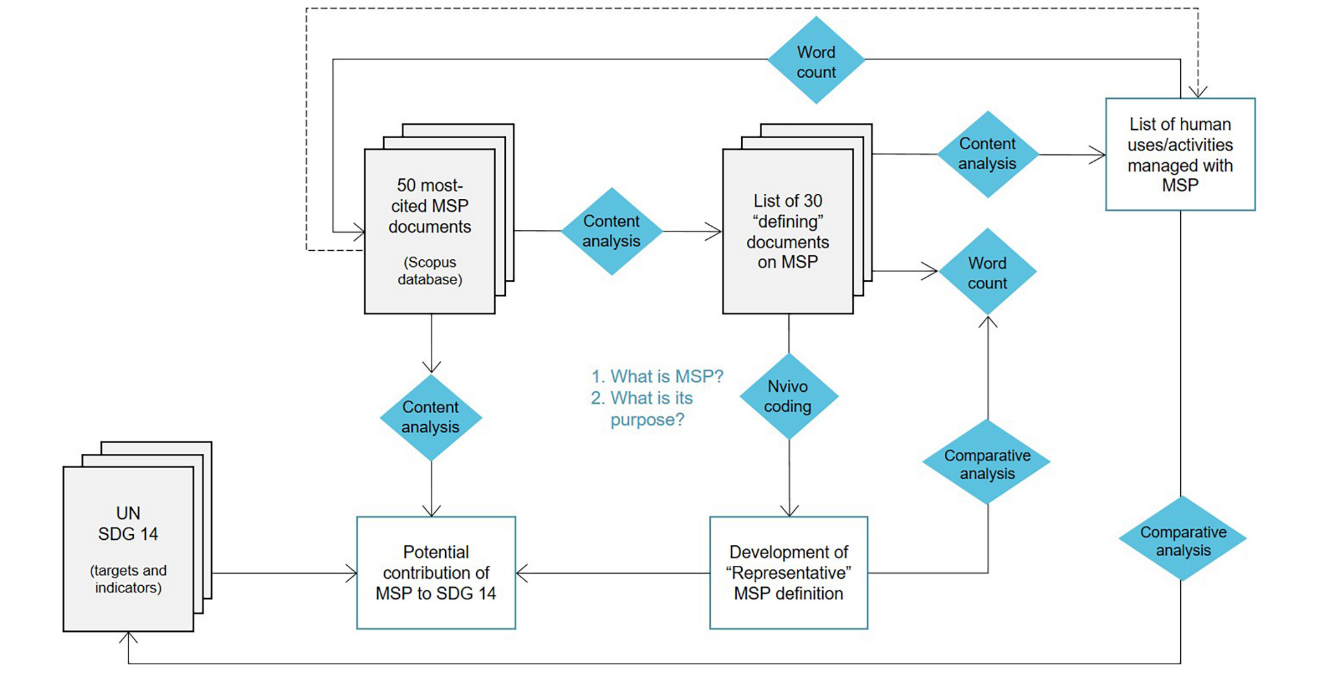

FIGURE 1 | Illustration of the methodological process used to establish a representative MSP definition, analyze human uses addressed in MSP, and investigate the contribution of MSP to each target of the SDG 14.

5 documents described MSP as being a type of "management," and 3 documents as a way to implement the ecosystem-based approach (EBA) [albeit there are some disagreements as to whether MSP implements EBA or is part of ecosystem-based management (Kirkfeldt, 2019)].

By combining the most applied terms, a preliminary MSP definition could be described as follows:

"Marine spatial planning is ***a public, planning process*** *and an* ***element of ecosystem-based sea use management****."*

During this preliminary search, the multifunctional purpose of MSP became vivid, with the 30 defining documents providing a long list of purposes for MSP (**Figure 4**). A shared element of the listed purposes was the focus on human uses and maritime activities, namely concerning solving potential conflicts among uses and between uses and the environment. A peculiar aspect, especially relevant when considering the role of MSP in achieving SDG 14, is that purposes including the words "sustainability" or "sustainable" are not among the top purposes in **Figure 4**. Indeed, among the 21 identified purposes, "Support sustainable development" and "Manage activities more sustainably" appear only in the 12th and 21st positions, respectively. Still, some of the most frequently mentioned purposes also relate to sustainability concepts. The latter is the case of the purposes "Achieve ecological, economic and social objectives" (the second most identified one, mentioned in 13 out of the 30 documents, and which addresses the three pillars of sustainable development) and "Sustain ecosystem services" (the fifth most identified purpose, identified in 7 out of the 30 documents).

Adding the purpose to the summarized description obtained earlier, MSP could be described as:

"Marine spatial planning is ***a public, planning process*** *and* ***an element of ecosystem-based sea use management,*** *that aims to prevent* ***conflicts among maritime uses and between human uses and the environment,*** *through a* ***strategic*** *and* ***rational, spatial and temporal, distribution of activities*** *in order* ***to achieve environmental, social and economic objectives,*** *such as* ***sustaining ecosystem services*** *and* ***improve decision-making.*** *The process involves the implementation of* ***environmental protection,*** *the facilitation of* ***co-location of compatible*** *uses, and the assessment and management of* ***cumulative impacts****."*

When comparing the formulation above with the word frequency test performed on the MSP definitions from the 30 defining documents, it became evident that this formulation was a valid representation of the word cloud (**Figure 2**).

The absence of sustainability concepts is, however, again evident. In effect, not a single sustainability concept appears among the 40 most applied words that constitute the word cloud. The word "sustainable" is the 95th most cited word, and therefore not displayed in the word cloud. By contrast, in the MSPD there is a substantive emphasis on sustainability. The word "sustainable" is the 11th most cited word (when excluding the term "maritime spatial planning"), being written 25 times over 11 pages (Kirkfeldt et al., 2020) and being the second most cited environmental-related word (Frazão Santos et al., 2015).

Human Activities and Uses to Address Through Marine Spatial Planning

The list of human uses and activities mentioned in the 30 defining documents is displayed in **Figure 5**, together with the

FIGURE 2 | Word cloud generated by Nvivo based on the definitions of MSP found in the 30 defining documents. The words "marine," "spatial," and "planning" were excluded from the word frequency analysis to not influence results. The size of each word represents the percentage of all citations relative to the other words. Baseline data can be found in **Supplementary Material C**.

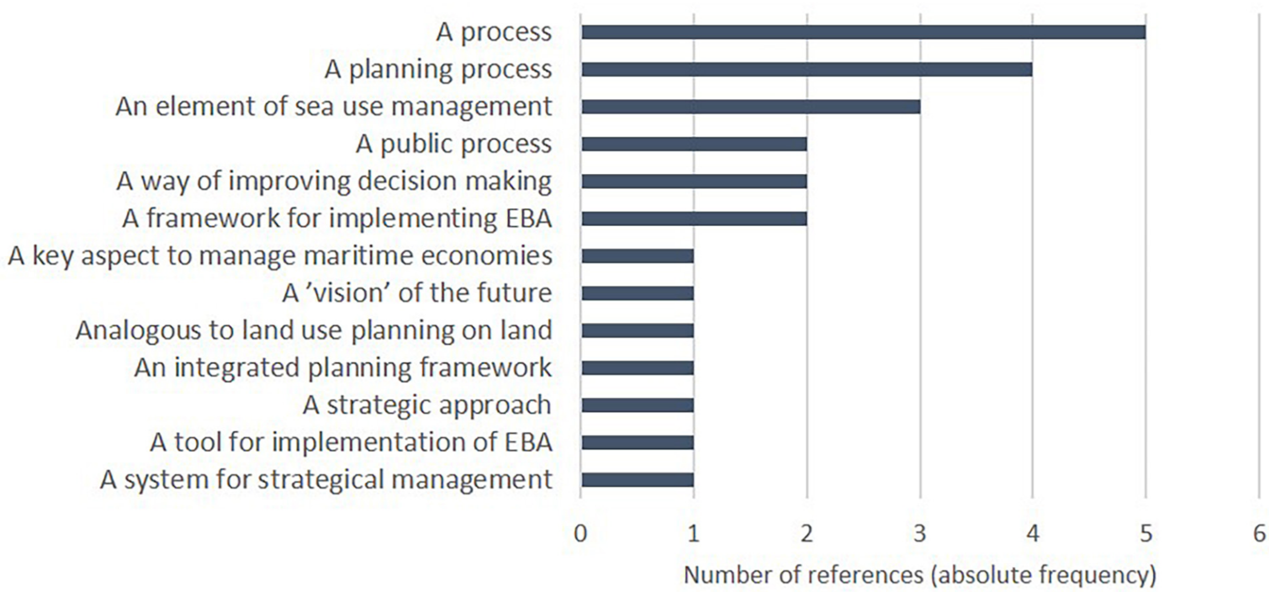

FIGURE 3 | Main definitions of what MSP is, found in the 30 "defining" documents. Five out of the 30 documents only defined what MSP does and not what MSP is. For that reason, they are not reflected into the graphic. Baseline data in **Supplementary Material B**.

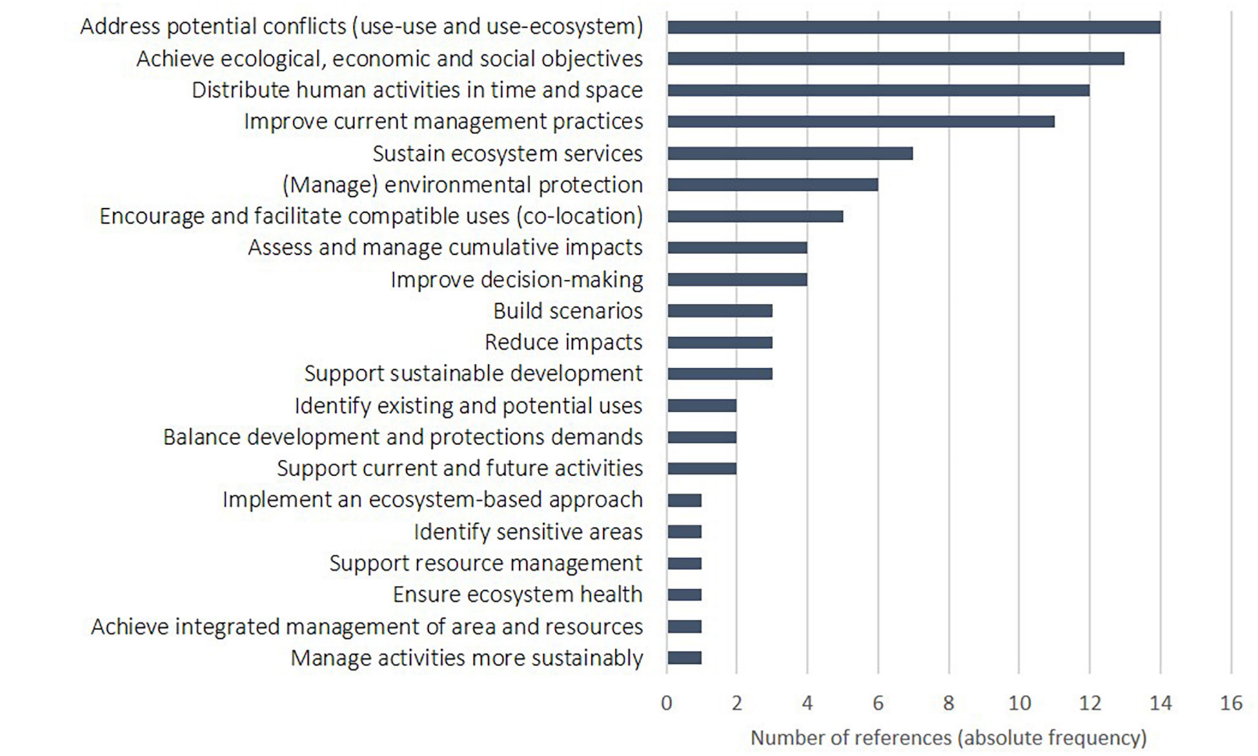

FIGURE 4 | Purposes of MSP found in the 30 "founding" documents and the number of times each purpose appears in such documents. Baseline data in **Supplementary Material B**.

corresponding word frequency results for the 50 most cited MSP documents. The list of uses and activities in **Figure 5** is diverse, and spans from on-shore, coastal activities (e.g., tourism, ports, and harbor activities) to off-shore activities (e.g., renewable energy, oil and gas activities, shipping, off-shore aquaculture). Many of these activities also correspond to sectors that were traditionally managed separately and through different institutional setups (Maes, 2008). Moreover, while some activities are managed nationally, others have a more transboundary nature. For example, where tourism is mainly managed at the country level, shipping and fishing activities are also managed through international frameworks, such as the International Council for the Exploration of the Seas (ICES, 2020) and the International Maritime Organization (IMO, 2020) (Blundell, 2004; Maes, 2008).

The word count showed that some activities receive much more attention in the MSP context. The most cited uses of the ocean space are those related to marine conservation and protection, renewable energy activities, and fishing (**Figure 5**). These activities are all known to be prone to conflicts, either among themselves or between them and other activities or stakeholders. Conflicts among the three activities can occur, for example, when fisheries are excluded from a new protected area or from a wind farm area (Agardy et al., 2011; White et al., 2012). Conflicts with stakeholders and other activities are often seen in relation to the establishment of a new wind farm, where conflicting interests of coastal residents and shipping and recreational activities exist (Ehler and Douvere, 2009; White et al., 2012). The level of potential conflicts surrounding these activities might explain the high citation numbers in the analyzed literature.

The Role of Marine Spatial Planning in Achieving SDG 14

The limited use of sustainability concepts in MSP definitions (discussed in section "Results") is noteworthy and especially relevant when considering the contribution of MSP to achieving SDG 14. This raises the question: Can MSP play an important role in achieving SDG 14, despite the lack of sustainability focus in the studied "defining" MSP documents? We address this question by analyzing the links between MSP and each SDG 14 target, as presented below and summarized in **Table 1**.

Target 14.1. Marine Pollution

The first SDG 14 target points to a sensitive issue in MSP. First, being a "spatial" practice, to which extent can MSP regulate pollution from sectoral activities? Second, being a "marine" practice, what is MSP potential to address land-based pollution sources? The indicator of target 14.1 is composed of two separate sub-indicators: (a) an index of coastal eutrophication; and (b) floating plastic debris density. Eutrophication is strongly linked to nutrient runoff from agricultural activities, and plastic debris has been found to derive primarily from land-based sources (c. 80% Jambeck et al., 2015; Sherrington, 2016). While the UN

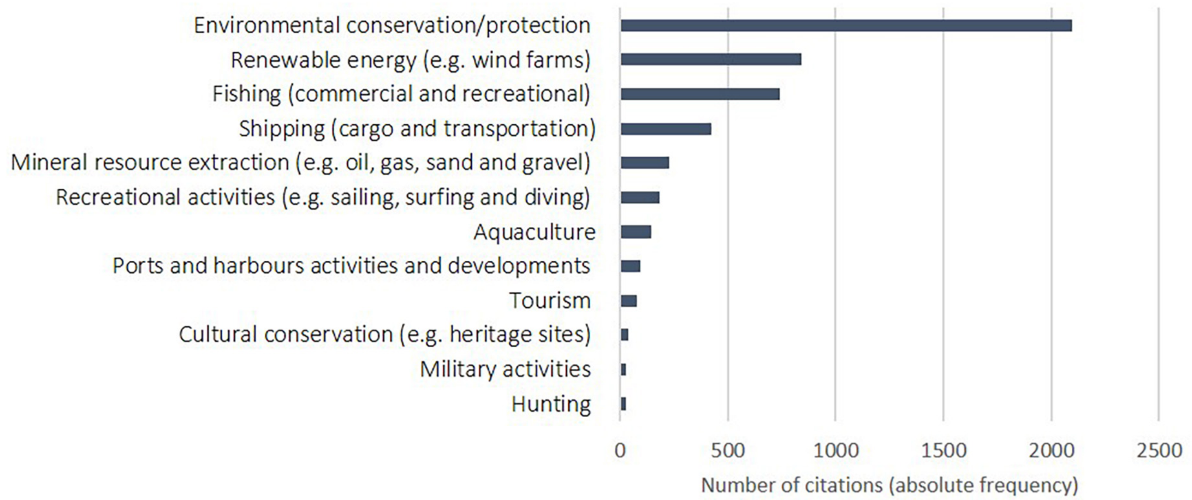

FIGURE 5 | A word count on key activities that can be addressed through MSP.

considers eutrophication—together with overfishing and ocean acidification—to be a key impact that is impossible to address with the current level of protection at sea (REF), the need to address land-based sources of pollution is highlighted. In one of the 30 defining MSP documents, the authors suggested that MSP can play a role in formulating regulations for "*the amount of fertilizers and pesticides applied to agriculture lands*"(Ehler and Douvere, 2009). Ehler and Douvere (2009) suggest this as a non-spatial management measure that might be necessary, albeit seldom applied, to achieve MSP objectives. However, the role of MSP in addressing what is called "land-sea interactions" (LSI) has been a topic for much debate and confusion. Indeed, in 2017 MSP practitioners met at a conference to discuss how to address land-sea interactions in MSP (Kidd et al., 2019). The practice of addressing LSI in MSP is, however, still limited and highly debated. Full integration of terrestrial and marine planning systems has been suggested as a way to facilitate better considerations for LSI, but it bears a number of challenges (Kidd and Ellis, 2012; EC, 2017; Kidd et al., 2019). While pollution from land is a dominant impact on marine ecosystems, marine pollution also derives from maritime activities (e.g., lost fishing gear and oil spills). It has been suggested that MSP could address the amount of lost fishing gear by making restriction zones for specified types of gear (e.g., bottom trawls) (Blundell, 2004), and that MSP could coordinate with risk and vulnerability analyses related to oil spills due to the shared spatial dimension of the two processes and a similar demand for data (Frazão Santos et al., 2013).

Target 14.2. Manage and Protect Marine and Coastal Ecosystems

To avoid adverse impacts on the marine environment, this target aims for a sustainable management and protection of marine and coastal ecosystems. The aim of target 14.2 is in line with the initial purpose of MSP, as exemplified for example by the case of the Great Barrier Reef Marine Park. The practice of MSP was originally considered (and is today still) a means to implement ecosystem-based management (Douvere, 2008)—as seen in the coded definitions. By implementing EBA, MSP could play a key role in achieving target 14.2, as the indicator pertains to the "*number of countries using ecosystem-based approaches to managing marine areas*" (UN, 2021). Indeed, three of the most cited "purposes" of MSP, as displayed in **Figure 4**, are related to target 14.2 (namely, manage "environmental protection," "assess and manage cumulative impacts," and "reduce impacts"), all of them being key elements of EBA (Kirkfeldt, 2019). As suggested by the "defining" documents (e.g., Blundell, 2004; Ehler and Douvere, 2007; Douvere, 2008), this indicates a high potential for MSP to contribute to target 14.2. The assessment of cumulative impacts has also been identified as of high importance if MSP is to prevent adverse environmental impacts (Halpern et al., 2008). Indeed, MSP can play a key role in reducing impacts on the marine environment through spatial restrictions (e.g., restrictions toward the use of bottom-trawling gear in certain areas), or restrictions of the total extent/intensity of high impact activities such as fishing, oil and gas extraction, and shipping (Blundell, 2004; Ehler and Douvere, 2009).

Target 14.3. Minimize and Address the Impacts of Ocean Acidification

Ocean acidification takes place because of the rising concentration of carbon dioxide in the atmosphere, which is absorbed by, and thus acidifies, the ocean (IPCC, 2019). While climate change in general is often neglected in MSP process, there are several potential pathways for how MSP can minimize and address climate-related impacts, including the ones from ocean acidification (Frazão Santos et al., 2020). Target 14.3 focuses on reducing and addressing the impacts of acidification, and this can include actions for climate change mitigation such as the development of wind farms. Indeed, by supporting the development of renewable energy production, allocating areas to blue carbon capture and storage, or limiting available space

TABLE 1 | Potential contribution of MSP initiatives to meeting each of the 10 targets of the UN SDG 14 (see detailed information in section "Discussion and Conclusion").

UN SDG 14 targets	Actions to be carried in MSP initiatives
Prevent and significantly reduce marine pollution of all kinds (Target 14.1)	• Encourage and support full integration with terrestrial planning • Exclusion of bottom-trawling activities from certain areas to prevent lost fishing gear • Cooperation with risk and vulnerability analyses carried for human hazards such as oil spills • Contribute to regulations for the amount of fertilizers and pesticides applied to agriculture
Sustainably manage and protect marine and coastal ecosystems (Target 14.2)	• Apply an ecosystem-based approach • Assess cumulative impacts • Establish spatial restrictions for high impact activities (e.g., fishing, oil and gas extraction or shipping) in particularly important marine areas • Allocate marine space for conservation areas
Minimize and address the impacts of ocean acidification (Target 14.3)	• Contribute to a green transition by prioritizing renewable energy developments (e.g., wind, wave and tidal) and reducing high-CO2 emitting activities (e.g., oil and gas, shipping) • Contribute to increased resilience of ecosystems by reducing non-climate human pressures (e.g., from pollution, overfishing and habitat losses)
Effectively regulate harvesting and end overfishing, illegal, unreported and unregulated fishing (Target 14.4)	• Establish "no-take" marine zones • Establish "trawling-free" marine zones • Regulate fishing activities through non-economic incentives and regulations (e.g., by setting limits for allowable catches) • Discourage IUU fishing activities (e.g., by establishing artificial reefs)
Conserve at least 10 per cent of coastal and marine areas (Target 14.5)	• Support the establishment of marine protected areas (MPAs) in at least 10% of the marine area • Ensure that MPAs are ecologically beneficial • Ensure proper monitoring and enforcement of MPAs
Prohibit certain forms of fisheries subsidies (Target 14.6)	• Combat IUU and overfishing through initiatives mentioned in target 14.4
Increase the economic benefits to Small Island developing States and least developed countries from the sustainable use of marine resources (Target 14.7)	• Support the development of sustainable fishing practices (e.g., by establishing MPAs, no-take zones or trawling-free zones to ensure healthy fish stocks) • Prioritize the allocation of space to eco-tourism • Prioritize zones for less polluting aquaculture activities (e.g., cultivation of seaweed, oysters, and mussels)
Increase scientific knowledge, develop research capacity and transfer marine technology (Target 14.a)	• Identify knowledge gaps when assessing environmental impacts and ocean health • Use geo-technologies such as remote sensing and GIS for the generation of new data and development of technologies • Make data and technologies available for other usage and further development
Provide access for small-scale artisanal fishers to marine resources and markets (Target 14.b)	• Prioritize areas to small-scale fisheries • Facilitate access to markets through stakeholder involvement and capacity building
Enhance the conservation and sustainable use of oceans and their resources by implementing international law (Target 14.c)	• Develop marine spatial plans in compliance with UNCLOS

for high-emission activities (Frazão Santos et al., 2020), MSP can play a key role in national strategies for climate change mitigation and thus the reduction of ocean acidification. Adverse impacts from acidification on marine species include reduced calcification and growth rates in skeletons and shells, changes in metabolism and in ecological connectivity (Committee on the Development of an Integrated Science Strategy for Ocean Acidification Monitoring, 2010; IPCC, 2019). These impacts influence the services that marine ecosystems deliver, something that MSP is intended to protect according to seven of the 30 defining MSP documents (see **Figure 4**). Ensuring healthy ecosystems and a good environmental status becomes even more relevant in face of climate change, as it provides for more resilient ecosystem components, thus increasing the chance of survival and potential adaptation to a more acidic environment (Committee on the Development of an Integrated Science Strategy for Ocean Acidification Monitoring, 2010). MSP can also contribute to such resilience by reducing non-climate related impacts from for example pollution, overfishing and habitat loss (Ehler and Douvere, 2009; Frazão Santos et al., 2020; Rilov et al., 2020). Increasing ecosystem resilience is part of target 14.2, and actions in MSP to increase ecosystem resilience will therefore support both the achievement of targets 14.2 and 14.3.

Target 14.4. Effectively Regulate Harvesting and End Overfishing, Illegal, Unreported and Unregulated Fishing, and Destructive Fishing Practices

The fourth target of the SDG 14 puts focus on the management of fishing activities with the goal to prevent the depletion of fish stocks. MSP can regulate the type and intensity of fishing activities within specified areas. No-take zones and zones where certain fishing equipment is not allowed (such as bottom trawls) have been found effective in securing benefits for both conservation and fishing (Blundell, 2004). While the creation of specific zones is one way that MSP can contribute to the achievement of target 14.4, indicator 14.4.1 focuses on the "*Proportion of fish stocks within biologically sustainable levels*" (UN, 2021) which indicates the need for a more holistic management of fishing activities—something that cannot be ensured solely through zoning. In addition to zoning procedures, MSP has been suggested to regulate fishing activities by supporting the implementation of non-economic incentives and regulations (e.g., setting limits for allowable catches) (Ehler and Douvere, 2009). While illegal, unreported and unregulated (IUU) fishing activities are difficult to manage through any planning or management initiative—MSP included—some spatial actions have been found to change IUU fishing activities indirectly. This is the case, for example, of establishing artificial reefs, which discourage potential IUU trawling in the area (Bishop et al., 2017).

Target 14.5 Conserve at Least 10 Per Cent of Coastal and Marine Areas

Conservation was the most cited use of the ocean space in section "Discussion and Conclusion" (**Figure 5**), and is seen as a key activity in MSP. A widespread way to ensuring conservation at sea is through the establishment of marine protected areas (MPAs). MPAs are, as well, the measuring factor of indicator 14.5.1: "*Coverage of protected areas in relation to marine areas*" (UN, 2021). MPAs can be defined as an area "*which has been reserved by law or other effective means to protect part or all of the enclosed environment*" (Lascelles et al., 2012), and are generally considered as one of the most effective conservation tools (Maes, 2008; Agardy et al., 2011). Initially, the practice of establishing MPAs was a key inspiration for the development of MSP practice (Douvere, 2008) and is now seen as a key element to ensuring an ecosystem-based approach in MSP (Ardron et al., 2008; Katsanevakis et al., 2011; Rilov et al., 2020). However, research on MPAs shows that many protected areas do not have the intended conservation effect, and that MPAs are not able to ensure ocean sustainability if not combined by other measures (Reimer et al., 2020). This can occur for several reasons, from poor management to issues in the initial scoping and design of protected area (Agardy et al., 2011). MSP can play a vital role in addressing some of these challenges and improving the current practice of MPAs (Agardy et al., 2011; Rilov et al., 2020), thus further contributing to target 14.5.

Target 14.6 Prohibit Certain Forms of Fisheries Subsidies

None of the analyzed literature suggested MSP as an ideal tool to the management of fisheries subsidies. This could be because of a clear lack of a spatial dimension in target 14.6. However, this target is strongly linked to target 14.4 (on the regulation of overfishing and IUU fisheries). Both targets aim to reduce the overall pressure from fisheries, with indicator 14.4.1 being dedicated to the status of fish stocks, and indicator 14.6.1 being more focused on management measures: "*Progress by countries in the degree of implementation of international instruments aiming to combat illegal, unreported and unregulated fishing*" (UN, 2021). While indicator 14.6.1 does not focus on the prohibition of certain subsidies, it does focus on the implementation of instruments to combat IUU. As the latter was considered as challenging, but not impossible for MSP to contribute to under target 14.4, it might constitute an indirect pathway to further contributions of MSP to target 14.6.

Target 14.7 Increase the Economic Benefits to Small Island Developing States and Least Developed Countries From the Sustainable Use of Marine Resources

Target 14.7 is the third target of SDG 14 to address fishing activities, the second most referred ocean use in section "Discussion and Conclusion" (**Figure 5**), with indicator 14.7.1 focusing on the economic development of sustainable fisheries: "*Sustainable fisheries as a percentage of GDP in small island developing States, least developed countries and all countries*" (UN, 2021). Small Island Developing States (SIDS) account for ca. 30% of the worlds' exclusive economic zones, and have thus a tremendous influence on the well-being of marine ecosystem globally. SIDS are extremely dependent on the ocean, and strongly rely on the ocean resources for human wellbeing and livelihood. Fishery is the primary economy in many SIDS and is intrinsic to their culture and lifestyles (Jumeau, 2013). However, target 14.7 goes further, focussing on activities other than fishing, such as sustainable aquaculture and tourism, to support the increase in economic benefits to SIDS and least developed countries. Fisheries, aquaculture and tourism are human activities commonly managed through MSP (**Figure 5**), and activities that rely on healthy ecosystems. The establishment of spatial restrictions (e.g., no-take protected areas, trawling-free zones) can therefore play an important role in supporting their sustainable development. For example, the definition of zones to the development of ecosystem-friendly tourism activities can provide important revenues, as well as better conditions for sustainable fishing activities (Douvere, 2008; Arkema et al., 2015). MSP can also facilitate the development of aquaculture in a strategic manner, by planning for a varied selection of aquaculture types and prioritizing least polluting activities, such as the cultivation of seaweeds, oysters and mussels (Guerry et al., 2012). However, due to the connectivity of the ocean and the mobility of marine species, local human activities depend largely on the activities that take place further off-shore (Gee and Zaucha, 2019). It is therefore important to consider the indirect contribution of MSP to target 14.7 through the role played in regards to other targets (e.g., targets 14.4 and 14.5).

Target 14.a. Increase Scientific Knowledge, Develop Research Capacity, and Transfer Marine Technology

Target 14.a focuses on increasing scientific knowledge and research capacity, in order to improve ocean health and marine biodiversity contribution to the development of developing countries, and is evaluated based on indicator 14.a.1 on the "*Proportion of total research budget allocated to research in the field of marine technology*" (UN, 2021). As MSP is a highly data-demanding practice, it often involves a large extent of data collection and analysis (Ehler and Douvere, 2009). MSP requires data on existing habitats, flora and fauna, existing and future maritime activities, and expected ecological, social and economic changes (including from climate change). Such data can be generated through geo-technologies such as remote sensing and data analysis in geographic information systems (Douvere, 2008; St. Martin and Hall-Arber, 2008). Thus, as formulated by Douvere (2008), MSP "*provides a management framework for new* *and previously inaccessible scientific information.*" It is therefore an ideal gateway for meeting 14.a, basing on the premise that data and technologies generated in MSP processes are made available to other usage and broader ocean management contexts. As target 14.a has a specific aim "*to enhance the contribution of marine biodiversity to the development of developing countries, in particular small island developing States and least developed countries*"(UN, 2021), the process of resource demanding data collection for MSP is an issue. As scientific research can be very costly, SIDS are more restricted than other states in meeting this target (Ehler and Douvere, 2009; Ehler, 2013; FAO, 2014).

Target 14.b. Provide Access for Small-Scale Artisanal Fishers to Marine Resources and Markets

Target 14.b is evaluated based on the "*Progress by countries in the degree of application of a legal/regulatory/policy/institutional framework which recognizes and protects access rights for small-scale fisheries*" (indicator 14.b.1). Because of its intrinsic characteristics, MSP can constitute such a framework. The most obvious role of MSP in this matter pertains to ensuring spatial access of small-scale fisheries to marine resources, for example, by establishing zones where only recreational and artisanal fishing are allowed, or where they have priority over other ocean uses (Blundell, 2004). However, MSP can also facilitate better access to markets, for example, by promoting communication among stakeholders. Stakeholder meetings, a key element of MSP, can bring actors in the fishing industry together, which in turn might facilitate new agreements and collaborations between small-scale fishers and market holders (Gopnik et al., 2012; Lewison et al., 2015).

Target 14.c. Enhance the Conservation and Sustainable Use of Oceans and Their Resources by Implementing International Law as Reflected in UNCLOS

The last target of SDG 14, target 14.c, focuses on nations implementation of international law, according to what is established in the United Nation Convention on the Law of the Sea (UNCLOS). Although UNCLOS does not refer to MSP as a concept, it does consider spatial planning as a facilitating tool that allows some countries to fulfill obligations within UNCLOS (Ardron et al., 2008; Maes, 2008). Indeed, the spatial boundaries set by UNCLOS, such as Territorial Waters and Exclusive Economic Zones, together with specifications for domestic rights within each zone, confirms the potential role to be played by MSP in managing marine resources (both living and non-living) within national jurisdictions (Papageorgiou and Kyvelou, 2018). While there is also a strong push for developing MSP initiatives in areas beyond national jurisdiction (Wright et al., 2019), international initiatives in the high seas are still scarce making MSP a predominantly national-level activity (Ardron et al., 2008). When considering the close connections between the legal framework of UNCLOS and MSP, especially in an ecosystem-based context, it can be said that any country with ongoing MSP initiatives is "*making progress in (...) implementing (...) ocean-related instruments that implement international law*" with the aim to "*enhance the conservation and sustainable use of oceans and their resources*" (UN, 2021) thus contributing to target 14.c.

DISCUSSION AND CONCLUSION

It is clear from this study that the practice of MSP can play an important role in ensuring sustainability for life below water and achieving SDG 14. However, it also became clear that while MSP is an ideal tool for some SDG 14 targets, others cannot be properly addressed through MSP and require alternative management approaches. In particular, spatial management measures like the establishment of conservation areas, such as MPAs, and restriction zones for fisheries, such as no-take zones or trawl-free zones, can contribute to the achievement of six out of the 10 SDG 14 targets.

Targets with a spatial dimension—such as targets 14.2 on sustainable ocean use, 14.5 on establishing MPAs, or 14.7 on fisheries, tourism and aquaculture in SIDS and least developed countries—are highly compatible with MSP practice. Indeed, the establishment of areas where certain types of fishing are prohibited would help in meeting several targets simultaneously (e.g., targets 14.2, 14.4, 14.5, 14.6, and 14.7), whereas the establishment of MPAs would contribute, both directly and indirectly, to meeting targets 14.2, 14.5, and 14.c. By contrast, targets that require non-spatial regulations such as target 14.6 on fisheries subsidies, or that address topics that go beyond the marine realm such as target 14.1 on marine pollution from land-based activities, can be more challenging to address through MSP. Indeed, while target 14.1 emphasizes the importance of considering land-sea interactions in MSP and ensuring ecosystems resilience to better endure impacts from marine pollution, ensuring this connection in practice is commonly challenging (Schlüter et al., 2020). In order to ensure a sustainable ocean, it is, however, necessary to address the problems from all angles. Actions should be ecosystem-based and should be coordinated holistically on a larger scale (Gjerde and Vierros, 2021). While the achievement of the UN SDG 14 has been estimated to be costly (Johansen and Vestvik, 2020), it is unfortunate that some of that largest challenges related to the ocean (such as loss of biodiversity) is to be found in the EEZ of developing states, of which many are highly reliant on the ocean to sustain livelihoods (Techera and Appadoo, 2019). This further emphasizes the importance of having a global and holistically coordinated effort, for which MSP could be a helpful tool.

But while this research supports the relevance of MSP to SDG 14, it also acknowledges that the current practice of MSP rather prioritizes the achievement of economic objectives against environmental goals (although some MSP cases are truly ecosystem-based) (Trouillet, 2020). Indeed, the assessment of MSP definitions showed a minimal attention to sustainability objectives and a high focus on how to manage human uses and potential conflicts, indicating a weak sustainability approach. This economic focus is reflected in the word cloud based on MSP definitions (**Figure 4**), in which the words "uses" and "activities" were the most frequently cited, and the words "ecosystem" and "sustainability" were far less predominant. The different prioritization of environmental and economic objectives in MSP practices is not new, and mirrors the ongoing debate of whether MSP is an abbreviation for "marine" or "maritime" spatial planning. While some use "marine" to indicate that the planning practice is ecosystem-based, and thus limited by ecosystem limits (with strong sustainability objectives), "maritime" is often used in EU contexts (as in the MSPD) or to emphasize the cross-sectoral character of MSP (Mee et al., 2008; Gilbert et al., 2015; Gee and Zaucha, 2019). While the choice of concepts does not in itself guarantee a particular outcome, the values associated with the terminology may play a role when objectives are set, and whether these aim for strong or weak sustainability objectives (Mee et al., 2008). Thus, despite its conceptual relevance to SDG 14, current MSP practices and definitions show that MSP is not yet fulfilling its full potential.

We are currently living in the period of history with the largest deterioration of nature, and the trend is accelerating (Diaz et al., 2019). The latest report from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services estimates that the current rate of species extinction is at least tens to a hundred times higher than it has ever been over the last 10 million years (Diaz et al., 2019). This extensive loss of biodiversity not only reduces ecosystems ability to deliver provisioning services, such as food, but it also decreases ecological resilience to overcome other anthropogenic threats such as climate change (Diaz et al., 2019; Dinerstein et al., 2020). Not only does the ocean provide livelihoods and income for humans, it also supports human wellbeing through non-monetary values, and is in many countries central to both socioeconomic and cultural dimensions (Allison et al., 2020). The current biodiversity loss can lead to various undesirable futures depending on the actions, strategies and plans we make today (Armstrong, 2020; Wyborn et al., 2020). This, together with the increasing need to achieve the UN SDG 14 for life below water emphasize the importance of implementing effective ecosystem-based MSP initiatives, with strong sustainability objectives that prioritize the health and resilience of the ocean above the achievement of blue growth objectives.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

TK carried out initial research design, data collection, analysis, and writing. CFS carried out writing and revision of the draft manuscripts. Both authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.713980/full#supplementary-material

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An Overview of Ocean Climate Change Indicators: Sea Surface Temperature, Ocean Heat Content, Ocean pH, Dissolved Oxygen Concentration, Arctic Sea Ice Extent, Thickness and Volume, Sea Level and Strength of the AMOC (Atlantic Meridional Overturning Circulation)

Carlos Garcia-Soto[1,2], Lijing Cheng[3], Levke Caesar[4,5], S. Schmidtko[6], Elizabeth B. Jewett[7], Alicia Cheripka[7], Ignatius Rigor[8], Ainhoa Caballero[9], Sanae Chiba[10], Jose Carlos Báez[11,12], Tymon Zielinski[13] and John Patrick Abraham[14]*

[1] Spanish National Research Council (CSIC), Santander, Spain, [2] Global Change Unit (BEGIK), Plentzia Marine Station (PiE), University of the Basque Country (UPV/EHU), Plentzia, Spain, [3] International Center for Climate and Environment Sciences, Chinese Academy of Sciences (CAS), Beijing, China, [4] Irish Climate Analysis and Research Unit (ICARUS), Department of Geography, Maynooth University, Maynooth, Ireland, [5] Potsdam Institute for Climate Impact Research (PIK), Postdam, Germany, [6] GEOMAR, Ocean Circulation and Climate Dynamics, Helmholtz Center for Ocean Research, Kiel, Germany, [7] Ocean Acidification Program, National Oceanic and Atmospheric Administration (NOAA), Silver Springs, MD, United States, [8] Polar Science Center, University of Washington, Seattle, WA, United States, [9] Technological Center Expert in Marine and Food Innovation (AZTI), Pasaia, Spain, [10] Japan Agency for Marine-Earth Science and Technology (JAMSTEC), Yokosuka, Japan, [11] Centro Oceanográfico de Málaga, Instituto Español de Oceanografía, Consejo Superior de Investigaciones Científicas (IEO, CSIC), Málaga, Spain, [12] Facultad de Ciencias de la Salud, Instituto Iberoamericano de Desarrollo Sostenible, Universidad Autónoma de Chile, Temuco, Chile, [13] Institute of Oceanology Polish Academy of Sciences (IOPAN), Sopot, Poland, [14] Department of Engineering, University of St. Thomas, Saint Paul, MN, United States

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Sebastian Villasante, University of Santiago de Compostela, Spain

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Aixue Hu, University Corporation for Atmospheric Research (UCAR), United States
Bárbara Cristie Franco, CONICET Centro de Investigaciones del Mar y la Atmósfera (CIMA), Argentina

****Correspondence:***
Carlos Garcia-Soto
carlos.soto@ieo.es

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Global ocean physical and chemical trends are reviewed and updated using seven key ocean climate change indicators: (i) Sea Surface Temperature, (ii) Ocean Heat Content, (iii) Ocean pH, (iv) Dissolved Oxygen concentration (v) Arctic Sea Ice extent, thickness, and volume (vi) Sea Level and (vii) the strength of the Atlantic Meridional Overturning Circulation (AMOC). The globally averaged ocean surface temperature shows a mean warming trend of 0.062 ± 0.013°C per decade over the last 120 years (1900–2019). During the last decade (2010–2019) the rate of ocean surface warming has accelerated to 0.280 ± 0.068°C per decade, 4.5 times higher than the long term mean. Ocean Heat Content in the upper 2,000 m shows a linear warming rate of 0.35 ± 0.08 Wm^{-2} in the period 1955–2019 (65 years). The warming rate during the last decade (2010–2019) is twice (0.70 ± 0.07 Wm^{-2}) the warming rate of the long term record. Each of the last six decades have been warmer than the previous one. Global surface ocean pH has declined on average by approximately 0.1 pH units (from 8.2 to 8.1) since the industrial revolution (1770). By the end of this century (2100) ocean pH is projected to decline additionally by 0.1–0.4 pH units depending on the RCP (Representative Concentration Pathway) and SSP (Shared Socioeconomic Pathways) future scenario. The time of emergence of the pH climate change signal varies from 8 to 15 years for open ocean sites, and 16–41 years for coastal sites. Global dissolved oxygen levels have decreased by 4.8 petamoles or 2% in the last 5 decades, with profound impacts on local and basin scale habitats. Regional trends are varying due to multiple processes impacting dissolved oxygen: solubility change, respiration changes, ocean circulation changes and multidecadal variability. Arctic sea ice extent has been declining by −13.1% per decade in summer (September) and by −2.6% per decade in winter (March) during the last 4 decades (1979–2020). The combined trends of sea ice extent and sea ice thickness indicate that the volume of non-seasonal Arctic Sea Ice has decreased by 75% since 1979. Global mean sea level has increased in the period 1993–2019 (the altimetry era) at a mean rate of 3.15 ± 0.3 mm year^{-1} and is experiencing an acceleration of ∼ 0.084 (0.06–0.10) mm year^{-2}. During the last century (1900–2015; 115y) global mean sea level (GMSL) has rised 19 cm, and near 40% of that GMSL rise has taken place since 1993 (22y). Independent proxies of the evolution of the Atlantic Meridional Overturning Circulation (AMOC) indicate that AMOC is at its weakest for several hundreds of years and has been slowing down during the last century. A final visual summary of key ocean climate change indicators during the recent decades is provided.

Keywords: ocean climate change indicators, sea surface temperature, ocean heat content, ocean pH, dissolved oxygen, Arctic sea ice, sea level, AMOC

INTRODUCTION

Rapid global warming over the past few decades has had consequences for weather, climate, ecosystems, human society and economy (IPCC, 2019). More heat available in the climate system is manifested in the oceans in many ways including increasing the ocean interior temperatures (Johnson et al., 2018; Cheng et al., 2019a), raising the sea level (Nerem et al., 2018), melting the ice sheets and permafrost (Shepherd et al., 2012; Meredith et al., 2019), altering the hydrological cycle (Durack et al., 2012), changing the atmospheric and oceanic circulation (Rahmstorf et al., 2015; Caesar et al., 2018), supporting stronger tropical cyclones with heavier rainfall (Trenberth et al., 2018), among others. Higher ocean heat content and sea surface temperatures invigorate tropical cyclones to make them more intense, bigger and longer lasting, and greatly increase their flooding rains.

In addition to global warming, rising concentrations of carbon dioxide (CO_2) in the atmosphere have a direct effect on the chemistry of the ocean through the absorption of CO_2 by surface waters. The oceans have absorbed about 25% of all CO_2 emissions since the pre-industrial period (Le Quéré et al., 2016; Gruber et al., 2019a,b; Friedlingstein et al., 2020). Increased CO_2 in the water lowers its pH, termed ocean acidification, making it harder for some marine organisms such as corals, oysters and pteropods (Hoegh-Guldberg et al., 2017; Lemasson et al., 2017) to form calcium carbonate shells and skeletons. In some cases, ocean acidification has also been shown to lower fitness in some species such as coccolithophores, crabs, sea urchins and early life stages of fishes (Baumann et al., 2012; Dodd et al., 2015; Campbell et al., 2016; Stiasny et al., 2016; Riebesell et al., 2017; Tasoff and Johnson, 2019). Research efforts over the past decade have built considerable understanding of how marine species, ecosystems, and biogeochemical cycles may be influenced by ocean acidification alone and in concert with other stressors including eutrophication, warming, and hypoxia (Breitburg et al., 2015; Baumann, 2019). Natural variability in carbonate chemistry, such as coastal upwelling and seasonal fluctuations in primary productivity, is also compounded by anthropogenic changes to create particularly extreme ocean acidification conditions in some regions of the global ocean (Feely et al., 2008; Cross et al., 2014).

Oxygen is the basis of life for the vast majority of all oceanic organisms and thus oceanic oxygen levels define habitat boundaries for marine life. Still oxygen can only be gained in the upper most waterlayers by photosynthesis or air sea gas exchange. Once a water mass has left the surface, oxygen is decreasing due to consumption. Global warming does reduce oxygen solubility at the surface, reducing the initial amount of subducted and convected oxygen. Furthermore, upper ocean warming has an impact on biological activity, oceanic stratification and overturning and other processes, which all in turn have the potential to decrease oceanic oxygen levels (Schmidtko et al., 2017; Stramma and Schmidtko, 2021).

Sea ice at the poles plays a critical role in maintaining global heat balance. Shortwave radiation from the sun bears down on the equator, while the global atmospheric and ocean circulations carry this heat to the relatively colder poles. The high albedo of sea ice and the cryosphere allows the global system to more effectively reflect insolation and radiate longwave heat to moderate the global heat balance. The loss of ice in the cryosphere (Meredith et al., 2019) lowers the planetary albedo, allows more heat from the ocean to flux to the atmosphere through the thinner sea ice and the more expansive areas of open water, and reduces Earth's ability to maintain global heat balance. Sea ice also plays a role in the fresh water and salt budget of the global ocean (Polyakov et al., 2020). Salt is expelled in areas of sea ice growth; this ice drifts with the winds and ocean currents transporting fresh water to areas where it may melt during summer. Sea ice has also a significant impact on wildlife, many species depend on the sea ice for habitat, subsistence, and culture (e.g., Meier et al., 2014; Thoman et al., 2020).

Global mean sea level encompasses several processes and climatic systems. Global mean sea level rise is comprised of the change in the sea water volume due to global temperature rise (the thermosteric component) and the change in sea water mass (the barystatic component). The latter is the sum of the melting of ice sheets (Antarctica, Greenland), glaciers and of the input to the sea of terrestrial water storage (e.g., Gregory et al., 2019; Frederikse et al., 2020). Sea level rise poses a significant threat to low lying islands, coasts and communities around the world through inundations, the erosion of coastlines and the contamination of freshwater reserves and foodcrops (Oppenheimer et al., 2019).

The Atlantic Meridional Overturning Circulation (AMOC), a large system of ocean currents in the Atlantic, is an important factor in climate variability and change for several reasons. Changes in AMOC strength can have global impacts on the oceanic carbon sink (Zickfeld et al., 2008; Fontela et al., 2016), the position of the Intertropical Convergence Zone (Timmermann et al., 2007) and, as a consequence Sahel precipitation (Mulitza et al., 2008), the Asian monsoon regions (Fallah et al., 2016), and affect marine ecosystems (Schmittner, 2005). Despite its importance, the evolution of the AMOC since the beginning of the industrial era is poorly known and the question of whether the AMOC has already been weakening in response to global warming remains unknown.

The ocean is currently in a phase of significant climate change and evaluation of the rate of change is of upmost importance. We present here a review of seven key ocean climate change indicators: (i) Sea Surface Temperature, (ii) Ocean Heat Content, (iii) Global Mean Sea level, (iv) Ocean pH, (v) Dissolved oxygen concentration (vi) Arctic Sea Ice extent, thickness, and volume and (vii) the strength of the Atlantic Meridional Overturning Circulation (AMOC). In addition to reviewing the current state of the art, we discuss some research gaps and future developments and present a final visual summary of ocean climate change indicators with emphasis in recent changes (1993–2019/20).

GLOBAL OCEAN WARMING

Sea Surface Temperature

Global Sea Surface Temperature (SST) values are derived from five datasets and are displayed in **Figure 1**. All five data sets show a robust increase of global SST since the late 1800s. The linear trends of SST over the period 1900–2019 for the respective datasets are: 0.060 ± 0.007°C (COBE1[1] data set) (Ishii et al., 2005), 0.062 ± 0.011°C (COBE2 data set) (Hirahara et al., 2014), 0.054 ± 0.007°C (HadISST[2] data set) (Rayner et al., 2003), and 0.073 ± 0.010°C (ERSST[3]) (Huang et al., 2017) per decade. The uncertainty range is 90% confidence interval. The mean SST rate averaged over the four datasets (satellite-based GMPE[4] data is after 1980, so it is excluded here) is 0.062 ± 0.013°C per decade over the same period (1900–2019). The differences between the methods used to fill the data gaps and correct the systematic errors mainly account for their differences between these data products. Since 1980, satellites began to provide high quality and high-resolution observations of SST. The consistency of satellite-based observations (GHRSST Multi-Product Ensemble: GMPE, **Figure 1**) with the *in situ* datasets gives more confidence for the observed ocean warming.

In each dataset, the 10 warmest years on record have all occurred since 1997, with the five warmest years occurring since 2014. The recent decade (2010–2019) shows a much higher rate of warming than the long-term trend: 0.287 ± 0.144 (COBE1), 0.270 ± 0.160 (COBE2), 0.240 ± 0.150 (HadISST), 0.323 ± 0.168°C per decade (ERSST). The mean rate is 0.280 ± 0.068°C per decade within 2010–2019. **Figure 1A** has been provided to graphically illustrate the changes to SST since ∼1850.

By regions, sea surface warming appears in most of the ocean areas (**Figure 1B**), which is an unequivocal signal of human-induced climate change (Bindoff et al., 2013). However, in the North Atlantic Ocean, it shows a long-term cooling trend (called the cold blob or North Atlantic warming hole), which extends from the sea surface to 2,000 m deep. Many studies indicate that this warming hole is a footprint of the AMOC (Caesar et al., 2018).

Ocean Heat Content

Because of the emission of heat-trapping greenhouse gases by human activities, the natural energy flows have been interfered and currently there is an energy imbalance in the Earth's climate system (Hansen et al., 2011; Trenberth et al., 2014; von Schuckmann et al., 2016, 2020). More than 90% of the excess heat is accumulated within the global oceans (Rhein et al., 2013) thus leading to an increase in ocean heat content (OHC). OHC is a fundamental indicator of global warming (Hansen et al., 2011; von Schuckmann et al., 2016; Wijffels et al., 2016; Cheng et al., 2018a; Trenberth et al., 2018). Compared with SST and global mean surface temperature records, the OHC record shows larger signal-to-noise ratio and is less impacted by natural variability (Wijffels et al., 2016; Cheng et al., 2018a,b). Therefore, OHC is better suited to detecting and attributing human influences than other climate records.

The first global OHC time series was provided by Levitus et al. (2000), where a robust long-term 0–3,000 m ocean warming was

[1] COBE (Centennial *in situ* Observation-Based Estimates of sea surface temperature).

[2] HadlSST (Hadley Centre Sea Surface Temperature).

[3] ERSST (Extended Reconstructed Sea Surface Temperature).

[4] GMPE: [GHRSST (Group for High Resolution Sea Surface Temperature) Multi-Product Ensemble dataset).

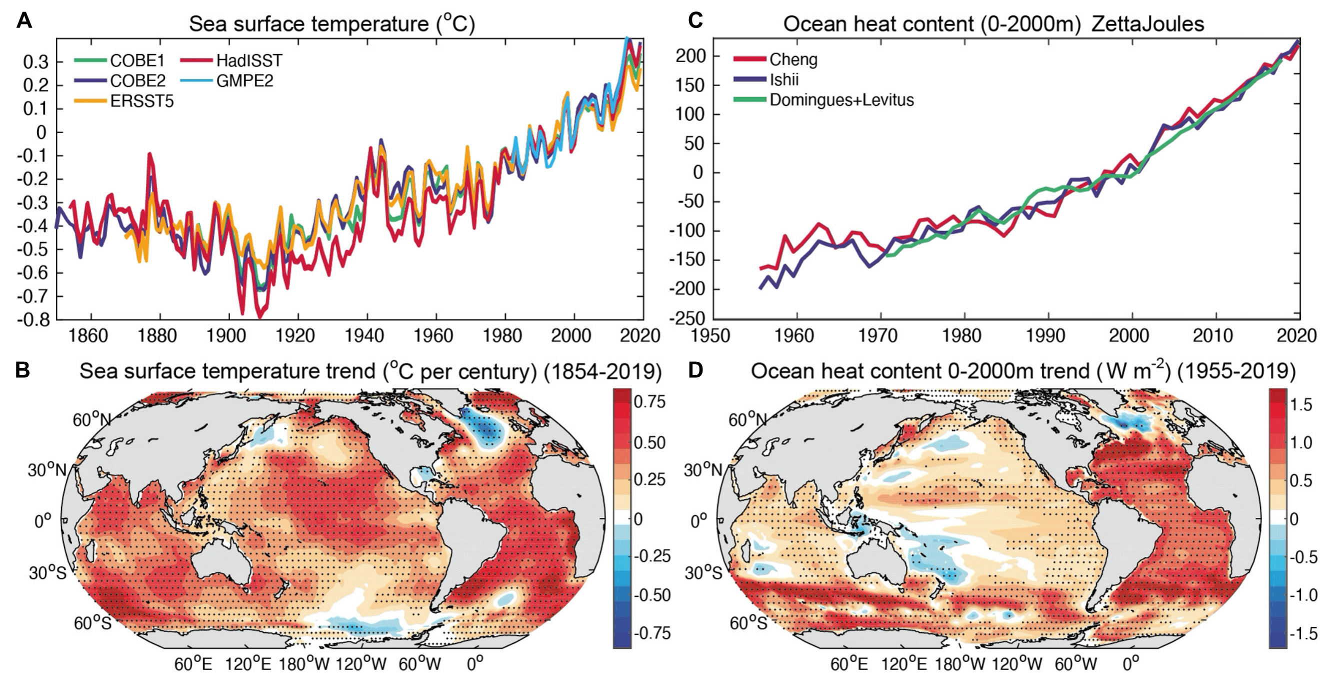

FIGURE 1 | (A) Global mean SST time series from1850 to 2019 (°C). Five datasets are used, including: COBE1, COBE2, ERSST5, HadISST, and GMPE. **(B)** Spatial distribution of the long-term SST trend from 1854 to 2019 for ERSST data (°C per century). **(C)** Global ocean heat content changes for the upper 2,000 m, three time series are provided (Domingues et al., 2008; Levitus et al., 2012; Cheng et al., 2017; Ishii et al., 2017). The Domingues estimate (0–700 m) is combined with the Levitus estimate (700–2,000 m) to produce a 0–2,000 m time series, according to IPCC (Rhein et al., 2013). The energy unit is the ZettaJoule (10^{21} Joules). **(D)** Spatial distribution of long-term ocean heat content trend (0–2000m) (W m^{-2}) for Cheng et al. (2017) data. All data are annual mean time series and use a 1981–2010 baseline. Black dots (in b and d) indicate grid boxes where trends are significant at a 90% confidence interval.

identified over the 1948–1998 period. Since 2000, a number of global and regional OHC data sets have been made available (Willis et al., 2004; Ishii et al., 2005; Palmer et al., 2007; von Schuckmann and Le Traon, 2011; Levitus et al., 2012; Lyman and Johnson, 2013; Desbruyères et al., 2017; Cheng et al., 2017; Zanna et al., 2019). However, the early global OHC time series show significant decadal variability, specifically, a warm period from the 1970s to the early 1980s. This pattern is not reproducible by climate models (Domingues et al., 2008). In 2007, Gouretski and Koltermann (2007) found that the time variation of the systematic errors in expendable bathythermographs (XBT) data is largely responsible for this decadal variation in OHC time series. Since then, scientific community efforts have been aimed to understand XBT errors and improve data quality. Scientific community consensus was developed in 2016 for the best practice of the correction of XBT bias (Cheng et al., 2016). After correcting the systematic errors, the XBT data quality has been improved and the OHC time series show a more homogeneous warming in the half century (Cheng et al., 2018a,b; Goni et al., 2019). In addition to the XBT error, several other sources of uncertainty in OHC estimates have been identified, including MBT biases (Gouretski and Cheng, 2020), mapping methods, and choice of climatology etc. Boyer et al. (2016) found that the major source of error in OHC estimates is the mapping method, which defines how the global map of a variable is created from incomplete observations and how the reconstructed field is smoothed.

It is becoming increasingly clear that many traditional gap-filling strategies introduced a conservative bias toward low-magnitude changes (Durack et al., 2014). To improve how spatial gaps are accounted for in historical ocean temperature measurements Cheng et al. (2017) proposed a new spatial interpolation method. Ishii et al. (2017) suggested a correction to their previous estimate. Based on these developments, we used three less-biased OHC estimates (here "less biased" means the global time series are less biased to the conservative error. This does not indicate that their regional signals are less biased), including Domingues et al. (2008); Cheng et al. (2017), and Ishii et al. (2017).

Estimates show highly consistent ocean warming since the late 1950s. **Figure 1C** provides data on ocean warming (down to a 2,000 m depth). The results reveal a linear warming rate of 0.36 ± 0.06 (Ishii et al., 2017), and 0.34 ± 0.10 (Cheng et al., 2017) Wm^{-2} over the 1955–2019 period (averaged over the Earth's surface), with the mean rate of 0.35 ± 0.08 Wm^{-2}. The new estimates are collectively higher than previous estimates (Rhein et al., 2013) and more consistent with each other (Cheng et al., 2019b). The past 10 years are the ten warmest on record for OHC (Cheng et al., 2019b).

The rate of ocean warming for the upper 2,000 m has increased since the 1990s, with linear trends of 0.59 ± 0.03 Wm^{-2} (Cheng et al., 2017), 0.57 ± 0.06 Wm^{-2} (Ishii et al., 2017), and 0.66 ± 0.02 Wm^{-2} (Domingues et al., 2008; Levitus et al., 2012) over 1990–2019. The mean rate during this period is 0.61 ± 0.05 Wm^{-2}. For the period 2010–2019, the rate of OHC increase is: 0.65 ± 0.07 Wm^{-2}(Cheng et al., 2017), 0.79 ± 0.08 Wm^{-2} (Ishii et al., 2017), and 0.66 ± 0.03 Wm^{-2} (Domingues et al., 2008; Levitus et al., 2012). The mean rate is 0.70 ± 0.07 Wm^{-2}. The latest data show that the upper 2,000 m of the world's oceans continued a trend of breaking records in 2020. Each of the last six decades have been warmer than the previous decade (Cheng et al., 2021).

Increases in OHC are evident throughout the global ocean from the surface down to 2,000 m over the 1960–2019 period (**Figures 1D**, **2**). There are some interesting local patterns for long-term OHC change. There has been stronger warming in the Southern Ocean (70°S∼40°S) and Atlantic Ocean (40°S∼50°N) than other regions and weaker warming throughout the Pacific and Indian Oceans (30°S∼60°N). Models suggest that the Southern Ocean has taken up most of the global warming heat in the past century (Cheng et al., 2017; Swart et al., 2018), driven predominantly by air-sea flux changes associated with upper-ocean overturning circulation and mixing (Swart et al., 2018). Despite of the broad scale 0–2,000 m ocean warming, the subtropical regions in the southwest Pacific and Indian oceans (near the eastern Australia coast and Madagascar), which extends from ∼200 to 1,000 m have displayed a different trajectory. The formation of these cooling signals has not been well understood before.

The Period 1998–2013

A slowdown in the increase of SST and global mean surface temperature has been observed from 1998 to 2013 and led to numerous assertions about a "global warming hiatus" (Hartmann et al., 2013). It has been increasingly clear that this temporal slowdown in surface temperature change is caused by a combination of internal variability, external forcing and the bias in data (Meehl et al., 2011; Kosaka and Xie, 2013; England et al., 2014; Santer et al., 2014; Schmidt et al., 2014;

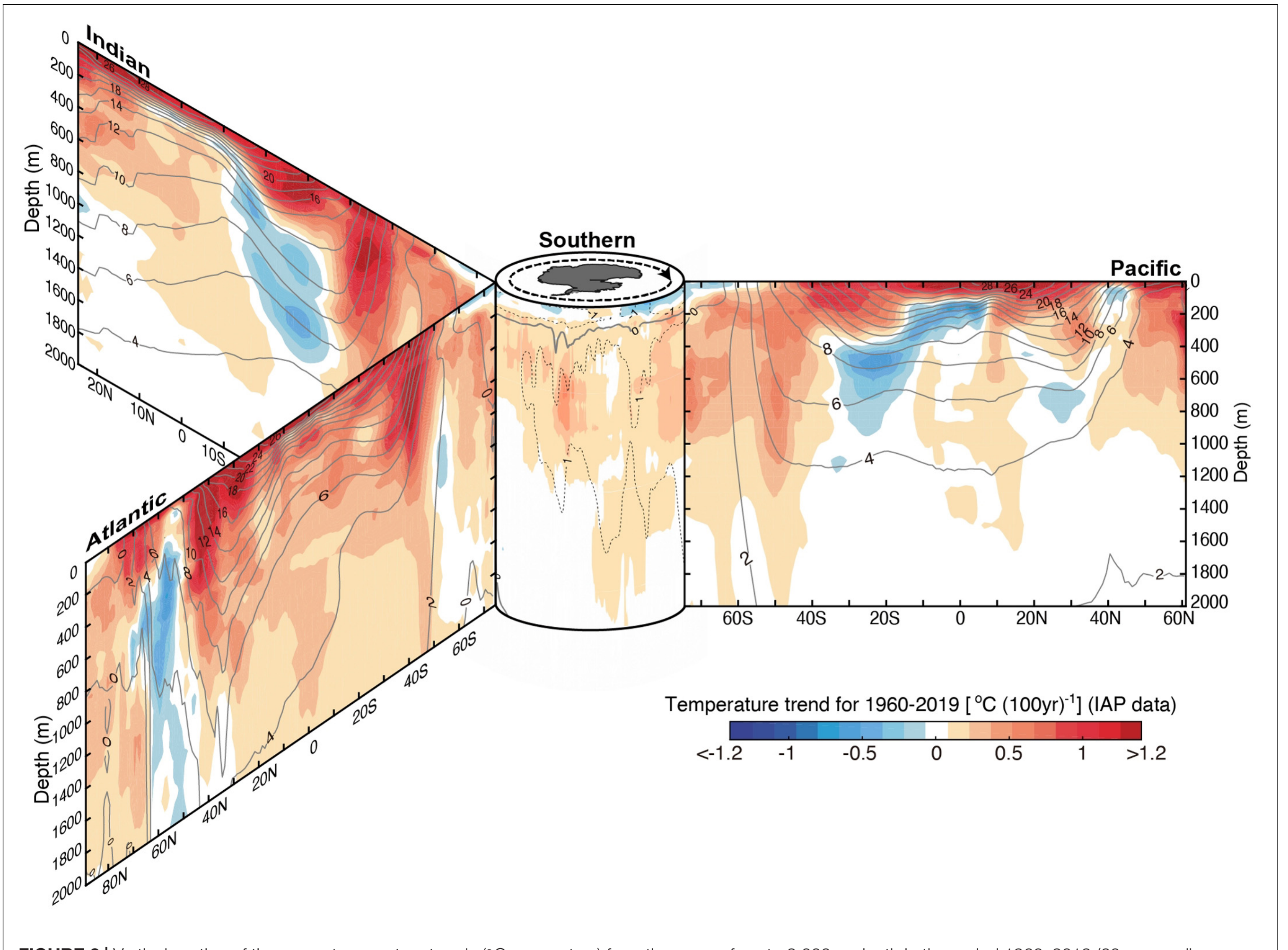

FIGURE 2 | Vertical section of the ocean temperature trends (°C per century) from the sea surface to 2,000 m depth in the period 1960–2019 (60-year ordinary least-squares linear trend). Shown are the zonal mean sections in each ocean basin organized around the Southern Ocean (south of 60°S) in the center. Black contours show the associated climatological mean temperature with intervals of 2°C (in the Southern Ocean, 1°C intervals are provided in dashed contours). Figure from Cheng et al. (2020).

Watanabe et al., 2014; Foster and Abraham, 2015). In particular, there are substantial interannual and decadal scale variability in surface records, which reduces the signal-to-noise ratio of these records. Consequently, a longer time is required to detect a robust trend from surface indicators compared to subsurface or integrated indicators such as OHC and sea level rise. The SST record until 2019 (**Figure 1A**) shows that the linear trend of SST for 1998–2019 is 0.137 ± 0.061°C per decade, greater than the linear trend during the previous decades (1982–1997; 0.100 ± 0.046°C per decade). This range includes the appearance of the extreme 2015/16 El Niño event (Hu and Fedorov, 2017). The rate of OHC increase has been more than doubled since 1990 (**Figure 1C**). The continuous increase in the rate of SST and OHC refute the concept of a slowdown of human-induced global warming (Gleckler et al., 2016; Cheng et al., 2020).

GLOBAL SURFACE OCEAN PH

Ocean acidification is the anthropogenic reduction in the pH of the ocean over an extended period of time, decades to centuries. The ocean has absorbed about 25% of all CO_2 emissions (1870–2015 period; Le Quéré et al., 2018; Gruber et al., 2019a,b; Friedlingstein et al., 2020) and the increased CO_2 in the water is lowering its pH through the formation of carbonic acid (**Figure 3**). Increased aqueous CO_2 is also leading to an increase in bicarbonate and decrease in carbonate ions.

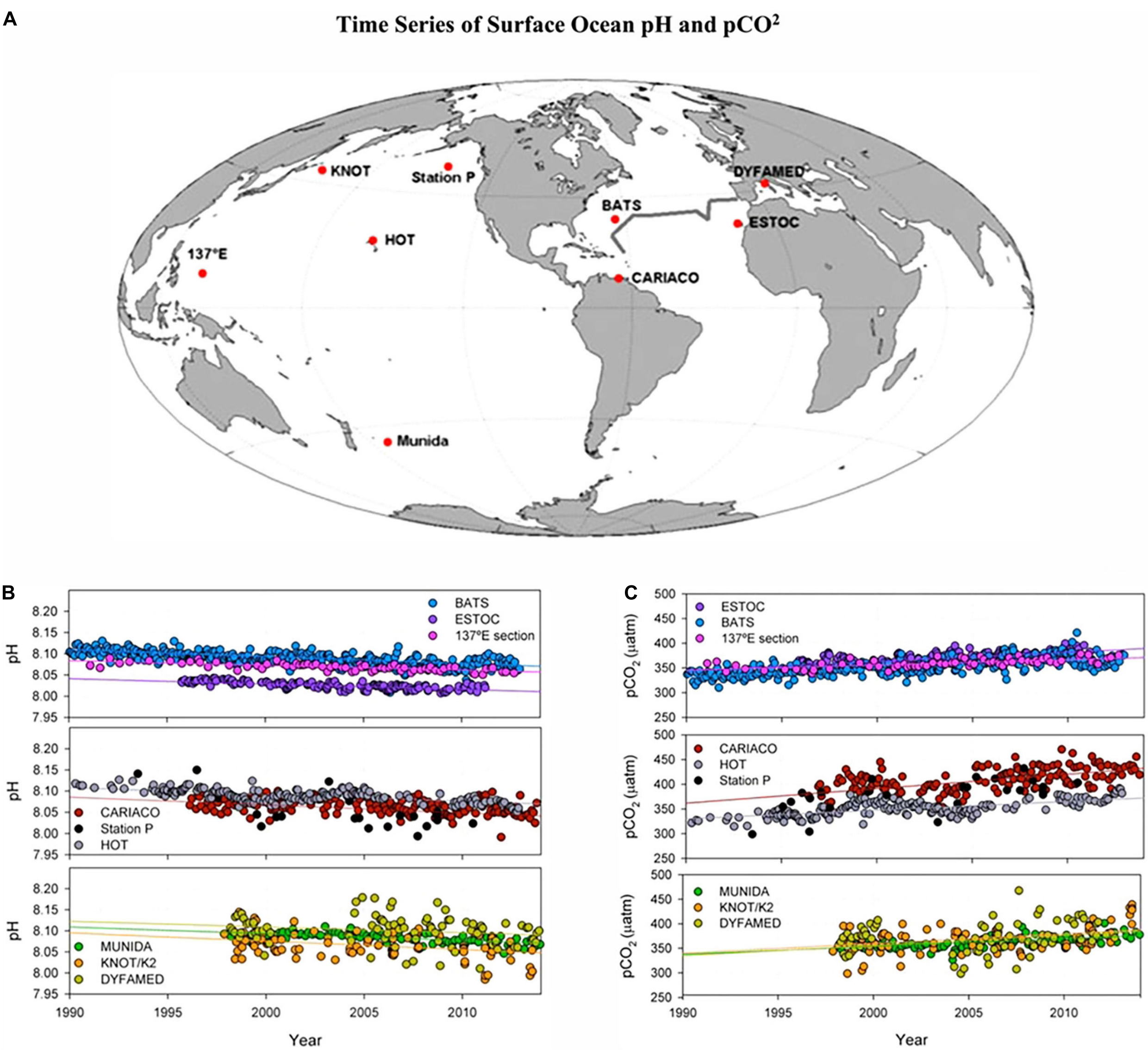

FIGURE 3 | Trends in surface (<50 m) ocean carbonate chemistry calculated from observations obtained at global monitoring stations. **(A)** Positions of time-series stations (red dots). **(B)** Time series of deseasonalized surface ocean pH measurements and trend lines. pH is on total scale for *in-situ* temperatures. **(C)** Time series of deseasonalized surface ocean pCO2 measurements and trend lines. Figure from Tanhua et al. (2015).

Global surface ocean pH has declined on average by approximately 0.1 (from 8.2 to 8.1) since the Industrial Revolution (Caldeira and Wickett, 2003; Orr et al., 2005). Jiang et al. (2019) reports a similar global decrease of −0.11 ± 0.03 pH units from 1770 to 2000. The Arctic Ocean has experienced the largest pH decrease with a pH decline of −0.16 ± 0.04 pH units (1,770–2,000). There is natural variability of the ocean's carbonate chemistry driven by a number of natural processes such as circulation, air-sea interchange, and remineralization for example, but carbonate chemistry at global scale is being driven by the increasing carbon dioxide in the atmosphere coming from emissions and land use change. The current changes can be observed in extended ocean time series and the rate of change is likely unparalleled in at least the past 66 million years (Hönisch et al., 2012; Zeebe et al., 2016). Ocean pH is projected to decline, approximately, by an additional 0.1–0.4 pH units by the end of century (2,100) depending on the future RCP (Representative Concentration Pathway) / SSP (Shared Socioeconomic Pathways) scenarios (Caldeira and Wickett, 2003; Feely et al., 2009; Jiang et al., 2019; Kwiatkowski et al., 2020).

Carbonate chemistry varies according to large-scale oceanic features including depth, distance from continents due to land influence, upwelling regime, freshwater/nutrient input and latitude (Jewett and Romanou, 2017). Due to this variability, as determined by these various characteristics, only longer term, observational time series can detect the predicted long-term increase in acidity at individual sites due to rising atmospheric CO_2 levels. Time of emergence of the signal varies from 8 to 15 years for open ocean sites, and 16 to 41 years for coastal sites (Sutton et al., 2019), making it necessary to commit to long-term observational records, especially in the coastal zone where most commercially and culturally important marine resources reside.

DISSOLVED OXYGEN CONCENTRATION

The ocean can only gain oxygen at the surface by air sea gas exchange and photosynthesis. Subsurface dissolved oceanic oxygen (DO) is advected along water mass distribution pathways and mixed into adjacent water masses while being consumed by respiration. Therefore, any change in solubility at the surface (due to warming), decrease of ventilation (due to stratification increase) and increase in deep ocean respiration (due to increased surface primary production and enhanced particle flux) can lead to oceanic deoxygenation. Thus, changes in deep ocean DO can be seen as an integrative long-term indicator for profound changes in physical or biogeochemical ocean dynamics.

Since oxygen is the marine biogeochemical parameter with little to no variations in analysis methods over time, long-term oceanic DO changes can be derived robustly with relatively high confidence (Carpenter, 1965; Wilcock et al., 1981; Knapp et al., 1991). Only limited data availability may compromise robust trends in all regions and depths. Winkler titration of water samples, established in 1903, has become the method of choice for DO measurements soon after its discovery and has since been used to calibrate DO measurements of all kind of platforms. More recent developments of computer-aided Winkler titration methods that provide higher accuracy, seem not to bias historical measurements (Schmidtko et al., 2017). Furthermore, a systematic relative bias due to reagent changes in the analysis was tested and determined as highly unlikely (Schmidtko et al., 2017). Therefore, any calibrated long-term DO observation can be used to derive long term trends and multi-decadal variability of timeseries spanning a century.

Coastal changes in DO are impacted on a very local scale by regional physical, biogeochemical and anthropogenic changes. These regional changes range from riverine run-off of nutrients, deposits of organic matter over heatwaves and tides, just to name a few. An observational study (Diaz and Rosenberg, 2008) reports an increased occurrence of coastal dead zones, with consequences for regional ecology and economy. While the occurrence of most of these coastal dead zones is locally driven, some low-oxygen events may have been affected by open ocean deoxygenation, making these events more likely by lowering background DO levels.

In the open ocean most long-term time series data from monitoring stations show decreasing DO levels despite temporal variations on annual to multidecadal time-cales (e.g., Keeling et al., 2010). Time stations with long-term increasing oxygen levels exist but are sporadic.

The long term monitoring stations support the findings of three major studies of global DO changes, covering the time period from the sixties to today (Helm et al., 2011; Ito et al., 2017; Schmidtko et al., 2017). **Figure 4** shows an overview of the relevant changes. Despite diverging methods all studies agree that the global ocean is losing oxygen at a significant rate. The rate of decrease does vary over depth and method but is on the order of 2% over 50 years (Schmidtko et al., 2017). This accumulates to a loss of 4.8 ± 2.1 petamoles since 1960. This loss is not homogeneously distributed in the global ocean. Oceanic oxygen loss varies with depth and region, resembling the several processes involved in oxygen distribution and consumption. All the works generally agree on the large scale deoxygenation patterns with most pronounced deoxygenation in the north Pacific and Southern Oceans with some smaller disagreement regarding the intensity of deoxygenation in the tropical oceans.

From a global perspective, temperature-driven solubility decrease is dominating the oxygen loss in the upper most water layers. A warming ocean is gaining less oxygen from air sea gas exchange. Schmidtko et al. (2017) attribute 50% of the oxygen loss in the upper 1,000 m to solubility changes. This number drops to about 25% for the upper 2,000 m and only on the order of 13% of the overall full water column oxygen loss. This solubility driven deoxygenation is attributed to the time period 1960–2010, assuming linear warming. For an accelerating warming process these numbers will likely change. Since solubility is responsible for only part of the observed changes, other processes are similarly important. Nevertheless, we cannot disregard temperature as the key source of those changes as well, since processes other than solubility change are also largely driven by a warming upper ocean. These temperature driven processes are not

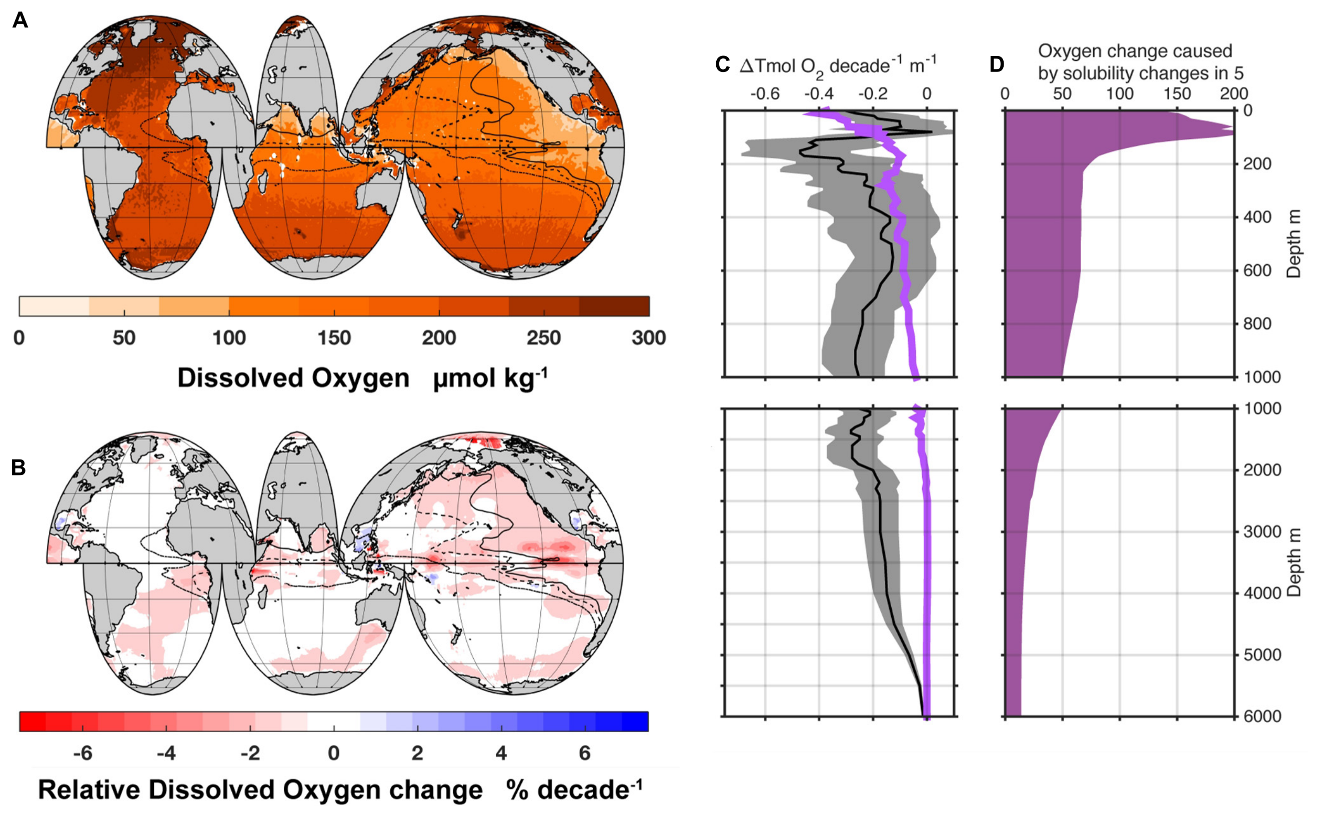

FIGURE 4 | Spatial and temporal trends of dissolved oxygen concentration. **(A)** Mean oxygen concentration, lines indicate the maximum extent of the oxygen minimum zone with 40, 80, and 120 μmol kg^{-1} dissolved oxygen anywhere in the water column. **(B)** Trend of oxygen over five decades 1960–2010 **(A,B)** data from Schmidtko et al., 2017. **(C)** Vertical distribution of deoxygenation (black curve) and error (gray area), the purple line indicates the loss expected from oceanic warming detected in the same data set. **(D)** Percentage of deoxygenation due to warming for the water column, values above 100% indicate that processes counter acting solubility deoxygenation are at play **(C,D)** (reworked from Schmidtko et al., 2017).

limited to, but do include stratification increase, circulation changes and thermal impacts on biogeochemical cycles (e.g., Keeling and Garcia, 2002; Stendardo and Gruber, 2012; Bianchi et al., 2013).

More recent analysis of available regional studies (Stramma and Schmidtko, 2021) attribute various processes to observed regional changes (**Figure 5**). While solubility and stratification dominate high latitudes and the Atlantic Ocean, multi-decadal variability is dominant throughout most basins. The reason for this can be seen in atmosphere-ocean indices like the North Atlantic Oscillation (NAO), El Nino-Southern-Oscillation (ENSO), and Pacific Decadal Oscillation (PDO), among many others, impacting regional ocean dynamics with subsequent influences on dissolved oceanic oxygen. Biological and nutrient stimulation causes are mainly found near the coast and in particular upwelling regions. Source water changes and circulation driven changes point to physical parameters that have shifted in ocean dynamics. Many of these processes are linked to changing ocean ventilation and respiration and are therefore challenging to appraise directly. Still, all tend to reinforce the impacts from warming (Oschlies et al., 2018).

Along with globally decreasing oceanic DO, the volume of so-called oxygen minimum zones has grown significantly. Oxygen minimum zones (OMZ) are generally defined as oceanic volume with less than 80 mmol l^{-1} DO, and thus not suited as habitat for many marine organisms that rely on continuous respiration although they may provide refuge for animals that can cope with low DO conditions. In areas where the OMZ DO levels are close to or completely depleted oxygen minimum zones have potential impacts on greenhouse gas driven climate warming, since they can emit large quantities of nitrous oxide, a potent greenhouse gas, owing to denitrification processes under anoxic conditions (e.g., Codispoti, 2010; Santoro et al., 2011). Such low OMZ DO levels can be found in the Pacific and Indian Ocean and have been found expanding.

With impacts on dissolved oxygen levels varying strongly on regional scales, predictions on future local oxygen can only be established knowing all local boundary conditions and predicted changes. At basin scale or even global scale, it can be stated that an increased warming of the upper ocean has impact on oxygen levels by solubility while simultaneously reinforcing other processes that are linked to ocean dynamics and

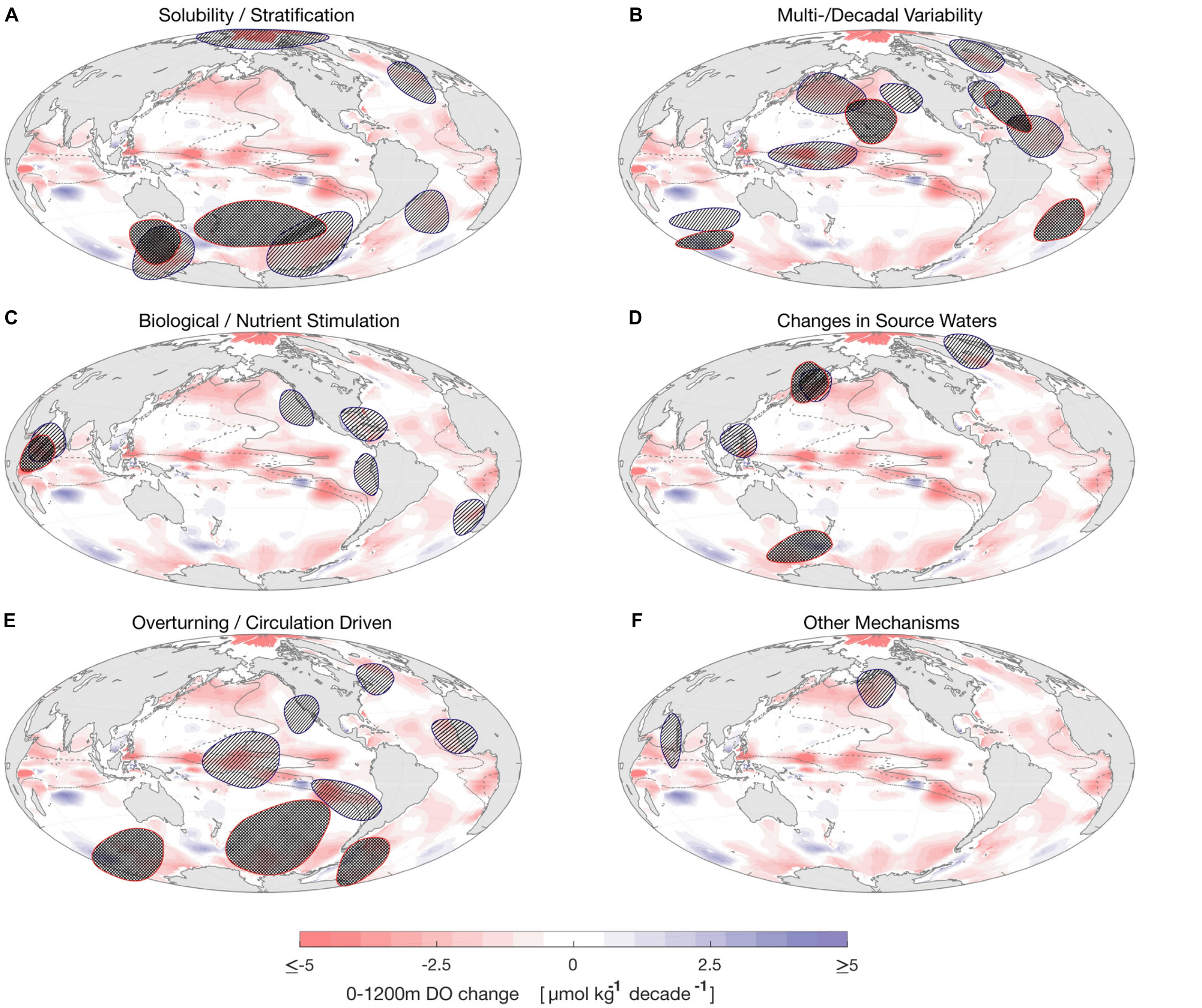

FIGURE 5 | Major drivers of regional oxygen variability marked as approximate areas as described in peer-reviewed articles from observations and model analyses (see details in Stramma and Schmidtko, 2021). Background colors indicate upper ocean oxygen changes in μmol kg^{-1} decade^{-1}. The areas with significant variability from **(A)** solubility and/or stratification, **(B)** multi-decadal or decadal variability, **(C)** biological and/or nutrients stimulation, **(D)** changes in source waters, **(E)** overturning and/or circulation driven, and **(F)** other mechanisms. The diagonal-hatched regions indicate upper ocean impact, the cross-hatched regions indicate deep ocean variability. (reworked after Stramma and Schmidtko, 2021).

biogeochemistry, and that are responsible for the majority of the observed deoxygenation.

ARCTIC SEA ICE EXTENT, THICKNESS AND VOLUME

The retreat of Arctic sea ice has been one of the most iconic indicators of climate change (Thoman et al., 2020). Arctic sea ice extent is declining by −13.1% per decade during summer (September 1979–2020), when it exhibits its seasonal minimum extent, and by −2.6% per decade during winter (March 1979–2018) (Fetterer et al., 2017; Perovich et al., 2020). **Figures 6A,B** present the mean sea ice concentration in summer and winter during the last 4 decades and **Figures 6C,D** the sea ice concentration trends during the same period and seasons. The decreasing trends during winter (**Figure 6C**) are observed in all the peripheral seas around the Arctic, with the greatest decreasing trends (−26% per decade) occurring in the Barents Sea. During summer the trends (**Figure 6D**) are almost twice as high in the Pacific and Asian sectors of the Arctic Ocean compared to the Atlantic Sector, with the greatest decreasing trends occurring in the Beaufort Sea (−29% per decade), which has been essentially ice free during summer over the past decade. The record minimum in summer sea ice extent was measured in September 2012, and the second lowest extent in the 42 years satellite record was measured in September 2020 (Perovich et al., 2020).

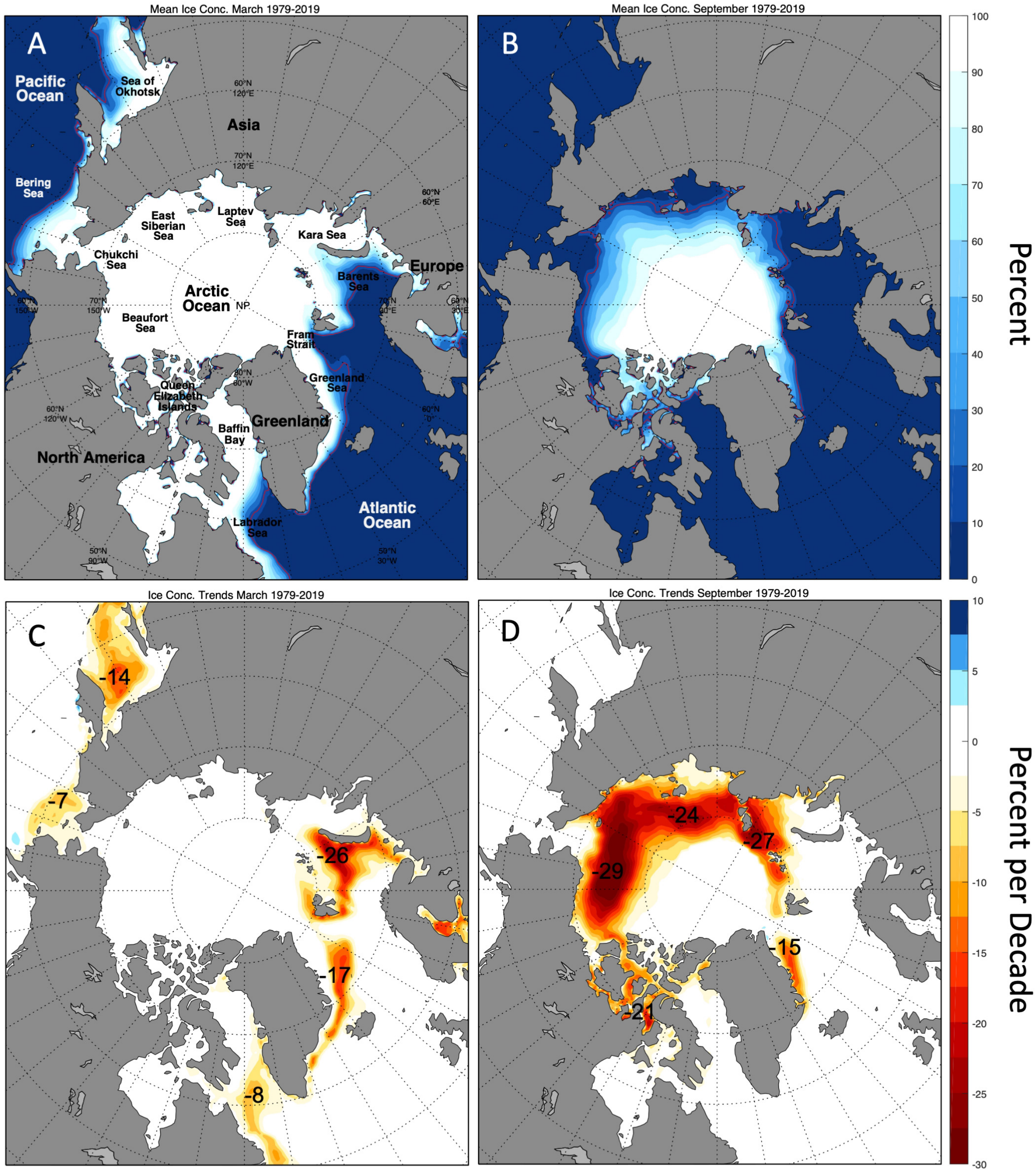

FIGURE 6 | Arctic Mean Sea Ice Concentration (percent) during the last 4 decades (1979–2019) in winter (March; **A**) and summer (September; **B**); and Arctic Sea Ice Concentration Trends (percent per decade) in winter (March; **C**) and summer (September; **D**) during the same 40 year period (1979–2019) estimated from Bootstrap Sea Ice Concentration analysis (https://nsidc.org/data/nsidc-0079/versions/3; Comiso, 2017).

Similarly, the thickness of Arctic sea ice has also decreased. In one of the first studies to document this change, using measurements from upward looking sonars on submarines, Rothrock et al. (1999) showed that the average of thickness of sea ice decreased from 3.1 meters in 1958–1976, to just 1.8 meters in 1993–1997, with the largest decreases occurring in the central and eastern Arctic. In an updated study which includes estimates of sea ice thickness from satellites, Kwok (2018) showed that the thickness has now decreased by 2.0 meters, comparing 2011–2018 ICESat and CryoSat-2 data to the 1958–1976 and 1993–1997 submarine cruise measurements, or about 66% over the six decades.

Taken together, the observed trends in sea ice extent and sea ice thickness indicate that the volume of Arctic sea ice has decreased by over 75% since 1979. This estimate is coincident with many modeling studies, including the Pan-Arctic Ice Ocean Modeling and Assimilation System (Zhang and Rothrock, 2003; Schweiger et al., 2011, 2019) which estimates that the average volume of Arctic sea ice of 11.5×10^3 km^3 in September, 1979–2010, has decreased with a rate of -2.8×10^3 km^3 per decade. The current record minimum in total ice volume estimate using PIOMAS is 3.8×10^3 km^3 set in September 2012. The summers of 2019 and 2020 are tied for second minimum with 4.2×10^3 km^3.[5]

These trends in the decline of Arctic sea ice are the result of the complex interplay between the atmosphere, sea ice and the ocean. During the winter, the cold halocline layer protects sea ice from the underlying warm Atlantic water (e.g., Steele and Boyd, 1998), allowing sea ice to grow thermodynamically driven by air temperatures which historically were around −32°C (Rigor et al., 2000). Winds and ocean currents may also change the thickness of sea ice dynamically by ridging and rafting of sea ice during storms or against a coastline. This process also creates areas of open water which would rapidly freeze over and thicken during winter, quickly increasing the thickness distribution of sea ice. Most of the sea ice in the Arctic Ocean is exported through Fram Strait, and later melts in the warmer waters of the Greenland Sea and North Atlantic. Heat from the atmosphere also melts sea ice on the Arctic Ocean during summer.

The global trends in air temperature are more dramatic in the Arctic due to the ice-albedo feedback and Polar Amplification of global warming (e.g., Manabe and Stouffer, 1980). As warmer temperatures melt the ice, the lower albedo of the surface allows more heat to be absorbed by the surface leading to a positive feedback that amplifies the global warming signal. Changes in wind related to the Arctic Oscillation (AO, Thompson and Wallace, 1998) have also been linked to the decline of sea ice. For example, Rigor et al. (2002) have shown that during high AO winters the winds blow more sea ice away from the Eurasian coast which allows more heat from the ocean to warm the atmosphere over these areas. Rigor and Wallace (2004) found that the trends in sea ice extent during summer (e.g., **Figure 6D**) are a lagged response, specifically, the younger, thinner sea ice that develops along the Eurasian coast during high AO winters, is less likely to survive the summer melt. The younger, thinner sea ice pack is also blown faster by the winds and is more prone to fracturing and increasing amounts of open water and leads during all seasons, allowing more heat to be released by the ocean during winter, and more heat to be absorbed by the ocean during summer, which will delay freeze up. Thus the increased kinematics of sea ice provides a dynamic complexity that strengthens the ice-albedo feedback even more (Rampal et al., 2011).

The high Arctic Oscillation conditions may have also shifted the ocean currents so that river runoff from the Eurasian continent was diverted to the east, weakening the cold halocline later and allowing heat in the Atlantic waters to reach the surface and melt sea ice (Morison et al., 2012, 2021). Polyakov et al. (2020) show that the weakening of the cold halocline layer is also observed in in the Eurasian Basin through 2018 and estimate that the Atlantic waters heat increased to over 10 Wm^{-2} in 2016–18 (from 3–4 Wm^{-2} in 2007–2008), decreasing winter ice growth by twofold.

GLOBAL MEAN SEA LEVEL

The IPCC Special Report on Oceans and Cryosphere (SROC, Oppenheimer et al., 2019) concluded that the rate of change of the Global Mean Sea Level (GMSL) was, respectively, 1.4 and 3.2 mm year^{-1} for the periods 1901–1990 and 1993–2015. Several studies have advanced the analysis of the trend of GMSL by improving the accuracy of the observational data adding observations by different platforms, using different analysis methods, reconstructing time series, or using climate models to simulate the sea level evolution. By means of a novel hybrid sea-level reconstruction applied to the global tide gauge time series Dangendorf et al. (2019) estimated a GMSL trend of 1.6 mm year^{-1} from 1900 to 2015, an increase of 0.19 m since 1900. From the analysis of altimetry for the period 1993–2017 Nerem et al. (2018) observed a GMSL rise of 2.9 ± 0.4 mm year^{-1}and an acceleration of 0.084 ± 0.025 mm year^{-2}. Dangendorf et al. (2019) also found a notable increase in the GMSL rate from 1993 (2.1 mm year^{-1}) to 2015 (3.4 mm year^{-1}) and a persistent acceleration of 0.06 mm year^{-2} since the 1960s. Cazenave et al. (2019) using the most recent time series (January 1993–February 2019) gives a mean rate of sea level rise of 3.15 ± 0.3 mm year^{-1}, with an acceleration of 0.10 ± 0.04 mm year^{-2}. This acceleration value agrees well with Nerem et al. (2018)'s estimate. GMSL is projected to rise by the end of the century (2100) between 0.43 m (0.29–0.59 m) under RCP2.6 and 0.84 m (0.61–1.10 m) under RCP8.5 and will continue to increase during centuries due to ocean heat uptake and the melting of the ice sheets and glaciers (Oppenheimer et al., 2019).

Sea level trend maps computed from satellite altimetry data reveal that even though a general GMSL rise has occurred during the altimetry era, the MSL change follows regional and local variability with diferences up to 8 mm year^{-1} reflecting the pattern of ocean currents, the large-scale oceanic and atmospheric oscillations or the contribution of the melting ice-sheets among other factors (**Figure 7**). Basin-scale MSL trend variability is also observed from the longer tide gauges series (e.g., Slangen et al., 2017; Dangendorf et al., 2019). For the period 1900–2018, Frederikse et al. (2020) found a positive MSL trend in all the ocean basins, being the highest for the subtropical North Atlantic (2.49 mm year^{-1}) and south Atlantic (2.07 mm year^{-1}), the lowest for the subpolar North Atlantic (1.08 mm year^{-1}) and East Pacific (1.20 mm year^{-1}), while intermediate values were observed in the Indian-south Pacific (1.33 mm year^{-1}) and in the northwest Pacific (1.68 mm year^{-1}).

Separating the global drivers of the MSL variability at a regional scale and even more at a local scale, becomes complex. In addition to the anthropogenically forced sea-level signal, the internal variability is time and location-dependant (Stammer et al., 2013). Thus, large atmospheric and ocean oscillations (e.g., El Niño Southern Oscillation, Pacific Decadal Oscillation, North

[5]http://psc.apl.uw.edu/research/projects/arctic-sea-ice-volume-anomaly/

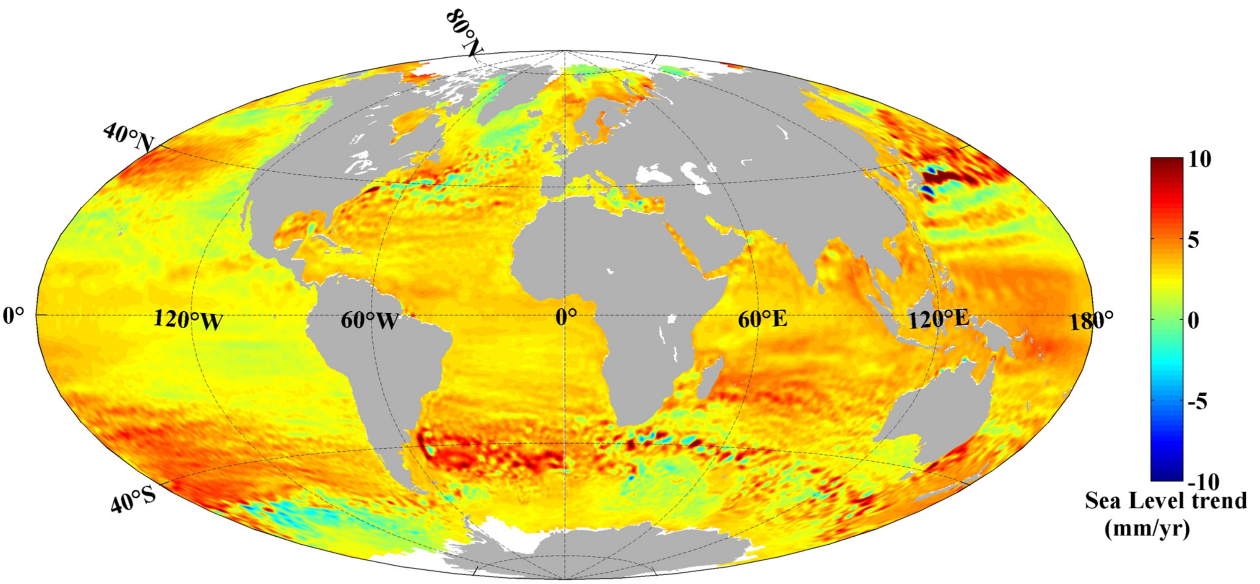

FIGURE 7 | Map of sea level trends (mm yr^{-1}) elaborated using 27 years of satellite multi-mission altimetry data (January 1993–October 2019). Credits EU Copernicus Marine Service (CMS).

Atlantic Oscillation, Indian Ocean Dipole) have interannual and decadal signals in MSL time series being of different period and intensity depending on the ocean basin. Analyzing the MSL trend from a regional perspective additionally allows development of a more detailed characterization of the global sea level budget. For instance, from a hybrid MSL reconstruction from 1900 to 2015, Dangendorf et al. (2019) demonstrate that a great part (∼76%) of the GMSL acceleration from the 1960s has its origin in the Indo-Pacific (0.07 ± 0.01 mm $year^{-2}$) and South Atlantic sea (0.06 ± 0.01 mm $year^{-2}$) as a consequence of an intensification and a displacement of the southern hemispheric westerlies that contributed to increased heat uptake and consequently a more intense thermal expansion.

With regard to local MSL trends, besides the complex processes occurring near the coast (e.g., Benveniste et al., 2019), and the particular importance of land subsidence, the different scale processes contributing to the sea level variability has not only a local origin, but it can be generated far away (Woodworth et al., 2019). The IPCC Special Report on Oceans and Cryosphere (Oppenheimer et al., 2019) further indicates that the local Extreme Sea Level (ESL) events happening once in one hundred years will become annual events in many low-lying cities and small islands by 2050 under all RCP scenarios due to the projected Global Mean Sea Level Rise (GMSL).

THE STRENGTH OF THE ATLANTIC MERIDIONAL OVERTURNING CIRCULATION (AMOC)

Direct continuous measurements of the AMOC only started in 2004 with RAPID-MOCHA (Smeed et al., 2014), an array of moored instruments that spans the width of the Atlantic at latitude 26.5 degrees north and provides continuous monitoring of the AMOC. Before, there had only been five individual snapshots of the AMOC, computed from seawater density measurements taken at hydrographic sections in the years 1957, 1981, 1992, 1998, and 2004 (Frajka-Williams et al., 2019). A couple of other trans-basin observing arrays at different locations in the Atlantic followed, including SAMBA in the South Atlantic in 2009 (Meinen et al., 2018) and OSNAP in the subpolar North Atlantic in 2014 (Li et al., 2017).

The RAPID observations recorded a notable decrease of 2.7 Sverdrup (Sv; 1 Sv = 10^6 m^3 s^{-1}), about 15%, in AMOC strength between April 2004 until roughly April 2008, followed by a fairly stable period until the end of the recovered data in September 2018 (Smeed et al., 2018; Moat et al., 2020). Yet such a short record cannot distinguish between decadal variability and long-term slowdown. Various studies have attempted to reconstruct the AMOC for the time period before 2004 using other climatic variables, so-called proxies, like sea surface temperatures (Latif et al., 2006; Rahmstorf et al., 2015) and sea level heights (Frajka-Williams, 2015) as well as the available snapshots (Bryden et al., 2005; Kanzow et al., 2010). Using the latter Bryden et al. (2005) estimated a decrease in AMOC transport at 26°N of about 30% between 1957 and 2004. Two main criticisms were levied against this conclusion. One, the first 4 years of the observed overturning strength provided by the RAPID data (Kanzow et al., 2010; Smeed et al., 2014) suggested that the seasonal variability of the AMOC has an amplitude of several Sverdrup, significantly larger than previously thought, thus, the five snapshots used by Bryden et al. (2005) might have sub-sampled intense high-frequency variability, rather than a robust trend. Correcting the measurements for the seasonal cycle Kanzow et al. (2010) found a much smaller weakening of only 13%. A different approach, estimating the strength of the AMOC mainly from the more widely available measurements of CTD (Conductivity-Temperature-Depth) end stations, found a reduction of about 2–4 Sv between 1980 and 2005, but concluded that this trend cannot be statistically validated due to the large variability in the layer transports found in the data (Longworth et al., 2011). This is in direct opposition to the findings of Latif et al. (2006) who concluded from the observed linear trend in the sea surface temperatures in the North Atlantic that the AMOC has strengthened since 1980. Combining the observed density change in the region of the Denmark Straight with the results from ocean model simulations they estimated the increase to be about 1 Sv between 1970 and 2000. These seemingly contradictory results could be reconciled with the AMOC reconstruction by Caesar et al. (2018), based on the relative cooling in the subpolar North Atlantic, who see a decline of AMOC strength since the 1950s with a short-lived recovery that is evident in the 1980s and 90s before a return to decline from the mid-2000s (**Figure 8**). This short-lived recovery of the AMOC is also found by Jackson et al. (2016) by analyzing a global-ocean reanalysis product, the GloSea5 data, which covers the years 1989–2015 as well as by Frajka-Williams (2015) who combined sea surface height data from satellites with cable measurements to reconstruct the AMOC from 1993 to 2014. Caesar et al. (2018), found an overall decline of the AMOC of about 3 ± 1 Sv (about 15%) since the mid-twentieth century.

To put these changes into an even longer-term context, researchers rely on different paleo-proxies, including data from ice and marine sediment cores, to reconstruct the strength of the AMOC over the last more than 1,000 years. Using grains from cores of sediments from a key site off Cape Hatteras Thornalley et al. (2018) found that the AMOC is now at its weakest in at least 1,600 years. They confirmed this finding with

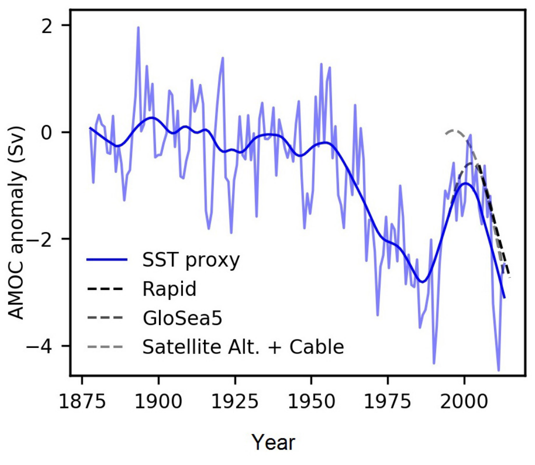

FIGURE 8 | Trend of the strength of the overturning circulation (AMOC) in observations. Shown are (i) the long-term (20-year LOWESS filtering, thin line are annual values) sea surface temperature proxy (blue), (ii) the quadratic trend of an ocean reanalysis product (GloSea5; Jackson et al., 2016), (iii) a reconstruction from satellite altimetry and cable measurements (Frajka-Williams, 2015) and (iv) the linear trend of *in situ* AMOC monitoring by the RAPID project. Figure from Caesar et al. (2018).

foraminiferal-based temperature proxies which, when taken from specific locations in the North Atlantic, reflect the strength of the North Atlantic sea surface temperature dipole which has been repeatedly linked to AMOC changes (Thornalley et al., 2018). Similar conclusions were reached by Rahmstorf et al. (2015) who used a proxy compilation of tree-rings and ice cores that represent the relative temperature changes in the subpolar North Atlantic caused by AMOC changes. Sherwood et al. (2011) studied the δ^{15}N concentration of deep-sea gorgonian corals and found a nutrient shift in the early 1970s that is unique in the context of the last approximately 1,800 years and indicates a decline in the presence of Labrador Slope Water associated with the AMOC. Thibodeau et al. (2018) found a similar decline in an AMOC record based on the δ^{18}O in benthic foraminifera from sediment cores retrieved from the Laurentian Channel. Caesar et al. (2021) compared all these different proxy types and found that they provide a consistent picture of the evolution of the AMOC since AD 400 with a long and fairly stable period (that is intermitted with an initial decline during the nineteenth century) followed by another, more rapid weakening in the middle of the twentieth century. Together, these proxies indicate that the AMOC over the last decades is weaker than ever before in the last 1,600 years. **Figure 9** shows collectively these observations since 1400 to the present.

Currently, while these findings provide strong evidence that the Atlantic Meridional Overturning Circulation has weakened relative to preindustrial times, there is insufficient data to quantify the exact magnitude of the weakening, or to properly attribute it to anthropogenic forcing (IPCC, 2019). This is also due to the fact that the ensemble means of the latest generation of climate models (CMIP6) show no trend in the strength of the overturning circulation over the historical period (Weijer et al., 2020). However, this might be due to an overestimation of the anthropogenic aerosol forcing in a majority of the models leading to a cooling of the subpolar North Atlantic and a subsequent AMOC strengthening. This is supported by the fact that those ensemble members that capture the North Atlantic cold blob, i.e., the SST fingerprint of a weaker AMOC, show a weakening of the AMOC over the historical period (Menary et al., 2020). For the future, the simulations of all CMIP6 models respond to the increasing greenhouse gas emissions with an AMOC weakening, showing on average a decline of 24–39%, depending on the emission scenario, over the course of the twenty-first century. When using the observed strength of the AMOC by RAPID/MOCHA as a constraint the mean decline increases, in particular for the low-emission scenarios, to 34–45% (Weijer et al., 2020).

ACTIONS FOR BETTER CLIMATE CHANGE MONITORING

The international ocean observing community has made persistent efforts in developing new technologies, observation networks and data sharing protocols to deliver credible climate change indicators and useful ocean information to a variety of users in a timely manner and at a global scale. Here we identify

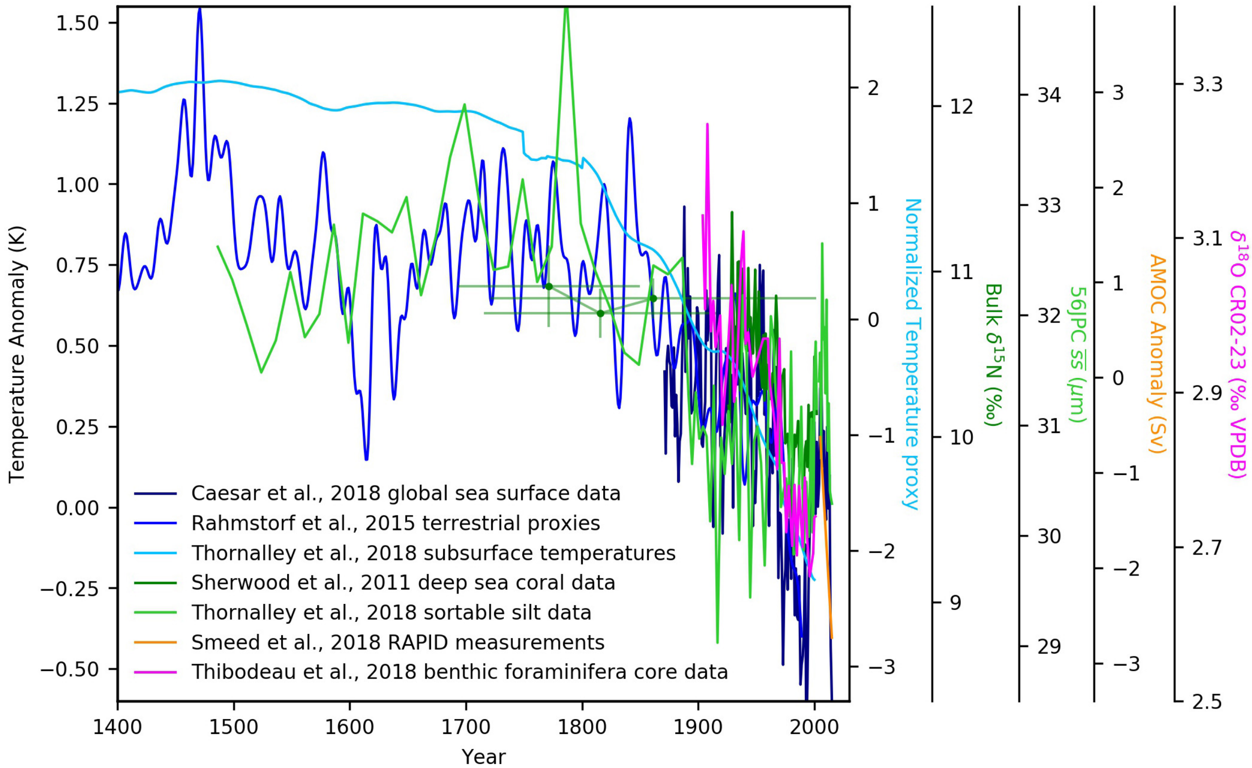

FIGURE 9 | Trend of the strength of the overturning circulation (AMOC) in observations since 1400 from various proxies. Shown are (i) the long-term evolution of the sea surface and land temperatures in the North Atlantic region (different shades of blue; Rahmstorf et al., 2015; Caesar et al., 2018; Thornalley et al., 2018), (ii) the long-term evolution of the ocean heat content of the Atlantic (red, Zanna et al., 2019), (iii) the long-term evolution using data from deep sea cores [light green (Thornalley et al., 2018), dark green (Sherwood et al., 2011), and magenta (Thibodeau et al., 2018)] and (iv) the linear trend of *in situ* AMOC monitoring by the RAPID project (orange, Smeed et al., 2018).

the progress of the global monitoring efforts and make some recommendations to fill some of the gaps in the coming years.

High quality and global coverage observations are essential to monitor the **ocean temperature** changes (Abraham et al., 2013). The primary instruments of the ocean subsurface observing system since the 1940s are MBTs, XBTs, Nansen/Niskin bottles, and CTDs (**Figure 10A**). MBTs typically go down to ∼125–250 m and were widely deployed from 1938 to the early 1960s. Shallow XBTs (e.g., T4/T6) reach 450 m, and were widely deployed during the 1970s∼1980s whereas deep XBTs (e.g., T7/DB) provide data to 800 m, and were widely used during the 1990s and early 2000s. The Argo Program, designed in 1998 achieved its initial goal of 3000 profiling floats in November 2007. Since 2007, the data coverage is > 80% of the global ocean area (3 by 3 degree box) from 0 to 1,200 m depth and > 70% for 1,200–2,000 m depth (**Figure 10B**; Meyssignac et al., 2019). Maintaining and improving the current ocean observation system are strongly recommended to ensure the accurate ocean climate monitoring. It is also essential to improve the historical record, for example, by recovering un-digitized temperature (OHC) and other observations.

Some limitations remain for the current ocean observation network, particularly for coastal regions, marginal seas, deep ocean regions below 2,000 m. It is important to establish a deep ocean system in the future to monitor ocean changes below 2,000 m, thus to provide a complete estimate of earth's energy imbalance (Johnson et al., 2015; von Schuckmann et al., 2016). Currently, boundary currents are not fully represented by Argo as floats can swiftly pass through the energetic regions, e.g., western boundary current (WBC) regions which could induce an inverse cascade of kinetic energy and affect the large scale low-frequency variability (Wang et al., 2017; Llovel et al., 2018). Achieving adequate sampling will require an observing system design based on a mixture of observing technologies adopted to the different operating environments. There is a need to develop/maintain multiple platform observations for cross-validation and calibration purposes (Meyssignac et al., 2019).

Intensive national and international efforts focused on carbonate chemistry monitoring, biological observations and biogeochemical/ecological forecast modeling over the past decade have shed light on the status and impacts of **ocean acidification** on local to global scales. International observing networks deployed around the world which use moorings, repeat hydrography research cruise transects, ships of opportunity, and fixed ocean time-series to track ocean chemistry include the Global Ocean Ship-based Hydrographic Investigations Program (GO-SHIP) surveys, the Surface Ocean CO_2 Observing Network (SOCONET), the Ship of Opportunity Program

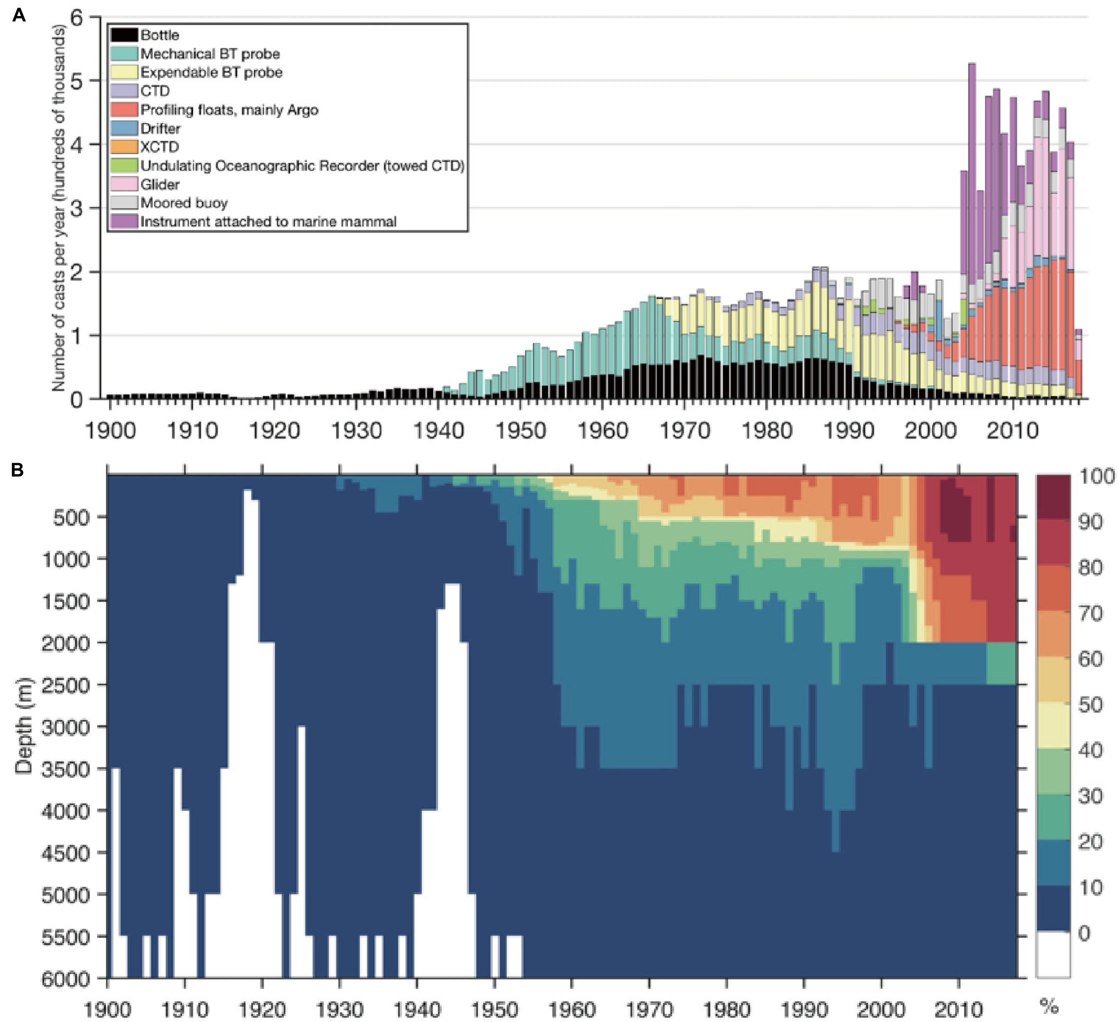

FIGURE 10 | (A) Number of subsurface ocean temperature profiles per year by instrument type 1900–2017. (BT, Bathythermograph; CTD, Conductivity-Temperature-Depth; XCTD, Expendable CTD). **(B)** Ocean subsurface observation coverage. Percentage (%) of data coverage for 3 × 3 boxes over the global ocean area from 5 to 6,000 m. Figure from Meyssignac et al. (2019).

(SOOP) volunteer observing ships, and the Ocean Sustained Interdisciplinary Time-series Environment Observation System (OceanSITES) time-series stations in the Atlantic, Pacific, and Indian oceans (**Figure 11**). These have provided essential, climate quality carbonate chemistry observations needed to understand ocean acidification in open waters. Biogeochemical Argo platforms, still under development, will increase the availability of pH profiles throughout the water column, along with other hydrographic parameters. In an effort to both coordinate with these international efforts and to coordinate with and expand national ocean acidification observing efforts, the Global Ocean Acidification Observing Network (GOA-ON) was launched in 2013. Through GOA-ON, organizations and scientists have established observation standards, enhanced data sharing, and quantified global and regional ocean acidification trends to identify areas of heightened vulnerability or resilience.

There have been many significant leaps in comprehending global ocean acidification trends and impacts and more research is needed to better inform models and improve predictions of the Earth system response to ocean acidification (Jewett et al., 2020). This includes the relationship between the impacts of ocean acidification and other stressors, such as warming, on organisms and communities. Understanding the direct and indirect impacts on marine populations and communities and the capacity of organisms to acclimate or adapt to the changes in ocean acidification-induced ocean chemistry will be extremely important in determining and predicting the economic, ecological, and societal impacts of ocean acidification. There remains a strong need for more extensive monitoring in coastal regions, including access to high quality, low-cost sensors to do this monitoring, and to satellite data and research into the long-term trends in ocean chemistry beyond the observational record (paleo-OA).

While **dissolved oxygen** data coverage has been proven to be sufficient to derive large scale and global trends, significant better long-term measurements are needed to address current

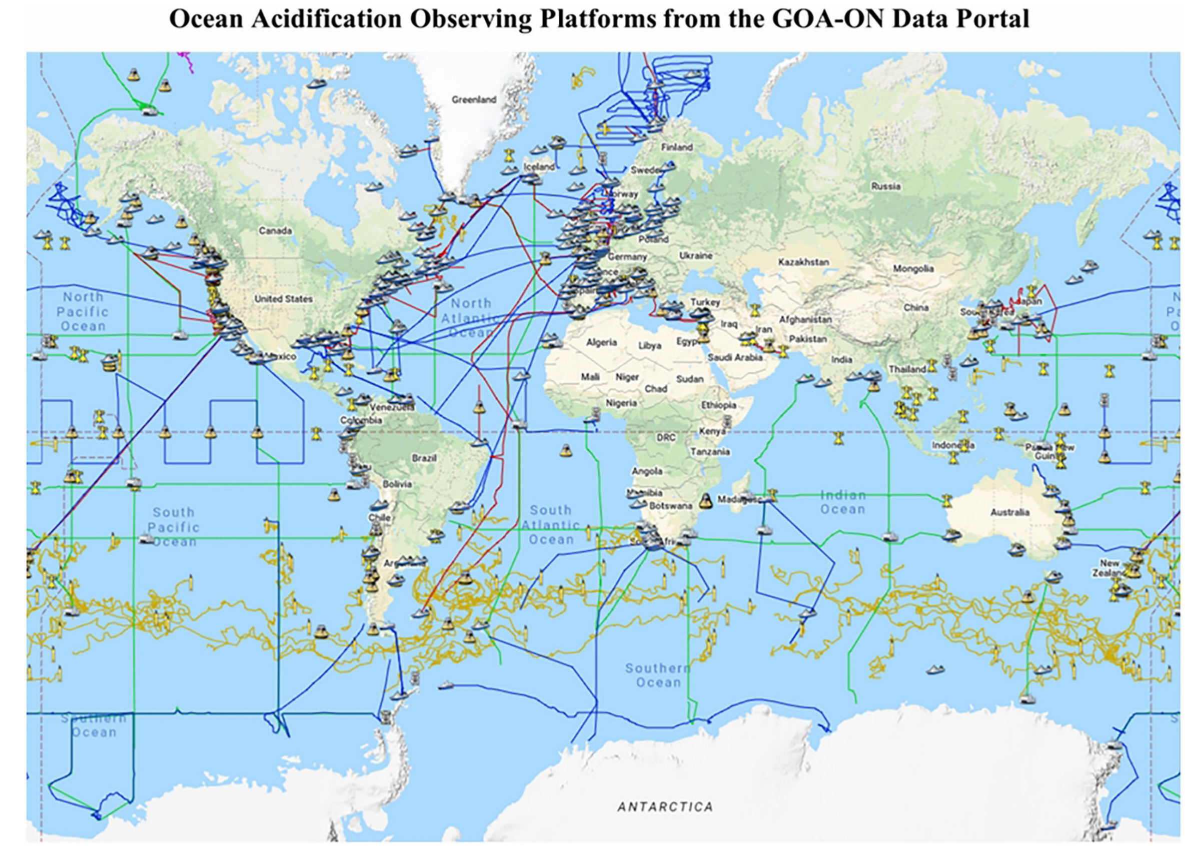

FIGURE 11 | Visualization of ocean acidification data and data synthesis products being collected around the world from a wide range of sources, including moorings, research cruises, and fixed time series stations. Figure source GOA-ON Data Portal (http://portal.goa-on.org/).

questions. Measurements at greater spatial and temporal extent are needed in many regions in order to capture for example the high variability in oxygen content in coastal areas. In this context it is of significant importance that a variety of biogeochemical parameters are increasingly added to the automated monitoring of the global ocean, since they serve as vital indicators for changes in the biogeochemical dynamics, which cannot be analyzed detached from changes in small- and large-scale ocean dynamics.

The International Arctic Buoy Programme (IABP)[6] maintains the fundamental Arctic Observing Network of drifting buoys which monitor ocean and sea ice circulation, as well as sea level pressure and surface temperature. While the IABP has been able to improve and maintain a denser network of drifting buoys, the prevailing winds and ocean circulation quickly carries these buoys away from the Eurasian coast of the Arctic Ocean, thus creating a recurring gap in the network during the winter that needs to be replenished since these gaps in the network hamper our ability to completely monitor and understand Arctic change (Thoman et al., 2020).

The best estimates of the long-term trends in **Arctic sea ice** volume are provided by models (e.g., Schweiger et al., 2019) given the paucity of *in situ*, pan-Arctic measurements of sea ice thickness. These models are compared to satellite retrievals of ice thickness and *in situ* measurements from field programs on the ice, aerial surveys (e.g., Haas et al., 2017), and submarine transits under the ice (e.g., Rothrock et al., 1999). The satellite retrievals are also compared to *in situ* measurements of ice thickness, and it has been shown that the primary source of uncertainty in these retrievals is the assumed depth of snow on top of the sea ice (Kwok and Cunningham, 2015; Kwok, 2018). Recently, the Multidisciplinary drifting Observatory for the Study of Arctic Climate (MOSAiC) Expedition completed a year-long drift across the Arctic Ocean (Shupe et al., 2020), and collected myriad of observations including measurements required to improve our understanding of Arctic climate processes, such as *in situ* measurements of snow and ice thickness, aerial and under ice surveys. Similar campaigns should be conducted routinely on a pan-Arctic scale (Haas et al., 2017; IPCC, 2019).

There are different programs dedicated to monitoring the different contributing factors of **sea level change**. The Argo program, mentioned before, is devoted to the monitoring of the temperature, salinity, currents and bio-optical properties of the global ocean reaching 2,000 m depths. From these globally distributed floats, changes in the density of the water column are estimated. These observations allow monitoring the contribution of the steric change of the GMSL of the ocean. With regard to the barystatic component, the GRACE space

[6]http://IABP.apl.uw.edu

gravimetry mission (covering the period 2002–2017) and the subsequent GRACE Follow-On (from 2018 on) satellite mission are registering global anomalies of the Earth's gravity field. To give continuity to the altimetry sea level record, Copernicus Sentinel-6 Michael Freilich satellite has been recently launched and has provided some first promising results. The launch of its twin, Sentinel 6B, is planned for 2025 (follow on of the Jason satellites). In the future the SWOT satellite will contribute to the improvement of the coastal data, due to its higher spatial resolution and in addition, this will measure river discharges and as such will be a valuable source of sea level budget from terrestrial contribution.

It is worth mentioning that when the footprint of the altimeter covers not only the sea but also the land the returning echoes are contaminated and consequently, it is not accurate enough at around 20–50 km from the coast. For retrieving altimetry data in those areas, the coastal altimetry community has investigated how to re-process these data by using different waveform retracking algorithms and applying different geophysical corrections (Vignudelli et al., 2019) and have provided different coastal altimetry databases (Birol et al., 2017; Cipollini et al., 2017; The Climate Change Initiative Coastal Sea Level Team, 2020) with more accurate data comparing to the conventional altimetry databases.

Information about the **ocean circulation** and its changes can be inferred from either direct measurements, proxies, model simulations and satellites. The main uncertainties regarding the trends in ocean circulation arise from the short time spans of the direct continuous measurements, the incompleteness when representing a circulation through proxies and the inherent uncertainties of the models. It is therefore essential that the existing observation programs like the Global Drifter Program (Dohan et al., 2010) and the Argo Program are sustained. This includes but is not limited to the main programs observing the AMOC, i.e., the RAPID programs (e.g., Smeed et al., 2014, 2018) that continuously measure the AMOC strength since 2004 at roughly 26°N, the SAMOC programs that measure AMOC strength in the South Atlantic and include the SAMBA array at about 34.5°S (Meinen et al., 2018; Kersale et al., 2020) and the OSNAP program (Lozier et al., 2017) measuring the overturning that feeds the AMOC since 2014.

A VISUAL SUMMARY OF OCEAN CLIMATE CHANGE INDICATORS

International data programs like Copernicus and others can play a relevant role to integrate the different data sets and provide the signs for ocean climate change. With that objective **Figures 12**, **13** present here a final visual summary of key ocean climate change indicators and current trends based on the individual analyses of the Copernicus Marine Services (CMS), that emphasizes recent changes (1993–2019/20).

Global Trends (1993–2019/20)

- According CMS, during the period 1993–2019 the global sea surface temperature (SST; **Figure 12A**) has increased at a mean rate of 0.15°C (±0.01°C) per decade, an increase of ~0.4°C in 27 years. The upper (0–700 m) near-global ocean (60°N-60°S) heat content (**Figure 12B**) shows during that period a warming rate of 0.9 ± 0.01 Wm^{-2}.
- During the years 1993–2019, the global mean sea level (**Figure 12C**) has been rising at a mean rate of 3.3 mm $year^{-1}$ with an uncertainty of ± 0.4 mm $year^{-1}$. This represents a sea level rise of 9 cm in the 27 year period. The upper (0–700 m) near-global ocean (60°N-60°S) thermosteric sea level (the sea level resulting of the volume expansion due only to the temperature increase; **Figure 12D**) has risen gradually at a rate of 1.5 ± 0.1 mm $year^{-1}$, which accounts for 45% of the global mean sea level increase.
- Since 1979 the Northern Hemisphere Sea ice extent (**Figure 12E**) has decreased at a mean rate of −0.52 million Km^2 per decade (1979–2017), and accordingly the freshwater of the Arctic Ocean (**Figure 12F**) has increased in volume by 4,230 ± 390 Km^3 per decade (data since 1993).
- The annual downward flux of CO_2 (**Figure 12G**) represents the ocean uptake of CO_2 over the whole ocean and has increased at a rate of 0.06 PgC $year^{-2}$ during the period 1985–2019. The annual global ocean CO_2 sink during the recent years 2017 and 2018 was, according CMS, 2.51 ± 0.17 and 2.61 ± 0.20 PgC $year^{-1}$ and the average sink over the full period (1985–2019) was 1.51 ± 0.14 PgC $year^{-1}$. The increasing concentrations of CO_2 in the ocean resulted in the ocean pH (**Figure 12H**) decreasing linearly at a mean rate of −0.0016 ± 0.0006 pH units per year (1985–2019) or a decrease of 0.056 pH units during the last 35 years.

Mediterranean Sea, Black Sea and Baltic Sea Trends (1993–2019/20)

CMS also provides regional analyses and we present here in **Figure 13** the results for three European semi-enclosed Seas: the Mediterranean Sea, the Black Sea and the Baltic Sea.

- The sea surface temperature (SST) during the period 1993–2019 has increased at a mean rate of 0.37 ± 0.03°C per decade in the Mediterranean Sea, 0.71 ± 0.04°C per decade in the Black Sea and 0.28 ± 0.03°C per decade in the Baltic Sea, though superimposed on these trends is a strong year-to-year variability. The sea surface temperature during this period (1993–2019; 27 years) has increased, respectively, at about 1°C (Mediterranean Sea), 1.9°C (Black Sea), and 0.7/0.8°C (Baltic Sea).
- The regional mean sea level has risen during the years 1993–2020 at a rate of 2.5 mm $year^{-1}$ in the Mediterranean Sea, 1.9 mm $year^{-1}$ in the Black Sea and 4.2 mm $year^{-1}$ in the Baltic Sea. This regional indicator also presents a high interannual variability (and possibly longer-term natural variability; Garcia-Soto et al., 2012) that impacts the trend with an uncertainty of ± 2.2 mm $year^{-1}$ in the three semi-enclosed seas.

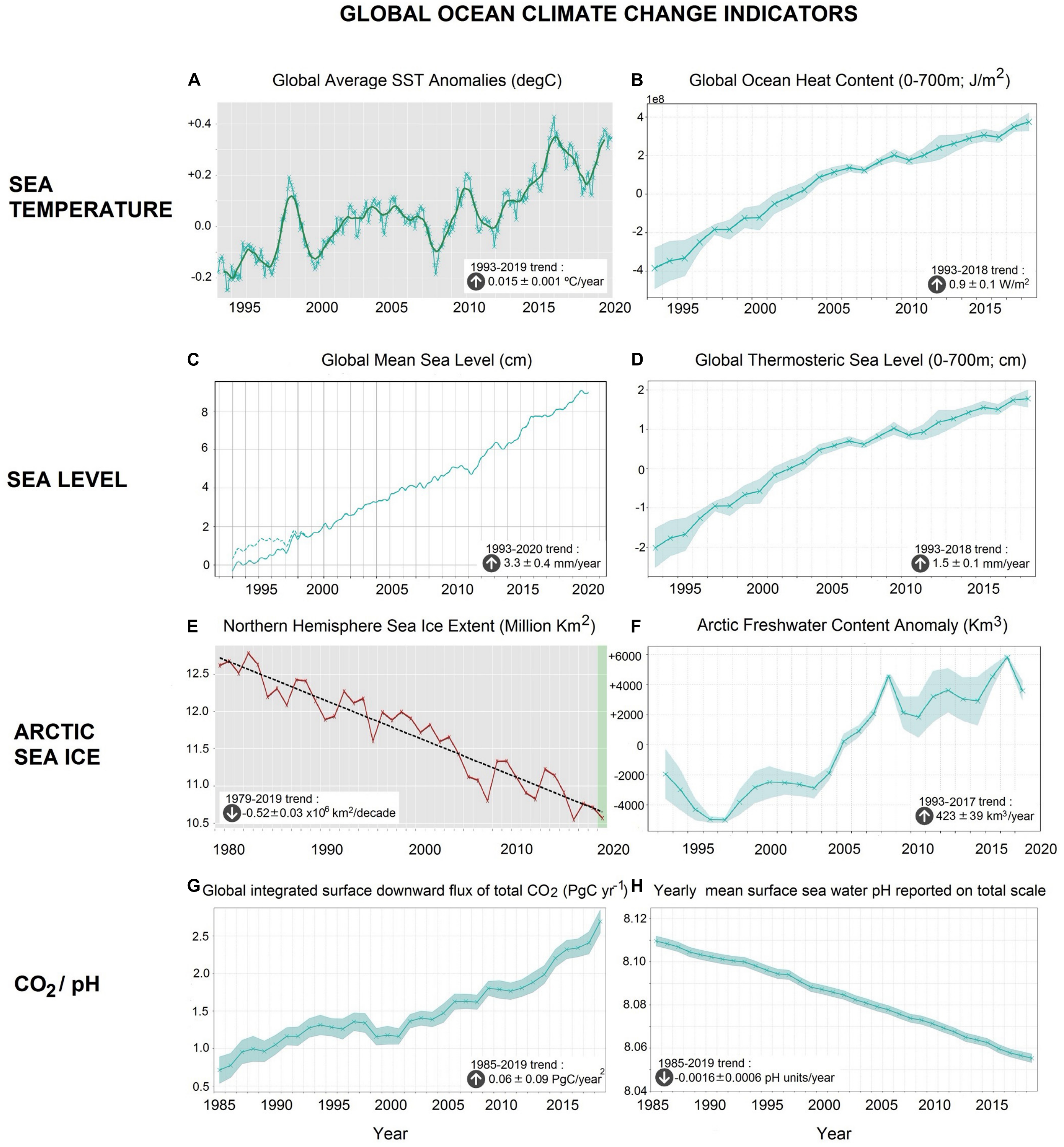

FIGURE 12 | Visual Summary of Global Ocean Climate Change Indicators **(A)** Global Sea Surface Temperature (SST) anomalies (°C) during the period January 1993–December 2019. The anomalies are relative to the climatological period 1993–2014. Blue crosses (monthly mean values) and green thick line (filtered values). **(B)** Global Ocean Heat Content (OHC; 0–700 m; J/m^2) for the period January 1993–December 2018. 60°N–60°S. Spread indicated with shade. **(C)** Global Mean Sea Level (cm) during the period January 1993–October 2019. Daily altimetric measurements. The time-series are low-pass filtered, the annual and semi-annual periodic signals adjusted and the curve corrected for the Glacial Isostatic Adjustment (GIA; Peltier, 2004). *The* dashed line indicates an estimate of the global mean sea level corrected for the drift of the TOPEX-A instrument during 1993–1998 (Ablain et al., 2017). **(D)** Global thermosteric sea level (0–700 m; cm) for the period January 1993–December 2018. **(E)** Northern Hemisphere sea ice extent (millions of Km^2) during the period January 1979–December 2019 (40 years). Based on satellite passive microwave data (SMMR, SSM/I, SSMIS). Sea Ice Extent is defined as the area covered by sea ice, or area having more than 15% sea ice concentration. All northern hemisphere sea ice is included, except for lake or river ice. **(F)** Arctic ocean freshwater content annual anomalies (Km^3) during the years 1993–2017. The regional domain is the Arctic Ocean basin with a depth > 500 m. **(G)** Global area integrated annual surface downward flux of total CO_2 (PgC/year) during the period January 1985–December 2019. **(H)** Annual global mean surface sea water pH over the period January 1985–December 2019 using a reconstruction methodology. Individual figures credited to EU Copernicus Marine Service (CMS).

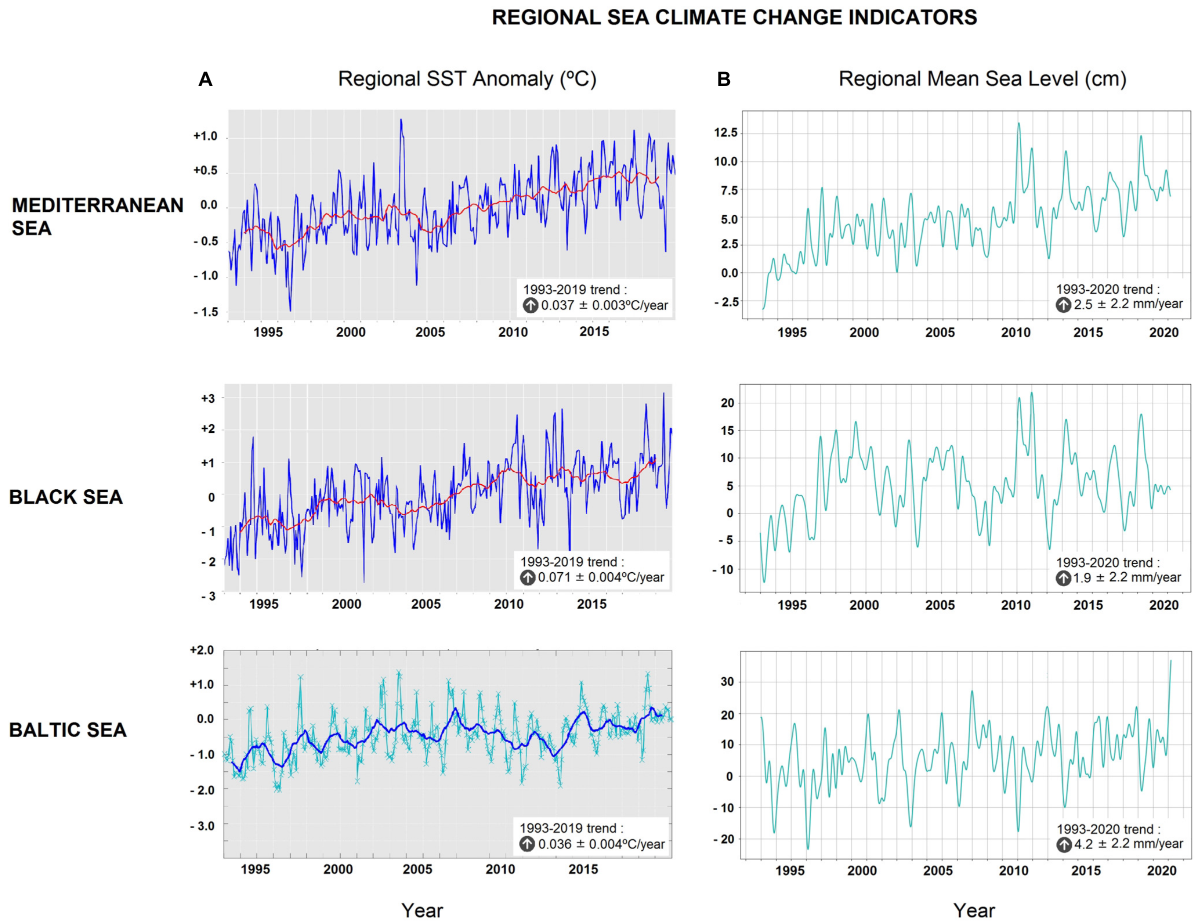

FIGURE 13 | (A) Time series of regional Sea Surface Temperature (SST) anomalies (°C) during the period January 1993–December 2019 (27 years) in the Mediterranean Sea, the Black Sea and the Baltic Sea. Mediterranean and Black Sea: blue line (monthly mean) and red line (24-month filtered). Baltic Sea: green crosses (monthly mean) and blue line (1 year trends). The anomalies are relative to the climatological period 1993–2014. **(B)** Time series of regional daily Sea Level (cm) during the period January 1993–February 2020 (27 years) in the Mediterranean Sea, the Black Sea and the Baltic Sea. Altimetric data derived from the average of gridded sea level maps weighted by the cosine of the latitude. The time-series are low-pass filtered, the annual and semi-annual periodic signals adjusted, and the curve corrected for the Glacial Isostatic Adjustment (GIA; Peltier, 2004). Individual figures credited to EU Copernicus Marine Service (CMS).

The Ocean Climate Change Indicators can be finally contextualized in larger international frameworks including the Sustainable Development Goals 13 (Climate Action) and 14 (Life Below Water) of UN Agenda 2030. Sea Surface Temperature (SST) is one of the essential climate variables (ECV) of the Global Climate Observing System (GCOS; Bojinski et al., 2014) and gives information about the flow of heat in the ocean and about modes of ocean and atmospheric variability (e.g., ENSO). Ocean Heat content is also an essential climate variable (ECV) and one of the 6 global climate indicators initially proposed by the World Meteorological Organization (WMO; Williams and Egglestob, 2017) for the Sustainable Development Goal 13 "Climate Action" ($SDG13_{WMO}$). Ocean Heat content variations produce changes in stratification and currents, impact sea ice, ice shelves and marine ecosystems, and play a role in sea level change and in the ocean-atmosphere interactions (WCRP, 2018; IPCC, 2019; von Schuckmann et al., 2020). Mean Sea Level (also an ECV) was proposed by WMO as an additional SDG-13 indicator ($SDG13_{WMO}$). It reflects the amount of heat added to the sea and the mass loss due to land ice melt, and has a direct impact on the coastal areas and population (e.g., WCRP, 2018; IPCC, 2019). Variations of sea ice cover (also ECV and $SDG13_{WMO}$ indicators) can modify the key role played by the cold poles in the Earth climate, and variations in the volume of Arctic freshwater can produce changes in ocean stratification, and influence the circulation and heat transport. As part of the Global Carbon Budget (Le Quéré et al., 2018) the ocean CO_2 storage (also ECV and $SDG13_{WMO}$ indicators) is evaluated every year. The ocean has absorbed about 25% of all anthropogenic CO_2 emissions since 1950 (Friedlingstein et al., 2020). A direct consequence of the uptake of carbon dioxide by the ocean is the decrease of surface ocean pH. Monitoring the surface ocean pH has become the focus of contributes to the Sustainable Development Goal 14 (SDG14) "Life below water."

The Global Climate Observing System (GCOS) and the UN Sustainable Development Goals 13 and 14 are setting in this way the set of key ocean indicators that ensure monitoring of the climate change signals in the global ocean in an integrated and coordinated manner. And these updated climate change indicators, as the ones presented here, highlighting the impacts in the present and future ocean, will allow a better action-taking towards an urgently needed mitigation and adaptation.

AUTHOR CONTRIBUTIONS

CG-S conceived and led the manuscript. All authors contributed to the writing of the manuscript.

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Who Is Where in Marine Food Webs? A Trait-Based Analysis of Network Positions

Anett Endrédi[1,2†], *Katalin Patonai*[1,3†], *János Podani*[3], *Simone Libralato*[4] *and Ferenc Jordán*[5,6*]

[1] Department of Community Ecology, Institute of Aquatic Ecology, Centre for Ecological Research, Budapest, Hungary, [2] Evolutionary Systems Research Group, Centre for Ecological Research, Budapest, Hungary, [3] Eötvös Loránd University, Budapest, Hungary, [4] Istituto Nazionale di Oceanografia e di Geofisica Sperimentale, Trieste, Italy, [5] Stazione Zoologica Anton Dohrn, Naples, Italy, [6] Democracy Institute, Central European University, Budapest, Hungary

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Edited by:
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Stazione Zoologica Anton Dohrn Napoli, Italy

****Correspondence:***
Ferenc Jordán
jordan.ferenc@gmail.com

† These authors share first authorship

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Networks of trophic interactions provide a lot of information on the functioning of marine ecosystems. Beyond feeding habits, three additional traits (mobility, size, and habitat) of various organisms can complement this trophic view. The combination of traits and food web positions are studied here on a large food web database. The aim is a better description and understanding of ecological roles of organisms and the identification of the most important keystone species. This may contribute to develop better ecological indicators (e.g., keystoneness) and help in the interpretation of food web models. We use food web data from the Ecopath with Ecosim (EwE) database for 92 aquatic ecosystems. We quantify the network position of organisms by 18 topological indices (measuring centrality, hierarchy, and redundancy) and consider their three, categorical traits (e.g., for mobility: sessile, drifter, limited mobility, and mobile). Relationships are revealed by multivariate analysis. We found that topological indices belong to six different categories and some of them nicely separate various trait categories. For example, benthic organisms are richly connected and mobile organisms occupy higher food web positions.

Keywords: food web, traits, network position, centrality, keystones, Ecopath with Ecosim

INTRODUCTION

In order to sustain the proper functioning of ecosystems, we need to better understand the simple question of Lawton (1994): What species do in ecosystems? Since ecological roles and food web positions are not independent (Luczkovich et al., 2003), we address the question what kind of species occupy certain kinds of network positions.

Since the very first attempts to identify keystone species (Paine, 1966, 1969), there has been an interest in their place in food webs (Mills et al., 1993; Power et al., 1996). First they were suggested to have been top predators, then also plants, herbivores, and parasites (Bond, 1994; Marcogliese and Cone, 1997). For both community ecology and conservation biology, it would be very useful to know where are they in complex trophic networks.

While it is clear that the relative importance of organisms varies with time and space, looking at a large database may provide some general insight into the problem. If certain types of organisms occupy certain types of network positions, results can increase the predictability of food web modeling. Comparisons of centrality indices with each other (the similarity of DC and CC: Jordán et al., 2007; K predicts KSI better than s: Endrédi et al., 2018b) and centrality indices with trophic level (most high-centrality species at medium trophic levels: Scotti and Jordán, 2010) were done to better understand critically important positions of organisms in food webs. Extending this interest by adding trait data to trophic groups helps the biological interpretation of the results. Relationships between centrality indices have been studied for other network types as

TABLE 1 | List of topological indices.

Index name	Description
Degree centrality (DC)	Number of other nodes connected directly to the considered node (Wasserman and Faust, 1994).
Weighted degree centrality (wDC)	Sum of weights of links adjacent to the considered node (Wasserman and Faust, 1994).
Betweenness centrality (BC)	Frequency of the considered node on the shortest paths connecting all pairs of other nodes (Wasserman and Faust, 1994). $$BC_i = \frac{2\sum_{j<k}\frac{g_{jk}(i)}{g_{jk}}}{(N-1)(N-2)}, i \neq j, k$$ g_{jk} is the number of equally shortest paths between nodes j and k, $g_{jk}(i)$ is the number of these shortest paths to which node i is incident in the length of the shortest path between nodes i and j in the network.
Closeness centrality (CC)	Quantifies how short are the minimal paths from a given node to all others. $$CC_i = \frac{N-1}{\sum_{j=1}^{N} d_{ij}}, i \neq j$$ d_{ij} is the length of the shortest path between nodes i and j in the network (Wasserman and Faust, 1994).
Topological importance (TI^3) Weighted importance (WI^3) Topological overlap (TO) Weighted overlap (WO)	The topological importance of species i when effects "up to" n steps are considered is the sum of effects originated from species i up to n steps averaged over by the maximum number of steps considered (i.e., n): $$TI_i^n = \frac{\sum_{m=1}^{n}\sigma_{m,i}}{n} = \frac{\sum_{m=1}^{n}\sum_{j=1}^{N} a_{m,ji}}{n}$$ $a_{m,ji}$ is the effect of j on i when i can be reached from j in n steps (Jordán et al., 2003). We analyzed indirect effects of maximum three steps ($n = 3$). WI_i^n is the same but with weighted links. We can assess the overlap in the neighbors of two nodes quantifying the uniqueness or redundancy of nodes (Jordán et al., 2009; Lai et al., 2015), as a function of a t threshold for the TI^n and the WI^n matrices, providing TO and WO, respectively.
Status (s), contra-status (s′) and net status (Δs)	In a directed strong hierarchy, the status is the sum of ij distances from node i to every other node j. Reversing the hierarchy (reverting the direction of the links), the same calculation will give the contrastatus of each node (s_i') (Harary, 1959): $$\Delta s_i = s_i - s'_i$$ Δs_i is called the net status of node i.
Keystone index and its components (K, K_{bu}, K_{td}, K_{dir}, and K_{indir})	The keystone index of a species i is defined as (Jordán et al., 1999): $$K_i = K_{bu,i} + K_{td,i} = K_{dir,i} + K_{indir,i} =$$ $$= \sum_{c=1}^{n}\frac{1}{d_c}(1 + K_{bc}) + \sum_{e=1}^{m}\frac{1}{f_e}(1 + K_{te})$$ n is the number of predators eating species i, d_c is the number of prey species of its c-th predator, K_{bc} is the bottom-up keystone index of the c-th predator, m is the number of prey eaten by species i, f_e is the number of predators of its e-th prey, K_{te} is the top-down keystone index of the e-th prey, $K_{bu,i}$ is the bottom-up keystone index, $K_{td,i}$ is the top-down keystone index, $K_{dir,i}$ represents the direct effects for node i, $K_{indir,i}$ represents the indirect effects for node i.

well, including habitat networks (Baranyi et al., 2011; Pereira et al., 2017).

With large databases and new statistical analyses, these questions can be re-investigated and our knowledge can be updated. In this article, we consider a large database of trophic networks, described by standard methodology for both data collection and network construction, making them comparable. We (1) characterize the network position of each trophic component by a variety of topological indices, quantifying centrality, hierarchy, redundancy, keystoneness, and trophic level, (2) characterize each trophic component by three traits, and (3) use multivariate methods for comparisons between various topological indices and between topological indices and traits.

MATERIALS AND METHODS

Data from 92 Ecopath with Ecosim (EwE) aquatic food web models were compiled using the EcoBase online database repository (Colléter et al., 2013) and previously published sources (Heymans et al., 2014). These networks have varying number of nodes (ranging from 8 to 63) but were assembled using comparable methodology of the EwE framework (Christensen and Walters, 2004; Heymans et al., 2016). For models of the same ecosystem described in different years, we used the most recent one (considering the year of publication). The compiled data represent five global regions with diverse ecosystems: 14 models from Africa, 14 from Australasia, 29 from Europe, 27 from North America, and 8 from South America (**Supplementary Material A**).

The network position of each trophic component in each trophic network was characterized by 18 topological indices (see **Table 1** for description of computed indices). Centrality was quantified by six indices (four binary and two weighted), we used eight indices for hierarchy (i.e., centrality in DAGs), two indices for redundancy (topological overlap), one for keystoneness (KSI, Libralato et al., 2006) and also the measure of trophic level as it is used in EwE. The last two indices were retrieved from previous publications (see Heymans et al., 2014 and the references in **Supplementary Material A**). All other topological indices were computed using programs UCINET (Borgatti et al., 2002) and CoSBiLab (Valentini and Jordán, 2010).

In order to be able to use a wide range of topological indices, some of them with specific requirements, it was necessary to pre-process the database in a few steps. This ensured the applicability of indices and the comparability of the results. Since we focus on the interactions among living organisms, we deleted (1) non-living network components (e.g., DOM) and (2) living components that became isolated nodes after deleting the non-living ones (e.g., holothuroids in the Kuosheng Bay network). From an energetic point of view, detritus and cycling are clearly crucial to ecosystem dynamics, however, topological indices (who interacts with whom) may provide biased results and artifacts if non-living components are not deleted (e.g., detritus can simply be connected to each living component). We double-checked if this data processing had a major effect on the KSI and TL index values and found the difference only minimal and safely negligible (TL was changed highly consistently across the networks, as almost the same trophic groups were removed from almost the same positions, while KSI-values still quantify nodes in the original networks but their re-calculation is not possible for the modified networks – from a comparative perspective, neither makes real difference). This process rendered one small network (Maspalomas Lagoon) without primary producers, thus not usable for our study. Altogether this resulted in the deletion of 150 network components (127 non-living and 23 living) (**Supplementary Material A**). On average, this means 1.63 node (6%) per network. One additional node, Stellar Sea Lion pup ("SSL pup") from the Aleutian Islands model was an outlier (due to asymmetric connections of only having one predator and no prey) and was omitted. Before computing non-hierarchical indices, networks were symmetrized by summing the interactions' strengths. All loops were eliminated from 57 food webs to be able to compute hierarchical indices (detailed methods can be found in **Supplementary Material B**).

TABLE 2 | List of categorical traits (mobility, habitat, and size), categories used, definitions, and data coverage of the data set (n = 2210).

Trait	Category	Definition	Coverage
Mobility	Sessile	Attached	2203 (99.7%)
	Drifter	Passive moving	
	Limited mobility	Slow active moving (burrowers and crawlers)	
	Mobile	Fast active moving (swimmers)	
Habitat	Benthic	Benthic and demersal organisms	2165 (98%)
	Water column	Pelagic groups	
Body size (cm)	10^{-4}	<0.001 cm	2056 (93%)
	10^{-3}	0.001–0.01 cm	
	10^{-2}	0.011–0.1 cm	
	10^{-1}	0.11–1.0 cm	
	10^{0}	1.01–10 cm	
	10^{1}	10.01–100 cm	
	10^{2}	100.01–1000 cm	
	10^{3}	>1000 cm	

Note that the size category trait was converted to cm (from Sieburth et al., 1978).

Functional groups were assigned to three categorical traits (i.e., feeding habitat, mobility, and size category, **Table 2**) and one continuous trait (maximum body size). In general, the trait for the foraging adult form was considered, unless age (e.g., juvenile) or size (e.g., small) was specifically noted. Species-level habitat preference and maximum length measurements (in cm) were extracted from the FishBase (FishBase, 2020) and SeaLifeBase (SeaLifeBase, 2020) online databases and assigned to larger functional groups.

Generalizations are inevitable where species are not listed or are aggregated into functional groups (common practice in food web studies). Below we describe the generalizations we encountered and the methods used for trait assignments. First it is noted that we needed to work with a small number of relatively large categories in order to keep the cross-ecosystem

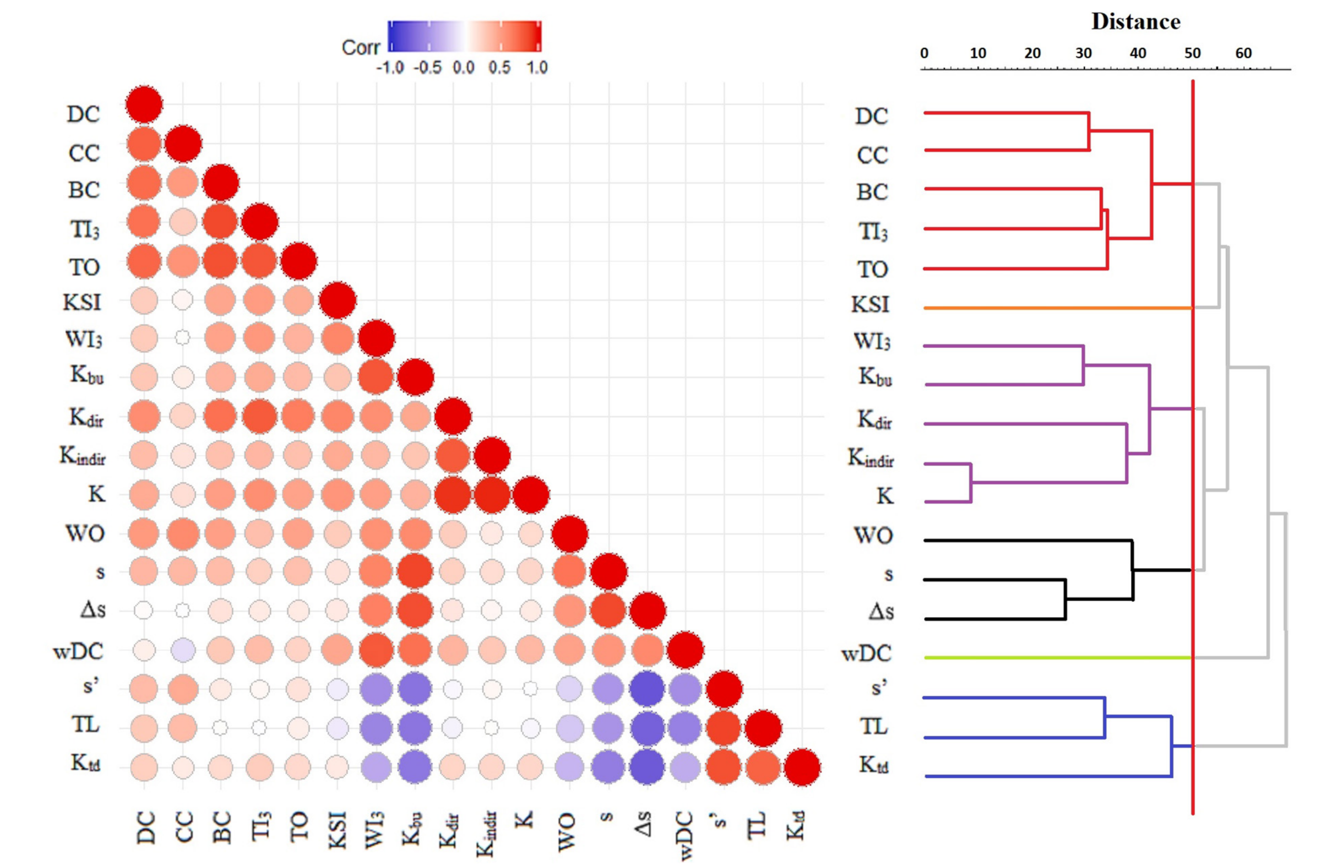

FIGURE 1 | Matrix plot of Spearman rank correlations among the 18 topological indices (left) and the UPGMA dendrogram based on Euclidean distances for standardized data. Negative correlations (e.g., between s′ and s) are shown in blue, while positive correlations (e.g., between DC and CC) are shown in red. The larger and darker the circle, the stronger the correlation. For the dendrogram, red vertical line denotes Euclidean distance at the level of 50, creating six clusters of indices ranking network nodes similarly in each cluster.

analysis feasible (more detailed classifications would reduce comparability). Several traits can be defined only for a smaller range of organisms, like "pigments" for phytoplankton (Weithoff and Beisner, 2019) or "dive duration" for the megafauna (Tavares et al., 2019). We tried to maintain the coverage of trait data for the possibly largest set of trophic groups (Kremer et al., 2017). For the habitat preference trait, benthic organisms included all of those associated with the benthos (infauna and epifauna) as well as demersal species (e.g., flatfish and rays) or those otherwise described living near the bottom (e.g., sandy or muddy surfaces) – all available in FishBase's species environment and biology descriptions. For other, non-specified fishes and sharks, we defaulted to the water column habitat. Phytoplankton, zooplankton, jellyfish, sea birds, sea turtles, and cetaceans were also assigned to the water column habitat. Other important categories (mesopelagic) were not considered for maintaining comparability and wide coverage among different ecosystem models, even if their importance is clear (Agnetta et al., 2019). The mobility trait was organized into four categories: sessile (attached), drifter (passive movers), limited mobility (slow active movers, including burrowers, and crawlers), and mobile (fast active movers and swimmers) (Costello et al., 2015). Sessile (e.g., macrophytes and barnacles) and drifter (e.g., plankton, bacteria, and fish larvae) organisms are biologically well-defined. Limited mobility organisms were mainly macroinvertebrates (e.g., echinoderms, gastropods, and annelids) and juvenile fish, whereas vertebrates capable of swimming (e.g., adult fish, turtles, birds, and marine mammals) were mobile. For non-species-specific size data (e.g., microzooplankton), we used Sieburth et al. (1978) plankton size fractions to extract maximum length (cm). Our data range from bacteria (0.0002 cm) to blue whales (3300 cm). Based on Sieburth's size fractions, functional groups were assigned to one of eight size categories (each category increasing by a factor of 10) (see **Table 2**).

Data coverage was relatively even (>93%) for the three categorical traits (**Table 2**). The continuous trait, maximum length, had the lowest data coverage (71%) and was not analyzed separately in this study, however, it was used to assign the nodes to body size classes. Distinction into trait categories was not always clear-cut due to ontogenetic shift in diet and habitat preferences (e.g., bathypelagic species) or food web aggregation problems (mixed groups or broad categories). For these functional groups, we made a case-by-case evaluation based on the detailed metadata (description based on original EwE

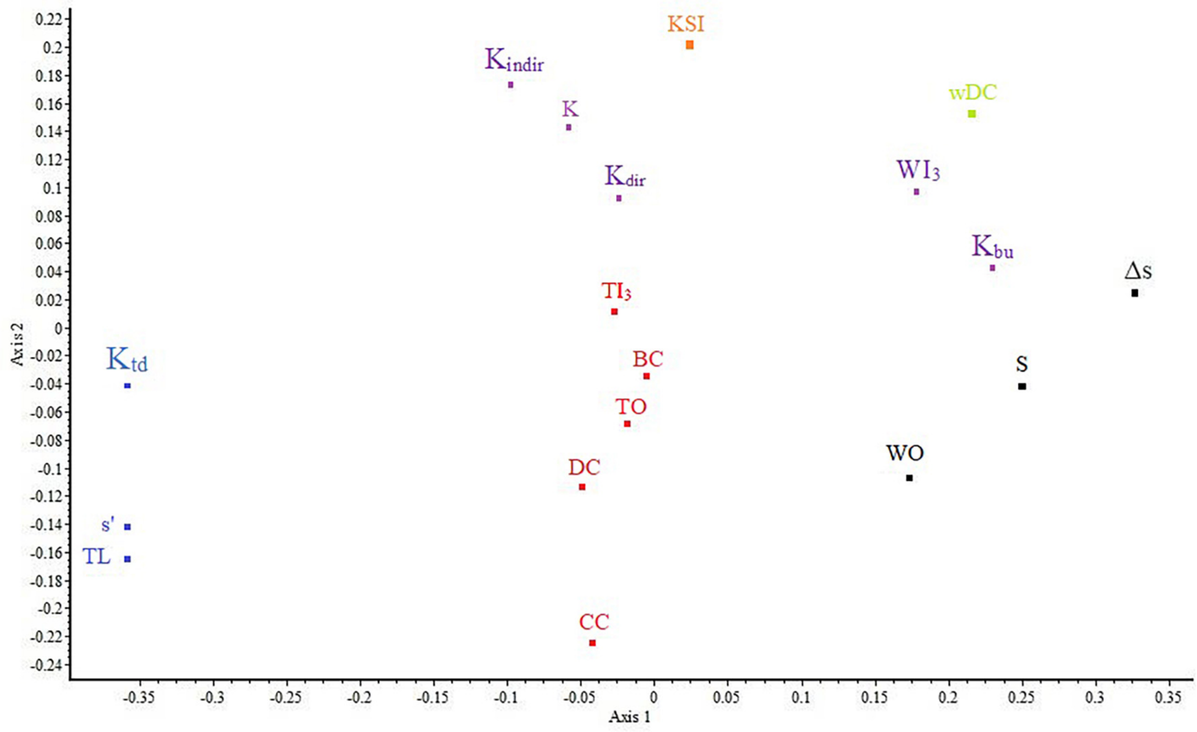

FIGURE 2 | Nonmetric Multidimensional Scaling ordination of the 18 topological indices (stress = 0.08) with the six-group classification of **Figure 1** superimposed. Although the ordination is nonmetric, the correspondence between the two results is remarkable, except for the positions of K-related indices.

publications) or left the trait blank ("NA"). If available from the metadata description, one representative was selected and categorized accordingly. Overall, our data sets are comparable in the sense that they have low resolution at the bottom (e.g., phytoplankton as a single group) and higher resolution at the top (e.g., fish species listed).

First, the relationship between topological indices was investigated using Spearman's rank correlation and multivariate analyses [Principal Component Analysis (PCA), hierarchical clustering, and Nonmetric Multidimensional Scaling (NMDS)]. PCA and hierarchical clustering were used as metric exploratory methods to reveal groups and correlations amongst the 18 indices. The results were compared with those obtained via ordinal methods (Spearman rank correlation and NMDS). PCA works well for linearly correlated data and requires few assumptions (e.g., accepts negative index values such as in s' or KS). Standardized PCA was applied to ensure commensurability of indices. Data for hierarchical clustering were standardized by the standard deviation of variables and then the indices were classified using Euclidean distance and the unweighted pair group method with arithmetic mean (UPGMA or group average method). While other clustering methods do exist, UPGMA was selected based on the highest cophenetic correlation value, which measures how closely the original distances are reproduced by distances in the dendrogram (Sokal and Rohlf, 1962). These methods are able to maintain much of the original metric information in the data, i.e., differences between the scores. Ordinal methods operate by reducing data to ranks thereby disregarding metric properties. From the Spearman's rank correlation coefficients (ρ), a dissimilarity semi-matrix was calculated according to the formula $d = 1 - \rho$, effectively converting the correlations to the interval [0,2]. Thus, $d = 0$ means complete similarity corresponding to identical rank orders, and $d = 2$ reflects complete dissimilarity, i.e., reverse rank orders. The matrix thus obtained was used as input to NMDS. Spearman's correlations were visualized by a matrix plot, while the dissimilarity values were subjected to NMDS to provide an ordination of indices. Analyses were computed and results were displayed using R software [R Core Team, 2020; packages: "stat" and "ggcorrplot " (Kassambara, 2019)], and the SYNTAX-2000 package (Podani, 2001).

Second, for testing the independence of the three categorical traits, Pearson's Chi-squared test and Fisher's exact tests were performed with simulated p-values, using the "stat" package in R (R Core Team, 2020).

Finally, the relationship between topological indices and functional traits was visualized in R ("ggpubr" package, see Kassambara, 2020) and analyzed using linear mixed-effects models, with the traits as fix effects and the networks as random effects (thereby accounting for network variability in the models). Before building the models, ten indices required transformation due to their positively skewed distribution (square-root transformation for moderate skew: BC, TO, K_{bu}, WO, and K_{td}; and log transformation for greater skew: wDC,

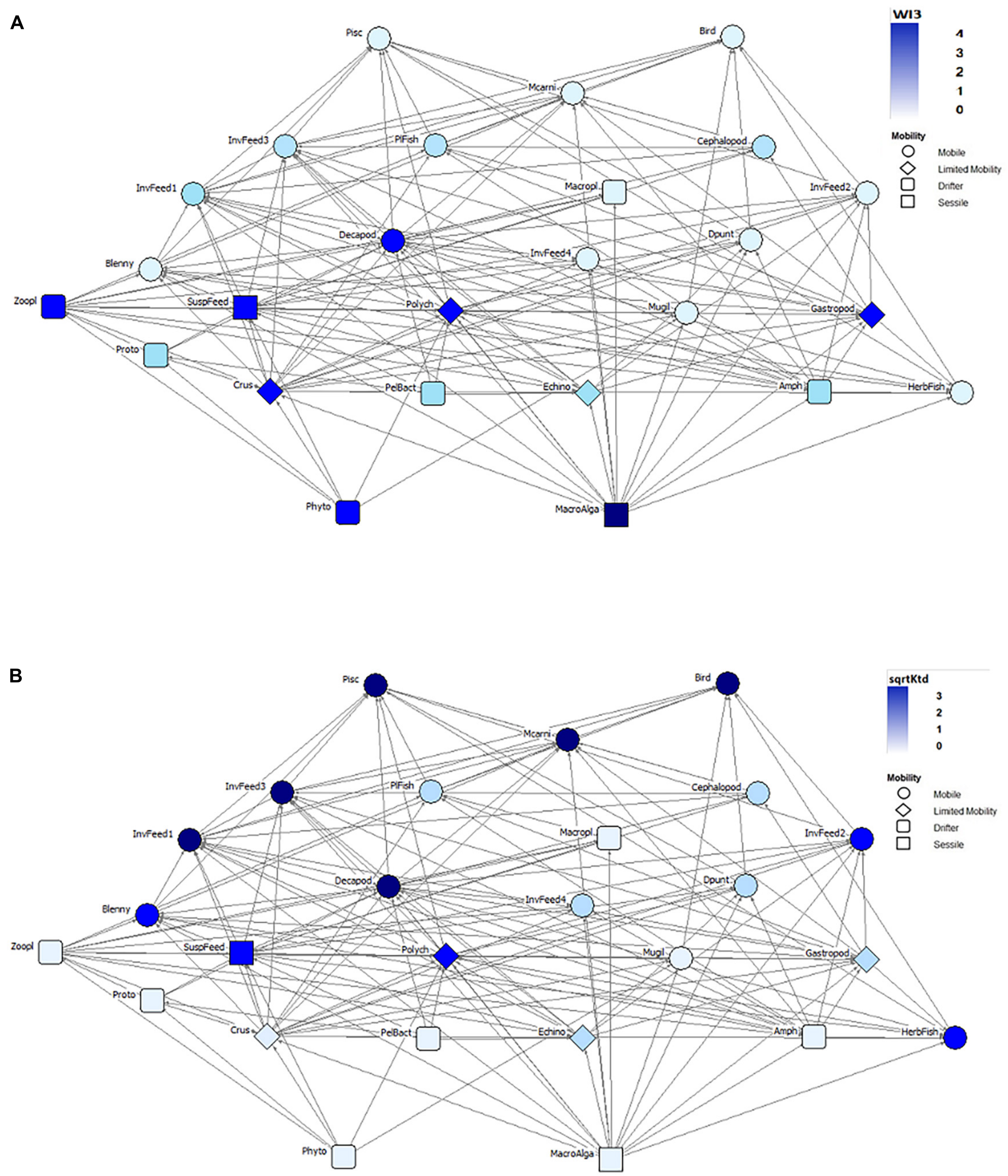

FIGURE 3 | The food web of Bay of Calvi (Pinnegar and Polunin, 2004) showing the relationship between the topological position of nodes (node color, see values in the inset) for logWI^3 **(A)** and sqrtK_{td} **(B)** and their mobility values (node shape, see categories in the inset). The abbreviations for the n = 26 trophic groups are: Phyto, phytoplankton; MacroAlga, macroalgae; Proto, pelagic protozoa; Crus, Crustacea; PelBact, pelagic bacteria; Echino, Echinoderms; Amph, Amphipods; HerbFish, herbivorous fish; Zoopl, zooplankton; SuspFeed, suspension feeders; Polych, polychaetes; Mugil, Mugilidae; Gastropod, gastropods; Blenny, omnivorous blennies; Decapod, decapods; Dpunt, *Diplodus puntazzo*; Macropl, macroplankton; PlFish, planktivorous fish; Cephalopod, cephalopods; Mcarni, macrocarnivorous fish; Pisc, piscivorous fish; Bird, seabirds; InvFeed1 through InvFeed4, benthic invertebrate feeders (groups 1–4).

WI^3, K, K_{dir}, and K_{indir}), and all indices were studentized within their network. The latter means that all index values were subtracted from the sample mean (mean value of the index in its network) and divided by the standard deviation of the sample. The transformations did not change the trends of the relationships between the indices and the traits but helped meet the model assumptions and make the values more independent from the network features. Mixed-effect models were built in R, using “lme4” (Bates et al., 2015) and “lmerTest” (Kuznetsova et al., 2017) packages.

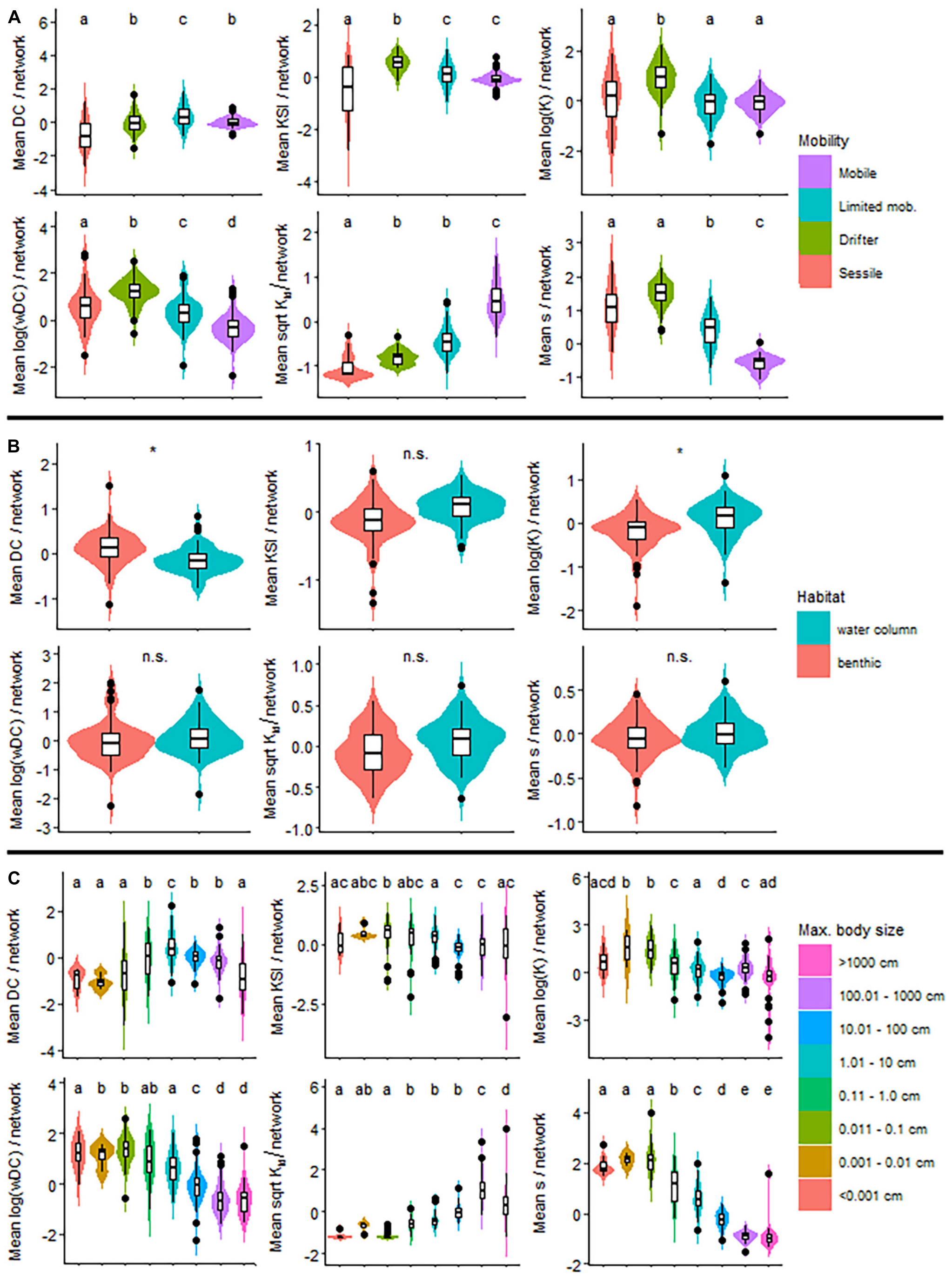

FIGURE 4 | The distribution of index values for trait categories. One representative index (DC, KSI, logK, logwDC, sqrtK_{td}, and s) is used for each of the six clusters in the dendrogram on **Figure 1**. Traits are mobility **(A)**, habitat **(B)**, and size **(C)**. * indicates significant difference between habitat categories. For each of the 18 indices, separately, see the same information in **Supplementary Material D**.

RESULTS

In the dendrogram resulting from the hierarchical classification of indices, six clusters are recognized at the level of 50 (**Figure 1**, right). Centrality indices (DC, CC, BC, TI^3, and TO) are grouped into the first cluster. The keystone index (KSI) is a singleton. The indirect component of the K index (K_{indir}) and K are the closest pair and comprise group three together with K_{dir}, K_{bu}, and WI^3. These two latter indices are related by both emphasizing bottom-up groups. The fourth cluster is somewhat mixed, containing two hierarchical indices (s and Δs) and a weighted index (WO). Weighted degree centrality (wDC) was found separately in group five. The sixth group is made up of three classical top-down indices (s′, TL, and K_{td}). The discussion of indices and traits will

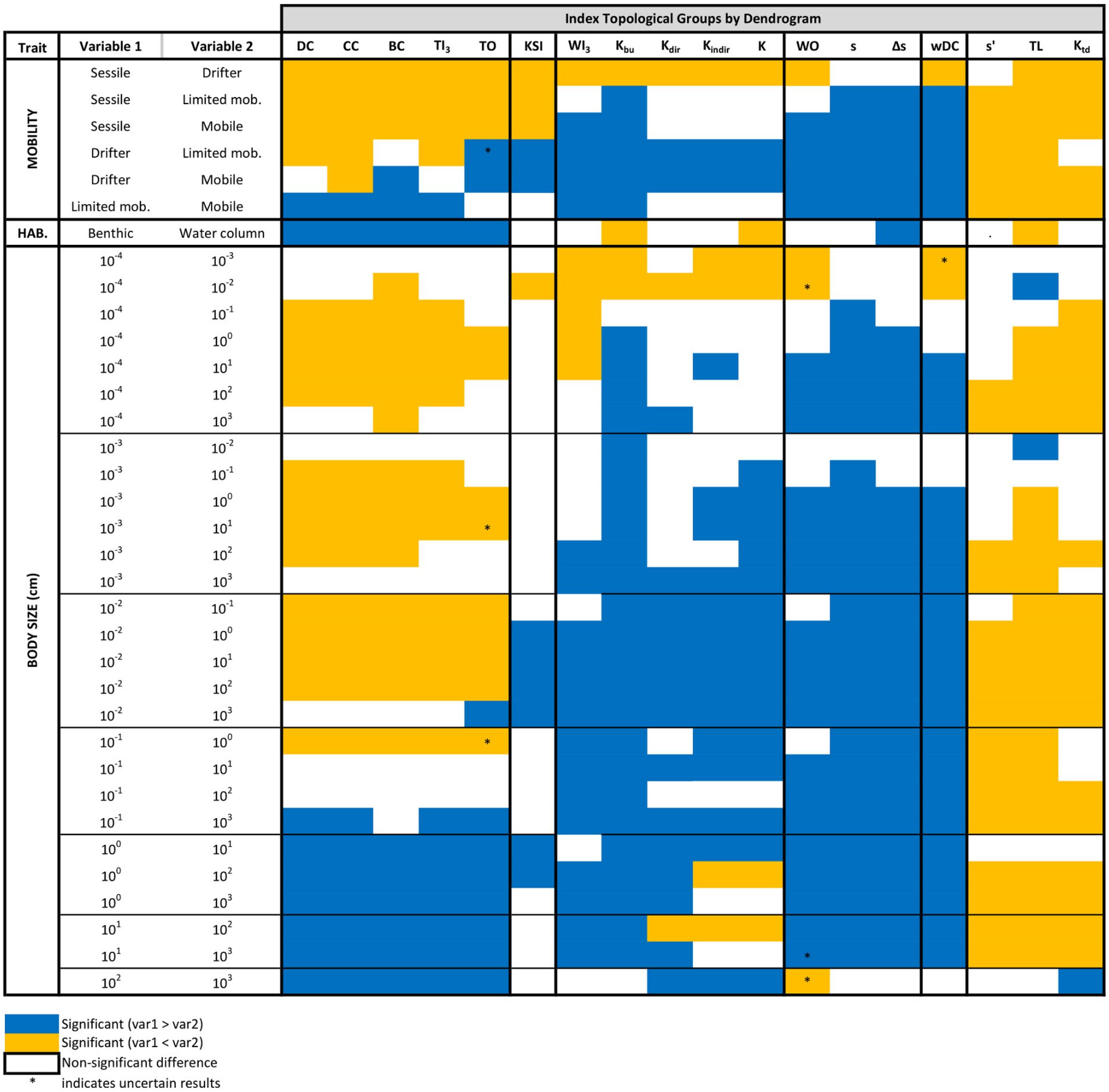

FIGURE 5 | The relevance of each topological index for separating each possible pair of trait values. Significant differences between the topological positions of nodes with different trait categories are marked by colors. For example, the first row shows that DC is different for sessile and drifter organisms but does not separate them.

be based on the groups classified in this dendrogram (**Figure 1**), since it had a high cophenetic correlation ($r = 0.8267$) indicating minimum distortion compared to the input Euclidean distances.

The NMDS ordination (**Figure 2**), even though an ordinal approach, identified largely the same clusters (stress = 0.08). The major difference is that K_{bu} and WI^3 fall away from the other three K components (K, K_{dir}, and K_{indir}) with which they formed a cluster in metric analysis, showing the inconsistent behavior of these K components. This pattern can also be observed on the matrix plot of Spearman rank correlations (**Figure 1**, left). In this diagram, rank correlations are contrasted with metric clustering, showing that the cluster membership of K_{dir} is the most ambiguous. The results of PCA can be found in **Supplementary Material C**. All four methods agree on the correlation of these indices, except for the above-mentioned K components (which are emphasized differently in metric versus ordinal approaches).

Next, we assessed the relationships of three common categorical traits (mobility, habitat, and size) with the 18 indices. We were interested in finding out which trait has predictive power in these aquatic ecosystems and which is negligible. We ran mixed-effects models on the combination of these traits to predict the importance of specific trophic groups in the networks (see **Supplementary Material E**). An example network for the food web of Bay of Calvi is shown for visualizing the relationships of the mobility trait with the indices WI^3 (**Figure 3A**) and K_{td} (**Figure 3B**). The former emphasizes bottom-up groups (e.g., sessile and drifters) and the latter brings attention to mobile groups at the top of the food web. **Figure 4** shows the relationship of the three traits with one representative index per dendrogram group, while **Figure 5** summarizes the results of all pair-wise comparisons based on the mixed-effect models.

For mobility, pairwise comparisons are almost always significant, especially for weighted (wDC and WI^3) and top-down indices (TL, ss′, and K_{td}) (**Figure 5**). Weighted indices emphasize drifter organisms, while top-down indices draw attention to mobile organisms. This is nicely visible in **Figure 4A** and in the violin plots of **Supplementary Material D**. Centrality indices highlight limited mobility animals. All other groups suggest the importance of drifters. Therefore, depending on what index we utilize, we can predict different groups with the mobility trait. Naturally, a balanced description of a network using one-two indices from each of the six groups is the best. For the mobility trait, groups 3, 4, and 5 are very similar and could be combined in the functional sense (see violin plots in **Supplementary Material D**).

The habitat trait only had two categories and is less useful in predicting the difference between groups (**Figure 4B**). Centrality indices were significantly larger values in the benthic than in the water column habitat (**Figure 5**). The TL, K, and K_{bu} indices were the opposite (benthic < water column). All other indices had no significant difference between habitat preference of the organisms (**Figure 5** and **Supplementary Material D**). It is somewhat difficult to interpret the biological meaning of these results. With too few, or too many categories, it becomes difficult to interpret the results. Simple traits, such as this one could be useful combined with other studies.

The third categorical trait, size had the opposite problem (with having many, eight categories). This trait behaved in a similar manner as the mobility trait (**Figure 4C**). Weighted indices along with the third and fourth index clusters highlight small organisms (0.001–0.1 cm), most likely a reiteration of the drifter mobility category. The keystoneness index (KSI) is not significant in relation to differences in size categories. The centrality cluster seems to favor medium-sized categories (1–10 cm) and top-down indices points out the large-sized groups (>10 cm) (**Figure 4C** and **Supplementary Material D**).

To summarize, mobility was the most reliable trait (>80% pairwise comparisons showing significant differences) and worked best combined with top-down (TL and K_{td}) or weighted indices (wDC). The size trait showed significant differences between 70% of pairs. Finally, habitat trait was only significant about 50% of the time (although works well for all centrality indices) (**Figure 5**). Regarding the relationships between the analyzed traits, all trait-combinations were significantly dependent (Chi-squared test and Fisher's exact test, $p < 0.001$).

CONCLUSION

The major component of sustainability is proper ecosystem functioning and different organisms play their distinct roles in ecosystems. Ecological roles and positions are interdependent, so studying food web position can help to assess functional importance. We addressed the question what kind of organisms (in terms of various traits) occupy what kinds of food web positions (in terms of various centrality indices).

Earlier work on the relationships between food web properties and ecosystem types provided valuable information on the use of indicators at the system level (Heymans et al., 2014). Here, we elaborated this kind of approach at the local level of organisms (trophic groups). The combination of a rich description of network position and the parallel analysis of multiple traits offers a way to improve ecological indication and predictive food web modeling.

For our analysis, it was crucial to set high standards for comparability. The EwE food web database is based on a constant and rigorous modeling approach (similar trophic components across food webs), the way of aggregation is also consistent (stronger at lower levels, e.g., phytoplankton) and mixed-effect models showed that networks (as random effects in the models) had zero or negligible explanatory power due to variance being around zero in most cases (n = 13 indices). The variance due to random effects (networks) was largest for five indices (BC, TO, WO, wDC, and K_{indir}), but still of minor importance (<0.30).

Our findings agree with the suggestions of Costello et al. (2015) that mobility and size should be included in describing aquatic systems. Some of the results are thus quite intuitive (e.g., more mobile organisms at the top of the food web): these are only confirmed and quantified by the present, large-scale statistical analysis. Other results may be more surprising, like the importance of benthic organisms in the food webs. These species or groups of species are fundamental for transferring matter and energy from the sea bottom to the water column through trophic flows contrasting the natural gravity-related flows and thus contributing to the cycling of energy and matter. Quantification and statistical significance are the ways for robust predictions.

Our study connects theoretical, network-based indicators of ecological role (i.e., topological position) and practical, ecologically meaningful categorizations (i.e., traits). Exploring this bridge is essential for giving the appropriate value to theoretical works also in supporting practical applications (Longo et al., 2015). Notably, the importance of such bridge is testified by the large discussions going on for finding the appropriate measures (Tam et al., 2017) to use in evaluating good environmental status for descriptor D4 (food webs) in the Marine Strategy Framework Directive (EU MSFD, 2008).

Certain pairs of centrality indices are consistently similar in different studies. For example, the weighted indices tend to provide similar node ranks (Jordán et al., 2006; Lai et al., 2015) with only a few exceptions (see Jordán et al., 2007). Closeness centrality is less predictable: it can be quite close (Jordán et al., 2006) or quite far (Lai et al., 2015) from degree. The classification depends also on whether it is based on ranks or distributions (Bauer et al., 2010).

It remains important to investigate what other traits are of potential significance in aquatic ecosystems (e.g., diet) and if the index-trait relationships vary by ecosystems (e.g., estuary versus reef). Research in trait-based aquatic food webs is ongoing

(Boukal, 2014) and effort should be made that trait databases are standardized (Kremer et al., 2017) and comparable across environments like freshwater to marine plankton (Weithoff and Beisner, 2019) and scales like megafauna (Tavares et al., 2019). The identification of relevant traits is an ongoing process. Simple, yet descriptive traits (as demonstrated here) can successfully supplement food web research. The choice and the relevance of traits largely depend on the resolution of the food web: for more resoluted networks, a number of traits can be used that make no sense or cannot be obtained for highly aggregated trophic networks. Yet, aggregation and using only the most basic traits make cross-system comparison feasible. Very sophisticated traits cannot be defined for a large number of species, only for a smaller taxonomic neighborhood.

With large databases, both biological information on organisms (e.g., size) and their characterization in a system context (e.g., centrality) can be richly described. Novel algorithms (e.g., machine learning) can further help in the future to provide quantitative analyses and to reveal hidden patterns. This way, trait-based analyses have a chance to offer more than just re-discovering biological knowledge *in silico* (Endrédi et al., 2018a). Combinations of traits, as a major future task, can be more informative than looking at them separately.

Contributing to the predictive power of food web modeling, by combining biological information and systems analysis, may help to understand and support the management of invasive species. Their trophic and other properties are partly known and but can also be adapted to some extent during invasion. The rules and their limits can be better understood by the present research.

Although the database we used is the largest one in community ecology, described by the highest standards for comparability, it is still loaded by the traditional problems of food web research. Aggregation (defining the nodes) and weighting (defining the links) are always problematic. It will be a interesting question for future research, whether and how omics data can provide larger, more reliable information (Lima-Mendez et al., 2015; Guidi et al., 2016; D'Alelio et al., 2019) and whether this can completely replace or only complement the information we have today.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

FJ, AE, and KP designed and evaluated the results, and wrote the manuscript. SL, AE, and KP managed the database. AE performed network analysis. KP and JP performed the statistical analysis. JP and SL provided comments on the manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.636042/full#supplementary-material

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Bycatch Estimates From a Pacific Tuna Longline Fishery Provide a Baseline for Understanding the Long-Term Benefits of a Large, Blue Water Marine Sanctuary

Vanessa Jaiteh[1,2], Tom Peatman[2,3], Steve Lindfield[4], Eric Gilman[5] and Simon Nicol[2,6]*

[1] Ministry of Natural Resources, Environment and Tourism, Koror, Palau, [2] Pacific Community (SPC), Nouméa, New Caledonia, [3] Independent Consultant, London, United Kingdom, [4] Coral Reef Research Foundation, Koror, Palau, [5] The Safina Center, Honolulu, HI, United States, [6] Centre for Conservation Ecology and Genomics, Institute for Applied Ecology, University of Canberra, Bruce, ACT, Australia

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***Correspondence:**
Vanessa Jaiteh
vanessa.jaiteh@outlook.com

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Bycatch on pelagic tuna longlines has contributed to population declines in several far-ranging, oceanic species and presents a conservation challenge that area-based management tools are increasingly promoted to address. In January 2020 the Republic of Palau, concerned about the impacts of longline fishing in its waters, closed 80% of its exclusive economic zone to all extractive activities, reserving the remaining 20% for a domestic fishing zone (DFZ). One of a growing number of very large marine protected areas, the Palau National Marine Sanctuary (PNMS) spans ~500,000 km^2 and was established *inter alia* to allow for the recovery of fish stocks adversely impacted by tuna longline fleets. Given that the main tuna stocks targeted in the western Pacific are not overexploited, the benefits of protection potentially afforded by the sanctuary are likely greater for vulnerable bycatch species. Evaluations of the sanctuary's performance require, in part, a baseline of historical catch rates and effort distribution in the distant-water fleet (DWF) and locally based fleet (LBF) operating in Palau prior to sanctuary implementation. We describe the fishing effort, catch rates, catch estimates and fishing mortality in Palau's longline fishery based on logbook, observer and electronic monitoring data. We defined bycatch as any species, retained or discarded, other than targeted tunas. Between 2010 and 2020, 104.8 million hooks were deployed, catching over 2 million individuals from 117 taxa at an overall target:bycatch ratio of 1:1, with a retention rate of ~62%. Pronounced differences in fishing strategies and spatial distribution of effort between fleets were associated with large variations in catch rates and composition. The LBF had a larger effect on populations of at-risk species relative to the DWF, with higher catch rates and magnitudes for several vulnerable species and higher observable fishing mortality rates (64% vs 50% in the DWF). The sanctuary reshaped Palau's longline fishery, contracting the fishery's area and capacity. The relocation of the DFZ eliminated the LBF and constrained the DWF to an area where the fleet's total catch rates and those of a number of vulnerable species were historically lower relative to former fishing grounds now closed by the sanctuary. Our results highlight the importance of consistent bycatch monitoring and emphasize the need for regional area-based approaches for managing longline fisheries.

Keywords: large scale marine protected area, CPUE, protected species, WCPO, elasmobranchs, marine turtles, access agreement, VLMPA

INTRODUCTION

Bycatch in tuna longline fisheries is an ecological and socioeconomic sustainability issue that is exacerbated by monitoring and management challenges arising from the fisheries' operational characteristics. Targeting highly mobile, far-ranging pelagic fishes, tuna longline vessels often spend weeks or months at sea before returning to port. In addition to fishing under license agreements in coastal states' Exclusive Economic Zones (EEZs), they often fish in Areas Beyond National Jurisdiction (ABNJ; the high seas), resulting in monitoring inefficiencies that impede the verification of fishing practices and self-reported catches in logbook data. An inherently unselective gear, longlines typically have relatively high rates of bycatch in comparison to other fisheries targeting tuna (Hall et al., 2017; Gray and Kennelly, 2018), although soak time, hook shape and size, depth, bait type, and various gear modifications are known to affect catch rate and composition (Bigelow et al., 2006; Clarke et al., 2015; Gilman et al., 2018). Here we define bycatch as the catch of any species, whether retained, released alive or discarded dead, other than the tuna species targeted by the fishery we describe (cf Clarke et al., 2015). This definition comprises by-product, i.e., lower value market species that are typically retained, including non-target members of the family Scombridae, billfishes (Istiophoridae, Xiphiidae) and other teleosts (bony fishes). It also encompasses unmarketable species, and threatened, endangered or protected marine megafauna including elasmobranchs (sharks and rays; some of which are commercially valuable), turtles, seabirds, cetaceans (whales, dolphins, and porpoises) and some teleosts, whose life histories render them vulnerable to fishing pressure. We provide this definition with the caveat that entirely unambiguous definitions of bycatch may not exist, even when applied to only one study or fishery.

Globally, five tuna regional management fisheries organisations (t-RFMOs) assess the status of target and non-target species of tuna fisheries. The reported levels of bycatch, discards and fishing mortality have prompted t-RFMOs to issue mitigation measures intended to improve the ecological sustainability of their longline fisheries, but limited collection and provision of catch data for bycatch taxa often impede their capacity to implement and assess the efficacy of these measures (Gilman et al., 2014; Juan-Jordá et al., 2018). Established in 2004, the Western and Central Pacific Fisheries Commission (WCPFC) is the newest t-RFMO, with the highest number of listed longline vessels (3,766 in 2013). In addition to target tuna species, its mandate encompasses the sustainable use, conservation and management of dependent and associated non-tuna species. This mandate is carried out through several Conservation and Management Measures (CMMs), which direct member states and cooperating non-members to report, for example, interactions with seabirds and sea turtles, and provide catch, effort and size data on 20 key shark species in vessel logbooks. Since 2009, the WCPFC oversees a regional observer program (ROP) which requires 5% observer coverage for all longline fisheries (WCPFC, 2018). Other regulatory measures to reduce bycatch in WCPO longline fisheries include modifications of fishing gear or strategies, retention bans, various measures to mitigate shark finning (WCPFC, 2010) and, on a broader level, fisheries closures (e.g., in high seas pockets). Despite these mitigation measures, sustainability risk analyses and stock assessments for several globally threatened species, including bigeye thresher (*Alopias superciliosus*), blue (*Prionace glauca*), silky (*Carcharhinus falciformis*) and oceanic whitetip sharks (*C. longimanus*), indicate substantial and ongoing population declines that may require more comprehensive measures to complement and strengthen those already in place (Harley and Rice, 2012; Rice et al., 2015; Fu et al., 2017). Worldwide, reported shark landings have declined by 15% since peaking in 2003. Although shark management measures may have played a role in driving these reductions, the more likely causes appear to be declines in abundance and possibly increased underreporting (Davidson et al., 2015; Pacoureau et al., 2021).

Large ($>$10,000 km^2) and very large marine protected areas (VLMPAs, $>$100,000 km^2) are increasingly promoted as a tool in addressing national and international conservation targets. Most VLMPAs were established in the last decade, encompassing approximately 6.5% of the global ocean (Marine Conservation Institute, 2020) and reflecting their growing popularity as a sweeping approach to sustainability issues, including fishing-induced population declines in large bodied, highly mobile marine fauna (Boerder et al., 2019). VLMPAs may protect core habitats or key life stages of highly migratory taxa or offer some respite from overfishing to species that exhibit predictable behaviors, such as philopatric blue (*Prionace glauca*), shortfin mako (*Isurus oxyrinchus*) and common thresher sharks (*Alopias vulpinus*) (Boerder et al., 2019). However, studies of – and empirical evidence for – the efficacy of large and very large marine protected areas (MPAs) in protecting highly mobile, large-bodied pelagic species, including threatened, endangered and protected species, are scarce (Ban et al., 2017; Gilman et al., 2019; Curnick et al., 2020a).

The offshore waters of Palau contain a diversity of far-ranging pelagic species, including tunas, billfishes such as swordfish, spearfish, sailfish, and marlin, elasmobranchs, cetaceans and sea turtles. While small-scale fishing on the archipelago's resource-rich barrier and fringing reefs is an important part of Palauan culture, industrial offshore fishing for tuna was pioneered by the Japanese, who introduced pole-and-line fishing for skipjack tuna (*Katsuwonus pelamis*) to Micronesia during their occupation of Palau in the 1920s (Gillett and Tauati, 2018). Industrial fishing was suspended during World War II and did not resume until 1964, when the US seafood company Van Camp established a transhipment base in Koror (**Figure 1**) to support a locally based pole-and-line fleet. The 1960s also saw the advent of Japanese distant-water tuna longline fishing for yellowfin tuna (*Thunnus albacares*) in Palau's EEZ. This fishery continues to this day, currently supporting a small fleet of around 20 longliners based out of the port of Ishigaki on Okinawa. During the Japanese fishery's presence in Palau's EEZ, its operations underwent two main changes: (1) vessels began targeting the higher value bigeye tuna (*Thunnus obesus*) by setting longlines deeper, and (2) in response to consumer demand for fresh fish over frozen product, smaller vessels started making shorter trips, chilling their catch in refrigerated seawater or brine until they returned to Okinawa (IPNLF, 2019). Beginning in the late 1980s, three longline fishing companies were established: Palau International Traders Incorporated (PITI), Palau Marine Industries Corporation (PMIC; closed in 2008) and Kuniyoshi Fishing Company (KFC). All three companies brought in foreign vessels from Taiwan and/or the People's Republic of China (PRC) to supply fish, nearly all of which was exported to Japan (higher quality product) or Taiwan (lower quality fish). In the last two decades, PITI established itself as the main company, with KFC operating a smaller export business and supplying the local market.

Tuna fishing in Palau is managed at the regional level by the WCPFC and at the sub-regional level by the Parties to the Nauru Agreement (PNA; est. 1992). Additionally, Palau has passed legislation to mitigate the effects of pelagic fishing in its waters. A 2003 Republic of Palau Public Law (Rppl 6-36, 2003) banned wire leaders and the retention of sharks, including their fins, by foreign fishing vessels. In 2009, then-President Toribiong declared his country's waters the world's first shark sanctuary and established Palau as a leader in marine conservation. However, despite being widely cited in the scientific literature (Vianna et al., 2012, 2016; Ward-Paige, 2017; Ward-Paige and Worm, 2017), the shark sanctuary does not have legal status: the Shark Haven Act (Senate Bill 8-105), proposed in 2009, was never adopted.

In 2015 the Olbiil Era Kelulau (OEK; Palau National Congress), concerned over the ecological and socio-economic impacts of foreign fishing activity in its waters, passed the Palau National Marine Sanctuary (PNMS) Act (Rppl 9-49, 2015). It established ~80% (500,000 km^2) of Palau's EEZ as a no-take reserve, banning all extractive activities from 1st January 2020 (**Figure 1A**). The remaining ~20% of the EEZ were declared a domestic fishing zone (DFZ; **Figure 1A**) where fishing by licensed vessels would be allowed, subject to the following specifications: (i) fishing by fishing vessels (the Act's definition of "fishing vessel" excluded most personal fishing boats) was prohibited within a boundary of 12 nm (the territorial seas) from a baseline of each island or island group, and within a 50 nm radius extending eastward from the reef entrance of Malakal fishing port near Koror; (ii) from 1st January 2020, 100% observer coverage would be mandatory for all fishing vessels operating in the DFZ; (iii) fish caught in the DFZ was to be made available for local sale only and its commercial export prohibited, with the exception of free-school purse-seine catches. These were to be landed in Palau before being exported – which, given the lack of purse-seine landing infrastructure at Malakal port, effectively countervailed their exemption to the export ban.

The PNMS Act also gave sweeping protections to sharks through an amendment of § 1204 ("Prohibited Acts") of the Marine Protection Act of 1994, prohibiting any person to fish for, remove the fins of or otherwise intentionally mutilate or injure, or possess any part of any shark within Palau's waters. This new provision, which effectively afforded the shark sanctuary legal standing, was superseded 2 days later by RPPL 9-50, a law regulating reef fish exports which also amended § 1204 of the Marine Protection Act, but without a provision for sharks. This presumably accidental cross-over of the two laws mainly affects sharks within the coastal waters of Palau, meaning they are not legally protected from injury, mutilation or taking through fishing or other means.

Following negotiations with various stakeholders, the OEK amended the PNMS Act in 2019 (RPPL 10-35). The amendments meant that: (i) fish caught on longlines in the DFZ were no longer subject to the export ban; (ii) longline and purse-seine catches could be exempted from the landing requirement through regulations promulgated by the Minister of Natural Resources, Environment and Tourism; (iii) the DFZ was reoriented to the west, bordering a high seas pocket to the northwest of Palau (**Figure 1B**); and (iv) the 50 nm exclusion area was replaced with a 24 nm contiguous zone surrounding the main island group, within which only pole-and-line and small personal vessels may fish, and only for domestic sale. Although this zone forms part of the DFZ, from here on we refer to the DFZ as the zone in which longline fishing is allowed (**Figure 1B**), which coincides with the fishing grounds of the Japanese distant water longline fleet.

Citing economic losses from the reduction in fishing grounds and the re-orientation of the DFZ, the two remaining locally based fishing companies closed down in late 2019, effectively ending locally based longline fishing by Taiwanese and Palau-chartered vessels. Japanese vessels, having historically landed their catch in Okinawa, were exempted from the landing requirement and continue to fish in the DFZ.

The PNMS Act lists the protection of overexploited fishery species as a primary objective in promoting their recovery and reproduction, claiming that "*[c]urrently, Palau's fishing stocks, including tuna and other bycatch, are being depleted by foreign fishing vessels [. . .]*" (Rppl 9-49, 2015). Given that the tuna stocks in the WCPO are not overfished and no overfishing is occurring (Hare et al., 2020), any benefits of protection afforded by the sanctuary are more likely to accrue for at-risk bycatch species. Measuring their responses to spatial protection will require an understanding of previous levels of incidental capture and resulting fishing mortality.

In light of criticisms of large and very large MPAs being politically driven (Leenhardt et al., 2013), with little scientific evidence backing their utility in conserving highly mobile pelagic

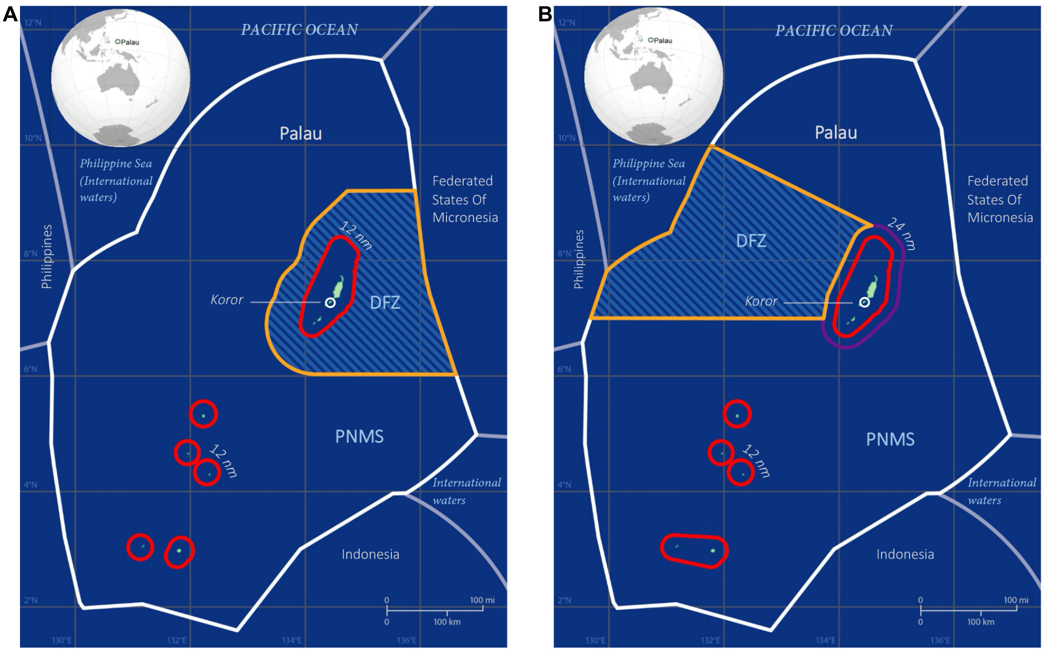

FIGURE 1 | Map of the Palau National Marine Sanctuary (PNMS) and Domestic Fishing Zone (DFZ), **(A)** as designated in the original PNMS Act 2015 and **(B)** as revised in the 2019 amendment to the PNMS Act, and implemented in January 2020. Red lines represent the 12 nm territorial seas boundary. Only pole-and-line and small personal vessels are permitted to fish within the 24 nm contiguous zone boundary (purple) around the main island group, which belongs to the DFZ. The orange boundary line denotes that part of the DFZ where longline fishing is permitted. The PNMS extends from the 12 and 24 nm boundary, respectively, to the edge of the Exclusive Economic Zone (EEZ) (white line). Malakal fishing port is situated near Koror.

fauna (Game et al., 2009; Boerder et al., 2019), we set out to establish a baseline of historical longline effort and catch in Palau's waters to allow for causal inference in future performance assessments of the PNMS. Using three sources of longline catch data we describe, with particular focus on bycatch species, the extent of fishing effort, catch rates, and estimated total bycatch in Palau's EEZ in the decade preceding the implementation of the PNMS. We examine whether bycatch mitigation measures at the regional and national level are reflected in changes in the fishing strategies and catch compositions of longline fleets operating in Palau. Finally, we explore how differences in fishing strategy and fishing grounds between fleets affect species composition, catch rates, and fishing mortality. Based on these findings, we discuss the potential benefits of static spatial protection afforded to fishery-associated species through the PNMS, recognizing that multiple factors will ultimately determine the sanctuary's impact on different species.

MATERIALS AND METHODS

Study Location

The Republic of Palau (hereafter "Palau") is the westernmost archipelago of the Caroline Islands in the western Pacific Ocean (**Figure 1**). Surrounded by an EEZ of 629,000 km^2, Palau's administrative and economic capitals, Ngerulmud and Koror, are located within the country's main island group (~7°N and 134°E). Five small oceanic islands, collectively known as the Southwest Islands, and a coral atoll, Helen Reef, lie 300–500 km southwest of Koror.

In contrast to many of its Micronesian neighbors, Palau was never home to a large purse-seine fishery, probably due to its location at the western limits of the regional fishery and its distance from ports with purse-seine landing facilities. Since the cessation of its pole-and-line fishery in 1982, the majority of tuna catches in Palau's waters have been made by longline fleets from Japan and Taiwan, and – until the early 2000s – Korea and the PRC.

During the time span of this study (2010–2019), the longline fleets that fished in Palau's EEZ consisted primarily of (i) a foreign-owned, distant-water fleet (DWF) of mainly Japan-flagged vessels operating out of Okinawa; (ii) a foreign-owned, locally based fleet (LBF) comprised of Taiwan-flagged, owned and operated vessels, and chartered (Palau-flagged, Taiwan-owned and operated) vessels, which closed down in late 2019; and (iii) a small locally based national fleet, with 1–3 domestically owned vessels licensed to fish since 2017. The main target species of all fleets were yellowfin and bigeye, the vast majority of which were air-freighted fresh to the Japanese sashimi market. Albacore *Thunnus alalunga* and skipjack tuna as secondary target species, and various billfish species were more commonly sold locally or shipped frozen. The LBF and national vessels landed their catch locally, at their base in Malakal harbor, with a small proportion of the catch retained and sold locally. The DWF has operated in Palau's waters under access agreements that exempt it from unloading its catch in Palau before exporting to Japan.

Datasets

We analyzed three non-public domain datasets comprising a decade of logbook data (2010–2019), 17 years of observer data (2003–2019) and 2 years of electronic monitoring data (2016–2018) collected in the national waters of Palau. Logbook data are collected by vessel captains and crew, and include information about the fishing vessel, the positions, timing and gear details of longline sets, and species caught. Although vessel crew are required to record tuna, billfish and shark species in their catch record, logbook reporting of shark catches and other species of scientific interest is often incomplete and may not be provided at a species-level (Rice, 2018). Logbook data are, however, the most complete source for calculating total fishing effort, and were used in conjunction with the other datasets to estimate total catch levels.

Human observers have been deployed on longline vessels fishing in Palau since the 1980s (WCPFC, 2017), initially through a Pacific Islands observer program that was later incorporated into the WCPFC's ROP, established in 2007. With an overall historical coverage of <1%, longline fleets within the convention area were required to increase their observer coverage to 5% as of June 2012. While compliance with this requirement has improved in recent years, it has not been met in some parts of the convention area, including Palau's LBF, in recent years (Peatman and Nicol, 2020; Williams and Ruaia, 2020).

In 2016, the Government of Palau agreed to trial an electronic monitoring system to assess its utility and potential to complement and augment observer coverage through the ROP. Three DWF and four LBF vessels participated in the trial, which ran until the end of 2018 and was coordinated by The Nature Conservancy, with Satlink providing technical services. Of 375 sets completed during 54 trips with EM, 261 sets from 39 trips were reviewed. The video footage collected during the trial was reviewed by fisheries observers in Palau (for DWF footage) and by scientific observers at Digital Observer Services (DOS) in Spain (for LBF footage).

The following links provide descriptions of the data collection forms and information fields for logbook, observer and EM data. The curator of these and other regional fisheries data is the scientific services provider and data manager of the WCPFC, the Oceanic Fisheries Programme of the Pacific Community (SPC).

Fishing Effort and Catch Composition

Due to the spatial overlap in their fishing effort and other fishery-specific similarities, the national fleet and the LBF were combined (hereafter collectively termed the LBF) for all analyses. Fishing effort was estimated from the entire available record of logbook data for Palau's EEZ. Throughout the paper we refer to reported (recorded in logbooks) vs. observed (recorded by human observers) effort and catch, respectively. Where observed catch includes EM data, this is noted. It was not possible to include time of day in this or other analyses, because set times in coordinated universal time (UTC) are not a minimum standard data field in the WCPFC ROP and observers often use vessel time (as opposed to local time).

To compare the catch composition between the three datasets, we calculated the percentage contribution of each species or species group to the total catch recorded in each data source. Teleosts with <10 records were excluded (**Supplementary Table 5**). For plotting, we selected species that contributed to at least 99% of the observations within each species grouping, with the exception of teleost fishes where we selected the top 80% of species. Two shark genera, thresher (*Alopias* spp.) and mako sharks (*Isurus* spp.), were pooled to the genus level because they were variously identified at species and genus groupings depending on the data source.

Definitions

Throughout this paper, the term "sharks" includes all sharks and rays, and "tunas" includes the four main target species (yellowfin, bigeye, skipjack, and albacore tuna) and seven additional species from the family Scombridae, unless stated otherwise. The 20 WCPFC "key shark" species are blue, oceanic whitetip, mako (two species), thresher (three species), silky, porbeagle, hammerhead (four species) and whale sharks, and mobulid rays (six species) (WCPFC, 2019a,b). The WCPFC lists marine turtles, seabirds, marine mammals, and key shark species as Species of Special Interest (SSI). We also added pelagic stingrays *Pteroplatytrygon violacea* to the SSI category in our analyses of bycatch condition. Given a lack of regional red list assessments for many species, we define species of conservation concern as any species classified within one of the IUCN Red List's threatened categories (VU, EN, and CR).

Catch Estimates

Total catch estimates were obtained using a stratified ratio estimation approach (Cochran, 1963). First, observer data were used to estimate catch per unit of effort (CPUE; the number of individuals caught per thousand hooks). Since longline observers record catch data specific to individuals, we used numbers of individuals as the unit for estimating catches. We did not convert catch numbers to weight to obtain biomass estimates. Given that only a small proportion of individuals were weighed or measured, this additional step would have likely rendered catch weight estimates less reliable than estimated catch numbers (Peatman et al., 2018). Data were then stratified by fleet, i.e., the LBF and DWF were separated to account for variation in catch rates and catch compositions due to fishing strategy. Additionally, data for the DWF were further stratified for fishing events inside the area that is now the DFZ vs. the PNMS to account for any spatial variation in catch rates. CPUE was estimated for each stratum, species code and individuals' fate, i.e., retained or discarded. These strata-specific estimates of catch rates were then applied to total reported effort in each stratum, to obtain estimates of total catch specific to each species code and fate. Higher-level estimates were then obtained by summing across species codes, e.g., to obtain total catch estimates of sharks. Estimates of uncertainty in catches and catch rates were obtained using a non-parametric bootstrap procedure, by first resampling at random from observer trips, and then for each trip resampling from observed sets. This approach was used as fishing events from the same trip are unlikely to be independent. We used 1,000 bootstrap replicates and obtained 95% confidence intervals using the 2.5 and 97.5% quantiles. Observer data from 2007 to 2019 were used to generate catch rate estimates. Observer data from earlier years were excluded due to differences in domestic regulations on shark retention. Available EM data were not used to estimate catch rates, as exploratory data analysis revealed lower taxonomic resolution in the EM dataset, particularly for shark species which were often only identified to genus or higher levels. There were insufficient observed sets to estimate catch rates inside the DFZ for the LBF, which has historically expended relatively little effort within that area (**Figure 2**).

Catch Clustering

To assess the extent to which species compositions have varied through time, we applied k-means clustering to catch compositions from both logbook and the combined observer and EM dataset. The clustering analysis was applied to catch proportions by number at a fishing trip resolution, with the number of clusters set at the point of inflection in variance explained as the number of clusters is increased. Trips with limited numbers of sets were excluded, i.e., logbook data with three sets or less, and observer trips with only one observed set. The clustering analysis of logbook data was applied to catch proportions of three species categories – bigeye, yellowfin, and total billfish catch. It was not possible to include shark catches in the logbook analysis, as these were not reported for all trips. The clustering analysis of observer data was applied to catch proportions of five categories – bigeye, yellowfin, total billfish catch, pelagic stingray, and total shark catch. We used the number of hooks between consecutive floats (HBF) as a proxy for relative gear depth, which can have a substantial impact on species' catch rates and therefore, catch composition.

Catch Rate Models

Species-specific catch rate models were constructed using the R package mgcv (Wood, 2011), focusing on elasmobranch species that were observed in sufficient numbers to allow robust statistical modeling: pelagic stingrays, blue (*Prionace glauca*) and silky sharks (*Carcharhinus falciformis*). We were specifically interested in spatial, temporal and fleet effects on catch rates, e.g., seasonal trends, differing catch rates across Palau's EEZ, and the effects of fishing strategy, particularly following the implementation of bycatch reduction measures. A negative-binomial likelihood was used, with a log link function. The response variable was numbers caught, and the natural log of observed hooks was included as an offset. Explanatory variables included in the model were: year, to account for temporal variation; month, to account for seasonal variation; fleet, to account for differences in fishing strategies; and a 2D gaussian process with a Matern covariance function, to account for spatial variation. Splines were used to account for potentially non-linear relationships between catch rates, and year and month. The model was:

$$E\left[Y_i\right] = \mu_i$$

$$Var\left[Y_i\right] = \mu_i + \frac{{\mu_i}^2}{\theta}$$

$$\ln \mu_i = \ln\left(thooks_i\right) + \beta_0 + fleet_i + f\left(year_i\right) + g\left(month_i\right) + h\left(lat_i,\ lon_i\right)$$

where Y_i denotes observed bycatch rate (individuals per thousand hooks), subscript i refers to set id, *fleet* is a categorical variable for the LBF and DWF, function f represents a thin plate regression spline, function g represents a cyclic cubic regression spline, function h represents the 2D gaussian process, and θ is an overdispersion parameter. All explanatory variables were included in each catch rate model. Models were fitted to observer data only, as exploratory data analysis suggested lower rates of species-specific catch records in the EM dataset for shark species. It was not possible to include HBF in the catch rate models along with fleet effects as the two variables were highly correlated.

RESULTS

Fishing Effort

Between 2010 and 2020, a reported 104′814′811 hooks were deployed during 70′959 longline sets by 183 vessels fishing in Palau's EEZ. Observer data were available for 980 (1.4%) of these sets, with 1,545,100 hooks observed. EM data were available from an additional 306 sets, with 458,100 hooks observed. Observations from the DWF accounted for 76 and 18% of the available observer and EM data, respectively. There was a marked difference in both the spatial distribution and the amount of fishing effort expended by the DWF and the LBF (**Figure 2**). The LBF expended 72% of the two fleets' combined effort and operated primarily to the south-east of Palau's main island group, in the area that was originally designated as the DFZ (Rppl 9-49, 2015; **Figure 2**), and east of Hatohobei State in the EEZ's south-west (**Figures 2A,B**). The DWF accounted for 28% of total effort and had a broader distribution along the western half of the EEZ. This included the area now encompassed by the DFZ (**Figures 2A,C**), where the DWF expended 29% of its effort, which accounted for two-thirds of the total effort in that area.

The two fleets also differed in terms of the depth at which they fished. Almost all observed effort (72% of reported effort) of the LBF were shallow sets with four to eight HBF (∼50–200 m), while 98% of the reported effort and all observed effort of the DWF were deep sets (20–24 HBF, ∼450–600 m). Furthermore, the majority of the LBF's observed effort was from pre-2015, whereas all observed effort of the DWF was from 2015 onward. The EM and observer datasets were imbalanced with respect to temporal and spatial coverage, with relatively limited overlap between the two.

Species Composition

A total of 117 taxa (species level or higher) were recorded in the longline catch of Palau, with the highest number (101) recorded by human observers, followed by EM (65) and logbooks (41) (**Supplementary Table 5**). Reported catches and species-level identification across all species except target tunas were

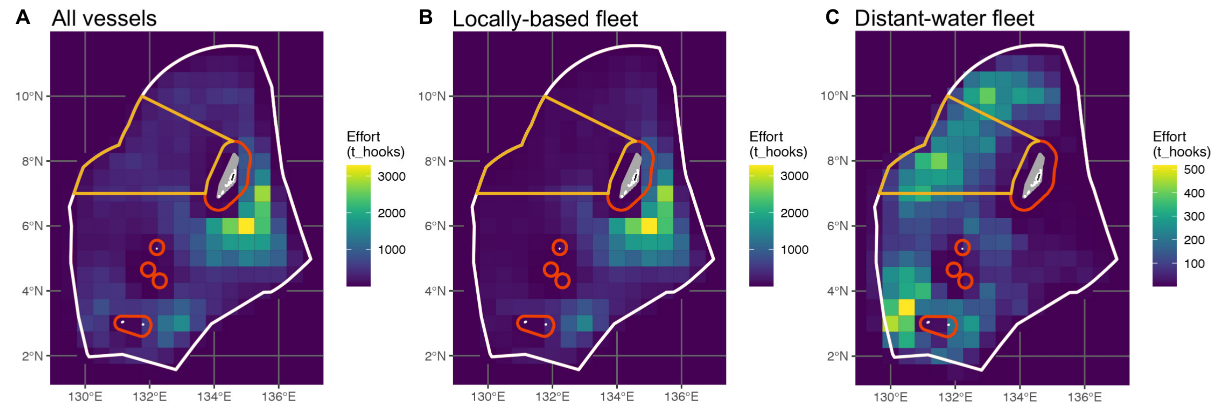

FIGURE 2 | Total fishing effort in Palau's EEZ from 2010 to 2019, expended by panel **(A)** all vessels combined, **(B)** vessels of the locally based fleet (LBF), and **(C)** vessels of the distant-water fleet (DWF; note different effort scale). Colored lines denote the boundaries of the DFZ (orange) as it pertains to longline fishing, and the PNMS (red to white). Only local fishing for domestic consumption is permitted within the red 12 and 24 nm boundaries. Note that the DFZ and PNMS were implemented after the timeframe of this study; their boundaries are shown here to illustrate subsequent results and their discussion.

proportionally lower in logbook data than those recorded by observers and EM (**Figure 3** and **Supplementary Table 5**). Comparing the EM and observer records, the pelagic stingray *Pteroplatytrygon violacea* was the most abundant bycatch species and the third most frequently recorded species in both datasets, although its percentage contribution to total catch was higher in the EM reported catch (**Figure 3**). Conversely, three of the key shark species, blue *Prionace glauca*, silky *Carcharhinus falciformis* and thresher sharks (*Alopias* spp.) were recorded in higher proportions by observers. The resolution of species identifications was lower in EM data, in particular for species that were predominantly not retained, with individuals more frequently identified to higher levels, e.g., "Carcharhinidae" for sharks and "Unknown teleost" for a majority of finfish. Differences were also pronounced for turtles, where olive ridley turtles *Lepidochelys olivacea* were the dominant species recorded by observers, while the majority of turtles recorded on EM were placed in the reptilian order Testudinata.

Catch Estimates

We estimated total catch of all species across both fleets between 2010 and 2020 at 2′122′279 individuals, of which 51% were tunas and the remainder was bycatch, equating to a target:bycatch ratio of ∼1:1. The ratio of retained:discarded individuals was 1.55:1, with an estimated 62% of all caught individuals being retained. Catch estimates based on logbook records accounted for 79% of the target catch and 8% of the bycatch estimated from observer records.

The difference between the estimated annual catch of the four main tunas ($n = 107'786$) and all tunas combined ($n = 108'800$) was 1,014 individuals. Of all bycatch species caught, just under half (49%) were SSI or of conservation concern. Sharks were the most abundant bycatch species group, with 48,400 individuals estimated to be caught annually, followed by the "other finfish" group (n = 32,600), billfish (n = 20,450), turtles (n = 2,350), marine mammals (n = 43) and seabirds (n = 34; **Table 1**).

Catch rates demonstrated strong between-fleet variation at both a species group and species level. Overall catch rates were lower for the LBF than for the DWF, with the exception of billfish and sea turtles (**Table 1**). Tunas were caught at 20.3 vs. 6.5 individuals per 1,000 hooks in the DWF and LBF, respectively. Compared to the target:bycatch ratio of the DWF for sharks (4:1), that of the LBF was nearly three times lower (1.5:1). For turtles, the LBF's ratio (22:1) was 27 times lower than in the DWF (592:1; **Table 1**). The retained:discarded ratio for the LBF (1.8:1) was higher than for the DWF (1.5:1), with retained proportions of 64.4% (95% CI 59.3 – 69.2%) and 59.5 % (95% CI 56.0 – 63.1%), respectively. The ratios of SSI to other species was higher for the LBF (0.23:1) than the DWF (0.1:1). Of the discarded portion of catch, the ratio of SSI to other species was 1.1:1 for the LBF, and 0.3:1 for the DWF.

Catch rates of pelagic stingray, blue shark, and thresher shark species were higher for the DWF, whereas catch rates of silky sharks were higher for the LBF (**Figure 4A** and **Supplementary Table 2**). The LBF had lower catch rates of yellowfin and bigeye tuna; swordfish catch rates were comparable between the fleets.

The DWF, despite expending only a third of the LBF's effort and operating smaller vessels, caught over twice the estimated annual number of bigeye tuna ($n = 28{,}900$), and higher numbers of most sharks, rays and several teleost species (**Figure 4B** and **Supplementary Table 3**). The LBF's higher effort was reflected in nearly twice the catch of pelagic stingrays, six times more billfish and 25 times more olive ridley turtles compared to the DWF. Estimated annual catches of silky sharks ($n = 13{,}200$), the third-most caught species in the LBF (after yellowfin, $n = 31{,}900$ and bigeye tuna, n = 13,300), were 11 times lower in the DWF ($n = 1{,}200$; **Supplementary Table 3**).

The majority of estimated retained individuals was accounted for by scombrid and billfish species. Approximately one-third of the catch of other teleost species was retained, including escolar *Lepidocybium flavobrunneum*, mahi-mahi *Coryphaena hippurus*, and great barracuda *Sphyraena barracuda*. The majority of the

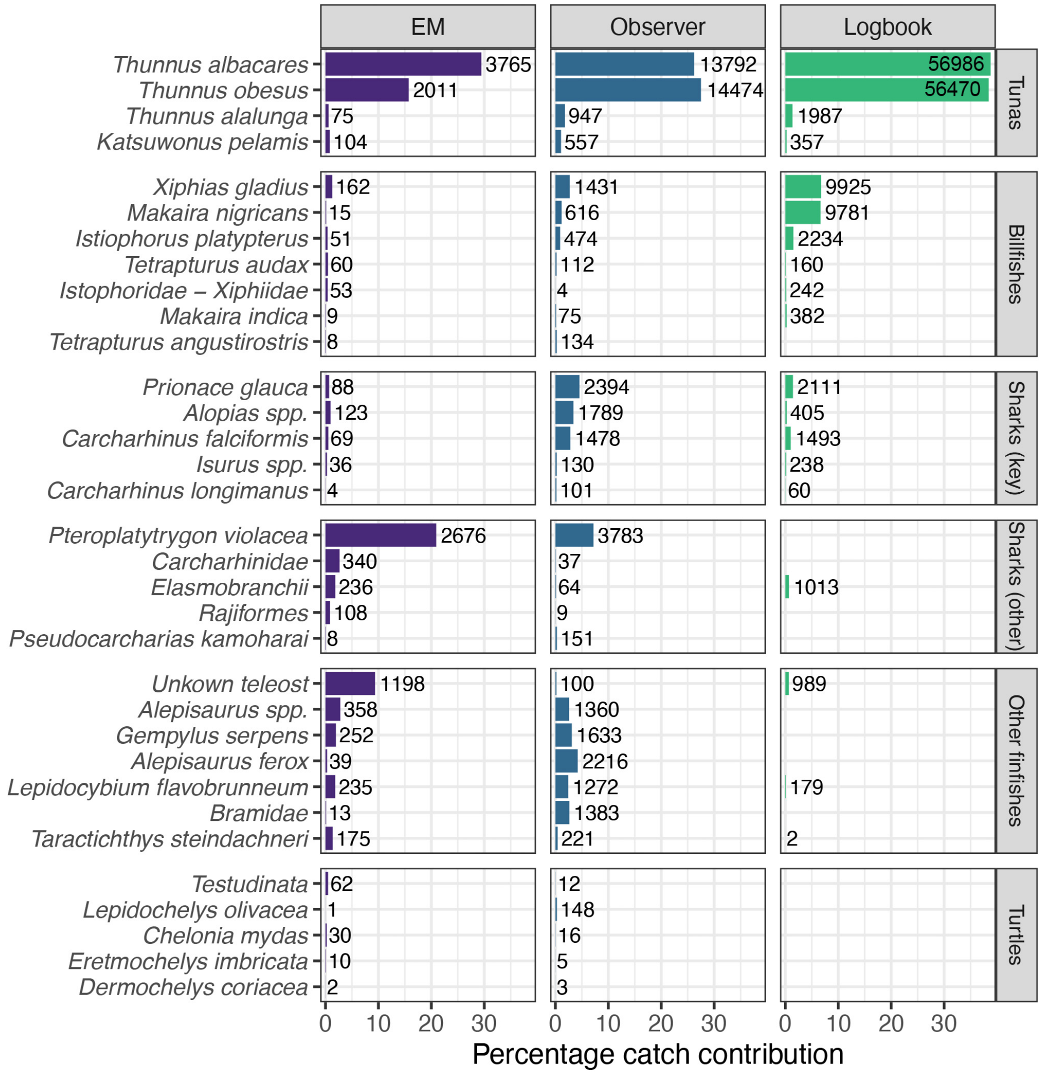

FIGURE 3 | Percentage contribution of target and bycatch species observations to total catch (number of individuals) as recorded in each data source [Electronic Monitoring (EM), Observer and Logbook]. Number labels next to each record show the total number of observations recorded from 2016 to 2018 (EM) and 2010–2019 (Observer and Logbook). Figure shows a subset of selected species (see Methods for selection criteria; note varying resolution of taxa). Key sharks as identified by the Western and Central Pacific Fisheries Commission (WCPFC).

retained catches of these teleosts was accounted for by the LBF, with minimal rates of retention for the DWF.

Sufficient observer data were available (2015–2019) to stratify the DWF catch rates spatially, i.e., inside and outside of what now constitutes the DFZ. Target:bycatch ratios for the DWF were lower inside the DFZ (∼1.2:1) than outside (∼1.8:1), predominantly driven by lower catch rates of yellowfin tuna inside the DFZ (see **Supplementary Table 4**). Retained:discarded ratios for the DWF were also lower inside the DFZ (∼1.2:1) than outside (∼1.5:1). SSI overall accounted for a higher proportion of total catch (0.15:1) and discarded catch (0.43:1) inside the DFZ compared to outside (0.09:1 and 0.26:1, respectively). Although catch rates of most species were lower in the DFZ, notably those of bigeye and yellowfin tuna, pelagic stingray and silky shark, the catch rates of thresher sharks (*Alopias* spp.) were almost twice as high inside the DFZ than outside (**Figure 5** and **Supplementary Table 4**). Catch rates of blue shark were comparable inside and outside the DFZ (**Figure 5** and **Supplementary Table 4**).

Catch Clustering

Catch clustering of observer data identified six clusters (**Supplementary Figure 2**). Yellowfin, as well as shark and billfish-dominated clusters tended to have fewer HBF, indicating shallower sets, while clusters with high proportions of bigeye tuna tended to reflect deeper sets with more HBF (**Supplementary Figure 2**). This was also reflected in assigned clusters for the different fleets, with the DWF having a higher observed effort assigned to target tuna clusters than the LBF.

TABLE 1 | Estimated catch rates (numbers per '000 hooks, 95% confidence interval (CI) in parentheses) and annual catch (numbers; 95% CI in parentheses) for the locally based and distant-water longline fleets by species group, based on observer data from 2007 to 2019.

	Estimated catch rates (CPUE)				Estimated annual catch (number)			
Species group	**Locally based fleet**		**Distant-water fleet**		**Locally based fleet**		**Distant-water fleet**	
Tunas	6.47	(5.12–7.82)	20.3	(17.5–23.5)	48,600	(38,400–58,700)	60,200	(51,900–69,800)
Billfish	2.33	(1.91–2.83)	0.994	(0.853–1.15)	17,500	(14,300–21,300)	2,950	(2,530–3,420)
Sharks	4.3	(3.57–5.15)	5.42	(4.49–6.51)	32,300	(26,800–38,700)	16,100	(13,300–19,300)
Other finfish	2.02	(1.60–2.52)	5.89	(5.19–6.62)	15,100	(12,000–18,900)	17,500	(15,400–19,700)
Turtles	0.299	(0.193–0.424)	0.0336	(0.018–0.051)	2,250	(1,450–3,190)	99.8	(53.6–152)
Seabirds	0.004	(0–0.016)	0.001	(0–0.005)	31.2	(0–120)	3.05	(0–14.4)
Mammals	0.004	(0–0.014)	0.004	(0–0.010)	30.6	(0–103)	11.9	(0–29.7)

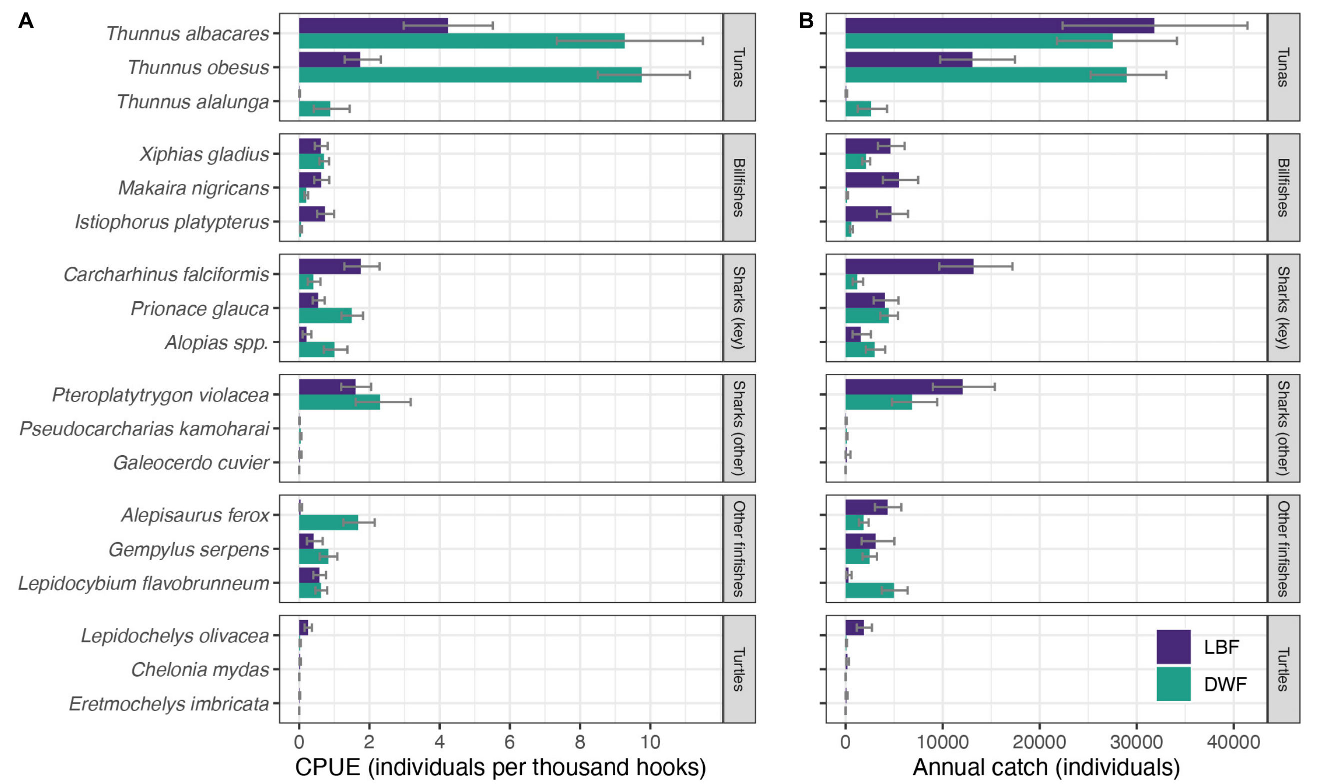

FIGURE 4 | Estimated **(A)** catch rates (individuals per thousand hooks) and **(B)** mean annual catch (individuals) of selected species for the LBF (purple) and the DWF (green), based on observer data collected from 2007 to 2019. Error bars represent 95% confidence intervals (CI).

There was also some suggestion of temporal trends in clusters for the LBF; a mean proportion of 0.42 of annual effort was assigned to the "SHK-BIL" and "BIL-SHK" clusters from 2004 to 2015, compared with 0.15 for 2016–2018. This suggests a shift away from sharks in the LBF's catch composition, though the limited number of observer trips for which data are available made it difficult to pinpoint the timing and extent of this apparent reduction in shark catches. It was not possible to assess temporal trends in assigned clusters for the DWF due to the relatively short time series of available observer data (2015–2019).

Catch clustering of logbook data identified four clusters and suggested that in 2014, the LBF's catch composition saw a further change with a shift from bigeye to yellowfin tuna. Sharks were not included when assigning logbook catch composition clusters due to very limited reported shark catch by both fleets.

Bycatch Condition

In both fleets, the majority of bycatch was discarded (dead, dying, healthy/injured, or in unknown condition based on a visual assessment by the observer), although billfishes and several other species of finfish were generally retained. The LBF discarded a substantially higher proportion of dead SSI (56%) than the DWF (23%), although this difference was less pronounced when dying individuals (6% vs. 27% in the LBF and DWF, respectively) were assumed not to have recovered, which would equate to a 62% and 50% mortality in SSI caught in the LBF and DWF,

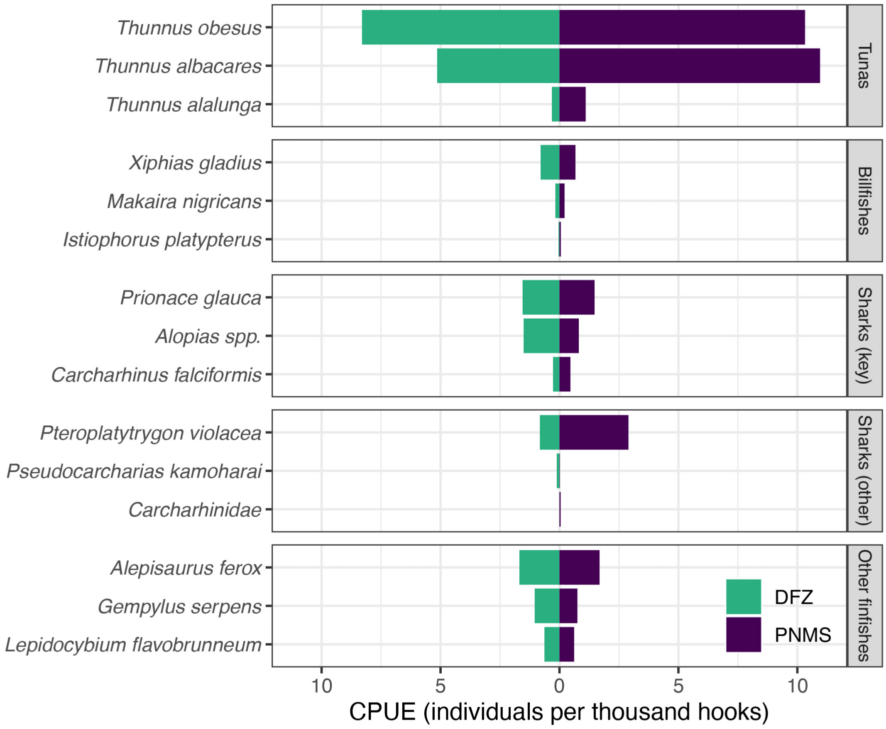

FIGURE 5 | Estimated catch rates catch per unit of effort (CPUE) of selected species caught by the DWF operating in what is now the DFZ (green) and the PNMS (purple). Catch rates were estimated using observer data collected from 2015 to 2019, i.e., the time between the signing into law of the PNMS Act (October 2015) and its full implementation (January 2020). Estimates for additional species are provided in **Supplementary Table 4**.

respectively (**Figure 6**). Furthermore, the DWF discarded a substantially higher proportion (although much lower number) of dead sea turtles than the LBF. The majority of pelagic stingrays, comprising the most frequently caught SSI, were discarded dead in the LBF (71%), and either dead (11%) or dying (58%) in the DWF (**Figure 6**).

Across all species and especially for SSI, observers recorded higher proportions of retained individuals and unknown fate outcomes than EM analysts, who recorded a higher proportion of discards. Individuals' condition at haulback and release also differed between the two datasets, with EM generally noting more "unknown" incidents (**Supplementary Figure 1**).

Catch Rate Models

Visual examination of quantile residuals did not suggest violation of assumed error distributions. Estimates of fleet effects were relatively imprecise for all models, which may result from the relatively distinct areas of operation for the different fleets (**Figure 2**). Chi squared statistics and approximate p-values are provided in **Supplementary Table 1**. All terms were significant except for the fleet effect for the pelagic stingray.

The DFZ was associated with lower catch rates of pelagic stingrays (**Figure 7**). No significant difference in pelagic stingray catch rates was detected between fleets ($p = 0.62$). The year effect for pelagic stingray catch rates demonstrated an increasing trend through time, though with variation. The month effect for pelagic stingray catch rates was complex and highly non-linear.

The spatial effect for blue shark catch rates demonstrated a generally increasing trend northward, though with an area of higher catch rates south of the DFZ (**Figure 7**). Blue shark catch rates for the DWF were estimated to be higher than for the LBF. The year effect for blue shark catch rates demonstrated a strong decline from 2004 to 2007 followed by a weaker increase from 2008 through to 2019, with no observer coverage available for 2005 and 2006. Blue shark catch rates were estimated to increase from November through to May, then decrease through to October.

The spatial effect for silky shark catch rates had lower catch rates at the latitudinal limits of Palau's EEZ, with catch rates declining from 8°N northward and 4°N southward (**Figure 7**). Silky shark catch rates were significantly higher for the LBF than the DWF. The year effect for silky shark catch rates was relatively imprecise from 2005 throughto 2015, with an increasing trend through time from 2015 onward. The month effect for silky shark was relatively imprecise, though catch rates were estimated to increase from March throughto September, before decreasing through to March.

DISCUSSION

The goal of this study was to establish a baseline of historical catches and catch rates, species composition, and effort distribution of longline fleets operating in the waters

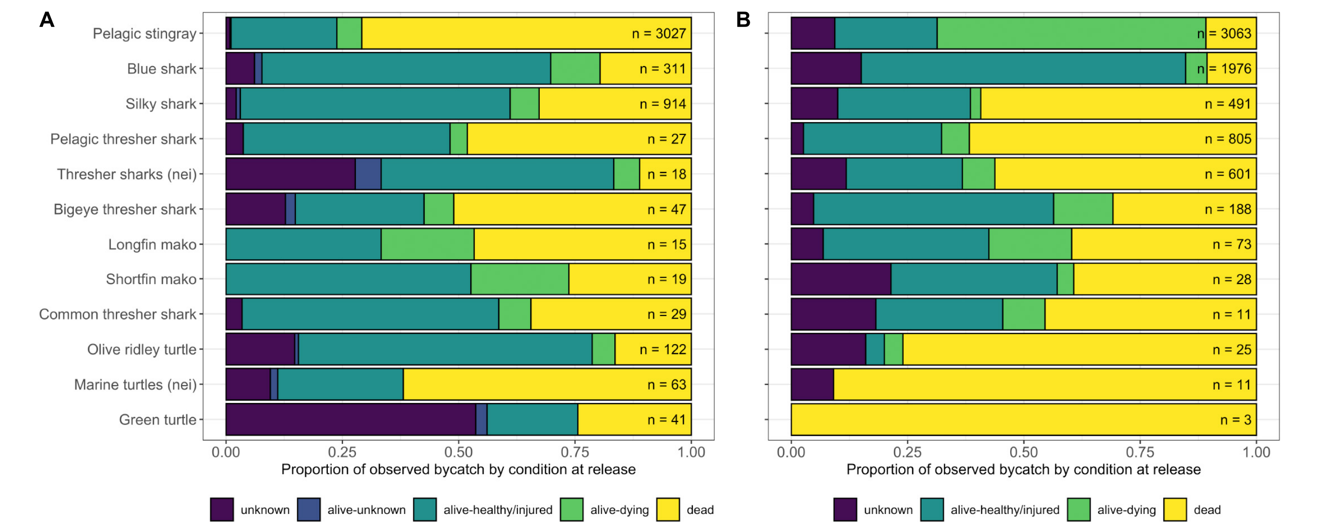

FIGURE 6 | Condition at release of selected Species of Special Interest (SSI) as a proportion of total observed catch (*n* =) of each species or taxon as recorded by observers in panel **(A)** the LBF and **(B)** the DWF. Species with at least 40 records were included. Nei (not elsewhere indicated) was used for individuals that were not identified to species level.

of Palau in the decade preceding the implementation of the PNMS in January 2020. This baseline is primarily intended to support future evaluations of the sanctuary's ecological performance. From a total longline effort of 104.8 million hooks deployed in Palau's waters between 2010 and 2020, 2′122′279 individuals from 117 taxa were estimated to have been caught. Target tuna species constituted 51% of the catch, resulting in a target:bycatch ratio of 1:1. The ratio of retained to non-retained catch for the combined fleets was 1.55:1, reflecting mainly the retention of billfishes as a seasonal byproduct in the LBF, but not the DWF. In terms of annual estimated catch, pelagic stingrays were the most frequently caught bycatch species (n = 18,890), while key sharks were the most abundant bycatch group (n = 48,400). With the exception of billfish and some finfishes, most bycatch was discarded, reflecting the generally high discard rates of pelagic longline fisheries: globally, longline fisheries contribute the majority (64%) of all tuna fishery discards, with discard rates of up to 40% of their catch in numbers (Gilman et al., 2017). In the LBF and DWF, respectively, 62% and 50% of all SSI were discarded dead or dying, with unknown levels of post-release mortality for individuals released alive and healthy or injured (but see Musyl and Gilman, 2018). Although Palau had implemented bycatch measures since 2003, this rate of mortality suggests that the impacts of fishing may have been continuing on vulnerable SSI populations, supporting the decision to implement the PNMS for species conservation.

In light of these findings and the ecological expectations of the PNMS, the question arises whether the sanctuary, which bans fishing in 80% (~500,000 km^2) of Palau's EEZ, is likely to provide conservation benefits to target and/or bycatch species. To answer this question, future assessments will be able to build on this baseline in part with data collected from fishing vessels operating in what is now the DFZ. However, given the historical differences in catch rates, species composition and spatial overlap between the LBF and DWF, future assessments of the PNMS may benefit, alongside other methods of census, from dedicated research fishing trips by the LBF in areas of the PNMS where fishing effort was once concentrated. Alternatively, a counterfactual approach could be used to assess the responses of various species to the PNMS by predicting what the LBF's catch would have been, had the designation of the sanctuary and the re-location of the DFZ not eliminated its fishing grounds (Gilman et al., 2020).

Fleet and Spatial Effects

We found strong between-fleet variation in catch rates and species composition, which were mostly explained by differences in fishing strategies and spatial distribution of effort. This implies that both how and where vessels fish matters in terms of fishery interactions with species. A key difference in fishing strategy between the two fleets was the depth at which their gear was set, whereby the LBF tended to fish in shallower waters (HBFs normally 4–8) while the DWF consistently set its gear deeper (20–24 HBF). While we could not include time of set (in UTC) in our analyses, an earlier assessment of the LBF's fishing strategy demonstrated that the locally based Taiwanese vessels used two strategies with different times of day, which also differed slightly in fishing depth (Gilman et al., 2015). Overall, these findings imply that differences in catch composition between fleets were primarily explained by differences in fishing strategy, rather than differences in the spatial distribution of their respective fishing effort. Nevertheless, it was not possible to clearly discern the effects of fishing strategy – in particular HBF – from spatial effects. For instance, the LBF caught 25 times more olive ridley turtles compared to the DWF. This difference could have arisen from an overlap of the fleet's preferred fishing grounds with important habitat for olive ridley turtles, or from the LBF's shallow sets. The latter may provide the more likely explanation,

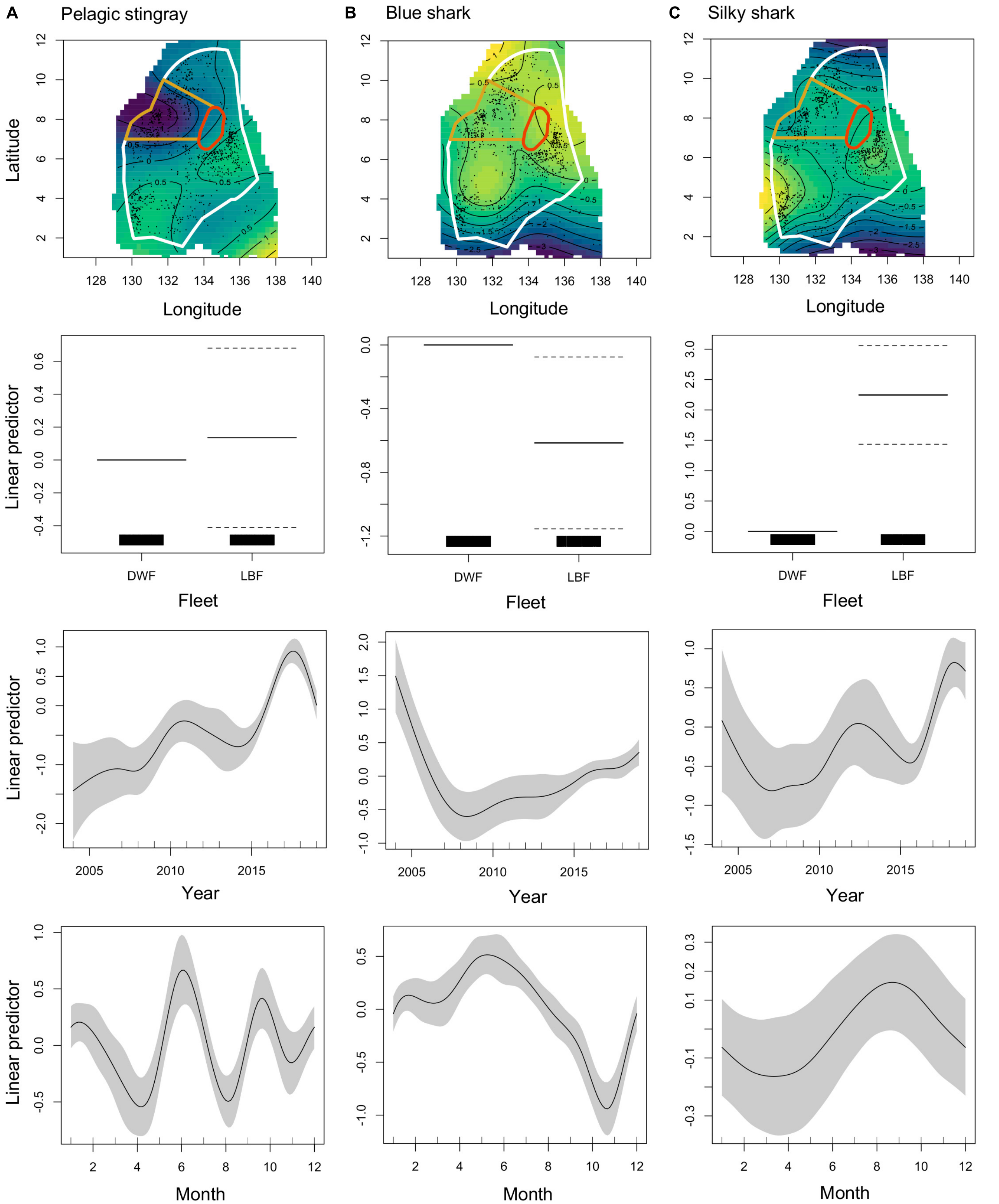

FIGURE 7 | Effect plots of the catch rate models for panel **(A)** pelagic stingray *(Pteroplatytrygon violacea)*, **(B)** blue shark *(Prionace glauca)* and **(C)** silky shark *(Carcharhinus falciformis)*. Darker and lighter colors on the maps (top row) indicate lower and higher catch rates, respectively. Black lines on the maps indicate contours of the spatial surface, and small black dots represent observations. In the second row, solid lines and dashed lines represent the parameter estimate and its standard error, respectively. For the year and month effects (bottom two rows), gray contours are provided for 95% confidence intervals (CI). Note that *y*-axis scale varies across plots.

given analyses of the effects of hook position on catch rates have shown that some species, including turtles and silky sharks, tend to be caught on the hooks nearest to the float (Watson and Bigelow, 2014; Huang et al., 2016).

Across all species, catch rates of the DWF were higher than those of the LBF, but lower in the DFZ than in the area that now constitutes the PNMS. In contrast, overall SSI catch was higher in the DFZ than outside, although this varied at the species-level: modeled catch rates for blue sharks, although higher in the DWF than the LBF, were also higher to the north, south and northwest of the DFZ, which suggests that the current ban on fishing in those areas could be beneficial to this species. Catch rates of vulnerable thresher sharks were almost twice as high inside the DFZ than outside, while those of several other at-risk species including endangered turtles and vulnerable silky sharks were lower in the DFZ. Likewise, catch rates for pelagic stingrays were lower in the DFZ. Although this species was not treated as an SSI in the catch rate analysis due to its large contribution to total catch, its risk status seems to warrant closer examination (see next paragraph). It is unclear whether the lower estimated catch rates of pelagic stingrays in the DFZ compared to the remaining EEZ are indicative of naturally lower abundance of this species, or of fleet- specific fishing strategies in that area.

Catches of target species were also lower in the DFZ. This could eventually result in intensified fishing effort or prompt political pressure to open a larger area for industrial fishing in Palau, potentially increasing risks to vulnerable bycatch species. However, at least in the short term these risks are outweighed by the benefits accrued to SSI through the re-orientation of the DFZ from east- to west-facing in 2019, which presented an unacceptable economic loss for the LBF and led to its departure. The LBF's catch rate for SSI relative to other species, as well as its shark:target and turtle:target ratios were substantially higher than that of the DWF. SSI also constituted a nearly four times higher proportion of discards in the LBF than the DWF, and the estimated annual catches of some of these species, such as silky sharks, olive ridley and green turtles, were magnitudes higher than those of the DWF. This suggests that the current location of the DFZ – if only by the fact that it effected the departure of the LBF – may offer a higher level of protection than if the DFZ had remained in its original location (it is assumed that in this case, the LBF would have remained in Palau, at least in reduced capacity). Compared to the LBF, the DWF was characterized by generally lower proportions of dead discards, particularly of SSI. This may be due to differences in bycatch handling techniques, depth, time of day or other fishing strategies (Poisson et al., 2019). Though more likely to be an unintended effect of techniques aimed at maximizing target catch rather than an attempt at lowering bycatch mortality rates, this characteristic of the DWF could be beneficial for bycatch species still at risk of fishing mortality in the DFZ. By example, pelagic stingrays are a widely distributed species and a common bycatch component of longline fisheries (Mollet, 2002). Although pelagic stingrays have long been regarded as a low risk species based on low mortality rates (Cortés et al., 2010), recent risk assessments of the effects of pelagic longline fisheries assign high relative risk to this species (Gilman et al., 2021). We treated this species as an SSI in our bycatch condition analysis, given the increasing trend in catch rates over the last 15 years indicated by the catch rate model, and their high mortality levels: in the DWF and LBF, respectively, 69 and 75% of discarded individuals of this species were dead or dying. Sea turtles were the exception to the fleet trend described here: they were caught at much lower rates, but discarded dead at substantially higher rates, in the DWF compared to the LBF. These results are consistent with those of an earlier study, where changing from shallow to deep sets was shown to reduce catch rates, but increase haulback mortality rates, for turtles (Gilman et al., 2015). Finally, given that the majority of observed sets in both fleets used Japanese tuna hooks, turtles might benefit from the use of circle hooks.

Effectiveness of Bycatch Mitigation Measures

Broadly, fisheries management agencies have been slow to implement effective bycatch mitigation practices, presumably due to a lack of political will (Soykan et al., 2008; Gilman and Lundin, 2010). We asked whether bycatch mitigation measures implemented at the national or regional level produced distinct changes in catch composition, and identified changes in the LBF on two occasions: first, a distinct reduction in blue shark and overall shark catches after 2003 and second, a shift away from sharks between 2012 and 2016. It is likely that the former was a consequence of a shift in fishing strategy in response to Palau's ban on wire leaders and shark retention (Rppl 6-36, 2003), passed in 2003 (Gilman et al., 2015) and to date the country's most important shark protection law. However, the year 2003 was also identified as the global peak of pelagic shark catches (Clarke et al., 2015), suggesting that the decline observed here could also be due, at least in part, to factors other than Palau's domestic legislation. The second shift away from sharks appears to have coincided with the adoption of several CMMs for sharks by members of the WCPFC, e.g., the silky shark retention ban of 2014 (WCPFC, 2013). No discernible change in catch composition was reflected in the data after Palau declared its waters the world's first shark sanctuary in 2009.

While it appears that these bycatch reduction measures had a discernible impact on longline catches, such measures can also have unintended effects. Retention bans, designed to reduce incentives to catch and retain certain species, might give a false sense of reduced mortality, but they mitigate neither at-vessel mortality nor post-release mortality rates, both of which can be high in some species (Musyl and Gilman, 2018; Braccini and Waltrick, 2019). Because retention bans often result in increased discard rates, they can undermine coastal states' food security and prevent them from fully realizing the benefits from the fisheries in their waters (Gilman et al., 2017). Furthermore, mitigation measures may be insufficient in improving the status of some sensitive species caught in the longline fisheries in Palau, such as critically endangered oceanic whitetip sharks. A 2012 stock assessment identified bycatch in longline fisheries as the greatest impact on the WCPO stock of this species, and found it to be overfished, with overfishing occurring (Rice and Harley, 2012). All three species of thresher shark (pelagic, bigeye, and common) were recorded in this study, collectively forming the third most abundant key shark taxon across all data sources. All are classified as vulnerable on the IUCN Red List, and are listed on CMS Appendix II and in the CMS MoU for sharks. Globally, the outlook for far-ranging elasmobranchs is grim: a recent study attributed an 18-fold increase in fishing pressure to a 71% decline in the global abundance of oceanic sharks and rays since 1970, which corresponds to three-quarters of the species in this group being threatened with extinction (Pacoureau et al., 2021). More comprehensive measures, such as fishery closures or fishing bans, may be required to halt rapid and ongoing population declines of these and other fishery-associated species, while avoiding some of the unintended outcomes of fisheries management approaches.

Static Pelagic Marine Protected Areas and Displacement of the Locally Based Fleet

A localized reduction in fishing pressure and the protection of important habitat has been shown to benefit populations interacting with fisheries, particularly those of threatened species (Jaiteh et al., 2016; Ban et al., 2017). VLMPAs are increasingly popular with NGOs and governments as a conservation tool for ecological issues that resist conventional fisheries management interventions. Yet VLMPAs are rarely designed for highly mobile marine fauna (O'Leary et al., 2018). The purported benefits of their large size, an important factor in MPA efficacy, has been questioned for these species, given that even the largest of MPAs are unlikely to fully encompass their home ranges (Agardy et al., 2011; Kaplan et al., 2014; Curnick et al., 2020b). There has been limited research and evidence of the ecological responses to static pelagic MPAs that are fixed in space and time, like the PNMS. Two studies provide relevant empirical evidence. First, small MPAs adjacent to African penguin (*Spheniscus demersus*) colonies that removed purse seine fishing for pelagic forage fishes may have improved penguin foraging efficiency, chick survival and condition, and increased population growth at one of the colonies. The local abundance of prey resources may have increased within the MPAs as a result of the cessation of fishing mortality, while at a "control" penguin colony with no MPA there may have been increased fishing mortality due to displaced fishing effort from the MPAs (Sherley et al., 2018). Second, a counterfactual assessment found that the U.S. Pacific Remote Islands Marine National Monument caused a reduction in blue shark catch rates by Hawaii's pelagic longline fishery (Gilman et al., 2020). The Monument was also found to have protected bycatch hotspots for some at-risk species (oceanic whitetip, silky and blue sharks, and olive ridley sea turtle) but cold spots for others (albatrosses, shortfin mako shark and striped marlin). Studies from other ocean basins suggest that protective benefits could accrue for species whose ranges, vulnerable life stages or critical habitats are highly concordant with the PNMS (Koldewey et al., 2010; Mee et al., 2017). For example, a tracking study of three species recorded in this study – blue marlin *Makaira nigricans*, sailfish *Istiophorus platypterus*, and silky sharks – indicated that they were effectively protected within the British Ocean Territory MPA, a sanctuary similar in size to the PNMS (Carlisle et al., 2019). Additionally, the PNMS might protect some core use areas of other frequently caught species, including parts of the foraging grounds or nesting routes of endangered green turtles *Chelonia mydas* and vulnerable olive ridley turtles, which were disproportionately caught by the LBF. While these studies suggest that the ban on fishing within the PNMS could result in some population-level benefits for certain species of conservation concern (Koldewey et al., 2010), they also highlight the need for robust assessments of the performance of pelagic MPAs, and to account for multispecies conflicts and other effects, such as displaced fishing effort.

Effort displacement in response to MPAs, if it occurs, affects ecological responses and can prevent MPAs from achieving objectives (Gilman et al., 2019). In Palau, the DFZ's re-orientation and consequent departure of the LBF might have tangible beneficial effects on some of the species that were frequently caught by its vessels. For example, the catch rate models suggested that silky sharks were caught at a significantly higher rate by the LBF than the DWF, with an increasing trend since 2015 and even catch rates throughout the DFZ and PNMS. As such, any benefits that may accrue for silky sharks through the PNMS are likely linked to the departure of the LBF rather than the location of the DFZ. However, movements and aggregations of pelagic fish and oceanic megafauna tend to be associated with particular environmental conditions upon which national boundaries have little or no influence (Harrison et al., 2018; Dunn et al., 2019). While the departure of the LBF reduced fishing effort in Palau's waters, it almost certainly did not cause a reduction in regional fishing effort, meaning that the displaced vessels are likely to have moved to neighboring EEZs (any reduction in regional fishing effort since early 2020 would likely be due to the COVID-19 pandemic). With regards to very highly migratory species whose home ranges are not wholly encompassed by the PNMS, the protective effects afforded by it may therefore be offset by the displacement of fishing effort out of Palau's waters or other coinciding external circumstances, such as the pandemic (see also Curnick et al., 2020b). Understanding to what – if any – extent these stocks may experience intensified fishing pressure in neighboring EEZs would be an interesting future research direction.

Outlook

With several countries struggling to meet the 10% protected national marine area requirement of Aichi Target 11, more large and very large MPAs are likely to be established in the coming decade (Failler et al., 2019). However, their anticipated benefits can raise unrealistic local expectations, divert attention and resources away from other means of addressing marine conservation targets, and demand substantial socioeconomic costs (Klein et al., 2008; Jones and De Santo, 2016; Christie et al., 2017). Mitigating these challenges warrants thorough assessments of LMPAs' effectiveness in delivering both conservation and socioeconomic outcomes. Ultimately, the benefits of protection potentially afforded to highly migratory species through the PNMS are inextricably linked to the sanctuary's future, which will be shaped in large part by the efficacy of its management and enforcement, sustainable financing, and local support for the sanctuary.

Unlike its neighboring Pacific Island economies, whose tuna fisheries constitute a primary source of revenue, Palau's economy relied heavily on (eco)tourism prior to the COVID-19 pandemic (Wabnitz et al., 2018). This was clearly reflected in ex-President Remengesau's introduction of the PNMS Act, which stated that "our future is in tourism, not tuna." The PNMS was intended to boost tourism, and a pre-implementation survey found that although most visitors were not aware of the PNMS, 43% of those who knew about it cited it as an important factor in their decision to visit Palau (Oleson et al., 2019). As it turned out, the pandemic had a devastating effect on Palau's tourism industry, potentially eroding the tourism-forward basis on which the PNMS was declared (EconMAP, 2020). It remains to be seen whether and how this unexpected development will affect continued political support for the PNMS.

Prior to 2020, one of the benefits to Palau from longline fishing in its waters was the supply of fresh tuna for the local market, an important contribution to local food security. Of note, in 2016 Palau was identified as one of four Pacific Island Countries in which landings from locally based tuna fisheries contributed the most to food security (Tolvanen et al., 2019). Following the implementation of the PNMS, an acute lack of locally available tuna and increased consumption of reef fish quickly became a point of contention, leading to claims that the sanctuary had "backfired" (Carreon, 2020). Disappointed residents expressed an unmet expectation of the PNMS resulting in higher, not lower, availability of pelagic fish. Indeed, one of the premises of the PNMS was that the transition from a foreign-dominated tuna fishery to a predominantly domestic one would reserve pelagic resources for Palauans. While a joint assessment by FFA and SPC deemed a domestic longline fishery unviable (Skirtun and Hare, 2017), a subsequent rapid assessment of Palau's tuna fishery development options identified a locally operated pole-and-line fishery as a promising alternative (IPNLF, 2019). A strategic plan for a nationwide network of anchored fish aggregating devices (FADs) within Palau's 12 nm territorial waters was developed in 2018–2019 and previously deployed FADs received maintenance in a bid to encourage pelagic fishing on local vessels. A Presidential Directive issued on World Tuna Day 2018 was meant to encourage Palauans to "Choose Pelagics" over reef fish, providing a further incentive for pelagic fishers. However, the beginning of 2020 saw the implementation of the sanctuary without a domestic pelagic fishery having been established. Spurred by the shortage of fresh tuna following the implementation of the PNMS, efforts to develop a domestic pelagic fishery have since been revived: one of the three locally owned longline vessels recommenced fishing in early 2021, supplying the newly formed fishers' association Belau Offshore Fishers, Incorporated (BOFI) with fish from the DFZ. Plans for a locally owned pole and line vessel, in discussion since 2019, are also expected to come to fruition in 2021. One advantage of focusing on local, relatively small-scale fisheries for pelagics is that bycatch events are likely to occur at much lower scales compared to industrial fishing operations. However, the likelihood of bycatch events is not negated, and well-managed local fisheries will need to be reliably documented, such as the 100% observer coverage called for in the PNMS Act and regulations.

CONCLUSION

A primary, although not grounded, expectation of the PNMS is the recovery of fish stocks and other oceanic megafauna that, prior to 2020, interacted with longline vessels throughout Palau's EEZ. We found that in the decade preceding the sanctuary's implementation, almost half of the longline catch constituted bycatch species (species other than the primary target tunas), and most of those were discarded, possibly resulting in high mortality levels. Annual catches of species of conservation concern, including an estimated 50,000 sharks provide a sobering perspective on sustainability in the world's first shark sanctuary, but also highlight the potential for population segments of these species to benefit from localized spatial protection through the PNMS, particularly if core use areas or key life history stages are demonstrably protected. While several studies have identified the design of a sanctuary as a key factor in its effectiveness, we could not clearly discern the effects of sanctuary location – and, by extension, the placement of the DFZ – from fleet effects, particularly differences in fishing strategies, on catch rates and composition. Our results identified the DFZ as an area of overall lower catch rates, while the fishing strategies of the fleet that continues to fish there seem to result in higher target:bycatch ratios and lower bycatch mortality levels compared to the LBF. Thus, the re-orientation of the DFZ in 2019 and the resulting changes in fleet presence may hold greater promise of potential conservation benefits than the original placement of the DFZ. The understanding that fishery interactions with species are influenced both by where and how vessels fish can be leveraged in the design and management of fishing zones contained within, or adjoining, LMPAs. As observer coverage on a reduced longline fleet in Palau's DFZ becomes more robust and representative, future studies might consider the effects of changes in the concentration of fishing effort within Palau's waters as well as neighboring EEZs as an effect of the PNMS. Coupled with an exploration of fishery-associated species' movements in Palau's waters and adjoining areas, such analyses could help to better discern the potential protective effects of the sanctuary on far-ranging species. Our results, as well as early local responses to some unexpected growing pains of the PNMS, illustrate that LMPA placement and implementation ought to be considered carefully to maximize potential benefits and manage local expectations.

DATA AVAILABILITY STATEMENT

The data analyzed in this study is subject to the following licenses/restrictions. The data used in this study are classified country data, which precludes their provision in a publicly accessible repository. Requests to access these datasets should be directed to the corresponding author. Aggregate catch and effort data for the WCPFC area are available from https://www.wcpfc.int/public-domain, and publicly available bycatch data can be downloaded from https://www.wcpfc.int/public-domain-bycatch.

AUTHOR CONTRIBUTIONS

VJ designed the study, obtained funding, analyzed the data, and wrote the manuscript. TP advised on data analysis, analyzed the data, and wrote the manuscript. SL analyzed the data and edited the manuscript. EG and SN advised on study design and edited the manuscript, and SN also provided logistical support. All authors approved of the version to be published and agreed to be accountable for all aspects of the work.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.720603/full#supplementary-material

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The Salient Dynamics of Cross-Border Ocean Governance in a Regional Setting: An Evaluation of Ocean Governance Systems and Institutional Frameworks in the Guinea Current Large Marine Ecosystem

Ibukun J. Adewumi[1,2,3,4], Juan Luis Suárez de Vivero[5] and Alejandro Iglesias-Campos[6]*

[1] Global Ocean Accounts Partnership, Institute for Global Development, University of New South Wales, Sydney, NSW, Australia, [2] Australian National Centre for Ocean Resources and Security, University of Wollongong, Wollongong, NSW, Australia, [3] African Marine Environment Sustainability Initiative, Lagos, Nigeria, [4] Erasmus Mundus Master Course on Maritime Spatial Planning, Università IUAV di Venezia, University of Sevilla, and University of Azores, Venice, Italy, [5] Department of Human Geography, University of Seville, Seville, Spain, [6] OSPAR Commission, London, United Kingdom

Edited by:
Jesper H. Andersen, NIVA Denmark Water Research, Denmark

Reviewed by:
Paulina Ramírez-Monsalve, World Maritime University, Sweden
William McClintock, University of California, Santa Barbara, United States

***Correspondence:**
Ibukun J. Adewumi
ibukunadewumi@gmail.com;
i.adewumi@oceanaccounts.org

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This article contributes to a growing body of research on the Large Marine Ecosystems Concept. It particularly shines the light on the Guinea Current Large Marine Ecosystem (GCLME), a biodiverse maritime domain providing essential ecosystem services for the survival of a large population while at the same time under intense pressure from both anthropogenic and natural factors. With the need for coordination and cross-border ocean management and governance becoming imperative due to the magnitude of challenges and maritime domain, we examine the factors that underpin ocean governance and those key elements necessary for cross-border ocean governance cooperation in the region. The research draws on qualitative data collected from peer-reviewed literature and documents sourced from different official portals. Three countries in the region (Benin, Nigeria, and Cameroon) are selected as the descriptive and comparative case studies to examine: (i) the factors that drive ocean governance (including geographical features, maritime jurisdictions, political framework, maritime activities, and associated pressures), and (ii) key enabling factors for cross-border ocean governance and cooperation in the GCLME (including marine and coastal related policy and legal framework convergence from international to national including, and shared experiences, common issues and joint solutions). We show that the biophysical maritime features, the implementation of the United Nations Convention on the Law of the Sea (UNCLOS), otherwise known as the Law of the Sea (LOS), inherent political characteristics and the relics of colonization, and increasing ocean use and pressure on the ecosystem make ocean governance challenging in the region. Our analysis also reveals a varying level of convergence on international, regional and national legal, policy and institutional frameworks between the case studies on ocean-related aspects. Significant convergence is observed in maritime security, ocean research, and energy aspects, mostly from countries adopting international, regional and sub-regional frameworks. National level convergence is not well established as administrative and political arrangement differs from country to country in the region. These different levels of convergence help reveal procedural and operational shortcomings, strengths, weaknesses, and functional capability of countries within a cooperative ocean governance system in the region. However, experience from joint-implementation of projects, pre- and post-colonial relations between countries and the availability of transboundary organizations that have mainly emerged due to sectoral ocean challenges would play a crucial role in fostering cross-border ocean governance cooperation in the region.

Keywords: ocean governance, ocean policy, Gulf of Guinea, integrated ocean management, cross-border cooperation, Guinea Current large Marine Ecosystem, Africa ocean governance

INTRODUCTION

The Guinea Current Large Marine Ecosystem (GCLME) is a total area of 1,958,802 km^2 bordering: Guinea-Bissau, Guinea, Sierra Leone, Liberia, Ivory Coast, Ghana, Togo, The Republic of Benin (Benin), Nigeria, Cameroon, Equatorial Guinea, Gabon, Congo, Angola, The Democratic Republic of Congo, São Tomé and Príncipe (IW:LEARN, 2016; **Figure 1**). It falls in the cluster of Large Marine Ecosystems exhibiting economic development levels within the low to medium range (based on the night light development index) and medium levels of collapsed and overexploited fish stocks (Ukwe et al., 2006; UNESCO/IOC, 2020a). According to UNESCO/IOC (2020a), the overall risk factor in the GCLME is rated high following a combined measure of the Human Development Index and the averaged indicators for fish and fisheries, pollution and ecosystem health modules. It is a marine region endowed with an extensive coastline and maritime space, which provides the basis for substantial economic and social proportion activities (Okafor-Yarwood et al., 2020). About 47% of the 248 million GCLME's people lives (200 km) off its coast and are dependent on the resources therein (Okafor-Yarwood et al., 2020), and projected to increase in share to 52% in 2100 (Barbier, 2015). However, intense competition and unsustainable use of resources by different sectors, coupled with climate change, negatively affect the ecosystem and people who depend on them (Abe et al., 2016; Okafor-Yarwood, 2018).

With the magnitude of marine space under the jurisdiction[1] of the GCLME countries (see **Table 1**), collaborative, management of different aspects of the maritime areas is therefore imperative to protect biodiversity and secure livelihoods. Weak collaborative processes in the GCLME impedes stakeholders to manage the ocean cohesively, minimize conflict, and maintain a long-term flow of ecosystem goods and services, just as resource mismanagement, degradation, and depletion become increasingly evident (IMS-UD/UNEP, 2015; Okafor-Yarwood et al., 2020). Likewise, the absence of adequate coordinating mechanisms for marine activities further entrenches fragmentation of governance architectures and duplication of efforts. However, the inadequate implementation/enforcement of the existing legal, policy, and institutional frameworks, combined with the significant extent of the maritime domain, might be why the required collaboration and coordination necessary to ensure sustainability in the GCLME needs unique attention. There have also been calls to strengthen cooperation across national boundaries to ensure ocean sustainability. This is principal because of specific governance gaps in Africa, such as the lack of a common political/economic agenda and coordinated approach to using and managing ocean resources (e.g., IMS-UD/UNEP, 2015).

How do we address these reprising challenges so that national and regional coordination and cross-border collaboration in the GCLME becomes possible to ensure the overall sustainability of coastal and marine spaces? Vivero and de Mateos (2015) believe that understanding the elements that shape the emergence of ocean governance, including geographical features (physical and biological), maritime jurisdictions, political framework, maritime activities, and associated pressures on different scales, should be the first prerogative. To Boateng (2006), a clear understanding of available frameworks and their consequent impact on resources and stakeholders' power is required. Boateng assertion holds true because the governance of coastal and marine space is viewed as the process of policymaking and negotiation nested between governmental institutions at several levels, civil society organizations and market parties (OECD, 2004; Momanyi, 2015; Horigue et al., 2016).

This paper aims to point out how cross-border collaboration for ocean governance in the GCLME may become possible by understanding the conceptual and normative construction, strength and weakness of ocean governance in the GCLME. To achieve this aim, the paper poses three research questions: (1) What are the underlining elements that shape the emergence of marine governance in the GCLME? (2) What are the enabling factors for cross-border ocean governance cooperation in the GCLME? (3) What is the capacity of the existing transboundary organizations to foster the most

[1]The term 'jurisdiction' under UNCLOS refers to coastal states' own maritime zones and encompasses the resources and activities therein as well as external impacts on them.

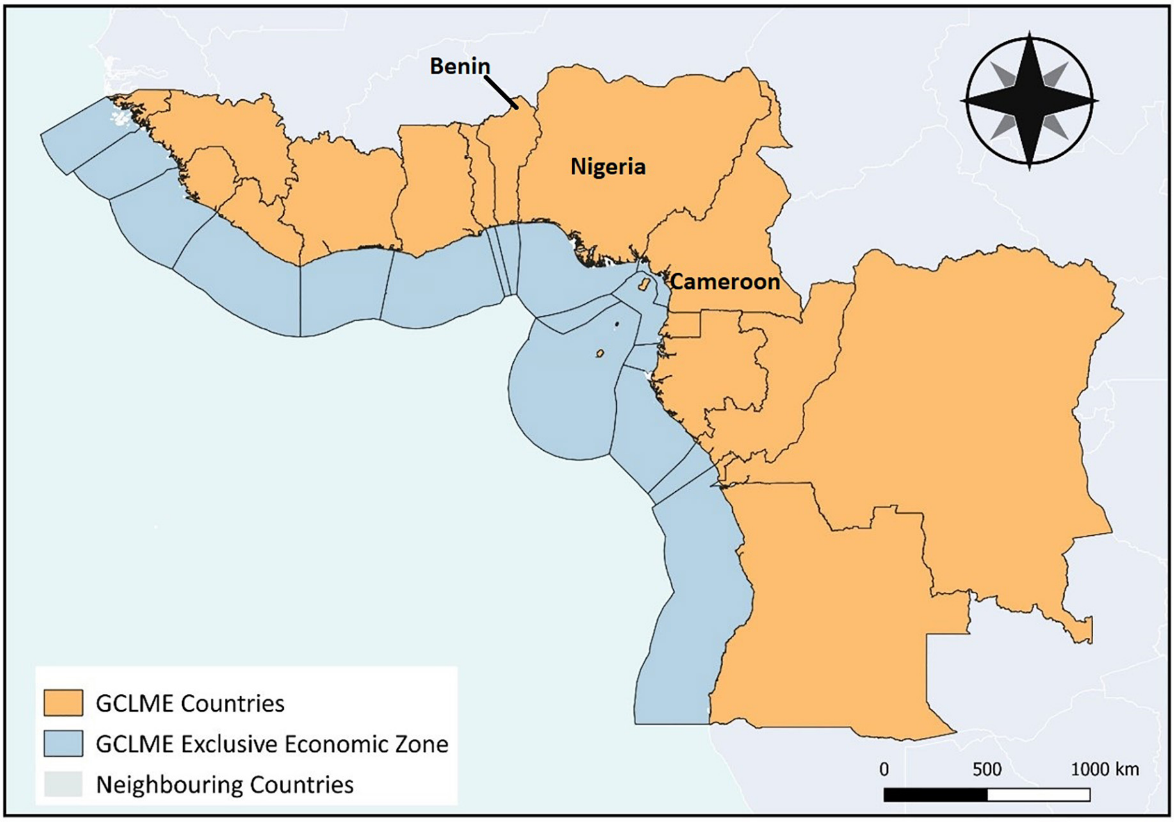

FIGURE 1 | Map showing the geographical scope of the GCLME (Data source: Flanders Marine Institute, 2019).

TABLE 1 | Maritime jurisdictions in the GCLME.

Countries	Jurisdictional waters (km^2)					
	Inland waters	Territorial sea	Contiguous zone	Exclusive economic zone	Extended continental shelf	Total
Angola	874	34,068	32,643	455,214	344,268	834,425
*Benin	–	30,069	–	–	–	30,069
*Cameroon	–	14,775	–	–	–	14,775
Congo	–	35,396	–	–	–	35,396
Equatorial Guinea	–	12,390	–	296,026	–	308,416
Gabon	5 439	16,212	14 798	156,094	18,130	195,874
Ghana	–	12,219	12 343	212,734	33,413	258,366
Guinea	–	8,447	–	101,181	–	109,628
Guinea Bissau	13 967	6,148	–	86,381	–	106,496
Ivory Coast	–	12,618	–	162,072	20,267	194,957
Liberia	–	12,389	12,525	233,935	–	246,325
Ivory Coast	–	12,618	–	162,072	20,267	194,957
*Nigeria	–	19,367	–	163,447	8,001	190,814
Sierra Leone	–	10,156	8,504	149,612	–	159,768
S. Tome and Principe	3,849	11,603	14,719	115,320	–	130,772
RD Congo	391	558	191	1,125	–	2,075
Togo	–	2,615	–	12,776	–	15,391
Nigeria–Sao Tome (joint)	–	–	–	34,539	–	34,539
Total	24,520	239,031	80,926	2,180,456	424,078	2,868,086

**Case study countries.*

significant cross-boundary ocean governance cooperation in the GCLME?

The GCLME provides the opportunity to explore the research questions in this paper, considering that the region: (1) in contrast to other regions on in the continent, is a setting where relatively all maritime boundary disputes have been resolved, (2) exhibits a wide range of biomes and ecoregions (Miller and Gosling, 2013) and, (3) consists of culturally diverse nations with different governance regimes (i.e., centralized/federal), which results in a wide range of ocean governance system transformations due to human action and social peculiarities. Also, it includes areas where the maritime space has been aggressively exploited for its resources, uniqueness and strategic location for more than five centuries (i.e., from the transatlantic slave trade era to the pre and post-colonization times). It is also an area where early European colonization expanded new forms of maritime trade and is currently the most active frontier of fisheries, agriculture, industrialization and population expansion in the world (Harley, 2015; Abobi and Wolff, 2020; Nwafor et al., 2020; OECD, 2020).

Cross-national research in the GCLME region poses many methodological and logistically challenges (Copans, 2020). These methodological and logistical challenges also come amidst an increasing call to decolonize academic research in the region (Adams, 2014; Seehawer, 2018). Therefore, answers to the research questions are explored using Benin, Nigeria, and Cameroon as descriptive and comparative case studies to highlight the functional capability of some GCLME countries to cooperate toward ocean governance and existing transboundary institutions.

Benin, Nigeria, and Cameroon are chosen because they share maritime and land borders and social and ethnographic affinity (Edung, 2015; Nwokolo, 2020). They have historically cooperated on several developmental areas pre-and post-independence. Also, many ocean development projects are currently taking place in these countries' maritime jurisdictions. These include developments in the oil and gas, maritime security, ports, coastal land concessions and reclamation sectors which have attracted the most significant attention from citizens, civil society groups and investors.

Although cross-national qualitative research presents many issues, including issues related to the selection process of countries and the analytical strategy (Gharawi et al., 2009), its application in this paper gives room for the development of new perspectives in the GCLME governance research. It also allows the development of robust and context-driven research in the Large Marine Ecosystems (LMEs) governance concept. Likewise, much of the academic literature on the LME concept focuses on the need for and the benefits of cross-border ocean management. However, little research has been conducted on how cross-border cooperation may be best advanced between neighboring jurisdictions in the GCLME or the political and institutional conditions that can facilitate practical cross-border cooperation at an LME scale.

The selection of Benin, Nigeria, and Cameroon as analytical and comparative case studies allows for a cross-national qualitative research approach to be applied, a method not commonly applied in ocean governance system research. It has also permitted comparisons between ocean governance systems in Francophone and Anglophone regions, differing political and post-colonial attitudes that affect cross-border participation in policy and development planning.

This paper is outlined in four sections. The first section reveals factors that shape the emergence of ocean governance in the GCLME by analyzing the three case studies' geopolitical variables, including geographical features, maritime jurisdictions, political framework, maritime activities, and associated pressures. The second section moves to identify the structures and mechanisms staged at international, regional and national levels that tend to promote or frustrate cross-border ocean. The third section assesses the current capacity of existing transboundary institutions in the GCLME to foster cross-national ocean governance cooperation based on Kidd and McGowan's (2013) analytical framework. It provides an opportunity to identify a spectrum of transnational ocean governance partnership approaches that could be applied in the region. The fourth concluding session discusses the study results by highlighting challenges facing coastal and marine governance and transboundary collaboration in the GCLME while emphasizing the need to enhance cross-sectoral coordination at the national and improve cooperation among regional institutions.

MATERIALS AND METHODS

The present paper is based on a desk review of secondary data collected from peer-reviewed literature and official documents sourced from the United Nations Food and Agricultural Organization (FAO) FAOLEX and ECOLEX databases, the UN treaty collection, the African Union (AU) database of treaties, conventions, protocols and charters, and other national repositories. It generally employs a qualitative research approach to understand factors that either bring weak or strong ocean governance. Likewise, it is used to explore mechanisms that foster or wreck cross-border cooperation and analyze the capacity of existing institutions to promote cross-border ocean governance coordination and cooperation. A combination of two political science approaches is adopted to guild the logic and analysis in this paper, including the Constructivist Institutionalism and Historical Institutionalism approach. Following Steinmo (2008) and Bell (2011), these two approaches are essential for this study to dissect the 'ideational' foundation of ocean governance and examine how institutions' creation, maintenance, and change can foster cross-border cooperation for ocean governance in a particular historical timeframe. After all, politics, policies and people constantly shape the ocean, just as political ecology themes (power and politics, narratives and knowledge, scale and history, and environmental justice and equity) are interconnected with governance and management (Bennett, 2019).

Given the previously mentioned aspects of geopolitical, sociological, historical, and developmental idiosyncrasies, the GCLME and the three case studies (Benin, Nigeria, and Cameroon) are chosen to undertake this study. Gerring (2013) and Devare (2015) had earlier raised concern about investigators believing they have full knowledge of a particular study area, and maintained that knowledge is always partial. However, information collected from existing documents is complemented with the first-hand knowledge of the authors about the environmental, political, and socio-economic realities of GCLME and the selected case studies.

To answer the questions posed by this paper, we carried out three types of investigations. Attending to the first research question "What are the underlining elements that shape the emergence of marine governance in the GCLME?", we employed a descriptive-analytical research approach to examine the ideational and normative factors which drive ocean governance in the GCLME using Benin, Nigeria, and Cameroon as analytical and comparative case studies. A descriptive-analytical research approach helps point toward causal understanding and reveals mechanisms behind causal relationships (Loeb et al., 2017).

Once the ideational and normative factors that shape ocean governance in the region are described, it became essential that answering our second research question, "what are the key enabling factors for cross-border ocean governance cooperation in the GCLME?" would require the examination of the operational and deliberative mechanisms staged at the international, regional and national levels to promote cooperative ocean governance. Previous studies on ocean governance (e.g., Rochette et al., 2015; Weiand et al., 2021) argues that this examination enables the understanding of how collaborative ocean governance in a particular context is constructed, particularly through inter-subjective operations embodied in the governance systems and institutional frameworks. A range of existing analytical frameworks from previous studies (e.g., Fanning et al., 2007, 2013; Hill and Kring, 2013; Herman, 2016) could be adopted to answer our second question. Pearce et al. (2015) posit that such frameworks improve validity and reliability in assessment, allowing researchers to create robust assessment instruments more easily. However, many of these frameworks focus more on the nature of cross-border ocean governance processes and their effects on managing marine resources. But the authors insist only on one dimension of the cross-border ocean governance or integration, typically favoring the functional capacity of governance systems and institutional frameworks dimensions of differing states. Therefore, taking a cue from a transboundary marine spatial planning perspective, we adopt Flannery et al. (2015) analytical framework. Flannery and colleagues believe that for cross-border ocean governance to be possible, it is critical to identify contextual factors that are likely to impact the success of transboundary partnership initiatives. These factors are identified as policy convergence, the common conceptualization of planning issues, joint vision and strategic objectives, shared experience, and existing transboundary institutions. We, however, categorize the factors into three broad elements, which are presented and explained in the **Table 2** below.

To answer the third question "what is the capacity of the existing transboundary organizations to foster the most significant cross-boundary ocean governance cooperation in the GCLME?", Kidd and McGowan's (2013) ladder of transnational partnership is adopted to assess existing transboundary organizations' nature in the region. Considering the region's and case study countries' multi-level ocean governance structure, this is to evaluate conditions and institutions that may affect cross-border ocean governance cooperation. Other cross-boundary institutional analysis frameworks such as Herrera et al. (2005) and Rahman et al. (2017) are based on institutional efficiency criteria and the relationship of different rule levels. In contrast, Kidd and McGowan's ladder provides the opportunity to explore further motivations for collaboration between cross-border institutions in particular marine settings. It also helps to grasp which institutions have reached an atmosphere of established understanding applicable to transboundary initiatives.

TABLE 2 | Explanation of the Flannery et al. (2015) theoretical framework.

Assessment elements	Explanation	Issues
Policy convergence	The degree of convergence in legal, policy and institutional arrangements is a critical element of successful cross-border ocean governance. The more alike the policy structures and discourses in neighboring jurisdictions, the more probable it is that transboundary ocean governance will succeed.	Can result in a 'race to the bottom, wherein jurisdictions compete to reduce the regulatory encumbrance on firms to develop a competitive advantage over one another.
Shared experiences, common issues and joint solutions	The development of cross-border initiatives can be expedited if the actors involved have previous experience in cross-border cooperation, regardless of the policy area, and have developed a sense of mutual understanding and trust. Identifying common issues and the collaborative formulation of mutually beneficial solutions can form the underpinning for lasting transboundary planning.	Institutional arrangements may often discourage cross-border ocean governance Identifying an area requiring collaboration amongst neighboring jurisdictions is not, however, sufficient to ensure effective cross-border ocean governance
Existing transboundary institutions	The existence of a network of well-developed transboundary institutions reduces transaction costs associated with cross-border ocean governance and facilitates cross-border working. These institutions may be formal or informal alliances and include supranational institutions, where the key actors know each other, have experience in cross-border cooperation and may have developed good working relations	Existing institutions may prescribe or limit the course of action that may be taken to address an issue

In Kidd and McGowan's ladder (see **Figure 2**), the first rung on the ladder concerns *Information Sharing*, *focusing* on trust-building among a range of stakeholders, understanding each other's perspectives, and building capacity to support integrated ocean approaches. *Administration Sharing* is the second rung that presents potential areas where collaborative advantages for ocean governance advantages are perceived. The third rung on the ladder is where stakeholders identify *Agreed Joint Rules* that can facilitate establishing standard procedures or protocols related to specific areas of activity. *Combined Organization* relates to the level where new joint research institutes, joint planning teams, or other formal institutional arrangements of a transnational nature are created. *Combined Constitution* occupies the fifth rung of the ladder and relates to how cooperative efforts are formalized through new legal agreements and may secure new political order for ocean governance and management.

Exploring the Underlining Elements That Shape the Emergence of Ocean Governance in the Guinea Current Large Marine Ecosystem Through the Lens of Benin, Nigeria and Cameroon

Biophysical Maritime Features and Ecosystem and Pressure

Benin, Nigeria, and Cameroon are coastal states in the GCLME (see **Figure 3**) with an extensive coastline and maritime space, characterized by a high degree of biodiversity and resources, which provides the basis for a substantial proportion of economic and social activities (UNDP, 2013; Oribhabor, 2016; Rice and Rosenberg, 2016). With one of the shortest coastal strips in the region, Benin's coastal zone comprises alluvia sand with a maximum depth of four meters with longitudinal depressions parallel to the coastline and swamps (Dossou and Gléhouenou-Dossou, 2007). Whereas, the barrier lagoon complex of Nigeria covers about 200 km from Benin/Nigeria border eastward to the western limit of transgressive mud beach and adjacent to the Gulf of Guinea (GoG) backed by the Badagry creek, Lagos Lagoon and Lekki, Lagoon (Amosu et al., 2012). Cameroon's different coastal ecosystems are prevalent, including estuaries (in Rio-del-Rey, Cameroon, and Ntem estuaries), mangroves, lagoons, deltas, mud and sand flats, coastal shelves, etc. (UNESCO/IOC, 2020b). On the other hand, the south-eastern part of the coast presents an alternation of rocky and sandy beaches and cliffs (Fonteh et al., 2009).

Mangrove swamps are the most biologically significant coastal ecosystems along these countries' coasts (Asangwe, 2006; Fonteh et al., 2009; Amosu et al., 2012; UNDP, 2013), with strands reaching heights of up to 40 m (FAO, 2007). As it will be described in section "Maritime uses, activities and pressures," these forests are now under severe pressure from anthropogenic activities, putting their ecosystem service roles and biological diversity at stake (Ukwe et al., 2006; Eke, 2015). Equally, several aquatic species are endangered due to unsustainable harvesting, oil pollution and habitat degradation (GCLME-RCU, 2006; Amosu et al., 2012). The severity of coastal erosion is high due to natural factors and habitat modification (Abessolo Ondoa et al., 2018; World Bank, 2019; Alves et al., 2020).

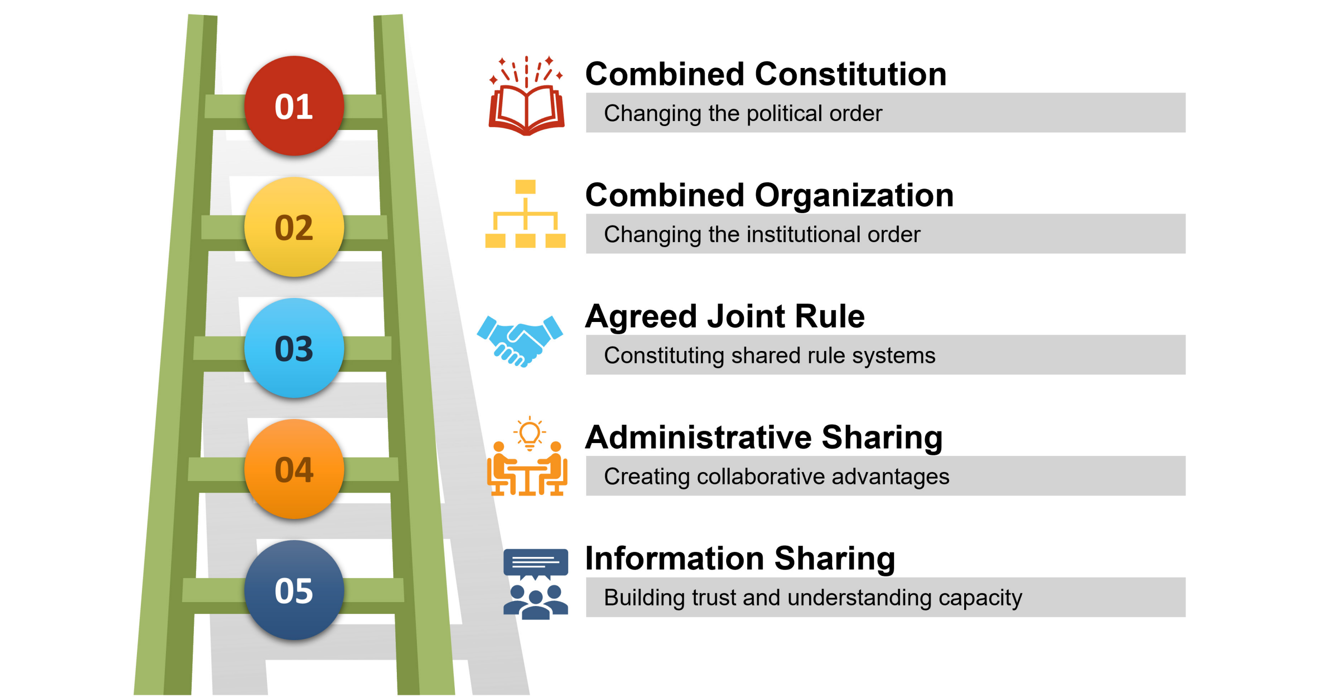

FIGURE 2 | A ladder of transnational partnership working to support marine spatial planning.

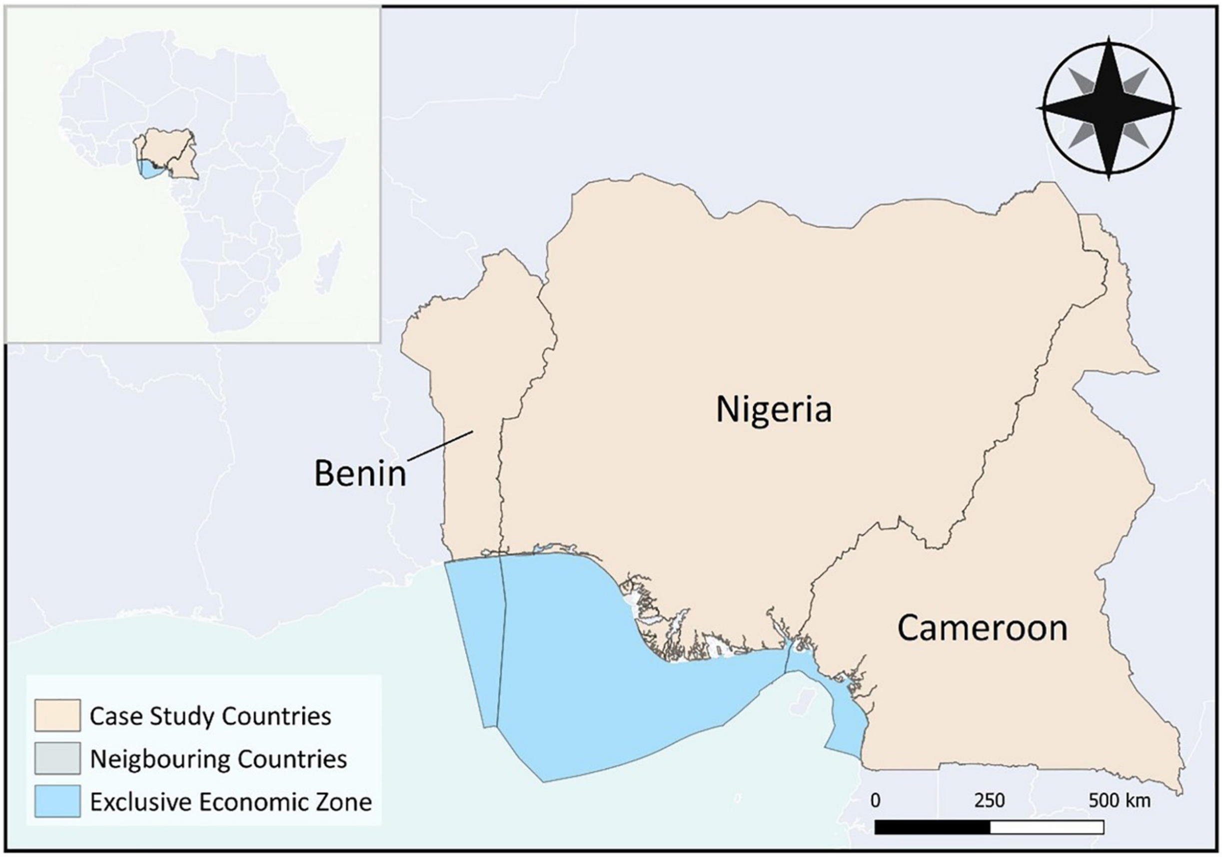

FIGURE 3 | Map of study area showing maritime jurisdictions (Data source: Flanders Marine Institute, 2019).

Maritime Jurisdiction

All three countries have since ratified the LOS—and like every other coastal state operating under UNCLOS, they are entitled to an Exclusive Economic Zone (EEZ) of 200 nautical miles, including territorial waters and contiguous zones. The three countries have since promulgated legislations to delimit their EEZ in 1976, 1978, and 2000, respectively (see **Table 1**). Likewise, they have submitted applications to the Commission on the Limits of the Continental Shelf for an extension beyond the 200 nautical miles[2]. Meanwhile, the fierce maritime and land dispute between Nigeria and Cameroon (Nigeria Vs Cameroon: Equatorial Guinea Intervening) was put to rest on the 10 of October 2002, following the International Court of Justice's grand judgment ruling in favor of Cameroon. It is interesting to note that the maritime jurisdictions of Benin, and Nigeria and Cameroon are recognized under various geographical contexts, including the greater Gulf of Guinea, GCLME, Southeast Atlantic, IHO Gulf of Guinea, and the Global International Water Assessment Region 42. Likewise, the maritime jurisdictions in the countries are found under different Universal Transverse Mercator Coordinate System Zones (Benin – 31N; Nigeria – 31N, 32N, 33N; and Cameroon – 32N, 33N).

Political Framework

The signing of UNCLOS marked the latest major international political step toward a universal regulation of the ocean. It has further jerked commitments at the political level in the GCLME, calling for a better understanding of the value and usefulness of the sea (Chatham House, 2013). However, Benin, Nigeria, and Cameroon are all products of colonial imperialism and exhibit inherent political characteristics that generally influence governance, but with various distinctions. Suárez-de Vivero and Rodríguez Mateos (2014) sees these countries as post-colonial maritime states shaped after maritime empires and powers.

In terms of the internal political system, Benin is a presidential representative democratic republic, where the President is both head of state and government. The current political system is derived from the 1990 Constitution giving the president executive power, while legislative power is vested in the government and the legislature. The judiciary is independent of the executive and the legislature. Nigeria is structured as a federation, having a three-tier government (legislative, executive and judiciary). Under the 1999 constitutions, governance is carried out within three federating units (Federal, States, and local governments).

[2]In accordance with Article 76, paragraph 8, of the United Nations Convention on the Law of the Sea through the Secretary-General.

Yet, power resides in the central government, which controls most of the country's revenues and resources.

The political system in Cameroon is a republic multiparty presidential regime that is structured on the French model. Under this model, power is distributed among the President, the Prime Minister, and the Cabinet ministers appointed by the president as proposed by the prime minister, allowing the president to control whoever comes into power. Under this system, the Republic is divided into ten regions supervised by a Governor appointed by the president, who coordinates Divisional officers and subdivision officers.

Apart from the role the internal political framework plays in each jurisdiction's maritime domain, various supranational bodies' roles have become increasingly important in managing marine space. These bodies include the African Union, the Economic Community of West Africa States (ECOWAS), the Economic Community of Central Africa States ECOWAS (ECCAS), Gulf of Guinea Commission (GGC), the United Nations Economic Commission for African (UNECA), etc.

Maritime Uses, Activities and Pressures

Benin, Nigeria, and Cameroon have a significant level of leisure-based coastal tourism with some beach and heritage-based interest. Ouidah, a coastal city in is the Voodoo religion's birthplace endowed with ancient temples (E.g. the Python temple) and grooves where the Voodoo festival attracts thousands of tourists yearly (Forte, 2009). Even though coastal tourism is still developing in Nigeria, the proportion of tourism on the coast is expected to be high. The expectation is partly owing to the coastal location of cities like Lagos and Port Harcourt, sizeable coastal towns and linked communities on the outskirts of cities (e.g., Badagry); pleasant sites on creeks in the Niger Delta; strong historic heritage linked to the slave trade and cultural events, etc. Cameroon has a diverse product, with some beach-related accommodation and a vital element of cultural tourism, including key coastal historical sites with mountain and rainforest experiences. Kribi stands out as the prime leisure tourism destination.

Of all these uses of the coastal-marine area, perhaps the one which has the most significant economic and environmental impact is maritime transport. Generally, the GCLME naval space offers seemingly idyllic shipping conditions (Ali, 2015; Osinowo, 2015; Richardson, 2015). Benin, Nigeria and Cameroon are hosts to numerous natural harbors that are weather friendly to vessels and primarily devoid of chokepoints (Osinowo, 2015). This unique feature provides a medium where raw materials like timber, cocoa, coffee, cotton and finished goods are being traded with other parts. Port and shipping activities are critical to Nigeria and Cameroon as the significant GDP earning of both countries depends on the exportation of hydrocarbon (UNCTAD, 2020a p. 24). Benin, Nigeria, and Cameroon are also open registry nations, registering 462, 10,882, and 448 ships respectively between 2011 and 2020 (UNCTAD, 2020b). However, besides the economic impact of maritime transport in the countries, there have been negative impacts. These include ship-based pollutants on the marine ecosystem (Onwuegbuchunam et al., 2017), coupled with security issues related to piracy and armed robbery at sea, which have escalated into a transboundary crisis (Ali, 2015; Eke, 2015; Okafor-Yarwood et al., 2020). For example, the extent of environmental pollution in the Niger-Delta region of Nigeria, mainly due to oil and gas activities, is unprecedented and has affected the health of ecosystems and the livelihood of those who depend on them (Eke, 2015; Okafor-Yarwood, 2018).

The significance of the export of hydrocarbons to the national income stream cannot be underemphasized, particularly in Nigeria and Cameroon. Nigeria produces an estimate of 2,317,000 Billion Barrels of crude oil per day (BPD) with an offshore output in 2019 estimated at 780,000 barrels per day (BPD), amounting to 39 percent of the country's total daily production (George, 2019). Cameroon received USD 1.152 billion revenue from extractive industry taxation in 2014, with 93.66% from upstream hydrocarbons, mainly from crude oil (EITI, 2020). Meanwhile, in Benin, oil and gas production stopped in the Sèmè field in 1998 with no further discovery. The Niger Delta of Nigeria, Kribi, and Limbe areas of Cameroon are prone to oil spills, destroying millions of people's livelihoods (Tiafack et al., 2014; Amesty International, 2015; Okafor-Yarwood, 2018).

Although resources and capacities related to fisheries vary significantly between Benin, Nigeria, and Cameroon, the sector has been vital to food and socio-economic security. Despite the lack of upwelling along Benin's coastline limits marine resources, the annual harvest is estimated at 12,000 MT for fish and 4,000 MT for shrimp (FAO, 2015), and provide an opportunity for artisanal fishing with an estimated 50,000 canoes and a maritime artisanal fleet of 825 pirogues (WASSDA, 2008; FAO, 2015). Meanwhile, the fisheries sector in Nigeria directly employs an estimated 8.6 million people and another 19.6 million indirectly (WorldFish, 2017). Cameroon's fisheries sector is crucial for socio-economic sustenance as it accounts for 1.8% of the country's estimated US$35 billion GDP and employs more than 200,000 people (Beseng, 2019). Generally, the governments cannot monitor fisheries effort and catch, which often results in a lack of data, scientific knowledge and inadequate management (Chan et al., 2019). Fishery bycatch is not mostly reported, while other illegal activities such as illegal fishing, trans-shipment are significant problems (Belhabib and Pauly, 2015).

Undoubtedly, for a long time, mining renewable and non-renewable resources in the coastlines and seabeds of Benin, Nigeria, and Cameroon contributes to the socio-economic development of coastal communities and substantial degradation of the marine ecosystem. Besides hydrocarbon exploration and mining, extraction of sand, gravel, rocks, sulfur and other construction materials both legally and illegally are ongoing, which has hitherto widely exacerbated land and coastal erosion (Ukwe et al., 2006). In Benin, illegal marine and beach sand mining thrive as sand diggers are paid between US$87 and US$125 per truckload—a value above Benin's average monthly salary is less than US$50 (WACA, 2018). Large-scale sand mining along Nigeria's coast raises concerns over erosion and other environmental damage (Aljazeera, 2014). Illegal and legal sand mining occurs in Cameroon, particularly around coastal cities and towns where industrial activity and construction are high, e.g., port development, land reclamation and housing construction, etc. (Asangwe, 2006; MINEP, 2011; Fotsi et al., 2019). All these puts together have exacerbated coastal erosion, habitat degradation and loss of livelihood (UNESCO/IOC, 2020b).

This section has identified the factors shaping ocean governance in the three case study countries, including biophysical maritime features, maritime jurisdiction, political framework, maritime uses, and associated ecosystem pressures. The next section of this paper examines the mechanisms, staged at the international, regional and national level, that would promote cooperative ocean governance. In addition, it is essential to ask what the enabling factors for cross-border ocean governance cooperation are in the GCLME from the lens of Benin, Nigeria, and Cameroon.

Key Enabling Factors for Cross-Border Ocean Governance Cooperation in the Guinea Current Large Marine Ecosystem From the Lens of Benin, Nigeria, and Cameroon

This section is structured following Flannery et al. (2015) framework, which presents several important factors to measure the possibility of cross-border ocean governance. Flannery and his colleagues identify these factors to include policy convergence, the common conceptualization of planning issues, joint vision and strategic objectives, shared experience, and existing transboundary institutions. However, we categorize the factors into three broad elements, as presented and explained in **Table 2** above. These elements allow us to examine the structures of operation, and deliberative mechanisms staged at international, regional and national levels that tend to promote or downplay cooperative ocean governance in the three case study countries and, by extension, in the GCLME. Also, the length of this section is extensive as it constitutes the core of our analysis

Ocean Related Policy and Legal Framework Convergence From International to National

Policy and Legal Framework Convergence From Ocean Related International Commitment

To a significant extent, Benin, Nigeria, and Cameroon rely on several international policy architecture and commitments to guild ocean management and governance, bringing about convergence in ocean policies and strategies. Besides the promulgation of legislations to delimit their territorial sea, contiguous zone, and EEZs, the countries in 2009 and 2018 for instance, submitted an application to the Commission on the Limits of the Continental Shelf to extend their Continental Shelves[3] beyond the 200 nautical miles. Also, under UNCLOS's limits of the Continental Shelf regime, Benin and Nigeria agreed in 2009 to commit to a "no objection note"[4] to cooperate on the boundary of their extended continental shelf.

Similarly, being contracting parties to the Convention on Biological Diversity (CBD), these jurisdictions must report the progress of biodiversity conservation under a common standard, ensuring they prepare a national biodiversity strategy that is expected to be mainstreamed into national conservation efforts. The same goes for the United Nations Framework Convention on Climate Change (UNFCCC) of 1992, under which the three countries have developed similar but individual Climate Change Action policies and plans. The three jurisdictions are also working with the Paris Agreement to actualize the global climate change targets. Meanwhile, the acceptance and ratification of the Kyoto Protocol of 1997 currently suffer a significant setback, probably due to the countries' progress anchoring on the UNFCC of 1992.

Agreements with international conventions and protocols governing aspects of maritime navigation and shipping seem to have brought a significant level of legal convergence in the three jurisdictions. Among the IMO conventions[5] to which the three countries are, to various degrees, in compliance with, are the 1972 Convention on the Prevention of Marine Pollution by Dumping Wastes and Other Matter (London Convention), International Convention for the Prevention of Pollution from Ships, 1973[6], Protocol for the Suppression of Unlawful Acts (SUA).

There is an average level of convergence on legal instruments in the three jurisdictions on ocean conservation matters, as several vital agreements and conventions have remained either unsigned, signed, or ratified. For instance, the 2001 Agreement on the Conservation of Albatrosses and Petrels is not recognized in the countries. Still, they are parties to the 1979 Bonn Convention on the Conservation of Migratory Species of Wild Animals by the basis of ratification and accession. Meanwhile, out of the three jurisdictions, Nigeria happens to be the only country party to the 1966 International Convention for the Conservation of Atlantic Tunas (ICCAT). Moreover, Benin and Cameroon have signed the MoU concerning the conservation of manatees and small cetaceans of Western Africa and Macaronesia to complement the Bonn Convention on the flip side. The three countries have also ratified the Convention on International Trade in Endangered Species (CITES) and the Cartagena Protocol on Biosafety.

Likewise, Benin and Cameroon are contracting parties to the International Tropical Timber Agreement (ITTA), Geneva, 1994, while Nigeria has not—possibly leaving Nigeria vulnerable to illegal logging of mangroves and other coastal timber species. The Ramsar Convention has also rallied the convergence of legal instruments in the three jurisdictions on coastal wetland conservation. At the same time, the ratification of the World Heritage Convention plays a significant role to protect a

[3] In accordance with Article 76, paragraph 8, of the United Nations Convention on the Law of the Sea through the Secretary-General.

[4] Minutes of Experts Meeting of ECOWAS member States on the Outer Limits of the Continental Shelf, Accra, 24–26 February 2009, Note 194/09 as part of the submission by Government of Nigeria for the Establishment of the Outer Limits of the Continental Shelf of Nigeria pursuant to Article 76, paragraph 8 of the United Nations Convention on the Law of the Sea.

[5] Including IMO convention 48 and its amendments 91 and 93 and Protocol of 1978 relating to the International Convention for Safety of Life at Sea (SOLAS).

[6] As modified by the Protocol of 1978 (MARPOL 73/78).

substantial number of heritage sites within their coastal zone are in alignment. Benin, Nigeria and Cameroon may gain from the recommendations of Article 4 of the European Union and the African, Caribbean and Pacific Group of States (ACP-EU) Agreement[7] which acknowledges and recognizes the complementary role and potential for contributions of non-state actors in the development process.

Although countries in the case studies generally see values aligning with international commitments, complying and implementing some of these commitments is taking a back foot. For example, despite Benin and Cameroon being parties to International Tropical Timber Agreement (ITTA), Geneva, 1994, illegal logging in these countries are still happening at an accelerated rate (Cannon, 2015; Teka et al., 2019). See **Table 3** for a summary of international conventions, protocols and agreements signed by countries in the case study.

Policy and Legal Framework Convergence From Marine and Coastal Related African Commitments

Due to the harmonization of several African and regional level policy instruments (including conventions, strategies, treaties and protocols), there is some degree of convergence of policy and legal frameworks between Benin, Nigeria and Cameroon (see **Table 4**). Besides promoting policy convergence, implementing several African Union (AU) conventions, protocols, treaties, and strategies emphasizes cooperation among the AU Member States. Nonetheless, the reactions of the three jurisdictions to these instruments varies significantly. The conservation of nature in Africa is within the African Convention on the Conservation of Nature and Natural Resources, 1968[8] and its 2017 revised version. While Benin and Cameroon only signed it, Nigeria has ratified the convention, signifying the policy and legal framework convergence toward ocean conservation.

Meanwhile, the formation of the African Ministerial Conference on the Environment (AMCEN) in 1985 has provided the necessary platform for environmental policy and legal framework convergence between Benin, Nigeria, and Cameroon on multilateral environmental agreements. In its objectives toward enhancing governance mechanisms for ecosystem-based management of the African ocean, the AMCEN has repeatedly called on African countries to fulfill their ocean-related commitments. For instance, Benin, Nigeria, and Cameroon are among the African countries through the AMCEN, which adopted 11 resolutions to accelerate action strengthen partnerships on marine litter microplastics at the third meeting of the UN Environment Assembly (UNEA) held in December 2017 in Nairobi (AMCEN, 2019).

[7]The ACP-EU Partnership Agreement signed in Cotonou on 23 June 2000. Since 2000, it has been the framework for EU's relations with 79 countries from Africa, the Caribbean and the Pacific (ACP). In 2010, ACP-EU cooperation has been adapted to new challenges such as climate change, food security, regional integration, State fragility and aid effectiveness. The Agreement entered into force in April 2003 and has been revised in 2005 and 2010 in accordance with the revision clause to re-examine the Agreement every 5 years. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=URISERV:r12101.

[8]African Convention on the Conservation of Nature and Natural Resources September 15, 1968 https://www.au.int/web/sites/default/files/treaties/7763-sl revised_african_convention_on_the_conservation_of_nature_and_natural_resources_18.pdf.

TABLE 3 | Summary of international conventions, protocols, and agreements signed by countries in the case study (Data source: Source: United Nations [UN], 2021).

	Atmosphere and Climate Change			Marine environmentrelated law		Marine living resources				Nature Conservation				
Countries	**UNFCCC[1]**	**PA[2]**	**KP[3]**	**LC[4]**	**UNCLOS[5]**	**ACAP[6]**	**CMS[7]**	**ICCAT[8]**	**ICRW[9]**	**CBD[10]**	**CITES[11]**	**ITTA[12]**	**RAMSAR[13]**	**WHC[14]**
Benin	R	R	X	P	R	X	P	X	P	R	A	P	R	R
Cameroon	R	R	X	X	R	X	P	X	P	R	A	P	R	R
Nigeria	R	S	X	P	R	X	P	P	X	R	R	X	R	R

S means that the Convention has been signed; R indicates ratification; A means accession to the Convention; P means party; X means not applicable.
P indicates Parties to the convention.
[1]*United Nations Framework Convention on Climate Change. New York, May 9 1992.*
[2]*Paris Agreement. Paris, December 12 2015.*
[3]*Kyoto Protocol to the United Nations Framework Convention on Climate Change. Kyoto, December 11 1997.*
[4]*Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention), London, 197.*
[5]*United Nations Convention on the Law of the Sea (UNCLOS) 1982.*
[6]*Agreement on the Conservation of Albatrosses and Petrels (ACAP) Hobart 2001.*
[7]*Convention on the Conservation of Migratory Species of Wild Animals (CMS), Bonn, 1979.*
[8]*International Convention for the Conservation of Atlantic Tunas (ICCAT), Rio de Janeiro, 1966.*
[9]*International Convention for the Regulation of Whaling (ICRW), Washington, 1946.*
[10]*Convention on Biological Diversity (CBD), Nairobi, 1992.*
[11]*Convention on the International Trade in Endangered Species of Wild Flora and Fauna (CITES), Washington DC, 1973.*
[12]*International Tropical Timber Agreement (ITTA), Geneva, 1994.*
[13]*Convention on Wetlands of International Importance, Ramsar, 1971.*
[14]*World Heritage Convention, Paris, 1972.*

TABLE 4 | Summary of Regional conventions, protocols, and agreements signed by Benin, Nigeria, and Cameroon countries (Data source: African Union [AU], 2021).

	Related African Conventions, Protocols, and agreement										Related Regional conventions, agreements and MoUs		
Countries	**ACCNNR[1]**	**CTMHW[2]**	**AMTC[3]**	**ACCNNR Revised[4]**	**AUCPCC[5]**	**ACDEG[6]**	**RAMTC[7]**	**AUCBC[8]**	**ACLGLD[9]**	**Abidjan Convention[10]**	**CPSIDF[11]**	**MoUCMSC[12]**	**AMoUPSC[13]**
Benin	S	R	S	S	R	R	S	S	X	R	S	S	P
Cameroon	R	R	S	X	S	R	X	X	X	R	X	S	P
Nigeria	R	S	R	S	R	R	X	X	X	R	S	X	P

S means that the Convention has been signed; R indicates ratification; A means accession to the Convention; P means party; X means not applicable
[1] African Convention on the Conservation of Nature and Natural Resources of September 15, 1968.
[2] Bamako Convention on the Ban of the Import into Africa and the Control of Transboundary Movement and Management of Hazardous Wastes within Africa of January 30.
[3] African Maritime Transport Charter of June 11, 1994.
[4] African Convention on the Conservation of Nature and Natural Resources (Revised Version) of July 01, 2003.
[5] African Union Convention on Preventing and Combating Corruption, of July 01, 2003.
[6] African Charter on Democracy, Elections and Governance of January 30, 2007.
[7] Revised African Maritime Transport Charter of July 26, 2010.
[8] African Union Convention on Cross-Border Cooperation (Niamey Convention) of June 27, 2014.
[9] African Charter on the Values and Principles of Decentralization, Local Governance and Local Development of June 27, 2014.
[10] The Convention for Cooperation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region of March 23, 1981.
[11] Convention on the Pooling and Sharing of Information and Data on Fisheries in the Zone of the Fisheries Committee for the West Central Gulf of Guinea of March 12, 2015.
[12] The Memorandum of Understanding (MoU) Concerning the Conservation of the Manatee and Small Cetaceans of Western Africa and Macaronesia of October 3, 2008.
[13] Abuja Memorandum of Understanding on Port State Control for West and Central African Region of October 10, 2019.

There is no consistency in the jurisdictions' commitment toward embracing African instruments concerning maritime transportation at different stages. The 1994 and 2010 versions of the African Maritime Transport Charter have low acceptance of accession and ratification. The Nigerian government has signed the 1994 version but has not acted on the 2010 version[7]. However, Benin and Cameroon have not reacted to either version of the Charter. The overall reaction of the Jurisdictions to this Convention shows absolute disregard of the jurisdictions' responsible authorities toward the plight of maritime transport, especially when the region is wallowing in the dismal affront of maritime piracy and related vices.

There is a significant level of policy convergence on information and data sharing, particularly in Benin and Nigeria. They are signatories to the Convention on the pooling and sharing of information and Data on Fisheries in the Zone of the Fisheries Committee of the West Central Gulf of Guinea. With this Convention, the three countries adopted a set of strategic objectives to ensure consistency in fisheries data and information to aid collaboration, joint-fact finding and decision making. The Jurisdictions being parties to the Abidjan Convention[9] allows them to work in tandem on coastal and marine issues, as the Convention provides the necessary platform for them to collaborate through the Conference of Party (CoP) and activities of the Focal Points. The Convention also commits the Jurisdictions to protect and manage their adjoining marine and coastal environment. Similarly, the Abuja Memorandum of Understanding on Port State Control for the West & Central African Region also binds the countries to adhere to crew adequacy and vessels best maintenance incompliance with the requirements of international conventions, such as SOLAS, MARPOL, etc.

Policy and Legal Framework Convergence From Ocean Related Sub-Regional Policies and Legal Instruments

Since the 1970s and early 1980s, the topics for regional-scale policies, protocols and actions have developed in West and Central Africa, either paralleling global/African environmental protection instruments or considering characteristic sub-regional challenges. Several aligned policies and legislations are in operation to enhance joint management and governance of ocean space within and across the two sub-regions. These policies and legislations have aims and objectives that stresses the move toward more integrated approaches. They address cross-cutting challenges, including security, fisheries, conservation, climate change, research and development, ocean renewable energy, etc. (see **Figure 4**).

More recently, some integrated policies have taken on goals for a sustainable ocean environment. In the ECCAS sub-region, the 2009 ECCAS Protocol on the Strategy to Secure Vital Interests at Sea aims to protect natural resources and artisanal maritime fisheries zones maritime routes and fight against illicit naval activities (ECCAS, 2009). Similarly, in the ECOWAS sub-region, the Integrated Maritime Strategy follows the AU Integrated Maritime Strategy (AIMS). There is a convergence in these two instruments as both focus on maritime security and identify the maritime domain's significant challenges and a set of comprehensive priority actions needed for a prosperous, safe and peaceful marine environment at the national and sub-regional level.

A convergence of policy approaches centered around an integrated regional maritime security architecture within the two sub-regions is also strongly noticeable. For example, the 2008

[9] Convention for Cooperation in the Protection, Management and Development of the Marine and Coastal Environment of the Atlantic Coast of the West and Central Africa Region.

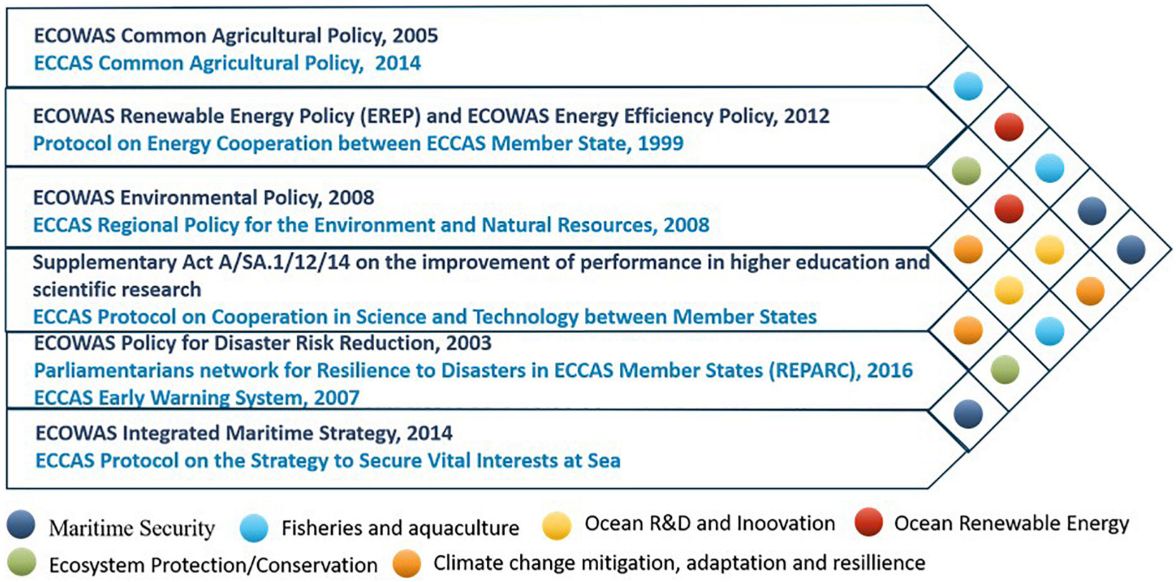

FIGURE 4 | Selected policies and legislations within ECOWAS and ECCAS with a matrix of their sector/activities convergence.

Memorandum of Understanding (MoU) on the Establishment of a Sub-regional Integrated Coast Guard Network in West and Central Africa signed by 14 Members of the Maritime Organization of West and Central Africa (MOWCA), laid down the framework to promote regional maritime cooperation, safety, law and order and surveillance for West and Central Africa (IMO/MOWCA, 2008). Additionally, the adoption of the 2013 Code of Conduct (CoC) concerning the repression of piracy, armed robbery against ships, and illicit maritime activity in West and Central Africa, also known as the Yaoundé CoC emphasises cooperation and information-sharing across the region as a panacea for addressing an array of maritime crimes.

On the environmental front, there are points of convergence between policies and plans in the ECCAS and ECOWAS sub-region on agriculture, environment, energy, research and development, etc. These policies and plans often have similar implementation strategies on management and governance objectives for coastal and marine spaces. For example, the ECOWAS Common Agricultural Policy adopted in 2005 includes two supplementary plans (Regional Agricultural Investment Plan and Food & Nutrition Security and the 2025 Strategic Policy Framework adopted 2016) with a high commitment to the maritime and continental fisheries/aquaculture sector. It promises to ensure a modern, competitive, inclusive, and sustainable fisheries sector to accelerate economic prosperity, guarantee decent jobs, and ensure food security (ECOWAS, 2017). Similarly, in the ECCAS sub-region, the 2014 Regional Common Agricultural Policy allowed reframing a set of strategies and programs (e.g., the Regional Program for Agricultural Investment, Food and Nutrition Security). Also, these two instruments' strategies and plans have similar focus on several topics, including fisheries. For the fisheries sectors, they both envisage a modern, competitive, inclusive, and sustainable sector to accelerate economic prosperity, guarantee decent jobs, and ensure food security (PDDAA, 2017).

Developing national platforms for cooperation, promoting and expanding various early warning systems, coordination and harmonization, and supporting public awareness advocacy are significant issues of interest in the existing ECOWAS and ECCAS sub-regional policies. For instance, the ECOWAS Policy for Disaster Risk Reduction, adopted in 2006, focuses on reducing disaster risks through development interventions by managing disaster risks as a development challenge. In response to disaster risk, the Parliamentarians Network for Resilience to Disasters in Central Africa was inaugurated in 2016 by the ECCAS in a drive to curb the impact of natural and human-made hazards by implementing the Sendai Framework for Disaster Risk Reduction. These policies and actions are essential for the two sub-regions, given that climate change risks pose a particular threat to coastal communities from increased marine erosion, sea flooding, and landslides (UNESCO/IOC, 2020b).

Although some of the available policies and instruments acknowledge integrated resources management principles, their implementation does not abide by these principles in practice. Likewise, a look into some of the policy documents shows their limitation to goals concerning resource exploration/exploitation and control, projections of future demands, or more on the needs for the financing of developmental projects. For example, the ECOWAS Renewable Energy Policy (EREP) and Protocol on Energy Cooperation between ECCAS Member States did not address critical issues that bother socioeconomic justice, such as equitable energy distribution.

Despite some of their lapses, the presented policies and legislations in the two sub-regions are starting points in determining the need to revise current laws, promulgating ocean-related regulations, or taking other steps to implement ocean laws effectively. Also, they are necessary to catalyze the creation of new legislative and institutional arrangements that accommodate novel policy prescriptions as the policies are periodically revised.

Convergence of National Institution, Planning and Policy Frameworks in Benin, Nigeria, and Cameroon

Constitutions typically outline a broader set of pronouncements for which implementation mechanisms are less exact (Lijphart, 2004). It often needs to be translated into laws and policies to have a widespread impact on citizens' lives. Constitutional provisions in Benin, Nigeria, and Cameroon revealed mechanisms for the legal enforcement and fundamental building blocks of government and laws for ocean-related concerns. Their commitments remain relatively stable and permanent even as different political parties assume power, which can help guard against governments' attempts to remove or weaken national coastal and marine management commitments.

Most of the recent sectoral laws on the countries' environment are derived from colonial laws, specifically from early 20th century English and French laws. These laws primarily deal with natural resource extraction to facilitate exploitation more than protection (Kameri-Mbote and Cullet, 1997). Questions arise as to the capacity of these laws to deal with traditional health and natural resource problems, let alone deal with new issues and needs not contemplated when the laws were initially enacted. However, several critical legal instruments exist directly or indirectly to the countries' management and control of coastal and maritime environments. As parties to UNCLOS, Benin, Nigeria, and Cameroon have sovereign rights over their EEZ, including soil and subsoil of their extended continental shelf. Various legal instruments are in place in the countries following UNCLOS's requirement for their Territorial Sea, Contiguous Zone and EEZ. Others bolster all aspects of sustainable development and align with objectives and goals for management on fisheries and aquaculture; conservation and environmental protection; coastal protection, waste management, land-use and development control; rural development. These legislations are in the form of Acts, Regulations, Orders, and Decrees, whose implications are clear if implemented and enforced correctly. After all, there are certain disadvantages of creating new coastal and marine management (including time-consuming, flexibility, undesired outcomes, and decreased political support) legislations, especially when considering Marine Spatial Planning (MSP) (IOC-UNESCO, 2009b).

Meanwhile, in the absence of a stand-alone national ocean policy in Benin, Nigeria and Cameroon some regulatory measures for managing coastal and marine resources are in place. These include issuing fishing, logging and mangrove harvesting permits, etc. – even though most of these have proven ineffective for various reasons (see, e.g., Ukwe and Ibe, 2010; Diop et al., 2012; Barnes-Dabban and Karlsson-Vinkhuyzen, 2018). Increasingly, the countries are enacting sectoral policies that can provide practical frameworks at the national level to implement ecological standards and regulate socio-economic activities in the light of sustainable development objectives.

In Benin, Nigeria, and Cameroon's ocean domain, the numerous pieces of sectoral policies and enacted legislative instruments of governance are devised, administered and enforced by a wide range of formally established institutions. Most departments within central government ministries, statutory authorities, or cabinet appointed multi-sectoral steering committees to manage single or multiple facets of the ocean and coastal sphere. Though sectoral in approach, these institutional frameworks for governance and management of ocean activities and resources are comprehensive. Coastal and marine management is mostly saddled on the environment and transportation's ministries with interwoven responsibilities for the three countries (see **Table 5**). Together with their various departments, the environment ministries oversee marine environmental protection, adherence to international, regional and national regulation and implementation of national policies and programs. In Nigeria, these institutions are also replicated at the state and local government levels and backed up by laws aligned with national legislation and policies. The ministry of transportation in Nigeria and Cameroon is responsible for activities that have to do with shipping, port development, and transportation. In Benin, this responsibility is carried out by the Ministry of Maritime Economic, with obligations mainly on transport and port infrastructure. Apart from the various government institutes and universities, the live wires of marine and coastal research and technical support are the the national institutes for oceanographic and marine research of the different countries. There are also some national NGOs serving as pressure groups to advance sustainable development.

Besides various spatial and territorial planning instruments in the case studies, there are few dedicated national legal frameworks for ICZM. Decree No. 86-516 of 1986 defining responsibilities for coastal management and law No 2018-10 of the 2 of July 2018, on the protection, development and theft of the coastal zone in Benin are in place to guide the ICZM process, with a proposition of inter-ministerial participation. In Nigeria, the National Coastal and Marine Area Protection Regulations, 2011 (S.I. No. 18 of 2011) and the National Wetlands, Riverbanks and Lake Shores Protection Regulations, 2009 (S.I. No. 26 of 2009) gives the Federal Ministry of Environment the coordination responsibility to develop and implement ICZM. However, there are existing comprehensive policies to realize ICZM at the national and regional levels in Cameroon. These include the National Action Plan (NAP) for Marine and Coastal Area Management (November 2010), the Management Plan of the Campo Ma'an National Park, and Kribi Campo Coastal Zone Management for Sustainable Tourism Development.

Concerning MSP, a mismatch of ministries has related competencies in the three countries. Based on several legal and essential institutional tools, developing and implementing MSP lies in an inter-ministerial arrangement. There exist several overlaps in mandates related to marine protection, development

TABLE 5 | The institutional and legal framework in the case studies.

	Benin	Nigeria	Cameroon
Level of responsibility for ocean governance	Central Government	Federal Government	Central Government
Responsible ministry for ocean governance	Ministry of Environment and Protection of Nature Ministry of National Defence Ministry of Urban Development, Land Reform and Erosion Prevention	Ministry of Environment, Ministry of Transportation, Ministry of Defence	Ministry of Transportation, Ministry of Environment
Legal and essential institutional tools for ocean governance l	Constitution of the Republic of Benin 1990, Law No 2010–March 11 07, 2011, on the maritime code in the Republic of Benin, The National Development Plan 2018–2025 (PND)	Nigeria constitution 1999, Exclusive Economic Zone Act (Cap. T.5), 2013, Territorial Waters (Amendment) Decree 1998; National Policy on the Environment (Revised 2016). Nigeria Agenda 2050 and the Medium-Term National Development Plan (MTNDP) 2021–2025	Constitution of Cameroon 2008, Strategy paper for growth and jobs (2010–2020); Poverty Reduction Strategy Paper, 2008, Law n° 39 PJL/AN of November 20 1974 fixing the limit of Cameroon's territorial waters.
Level of responsibility for coastal planning	The central government, Municipality	Federal and State government	Central Government
Responsible ministry for coastal planning	Ministry of Transport, Ministry of Finance and Economy; Ministry of Environment and Protection of Nature, Ministry of Tourism, Ministry of National Defence; Ministry of Mines, Energy and Water	Federal Ministry of Environment, Department of Erosion, Floods and Coastal Zone Management, Coastal Zone Division Federal and States Ministry of Physical Planning and Urban Development[1]	Inter-ministerial
Legal and essential institutional tools for coastal planning	Decree No. 86-516 of 1986 defining responsibilities for coastal management; Law No 2018–July 10 2, 2018, on the protection, development, and theft of the Republic of Benin's coastal zone.	Nigeria constitution 1999, Exclusive Economic Zone Act (Cap. T.5), 2013; Territorial Waters (Amendment) Decree 1998; Landuse Act of 1978; The Nigeria Urban and Regional Planning (Decree No. 88, 1992)[2]; National Environmental Standards and Regulations Enforcement Agency (Establishment) Act, 2007 (No. 25 of 2007); National Environmental (Coastal and Marine Area Protection) Regulations, 2011 (S.I. No. 18 of 2011); National Environmental (Wetlands, River Banks and Lake Shores Protection) Regulations, 2009 (S.I. No. 26 of 2009); Sea Fisheries Act, 1992. Date of original text: December 31 1992 (February 28 2013), National Policy on the Environment (Revised 2016).	Constitution of Cameroon of 2008 Law n° 96/12 of August 5 1996 on a framework law on environmental management in the Republic of Cameroon; The National Action Plan (NAP) for Marine and Coastal Area Management (November 2010), National Poverty Reduction Strategic Document; Regional Development and Management Master Plan, Management Plan of the Campo Ma'an National Park; Forest Environment Sector Programme; Kribi Campo Coastal Zone Management for Sustainable Tourism Development
Level of responsibility for maritime planning	Central government	Federal Government	Central Government
Responsible ministry for maritime planning	Ministry of Environment and Protection of Nature; Ministry of National Defence; Ministry of Urban Development, Land Reform and Erosion Prevention; Ministry of Transport	Inter-ministerial	Inter-ministerial
Legal and essential institutional tools for Maritime Spatial Planning	Constitution of the Republic of Benin 1990, Law No 2010-March 11 07, 2011, on the maritime code in the Republic of Benin, The National Development Plan 2018-2025 (PND Decree No. 86-516 of 1986 defining responsibilities for coastal management; Law No 2018-July 10 2, 2018, on the protection, development, and theft of the Republic of Benin's coastal zone. Decree 2015-029 of January 29, 2015, fixing the modalities of acquisition of rural land in the Republic of Benin	Nigeria constitution 1999, Exclusive Economic Zone Act (Cap. T.5), 2013; Territorial Waters (Amendment) Decree 1998; Landuse Act of 1978; The Nigeria Urban and Regional Planning (Decree No. 88, 1992)[3]; National Environmental Standards and Regulations Enforcement Agency (Establishment) Act, 2007 (No. 25 of 2007); National Environmental (Coastal and Marine Area Protection) Regulations, 2011 (S.I. No. 18 of 2011); National Environmental (Wetlands, River Banks and Lake Shores Protection) Regulations, 2009 (S.I. No. 26 of 2009); Sea Fisheries Act, 1992. Date of original text: December 31 1992 (February 28 2013), National Policy on the Environment (Revised 2016); Nigeria Agenda 2050 and the Medium-Term National Development Plan (MTNDP) 2021-2025	Constitution of Cameroon of 2008 Law n° 96/12 of August 5 1996 on a framework law on environmental management in the Republic of Cameroon; Presidential Decree No 99/195 of September 10, 1999, establishing the Ocean Division Development Authority, supplemented by law No 99/016 of December 22, 1999. The National Action Plan (NAP) for Marine and Coastal Area Management (November 2010), National Poverty Reduction Strategic Document; Regional Development and Management Master Plan, Management Plan of the Campo Ma'an National Park; Forest Environment Sector Programme; Kribi Campo Coastal Zone Management for Sustainable Tourism Development; Strategy paper for growth and jobs (2010–2020); Poverty Reduction Strategy Paper, 2008; Vision 2035 Plan, 2009. National Development Strategy 2020–2030

(Continued)

TABLE 5 | (Continued)

	Benin	Nigeria	Cameroon
Level of responsibility for Blue Economy	Central government	Federal Government	Central government (Presidency)
Responsible ministry for Blue Economy	Inter-ministerial	Inter-ministerial	Presidency, Ocean Division Development Authority (MEAO)
Legal and essential institutional tools for the Blue Economy	Constitution of the Republic of Benin 1990; The National Development Plan 2018–2025 (PND)	Nigeria constitution 1999, National Policy on the Environment (Revised 2016). Nigeria Agenda 2050 and the Medium-Term National Development Plan (MTNDP) 2021–2025	Constitution of Cameroon 2008, Strategy paper for growth and jobs (2010–2020); Poverty Reduction Strategy Paper, 2008. Decree No 99/195 of September 10, 1999, supplemented by law No 99/016 of December 22, 1999. It is since January 7, 2002; Vision 2035 Plan, 2009. National Development Strategy 2020–2030

[1] *Each of the 36 states has its own environmental protection laws an legislation.*
[2] *Each States has its own Urban and Regional Planning laws and regulations.*
[3] *Each States has its own Urban and Regional Planning laws and regulations.*

and administration. In the three countries, the inter-ministerial arrangement involves ministries responsible for the:

- Management and protection of inland waters, prevention of pollution and the protection of the sea and coastal.
- Spatial and physical planning.
- National Defence; Ministry of Urban Development, Land Reform & Erosion Prevention; and Ministry of Transport the three countries.
- Infrastructure, transport and management of maritime properties of national interest.
- Coordination of policies on food, forestry, aquaculture and fisheries.

Besides countries' constitutions in the case studies, various high-level national policies are the basis for Blue Economy development (see **Table 5**). For example, the National Development Plan 2018-2025 of Benin has one of its objectives "to make agro-industry and services the engine of inclusive and sustainable economic growth within the framework of more effective national and local governance by focusing on the development of human capital and infrastructure." This it plans to achieve by consolidating the rule of law and good governance; ensuring the sustainable management of the living environment, the environment, and the emergence of regional development poles; sustainably increasing the Beninese economy's productivity and competitiveness healthy, competent and competitive human capital.

With the eradication of poverty expected to be at its center, the Nigerian Medium-Term National Development Plan 2021–2025 and 2026–2030, which is currently under preparation, would invariable aid the realization of the Blue Economy in the country.

Meanwhile, the prospect for the Blue Economy development in Cameron aligns with the expectations of the Vision 2035 Plan and the National Development Strategy 2020–2030. These two strategic documents highlight Cameroon's overall policy direction and developmental pursuit, focusing on poverty reduction, becoming a middle-income country; industrialization, consolidating democracy and enhancing national unity.

Shared Experiences, Common Issues and Joint Solutions

Important Cross-Border Ocean Related Projects and Initiatives Involving Benin, Nigeria, and Cameroon

Several strategic projects and programs have set the foundation for developing transboundary ocean science capacities between Benin, Nigeria, Cameroon, and beyond. For example, the Global Environment Facility financed GCLME program introduced the countries and others to the ecosystem-based approach for marine goods and services assessment and management (GCLME-RCU, 2006). The program commits the GCLME countries to conduct transboundary marine resources assessments and support resource recovery and sustainability actions. Another important project that brought Benin, Nigeria and Cameroon together is the Monitoring of the Environment for Security in Africa (MESA) project implemented through ECOWAS and ECCAS. By providing information to relevant agencies using Earth Observation data and information products, the project helps the countries to work together to enhance coastal monitoring, improve fishery management and reduce illegal, unreported and unregulated fishing practices. Under the Ocean Data and Information Network for Africa (ODINAFRICA) project initiated by UNESCO's Intergovernmental Oceanographic Commission, Benin, Nigeria, and Cameroon have been working on ocean science and observation since 2011 (IOC-UNESCO, 2009a). The National Oceanographic Data Centre (NDC) in Nigeria coordinates Benin, Cameroon and other NDCs in the region (IOC-UNESCO, 2010).

Similarly, several projects aim to strengthen national and regional action through knitted activities and integrated approaches to accelerating integrated coastal and marine management in the three countries and beyond. This includes the Mami Wata transboundary project, which has built technical and institutional capacity for marine ecosystem-based using

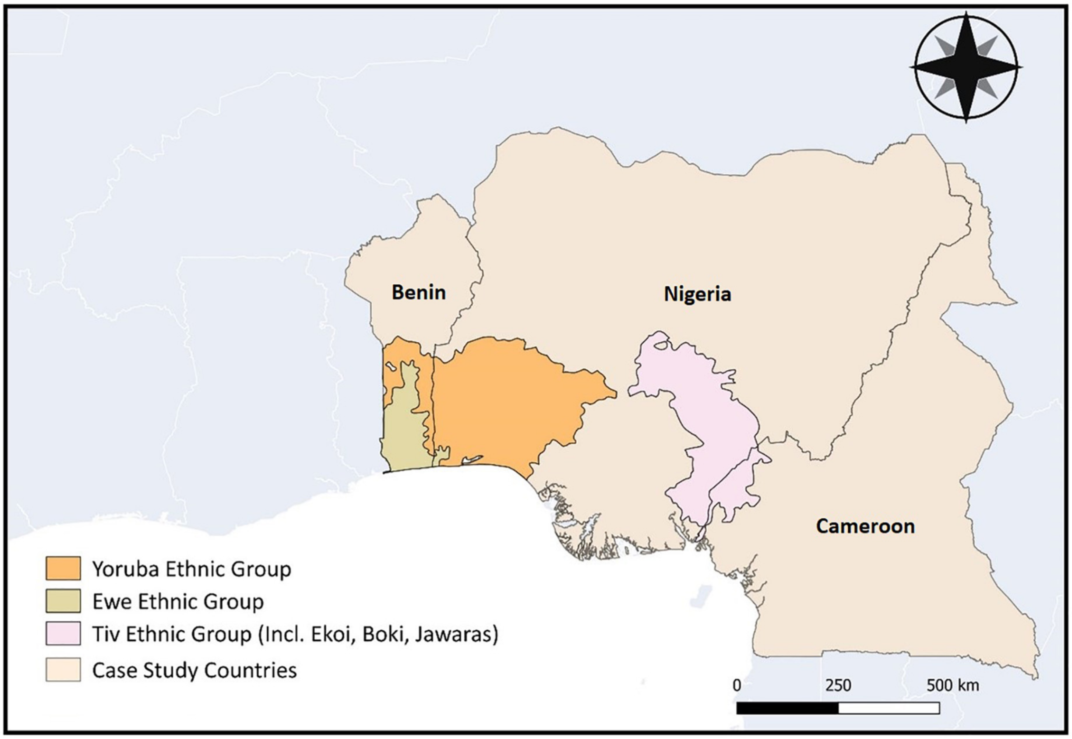

FIGURE 5 | Map showing the coastal communities in Benin, Nigeria, and Cameroon cultural and ethnic affinity (Data source: Weidmann et al., 2010).

integrated Ocean Management frameworks, including MSP. Also, the two phases of the World Bank's West Africa Coastal Areas Management Program (WACA) have improved countries' capacities to manage their growing coastal erosion and flooding problems and access expertise and finance to manage their coastal areas (World Bank, 2016). The "Enhancing Adaptation and Resilience against multi-hazards along West Africa's Coasts (EARWAC)" is a recent project supported by the European Space Agency and Future Earth and developed by The Sixth Avis Ltd, which presents an interactive dashboard (https://earwac.com/). The dashboard is built using long-term climate records derived from Earth Observation (EO) and other sources, allowing 10 West African countries (including Benin, Nigeria and Cameroon) to understand better, prepare for, monitor, and manage coastal degradation and hazards.

Historical Relationship Between Benin, Nigeria, and Cameroon

Historically, the Nigeria–Benin and Nigeria–Cameroon relationship has been that based on cultural and socio-economic nerves. Coastal communities in the three countries have similar ethnological composition and culture (Familigba and Ojo, 2013; Mark, 2015). For example, the Badagry division's people are mainly the Egun-speaking people with a direct cultural affinity with the Aja-speaking people of Benin, ditto, the Ewes in Benin and Nigeria. The language, culture and traditional administration of the people on either side of the border are identical. Similarly, the same ethnocultural stock is on both sides of the ostensible international divide between Nigeria and Cameroon (Edung, 2015; Nwokolo, 2020). The Ibibio, Efik, Ekoi, some Bantu and semi-Bantu people are the five original ethnic groups that settled at the boundary area (Njoku, 2012; see **Figure 5**).

Likewise, these three jurisdictions were among African countries affected by this colonial division between British and French rule (Omede, 2006). However, despite different western powers colonizing the countries, long-standing bilateral relations between Nigeria–Benin and Nigeria–Cameroon persist post-independence. The three countries have signed many bilateral agreements (in pair – Nigeria–Benin and Nigeria–Cameroon) relevant to ocean governance (see **Tables 6**, **7**).

To fight maritime piracy, Nigeria and Benin in 2011 set up the Operation Prosperity initiative, the first of its kind in the region, aimed at a combined maritime patrol of their waters.

Similarly, peace, environmental, socio-economic development, and other partnership forces are creating interlinked and overlapping identities that influence the form and function of the relationship between Nigeria and Cameroon. Several bilateral agreements have been signed between both countries, which governs their relationships (see **Table 6**).

The impression from the historical relationship and bilateral agreements/cooperation in these three Jurisdictions is that the relationship between the Benin and Nigeria seems more effortless than that of Nigeria-Cameroon for several reasons. Firstly, cordial symbiotic relationship between two ethnic stocks (Ewe, Yoruba, Egun) found between the coastal boundary of Benin

TABLE 6 | Post-colonial Bilateral relationship between Benin and Nigeria.

Document	Date signed	Relevance to ocean governance
The Benin–Nigeria Agreement concerning the exchange of workers		Joint fact-finding, ocean governance and management
Protocol to the Agreement on the free movement of persons	29 of May 1975	Cooperation to share information and management of (il)legal maritime migration and movement of goods at sea
Benin and Nigeria four-party agreement (including Ghana and Togo) on measures for the repatriation, deportation, safety and property of foreigners, and security in both countries.		Cooperation to share information and management of (il)legal maritime migration and movement of goods at sea
Cooperation Agreement to Prevent, Suppress and Punish Trafficking in Persons with an Emphasis on Trafficking in Women and Children (Benin and Nigeria)	9 of June, 2005	Cooperation to share information and management of (il)legal maritime migration and movement of goods at sea
Treaty providing a legal and fiscal framework for a US $500-million regional gas pipeline project.	2003	Joint maritime control and surveillance
Memorandum of Understanding (MoU) on border security and trade facilitation to combat border crimes and create a business atmosphere beneficial to both countries.	29 of March 2006	Cooperation to share information and management of (il)legal migration and movement of goods at sea, transboundary conservation of coastal resources

and Nigeria are well established till today (Babatunde, 2014). However, the same cannot be said of the Ibibio, Efik, Ekoi, and Tivs people found between the Nigeria–Cameroonian border. The implication of this to coastal and marine management is that the ease of communication and cultural acceptance that cross-border management and governance initiatives will gain with stakeholders between Benin and Nigeria would be more than that of Nigeria–Cameroon. Secondly, the fierce maritime and land dispute[10] between Nigeria and Cameroon has brought about a certain level of animosity between governments and people at the two divides even though the dispute has been settled since 2002 (Kadagi et al., 2020). Confrontations between the military,

[10]The dispute between Nigeria and Cameroon (Nigeria vs. Cameroon: Equatorial Guinea Intervening) was put to rest on October 10, 2002, following the grand judgment by the International Court of Justice ruling in favor of Cameroon.

TABLE 7 | Post-colonial Bilateral relationship between Nigeria and Cameroon.

Document	Date signed	Relevance to ocean governance
The Agreement of Friendship and Cooperation	6 of February 1963	Joint fact-finding, ocean governance and management
The Memorandum of Understanding on the control of the movement of persons and goods	6 of February 1963	Cooperation to share information and management of (il)legal maritime migration and movement of goods at sea
The cultural, social, and technical agreement	22 of March 1972	Ocean literacy, indigenous knowledge capacity building, and conservation of maritime cultural heritage
The Agreement on Police Cooperation	27 of March 1972	Joint maritime control and surveillance
Cooperation agreement	27 of March 1972	Transboundary management of coastal and marine resources
Air Services Agreement	19 of May 1978	Joint maritime control and surveillance
The Memorandum of Understanding on the transnational highway project to facilitate transportation between Cameroon and Nigeria	29 of March 2006	Cooperation to share information and management of (il)legal migration and movement of goods at sea, transboundary conservation of coastal resources
The Green Tree Agreement of	12 of June 2006	Conservation of coastal fauna and flora resources
The Cameroon–Nigeria electrical interconnection Agreement	18 of February 2011	Joint development of Ocean renewable Resources
Agreement of cooperation in the field of Sports and Physical Education	18 of February 2011	Youth empowerment and socio-economic livelihood development in coastal communities
The Agreement Establishing Cameroon-Nigeria Border Security Committee	28 of February 2012	Join management of coastal and marine resources
The Trade Agreement	6 of February 1963, revised on 13 of January 1982, and the 11 of April 2014	Maritime trade development and cooperation
Agreement of Cooperation in the fields of Science and Technology	11 of April 2014	Joint development of marine science, ocean observation, technology, and innovation capacity
Memorandum of Understanding on the implementation of the program on cooperation and cultural exchanges	11 of April 2014	Cooperation the conservation of heritage in transboundary areas
Agreement on Youth Development	11 of April 2014	Youth empowerment and socio-economic livelihood development in coastal communities

fishers, and people from Nigeria and Cameroon, particularly in the Bakassi Peninsula axis, are still periodic (BBC, 2017).

Following Flannery et al. (2015) analytical framework, this section so far has focused on examine the factors that shape ocean governance in the three case study countries, including policy convergence, the common conceptualization of planning issues, joint vision and strategic objectives, and shared experience. The following section will explore the potential of existing transboundary organizations in the GCLME to impact cross-boundary ocean governance cooperation using Kidd and McGowan's (2013) ladder of transnational partnership as an analytical framework.

Analysis of the Capacity of Existing Transboundary Organizations to Foster the Most Significant Cross-Boundary Ocean Governance Cooperation in the Guinea Current Large Marine Ecosystem

In this section, the nature of selected existing institutions with cross-border mandates in the GCLME and their capacity to foster transboundary cooperation toward sustainable coastal and marine management is analyzed using Kidd and McGowan's (2013) ladder of transnational partnership, is used to examine (see **Table 4**). As described in section "Materials and Methods," the ladder uses five 'rungs' to describe the different partnership categories, with informal partnerships at the bottom and more formalized partnerships on top. The analysis focuses on organizations in four key marine sectors, maritime security, fisheries, port and shipping, conservation and ecosystem-based management. These policy domains are selected for analysis because they represent critical sectors of activity and aspects in the GCLME and in the Jurisdictions understudy, and are likely areas of interest for cross-border ocean governance.

Maritime Security

The signing of the Yaoundé Code of Conduct in 2013 led to the formation of the Interregional Coordination Centre (ICC) based in Yaoundé, Cameroon. The center coordinates the Regional Centre for maritime Security in Central Africa (CREAMAC) located in Pointe-Noire, the Republic of Congo for the Central Africa Region, and the Regional Coordination Centre for Maritime Security in West Africa (CRESMAO) based in Abidjan, Cote d'Ivoire. Strengthening the cooperation, coordination, mutualization and interoperability of resources while ensuring maritime safety and security in the West and Central Africa region is the principal role of this center. These roles make the ICC correspond to Kidd and McGowan's description of a "Combined Organization" and "Administration sharing." Besides playing a prominent role in the emergence of the Yaoundé Code of Conduct, Member States of the GGC consults with each other and cooperate on preventing, managing and resolving conflicts that may include maritime border delimitation, exploitation of resources with their EEZs. The GCC would therefore occupy the "Combined Organization" rung on the Kidd and McGowan's ladder. For the Northwest Africa Maritime Safety and Security Agency (NWAMSA), the provision of scientific and intelligence assistance to the Member States and other Maritime Stakeholders on issues relating to the safe, secure and clean movement of maritime transport and the prevention of the loss of human lives at sea is its primary mission. Three iterations of the NWAMSA Work Plan (2008, 2009/2010, and 2011) developed a communication system and outline plan, harmonized methodologies for analytical purposes and information sharing, and a common information-sharing platform (NWAMSA, 2008). Following Kidd and McGowan's ladder, NWMSA would sit on the "Information Sharing" rung.

Fisheries

Following approval for its establishment by the directors of fisheries in Benin, Côte d'Ivoire, Ghana, Liberia, Nigeria and Togo in 2006, and the 2007 approval of the Ministers of Fisheries establishing its Convention and the Rules of Procedure establishing, the Committee of Fisheries for the West Central (FCWC) of the Gulf of Guinea (FCWC) being working to promote regional integration through practical implementation of sound fisheries initiatives. The FCWC increased its commitment to transboundary fisheries management by recently conveying the West Africa Task Force composed of representatives from its six Member States to stop illegal fishing activities and trade. Rule 15 of the FCWC's Rules of Procedure leaves the final decision-making power in the hands of the Conference of Ministers (Adewumi, 2020a). The FCWC would thus sit at the highest rung, "Combined Constitutions," and could also pass as "Agreed Joint Rule" on the Kidd and McGowan's ladder. Another organization of note relevant for the fisheries is the Ministerial Conference on fisheries cooperation among African States bordering the Atlantic Ocean (ATLAFCO), an intergovernmental organization founded in 1989 with 22 Member States covering from Morocco to Namibia. Cooperation between its Member States is fostered through two instruments: (1) the Constitutive Convention[11], which sets out the areas and modalities of Regional Fisheries Cooperation, and (2) the institutional Framework Protocol, which commits the States to actively cooperate to the sustainable management of fisheries in the region. With these instruments, ATLAFCO promotes cooperation develops coordination and harmonization of Member States' efforts and capabilities to manage fisheries resources. Member States exerts rights to influence decision making through nominees to the different ordinary and extraordinary sessions, a position mandated by ATLAFCO's general rules of procedure. Through the regional professional and institutional networks in the fisheries sector established by ATLAFCO, states also share a common platform to work together on issues of mutual concern. The *modus operandi* and responsibilities of ATLAFCO indicate that it rightly fits the "Combined Organization" and "Agreed Joint Rule" rung on the Kidd and McGawan's ladder.

[11]The Regional Convention on Fisheries Cooperation among African States bordering the Atlantic Ocean. https://www.comhafat.org/en/files/Présentation/Conventionfr.pdf.

Port and Shipping

The port and shipping sector in the GCLME provides a significant advantage for socio-economic development and the potential for the region to realize its growth ambition. Institutions and agencies with maritime administration mandate in the region are aware of this potential and engage in various transboundary cooperation forms. For example, the Maritime Organisation for the West and Central Africa (MOWCA)[12] unifies 25 countries on the West and Central African shipping range and offers a platform to cooperate on maritime security and environmental safety security. Its 2008–2010 Action Plan and 2011–2013 program saw the adoption of processes for information sharing, formation of collaborative projects, and building strategic networks by the Assembly of Ministers of Transport of Member States. The coordination responsibility of MOWCA also extends to the Port Management Association of West and Central Africa, the Union of African Shippers Councils, and the Association of African Shipping Lines, three specialized units governed its mechanisms. Therefore, MOWCA's position on the Kidd and McGowan's ladder would be between "Information Sharing" and "Joint Administration."

Conservation

Taking a holistic view of the region in terms of geography, ecosystem and governance, the Abidjan Convention stands as the regional legally binding institution for coastal and marine conservation and management within Central and West African and beyond. Through its Conference of Party and Secretariat, the role of the Abidjan Convention is to develop consultation, co-operation and actions within its jurisdiction on coastal and marine matters. The Party States have jointly signed several important protocols to the Convention, making it a critical regional platform influencing coastal and marine policies at the national level (Adewumi, 2020b). With this, the Abidjan Convention correspond to Kidd and McGowan's description of a "Combined Constitution" and "Combined Organization."

The introduction of the Monitoring for Environment and Security in Africa (MESA) under the Global Monitoring for Environment and Security and Africa (GMES and Africa) initiative[13] has strengthened partnerships between countries through two specialized technical institutions in the GCLME. These are the International Commission for Congo-Oubangui-Sangha Basin (CICOS) for the Central Africa sub-region and the ECOWAS Coastal and Marine Resources Management Centre (ECOMARINE) for the West African sub-region. With the coordination of CICOS and CECOMARINE, relevant national agencies are committed to sharing and receiving earth observation data from the satellite to enhance their early warning system on ocean conditions, thereby helping make informed conservation and management decisions. For example, a dedicated interactive web-based platform allows both users and ECOMARINE to provide information to relevant agencies in the region to enhancing coastal monitoring and improve fishery management. Likewise, CICOS has developed consolidated operational applications to monitor water heights for river navigation and the dynamics of the wetlands, thereby increasing data, knowledge and access to information for natural resources management. The commitment of these institutions to support conservation efforts in the region implies they fulfill criteria for "Information Sharing" on the Kidd and McGowan ladder.

The GCLME program has offered significant background and knowledge for implementing ecosystem-based management for the maritime domain in the region. It gave credence to the Interim Guinea Current Commission (IGCC), established in 2006 by the Abuja Ministerial Declaration for leadership and coordination of the GCLME Projects. The success of the IGCC has generated some new momentum to establish a permanent Guinea Current Commission (GCC) to oversee the sustainable development of the GCLME. Besides the financial and technical support from the Global Environment Facility (GEF), the World Bank, UNEP, UNDP, UNIDO, FAO, etc., solid political buy-in from ministers of the 16 participating countries and an array of top-notch scientists and professionals from the region with extant experience in the LME approach, the emerging GCC is poised to enhance integrated management of the GCLME region. Correspondingly, a new "protocol" has been decided at the 2nd Ministerial Meeting of the Abidjan Convention to support ecosystem-based assessment and management practices for sustainable development of the GCLME through the proposed GCC (Abe and Brown, 2020). The proposed GCC will occupy the "Combined Constitution" and "Combine Organization" rung by Kidd and McGowan ladder.

The second column on the table, "Function of cooperation," follows Glasbergen (2011) and helps us better understand the heuristic of the ocean cooperation development processes in the GCLME in terms of critical issues. In this context, the coming together of stakeholders from different countries to resolve particular or complex marine challenges and realize opportunities reflects a functional image of "joint conceptualization" and partnership for ocean governance. The highest point of this partnership function is "Changing political order" while the lowest being "Building trust, understanding capacity." **Table 8** shows that one of the steps to achieving cross-border cooperation for ocean governance in the GCLME is to have a basis for collaborative interactions between various stakeholder institutions across borders in an atmosphere of mutual trust. Organizations such as MOWCA, FCWC, PMAWCA, CICOS, and ECOMARINE have built both internal and external trust, thereby guaranteeing positive intentions of national institutions, their capacity to contribute and reaction to broader ocean governance cooperation. These organizations provide an atmosphere for ocean governance cooperation to foster because national institutions would have been used to (1) operating in a system where coastal and marine-related information is shared, and (2) partnership working where the primary goal is the creation of comparative value for

[12]Formerly the Ministerial Conference of West and Central African States on Maritime Transport (MINCONMAR) established in 1975 through the Charter of Abidjan by the General Assembly of Ministers of Transport.

[13]The Global Monitoring for Environment and Security and Africa (GMES and Africa) Support Program is a 30 million Euro joint program co-financed by the European Commission and the African Union Commission (http://gmes.africa-union.org/about-us).

TABLE 8 | Evaluation of transboundary organizations against Kidd and McGowan's ladder.

Kidd and McGowan's ladder	Function of cooperation	Transboundary institution
Combined constitution	Changing political order	Fisheries Committee of West and Central Africa Abidjan Convention Guinea Current Commission (GCC) (proposed)
Combined organization	Changing institution order	ICC (CREAMAC and CRESMAO) Ministerial Conference on fisheries cooperation among African States bordering the Atlantic Ocean (ATLAFCO) Guinea Current Commission (GCC) (proposed) Abidjan Convention
Agreed joint rules	Constituting shared system	Fisheries Committee of West and Central Africa Ministerial Conference on fisheries cooperation among African States bordering the Atlantic Ocean (ATLAFCO)
Administration sharing	Creating collaborative advantage	The Maritime Organization of West and Central Africa (MOWCA) Port Management Association of West and Central Africa Union of African Shippers Councils Association of African Shipping Lines ICC (CREAMAC and CRESMAO)
Information sharing	Building trust, understanding capacity	The Maritime Organization of West and Central Africa (MOWCA) Union of African Shippers Councils Association of African Shipping Lines Northwest African Maritime Safety and Security Agency West Central Gulf of Guinea (FCWC) Port Management Association of West and Central Africa (PMAWCA) International Commission for Congo-Oubangui-Sangha Basin (CICOS) ECOWAS Coastal and Marine Resources Management Centre (ECOMARINE)

sustainable ocean development beyond the material interest of one single country.

Values and mechanisms exhibited by transboundary organizations like MOWCA, PMAWCA, Union of African Shippers Councils, Association of African Shipping Lines, and ICC provide a basis for participating countries to explore how they work together, find common ground, and distribute opportunities and risks for ocean governance. They captures the potential for collaborative advantage for ocean governance, which could not be achieved by any of the GCLME countries working alone. In other words, participating countries can connect their ocean interest with the common objectives across the GCLME.

The joint rule system practiced by transboundary organizations such as FCWC and ATLAFCO is a tool for coordinating and resolving unforeseen contingencies. Therefore, they are, thus, capable of advancing cross-border ocean cooperation in the GCLME based on trust-building and achieving collaborative advantage. Although not legally binding, the system indicates that the rights of participating countries are covered, duties are well articulated, and there are implementation and evaluation mechanisms. It, therefore, constitutes a shared system that motivates participating countries to (1) build cross-border ocean governance cooperation and develop areas of joint working and common practice, (2) develop a coordinated approach to major cross-boundary development issues, and (3) facilitate a coordinated approach to international/regional/sub-regional obligations.

Mainstreaming cross-border ocean governance cooperation in the GCLME would mean that forms of partnership that build trust, create collaborative advantage and shared system functions are implemented on a broader scale. At this scale, cross-border ocean governance cooperation implies transcending beyond a single ocean policy area or sector to an integrated ocean governance structure that countries associate with while changing institutions order. Transboundary institutions such as the proposed Guinea Current Commission, the Abidjan Convention, FCWC, ICC, and ATLAFCO are examples of organizations that can be leveraged to achieve this type of cooperation. This is because they (1) have the legitimacy to influence how their Party States manage and govern their maritime domain, and (2) will manifest themselves in the political sphere of ocean governance actions and structure in the GCLME.

With the diversity, dynamism and complexities of the maritime domain in the GCLME, cross-border ocean governance cooperation will not only be fostered based on their merit but a more significant societal, political order. With transboundary institutions like the Abidjan Convention and the proposed GCC, ocean governance cooperation in the GCLME has undoubtedly become part of the networks that govern the society, as political power has become disperse among the Party States. As such, cross-border ocean governance cooperation from the perspective of these organizations is seen as a new political space where stakeholders come together for negotiations and deliberate on ocean issues and decide concerted action of change.

CONCLUSION

The GCLME is a highly biodiverse marine area endowed with enormous marine resources that are important for livelihood sustenance and provide significant sources of governments' GDP earnings. However, anthropogenic and natural factors pose considerable threats to the marine environment, reducing its capacity to continue performing its ecosystem services. A look at the GCLME through the lens of Benin, Nigeria, and Cameroon provides an opportunity to examine the dynamics of ocean governance mechanisms in the region and reveal challenges and opportunities for a cross-bother ocean governance cooperation. The geopolitical characterization underpinning ocean governance in the region is brought to the fore by highlighting factors that drive ocean governance, including geographical features, maritime jurisdictions, political framework, maritime framework activities, and associated pressures.

The paper further assessed the key enabling factors for transboundary planning and governance in the GCLME from the perspective of the selected case studies, looking at how (i) ocean-related instruments from international to national scales bring convergence of policies at the country level, and (ii) shared experiences, common issues and joint solutions. The convergence in policy and legislative arrangements across borders will be of utmost importance and a prime contributor to successful transboundary governance of the ocean space (Flannery et al., 2014). Strong policy convergence in maritime boundary delimitation, security and safety, and climate change adaptation and resilience are evident from commitments to various international governance mechanisms like UNCLOS, IMO, UNFCC, etc. In contrast, there is a limited policy convergence on conservation issues from obligations to international level environmental mechanisms. Meanwhile, implementing several AU and sub-regional level environmental instruments and commitment, including arrangements such as AMCEN, Abidjan Convention, and the FCWC promotes policy convergence on ocean protection, conservation, integrated resource management and data sharing in the region. Policy convergence is also visible through various ECOWAS and ECCAS instruments on maritime security, disaster early warning system, fisheries and energy. Although a dedicated national ocean governance policy does not exist so far in the countries under study, government institutions and legal instruments are necessary to galvanize ocean governance and administration. In most cases, ocean governance competencies are usually within the federal or central government's extant powers, depending on government operations in the countries. In Nigeria for instance, the overall responsibility for ocean governance, ICZM, MSP, and the Blue Economic rest on the federal government's shoulders through various competent ministries.

According to Blæsbjerg et al. (2009), documenting the expected values from transboundary marine management implies a need for signaling that countries can work together. Likewise, identifying common challenges and joint-solutions formulation can lay a solid foundation for transboundary planning (Flannery et al., 2014). There are shreds of evidence on the experience of countries in the GCLME toward working on common ocean issues and joint solutions. Firstly, this is exhibited in their participation in various cross-border marine and coastal related projects and initiatives on joint spatial planning across national borders covering parts of or even the entire EEZ of the countries as they share common concerns. These projects and initiatives are either short or long-term, usually donor-funded and focusing on capacity building for integrated ocean governance, ecosystem assessment, long term data collection, etc. Uitto and Duda (2002) and Chikozho (2015) observed that strategic, transboundary projects could build confidence and develop capacity among actors from different countries, facilitating cross-border working relations and eliminating obstacles to collaboration. Invariably, these connecting programs, projects, and initiatives are either ongoing or implemented in the GCLME and have brought a pair or all the countries in the GCLME together toward collaborative coastal and marine management. Unsurprisingly, there are signals that these collaborative projects, after all, are not inconceivable in the GCLME, as the countries already have potent pre- and post-colonial historical relationships (including social-cultural and development) with implications for cooperative ocean governance. Coastal communities in the GCLME usually share similar ethnological composition, despite the effect of colonial division between British and French rule. This is capture in terms of the language, culture and traditional administration of the people on either side of the borders. Similarly, there are existing long-standing bilateral relations countries in the GCLME on migration, trade, energy, transportation, science and technology, and education that can be leveraged to foster cross -border ocean governance cooperation in the region.

Meanwhile, an analysis of the capacity of existing regional organizations within four key marine sectors (maritime security, fisheries, port and shipping, conservation and ecosystem-based management) reveals the practicality of cross-border governance cooperation in the GCLME. These organizations either operates through a combined constitution, combined organization, agreed on joint rules, and administration sharing mechanisms. The Abidjan Convention and the proposed Guinea Current Commission appear to be the institutions that will help foster the most significant cross-boundary ocean governance cooperation in the GCLME because of their legal and political grounding. However, the conclusion is that existing organization arrangements are critical to effectuating cross-border ocean governance commitments, which may take many forms, including legally or non-legally binding alliances and supranational organizations. Flannery et al. (2014) and Sanchez and Roberts (2014) agrees that these arrangements may bring a new political order to managing a particular sea area, regardless of if they are legally binding or not. The challenge here is that although some of these organizations, particularly those designed to address sectoral ocean issues, can function across borders, their capacity to contribute to join-management and polycentric governance may be limited. This is because integrated management is broader in its intersectoral focus, and implementation is often still within the various sectoral organizations' mandate.

Establishing an effective cross-sectoral structure will enhance these organizations' effectiveness and efforts to foster integrated cross-border ocean governance cooperation in the region. However, the momentum to achieve this must be matched up with actions at national levels where the legal and institutional framework appears uncoordinated, follows a top-down management approach and is characterized by an assortment of agencies doing the same thing. Therefore, strengthening legal, policy and institutional capacity at the national level, prioritizing well-functioning institutions and legal frameworks as a basis for action supporting responsible and effective ocean management, and greater participation in transboundary ocean initiatives and structures are required. Also, long-term mechanisms to overcome and, if possible, avoid multiplicity of responsibilities are essential to addressing the impediments to inadequate human and marine natural resources management. Therefore, Education and training on integrated ocean management frameworks and concepts such such as MSP, ICZM and ocean accounting will be necessary.

Giving the transboundary nature of maritime activities and challenges in the GCLME, the MSP concept appears as a veritable go-to framework necessary to facilitate cross-border ocean governance in the region. Countries in the region are already directly or indirectly working within the MSP framework for integrated management of their maritime domain. Key regional organizations now have mechanisms (including the Abidjan Convention Draft Decision on MSP and the Convention's MSP Working Group) to mainstream MSP into their activities, while projects are now increasingly emerging. A transboundary MSP will enable the development of integrated ocean management legislation or policy at national levels, which will then be harnessed to galvanize cooperative ocean governance support at the GCLME through various regional and multilateral mechanisms. This is also in tandem with the strategic objective of the "Priority Area 1— Transboundary Maritime/Marine Spatial Planning" in the international Joint IOC/European Commission Roadmap for MSP, and in line with regional and global efforts to promote the development of strategic action plans at transboundary scale to achieve long-term sustainable use of ocean resources.

Finally, achieving a long-term ocean governance cooperation in the GCLME may warrant that specific departments, inter-ministerial committees or commissions created by ECOWAS and ECCAS. The success of such actions will depend upon the capacity of the coordinating mechanism to bring together the broadest possible range of institutions concerned with coastal and ocean management and assist them in including marine concerns in their work, rather than replacing their existing functions. However, efforts and work toward understanding cross-sectoral and cross-border ocean management and governance globally and in the GCLME are still developing. Further research is needed to comprehensively analyze ocean policy, legal and institutional frameworks' performance while identifying the main constraints and opportunities for an efficient ocean governance structure at national and regional scales.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

IJA conceived and designed the manuscript's ideas. IJA and JLS analyzed the data and sources of information. IJA, JLS, and AI-C wrote and reviewed the manuscript. All authors contributed to the article and approved the submitted version.

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