Research Topics

Ecocentric fisheries management in european seas: Data gaps, base models and initial assessments,

volume I

Edited by
Athanassios C. Tsikliras, Georgi M. Daskalov,
Georgios Sylaios, Gianpaolo Coro, Marco Scotti
and David Grémillet

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Ecocentric fisheries management in european seas: Data gaps, base models and initial assessments, volume I

Topic editors
Athanassios C. Tsikliras — Aristotle University of Thessaloniki, Greece
Georgi M. Daskalov — Institute of Biodiversity and Ecosystem Research, Bulgarian Academy of Sciences, Bulgaria
Georgios Sylaios — Democritus University of Thrace, Greece
Gianpaolo Coro — Institute of Information Science and Technologies Alessandro Faedo, Department of Engineering, ICT and Technology for Energy and Transport, National Research Council (CNR), Italy
Marco Scotti — National Research Council (CNR), Italy
David Grémillet — UMR5175 Centre d'Ecologie Fonctionnelle et Evolutive (CEFE), France

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Stephen J. Newman,
Western Australian Fisheries and Marine Research Laboratories, Australia

*CORRESPONDENCE
Athanassios C. Tsikliras
atsik@bio.auth.gr

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Editorial: Ecocentric fisheries management in European seas: Data gaps, base models and initial assessments, volume I

Athanassios C. Tsikliras[1]*, Gianpaolo Coro[2], Georgi Daskalov[3], David Grémillet[4,5], Marco Scotti[6,7] and Georgios Sylaios[8]

[1]Laboratory of Ichthyology, School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece, [2]Institute of Information Science and Technologies "Alessandro Faedo", Pisa, Italy, [3]Institute of Biodiversity and Ecosystem Research, Bulgarian Academy of Sciences, Sofia, Bulgaria, [4]Centre d'Ecologie Fonctionnelle et Evolutive, Université de Montpellier, Centre National de la Recherche Scientifique, Ecole Pratique des Hautes Etudes, Institut de Recherche pour le Développement, Montpellier, France, [5]FitzPatrick Institute, Department of Science and Technology (DST)/National Research Foundation (NRF) Excellence Centre at the University of Cape Town, Cape Town, South Africa, [6]GEOMAR Helmholtz Centre for Ocean Research Kiel, Kiel, Germany, [7]Institute of Biosciences and Bioresources, National Research Council of Italy, Sesto Fiorentino, Italy, [8]Democritus University of Thrace, Xanthi, Greece

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Editorial on the Research Topic
Ecocentric fisheries management in European seas: Data gaps, base models and initial assessments, volume I

Fisheries exploitation has historically been considered as the strongest driver of fish population dynamics (Jackson et al., 2001). Existing fisheries management practices have not always been successful in sustainably exploiting fish stocks (Froese et al., 2018) because of weaknesses in management approaches (Froese et al., 2021), the lack of consideration of ecosystem processes (Skern-Mauritzen et al., 2016), improper management implementation (Schnute et al., 2007), harmful government subsidies (Sumaila et al., 2021), and illegal, unreported and unregulated fishing (Agnew et al., 2009). Moreover, data deficiencies often constrain stock assessments and ecosystem model construction, tools that provide the basis for decision-making in fisheries management (Dimarchopoulou et al., 2017). Stocks that do not exhibit commercial interest have generally been given a low priority (Tsikliras et al., 2021), and the conservation status of marine megafauna is unknown in many European areas, especially in the Mediterranean and the Black Sea (Gremillet et al., 2022). Commercial and recreational fishing strongly influence all levels of biological organisation, from populations to ecological communities, directly affecting ecosystem structure and function (Pauly et al., 1998). Such circumstances raise the pressing need for a paradigm shift from the anthropocentric (=human-centred) perception of commercial stock surplus yield (Chapman, 1949) to an ecocentric (=ecosystem-centred) fisheries management that incorporates all ecosystem components, including abiotic and socio-economic factors (Link, 2010), aiming at rebuilding fish stocks, and restoring ecosystems and habitats (Lotze et al., 2011). This whole-ecosystem approach to managing fisheries aspires to ensure a balance between food security and healthy seas (Schiller et al., 2018), secure fisheries jobs (Teh and Sumaila, 2013), and even minimising exploitation while prioritising sustainable fishing practices that feed people (Jacquet and Pauly, 2022).

The design of interactive policies that aim to implement the ecocentric fisheries management is better served through a holistic and integrated perspective to maintain ecosystem integrity. They should be supported by the adoption of a precautionary approach to fisheries with broad stakeholder participation, while also promoting sector integration (Pikitch et al., 2004). This will require interdisciplinarity among scientists to achieve the best possible knowledge of biotic, abiotic, and human components of ecosystems and their interactions (Cury, 2004), as well as information on social and economic factors that affect the availability of ecosystem services within marine ecoregions (FAO, 2021).

Understanding the biology and ecology of most species within an ecosystem is, in some areas, far from complete, and appropriate assessment models and decision-support tools need to be further developed. Sources of uncertainty, such as ecological and biological knowledge gaps and future climatic conditions must be accounted for (Link et al., 2012). Basic biological data are essential for stock and vulnerability assessments as well as for ecosystem models that facilitate the understanding of ecosystem functioning and responses to change and can, therefore, inform ecosystem-based fisheries management (EBFM: Daskalaki et al., 2022). Measuring the human impact on organisms, habitats and ecosystems (Piroddi et al., 2017), assessing the status and trends of fisheries and stocks (Froese et al., 2018), and evaluating ecosystem health using novel indicators (Link and Watson, 2019) and technologies (Coro et al., 2013) are also prerequisites for effective ecocentric fisheries management. These measures, in fact, provide the baseline of current and past ecosystem status, which in turn can be used to define future management targets. Finally, ecosystem models (Christensen and Walters, 2004) are the key tool to studying marine ecosystems (Heymans et al., 2020), and exploring fisheries management and climate change scenarios by incorporating temporal and spatial ecosystem dynamics (Heymans et al., 2016).

Addressing trade-offs among the wide range of issues involved in EBFM, such as ecological principles, legal mandates, climate change and economic interests (Hart and Fay, 2020) will hopefully lead to a shift towards the perception that ecocentric fisheries management, albeit demanding and challenging, is a necessary and feasible option. In Europe, the specific objectives of fisheries management within the full ecosystem framework will have to be designed to fulfil the requirements of the Common Fisheries Policy (CFP) and the Blue Growth Strategy (BG). They will also have to aim at the Good Environmental Status (GES) as defined for the Marine Strategy Framework Directive (MSFD) descriptors and will have to be aligned with the EU Green Deal, the EU Biodiversity Strategy for 2030 and the Maritime Spatial Planning (MSP) Directive.

The present Research Topic aimed to (i) identify gaps in biological and ecological knowledge across marine ecosystem components, ecosystem models, food web models, fisheries catch statistics, and survey data; (ii) assess the current status of exploited populations; (iii) evaluate the efficiency of current fisheries management approaches within the context of ecosystem and anthropogenic impact including climate change; and (iv) create base ecosystem models across the European Seas that will be used to examine ecosystem management scenarios in the future.

The present Research Topic included six original articles, two reviews, and one policy and practice review. The six original articles focused on Mediterranean ecosystems, where data gaps and deficiencies are wider. Spatiotemporal and environmental modelling, that can be applied to areas with varying heterogeneity, was used to identify and fill gaps in trawl surveys in the Adriatic Sea (Coro et al.). In the same area, a timely manuscript examined the (beneficial) effects of the COVID-19 lockdown related fishing restrictions and the involuntary fishing effort reduction on the status of target stocks (Scarcella et al.). The impact of fisheries on ecosystem structure and functioning was examined in the Sea of Marmara over a period of thirty years (Saygu et al.), while data availability and the participatory approach for promoting fisheries sustainability were reported for the central and western Mediterranean Sea (Malvarosa et al.). Two trophic structure models were developed using the EwE modelling approach, one focusing on the oligotrophic ecosystem of the Balearic Islands in the western Mediterranean Sea (Sánchez-Zulueta et al.) and the other evaluating fisheries management policies in the alien-rich eastern Mediterranean Sea (Ofir et al.). An exhaustive review of all EwE models applied in European marine ecosystems stressed the importance of ecosystem modelling in studying the structure and dynamics of ecosystems and examining management and climate scenarios (Keramidas et al.). Another article reviewed all available biological information on non-fish marine organisms and identified gaps in knowledge across European Seas (Abucay et al.). Finally, the policy and practice review included the global and European policies and implementing bodies which have a repercussion on the implementation of EBFM, highlighting specific stakeholder needs, which ecosystem models could help address (Rodriguez Perez et al.).

Author contributions

AT: Conceptualization, Writing – original draft. GC: Writing – review & editing. GD: Writing – review & editing. DG: Writing – review & editing. MS: Writing – review & editing. GS: Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

Agnew, D. J., Pearce, J., Pramod, G., Peatman, T., Watson, R., Beddington, J. R., et al. (2009). Estimating the worldwide extent of illegal fishing. *PloS One* 4, e4570. doi: 10.1371/journal.pone.0004570

Chapman, W. M. (1949). *United States Policy on High Seas Fisheries* (Washington, D.C: Department of State Bulletin), 67–80.

Christensen, V., and Walters, C. (2004). Ecopath with Ecosim: methods, capabilities and limitations. *Ecol. Model.* 172, 109–139. doi: 10.1016/j.ecolmodel.2003.09.003

Coro, G., Pagano, P., and Ellenbroek, A. (2013). Combining simulated expert knowledge with Neural Networks to produce Ecological Niche Models for Latimeria chalumnae. *Ecol. Model.* 268, 55–63. doi: 10.1016/j.ecolmodel.2013.08.005

Cury, P. M. (2004). Tuning the ecoscope for the ecosystem approach to fisheries. *Mar. Ecol. Prog. Ser.* 274, 272–275.

Daskalaki, E., Koufalis, E., Dimarchopoulou, D., and Tsikliras, A. C. (2022). Scientific progress made towards bridging the knowledge gap in the biology of Mediterranean marine fishes. *PloS One* 17, e0277383. doi: 10.1371/journal.pone.0277383

Dimarchopoulou, D., Stergiou, K. I., and Tsikliras, A. C. (2017). Gap analysis on the biology of Mediterranean marine fishes. *PloS One* 12, e0175949. doi: 10.1371/journal.pone.0175949

FAO (2021). *Legislating for an ecosystem approach to fisheries – Revisited – An update of the 2011 legal study on the ecosystem approach to fisheries* (Rome: FAO EAF-Nansen Programme).

Froese, R., Tsikliras, A. C., Scarcella, G., and Gascuel, D. (2021). Progress towards ending overfishing in the Northeast Atlantic. *Mar. Policy* 125, 104282. doi: 10.1016/j.marpol.2020.104282

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/j.marpol.2018.04.018

Grémillet, D., Chevallier, D., and Guinet, C. (2022). Big data approaches to the spatial ecology and conservation of marine megafauna. *ICES J. Mar. Sci.* 79, 975–986. doi: 10.1093/icesjms/fsac059

Hart, A. R., and Fay, G. (2020). Applying tree analysis to assess combinations of Ecosystem-Based Fisheries Management actions in Management Strategy Evaluation. *Fisheries Res.* 225, 105466. doi: 10.1016/j.fishres.2019.105466

Heymans, J. J., Bundy, A., Christensen, V., Coll, M., de Mutsert, K., Fulton, E. A., et al. (2020). The Ocean Decade: a true ecosystem modelling challenge. *Front. Mar. Sci.* 7, 554573. doi: 10.3389/fmars.2020.554573

Heymans, J. J., Coll, M., Link, J. S., Mackinson, S., Steenbeek, J., Walters, C., et al. (2016). Best practice in Ecopath with Ecosim food-web models for ecosystem-based management. *Ecol. Model.* 331, 173–185. doi: 10.1016/j.ecolmodel.2015.12.007

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., et al. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293, 629–638. doi: 10.1126/science.1059199

Jacquet, J., and Pauly, D. (2022). Reimagining sustainable fisheries. *PloS Biol.* 20, e3001829. doi: 10.1371/journal.pbio.3001829

Link, J. S. (2010). *Ecosystem-Based Fisheries Management: Confronting Tradeoffs* (Cambridge, UK: Cambridge University Press).

Link, J. S., Ihde, T. F., Harvey, C. J., Gaichas, S. K., Field, J. C., Brodziak, J. K. T., et al. (2012). Dealing with uncertainty in ecosystem models: The paradox of use for living marine resource management. *Prog. Oceanography* 102, 102–114. doi: 10.1016/j.pocean.2012.03.008

Link, J. S., and Watson, R. (2019). Global ecosystem overfishing: Clear delineation within real limits to production. *Sci. Adv.* 5, eaav0474. doi: 10.1126/sciadv.aav0474

Lotze, H. K., Coll, M., Magera, A. M., Ward-Paige, C., and Airoldi, L. (2011). Recovery of marine animal populations and ecosystems. *Trends Ecol. Evol.* 26, 595–605. doi: 10.1016/j.tree.2011.07.008

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. Jr. (1998). Fishing down marine food webs. *Science* 279, 860–863. doi: 10.1126/science.279.5352.860

Pikitch, E. K., Santora, C., Babcock, E. A., Bakun, A., Bonfil, R., Conover, D. O., et al. (2004). Ecosystem-based fishery management. *Science* 305, 346–347. doi: 10.1126/science.1098222

Piroddi, C., Coll, M., Liquete, C., Macias, D., Greer, K., Buszowski, J., et al. (2017). Historical changes of the Mediterranean Sea ecosystem: modelling the role and impact of primary productivity and fisheries changes over time. *Sci. Rep.* 7, 44491. doi: 10.1038/srep44491

Schiller, L., Bailey, M., Jacquet, J., and Sala, E. (2018). High seas fisheries play a negligible role in addressing global food security. *Sci. Adv.* 4, eaat8351. doi: 10.1126/sciadv.aat8351

Schnute, J. T., Maunder, M. N., and Ianelli, J. N. (2007). Designing tools to evaluate fishery management strategies: can the scientific community deliver? *ICES J. Mar. Sci.* 64, 1077–1084. doi: 10.1093/icesjms/fsm109

Skern-Mauritzen, M., Ottersen, G., Handegard, N. O., Huse, G., Dingsør, G. E., Stenseth, N. C., et al. (2016). Ecosystem processes are rarely included in tactical fisheries management. *Fish Fisheries* 17, 165–175. doi: 10.1111/faf.12111

Sumaila, R., Skerritt, D. J., Schuhbauer, A., Villasante, S., Cisneros-Montemayor, A. M., Sinan, H., et al. (2021). The WTO must ban harmful fisheries subsidies. *Science* 374, 544. doi: 10.1126/science.abm1680

Teh, L. C., and Sumaila, U. R. (2013). Contribution of marine fisheries to worldwide employment. *Fish Fisheries* 14, 77–88. doi: 10.1111/j.1467-2979.2011.00450.x

Tsikliras, A. C., Touloumis, K., Pardalou, A., Adamidou, A., Keramidas, I., Orfanidis, G., et al. (2021). Status of 74 non-commercial fish and invertebrate stocks in the Aegean Sea using abundance and resilience. *Front. Mar. Sci.* 7, 578601. doi: 10.3389/fmars.2020.578601

Ulanowicz, R. E. (1993). "Inventing the ecoscope," in *Trophic Models of Aquatic Ecosystems ICLARM Conf. Proc*, vol. 26 . Eds. V. Christensen and D. Paul. (Philippines: ICLARM).

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Filling Gaps in Trawl Surveys at Sea through Spatiotemporal and Environmental Modelling

Gianpaolo Coro [1]*, ***Pasquale Bove*** [1], ***Enrico Nicola Armelloni*** [2,3], ***Francesco Masnadi*** [2,3], ***Martina Scanu*** [2,3] ***and Giuseppe Scarcella*** [2]

[1] Institute of Information Science and Technologies (ISTI), National Research Council of Italy (CNR), Pisa, Italy, [2] Institute for Biological Resources and Marine Biotechnology (IRBIM), National Research Council of Italy (CNR), Ancona, Italy, [3] Department of Biological, Geological and Environmental Sciences (BiGeA), University of Bologna, Bologna, Italy

International scientific fishery survey programmes systematically collect samples of target stocks' biomass and abundance and use them as the basis to estimate stock status in the framework of stock assessment models. The research surveys can also inform decision makers about Essential Fish Habitat conservation and help define harvest control rules based on direct observation of biomass at the sea. However, missed survey locations over the survey years are common in long-term programme data. Currently, modelling approaches to filling gaps in spatiotemporal survey data range from quickly applicable solutions to complex modelling. Most models require setting prior statistical assumptions on spatial distributions, assuming short-term temporal dependency between the data, and scarcely considering the environmental aspects that might have influenced stock presence in the missed locations. This paper proposes a statistical and machine learning based model to fill spatiotemporal gaps in survey data and produce robust estimates for stock assessment experts, decision makers, and regional fisheries management organizations. We apply our model to the SoleMon survey data in North-Central Adriatic Sea (Mediterranean Sea) for 4 stocks: *Sepia officinalis*, *Solea solea*, *Squilla mantis*, and *Pecten jacobaeus*. We reconstruct the biomass-index (i.e., biomass over the swept area) of 10 locations missed in 2020 (out of the 67 planned) because of several factors, including COVID-19 pandemic related restrictions. We evaluate model performance on 2019 data with respect to an alternative index that assumes biomass proportion consistency over time. Our model's novelty is that it combines three complementary components. A spatial component estimates stock biomass-index in the missed locations in one year, given the surveyed location's biomass-index distribution in the same year. A temporal component forecasts, for each missed survey location, biomass-index given the data history of that haul. An environmental component estimates a biomass-index weighting factor based on the environmental suitability of the haul area to species presence. Combining these components allows understanding the interplay between environmental-change drivers, stock presence, and fisheries. Our model formulation is general enough to be applied to other survey data with lower spatial homogeneity and more temporal gaps than the SoleMon dataset.

Keywords: time series analysis, spatial interpolation, ecological niche modelling, scientific surveys, Adriatic sea, marine surveys, time series forecasting

Edited by:
Tomaso Fortibuoni,
Istituto Superiore per la Protezione e la Ricerca Ambientale (ISPRA),
Italy

Reviewed by:
Matteo Zucchetta,
National Research Council (CNR),
Italy
Maria Grazia Pennino,
Spanish Institute of Oceanography (IEO),
Spain

****Correspondence:***
Gianpaolo Coro
gianpaolo.coro@cnr.it

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1 INTRODUCTION

Understanding and estimating the status of fish stocks residing in a marine area, requires continuously collecting stock biomass and abundance samples through scientific surveys. After processing these data, scientific advice can be produced for policymakers to assess the stocks' status and prevent their depletion. Since 2000, European Member States have been collecting fisheries data in a structured way within the Data Collection Framework (DCF) multi-annual programme (JRC, 2021), and more recently under the EU-MAP programme (EUR-Lex, 2021). They advise for the EU Common Fisheries Policy (CFP) (Frost and Andersen, 2006), collect data according to national work plans, and report the results annually. In the Mediterranean context, the data are eventually analysed by fishery experts of European Regional Fisheries Management Organisations (RFMOs), such as the EU Scientific, Technical and Economic Committee for Fisheries (STECF), and the General Fisheries Commitee for the Mediterranean Sea (GFCM). The resulting recommendations are used in the CFP decision-making processes to regulate fishing activity, monitor Essential Fish Habitat conservation, and predict future resource exploitation scenarios (Rosenberg et al., 2000; Hilborn and Walters, 2013; Froese et al., 2017). The data collected within the DCF are integral to several societal challenges of the EU Programmes and the European Marine Strategy Framework Directive (MSFD) (Long, 2011). In this context, fishery-independent data can come with gaps that must be filled to improve quality and reliability. For example, biomass measurements collected through trawl surveys, across several hauls in a marine area, might miss data for some locations in specific years. These data gaps also affect the estimation of catchability during the survey - a measure of fishery efficiency - which requires that the survey protocol and locations remain constant over the years (Swain et al., 2000; Aeberhard et al., 2018). Other drivers of data biases are the possible non-uniform spatial and temporal sampling and the change of the measurement tools. Various uncontrollable causes contribute to these drivers, such as funding delays, vessel unavailability or damage, long bureaucracy, adverse weather and sea conditions, and lastly, the COVID-19 pandemic (Coro et al., 2022b).

Producing accurate and unbiased spatial time series for fishery-independent surveys is crucial to inform stock assessment models and produce valuable results for decision-makers (Maunder, 2001; Coro, 2020b). However, filling the data with stock biomass estimates requires modelling complex and complementary aspects such as (i) the spatial biomass distribution in the surveyed hauls, (ii) the historical stock presence and biomass in the unsurveyed hauls, and (iii) the environmental conditions that may have favoured or penalised the stock presence in the unsurveyed hauls (Jouffre et al., 2010). Artificial Intelligence, and in particular machine learning, can help model these factors and produce valuable estimates with measured uncertainty.

One of the most commonly used models for geospatial time series reconstruction is the Vector Autoregressive Spatio-Temporal (VAST) model (Thorson and Kristensen, 2016; Thorson, 2019). VAST combines two estimators of average density variation in space and time, modelled as two linear predictors. One predictor approximates the probability of encountering the analysed species in an unsurveyed haul, and the other approximates the expected catch rate. VAST combines these two predictors to estimate stock biomass density in the unsurveyed hauls of a specific survey year. Despite the valuable results this technique can produce (Eisner et al., 2020), it is potentially limited by (i) the exclusion of an explicit modelling of environmental aspects, (ii) the fixed prior assumptions on the predictors' shapes, and (iii) the linear approximations used. Other studies have applied statistical approaches to infer stock structure (i.e., stock abundance-at-length) from incomplete survey data. In Breivik et al. (2021), a model predicts the number of fishes per year and length class in the unsurveyed hauls. It uses a linear combination of multi-variate Gaussian functions dependent on time, location, and length class. The model assumes that each spatial distribution depends only on the previous year's distribution. The potential limitations of this modelling approach are (i) the high computational complexity to optimise the multi-variate Gaussian functions, (ii) the weak temporal dependency assumed between the spatial distributions (i.e., one year instead of long-term), and (iii) the ambitious goal to infer the full stock structure from scattered and fragmented spatiotemporal data. Other similar modelling approaches have addressed the same goal using a more complex multi-variate function modelling. For example, state-space statistical models have been used to model biomass alongside recruitment, mortality, and growth (Payne, 2010; Aeberhard et al., 2018). These models infer the principal statistical moments of their target distributions through iterative sampling (Fournier et al., 2012; Coro, 2013), but still assume a one-year time dependency between the samples. Other studies have explored - especially through machine learning modelling - long-term dependencies in non-stationary geospatial time series to predict species presence and temporal persistence, and infer species abundance (Paradinas et al., 2020; Lou et al., 2021). Several other modelling approaches assume that the ratio between the stock biomass (or abundance) in a specific haul and the total biomass remains averagely constant in the survey years. The generated biomass indexes (hereafter named *equiproportional*) are easily implementable and applicable to heterogeneous survey data. They have been used to fill gaps in the Arctic, North Sea, Norwegian Sea, and the Barents Sea surveys (Schmidt et al., 2009; ICES, 2020; Bergenius et al., 2021). Some studies have tried to enhance these approaches by better modelling the co-variation between the missed hauls and known hauls over the years (Gröger et al., 2001). Although these methodologies are widely used, they are more suited for short time series with few gaps where their basic assumptions are approximately valid.

This paper proposes a new model - made up of three machine learning and statistical sub-models - to fill gaps in the geospatial time series of stock biomass indexes collected by the SoleMon fishery-independent surveys in 2020 (Grati et al., 2013; Scarcella, 2018). SoleMon is an experimental trawl survey collecting fishery-independent data since 2005 to facilitate the sustainable management of fisheries-exploited resources in the

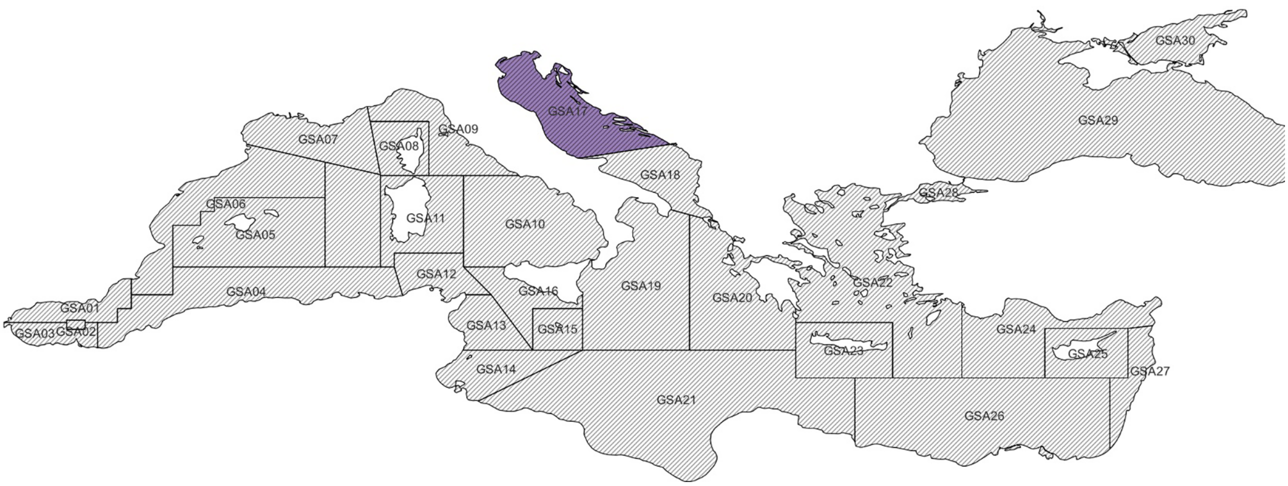

FIGURE 1 | Distribution of Mediterranean geographical subareas (GSAs) of the General Fisheries Commission for the Mediterranean, with the highlight of the GSA-17 addressed by our experiment.

North and Central Adriatic Sea, i.e., the GFCM Geographical Sub Area (GSA) 17 (FAO, 1999) (**Figure 1**). The SoleMon data presented gaps in 2020 due to unfavourable sea weather conditions and restrictions consequent to the COVID-19 pandemic, which limited research vessel availability and survey duration and constrained access to territorial waters. These restrictions prevented surveying 10 hauls out of the 67 planned in 2020. The unsurveyed hauls were mostly concentrated on the Croatian side of the Adriatic, and potentially introduced a sampling bias that could affect the overall biomass estimates (Colloca et al., 2015).

We analysed the 2020 data gaps of four Adriatic commercial stocks targeted by SoleMon: *Sepia officinalis*, *Solea solea*, *Squilla mantis*, and *Pecten jacobaeus*. To this aim, we introduced a new model to estimate the biomass-index of these stocks in the 2020 missed hauls. Our model combines three sub-models: one sub-model uses a spatial analysis of the surveyed hauls in 2020; a second sub-model processes the historical information on the missed hauls to forecast values in 2020; the third sub-model estimates the environmental suitability of the missed hauls to species persistence. We implemented the three analysis dimensions as different machine learning and statistical models and eventually combined them into one overall model. We trained the sub-models with data up to 2019. Finally, we evaluated model accuracy by forecasting 2019 known data, using data up to 2018 to train the sub-models.

The proposed model is general enough to be re-used for other areas, years, stocks and survey programmes, reconstruct data in time and space, and produce valuable information for stock assessment models.

This paper is organised as follows: Section 2 describes our model and sub-models; Section 3 reports our model's optimal parametrisation and accuracy to predict known 2019 data; Section 4 discusses the results and draws the conclusions.

2 METHODS

2.1 Model Overview

This paper proposes a machine learning and statistical modelling solution to reconstruct a biomass density index (biomass over surface, expressed in *kg/km²*) over a set of survey hauls monitored by SoleMon (**Figure 2**). We targeted 4 stocks and 10 hauls (over 67) in North-Central Adriatic that were not visited in 2020.

The premises of our experiment can be summarised as follows:

1. Scientists estimated stock biomass-index for 67 fixed-location hauls between 2006 and 2019. The 2005 survey was structured with another set of hauls and sampling plan, and was thus excluded from the analysis;
2. in 2020, biomass-index measurements were missed for 10 hauls;
3. the biomass-indexes of the previous years - with possible sporadic gaps - were available for the unsurveyed haul;
4. the survey period was always late fall.

Our goal was to estimate:

1. the biomass-index of each missed haul in 2020 for the 4 selected stock;
2. the 2020 total biomass-index for each stock, to be proposed as a fishery-independent tuning index in stock assessment models;
3. the contribution of each missed haul to the total biomass-index as an indication of the priority to survey these hauls (*haul contribution* to total biomass-index);
4. the relation between model uncertainty and haul contribution.

We propose a *haul biomass-index estimator* (HBIE) that combines three components (**Figure 3**):

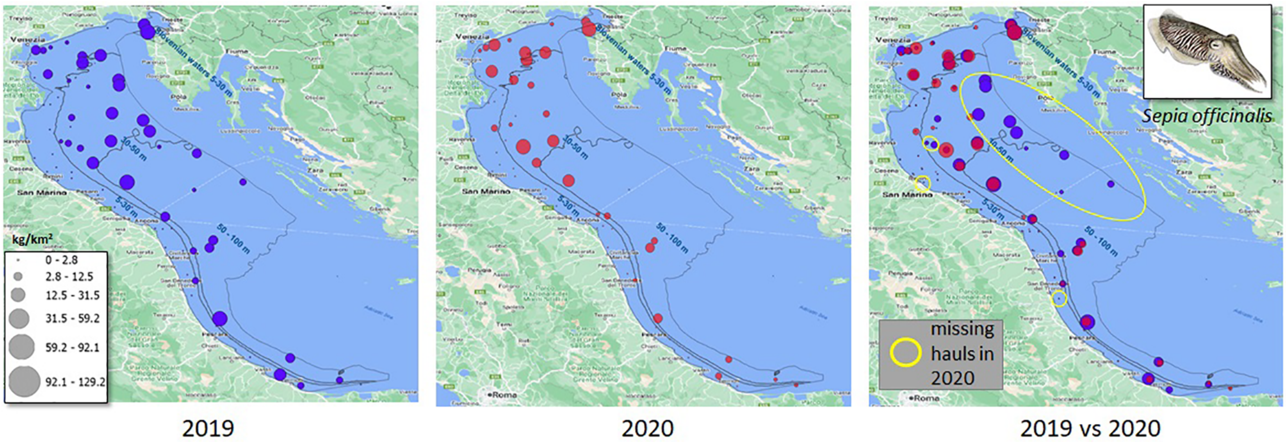

FIGURE 2 | Explanatory example of our model's scope, i.e., estimating stock biomass-index (*Sepia officinalis*, in the example) in the hauls missed by the SoleMon programme surveys in 2020. The depth strata used by the total biomass index calculation are also reported.

1. A spatial component that estimates the biomass index of a missed haul given the biomass index of the hauls surveyed in the same year. This model uses oceanographic data to estimate the spatial correlation between the surveyed hauls and the stock biomass index in the missed hauls (Section 2.3);
2. A temporal component that forecasts a missed haul's biomass-index in the analysis year based on the historical biomass-index measurements in that location (Section 2.4). Differently from alternative models, this model can also discover long-term correlations;
3. An environmental component that penalises or increments the biomass-index estimates in a missed haul by evaluating if it presents favourable environmental conditions for species presence (Section 2.5). This model represents a novelty in survey data gap filling because it hypothesises that favourable environmental conditions are key factors to compensate for fishing mortality (Froese et al., 2017).

The following sections explain how these components were implemented and combined through machine learning and statistical models and applied to the SoleMon 2020 survey data. Since independent measurements were not available for the missed hauls, model optimisation had to rely on the data at hand. Therefore, we used a *precautionary* optimisation constraint that assumed the estimated total biomass-index was not too far from the one measured in the last year (Section 2.7). Abrupt and unpredictable events of stock absence or boost from one year to the next are indeed uncommon, especially in a circumscribed area like the Adriatic (Stergiou and Pollard, 1994; Coro et al., 2016a).

2.2 Total Biomass-Index Calculation

The total biomass-index produced by the SoleMon surveys is a biomass density index (expressed in kg/km^2) based on weighted depth strata, where larger strata have higher weights. It was first introduced by Cochran (1977) and later revised by Souplet (1995). Its calculation was adapted by Grati et al. (2013) to the Adriatic by assigning specific strata weights. This process currently uses three depth strata (at 5-30°m, 30-50°m, and 50-100°m, **Figure 2**), corresponding to those where the target stocks (mostly flatfishes) are more abundant. Each stratum u is assigned a predefined and fixed weight $W(u)$ proportional to its extension. The input is the set of biomass-indexes$b(h, s, y)$ estimated by a survey campaign for each year y, target stock s, and haul h. The index is the observed biomass (in *kg*) divided by the haul swept-area (in km^2). The total biomass-index $tb(s, y)$ of stock s in year y is calculated by (i) transforming each haul biomass index into a biomass estimate through multiplication with the haul's swept area, (ii) summing all haul biomass estimates, (iii) dividing the total biomass by the total stratum area (to obtain a stratum biomass-index), and finally (iv) calculating the weighted sum of the stratum biomass-indexes. The following algorithm summarises the process:

```
Algorithm 1 Total biomass index calculation algorithm
for each stock $s$ and year $y$
for each stratum $u$
for each haul $h$
get the swept area $a(h,u)$
calculate the biomass of the haul and stratum: $B(h,u,s,y)=b(h,s,y)\cdot a(h,u)$
calculate the overall stratum biomass across the hauls: $B(u,s,y)=\sum_h B(h,u,s,y)$
calculate the overall swept area of the stratum: $A(u)=\sum_h a(h,u)$
calculate the biomass-index of the stratum: $b(u,s,y)=B(u,s,y)/A(u)$
calculate total biomass-index as the weighted sum of the strata biomass-indexes: $tb(s,y)=\sum_u b(u,s,y)\bullet W(u)$
```

The main aim of the present experiment was thus to estimate $b(h^*,s,y)$ in the hauls (h^*) missed by the SoleMon surveys in 2020, and then calculate $tb(s,2020)$ for 4 target stocks. The time series {*tb*(*s*,2006),*tb*(*s*,2007),⋯,*tb*(*s*,2019),*tb*(*s*,2020)} of the 4 stocks was meant to be proposed to the GFCM and STECF

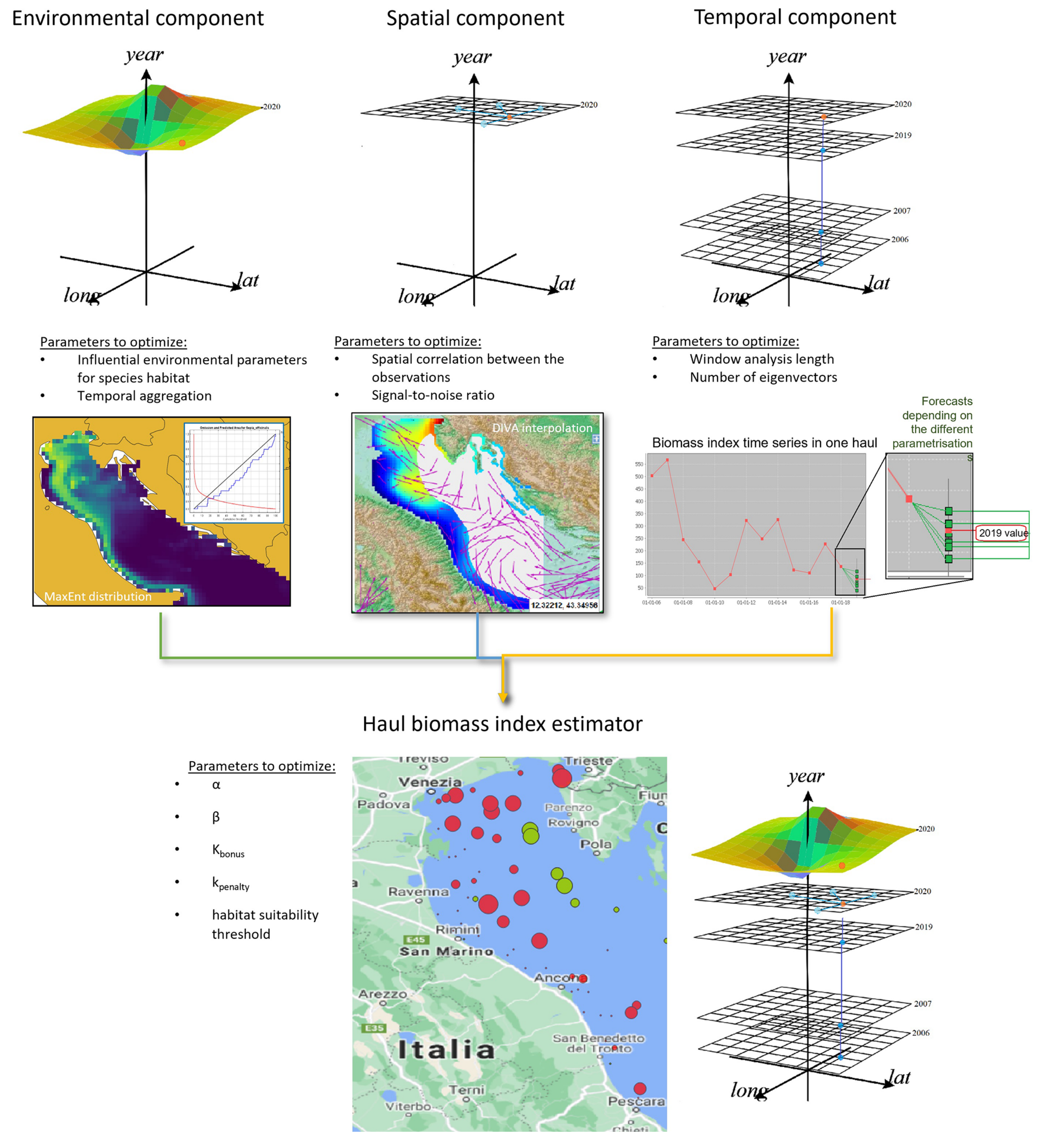

FIGURE 3 | Overview of our overall biomass-index estimation model and its three components, alongside the parameters required by each model.

working groups as a fishery-independent support to stock assessment models.

2.3 Spatial Component

Our model's spatial component estimates the stock biomass-index in the hauls missed in a specific survey year (e.g., 2020) given the biomass-index distribution in the surveyed hauls. To this aim, it interpolates the measured biomass-indexes to produce a homogeneous distribution over the area. The model assumes that the measured biomass-indexes are punctual scattered observations of a parameter uniformly defined over the analysed area. It assumes that the spatial correlation between these observations relates to the species' geographical spread, its ecological region in the water column, and the oceanic currents (Troupin et al., 2010; Watelet et al., 2016). To implement our spatial component, we used the Data-Interpolating Variational Analysis (DIVA) model (Barth et al., 2010). DIVA is typically used to estimate the uniform spatial distribution of a marine parameter from scattered observations, assuming that it is subject to currents and dependent on sea depth (Schaap and Lowry, 2010; Coro et al., 2018a; Coro and Trumpy, 2020). To this aim, DIVA solves the advection equation. As input parameters, it requires a prior estimate of the spatial correlation between the observations and the amount of noise in the data (signal-to-noise ratio) (Troupin et al., 2010; Troupin et al., 2012; Coro et al., 2016b). Internally, the model reconstructs a continuous vector field from the scattered measurements through the Variational Inverse Model (Bennett, 1992). It fits a generic continuous field to the data based on a minimization cost-function (Watelet et al., 2016). The fit algorithm is a finite-element statistical method that uses bathymetry and oceanic-current values in the observation locations as constraints. The fitted field is eventually projected on a regular spatial grid, and a triangular-element mesh is traced over the interpolation area. The characteristic length of the mesh elements is related to the spatial correlation between the input observations.

Our spatial component was a DIVA model, which we trained on the $b(h,s,2020)$ biomass-index available estimates of the SoleMon surveyed hauls in 2020 (57 values). We used the DIVA interpolated values in the 10 missed hauls h^* as the biomass-index estimates $b(h^*,s,2020)$ of the spatial component. As further input to the DIVA model, we used the 2020 annual water-column averaged oceanic-current components, as NetCDF files, from the Copernicus Global Ocean Physic Analysis (Von Schuckmann et al., 2018). Another input was a bathymetry NetCDF file from the high-resolution GEBCO-2020 dataset (GEBCO, 2020). To speed up processing, we executed the model on the D4Science cloud computing platform (Coro et al., 2015a; Candela et al., 2016; Coro et al., 2017; Assante et al., 2019; Assante et al., 2020) that freely offers the DIVA software for notebook development (Blue Cloud, 2022). The used notebooks and platform are linked in the **Supplementary Material**.

2.4 Temporal Component

Our temporal component was based on Singular Spectrum Analysis (SSA), a signal processing model to forecast time series values based on long-term sample dependency (Vautard et al., 1992). SSA decomposes the input time series into the sum of simpler time series (*hidden* components), which represent its hidden structure. It eventually combines these components to reconstruct possible gaps and project the time series in the future. For the present experiment, we used our own open-source JAVA implementation of this algorithm (Coro et al., 2016a), linked in the **Supplementary Material**.

One SSA main input parameter is the number of samples (M) of a signal window that contains sufficient information to capture the time series structure. This parameter also represents the maximum temporal dependency between the samples. The algorithm can be summarised as follows (Golyandina and Osipov, 2007; Elsner and Tsonis, 2013):

Algorithm 2 Singular Spectrum Analysis algorithm

1. divide the time series $X(t)$ (with $t_0 \leq t \leq T$) into N sub-segments (*chunks*) using an M -sample window to cut the signal sequentially;

2. build a $M \times M$ matrix so that the (i,j) element is the cross-covariance between the i th and j th chunks (*lag-covariance matrix*);

3. extract the lag-covariance matrix *eigenvectors* $\{e_1,e_2,\ldots,e_M\}$ and *eigenvalues* through matrix decomposition;

4. project the time series $X(t)$ onto the eigenvectors e_k to estimate its *components*: $a_k(t) = \sum_{j=1}^{M} X(t+j-1) \bullet e_k(j)$;

5. combine the components $\{a_1,a_2,\ldots,a_M\}$ to reconstruct the time series (including possible missing samples): $a_k(t) = \sum_{i-1}^{M} X(t+j-1) \bullet e_k(j)$; with N_t being a time-dependent normalization factor;

6. literate the process to forecast additional samples after T.

Differently from techniques based on Fourier Analysis, SSA does not use time series frequency information. This feature improves algorithm speed and allows processing also non-stationary time series (Coro et al., 2016a). The estimated eigenvectors represent the time series structure, and each eigenvalue represents the partial variance of the time series in the eigenvector direction. The sum of all eigenvalues is the time series total variance. Reducing the number of eigenvectors for reconstruction and forecast is essential to lowering data noise. The eigenvectors contain essential information about the time series, including noise, but discarding too many of them would generate trivial forecasts. The number of eigenvectors to keep for time series reconstruction and forecast is a crucial parameter to optimize.

In our experiment, the optimal SSA parameters for the time series of $b(h^*,s,t)$ values (with $2006 \leq t \leq 2019$) were found for each target stock s and missed haul h^* (Section 3.7). The process finally estimated the biomass-index forecasts $Xr(t=T+1)= Xr(2020)=b(h^*,s,2020)$. The SSA components $\{a_1,a_2,\ldots,a_M\}$ were used to fill possible gaps in the time series (which were up to one missing year for each haul) before forecasting data in the future.

2.5 Environmental Component

Our environmental component was based on the Maximum Entropy model (MaxEnt) model, a machine learning-based ecological niche model that estimates species subsistence (i.e., habitat suitability) as a function of environmental parameters (Phillips and Dudík, 2008). MaxEnt can learn from species presence locations only (i.e., without using absence information), which in our case were the hauls surveyed in the analysis year that reported non-zero biomass. We used MaxEnt to simulate the probability that a missed haul fell in suitable habitat for each analysed stock. This probability was used to set a penalty/bonus weight for the biomass estimates produced by the spatial and temporal components (Section 2.6). MaxEnt was trained on expert-identified sea-water parameters potentially correlated (either directly or indirectly) with the analysed stocks (Mancinelli et al., 1998; Zavatarelli et al., 1998; Cibic et al., 2007; Spagnoli et al., 2010; Lotze et al., 2011; Ninˇcevi´c-Gladan et al., 2015), i.e.:

1. average chlorophyll-a in the water column (mg/m^3);
2. average mole concentration of dissolved molecular oxygen in the water column (mol/m^3);
3. average moles of nitrate per unit of mass in the water column (mol/kg);
4. average moles of phosphate per unit of mass in the water column (mol/kg);
5. sea-bottom temperature (°*C*);
6. sea-surface temperature (°*C*);
7. average salinity in the water column (PSU);
8. bathymetry (m);
9. average size of grains in a sediment sample (m).

These data were mainly retrieved from Copernicus (Sauzède et al., 2017; Salon et al., 2019; Clementi et al., 2021; Feudale et al., 2021) to have spatially aligned and verified data. Bathymetry was retrieved from GEBCO-2020 (GEBCO, 2020). Grain size data belonged to CNR historically-collected Adriatic data (Santelli et al., 2017). Data were retrieved for 2019 (for model evaluation) and 2020 (for data gap filling). The spatial resolution was 0.1°, consistent with the average haul swept area. We evaluated different temporal aggregations of the environmental parameters to train MaxEnt: annual (average over the year), seasonal (average per season), trimester (average per trimester), hot-cold months (separate averages for July-September and October-December), and survey period (November-December average). For each species, we also used MaxEnt to select the parameters with the highest correlation with presence and tested them for optimal modelling.

MaxEnt is widely used in ecological niche modelling (Raybaud et al., 2015; Capezzuto et al., 2018; Angeletti et al., 2020). It is naturally suited for modelling the distribution of a fixed number of events in a delimited space (such as survey hauls) and is equivalent to a Poisson-regression generalized linear model (Renner and Warton, 2013). In the training phase, MaxEnt estimates a function $\pi(\overline{x})$ of environmental parameter vectors $\overline{x}$ constrained to have maxima on species presence locations and minima on simulated absence locations. It is common to consider $\pi(\overline{x})$ a proxy of a probability density of species presence (Phillips and Dudík, 2008; Elith et al., 2011; Merow et al., 2013; Coro et al., 2015b, Coro et al., 2018b). Therefore MaxEnt estimates a functional relation between environmental parameters and the species' presence to generalise the species' distribution (Pearson, 2007). We trained and tested one MaxEnt model for each target species and every environmental parameter temporal aggregation (Section 2.7).

MaxEnt model inherits the spatial resolution of the environmental parameters (0.1°, in our experiment). The optimization algorithm estimates $\pi(\overline{x})$ after maximising the entropy function $H = -\sum \pi(\overline{x}) ln(\pi(\overline{\wedge x}))$ on the training locations (e.g., non-zero biomass surveyed hauls in 2020) with respect to randomly-selected vectors in the study area (*background points*). During the process, it estimates the coefficients of a linear combination of the environmental parameters that represent the importance of each parameter to predict the species' distribution (*percent contribution*). These coefficients can be used to select the parameters carrying the highest quantity of information for the model and re-train/re-test it (Phillips et al., 2017; Coro, 2020a; Coro and Bove, 2022). We used the estimated $\pi(\overline{x})$ function to build up a bonus/malus factor for the biomass estimates produced by the other two components (Section 2.6).

We used and configured a MaxEnt software implementation (Phillips et al., 2017) (linked in the **Supplementary Material**) to reduce over-fitting risk by (i) allowing random background point selection (i.e., pseudo-absence location estimation) to possibly include also surveyed hauls with non-zero biomass (Coro et al., 2022a), and (ii) using *hinge* features to model complex presence-environment relations (Hengl et al., 2009).

2.6 Haul Biomass-Index Estimator

We built the overall *haul biomass-index estimator* (HBIE) model as an open-source R program (linked in the **Supplementary Material**) that combined the three components described in the previous sections. Being $\overline{x}_{h^*}$ the set of environmental feature values in missed haul *h**, HBIE estimates the biomass-index $b_{HBIE}(h^*, \overline{x}_{h^*}, s, y)$ of stock *s* in year *y* and haul *h** as:

$$b_{HBIE}(h^*, \overline{x}_{h^*}, s, y) = W(\overline{x}_{h^*}) \bullet \frac{\alpha \bullet b_{spatial}(h^*, s, y) + \beta \bullet b_{temporal}(h^*, s, y)}{\alpha + \beta}$$

where

$$W(\overline{x}_{h^*}) = \begin{cases} k_{bonus}\ if\ \pi(\overline{x}_{h^*}) > habitat\ suitability\ threshold \\ k_{penalty}\ otherwise \\ 1\ if\ environmental\ information\ is\ unavailable \end{cases}$$

The $W(\overline{x}_{h^*})$ term acts as a bonus multiplier if habitat is suitable in *h**, and as a penalty factor otherwise. A habitat suitability threshold set on top of the $\pi(\overline{x}_{h^*})$ values distinguishes between these two conditions.

In our experiment, we calculated $b_{HBIE}(h^*, \overline{x}_{h^*}, s, 2020)$ for the 4 selected SoleMon stocks in the 10 hauls missed in 2020, but the HBIE model could be applied beyond the SoleMon data. Generally, it is applicable to stocks and survey data with temporal, spatial, and environmental information associated. It would work even if either the spatial or the temporal components were missing. Additionally, if environmental data were missing, the corresponding component factor would be 1.

HBIE introduces new parameters to be estimated in the optimization phase (Section 2.7), i.e., α; β ; k_{bonus}; $k_{penalty}$, and the habitat suitability threshold.

2.7 Model Optimization and Evaluation

2.7.1 Optimisation

The complete list of parameters to optimise is reported in **Table 1**. Of course, the optimal parametrisation depends on the stock. We translated the *precautionary* modelling assumption explained in Section 2.1 into the assumption that the optimal model was the one producing the minimum total biomass-index difference with respect to the last year. Therefore, in our case the optimised parameters were those that ended in the minimum total biomass-index difference between 2019 and 2020.

To select the optimal DIVA parametrisation, we fit DIVA to the biomass-indexes of the 57 surveyed hauls of 2020 by testing several combinations of spatial correlation and signal-to-noise values. We searched for the parameters that minimised the difference between the total biomass-index in 2020 and 2019 after the DIVA estimations. DIVA embeds the *DIVAfit* tool, a statistical tool that produces an initial estimate of the parameters. This tool estimates spatial correlation after fitting the target vector field to the data, under spatial homogeneity hypothesis. It also estimates signal-to-noise ratio based on the anomaly range of this fit (Troupin et al., 2010). Based on the *DIVAfit* indications on our data, we tested spatial correlations between 0.5° and 2° (by 0.5°steps) and signal-to-noise ratios between 0.1 and 10 (by 0.2° steps).

We trained SSA for each of the 10 missed hauls separately to select the optimal temporal component parametrisation. We used historical biomass-index data from 2006 to 2019 (i.e., 14 values) to forecast the 2020 haul biomass-index. We selected the individual-haul parameters minimising the total biomass-index difference between 2020 and 2019. The optimal temporal correlation and number of eigenvectors depended on the haul and the stock. Thus, we optimised 4 *stocks*×10 *hauls*=40 SSA models. For each model, we tested all analysis window lengths between 2 (short-term dependency) and the maximum length of the time series (long-term dependency). We also iteratively incremented and tested the number of eigenvectors to keep for the forecast (Ding et al., 2008).

To select the optimal environmental component parametrisation, we used the non-zero biomass hauls in 2020 as observation locations and tested different environmental parameter sets and temporal aggregations. We tested annual, seasonal, trimester, hot-cold months, and survey period aggregations of the 9 parameters listed in Section 2.4. The non-zero biomass locations used as observation records were 31 for *S. officinalis*, 51 for *S. solea*, 51 for *S. mantis*, and 11 for *P. jacobaeus*. MaxEnt was configured to generate a maximum of 1000 background points as pseudo-absence locations and conduct 500 training iterations. Following the indications to reduce overfitting risk reported in Section 2.5, pseudo-absence locations were randomly taken with the possible inclusion of the surveyed hauls, and *hinge* feature usage was enabled. The projection area was made up of ~ 2900 locations. In the selection process, we first identified the optimal temporal aggregation by tracing the *Receiver Operating Characteristic* (ROC) curve. This curve allowed us to conduct a sensitivity analysis by calculating true-positive and false-positive rates using various decision thresholds on the model output. The ROC curve integral is the *Area Under the Curve* (AUC) and was used as a model-selection criterion (Coro et al., 2015b; Coro et al., 2018b). The higher the AUC, the better the model because a high AUC indicates that the model simulates a probability distribution with significantly higher values on species-presence locations than on random locations. To further test the parameter set, we compared the model using all variables against one using the features carrying 95% of the total percent contribution (Coro et al., 2015b). Eventually, we selected the model with the highest AUC. The habitat suitability threshold used by the HBIE model was the number that resulted in an omission rate (percentage of false absences over estimated absences) below 1% (Coro and Trumpy, 2020; Coro, 2020a; Coro and Bove, 2022).

After optimising the individual components, we optimised the HBIE model by testing all parameter combinations within the following prior ranges: [0.1;2] (by 0.1 steps) for α and β; [0;2] (by 0.1 steps) for kbonus and kpenalty. Eventually, we selected the set resulting in the minimum total biomass-index difference between 2020 and 2019.

2.7.2 Evaluation

In order to evaluate the HBIE model, we used 2019 as the analysis year and hypothesised that the missed hauls were the same 10 hauls missed in 2020. We used the time series of 2006-2018 data of these hauls (i.e., 13 values for each haul) to train the temporal component and forecast the 2019 values. We used 57 biomass-index values in 2019 (i.e., those from the same surveyed hauls of 2020) to train the spatial component and project its estimates in the missed hauls. The same 57 locations were used as observation records (when biomass-index was non-zero) to train the environmental component with the 9 selected environmental parameters and, iteratively, on 5 temporal aggregations (from annual to November-December period). We used 2019 values for all environmental and oceanic parameters involved. Non-zero biomass observation records were 32 for *S. officinalis*, 52 for *S. solea*, 32 for *S. mantis*, and 17

TABLE 1 | Complete set of parameters used by our models and optimised in the training phase.

Model	Parameters to estimate
Spatial component (DIVA)	Spatial correlation between the observations; Signal-to-noise ratio
Temporal component (SSA)	Window analysis length (*M*); Number of eigenvectors to keep
Environmental component (MaxEnt)	Most influential environmental parameters for species habitat; Temporal aggregation
Overall model (HBIE)	α; β ; k_{bonus}; $k_{penalty}$; habitat suitability threshold

for *P. jacobaeus*. MaxEnt was configured to generate a maximum of 1000 background points as pseudo-absence locations and 500 training iterations.

We used the measured 2019 biomass-indexes in the missed hauls to calculate model accuracy, i.e., the percentage of correctly predicted indexes within statistical confidence limits. We also estimated the correct prediction of the 2019 total biomass-index.

As a baseline comparison index, we adopted an *equiproportional* index that assumed, for each missed haul, that the average ratio between the total biomass-index of the surveyed hauls and the missed hauls' index remained constant over the years. Therefore, after calculating the average ratio for each unsurveyed haul, this index easily allowed estimating the unsurveyed hauls' values. The equiproportional index calculation algorithm is summarised as follows:

Algorithm 3 Equiproportional index calculation algorithm

```
for each missed haul
for each year before the analysis year
estimate the ratio between the total biomass-index in the surveyed hauls and the biomass-index in the missed haul
average the ratios over the years
use the ratio to estimate the biomass-index in the missed haul, in the analysis year, given the total biomass-index of the surveyed hauls
estimate the total biomass-index using the surveyed values and the estimates for the missed hauls
```

We also analysed the relation between our HBIE model uncertainty and the hauls' contributions to the total biomass-index. Haul contribution was estimated as the average relative variation of the total biomass-index over the years when the haul (and its associated strata) was removed from the calculation. Evaluating the relation between haul contribution and HBIE model precision shed light on accuracy calculation reliability and stock biomass distribution homogeneity. It is worth noting that HBIE uncertainty comes from the DIVA model after propagating the confidence limits into the HBIE formula. In fact, the canonical SSA algorithm does not produce statistical uncertainty for its estimates (Allen and Smith, 1997) and MaxEnt was used as a thresholded factor.

3 RESULTS

3.1 Optimal Parameters

The optimal model parameters for the 2020 SoleMon data are reported in **Table 2**. The DIVA spatial correlation reflects the average spatial geographical distance from an abundant location to the other, with less mobile species (e.g., *P. jacobaeus*) having lower spatial correlation values. The signal-to-noise ratio was averagely low for all species, but was sensibly higher for *S. solea*. The average SSA temporal dependencies indicate that long-term dependency modelling (from 7 to 9 years) was necessary for good forecasts. MaxEnt gave optimal results when all parameters were used because they all brought essential information to properly model species presence. The optimal temporal aggregation was hot and cold months (i.e., separate averages over July-September and October-December). Cold months indeed included the environmental conditions of the survey period, and hot months included summer conditions that might have influenced species distribution in winter (Henderson et al., 2017). The MaxEnt habitat suitability threshold depended on the species. Interestingly, these values almost corresponded to the lower confidence limit of a log-normal distribution traced over all MaxEnt values on low-biomass locations. In this case, low-biomass locations were those with a biomass-index falling at the lower log-normal tail of the overall biomass-index distribution.

The HBIE optimal parameter values indicate that no component outperformed the other. Therefore the weighted average in the HBIE formula was a standard average. This condition was likely related to the specific SoleMon data, with few temporal gaps and a peculiarly invariant haul distribution over the years. We anticipate that conditions such as worse temporal sampling, less homogeneous spatial sampling, and under-representative data would result in different component weights. The environmental suitability bonus was 1 for all species, which indicates that the models directly reported the average biomass-index estimate for suitable habitat locations in the analysis year. Instead, all models applied a 0.4 penalty (i.e., a 60% reduction) on unsuitable habitat locations. Therefore, the environmental component only intervened in unsuitable habitat hauls to soften the biomass-index estimate.

TABLE 2 | Optimal model parameters estimated for the analysed stocks based on the SoleMon data.

	Sepia officinalis	*Solea solea*	*Squilla mantis*	*Pecten jacobaeus*
DIVA spatial correlation between observations	1°	0.2°	1°	0.15°
DIVA signal-to-noise ratio	2.4	6.5	4.5	2
SSA window analysis length (samples) - avg across hauls	9	7	7	9
SSA n. of eigenvectors to keep - avg across hauls	3	2	6	6
MaxEnt environmental parameters	All	All	All	All
MaxEnt parameter temporal aggregation	hot-cold	hot-cold	hot-cold	hot-cold
MaxEnt habitat suitability threshold	0.021	0.022	0.036	0.037
HBIE α	1	1	1	1
HBIE β	1	1	1	1
HBIE k_{bonus}	1	1	1	1
HBIE $k_{penalty}$	0.4	0.4	0.4	0.4

FIGURE 4 | Two cases demonstrating substantial differences between the biomass-index estimates of our combined model (HBIE) and its temporal and spatial components with respect to a baseline estimate (equiproportional index). The two cases show a quasi-periodic and a non-periodical time series, respectively. The rightmost charts report forecasts of 2019 values and comparison with known data. The middle charts report forecasts of 2020 values. The dashed lines highlight the correspondence between the measurements (real data) and the same points in the forecast charts. The colours of the numbers and lines in the forecast charts correspond to the legend indications.

Two examples on *S. officinalis* missed hauls show the difference between the HBIE model and its components (**Figure 4**). The first example reports a haul's historical biomass-index with tri-annual periodicity between 2011 and 2017. The equiproportional index coarsely identified a decreasing trend in the last years and thus estimated a slightly lower value for 2020 (32.9 kg/km^2) than the 2019 value (33.48 kg/km^2). Our spatial component also estimated a slightly lower value for 2020 (32.62 kg/km^2) than the 2019 value. The temporal component better captured the decreasing trend in 2020 and reported a 23% lower value (25.11 kg/km^2) than the other indexes. The environmental component classified the habitat as unsuitable for the species in the haul in 2020, and thus further decreased the estimated biomass-index to 11.55 kg/km^2. This penalty resulted in better capturing the low biomass that experts expected in the haul due to a delayed species absence periodicity and unsuitable habitat. It is worth noting that habitat was instead suitable in 2019, with a relatively high biomass-index (33.48 kg/km^2), and all HBIE components achieved a good prediction of this value (between 33.11 and 36.2 kg/km^2). Instead, the equipropotional index overestimated the 2019 value as 44.9 kg/km^2.

The second example shows a particular non-periodical biomass-index time series associated with a missed haul. The equiproportional index estimate for 2020 (24.7 kg/km^2) was higher than the 2019 value (16.13 kg/km^2) because it captured an averagely increasing trend since 2012. The spatial model reported a similar estimate for the same year (24.95 kg/km^2). Instead, the unpredictability of the time series of the last years made the temporal component estimate complete stock absence in the haul for 2020. Since habitat was estimated as suitable in the haul area in 2020, HBIE directly returned half of the spatial model estimate as the final result without further penalties (12.47 kg/km^2). This estimate compensated for the potential bias of the temporal component. It is worth noting that this value is consistent with the time series values because it is close to the last 10-year average (14.3 kg/km^2), if the 2018 value (55.63 kg/km^2) were considered an anomaly. The evaluation of the 2019 value prediction shows that all HBIE components returned very close values (from 15.8 to 16.04 kg/km^2) to the real value (16.13 kg/km^2), whereas the equiproportional index sensibly overestimated it (34.34 kg/km^2). The temporal component prediction was particularly close to the real value, which demonstrates the SSA effectiveness with non-stationary time series, but - considering the 2020 estimate - also its sensitivity on the number of samples and abrupt variations. All the time series comparisons for the missed hauls are reported in the **Supplementary Material**.

3.2 Performance

We trained a model using 2019 data, while excluding the same hauls missing in 2020. We used the 2006-2018 data for time series analyses and took 2018 data as a reference for model training. Since temporal and spatial sampling data were constant in the surveyed area over the years, the estimated optimal HBIE parametrisation - apart from the habitat suitability thresholds - was equal to the one for 2020 data (**Table 2**). The MaxEnt environmental parameters were all confirmed to carry important information for optimal modelling. The hot-cold-months aggregation was confirmed to be optimal, and thus was not specific to the 2020 data. The average SSA and the DIVA parameters were not sensibly different from the 2020 model's ones, thus they only depended on the spatiotemporal structure of the data.

Average accuracy on haul-biomass recognition ranged from 80% to 100% (**Table 3**), which was higher than the 30%-80% range of the equiproportional index. The lowest accuracy was obtained for *S. mantis*, and was probably due to the very low biomass in the missed hauls, going down to complete absence in some cases. The total biomass-index fell within the confidence ranges for all stocks, whereas the equiproportional index correctly estimated the total biomass-index of *P. jacobaeus* only. The comparison table also reports the estimated 2020 total biomass-indexes, which are meant to feed RFMOs' stock assessment models (Froese et al., 2020).The overall biomass-index distributions are displayed in **Figure 5**.

3.3 Model Uncertainty and Haul Contribution to Total Biomass-Index

Highly contributing hauls to the total biomass-index were present throughout the entire area (**Figure 6**). However, no stock presented an isotropic and homogeneous distribution of highly contributing hauls. One small homogeneous area of lowly contributing hauls can only be observed for *S. mantis* in the deep area halfway between the Italian and Croatian coasts.

It is worth noting that the unsurveyed 2020 hauls were not randomly distributed, but mostly concentrated off the Croatian coasts with generally high contributions to the total biomass index. Therefore, it was crucial to estimate these values correctly because they sensibly influenced the total biomass-index estimates.

Due to inhomogeneous distribution, low-contribution locations could reside very close to high-contribution locations because low-biomass hauls could surround large-biomass hauls. Therefore, high-contribution hauls were peaks of the contribution distribution close to minima. This scenario increased the estimation uncertainty on high-contribution hauls. This observation is confirmed by a direct linear relation between the 2020 HBIE model uncertainty and the haul contribution to the total biomass-index (**Figure 7**). The correlation strengths range between moderate (0.36 for *S. solea*, 0.44 for *P. jacobaeus*, and 0.46 for *S. officinalis*) and high (0.95 for *S. mantis*). The higher the contribution, the higher the uncertainty. Understanding this relationship is important when re-using our model for other stocks and areas. Generally, this relation complies with the expected properties of a biomass estimation model. It is reasonable that such a model predicts missing data with higher precision over a small area with homogeneous biomass, and with lower precision over a wide area with jeopardised large-biomass distribution.

4 DISCUSSION AND CONCLUSIONS

We have presented a model to estimate stock biomass density in occasionally unsurveyed areas, with an application to the 2020 SoleMon survey data in North-Central Adriatic Sea. The model combines three complementary components: spatial, temporal, and environmental. When applied to the 2019 SoleMon data, our model was able to estimate the total biomass-index of all analysed stocks correctly. The accuracy over individual haul biomass-index estimation was also high (80-100%). We observed that model uncertainty was higher for larger biomass-index hauls, probably because of the jeopardised biomass distribution of the analysed stocks. Moreover, the model achieved a higher estimation accuracy than an alternative, widely used index that assumed the conservation of average surveyed/unsurveyed biomass proportion over time. The advantage of this alternative index is its fast implementation, but our results showed that it is more suited for coarse approximations. Our model implementation

TABLE 3 | Performance of our model with respect to measured 2019 biomass-indexes across the four analysed stocks.

	Accuracy of the 2019 model at predicting the 2019 hauls	Accuracy of the 2019 equiproportional index at predicting the 2019 hauls	Total biomass-index in 2019	Total model-predicted biomass-index in 2019	Total equiproportional-index predicted biomass in 2019	Model-predicted biomass-index in 2020
Sepia officinalis	90%	30%	37.03	37.27 [35.16; 39.39]	46.29 [45.78; 46.81]	**27.98 [26.53; 29.44]**
Solea solea	90%	60%	72.71	68.10 [56.39; 82.32]	66.30 [66.07; 66.53]	**71.09 [57.86; 85.84]**
Squilla mantis	80%	50%	25.75	25.72 [25.29; 28.60]	26.20 [26.08; 26.32]	**26.76 [26.56; 28.06]**
Pecten jacobaeus	100%	80%	7.47	9.11 [5.14; 15.61]	8.82 [6.82; 10.83]	**4.12 [3.30; 5.04]**
Total Accuracy				4/4	1/4	

A comparison with a conservative surveyed/unsurveyed haul proportion index (equiproportional) is reported. The last column (bold-highlighted) reports final 2020 estimates to be used in stock assessment models. Square brackets indicate lower and upper confidence values.

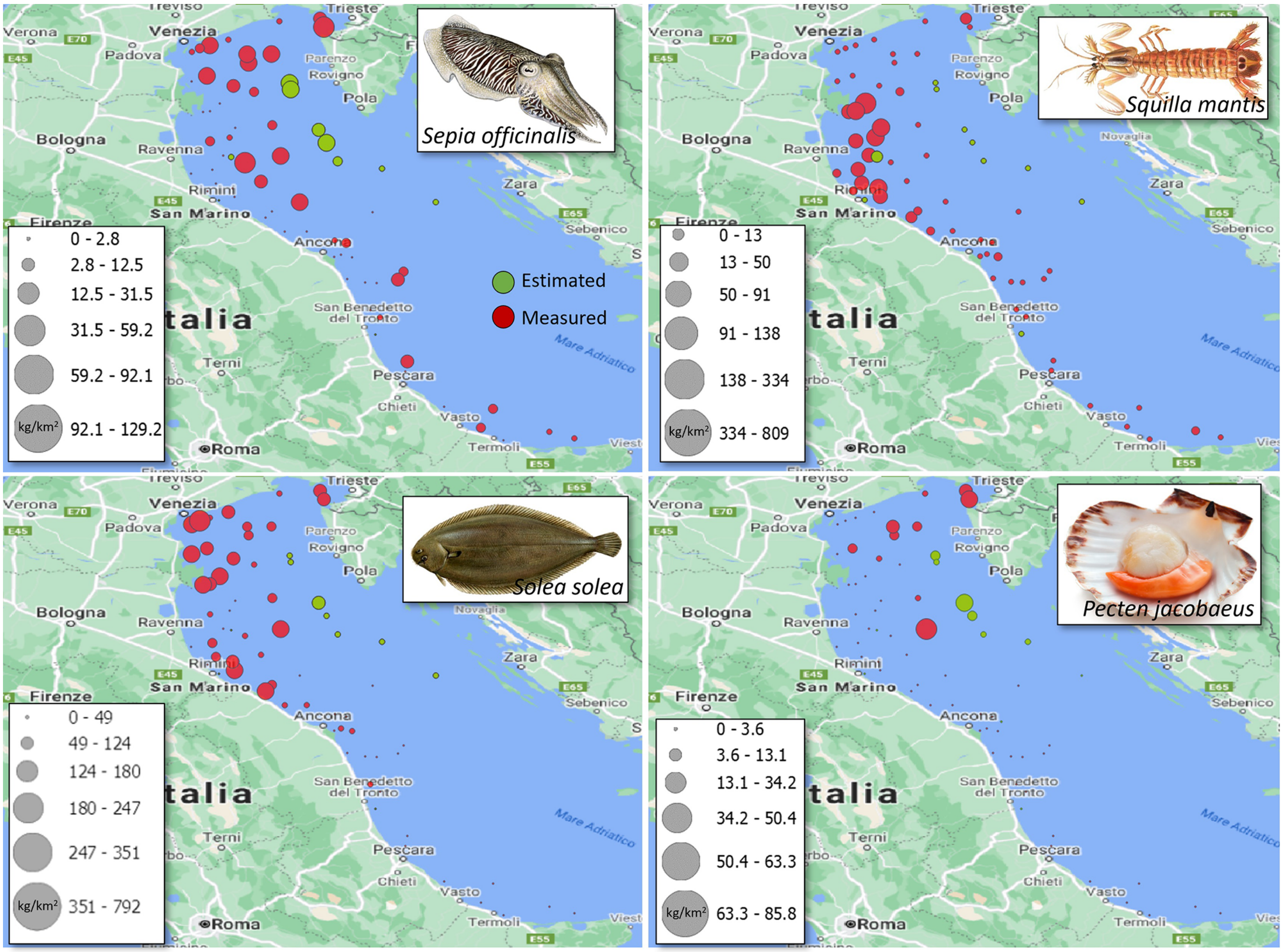

FIGURE 5 | Distribution of measured (red) and estimated (green) SoleMon biomass-indexes per haul for 2020.

is fully based on open-source software, and every sub-model is available either as desktop software or notebook (**Supplementary Material**). After data preparation, running the sub-models for one species on a modern desktop PC or laptop - e.g., endowed with an Intel i9 CPU with 8 GB of Random Access Memory - requires about 1 hour. Moreover, all sub-models can be used through free-to-use Web interfaces based on cloud computing systems that simplify model configuration and speed up data processing. One limitation of our current implementation is that the three sub-models are not integrated into an all-in-one offline process because DIVA is currently released as a notebook that can be hardly transformed into an automatic process. Our next-future plan is to transform DIVA into a Web service to facilitate its automatic integration with the other sub-models, which will require preparing specific cloud services and infrastructures (Assante et al., 2020).

One similarity between our model and VAST is that they both include spatial and temporal models, although they are modelled and combined differently. VAST uses two functions to estimate stock biomass density in the unsurveyed hauls for a specific survey year: one is the probability $p(s_i,t_j)$ of encountering the species in unsurveyed haul s_i in year t_j, and the other is the expected catch rate $r(s_i,t_j)$. The expected stock biomass density $d(s_i,t_j)$ in s_i is calculated as the product of these two terms, i.e., $d(s_i,t_j)=p(s_i,t_j)\cdot r(s_i,t_j)$. VAST models $p(s_i,t_j)$ as a logit distribution approximated by a linear combination of unknown random variables defined on s_i and t_j. Moreover, it models $r(s_i,t_j)$ as the mean of a log-normal distribution approximated by another linear combination of random variables. The probability (p) of encountering the species in the unsurveyed hauls in the analysis year coarsely corresponds to our environmental and spatial components, although VAST does not explicitly use environmental variables. The VAST catch rate term (r) is a time-dependent model that, differently from our temporal component, does not estimate a biomass index directly. Moreover, being d the product of the two r and p terms, the two models should be very accurate because multiplication is highly sensitive to individual function biases. Conversely, in our model, one of the biomass-index estimators could even be missing. Finally, VAST finds the optimal distributions using the Akaike Information Criterion as a model quality measurement, which introduces the potential bias to always select models with a higher number of parameters among equal-likelihood models (Guthery et al., 2005; Arnold, 2010; Coro et al., 2022a). Conversely, our model

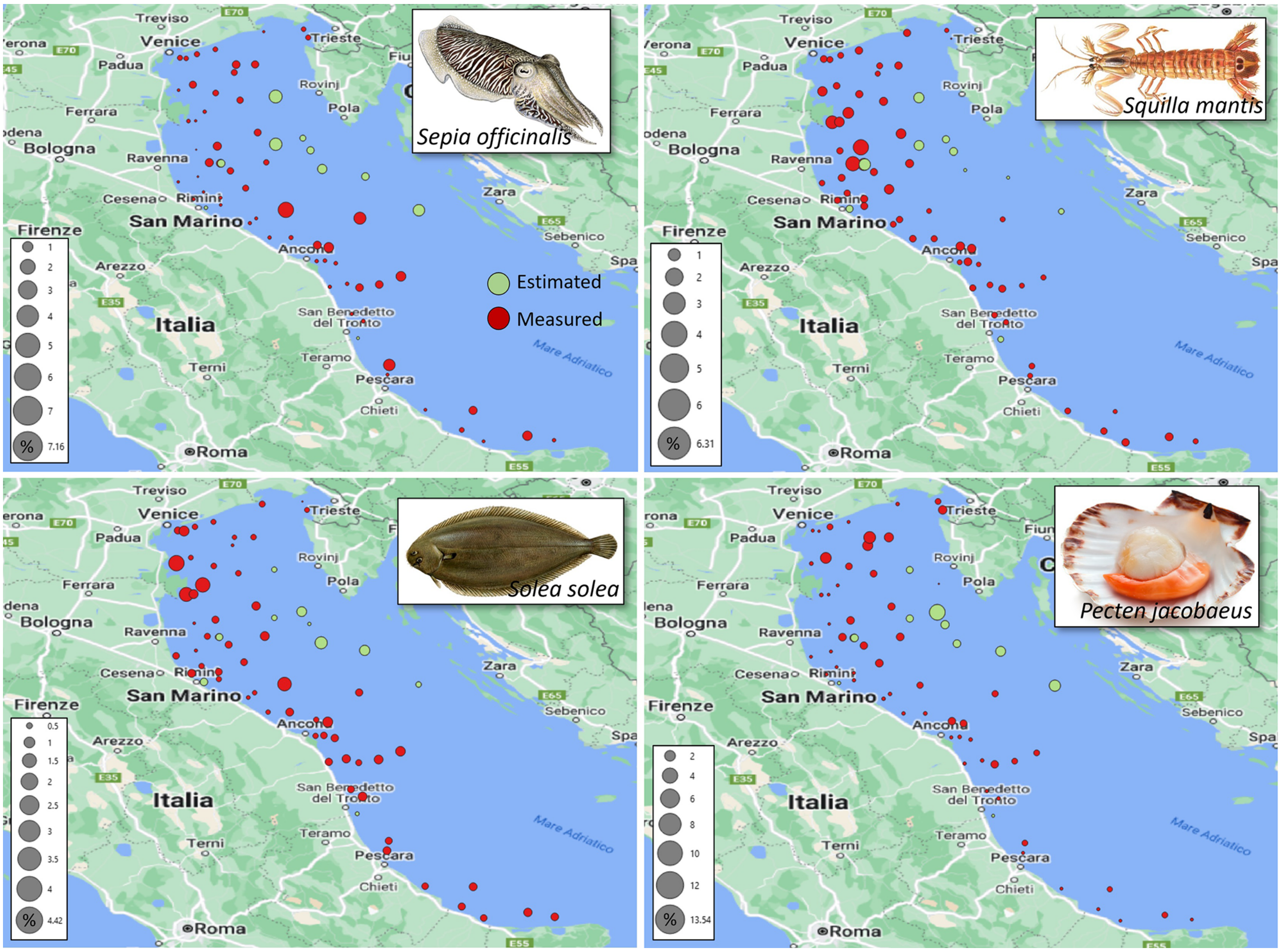

FIGURE 6 | Percent average contributions, over the survey years, of the SoleMon hauls to the total biomass-index. Colours highlight the 2020 measured (red) and estimated hauls (green).

trains the components independently of each other using the last known biomass index as a reference. Moreover, each component models a more complex function than a linear combination of random variables.Our model shares characteristics with general spatiotemporal data gap filling models for remote sensing imagery reconstruction, which separately fill spatial and temporal gaps and eventually combine the estimates (Weiss et al., 2014; Metz et al., 2017; Yan and Roy, 2018). With respect to these models, our model uses an ocean-specific kriging model for spatial modelling. Moreover, it uses a general signal processing technique for temporal modelling that is more complex than the pixel-wise temporal smoothing functions used by most alternative models. One interesting comparison is with deep-learning-based models that directly simulate a space-time data reconstruction function and can reach very high performance in specific contexts (Belda et al., 2020; Varshney et al., 2021; Goodman, 2021). Differently from our model, deep-learning models can difficultly be re-implemented and adapted to new contexts - that usually require new model topologies and specific large training sets - and optimisation is very time-consuming. Moreover, performance and bias interpretability are easier for our type of model components than for deep learning models (Chakraborty et al., 2017; Zhang and Zhu, 2018).

We conjecture that our model is general enough to be applied also to other fishery trawl survey data. However, we acknowledge that the performance on SoleMon data were facilitated by favourable conditions such as a low inter-annual spatio-temporal variability of the haul distribution. Unfortunately, such conditions are uncommon and unlikely in more extended and multi-country survey programmes. For example, the Mediterranean MEDITS programme (Spedicato et al., 2019) is a 30-year data collection action that has been subject to changes due to revisions, optimisation, and re-planning. These changes corresponded to data gaps and inhomogeneity in time and space. Our model can manage this scenario by giving the highest weight to the component using the most informative data. Generally, in our future applications we will test our model on survey data that include issues such as (i) haul distribution change across the years, (ii) survey season change, and (iii) haul historical data containing several gaps. A potential limitation of our model when applied to other trawl surveys is that it cannot predict stock abundance directly, which will require integrating more data with the model.

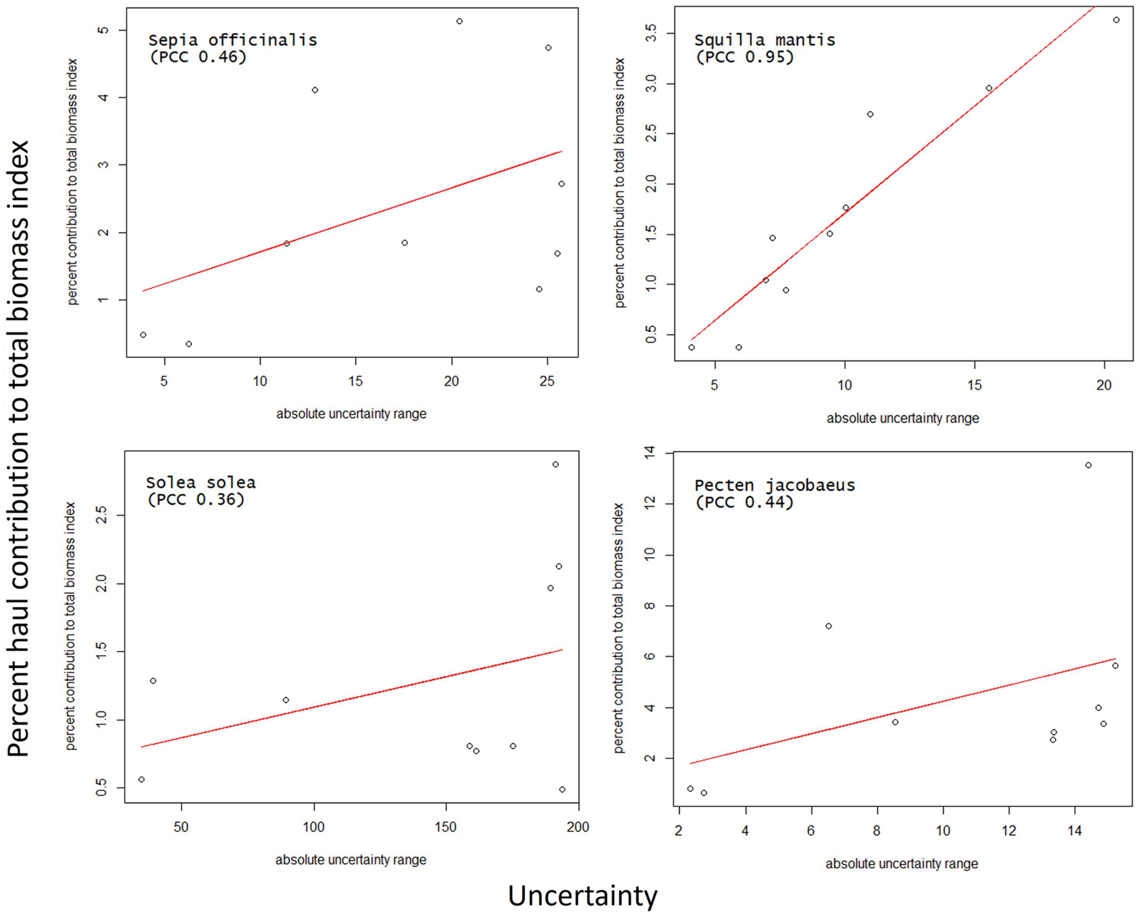

FIGURE 7 | Linear fit between the HBIE 2020 model uncertainty and the percent haul contribution to the total biomass, with the indication of the Pearson Correlation Coefficient (PCC).

We believe that our model can improve the quality of the information used by the GFCM, STECF, and MSFD, and improve stock status evaluation. Indeed, the biomass indexes reported in **Table 3** have already been proposed and used for the 2022 GFCM stock assessments, after experts' consistency evaluation of the model (Scientific Advisory Committee on Fisheries, 2022).

4.1 Model Applications

The major applications of our model can be summarised as follow:

- *Data enhancement*: The estimated biomass indexes can independently enrich the data coming from fishery survey, especially when major issues prevented complete monitoring. They also help monitor the correlation between biomass distribution and environmental conditions;
- *Re-application to other scientific survey data*: other scientific survey programmes can reuse our models to reconstruct biomass-indexes and compare the results to their current estimates;
- *Haul contribution analysis*: In critically limiting survey conditions, surveys could be prioritised to visit the hauls with the highest contribution to the total biomass-index calculation;
- *Supporting stock assessment and harvest control rules*: Stock status assessment is the basis for setting management rules, i.e., the amount of days fishing vessels can spend at sea and the harvest control rules that limit the catches. Indexes of relative abundance – such as the survey biomass index - are primary input data for stock assessment (Maunder and Punt, 2004). Using robust model-based input data is encouraged when raw observations are not sufficiently reliable (Thorson and Haltuch, 2019). Having access to complete time series with spatial gaps reliably filled would help experts parametrise stock assessment models and increase result reliability and precision;
- *Understanding the interplay between environmental change and fisheries*: Environmental change may affect stock distribution and productivity (Free et al., 2019). The stock-specific intrinsic rate of increase and carrying capacity depend on the interaction between the species and the environment where it lives (Froese et al., 2017). Understanding the interplay between environmental conditions and stock dynamics is crucial for integrated environmental assessment and

ecosystem approaches to fishery management (Antunes and Santos, 1999; Rosenberg et al., 2000; Karp et al., 2019; Marshall et al., 2019; Coro et al., 2021). Our model can contribute to this context because it can model species' habitat suitability change over the years and attach this information to the survey data.

DATA AVAILABILITY STATEMENT

All datasets and software presented in this study can be found in a GitHub free access online repository, which corresponds to the Supplementary Material of the manuscript: https://github.com/cybprojects65/SoleMonGeospatialModelling.

AUTHOR CONTRIBUTIONS

GC conceived, designed and implemented the model, and conducted the experiments; PB contributed to model design and conducted the experiments; EA and FM provided and prepared biomass and environmental data, and contributed to modelling context definition, environmental parameter selection and result interpretation; MS provided data on Squilla mantis; GS supervised and organised the SoleMon survey campaigns, validated the data and provided information about the surveys. All authors contributed to paper writing and revision.

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REFERENCES

Aeberhard, W. H., Mills Flemming, J. and Nielsen, A. (2018). Review of State-Space Models for Fisheries Science. *Annu. Rev. Stat Its Appl.* 5, 215–235. doi: 10.1146/annurev-statistics-031017-100427

Allen, M. R. and Smith, L. A. (1997). Optimal Filtering in Singular Spectrum Analysis. *Phys. Lett. A* 234, 419–428. doi: 10.1016/S0375-9601(97)00559-8

Angeletti, L., Prampolini, M., Foglini, F., Grande, V. and Taviani, M. (2020). "Cold-Water Coral Habitat in the Bari Canyon System, Southern Adriatic Sea (Mediterranean Sea)," in *Seafloor Geomorphology as Benthic Habitat* (Amsterdam, The Netherlands: Elsevier), 811–824. doi:

Antunes, P. and Santos, R. (1999). Integrated Environmental Management of the Oceans. *Ecol. Economics* 31, 215–226. doi: 10.1016/S0921-8009(99)00080-4

Arnold, T. W. (2010). Uninformative Parameters and Model Selection Using Akaike's Information Criterion. *J. Wildlife Manage.* 74, 1175–1178. doi: 10.1111/j.1937-2817.2010.tb01236.x

Assante, M., Candela, L., Castelli, D., Cirillo, R., Coro, G., Dell'Amico, A., et al. (2020). Virtual Research Environments Co-Creation: The D4science Experience. *Concurrency Computation: Pract. Exp.*, 35: e6925. doi: 10.1002/cpe.6925

Assante, M., Candela, L., Castelli, D., Cirillo, R., Coro, G., Frosini, L., et al. (2019). Enacting Open Science by D4science. *Future Generation Comput. Syst.* 101, 555–563. doi: 10.1016/j.future.2019.05.063

Barth, A., Alvera-Azcárate, A., Troupin, C., Ouberdous, M. and Beckers, J.-M. (2010). A Web Interface for Griding Arbitrarily Distributed in Situ Data Based on Data-Interpolating Variational Analysis (Diva). *Ad. Geo.* 28, 29-37 doi: 10.5194/adgeo-28-29-2010

Belda, S., Pipia, L., Morcillo-Pallarés, P., Rivera-Caicedo, J. P., Amin, E., De Grave, C., et al. (2020). DATimeS: A Machine Learning Time Series GUI Toolbox for Gap-Filling and Vegetation Phenology Trends Detection. *Environ. Model. Software* 127, 104666. doi: 10.1016/j.envsoft.2020.104666

Bennett, A. F. (1992). *Inverse Methods in Physical Oceanography* (Cambridge University Press).

Bergenius, M., Eigaard, O. R., Orio, A., Søvik, G. and Ulmestrand, M. (2021). Joint NAFO/ICES Pandalus Assessment Working Group (NIPAG) Report. Tech. rep., International Council for the Exploration of the Sea (ICES).

Blue Cloud (2022). *The Blue Cloud Lab D4Science Virtual Research Environment*.

Breivik, O. N., Aanes, F., Søvik, G., Aglen, A., Mehl, S. and Johnsen, E. (2021). Predicting Abundance Indices in Areas Without Coverage With a Latent Spatio-Temporal Gaussian Model. *ICES J. Mar. Sci.* 78, 2031–2042. doi: 10.1093/icesjms/fsab073

Candela, L., Castelli, D., Coro, G., Pagano, P. and Sinibaldi, F. (2016). Species Distribution Modeling in the Cloud. *Concurrency Comput. Pract. Exp.* 28, 1056–1079. doi: 10.1002/cpe.3030

Capezzuto, F., Sion, L., Ancona, F., Carlucci, R., Carluccio, A., Cornacchia, L., et al. (2018). Cold-Water Coral Habitats and Canyons as Essential Fish Habitats in the Southern Adriatic and Northern Ionian Sea (Central Mediterranean). *Ecol. Questions* 29, 9–23. doi: 10.12775/EQ.2018.019

Chakraborty, S., Tomsett, R., Raghavendra, R., Harborne, D., Alzantot, M., Cerutti, F., et al. (2017). "Interpretability of Deep Learning Models: A Survey of Results," in *2017 IEEE Smartworld, Ubiquitous Intelligence & Computing, Advanced & Trusted Computed, Scalable Computing & Communications, Cloud & Big Data Computing, Internet of People and Smart City Innovation (Smartworld/SCALCOM/UIC/ATC/CBDcom/IOP/SCI)* (IEEE), 1–6.

Cibic, T., Blasutto, O., Falconi, C. and Umani, S. F. (2007). Microphytobenthic Biomass, Species Composition and Nutrient Availability in Sublittoral Sediments of the Gulf of Trieste (Northern Adriatic Sea). *Estuarine Coast. Shelf Sci.* 75, 50–62. doi: 10.1016/j.ecss.2007.01.020

Clementi, E., Aydogdu, A., Goglio, A., Pistoia, J., Escudier, R., Drudi, M., et al. (2021). *Mediterranean Sea Physical Analysis and Forecast (Cmems Med-Currents, Eas6 System) (Version 1)* (Copernicus Monitoring Environment Marine Service (Brussels, Belgium: CMEMS). doi: 10.25423/CMCC/MEDSEA

Cochran, W. G. (1977). *Sampling Techniques* (New York: Wiley).

Colloca, F., Garofalo, G., Bitetto, I., Facchini, M. T., Grati, F., Martiradonna, A., et al. (2015). The Seascape of Demersal Fish Nursery Areas in the North Mediterranean Sea, a First Step Towards the Implementation of Spatial Planning for Trawl Fisheries. *PLos One* 10, e0119590. doi: 10.1371/journal.pone.0119590

Coro, G. (2013). A Lightweight Guide on Gibbs Sampling and JAGS. *Tech. rep. Istituto di Scienza e Tecnologie dell'Informazione A. Faedo - CNR*. 1-16

Coro, G. (2020a). A Global-Scale Ecological Niche Model to Predict Sars-Cov-2 Coronavirus Infection Rate. *Ecol. Model.* 69*:* 431, 109187. doi: 10.1016/j.ecolmodel.2020.109187

Coro, G. (2020b). Open Science and Artificial Intelligence Supporting Blue Growth. *Environ. Eng. Manage. J. (EEMJ)* 19, 1719-1729. doi: 10.30638/eemj.2020.162

Coro, G. and Bove, P. (2022). A High-Resolution Global-Scale Model for Covid-19 Infection Rate. ACM Transactions on Spatial Algorithms and Systems (TSAS) 8, 1–24. doi: 10.1145/3494531

Coro, G., Bove, P. and Ellenbroek, A. (2022a). Habitat Distribution Change of Commercial Species in the Adriatic Sea During the Covid-19 Pandemic. *Ecol. Inf.*, 101675. doi: 10.1016/j.ecoinf.2022.101675

Coro, G., Candela, L., Pagano, P., Italiano, A. and Liccardo, L. (2015a). Parallelizing the Execution of Native Data Mining Algorithms for Computational Biology. *Concurrency Computation: Pract. Exp* 27, 4630–4644. doi: 10.1002/cpe.3435

Coro, G., Ellenbroek, A. and Pagano, P. (2021). An Open Science Approach to Infer Fishing Activity Pressure on Stocks and Biodiversity From Vessel Tracking Data. *Ecol. Inf.* 64, 101384. doi: 10.1016/j.ecoinf.2021.101384

Coro, G., Large, S., Magliozzi, C. and Pagano, P. (2016a). Analysing and Forecasting Fisheries Time Series: Purse Seine in Indian Ocean as a Case Study. *ICES J. Mar. Sci.* 73, 2552–2571. doi: 10.1093/icesjms/fsw131

Coro, G., Magliozzi, C., Ellenbroek, A. and Pagano, P. (2015b). Improving Data Quality to Build a Robust Distribution Model for Architeuthis Dux. *Ecol. Model.* 305, 29–39. doi: 10.1016/j.ecolmodel.2015.03.011

Coro, G., Pagano, P. and Ellenbroek, A. (2018a). Detecting Patterns of Climate Change in Long-Term Forecasts of Marine Environmental Parameters. *Int. J. Digital Earth*, 13: 1–19. doi: 10.1080/17538947.2018.1543365

Coro, G., Pagano, P. and Napolitano, U. (2016b). Bridging Environmental Data Providers and Seadatanet Diva Service Within a Collaborative and Distributed E-Infrastructure. *Bollettino di Geofisica Teorica ed Applicata*, 57: 23–25.

Coro, G., Panichi, G., Scarponi, P. and Pagano, P. (2017). Cloud Computing in a Distributed E-Infrastructure Using the Web Processing Service Standard. *Concurrency Computation: Pract. Exp.* 29, e4219. doi; 10.1002/cpe.4219

Coro, G., Tassetti, A. N., Armelloni, E. N., Pulcinella, J., Ferrà, C., Sprovieri, M., et al. (2022b). Covid-19 Lockdowns Reveal the Resilience of Adriatic Sea Fisheries to Forced Fishing Effort Reduction. *Sci. Rep.* 12, 1–14. doi: 10.1038/s41598-022-05142-w

Coro, G. and Trumpy, E. (2020). Predicting Geographical Suitability of Geothermal Power Plants. *J. Cleaner Production* 267, 121874. doi: 10.1016/j.jclepro.2020.121874

Coro, G., Vilas, L. G., Magliozzi, C., Ellenbroek, A., Scarponi, P. and Pagano, P. (2018b). Forecasting the Ongoing Invasion of Lagocephalus Sceleratus in the Mediterranean Sea. *Ecol. Model.* 371, 37–49. doi: 10.1016/j.ecolmodel.2018.01.007

Ding, H., Trajcevski, G., Scheuermann, P., Wang, X. and Keogh, E. (2008). "Querying and Mining of Time Series Data: Experimental Comparison of Representations and Distance Measures," in Proceedings of the VLDB Endowment, *Proceedings of the VLDB Endowment,* Auckland, New Zealand Vol. 1. 1542–1552.

Eisner, L. B., Yasumiishi, E. M., Andrews, III, A. G. and O'Leary, C. A. (2020). Large Copepods as Leading Indicators of Walleye Pollock Recruitment in the Southeastern Bering Sea: Sample-Based and Spatio-Temporal Model (Vast) Results. *Fisheries Res.* 232, 105720. doi: 10.1016/j.fishres.2020.105720

Elith, J., Phillips, S. J., Hastie, T., Dudík, M., Chee, Y. E. and Yates, C. J. (2011). A Statistical Explanation of Maxent for Ecologists. *Diversity distributions* 17, 43–57. doi: 10.1111/j.1472-4642.2010.00725.x

Elsner, J. B. and Tsonis, A. A. (2013). *Singular Spectrum Analysis: A New Tool in Time Series Analysis* (Berlin, Germany: Springer Science & Business Media).

EUR-Lex (2021). *Commission Delegated Decision (Eu).*

FAO (1999). "Report of the First Session of the Scientific Advisory Committee," in *Report N. 601* (Mediterranean (Rome, Italy :FAO), 1–185.

Feudale, L., Bolzon, G., Lazzari, P., Salon, S., Teruzzi, A., Di Biagio, V., et al. (2021). *Mediterranean Sea Biogeochemical Analysis and Forecast (Cmems Med-Biogeochemistry, Medbfm3 System)(Version 1)* (Copernicus Monitoring Environment Marine Service (Brussels, Belgium: CMEMS). doi: 10.25423/cmcc/medsea_analysis_forecast_bio_006_014

Fournier, D. A., Skaug, H. J., Ancheta, J., Ianelli, J., Magnusson, A., Maunder, M. N., et al. (2012). Ad Model Builder: Using Automatic Differentiation for Statistical Inference of Highly Parameterized Complex Nonlinear Models. *Optimiszation Methods Software* 27, 233–249. doi: 10.1080/10556788.2011.597854

Free, C. M., Thorson, J. T., Pinsky, M. L., Oken, K. L., Wiedenmann, J. and Jensen, O. P. (2019). Impacts of Historical Warming on Marine Fisheries Production. *Science* 363, 979–983. doi: 10.1126/science.aau1758

Froese, R., Demirel, N., Coro, G., Kleisner, K. M. and Winker, H. (2017). Estimating Fisheries Reference Points From Catch and Resilience. *Fish Fisheries* 18, 506–526. doi: 10.1111/faf.12190

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2020). Estimating Stock Status From Relative Abundance and Resilience. *ICES J. Mar. Sci.* 77, 527–538. doi: 10.1093/icesjms/fsz230

Frost, H. and Andersen, P. (2006). The Common Fisheries Policy of the European Union and Fisheries Economics. *Mar. Policy* 30, 737–746. doi: 10.1016/j.marpol.2006.01.001

GEBCO (2020). *Gridded Bathymetry Data.*

Golyandina, N. and Osipov, E. (2007). The "Caterpillar"-Ssa Method for Analysis of Time Series With Missing Values. *J. Stat. Plann. Inference* 137, 2642–2653. doi: 10.1016/j.jspi.2006.05.014

Goodman, S. M. (2021). *Filling in the Gaps: Applications of Deep Learning, Satellite Imagery, and High Performance Computing for the Estimation and Distribution of Geospatial Data* (Williamsburg, VA, United States of America: The College of William and Mary).

Grati, F., Scarcella, G., Polidori, P., Domenichetti, F., Bolognini, L., Gramolini, R., et al. (2013). Multi-Annual Investigation of the Spatial Distributions of Juvenile and Adult Sole (Solea Solea L.) in the Adriatic Sea (Northern Mediterranean). *J. Sea Res.* 84, 122–132. doi: 10.1016/j.seares.2013.05.001

Gröger, J., Schnack, D. and Rohlf, N. (2001). Optimisation of Survey Design and Calculation Procedure for the International Herring Larvae Survey in the North Sea. *Arch. Fishery Mar. Res.* 49, 103–116.

Guthery, F. S., Brennan, L. A., Peterson, M. J. and Lusk, J. J. (2005). Information Theory in Wildlife Science: Critique and Viewpoint. *J. Wildlife Manage.* 69, 457–465. doi: 10.2193/0022-541X(2005)069[0457:ITIWSC]2.0.CO;2

Henderson, M. E., Mills, K. E., Thomas, A. C., Pershing, A. J. and Nye, J. A. (2017). Effects of Spring Onset and Summer Duration on Fish Species Distribution and Biomass Along the Northeast United States Continental Shelf. *Rev. Fish Biol. Fisheries* 27, 411–424. doi: 10.1007/s11160-017-9487-9

Hengl, T., Sierdsema, H., Radović, A. and Dilo, A. (2009). Spatial Prediction of Species' Distributions From Occurrence-Only Records: Combining Point Pattern Analysis, Enfa and Regression-Kriging. *Ecol. Model.* 220, 3499–3511. doi: 10.1016/j.ecolmodel.2009.06.038

Hilborn, R. and Walters, C. J. (2013). *Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty* (Berlin, Germany: Springer Science & Business Media).

ICES (2020). *Arctic Fisheries Working Group 2020 Report.*

Jouffre, D., Borges, M., Bundy, A., Coll, M., Diallo, I., Fulton, E. A., et al. (2010). Estimating Eaf Indicators From Scientific Trawl Surveys: Theoretical and Practical Concerns. *ICES J. Mar. Sci.* 67, 796–806. doi: 10.1093/icesjms/fsp285

JRC (2021) *EU Data Collection Web Site* (Accessed Apr. 2022).

Karp, M. A., Peterson, J. O., Lynch, P. D., Griffis, R. B., Adams, C. F., Arnold, W. S., et al. (2019). Accounting for Shifting Distributions and Changing Productivity in the Development of Scientific Advice for Fishery Management. *ICES J. Mar. Sci.* 76, 1305–1315. doi: 10.1093/icesjms/fsz048

Long, R. (2011). The Marine Strategy Framework Directive: A New European Approach to the Regulation of the Marine Environment, Marine Natural Resources and Marine Ecological Services. *J. Energy Natural Resour. Law* 29, 1–44. doi: 10.1080/02646811.2011.11435256

Lotze, H. K., Coll, M. and Dunne, J. A. (2011). Historical Changes in Marine Resources, Food-Web Structure and Ecosystem Functioning in the Adriatic Sea, Mediterranean. *Ecosystems* 14, 198–222. doi: 10.1007/s10021-010-9404-8

Lou, R., Lv, Z., Dang, S., Su, T. and Li, X. (2021). Application of Machine Learning in Ocean Data. *Multimedia Syst.*, 29: 1–10. doi: 10.1007/s00530-020-00733-x

Mancinelli, G., Fazi, S. and Rossi, L. (1998). Sediment Structural Properties Mediating Dominant Feeding Types Patterns in Soft-Bottom Macrobenthos of the Northern Adriatic Sea. *Hydrobiologia* 367, 211–222. doi: 10.1023/A:1003292519784

Marshall, K. N., Koehn, L. E., Levin, P. S., Essington, T. E. and Jensen, O. P. (2019). Inclusion of Ecosystem Information in Us Fish Stock Assessments Suggests Progress Toward Ecosystem-Based Fisheries Management. *ICES J. Mar. Sci.* 76, 1–9. doi: 10.1093/icesjms/fsy152

Maunder, M. N. (2001). A General Framework for Integrating the Standardization of Catch Per Unit of Effort Into Stock Assessment Models. *Can. J. Fisheries Aquat. Sci.* 58, 795–803. doi: 10.1139/f01-029

Maunder, M. N. and Punt, A. E. (2004). Standardizing Catch and Effort Data: A Review of Recent Approaches. *Fisheries Res.* 70, 141–159. doi: 10.1016/j.fishres.2004.08.002

Merow, C., Smith, M. J. and Silander, J. J. A. (2013). A Practical Guide to Maxent for Modeling Species' Distributions: What it Does, and Why Inputs and Settings Matter. *Ecography* 36, 1058–1069. doi: 10.1111/j.1600-0587.2013.07872.x

Metz, M., Andreo, V. and Neteler, M. (2017). A New Fully Gap-Free Time Series of Land Surface Temperature From Modis Lst Data. *Remote Sens.* 9, 1333. doi: 10.3390/rs9121333

Ninˇcevi´c-Gladan, ˇZ., Buˇzanˇci´c, M., Kuˇspilic´, G., Grbec, B., Matijevic´, S., Skejic´, S., et al. (2015). The Response of Phytoplankton Community to Anthropogenic Pressure Gradient in the Coastal Waters of the Eastern Adriatic Sea. *Ecol. Indic.* 56, 106–115. doi: 10.1016/j.ecolind.2015.03.018

Paradinas, I., Conesa, D., López-Quílez, A., Esteban, A., López, L. M. M., Bellido, J. M., et al. (2020). Assessing the Spatiotemporal Persistence of Fish Distributions: A Case Study on Two Red Mullet Species (Mullus Surmuletus and M. Barbatus) in the Western Mediterranean. *Mar. Ecol. Prog. Ser.* 644, 173–185. doi: 10.3354/meps13366

Payne, M. R. (2010). Mind the Gaps: A State-Space Model for Analysing the Dynamics of North Sea Herring Spawning Components. *ICES J. Mar. Sci.* 67, 1939–1947. doi: 10.1093/icesjms/fsq036

Pearson, R. G. (2007). Species' Distribution Modeling for Conservation Educators and Practitioners. *Synthesis. Am. Museum Natural History* 50, 54–89.

Phillips, S. J., Anderson, R. P., Dudík, M., Schapire, R. E. and Blair, M. E. (2017). Opening the Black Box: An Open-Source Release of Maxent. *Ecography* 40, 887–893. doi: 10.1111/ecog.03049

Phillips, S. J. and Dudík, M. (2008). Modeling of Species Distributions With Maxent: New Extensions and a Comprehensive Evaluation. *Ecography* 31, 161–175. doi: 10.1111/j.0906-7590.2008.5203.x

Raybaud, V., Beaugrand, G., Dewarumez, J.-M. and Luczak, C. (2015). Climate-Induced Range Shifts of the American Jackknife Clam Ensis Directus in Europe. *Biol. Invasions* 17, 725–741. doi: 10.1007/s10530-014-0764-4

Renner, I. W. and Warton, D. I. (2013). Equivalence of Maxent and Poisson Point Process Models for Species Distribution Modeling in Ecology. *Biometrics* 69, 274–281. doi: 10.1111/j.1541-0420.2012.01824.x

Rosenberg, A., Bigford, T. E., Leathery, S., Hill, R. L. and Bickers, K. (2000). Ecosystem Approaches to Fishery Management Through Essential Fish Habitat. *Bull. Mar. Sci.* 66, 535–542.

Salon, S., Cossarini, G., Bolzon, G., Feudale, L., Lazzari, P., Teruzzi, A., et al (2019). Marine Ecosystem Forecasts: Skill Performance of the Cmems Mediterranean Sea Model System. *Ocean Sci. Discuss.* 145, 1-35. doi: 10.5194/os-2018-145

Santelli, A., Cvitković, I., Despalatović, M., Fabi, G., Grati, F., Marčeta, B., et al. (2017). Spatial Persistence of Megazoobenthic Assemblages in the Adriatic Sea. *Mar. Ecol. Prog. Ser.* 566, 31–48. doi: 10.3354/meps12002

Sauzède, R., Bittig, H. C., Claustre, H., Pasqueron de Fommervault, O., Gattuso, J.-P., Legendre, L., et al. (2017). Estimates of Water-Column Nutrient Concentrations and Carbonate System Parameters in the Global Ocean: A Novel Approach Based on Neural Networks. *Front. Mar. Sci.* 128. doi: 10.3389/fmars.2017.00128

Scarcella, G. (2018). *The SoleMon CNR Project*.

Schaap, D. M. and Lowry, R. K. (2010). Seadatanet–pan-European Infrastructure for Marine and Ocean Data Management: Unified Access to Distributed Data Sets. *Int. J. Digital Earth* 3, 50–69. doi: 10.1080/17538941003660974

Schmidt, J. O., Van Damme, C. J., Röckmann, C. and Dickey-Collas, M. (2009). Recolonisation of Spawning Grounds in a Recovering Fish Stock: Recent Changes in North Sea Herring. *Scientia Marina* 73, 153–157. doi: 10.3989/scimar.2009.73s1153

Scientific Advisory Committee on Fisheries (2022). "Report of Working Group on Stock Assessment of Demersal Species (WGSAD)," in *Report of 24–28 January 2022 Workshop* (Mediterranean (Rome, Italy:FAO)), 1–122.

Souplet, A. (1995). *Calculation of Abundance Indices and Length Frequencies in the Medits Survey* (Sète, France: Campagne internationale du chalutage démersal en Méditerraneé. Campagne).

Spagnoli, F., Dell'Anno, A., De Marco, A., Dinelli, E., Fabiano, M., Gadaleta, M. V., et al. (2010). Biogeochemistry, Grain Size and Mineralogy of the Central and Southern Adriatic Sea Sediments: A Review. *Chem. Ecol.* 26, 19–44. doi: 10.1080/02757541003689829

Spedicato, M. T., Massutí, E., Mérigot, B., Tserpes, G., Jadaud, A. and Relini, G. (2019). The Medits Trawl Survey Specifications in an Ecosystem Approach to Fishery Management. *Sci. Mar.* 83, 9–20. doi: 10.3989/scimar.04915.11X

Stergiou, K. and Pollard, D. (1994). A Spatial Analysis of the Commercial Fisheries Catches From the Greek Aegean Sea. *Fisheries Res.* 20, 109–135. doi: 10.1016/0165-7836(94)90078-7

Swain, D., Poirier, G. and Sinclair, A. (2000). Effect of Water Temperature on Catchability of Atlantic Cod (Gadus Morhua) to the Bottom-Trawl Survey in the Southern Gulf of St Lawrence. *ICES J. Mar. Sci.* 57, 56–68. doi: 10.1006/jmsc.1999.0516

Thorson, J. T. (2019). Guidance for Decisions Using the Vector Autoregressive Spatio-Temporal (Vast) Package in Stock, Ecosystem, Habitat and Climate Assessments. *Fisheries Res.* 210, 143–161. doi: 10.1016/j.fishres.2018.10.013

Thorson, J. T. and Haltuch, M. A. (2019). Spatiotemporal Analysis of Compositional Data: Increased Precision and Improved Workflow Using Model-Based Inputs to Stock Assessment. *Can. J. Fisheries Aquat. Sci.* 76, 401–414. doi: 10.1139/cjfas-2018-0015

Thorson, J. T. and Kristensen, K. (2016). Implementing a Generic Method for Bias Correction in Statistical Models Using Random Effects, With Spatial and Population Dynamics Examples. *Fisheries Res.* 175, 66–74. doi: 10.1016/j.fishres.2015.11.016

Troupin, C., Barth, A., Sirjacobs, D., Ouberdous, M., Brankart, J.-M., Brasseur, P., et al. (2012). Generation of Analysis and Consistent Error Fields Using the Data Interpolating Variational Analysis (Diva). *Ocean Model.* 52, 90–101. doi: 10.1016/j.ocemod.2012.05.002

Troupin, C., Machin, F., Ouberdous, M., Sirjacobs, D., Barth, A. and Beckers, J.-M. (2010). High-Resolution Climatology of the Northeast Atlantic Using Data-Interpolating Variational Analysis (Diva). *J. Geophysical Research: Oceans* 115, C08005 doi: 10.1016/j.ocemod.2012.05.002

Varshney, S. K., Kumar, J., Tiwari, A., Singh, R., Gunturi, V. and Krishnan, N. C. (2021). Deep Geospatial Interpolation Networks. *arXiv*, 2108.06670. doi: 10.48550/arXiv.2108.06670

Vautard, R., Yiou, P. and Ghil, M. (1992). Singular-Spectrum Analysis: A Toolkit for Short, Noisy Chaotic Signals. *Physica D: Nonlinear Phenomena* 58, 95–126. doi: 10.1016/0167-2789(92)90103-T

Von Schuckmann, K., Le Traon, P.-Y., Smith, N., Pascual, A., Brasseur, P., Fennel, K., et al. (2018). Copernicus Marine Service Ocean State Report. *J. Operational Oceanography* 11, S1–S142. doi: 10.1080/1755876X.2018.1489208

Watelet, S., Back, Ö., Barth, A. and Beckers, J.-M. (2016). "Data-Interpolating Variational Analysis (Diva) Software: Recent Development and Application," in *Proceedings of the EGU General Assembly 2016*, vol. 1. (Vienna, Austria: European Geosciences Union General Assembly).

Weiss, D. J., Atkinson, P. M., Bhatt, S., Mappin, B., Hay, S. I. and Gething, P. W. (2014). An Effective Approach for Gap-Filling Continental Scale Remotely Sensed Time-Series. *ISPRS J. Photogrammetry Remote Sens.* 98, 106–118. doi: 10.1016/j.isprsjprs.2014.10.001

Yan, L. and Roy, D. P. (2018). Large-Area Gap Filling of Landsat Reflectance Time Series by Spectral-Angle-Mapper Based Spatio-Temporal Similarity (Samsts). *Remote Sens.* 10, 609. doi: 10.3390/rs10040609

Zavatarelli, M., Raicich, F., Bregant, D., Russo, A. and Artegiani, A. (1998). Climatological Biogeochemical Characteristics of the Adriatic Sea. *J. Mar. Syst.* 18, 227–263. doi: 10.1016/S0924-7963(98)00014-1

Zhang, Q.-s. and Zhu, S.-C. (2018). Visual Interpretability for Deep Learning: A Survey. *Front. Inf. Technol. Electronic Eng.* 19, 27–39. doi: 10.1631/FITEE.1700808

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EDITED BY
Tomaso Fortibuoni,
Istituto Superiore per la Protezione e la Ricerca Ambientale (ISPRA), Italy

REVIEWED BY
Pablo Del Monte-Luna,
Centro Interdisciplinario de Ciencias Marinas (IPN), Mexico
Juan Zwolinski,
University of California, Santa Cruz, United States

*CORRESPONDENCE
Enrico Nicola Armelloni
enrico.armelloni@irbim.cnr.it

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The potential effects of COVID-19 lockdown and the following restrictions on the status of eight target stocks in the Adriatic Sea

Giuseppe Scarcella[1], Silvia Angelini[1,2], Enrico Nicola Armelloni[1,3]*, Ilaria Costantini[1], Andrea De Felice[1], Stefano Guicciardi[1], Iole Leonori[1], Francesco Masnadi[1,3], Martina Scanu[1,3] and Gianpaolo Coro[4]

[1]Institute for Biological Resources and Marine Biotechnology (IRBIM), National Research Council of Italy (CNR), Ancona, Italy, [2]Fano Marine Center, The Inter-Institute Center for Research on Marine Biodiversity, Resources and Biotechnologies, Pesaro Urbino, Italy, [3]Department of Biological, Geological and Environmental Sciences (BiGeA), University of Bologna, Bologna, Italy, [4]Institute of Information Science and Technologies (ISTI), National Research Council of Italy (CNR), Pisa, Italy

The COVID-19 pandemic had major impacts on the seafood supply chain, also reducing fishing activity. It is worth asking if the fish stocks in the Mediterranean Sea, which in most cases have been in overfishing conditions for many years, may have benefitted from the reduction in the fishing pressure. The present work is the first attempt to make a quantitative evaluation of the fishing effort reduction due to the COVID-19 pandemic and, consequently, its impact on Mediterranean fish stocks, focusing on Adriatic Sea subareas. Eight commercially exploited target stocks (common sole, common cuttlefish, spottail mantis shrimp, European hake, red mullet, anchovy, sardine, and deepwater pink shrimp) were evaluated with a surplus production model, separately fitting the data for each stock until 2019 and until 2020. Results for the 2019 and 2020 models in terms of biomass and fishing mortality were statistically compared with a bootstrap resampling technique to assess their statistical difference. Most of the stocks showed a small but significant improvement in terms of both biomass at sea and reduction in fishing mortality, except cuttlefish and pink shrimp, which showed a reduction in biomass at sea and an increase in fishing mortality (only for common cuttlefish). After reviewing the potential co-occurrence of environmental and management-related factors, we concluded that only in the case of the common sole can an effective biomass improvement related to the pandemic restrictions be detected, because it is the target of the only fishing fleet whose activity remained far lower than expectations for the entire 2020.

KEYWORDS

COVID-19, CMSY and BSM, Adriatic sea, stock assessment and management, fishing effort and co-management

Introduction

The COVID-19 global pandemic has forced many governments to temporarily shut down large segments of their economies to promote social distancing and reduce the infection rate, including businesses, restaurants, and schools (Althouse et al., 2020; White and Hébert-Dufresne, 2020; Hale et al., 2021). The unpredictable impacts of COVID-19 itself, and responses to it, were indeed felt throughout the food supply chains, including the seafood sector, at local and global scales (Bennett et al., 2020; FAO, 2020a; Love et al., 2021). Depending on the typology of the fishery, it is possible to detect a sort of gradient in the effect of COVID-19, going from a fishing effort reduction that was almost negligible (Coro et al., 2022c) to a complete shutdown of some fisheries (Pita et al., 2021).

In areas such as the Mediterranean Sea, where the status of the stock was considered for many years worrisome (Froese et al., 2018; FAO, 2020b), it is of great interest to understand whether the effects of the 2020 lockdown and the following restrictions in terms of social distancing have had any positive influence on the restocking of fishery resources. The present work aims to attempt an evaluation of the short-term effects of the pandemic-related exploitation pressure release on commercially exploited fish stock status. We focused on the most important target stocks of the Adriatic Sea, a subarea of the Mediterranean Sea (Geographical Sub-Areas 17 and 18; FAO, 1999) and one of the most exploited areas in the world in terms of trawling (Eigaard et al., 2017; Amoroso et al., 2018), with a high rate of productivity as well (Campanelli et al., 2011). The five countries bordering the Adriatic Sea and involved in fisheries, namely, Italy, Croatia, Slovenia, Albania, and Montenegro, all implemented several restrictions in 2020 due to COVID-19. In Italy, a lockdown period was imposed from March 11 to May 17. In almost the same period, Croatia, Slovenia, Albania, and Montenegro also adopted similar measures (Coro et al., 2022c). The restrictions in terms of social distancing affected the fishery sector, and a strong reduction in seafood requests caused a decrease in fishing pressure (Coro et al., 2022c) and related activities, such as fish markets (Pititto et al., 2021). As soon as the harsh restrictions were loosened, fishing intensity in the Adriatic Sea quickly reverted to pre-COVID levels for the majority of the fleet categories (Russo et al., 2021; Russo et al., 2022; Coro et al., 2022c).

COVID-related restrictions in the Adriatic Sea need to be examined within a fishery system where factors such as environmental conditions and management measures can be seen as major forces acting on the stock dynamics (Coll et al., 2009; Fortibuoni et al., 2017). Therefore, we ought to make some assumptions to try to disentangle these factors. Concerning environmental conditions, the climate change effect in this area has been documented for many years (Ben Rais Lasram et al., 2010; Fortibuoni et al., 2015). Considering the inertia underlining these processes, we can assume that the environmental drivers observed in 2020 trace a continuum with 2019. Regarding the effect of management, the major instruments in place for 2020 were the multiannual management plan for the sustainable demersal fisheries in the Adriatic Sea (GFCM/43/2019/5), the management plan for the sustainable small pelagic (anchovy and sardine) fisheries in the Adriatic Sea (last update GFCM/42/2018/8), and the recommendation defining the Pomo/Jabuka pits as a Fishery Restricted Area (FRA; GFCM/41/2017/3). Among these measures, only GFCM/43/2019/5 was deemed to start in 2020, while other recommendations were already put in place in previous years. Due to the lack of responsiveness of the management system in the Mediterranean Sea (Cardinale et al., 2017), the measures that were decided before the start of the global pandemic were applied without any change: for example, the reduction in fishing days foreseen in management plans on demersal or small pelagic stocks was applied in 2020. Therefore, in some cases the pandemic-related restrictions added up to the fishing ban foreseen by the management plan.

Before discussing any potential effect of the pandemic restrictions, there is the need to quantify the effective fishing pressure reduction attributable to the anti-COVID measures and determine which fleets are mostly affected. To account for interactions with management measures already in place, we accompany a description of the fishing effort alterations between 2019 and 2020 with a brief review of the ongoing management actions, trying to distinguish the fishing activity reduction due to management from those due to the pandemic. We then proceed to quantify the potential reduction in fishing mortality and the potential increase in biomass at sea of the most important target stocks exploited by Adriatic fleets, namely, six demersal and two small pelagic stocks routinely assessed within the General Fishery Commission for the Mediterranean (GFCM) and the

Scientific, Technical and Economic Committee for Fisheries of the European Commission (STECF). We conclude by discussing the merits of a management system based on fishing effort reduction, especially when using fishing days as a unit of measure.

Material and methods

Fishing effort alteration

Effort data in terms of fishing days and hours at sea are available for European fleets from the STECF FDI 2021 data call (STECF, 2021a); fishing hours from AIS data processing are available from a regional study (Coro et al., 2022c) and were used for comparison. In the FDI dataset, data for pots, nets, and longlines were grouped under the "Passive" category; data for Croatian Dredges were grouped with Italian beam trawlers (Armelloni et al., 2021). The resulting information was aggregated by quarter and the effort change calculated by dividing the value shown in 2020 by the value shown in 2019.

Target stocks and associated data

The demersal and small pelagic stocks considered in the present study are the following: common sole (*Solea solea*), common cuttlefish (*Sepia officinalis*), and spottail mantis shrimp (*Squilla mantis*) in Geographical Subarea (GSA) 17, that is, the North and Central Adriatic Sea; European hake (*Merluccius merluccius*), red mullet (*Mullus barbatus*), anchovy (*Engraulis encrasicolus*), and sardine (*Sardina pilchardus*) in GSAs 17-18, which include the whole Adriatic basin (North, Central, and South Adriatic Sea); and deepwater pink shrimp (*Parapenaeus longirostris*) in GSAs 17–18–19, corresponding to the entire Adriatic Sea and the Western Ionian Sea. In the 2015–2020 period, the listed stocks represented on average almost 70% and 50%, respectively, in landing weight (tonnes) and value (euros) of the entire commercial fishing activity of EU fleets working in GSAs 17 and 18 over the last 5 years (source: STECF, 2020a). Moreover, the species are the main target of important fleets operating in the area as bottom otter trawlers (OTB), bottom beam trawlers (TBB), pelagic pair trawlers (PTM), purse seiners (PS), and small-scale fishery using passive gears (PGP). In addition, they are also subject to yearly stock assessments carried out within the framework of both STECF and GFCM working groups. Different sources were used to reconstruct fishery-dependent and -independent time series to be used in the following model. Data for demersal species were gathered from STECF FDI (STECF, 2021a), STECF stock assessment, GFCM stock assessment reports (FAO, 2021a; FAO, 2021b; FAO, 2022a), and GFCM stock assessment forms (SAFs updated to the reference year 2019 at the time of writing; available at https://www.fao.org/gfcm/data/safs/zh/); catch data for small-pelagic species were extracted from the graphs contained in FAO (2022b), and fishery-independent data for small pelagic species were provided by CNR-IRBIM. Stock assessment models used in the official stock assessments are Stock Synthesis 3 (SS3, Methot and Wetzel, 2013), assessment 4 all (a4a, Jardim et al., 2014), CMSY (Froese et al., 2017), and the FLR implementation of the SAM method (FLSAM, Payne and Hintzen, 2013) (details by species are reported in Table 1).

TABLE 1 Summary of the stocks analysed in the present paper, with references to the FAO 3 alpha code, the assessment method used in the official context, the geographical aggregation (GSAs), and data and priors used in the CMSY analysis.

Stock	FAO 3 alpha code	Official Stock Assessment method	GSAs	Start year	*r* range	Biomass start	Biomass int. yr	Biomass int	Biomass end
Engraulis encrasicolus	ANE	FLSAM	17-18	2000	0.39-0.91	0.01-0.4	NA	NA	0.01-0.4
Merluccius	HKE	SS3	17-18	1998	0.35-0.8	0.01-0.4	NA	NA	0.01-0.4
Mullus barbatus	MUT	A4a	17-18	2006	0.42-1.04	0.01-0.4	NA	NA	0.01-0.4
Parapenaeus longirostris	DPS	A4a	17-18-19	2002	0.68-1.54	0.01-0.4	NA	NA	0.2-0.6
Sardina pilchardus	PIL	FLSAM	17-18	2000	0.4-0.9	0.01-0.4	NA	NA	0.01-0.4
Sepia officinalis	CTC	CMSY	17	1974	0.37-0.84	0.4-0.8	2007	0.1-0.5	0.05-0.4
Solea solea	SOL	SS3	17	1958	0.33-0.76	0.4-0.8	NA	NA	0.01-0.4
Squilla mantis	MTS	SS3	17	1953	0.37-0.84	0.2-0.6	NA	NA	0.01-0.4

The reference year for all the assessments was 2020. NA, not assigned.

CMSY model

The stock assessment of the selected stocks was performed using the CMSY model (Froese et al., 2017). CMSY includes a Bayesian Schaefer model (BSM), which fits catch and—optionally—biomass (or catch-per-unit-of-effort) data through a Markov chain Monte Carlo method based on the Schaefer function for biomass dynamics. The model estimates fisheries reference points (MSY, F_{MSY}, B_{MSY}), as well as relative stock size (B/B_{MSY}) and exploitation (F/F_{MSY}) from catch data and broad priors for "resilience" (approximated by r) and stock's relative biomass (B/k) at the beginning and end of the catch time series. For the purposes of this paper, BSM was applied on landing data and biomass indices. Table 1 summarizes the input data and priors used for each stock. Catch data for all species were derived from published stock assessment working group reports (see the previous section). The biomass indices for common sole, common cuttlefish, and spottail mantis shrimp in GSA 17 were obtained from the SoleMon project (Grati et al., 2013), a trawl survey carried out from 2005 up to present times with Rapido trawl (Hall-Spencer et al., 1999) over a 36,742-km^2 area of the Northern and Central Adriatic Sea (Scarcella et al., 2014). In 2020, no SoleMon survey was carried out in the area comprised between the midline and Croatian waters, and missing hauls were provided by model-based estimates (Coro et al., 2022a). The biomass indices for European anchovy and sardine in GSAs 17–18 were obtained from the MEDiterranean International Acoustic Survey (MEDIAS; Leonori et al., 2021). MEDIAS surveys were carried out in the European Mediterranean Sea following a standardised protocol to provide inputs for the management of small pelagic fish. MEDIAS data used in this paper are in tonnes by NM^2 obtained from the survey conducted in the western Adriatic Sea (Italian and Slovenian water), where the acoustic survey has been performed since 1976. The biomass indices for European hake and red mullet in GSAs 17–18 and deep-water rose shrimp in GSAs 17–18–19 were collected from open-source stock assessment reports (STECF, 2021b), which include the annual biomass at sea values estimated from the International Bottom Trawl Survey in the Mediterranean Sea (MEDITS; Anonymous, 2017; Spedicato et al., 2019). The MEDITS is a European survey which started in 1994 in order to collect data and information about the demersal communities inhabiting the Mediterranean basin. Priors for r were either taken from previous specific studies in this area (Froese et al., 2018; Armelloni et al., 2021) or inferred from their averages in FishBase and SeaLifeBase (Palomares and Pauly, 2018; Froese and Pauly, 2019). The choice of an increasing pattern from the initial to final depletion priors in the reference models was supported by an overall increase in fishing pressure in the Adriatic Sea (Colloca et al., 2017) followed by a reduction in the productivity of the commercial fishery over the study period (Marini et al., 2017). Moreover, the assessment results available from official stock assessment reports and GFCM SAF reports 2020 were also used to infer the depletion priors. The CMSY-BSM models were employed with time series until 2019 (before COVID-19) and until 2020 (during COVID-19). The results were compared to understand the impact of the COVID pandemic using the ratio of F_{2020}/F_{2019} and B_{2020}/B_{2019}. An overview of the official stock assessment results for 2019 and 2020 is provided for completeness (Table 1).

Statistical comparisons of the CMSY outputs

The statistical comparisons between 2019 and 2020 F/F_{MSY} and B/B_{MSY} data were carried out using bootstrap (Efron and Tibshirani, 1993). For each species and each variable of interest, the difference in medians between the years 2020 and 2019 was tested as follows (Efron and Hastie, 2016). Raw data were the 6,000 point estimates for B/B_{MSY} and F/F_{MSY} coming from the JAGS implementation in the CMSY model, which are the data already providing the CMSY results. Starting from the raw data, for each species, 10,000 replicates with replacements for both 2020 and 2019 were carried out. From these replicates, 10,000 values of the difference between 2020 and 2019 medians were derived. On this set of 10,000 differences, the mean and the relative 95% CI were then calculated. If the 95% CI did not capture the zero value, the difference was deemed statistically significant. All the calculations were carried out using the free statistical software R (R Core Team, 2022).

Results

In the present section, comparisons between 2020 and 2019 fishery-dependent and -independent data were carried out. A similar comparison was carried out on the CMSY/BSM outputs as well.

Fishing effort alteration

When commenting on the results, it must be considered that some differences between AIS data and FDI data can be imputed to the following factors: (i) AIS gear categorization is model-based (Galdelli et al., 2021), while FDI data are assigned to fishing gear based on official registers (EU, 2020); (ii) AIS data do not include small vessels (smaller than 15 m of LFT) that are not equipped with AIS transponder; non-EU fleets (Albania and Montenegro) are included only in the AIS data. The comparison of 2020 *vs.* 2019 reduction among fishing days, hours at sea (from FDI data), and fishing hours (from AIS data) did not yield a coherent picture of the fleets considered (Figure 1), since in

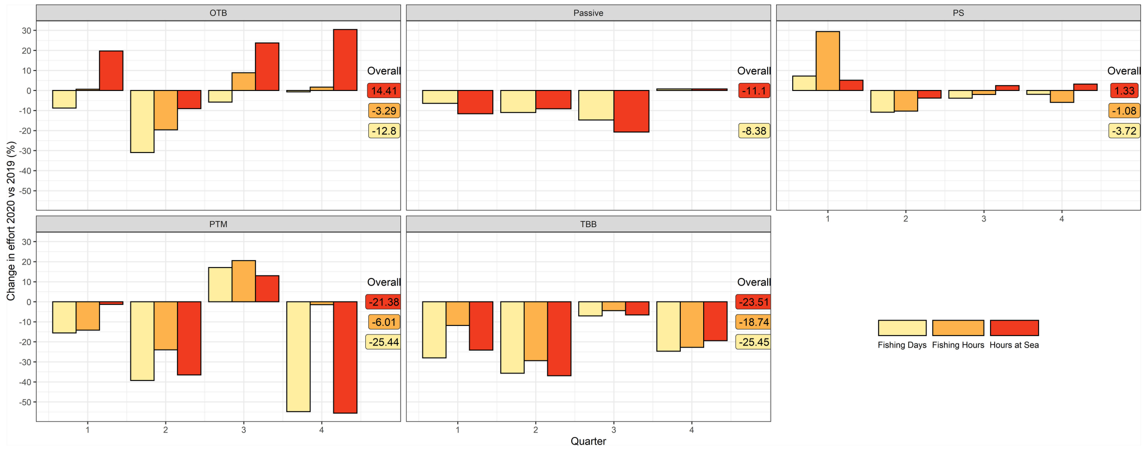

FIGURE 1
Seasonal change in fishing effort in the period 2019–2020 in the Adriatic Sea (GFCM-GSAs 17 and 18). Fishing days and hours at sea are derived from FDI data (STECF, 2021a) fishing hours from AIS data taken from Coro et al., 2022c. Metièrs are otter bottom trawlers (OTB), purse seiners (PS), pelagic midwater trawlers (PTM), and beam bottom trawlers (TBB). TBB comprises dredgers (DRB) from Croatia that are using beam trawl. Passive gears include the following gears: FPO, FYK, GND, GNS, GTN, GTR, HMD, LHM, LHP, LLD, LLS, LTL; Other include the following gears: OTM, NK, SB, SV. Acronyms full list available at https://datacollection.jrc.ec.europa.eu/wordef/gear.

most of the cases the magnitude of the difference observed between 2019 and 2020 depends on the indicator selected. Fishing activity metrics agreed on a reduction in Bottom trawlers (OTB) activity only in the second quarter of 2020. In detail, during the first quarter an increase was observed in fishing activity when focusing on hours at sea not confirmed by other metrics (barely any change in fishing hours and a decrease in fishing days). In the second quarter, a marked reduction was observed in fishing days (as much as -30% in the second quarter), corresponding to a less marked reduction in fishing hours (-20%) and a slight decrease in hours at sea (-10%). In the third quarter, fishing days were indicating a slight reduction (-6%), while fishing hours (+ 9%) and hours at sea (+24%) suggested that the activity was higher than pre-COVID levels. In the fourth quarter, hours at sea indicated a steep increase while the other indicator suggested a fishing activity very similar to the previous year. Passive gears (not available from AIS data) showed a slight activity reduction in both fishing days and hours at sea for quarters 1–3, with the third quarter showing the most severe reduction. The fourth quarter registered a negligible variation vis-à-vis the year 2019. Purse seiners (PS) showed a less marked variation in terms of fishing activity; namely, a sudden increase in fishing hours was observed in the first quarter (not confirmed by the other metrics), followed by a moderate reduction in the second quarter, not exceeding 10%, as confirmed by all indicators. In the rest of the year, the activity variation remained almost negligible, generally comprised within ±6%. Pelagic pair trawlers (PTM) experienced a strong activity reduction already in the first quarter (not confirmed by the hours at sea). In the second quarter, indicators confirmed a strong reduction, exceeding -20% at least. The fishing activity reverted to a value higher than the pre-COVID period in the third quarter, with an increase of 10%–20% (depending on the indicator). During the fourth quarter, the FDI indicators reported a fishing activity drop of over 50%, while the AIS data suggested values comparable to 2019. Beam trawlers (TBB) were the only fleet showing decreasing values in all quarters, regardless of the indicator considered. During the first quarter, a moderate decrease was observed in each indicator, with values comprised between -10% and -28%. In the second quarter, a marked reduction, around -30% on average, was discernible. In the third quarter, the fishing activity recovered but remained slightly below the previous year levels. In the fourth quarter, the fishing activity dipped again, with a reduction of close to -20% confirmed by all indicators.

Target stocks and associated data

According to FDI statistics and data available from stock assessment reports, a clear increase of catches (C) in 2020, around 15%, has been observed only for European hake, unlike the remaining stocks that showed a decrease in catches, more evident for demersal species such as common cuttlefish, red mullet, and common sole. The biomass indexes (I) available from bottom trawl surveys (SoleMon and MEDITS) and an acoustic survey (MEDIAS) showed a notable reduction for deepwater pink shrimp and common cuttlefish, approximately

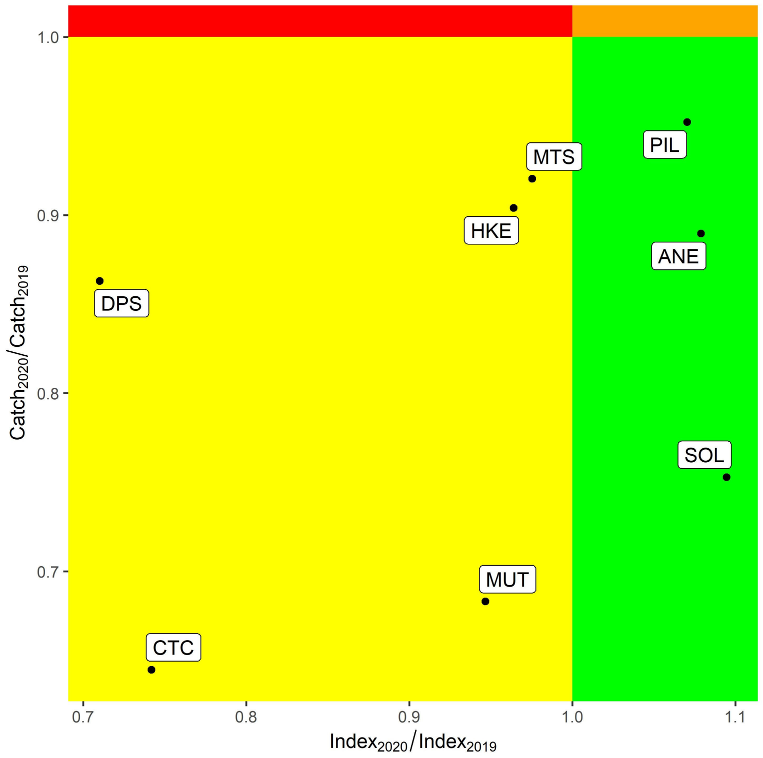

FIGURE 2
Kobe plot with ratio of catches (C) and biomass indexes (I) in 2020 and 2019 of the eight Adriatic Sea target stocks.

TABLE 2 Results of the official stock assessment models and from the CMSY models.

Stock	Stock assessment	F/F_{ref} 2019	F/F_{ref} 2020	F/F_{ref} 2020 *vs.* F/F_{ref} 2019	B/B_{ref} 2019	B/B_{ref} 2020	B/B_{ref} 2020 *vs.* B/B_{ref} 2019
Engraulis encrasicolus	Official	1.51	1.15	0.762	0.422	0.49	1.161
	CMSY	1.833	1.252	0.683	0.8	1.1	1.375
Merluccius merluccius	Official	2.72	2.47	0.908	1.56	1.62	1.038
	CMSY	0.819	0.735	0.897	0.823	0.84	1.021
Mullus barbatus	Official	2.029	1.028	0.506	8306*	10411*	1.253
	CMSY	0.931	0.69	0.74	0.864	0.872	1.01
Parapenaeus longirostris	Official	2.98	2.256	0.757	3245.5*	3246.833*	1
	CMSY	1.093	1.086	0.994	1.078	0.973	0.903
Sardina pilchardus	Official	4.43	NA	NA	0.67	NA	NA
	CMSY	2.111	1.938	0.918	0.535	0.549	1.027
Sepia officinalis	Official	0.81	1.17	1.444	0.49	0.36	0.735
	CMSY	1.096	1.171	1.068	0.417	0.363	0.871
Solea solea	Official	1.125	0.81	0.72	0.69	0.73	1.058
	CMSY	1.202	0.997	0.829	0.893	0.911	1.02
Squilla mantis	Official	0.917	0.79	0.862	0.74	0.92	1.243
	CMSY	1.013	0.874	0.863	0.797	0.807	1.013

For the official models, 2019 sources for Sepia officinalis, Merluccius merluccius, Squilla mantis is (FAO, 2021a), common sole in 2019 (FAO, 2021b), Parapenaeus longirostris and Mullus barbatus (STECF, 2020b), Engraulis encrasicolus and Sardina pilchardus (FAO, 2021c) and 2020 for Sepia officinalis, Merluccius merluccius, Squilla mantis and common sole (FAO, 2022a), Parapenaeus longirostris and Mullus barbatus (STECF, 2021b), Engraulis encrasicolus, and Sardina pilchardus (FAO, 2022b). *Unit of measure is SSB; NA, not available. Fref and Bref correspond to the estimated F_{MSY} and B_{MSY} values for the stock assessment developed with CMSY, whereas Fref and Bref for the official stock assessments agree with those officially selected and set out in the reports cited above.

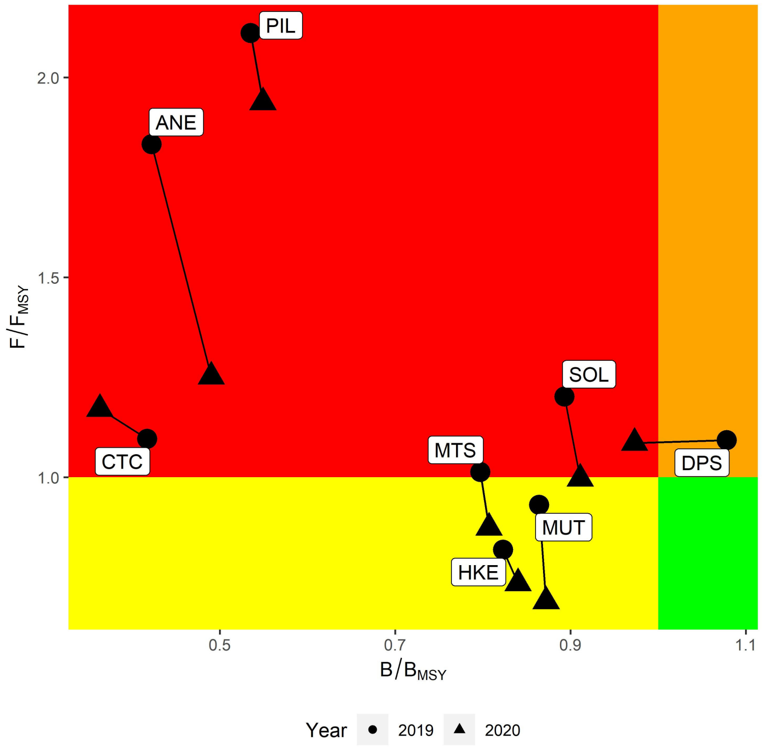

FIGURE 3
Kobe plot showing the outputs of CMSY/BSM of eight Adriatic Sea target stocks in 2019 (triangle) and 2020 (circle). Lines link the outputs for the same species in different years.

30%, while the remaining stocks showed a stable trend from 2019 to 2020, with a moderate increase for small pelagic species and common sole (Figure 2).

CMSY model

The results of the stock assessment model used to investigate the status of the eight Adriatic Sea target stocks (Table 2 and Figure 3) revealed that both in 2019 and in 2020, most of the stocks were in overfishing ($F_{current}$ higher than F_{MSY}) and overfished ($B_{current}$ lower than B_{MSY}) conditions. Stock trajectories for B/B_{MSY} and F/F_{MSY} for the CMSY models having as reference year 2019 and 2020, respectively, are available in the Supplementary Information (Figures SI 2, 3). Common cuttlefish, anchovy, sardine, and hake proved to be in overfishing and overfished conditions in both years considered. Common sole and spottail mantis shrimp showed an improvement in terms of reduction in fishing mortality, which was below the F_{MSY} in 2020. Deepwater pink shrimp was in overfishing conditions in 2019 only and in both overfishing and overfished conditions in 2020. Red mullet stock showed a current F below F_{MSY} in both 2019 and 2020. All the stock assessment outputs showed wide ranges of uncertainty. All the CMSY model outputs and diagnostics are available in the Supplementary Information (Figures SI 4 to 83). The direction of the difference (whether positive or negative) between results for 2019 and 2020 is coherent between the estimates of the present paper and the official stock assessment, except for deepwater rose shrimp biomass that was stable in the official stock assessment and slightly decreasing in this paper.

Statistical comparison of the CMSY outputs

CMSY point estimation for the reference years 2019 and 2020 is available in the Supplementary Information, Figure SI 1. Looking at the comparative plot in Figure 4 and Tables 2, 3, it is possible to detect a small but significant improvement in stock conditions for most of the stocks in terms of both biomass at sea and reduction in fishing mortality. The countertrend stocks were common cuttlefish and deepwater pink shrimp, which showed a reduction in biomass at sea and, only for common cuttlefish, an increase in fishing mortality. The only non-significant

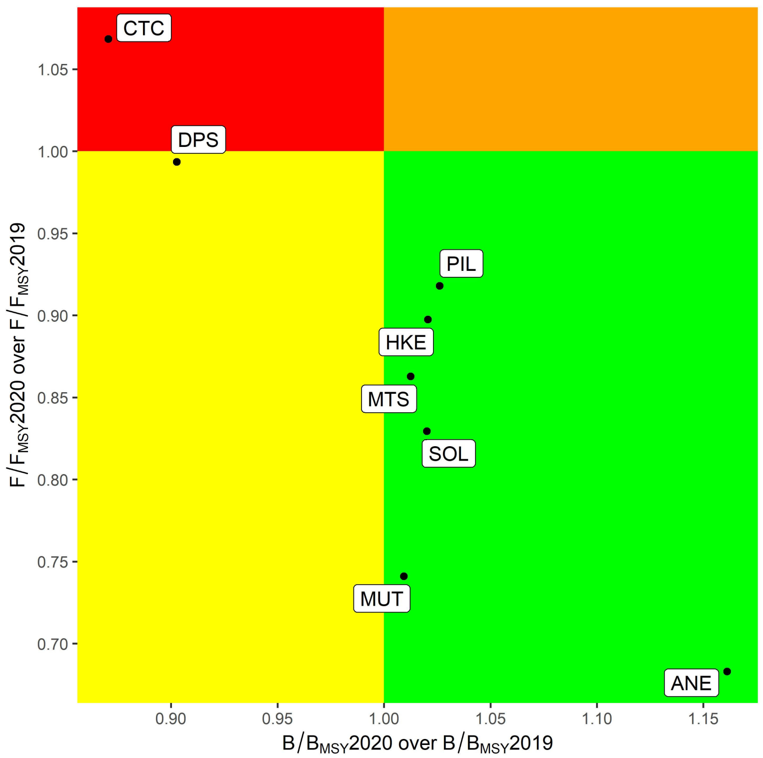

FIGURE 4
Comparative Kobe plot showing the 2020/2019 ratio of the F/F_{MSY} and B/B_{MSY} outputs from the CMSY/BSM outputs of eight Adriatic Sea Target stocks.

differences were observed for deep-water pink shrimp in respect of fishing mortality.

Discussion

According to our knowledge, the present study is the first attempt to quantify the status of target stocks concerning the 2020 COVID-19 lockdown and related restrictions that occurred in the countries exploiting the resources in the entire Adriatic Sea. It is important to stress that pre-crisis fisheries evaluations which showed that the situation of several of our target stocks was already worrying, in agreement with historical data and recent studies (Colloca et al., 2017; Fortibuoni et al., 2017; Armelloni et al., 2021).

TABLE 3 Mean and 95% confidence intervals of the B/B_{MSY} and F/F_{MSY} differences in medians between 2020 and 2019 calculated by bootstrap as a function of species.

Stock	Area	B/B_{MSY}	LCI	UCI	Sign.	F/F_{MSY}	LCI	UCI	Sign.
Engraulis encrasicolus	17,18	0.068	0.060	0.076	*	0.579	0.521	0.631	*
Merluccius merluccius	17,18	0.017	0.014	0.021	*	0.084	0.078	0.091	*
*Mullus barbatus**	17,18	0.009	0.004	0.014	*	0.242	0.233	0.251	*
*Parapenaeus longirostris**	17,18,19	-0.105	-0.112	-0.097	*	0.006	-0.007	0.019	
Sardina pilchardus	17,18	0.015	0.008	0.021	*	0.174	0.139	0.209	*
Sepia officinalis	17	-0.054	-0.060	-0.049	*	-0.075	-0.106	-0.044	*
Solea solea	17	0.017	0.013	0.021	*	0.206	0.195	0.217	*
Squilla mantis	17	0.01	0.004	0.016	*	0.138	0.127	0.149	*

LCI and UCI represent the lower and upper 95% confidence interval limit, respectively. A difference is considered statistically significant when its confidence interval does not include 0. In the table, these differences are highlighted in bold and marked with the * symbol in the "Sign." Column.

Management of target stocks and COVID-related effects on fishing

The review of fishing effort data done in the present paper highlights that all major commercial fleets fishing in the Adriatic Sea reduced their activity during the months when severe COVID-19 restrictions were in place. When the social limitations were relaxed, the fishing fleets responded in a heterogeneous way: some fisheries were unable to fully recover to pre-COVID rates, while others increased their activity to levels higher than in 2019. The multiannual management plan for the sustainable exploitation of demersal stocks in the Adriatic Sea adopted in 2019 (GFCM/43/2019/5) required a substantial fishing effort reduction for the 2020–2021 period to foster stock rebuilding. In particular, in 2020 it established a 12% fishing-day reduction for OTB and a 16% fishing-day reduction for TBB, compared to their average values for the 2015–2018 period. Based on 2021 FDI data, on a subset considering European fleets in GSAs 17 and 18, the fishing-day values observed in 2020 suggest that the effort reduction was in line with the expectation for OTB (-12.8%) and far higher for TBB (-25.4%). However, when fishing hours are used as an indicator, the change in fishing effort drops to just -3.3% for OTB and to -18.7% for TBB. Seasonal fishing data give us more detail on the topic. OTB experienced a sharp activity reduction in spring (detected by all the indicators) followed by a quick recovery, and activity in summer/fall was comparable to or even higher than the previous year depending on the indicator. Therefore, the effect of COVID-19 restrictions on OTB mostly concentrated on the fishing-day reduction foreseen by the management plan within the spring period, and the dynamics observed in the second half of 2020 suggests a reframing of the fishing activity instead of an effective fishing pressure reduction. On the contrary, all indicators agree on depicting TBB activity in 2020 at levels lower than 2019 for the entire year, in any case describing a more severe reduction than the management plan claims. As a consequence, there is evidence that the pandemic restriction effectively forced TBB activity to be far lower than expectations. Another management aspect for demersal fisheries that might have interacted with COVID restrictions is GFCM/41/2017/3, which defines a Fishery Restricted Area (FRA) in the Pomo/Jabuka pit, a heavy exploited and productive area. The Pomo FRA is deemed to protect essential fish habitat for European hake and Norway lobster, with the first experiencing a positive biomass trend over recent years (Chiarini et al., 2022). Considering that the effective fishing effort reduction in the Pomo area was limited to a few weeks (FAO, 2021d) and that this area is not fished by TBB (Armelloni et al., 2021; Coro et al., 2022c), we assume a negligible interaction of this management measure with COVID-related restrictions.

The management plan for the sustainable exploitation of small pelagic (anchovy and sardine) stocks in the Adriatic Sea was adopted in 2013 (GFCM/37/2013/1). After several updates, the latter regulation (GFCM/42/2018/8) recommends for 2019–2021 a limitation of fishing days equal to 180 per year (not exceeding 144 days targeting a single species) and a reduction in catches of 5% year after year. In addition, seasonal closures of 30 continuous days within the period 1 October–31 March for sardine and 1 April–30 September for anchovy are foreseen; these fishing bans are differentiated by country, region, and fishing gear (purse seine and pelagic trawl). The only fishing ban falling within the lockdown period was the one for Italian purse seiners targeting sardine, which was planned for the period 20 February–21 March 2020. All the other fishing bans were planned in different periods and did not overlap with the most severe COVID restrictions. Therefore, if the management plan was not imposing a fishing effort reduction from 2019 to 2020, based on the interaction between management recommendations and COVID restrictions, it is reasonable to expect a slight reduction in fishing activity for the gears targeting small pelagic species, especially for PTM, because of the cumulative effect of COVID restrictions and the planned fishing ban. When looking at the data, PS activity was almost unchanged, while a marked drop is detected in PTM fishing days (although fishing hours indicate a less severe reduction). Seasonal data confirm that PS activity remained quite stable throughout the year, experiencing a less severe contraction during the 2020 spring among the analysed fleets. As a consequence, there is no evidence that the pandemic restrictions caused an effective reduction in PS activity. PTM activity underwent a strong reduction during the lockdown period, then the fishery quickly recovered in summer according to all indicators. The discrepancy observed among the indicators for PTM in fall complicates the interpretation of data, because it may either indicate a dramatic reframing of the fishing practice or a bias in the fishing data model estimate. However, when looking at FDI data, including years from 2014, what emerges is a slight but continuous contraction of PTM activity over the last 5 years. Therefore, for PTM there are indications that the pandemic restrictions added up as an extra driver of fishing effort reduction to a fleet segment already experiencing a gradual contraction.

Environmental drivers

Environmental drivers that could influence a population increase in 2020 were scarcely effective for common sole, since its habitat distribution is not sensible enough to the magnitude of alteration of chlorophyll, dissolved oxygen, and temperature that occurred in 2020 compared to the previous years (Coro et al., 2022b). Environmental change was also unlikely to be strong enough to change the habitat distribution of spottail mantis shrimp and anchovy. However, temperature increase and dissolved oxygen decrease in 2020—which were climate change-related trends—might have fostered the habitat distribution of common cuttlefish and penalised that of pink shrimp with a resultant influence on biomass. Habitat unsuitability might indeed be one additional reason for the particular condition of deepwater pink shrimp shown in Figure 4. Instead, COVID-specific environmental changes, like chlorophyll-a decrease, slightly penalised sardine (Coro et al., 2022b). A substantial water-column chlorophyll-a decrease from 2019 to 2020 was indeed measured in the Adriatic and was likely a consequence of the COVID-19 pandemic. This reduction was observed worldwide and potentially corresponded to CO_2 emission dropping in several areas as a result of human activity reduction (Coro et al., 2022b). One logical explanation is that chlorophyll-a is an integral part of the carbon cycle, because this cycle strongly depends on CO_2 consumption during photosynthesis. Thus, in the global balance of the natural carbon cycle, a significant CO_2 decrease likely corresponds to a chlorophyll-a level decrease because of the lower demand for CO_2 uptake. When coupled with temperature increase, a lower chlorophyll-a creates new environmental conditions that may influence species' presence and abundance (Coro et al., 2022a).

Comparison between CMSY/BSM outputs and official assessments both in 2019 and in 2020

Most of the stocks analysed in the present work are routinely assessed, generally with age-based models, in the framework of the Scientific Advisory Committee (SAC) of the GFCM. We acknowledge that in this work we applied a surplus production model (SPM) to every stock, a methodology that is sometimes defined as too simplistic to account for the variability of size structure, ecological drivers, and catchability of a real-world stock (Pedersen and Berg, 2017). Nevertheless, modern SPMs greatly improved on these caveats by modulating the error component, thereby improving the model parameter estimation, and nowadays these models are widely used in the official stock assessment context as well. Since the purpose of this paper is not to revise the stock assessment methodology in the Adriatic Sea but to assess whether COVID-19-related restrictions have in some way affected the stock status, we preferred to adopt a common methodology for all stocks. In this case, an SPM such as CMSY was the best candidate because of its flexibility, ease of use, and successful applications to a variety of stocks worldwide (Froese et al., 2018; Palomares et al., 2020). Nevertheless, to verify consistency with the official stock assessment, before commenting on the possible effect of COVID-19 restrictions on the stocks, we provide a brief qualitative comparison of the results obtained in the present work and official stock assessments (STECF, 2020c; FAO, 2021a; FAO, 2021b; STECF, 2021b; FAO, 2021c; FAO, 2022a; FAO, 2022b). In the SI, all the CMSY model outputs (Figures SI 4 to 83) are included, to allow the reader to carry out a detailed comparison with the results provided in the official stock assessment reports.

Regarding demersal species, the situation of hake depicted by the official stock assessment differs from the one described by the CMSY model. F results are below the reference values for both years in the CMSY model, describing a downward trend from 2019 to 2020; this reduction is also depicted by the official stock assessment, estimating however a value of F above the respective reference points. SSB increases from 2019 to 2020, as underlined by all the assessments; however, the official stock assessments depict a more positive situation in which SSB is above the reference points for 2019 and 2020 alike, whereas the biomass is below B_{MSY} for the CMSY models. One of the main reasons for these contrasting results is the fact that CMSY and the official stock assessment carried out in SS3 are very difficult to compare (Bouch et al., 2020). The main source of information for the CMSY model consists in the catch amount and the survey index, whereas SS3 also considers the population's structure of the target stock, as well as different biological features. In addition, SS3 includes a wide number of parameters related to catchability and other aspects (Methot and Wetzel, 2013). However, if SS3 is considered the best model for assessing the hake stock in Adriatic waters (FAO, 2020a), the CMSY model proves valid for the aim of this study, and they substantially agree on a significant reduction in fishing mortality in the last year. For common sole, this study and the official benchmark assessment substantially agree on a significant reduction in fishing mortality in the last year: the state of the Adriatic stock in 2019 was one of overfishing with low biomass, while the result for 2020 reflects the sharp drop in fishing mortality with a value below the reference point in the last year. The stock status of spottail mantis shrimp in GSA 17 in the GFCM context showed in 2019 and 2020 the same trend resulting from the analysis conducted through the CMSY model: a moderate reduction in fishing mortality mirrored by an increase in biomass. The status of spottail mantis shrimp slightly differs only in the final classification to the reference point: indeed, it was in the yellow area of the Kobe plot in 2019 and 2020 GFCM results, while it moved from the red to the yellow area in the same CMSY outputs. For cuttlefish in GSA 17, the configuration in the GFCM context was revised from 2019 to 2020 by changing crucial prior information, compromising any possible comparison with the results of this work. The parameters of the most recent assessment were adopted here. Biomass trajectory decreased from 1980 to 2010 when it dropped below 0.5 B/B_{MSY}. In recent years, the poor biomass status had some minor oscillation but remained quite stable, with a slight decrease from 2019 to 2020. Fishing mortality extensively fluctuated along the time series, and it is generally declining, but the model having 2020 as the reference year estimated an increase compared to 2019. The deepwater pink shrimp official assessments show a biomass trajectory increase from 2012, which stabilizes in 2019 and 2020. F trajectories, albeit with some minor oscillation, remained barely constant from 2012 to 2019, indicating an overfishing status for 2019 and 2020, with a slight increase in fishing mortality in the last year. The CMSY model also captured the large biomass increase from 2012, although it estimated a slight decrease in recent years. The F trajectories of the CMSY model were in a flux but stabilised in recent years, indicating a moderate increase of the F from 2019 to 2020 in line with the official assessment. For red mullet, the available official assessments describe an upward biomass trajectory from 2010, with a sudden increase in 2018. F trajectories are mirroring the biomass and were halved from 2018 to 2020, switching from an overfishing status for 2019 to a condition of F in line with the reference point for 2020. In the present paper, the CMSY estimated a biomass trajectory similar to the official assessment but much smoothed, with a more constant increase from 2010 to 2020. The F trajectories of the CMSY model are in line with the official assessment, below the reference point in both years. Nevertheless, in both cases, a reduction in fishing mortality from 2019 to 2020 is estimated, with a marked drop in the official assessment. Comparing the stock assessment results of demersal species to the fishing effort dynamics of bottom trawlers, it is possible to recognize a slightly different stock status between the main target species of OTB and TBB, with the target species of the second gear (common sole, spottail mantis shrimp) experiencing high benefit from the extra fishing activity reduction imposed by the pandemic restrictions. For the main target of the OTB fleet (hake, deepwater shrimp), just a minor decrease in fishing pressure was discerned. As regards hake, the official stock assessments over recent years were describing an increasing biomass trend from 2017, when the Pomo pit FRA was established. The biomass increase slightly improved in 2020, suggesting that the effect of the COVID pandemic on hake stock was negligible when compared to the effectiveness of the FRA (Chiarini et al., 2022). Red mullet was an exception; it is a target of OTB, and a large fishing mortality decrease was observed. Indeed, fishing mortality for red mullet has been decreasing almost steadily during the period comprised between 2010 and 2020, probably due to a positive effect of the spatial management measures in place. In fact, in the Adriatic Sea coastal bottom trawl (within 6 nautical miles) is banned for a few weeks after the summer fishing ban, protecting red mullet recruits starting their seasonal offshore migration. However, red mullet results have to be interpreted with caution because of the important flaws in the input data used both in the official assessment and in the present paper. As was already noted in STECF (2021b) and as it was explored in depth during the tentative benchmark assessment carried out by GFCM in 2022 (FAO, 2022c), issues were detected in the discard data and the fishery-independent data. Discarded data suffered from a reporting incoherence, while fishery-independent biomass is estimated from a survey plagued by important violations of the protocol, which influenced the detection of the recruitment. Therefore, caution is required when interpreting the red mullet results. Common cuttlefish is a separate case, since it is targeted by a mix of gears (TBB, as well as passive gears and OTB). This stock shows a countertrend compared to the other species analysed and lies alone in the red quadrant of the Kobe plot. The decreasing biomass trend was not curbed, and the fishing pressure was even increased in 2020, despite official statistics depicting a significant fishing effort reduction for passive gears and TBB. Nevertheless, the alarming status of cuttlefish had already been noted well before the pandemic (FAO-GFCM, 2019a), and the most recent stock assessment states that the biomass is below 0.5 MSY—a threshold that indicates possible recruitment impairment (Froese et al., 2017).

Small pelagic species, anchovy and sardine, proved to be under overexploitation and overexploited for both the 2019 and 2020 stock assessments. However, the most recent assessment shows an important increase in biomass and a decrease in fishing mortality for the anchovy stock. The critical situation of these two stocks is also confirmed by the official stock assessment in which $F_{current}$ overpass F_{MSY} and $B_{current}$ are well below the respective reference points, B_{lim} and B_{pa}, with a more critical situation for sardines (FAO, 2021c; FAO, 2022b). When considering the biomass estimated by the MEDIAS acoustic survey, the COVID pandemic does not seem to have any direct effect on these stocks. However, the CMSY model depicts a more marked increase in anchovy biomass. Since anchovies are mainly targeted by Italian PTM (STECF, 2020a), the biomass increase can be partially explained by the fishing effort reduction. In the case of sardines, the results of the CMSY models depict a minor change from 2019 to 2020. Considering the negligible fishing effort reduction attributed to PS, the fleet mainly targeting this resource, there is no evidence that COVID-19 had a positive effect on sardine stock in the Adriatic Sea.

Conclusions

The COVID-19 impact on fishery resources was unexpected and therefore difficult to study. The pandemic can be considered a sort of benchmark helping us to better understand how the effort management alone can have limited success in rebuilding the stocks in the short term by simply limiting the fishing days.

Accounting for the fishing effort reduction already foreseen in the management plan, the COVID imposed extra-activity reduction not balanced over the studied *metièrs*. On a 1-year basis, only a few fleets were affected while for others no effective activity reduction was observed. Moreover, the comparison between fishing effort metrics indicates that fishing vessels may have balanced the reduced number of fishing days by remaining more time at sea, thus potentially limiting the effects of the observed fishing pressure reduction. The case of TBB was unique, indicating a pandemic-related reduction in effort evenly allocated over 2020, which was confirmed by all the fishing effort metrics. On the contrary, the decrease of around 12% in fishing days observed for OTB (in line with the management plan objectives) was translated into a very limited reduction in terms of fishing hours and an increase in hours at sea.

The trend in catches was generally on a downward trend from 2019 to 2020. In some cases, the catch reduction was due to diminished fishing activity, while in other cases catch reduction was linked to diminished stock productivity. Based on the CMSY model results, some examples could be identified. The main target stock of TBB, common sole, as well as spottail mantis shrimp, increased in terms of biomass and showed fishing mortality at the MSY level after many years of overfishing. That said, a recovering trend status for these stocks had already been documented in assessment reports (FAO-GFCM, 2019b; FAO, 2021b), probably due to the effective management actions underway in the area, such as the coastal trawling ban (up to 4 nm) for 8 weeks from 2006 and the temporary extension of this spatial restriction up to 6 nm for 10 weeks since 2012 (EC, 2006; Armelloni et al., 2021). Conversely, the slight reduction in fishing pressure seems to have had a limited effect on those stocks in poor biomass status. As an example, the catch trend for common cuttlefish has been steadily declining over the last 6 years. The common cuttlefish biomass trend has been well below half of the B_{MSY} reference point from 2010, and it did not show any response to fishing effort reductions already noticed in the 2010–2019 period. Catch drop for this species may therefore be likely explained by low stock productivity instead of a decreased fishing pressure.

Therefore, the COVID-19 effect can be considered a positive accelerator of a recovery process already underway only when the fishing activity was effectively reduced. When the stock was not increasing its biomass, a sporadic fishing effort reduction might have had a negligible effect. The case of cuttlefish falls within this category and may suggest that stock rebuilding needs *ad-hoc* actions (i.e., more attention on restoring alive at sea the egg masses left on fishing devices or actions aimed at increasing spawning stock substrates; Grati et al., 2018).

In conclusion, similarly to other areas (Coll et al., 2021; Pita et al., 2021), our study shows that the COVID-19 restrictions that occurred in 2020 resulted in a low recovery effect on the status of target stocks. The severe COVID-19 effects on effort and catches were limited to a short period (March-May 2020), and the main impacts were on the markets and the supply chain, affected by price instability for a longer period (Pititto et al., 2021). Nevertheless, the study outputs might help us understand some strengths and caveats of a management system based on effort control. In particular, it emerged that limiting the fishing days was a measure capable of being circumvented by increasing the duration of fishing trips and/or by increasing fishing efficiency (Palomares and Pauly, 2019), especially if the foreseen reduction is moderate. There is no drive to encourage more severe limitations to effort, which is likely to create conflicts between the fishery sector and the management system, but we stress the importance of evaluating alternative management measures. Echoing a recent work focusing on the revision of management measures (Fiorentino and Vitale, 2021), and considering also the conclusion from Cardinale et al. (2017) evidencing the ineffectiveness of the putative effort reductions to control fishing mortalities, we support the need to integrate input controls with stock- and fleet-specific measures, which are going to decrease fishing mortality toward MSY levels.

Data availability statement

Publicly available datasets were analyzed in this study. This data can be found here: https://github.com/CNRFisheries/COVID-19-effects-on-target-stocks-in-the-Adriatic-Sea.

Author contributions

GS conceived the original idea, collated data, and implemented the stock assessment models. SA provided information on official stock assessment. EA validated the data and provided information on official stock assessment. IC, AD, and IL provided information on the MEDIAS survey. FM and MS provided information on official stock assessment. SG performed the statistical analysis. GC provided information on the environmental parameters and supervised the experiment. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.920974/full#supplementary-material

References

Althouse, B. M., Wallace, B., Case, B., Scarpino, S. V., Berdhal, A., White, E. R., et al. (2020)The unintended consequences of inconsistent pandemic control policies. In: *medRxiv : the preprint server for health sciences* (Cold Spring Harbor Laboratory Press). (Accessed 11 April 2022).

Amoroso, R. O., Pitcher, C. R., Rijnsdorp, A. D., McConnaughey, R. A., Parma, A. M., Suuronen, P., et al. (2018). Bottom trawl fishing footprints on the world's continental shelves. *Proc. Natl. Acad. Sci.* 115, E10275–E10282. doi: 10.1073/pnas.1802379115

Anonymous (2017). *MEDITS-handbook* (MEDITS Working Group), 106. Available at: http://www.sibm.it/MEDITS 2011/principaledownload.htm.

Armelloni, E. N., Scanu, M., Masnadi, F., Coro, G., Angelini, S., and Scarcella, G. (2021). Data poor approach for the assessment of the main target species of rapido trawl fishery in Adriatic Sea. *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.552076

Bennett, N. J., Finkbeiner, E. M., Ban, N. C., Belhabib, D., Jupiter, S. D., Kittinger, J. N., et al. (2020). The COVID-19 pandemic, small-scale fisheries and coastal fishing communities. *Coast. Manage.* 48, 336–347. doi: 10.1080/08920753.2020.1766937

Ben Rais Lasram, F., Guilhaumon, F., Albouy, C., Somot, S., Thuiller, W., and Mouillot, D. (2010). The Mediterranean Sea as a 'cul-de-sac' for endemic fishes facing climate change. *Global Change Biol.* 16, 3233–3245. doi: 10.1111/j.1365-2486.2010.02224.x

Bouch, P., Minto, C., and Reid, D. G. (2020). Comparative performance of data-poor CMSY and data-moderate SPiCT stock assessment methods when applied to data-rich, real-world stocks. *ICES J. Mar. Sci.* 78.1:264–276. doi: 10.1093/icesjms/fsaa220/6032767

Campanelli, A., Grilli, F., Paschini, E., and Marini, M. (2011). The influence of an exceptional po river flood on the physical and chemical oceanographic properties of the Adriatic Sea. *Dynamics Atmospheres Oceans* 52, 284–297. doi: 10.1016/j.dynatmoce.2011.05.004

Cardinale, M., Osio, G. C., and Scarcella, G. (2017). Mediterranean Sea: A failure of the European fisheries management system. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00072/full

Chiarini, M., Guicciardi, S., Zacchetti, L., Domenichetti, F., Canduci, G., Angelini, S., et al. (2022). Looking for a simple assessment tool for a complex task: Short-term evaluation of changes in fisheries management measures in the Pomo/Jabuka pits area (Central Adriatic Sea). *Sustainability* 14, 7742. doi: 10.3390/su14137742

Colloca, F., Scarcella, G., and Libralato, S. (2017). Recent trends and impacts of fisheries exploitation on Mediterranean stocks and ecosystems. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00244

Coll, M., Ortega-Cerdà, M., and Mascarell-Rocher, Y. (2021). Ecological and economic effects of COVID-19 in marine fisheries from the northwestern Mediterranean Sea. *Biol. Conserv.* 255, 108997. doi: 10.1016/j.biocon.2021.108997

Coll, M., Santojanni, A., Palomera, I., and Arneri, E. (2009). Food-web changes in the Adriatic Sea over the last three decades. *Mar. Ecol. Prog. Ser.* 381, 17–37. doi: 10.3354/meps07944

Coro, G., Bove, P., Armelloni, E. N., Masnadi, F., Scanu, M., and Scarcella, G. (2022a). Filling gaps in trawl surveys at Sea through spatiotemporal and environmental modelling. *Front. Mar. Sci.* 1106. doi: 10.3389/fmars.2022.919339/abstract

Coro, G., Bove, P., and Ellenbroek, A. (2022b). Habitat distribution change of commercial species in the Adriatic Sea during the COVID-19 pandemic. *Ecol. Inf.* 69, 101675. doi: 10.1016/j.ecoinf.2022.101675

Coro, G., Tassetti, A. N., Armelloni, E. N., Pulcinella, J., Ferrà, C., Sprovieri, M., et al. (2022c). COVID-19 lockdowns reveal the resilience of Adriatic Sea fisheries to forced fishing effort reduction. *Sci. Rep.* 12, 1052. doi: 10.1038/s41598-022-05142-w

EC (2006). *Council regulation (EC) no 1967/2006 of 21 December 2006 concerning management measures for the sustainable exploitation of fishery resources in the Mediterranean Sea, amending regulation (EEC) no 2847/93 and repealing regulation (EC) no 1626/94.*

Efron, B., and Hastie, T. (2016) *Computer age statistical inference: Algorithms, evidence, and data science*, 1–475.

Efron, B., and Tibshirani, R. (1993) *A n introduction to the practice. cindy.informatik.uni-bremen.de*. Available at: http://cindy.informatik.uni-bremen.de/cosy/teaching/CM_2011/Eval3/pe_efron_93.pdf (Accessed 6 April 2022).

Eigaard, O. R., Bastardie, F., Hintzen, N. T., Buhl-Mortensen, L., Buhl-Mortensen, P., Catarino, R., et al. (2017). The footprint of bottom trawling in European waters: Distribution, intensity, and seabed integrity. *ICES J. Mar. Sci.* 74, 847–865. doi: 10.1093/icesjms/fsw194

EU (2020) *European Union fleet register*. Available at: https://webgate.ec.europa.eu/fleet-europa/search_en (Accessed 1 June 2020).

FAO (1999). *Report of the first session of the scientific advisory committee* (Rome: Rome, Italy), 52. 23-26 March 1999. FAO Fisheries Report No. 601.

FAO (2020a) *How is COVID-19 affecting the fisheries and aquaculture food systems* (Rome). Available at: https://www.fao.org/policy-support/tools-and-publications/resources-details/fr/c/1271431/ (Accessed 14 April 2022).

FAO (2020b). *The state of Mediterranean and black Sea fisheries 2020*. 172.

FAO (2021a) *Scientific advisory committee on fisheries (SAC) working group on stock assessment of demersal species (WGSAD), online, 18–23 January 2021 (western Mediterranean) online, 25–30 January 2021 (central and eastern Mediterranean and Adriatic Sea)*. Available at: https://www.fao.org/gfcm/technical-meetings/detail/ru/c/1412431/.

FAO (2021b). *Scientific advisory committee on fisheries (SAC) working group on stock assessment of demersal species (WGSAD) - benchmark session for the assessment of common sole in GSA 17 , online, 12-16 April 2021*. 70.

FAO (2021c) *Working group on stock assessment of small pelagic species (WGSASP)*. Available at: https://www.fao.org/gfcm/technical-meetings/detail/ru/c/1412431/.

FAO (2021d) *Scientific advisory committee on fisheries (SAC) fifth meeting of the subregional committee for the Adriatic Sea (SRC-AS) online, 20–23 April 2021*. Available at: https://www.fao.org/gfcm/technical-meetings/detail/en/c/1410712/.

FAO (2022a) *Scientific advisory committee on fisheries (SAC) working group on stock assessment of demersal species (WGSAD) (western Mediterranean) (central and eastern Mediterranean and Adriatic Sea), 2022*. Available at: https://www.fao.org/gfcm/technical-meetings/detail/en/c/1506058/.

FAO (2022b). *Scientific advisory committee on fisheries (SAC) working group on stock assessment of small pelagic species (WGSASP)* Vol. 11 (Session on the assessment of European anchovy and sardine in the Adriatic Sea Online.), 67. Available at: https://www.fao.org/gfcm/technical-meetings/detail/en/c/1542569/.

FAO (2022c) *Working group on stock assessment of demersal species (WGSAD) benchmark session for the assessment of red mullet in GSAs*. Available at: https://www.fao.org/gfcm/technical-meetings/detail/en/c/1507459/.

FAO-GFCM (2019a). *Scientific advisory committee on fisheries (SAC) working group on stock assessment of demersal species (WGSAD)*. 1–68.

FAO-GFCM (2019b). *Working group on stock assessment of demersal species (WGSAD) FAO headquarters, Rome, Italy, 19–24 November 2018 FINAL REPORT*. 1–68.

Fiorentino, F., and Vitale, S. (2021). How can we reduce the overexploitation of the Mediterranean resources? *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.674633

Fortibuoni, T., Aldighieri, F., Giovanardi, O., Pranovi, F., and Zucchetta, M. (2015). *Climate impact on Italian fisheries (Mediterranean sea). regional environmental change* Vol. 15 (Springer Verlag), 931–937.

Fortibuoni, T., Giovanardi, O., Pranovi, F., Raicevich, S., Solidoro, C., and Libralato, S. (2017). Analysis of long-term changes in a Mediterranean marine ecosystem based on fishery landings. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00033

Froese, R., Demirel, N., Coro, G., Kleisner, K. M., and Winker, H. (2017). Estimating fisheries reference points from catch and resilience. *Fish Fisheries* 18, 506–526. doi: 10.1111/faf.12190

Froese, R., and Pauly, D. (2019). *Fishbase* (World Wide Web electronic publication).

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/j.marpol.2018.04.018

Galdelli, A., Armelloni, E. N., Ferrà, C., Pulcinella, J., Scarcella, G., and Tassetti, A. N. (2021). *R4AIS: An r workflow to process AIS data for fishery* (Zenodo). doi: 10.5281/zenodo.4761890

Grati, F., Fabi, G., Scarcella, G., Guicciardi, S., Penna, P., Scanu, M., et al. (2018). Artificial spawning substrates and participatory research to foster cuttlefish stock recovery: A pilot study in the Adriatic Sea. *PloS One* 13, e0205877. doi: 10.1371/journal.pone.0205877

Grati, F., Scarcella, G., Polidori, P., Domenichetti, F., Bolognini, L., Gramolini, R., et al. (2013). Multi-annual investigation of the spatial distributions of juvenile and adult sole (Solea solea l.) in the Adriatic Sea (northern Mediterranean). *J. Sea Res.* 84, 122–132. doi: 10.1016/j.seares.2013.05.001

Hale, T., Angrist, N., Goldszmidt, R., Kira, B., Petherick, A., Phillips, T., et al. (2021). A global panel database of pandemic policies (Oxford COVID-19 government response tracker). *Nat. Hum. Behav.* 5, 529–538. doi: 10.1038/s41562-021-01079-8

Hall-Spencer, J., Froglia, C., Atkinson, R. J. A., and Moore, P. G. (1999). The impact of rapido trawling for scallops,Pecten jacobaeus(L.), on the benthos of the gulf of Venice. *ICES J. Mar. Sci.* 56, 111–124. doi: 10.1006/jmsc.1998.0424

Jardim, E., Millar, C. P., Mosqueira, I., Scott, F., Osio, G. C., Ferretti, M., et al. (2014). What if stock assessment is as simple as a linear model? the a4a initiative. *ICES J. Mar. Sci.* 72, 232–236. doi: 10.1093/icesjms/fsu050

Leonori, I., Tičina, V., Giannoulaki, M., Hattab, T., Iglesias, M., Bonanno, A., et al. (2021). History of hydroacoustic surveys of small pelagic fish species in the European Mediterranean Sea. *Mediterr. Mar. Sci.* 22, 751–768. doi: 10.12681/mms.26001

Love, D. C., Allison, E. H., Asche, F., Belton, B., Cottrell, R. S., Froehlich, H. E., et al. (2021). Emerging COVID-19 impacts, responses, and lessons for building resilience in the seafood system. *Global Food Secur.* 28, 100494. doi: 10.1016/j.gfs.2021.100494

Marini, M., Bombace, G., and Iacobone, G. (2017). *Il mare adriatico e le sue risorse*. Carlo Saladino Editore (Palermo, Italy), 267.

Methot, R. D., and Wetzel, C. R. (2013). Stock synthesis: A biological and statistical framework for fish stock assessment and fishery management. *Fisheries Res.* 142, 86–99. doi: 10.1016/j.fishres.2012.10.012

Palomares, M. L. D., Froese, R., Derrick, B., Meeuwig, J. J., Nöel, S. L., Tsui, G., et al. (2020). Fishery biomass trends of exploited fish populations in marine ecoregions, climatic zones and ocean basins. *Estuarine Coast. Shelf Sci.* 243, 106896. doi: 10.1016/j.ecss.2020.106896

Palomares, M., and Pauly, D. (2018). *SeaLifeBase. version (02/2018)*.

Palomares, M. L. D., and Pauly, D. (2019) *On the creeping increase of vessels' fishing power. ecology and society*. Available at: https://www.jstor.org/stable/26796991?casa_token=2VHjx1yYs9gAAAAA:I4CSmzHEIhCa1yRL_5cO71994yZ34Xao7e_KBkq5KJxo9Z98ehGh-latk2K–bUSSjeIzgOTO5oiT7U3ARkIvav8VHDm3GMEr8tXBxbIXquB2lbXq9SkZg (Accessed 4 May 2020).

Payne, M. R., and Hintzen, N. T. (2013). *FLSAM: FLSAM, an implementation of the state-space assessment model for FLR.*

Pedersen, M. W., and Berg, C. W. (2017). A stochastic surplus production model in continuous time. *Fish Fisheries* 18, 226–243. doi: 10.1111/faf.12174

Pita, P., Ainsworth, G. B., Alba, B., Anderson, A. B., Antelo, M., Alós, J., et al. (2021). First assessment of the impacts of the COVID-19 pandemic on global marine recreational fisheries. *Front. Mar. Sci.* 8, 735741. doi: 10.3389/fmars.2021.735741

Pititto, A., Rainone, D., Sannino, V., Chever, T., Herry, L., Parant, S., et al. (2021) *Research for PECH committee – impacts of the COVID-19 pandemic on EU fisheries and aquaculture, epartment for structural and cohesion policies, Brussels*. Available at: https://www.europarl.europa.eu/thinktank/en/document/IPOL_STU(2021)690880.

R Core Team (2022). *R: A language and environment for statistical computing* (Vienna, Austria: Version 4.1.3. R Foundation for Statistical Computing).

Russo, E., Anelli Monti, M., Toninato, G., Silvestri, C., Raffaetà, A., and Pranovi, F. (2021). Lockdown: How the COVID-19 pandemic affected the fishing activities in the Adriatic Sea (Central Mediterranean Sea). *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.685808

Russo, T., Catucci, E., Franceschini, S., Labanchi, L., Libralato, S., Sabatella, E. C., et al. (2022). Defend as you can, react quickly: The effects of the COVID-19 shock on a Large fishery of the Mediterranean Sea. *Front. Mar. Sci.* 9. doi: 10.3389/fmars.2022.824857

Scarcella, G., Grati, F., Raicevich, S., Russo, T., Gramolini, R., Scott, R. D., et al. (2014). Common sole in the northern and central Adriatic Sea: Spatial management scenarios to rebuild the stock. *J. Sea Res.* 89, 12–22. doi: 10.1016/j.seares.2014.02.002

Spedicato, M. T., Massutí, E., Mérigot, B., Tserpes, G., Jadaud, A., and Relini, G. (2019). The medits trawl survey specifications in an ecosystem approach to fishery management. *Scientia Marina* 83, 9–20. doi: 10.3989/scimar.04915.11X

STECF (2020a). *Scientific, technical and economic committee for fisheries (STECF) - the 2020 annual economic report on the EU fishing fleet (STECF 20-06)* (Luxembourg: Publications Office of the European Union), 302p. Available at: http://stecf.jrc.ec.europa.eu/documents/43805/581354/2013-09_STECF+13-15+-+AER+EU+Fleet+2013_JRC84745.pdf.

STECF (2020b). *Scientific, technical and economic committee for fisheries (STECF)- stock assessments: demersal stocks in the western Mediterranean Sea (STECF-20-09)* (Luxembourg: EUR 28359 EN, Publications Office of the European Union). Available at: https://stecf.jrc.ec.europa.eu/documents/43805/1446742/STECF+16-14+-+Methods+for+MED+stock+assessments_JRC102680.pdf.

STECF (2020c). *Scientific, technical and economic committee for fisheries (STECF) stock assessments in the Mediterranean Sea – Adriatic, Ionian and Aegean seas (STECF-20-15)* (Luxembourg: Publications Office of the European Union). Available at: https://stecf.jrc.ec.europa.eu/documents/43805/1446742/STECF+16-14+-+Methods+for+MED+stock+assessments_JRC102680.pdf.

STECF (2021a). *Scientific, technical and economic commi ttee for Fi sheries (STECF) – fisheries dependent -information – FDI (STECF-21-12)* (Luxembourg,2021: Publications Office of the European Union).

STECF (2021b). *Scientific, technical and economic committee for fisheries (STECF) – stock assessments in the Mediterranean Sea 2021 – (Adriatic and Ionian seas) (STECF-21-15)* (Luxembourg,2021: Publications Office of the European Union).

White, E. R., and Hébert-Dufresne, L. (2020). State-level variation of initial COVID-19 dynamics in the united states. *PloS One* 15, e0240648. doi: 10.1371/journal.pone.0240648

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Fisheries impact on the Sea of Marmara ecosystem structure and functioning during the last three decades

İsmet Saygu[1], Ekin Akoglu[2], Güzin Gül[3],
Dalida Bedikoğlu[3] and Nazli Demirel[3]*

[1]Fisheries Faculty, Cukurova University, Adana, Türkiye, [2]Institute of Marine Sciences, Middle East Technical University, Erdemli, Türkiye, [3]Institute of Marine Sciences and Management, Istanbul University, Istanbul, Türkiye

The Sea of Marmara (SoM), without doubt, is one of the most disturbed marine ecosystems of the Mediterranean basin. As a semi-enclosed and a recognized eutrophic basin, it has a unique natural characteristic by permanent stratification at ~25 m depth. The SoM ecosystem is under threat by multiple stressors from excessive nutrient enrichment, overfishing, invasive species, habitat loss to the climate change. Within this study, an assessment on the SoM ecosystem structure and functioning and its changes over time have been achieved. Hence, we firstly evaluate the SoM ecosystem over a period of three decades from the 1990s to the 2010s by using three mass-balance Ecopath models, secondly, we delineate the prevailing ecosystem structure and functioning in each period, thirdly we determine how the fishing impact contributed to the changes in the SoM since the 1990s and finally establish a baseline for the management and future studies of the ecosystem. The comparison of the models' results for three consecutive time periods of the SoM showed that the energetic capacity of the ecosystem decreased significantly over the three decades while the decrease in cycling indices indicated that the SoM ecosystem became fragile to anthropogenic disturbances. According to ecosystem theory indices, the SoM is an immature ecosystem at a stage of autotrophic succession, with a very high total primary production, much more above unity that greatly exceeds total respiration. Our results highlighted that the SoM ecosystem was under bottom-up control exerted by mesozooplankton; however, anchovy was a bottleneck in energy transfers to higher trophic levels creating also a wasp-waist control in the food-web. The bycatch of demersal species with long life span, sharks and rays, brought them to the brink of going commercial extinction. Overall, the changes in the SoM ecosystem were due to the trophic cascades exerted by fisheries exploitation in addition to the changes in the productivity of the system. We conclude that the concomitant impacts of stressors on the SoM have been so diverse that an immediate action plan is required. Therefore, we propose a tentative outlook that will help in ecosystem monitoring and better management of the SoM.

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EDITED BY
Georgios Sylaios,
Democritus University of Thrace, Greece

REVIEWED BY
Donna Dimarchopoulou,
Dalhousie University, Canada
Sukran Yalcin Ozdilek,
Çanakkale Onsekiz Mart University, Türkiye
Mustafa Zengin,
Central Fisheries Research Institute (CFRI), Türkiye

*CORRESPONDENCE
Nazli Demirel
✉ ndemirel@istanbul.edu.tr

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Ecopath, ecological network analyses, food web, trophic flow, ecosystem approach to fisheries (EAF), data-poor region

Introduction

Overexploitation of marine fishery resources is known to degrade the productivity of fisheries and marine ecosystems around the world (Watson et al., 2013), with only a few exceptions of healthy fisheries as a result of good management practices (Froese et al., 2018). However, the precise level of exploitation is a matter of controversy among scientists (Worm et al., 2009; Froese et al., 2017). Existing fisheries statistics clearly indicate that the limits to the amount of fish that can be extracted from wild stocks have already been reached (Pauly and Zeller, 2016). Fishing directly affects the dynamics of stocks and its impacts can reach far beyond the targeted species through trophic cascades in the food web. Therefore, the delineation of ecosystem response to fisheries exploitation and other anthropogenic stressors; e.g., climate change, pollution, invasive species, is crucial to ensure and maintain a sustainable fishing activity and a healthy marine ecosystem. However, maintaining the existing volumes of catches and potentially increasing them will depend on our ability to rebuild depleted stocks (Froese et al., 2018; Palomares et al., 2020) and restore degraded ecosystem conditions.

The major changes in the size of a stock are attributed to the temporal dynamics of both predator and prey stocks, namely possible effects of density dependence on the development of stock biomass (Horbowy and Luzenczyk, 2016). The rebuilding of fish populations with low biomass to sustainable levels, aside from reducing fishing effort, is also dependent on their life history traits, food availability and trophic interactions amongst species (Hutchings and Reynolds, 2004; Audzijonyte and Kuparinen 2016). Classical fisheries science theory proposes that when a stock is first fished, its catches initially increase and then stabilize at a maximum level and can be sustained long time if fishing effort is optimal. This level of stability is termed maximum sustainable yield (MSY, Beverton and Holt, 1957; Tsikliras and Froese, 2019; Pauly and Froese, 2021). Although drastic reductions in fishing pressure may contribute to fish biomass recoveries (Hutchings, 2000; Neubauer et al., 2013) unless there are cultivation/depensation effects already in play (Walters and Kitchell, 2001), the historically high overexploitation levels may result in slower recovery rates than predicted by stock assessment models (Neubauer et al., 2013). However, years long stock assessment and fisheries management practices have shown that changes in fishing and fisheries regulations (McGarvey et al., 2015) as well as environmental variability and climate change (Alheit et al., 2014; Tu et al., 2018) affect stock dynamics.

A healthy safety margin must remain in fisheries management practices to account for inter- and intra-trophic interactions such as prey-predator relationships, and competition for resources (Horbowy and Luzeńczyk, 2017) and changes in the carrying capacity of the ecosystems. Hiddink et al. (2016) found that the prey composition and abundance were remarkably affected by exploitation of predators which is, further, very important to determine their impact on life history characteristics of predator species from food intake to condition and growth rates, to spawning cycle and recruitment potential. Further, the exploited stocks, their relationships with their preys and predators are intertwined with the dynamics of lower-trophic-level organisms, primary producers and zooplankton, and affect the whole food web. Finally, fisheries exploitation can have cascading impacts (Daskalov, 2002), indirectly affecting plankton dynamics (Lynam et al., 2017) and the carrying capacity of the ecosystem. Uncontrolled fishing, combined with other human-induced effects, namely climate change and pollution, causes the resilience of marine ecosystems to weaken, their health to deteriorate and their changes become irreversible (Halpern et al., 2015). Delineating the impacts of anthropogenic activities such as fishing on the whole food web may provide knowledge for ecosystem approach to fisheries (EAF) management. Therefore, ecosystem models are useful tools to provide knowledge on the changes in structure and functioning of the aquatic ecosystems.

The Sea of Marmara (SoM) is a nearly enclosed basin and recognized as a biological corridor that connects the Mediterranean and the Black Seas *via* Bosphorus (Istanbul) and Dardanelles (Canakkale) Straits. The SoM is surrounded by the Marmara geographical region which is the most populated and most urbanized region in Turkey including Metropolitan city of Istanbul with a population of nearly 15.5 million, located in the northeast, and Gulf of İzmit in the east with the center of industrial activities. Over the last 30 years, the marine ecosystem of the SoM was changed by human induced pressures and it lost its large pelagic and demersal predator species (Demirel et al., 2023). In the 2000s, carnivorous pelagic species seriously decreased, 22 marine species with high trophic levels including swordfish (*Xiphias gladius*), leerfish (*Lichia amia*), angelshark (*Squatina squatina*) and picked dogfish (*Squalus acanthias*) have become commercially extinct, and biodiversity has evolved towards the dominance of planktivorous fish species (Ulman et al., 2020). Unhealthy conditions such as harmful phytoplankton blooms (Tas et al., 2020), population increase of gelatinous organisms (İşinibilir and Yılmaz, 2016), and the effect of climate change *via* increasing sea surface temperatures (Demirel et al., 2023) have been observed since the late 1990s. Being a continuously degraded ecosystem over the last thirty years, the SoM is in severe need of understanding the changes in the structure and functioning of its ecosystem under these multiple stressors for the development of EAF as well.

Here, we have developed for the first time a food-web model of the SoM by investigating the changes since the 1990s using trophodynamic mass-balance models set up to represent three decades separately; the 1990s, the 2000s and the 2010s. While we set up three models for each decade, we considered a series of changes reported in the SoM (Demirel et al., 2023); i) the 1990s are characterized with fast development of urbanization and industrialization especially in the Istanbul and surrounding area, ii) the 2000s are mainly dedicated to overfishing and serious eutrophication and pollution including increase in the frequency of harmful algal blooms, and iii) the 2010s are dedicated to public awareness and starting of pollution monitoring activities, a buyback programme in fisheries which lowered active number of fishing boats. In line with the different characteristics of the three decades, we first aimed to understand the state of the SoM ecosystem and then to compare outputs of the three models. Therefore, the objectives of our study were; i) describing the structure and functioning of the SoM ecosystem, ii) understanding the changes exerted by fisheries exploitation on the SoM food web, iii) determining the shifts in the structuring role of the organisms on the food web, and iv) delineating the energy flows between the trophic levels to develop advice for increasing the ecosystem's well-being for ensuring sustainable use of the SoM's living resources.

Material and methods

Study area and its fisheries

The SoM, a semi-enclosed basin with an area of 11,500 km^2 (Figure 1), has a permanently stratified water column where upper low saline waters of the Black Sea enter from the Bosphorus Strait (upper ~20 m layer), while the Mediterranean Sea originated dense waters entering from Dardanelles Strait form the lower layer (> ~40 m) (Beşiktepe et al., 1994). It is recognized eutrophic and has an average primary productivity about 100 gC m^{-2} y^{-1} (Ergin et al., 1993). Due to strong stratification, the primary production is limited to the upper 25-30 m, where euphotic zone meets the pycnocline, and in addition to the Black Sea inflow and effluents, this high primary productivity significantly limits the light penetration (Ediger and Yilmaz, 1996; Altıok et al., 2014).

The total fish production of the SoM increased by 39% in the 1980s compared to the 1970s (Supplementary Figure A1) due to the development of the fishing fleet by subsidies provided to the fishers and the increased demand by the establishment of fish oil factories during high industrialization era (Zengin et al., 2017). The annual total catch in the SoM dropped from approximately 45,000 tons during the 1980s to just over 30,000 tons in the 2010s (FAO, 2022). The fisheries characteristic is mainly coastal for benthic/demersal species by beam trawling and rest is pelagic fishery by gillnets and purse-seines during seasonal migrations of the target species (Yıldız et al., 2020). Although an average of 64 fish and 28 invertebrate species are subject to fisheries, up to 79% of the total catch consisted of anchovy, horse mackerel, and deep water rose shrimp and 10% percent of sardine, bluefish, bonito, whiting, and hake in 2020. The catches of medium pelagic and demersal fish dramatically decreased from a maximum of 20,000 tons in the 2000s to approximately 10,000 tons in the second half of the 2010s. Overall, the decrease in total catch has not greatly fluctuated since 2008 because the fishery in the SoM depends on small pelagics, mainly anchovy (Demirel et al., 2020). Therefore, ecological groups contributing to fisheries catches are mainly small pelagic fish, followed by benthic invertebrates, and small amount of medium pelagic and demersal fish (Supplementary Figure A1).

The Ecopath modelling approach

Three mass-balanced food-web models representing the average state of the SoM for 1993-97 (1990s), 2003-07 (2000s), and 2013-17 (2010s) were developed using Ecopath with Ecosim (EwE) version 6.6.7 (Christensen et al., 2005). In Ecopath modelling, two master equations constitute the energy and mass balances (Christensen and Walters, 2004):

Consumption = production + respiration + unassimilated food

Production = catches + predation mortality + net migration + biomass accumulation + other mortality

The latter equation can be re-expressed as:

$$P_i = Y_i + B_i \cdot M2_i + E_i + BA_i + P_i \cdot (1 - EE_i)$$

where, for group i, P_i is the total production, Y_i is the total catch rate, B_i is the biomass, $M2_i$ is the total predation rate per year; E_i is the net migration rate per year (emigration - immigration); BA_i is the biomass accumulation rate per year and EE_i is the ecotrophic efficiency and $M0_i=P_i*(1-EE_i)$ is the annual 'other mortality' rate due to old age, diseases and starvation. Biomass (B), Production/Biomass (P/B),

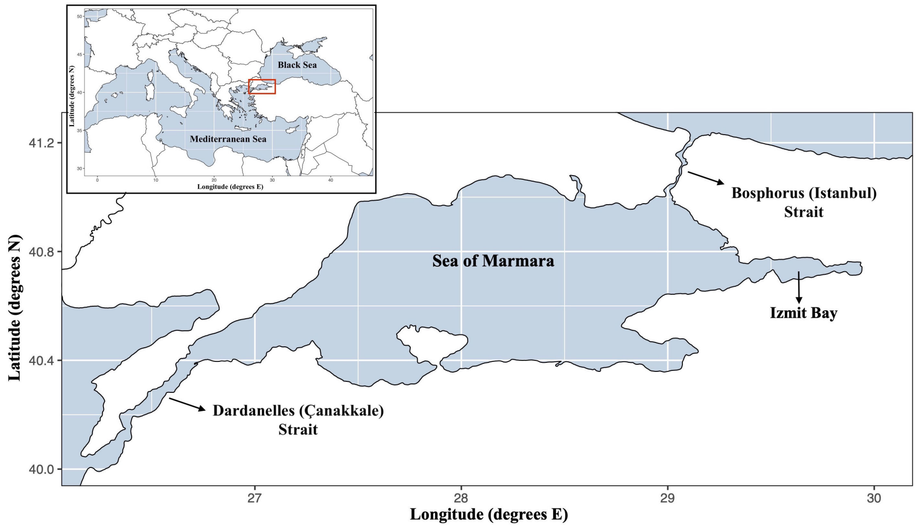

FIGURE 1
Modelling area: Sea of Marmara. The model area is limited by excluding the two strait systems of the Sea of Marmara.

Consumption/Biomass (Q/B) or Production/Consumption (P/Q), and Ecotrophic Efficiency (EE) are the four basic input parameters and three of them need to be provided and then the remaining parameter can be estimated by the model. If gross growth efficiency; i.e., Production/Consumption (P/Q) ratio, is entered, then Q/B or P/B ratio can be omitted and is estimated by Ecopath. In addition, a matrix of relative diet compositions by weight is required as input.

The food web topology (number of compartments, links between compartments, etc.) and methodology of construction for the Ecopath models were identical in all three models allowing their outputs to be directly compared (Heymans et al., 2014; Heymans et al., 2016). All models were balanced using the same diet matrix to ensure similar topology (Supplementary Table A5), with the differences being the biomasses and the catches estimated for the different periods.

We followed best practice guidelines presented by Heymans et al. (2016) to balance the models. Pre-balance (PREBAL) diagnostics (Supplementary Figures A2–A4) were used to check the slopes of B, P/B and Q/B across trophic levels before the model was balanced (Link, 2010; Heymans et al., 2016). Further, the pedigree routine, which assigns a different uncertainty range to each data used to parameterize an Ecopath model based on its source; e.g., from surveys, published literature, guesstimates or similar species/ecosystems, was used to calculate the pedigree index to assess the quality of the input data used in the models.

Default procedures were used to ensure mass-balance for the three models. The models were considered balanced when ecotrophic efficiencies were smaller than unity (EE<1), gross growth efficiencies (P/Q ratios) ranged between 0.1 and 0.3 as per the 2nd law of thermodynamics, respiration to assimilation (R/A) and production to respiration (P/R) ratios were lower than unity, and respiration to biomass (R/B) ratios were in the range of 1-10 year^{-1} for fish groups (Christensen and Walters, 2004; Darwall et al., 2010; Heymans et al., 2016).

Input parameters and functional groups

The Sea of Marmara was represented with a total of seventeen functional groups (FGs) and species, from primary producers to top predators including two planktonic groups, one invertebrate group, one invertebrate and eleven fish species, and one detritus group as well as dolphins (Table 1). We defined functional groups mainly based on their commercial importance as well as considering trophic diversity of the SoM ecosystem. Specifically, modeled main target species were anchovy (*Engraulis encrasicolus*), sardine (*Sardina pilchardus*), Mediterranean horse mackerel (*Trachurus mediterraneus*), Atlantic bonito (*Sarda sarda*), bluefish (*Pomatomus saltatrix*), European hake (*Merluccius merluccius*), whiting (*Merlangius merlangus*), turbot (*Scophthalmus maximus*), mullets (*Mullus* spp.) thornback ray (*Raja clavata*), picked dogfish (*Squalus acanthias*), and deep water rose shrimp (*Parapenaeus longirostris*). In addition, three fishing fleets, purse seiners, beam trawlers and gillnetters were included in the model. Since there was no regional statistical data describing species-specific catch by fishing fleet, purse seiners, beam trawlers and gillnetters are considered majorly responsible for fish removals based on the questionnaire survey that was carried out amongst local fishermen by Akyol and Perçin (2006) and field study on by-catch amounts by fishing gear in the Marmara Sea by Yazıcı et al. (2006). Purse seiners were the main fishing fleet catching

TABLE 1 Inputs (bold) and outputs of the Sea of Marmara (SoM) models by functional groups: Trophic level (TL); biomass (B, t km^{-2}); production/biomass (P/B, y^{-1}); consumption/biomass (Q/B, y^{-1}); ecotrophic efficiency (EE); production/consumption (P/Q); exploitation rate (E=F/Z); SoM 1990s model (90s); SoM 2000s model (00s); SoM 2010s model (10s).

#	Functional groups	TL	B			P/B			Q/B			EE			P/Q	E		
		All	90s	00s	10s	90s	00s	10s	90s	00s	10s	90s	00s	10s	All	90s	00s	10s
1	Phytoplankton	1.00	**22.23**	**19.38**	**17.63**	**132.63**	**113.77**	**114.57**				0.14	0.18	0.12				
2	Mesozooplankton	2.05	4.68	4.28	2.66	**38.29**	**38.29**	**38.29**	127.6	127.6	127.6	**0.90**	**0.90**	**0.90**	**0.30**			
3	Benthic invertebrates	2.11	7.00	8.30	4.29	**2.00**	**2.00**	**2.00**	**10.00**	**10.00**	**10.00**	**0.90**	**0.90**	**0.90**	0.20			
4	Deep water rose shrimp	3.02	**0.29**	**0.44**	**0.31**	**1.28**	**1.36**	**1.24**	5.12	5.43	4.95	0.80	0.72	0.54	**0.25**	0.35	0.42	0.32
5	European anchovy	3.00	**10.02**	**8.39**	**6.74**	**1.30**	**1.40**	**1.15**	**9.05**	9.72	7.98	0.98	0.97	0.76	0.14	0.09	0.14	0.17
6	European pilchard	3.02	**3.06**	**2.14**	**1.47**	**0.99**	**1.10**	**1.06**	**9.20**	10.28	9.87	0.87	0.98	0.86	0.11	0.15	0.21	0.32
7	Horse mackerel	3.09	**2.98**	**2.84**	**1.50**	**0.95**	**1.20**	**1.00**	**7.00**	8.82	7.35	0.98	0.93	0.95	0.14	0.11	0.29	0.24
8	Mullets	3.13	**0.32**	**0.23**	**0.16**	**0.54**	**0.70**	**0.59**	**4.00**	5.17	4.32	0.66	0.65	0.47	0.14	0.29	0.26	0.18
9	Turbot	4.09	**0.06**	**0.04**	**0.02**	**0.35**	**0.58**	**0.52**	**2.86**	4.83	4.31	0.24	0.45	0.13	0.12	0.24	0.45	0.13
10	Whiting	4.00	**0.62**	**0.41**	**0.17**	**0.50**	**0.90**	**0.70**	**2.00**	3.60	2.80	1.00	0.82	1.00	0.25	0.35	0.28	0.29
11	European hake	4.04	**2.00**	**0.50**	**0.28**	**0.55**	**1.17**	**0.65**	**3.50**	7.52	4.18	0.69	0.91	0.60	0.16	0.26	0.45	0.06
12	Bluefish	4.00	**1.60**	**1.14**	**0.58**	**0.55**	**0.82**	**0.89**	**5.00**	7.44	8.05	0.53	0.78	0.60	0.11	0.22	0.47	0.27
13	Atlantic bonito	4.02	**1.03**	**0.80**	**0.39**	**1.17**	**1.30**	**1.50**	**5.00**	5.54	6.37	0.21	0.25	0.21	0.23	0.20	0.23	0.17
14	Thornback ray	3.72	**0.18**	**0.17**	**0.08**	**0.31**	**0.44**	**0.39**	**2.00**	2.78	2.48	0.12	0.35	0.29	0.16	0.12	0.35	0.29
15	Picked dogfish	4.00	**0.12**	**0.08**	**0.03**	**0.28**	**0.45**	**0.47**	**1.80**	2.90	3.06	0.21	0.30	0.14	0.16	0.21	0.30	0.14
16	Dolphins	4.18	**0.04**	**0.04**	**0.04**	0.12	0.12	0.12	**12.05**	**12.05**	**12.05**	0.00	0.00	0.00	**0.01**			
17	Detritus	1.00	**152.52**	**115.57**	**106.31**							0.08	0.11	0.07				

European anchovy and pilchard. Both purse seiners and gillnetters targeted horse mackerel, bluefish and Atlantic bonito. Beam trawlers, on the other hand, was responsible for catching mainly deep water rose shrimp, and by-catching whiting, European hake, mullets and thornback rays. European hake, turbot, mullets, thornback ray and picked dogfish were largely caught by gillnets.

Input parameters for the species and functional groups were mainly obtained from published literature and unpublished information in the study area such as national reports and dissertations. For fished groups, biomass (B, tonnes) and fishing mortality (F, y^{-1}) values were obtained from the only stock assessment study based on catch-based analysis in the SoM (Demirel et al., 2020). Many countries such as Turkey lack detailed long-term data sets for fish stocks which are needed for the evaluation of proper stock assessment practice, and SoM stocks are defined in the "data-poor stocks" category (FAO, 2022). Existing data are mostly limited to 2/3-year data sets which themselves are not planned properly to provide what is needed to understand trends easily. Demirel et al. (2020) provided the first stock assessment study to the commercial fish stocks of the SoM between the years 1967-2020 through implementation of the catch-based data-limited CMSY method (Froese et al., 2017). The annual landings (tonnes) of the species were extracted from the General Fisheries Commission for the Mediterranean (GFCM) database of United Nations Food and Agriculture Organization (FAO) for the study period (FAO, 2022). The P/B ratios were calculated for each model assuming that the P/B ratio is equivalent to the total mortality (Z, y^{-1}) under steady-state conditions (Allen, 1971) and Z is equal to the sum of fishing mortality (F, y^{-1}) and natural mortality (M, y^{-1}). Therefore, P/B ratios for fishes were calculated by adding fishing mortality values from the CMSY analysis to the estimated natural mortality values using empirical equation by Pauly (1980). The Q/B ratios were calculated following Palomares and Pauly (1998) for fish groups and for other groups details are provided in Supplementary Table A1. Production/Consumption (P/Q) ratios were obtained from the 1990s Ecopath model and were assumed not to be time-dependent, and therefore, used to estimate the Q/B ratios in the 2000s and 2010s models. Functional group-specific descriptions of the input data, including B, P/B, Q/B, EE, P/Q, and catch were given in the Supplementary Table A2. Diet composition data were extracted from published studies preferably from the SoM or adjacent areas, western Black Sea and/or northern Aegean Sea (Supplementary Table A3).

Ecological indices for ecosystem structure

The trophic flows in the food web were investigated using discrete trophic levels (TL) and integrated trophic flows by TL. The flows were demonstrated using Lindeman spine graphs (Lindeman, 1942; Ulanowicz, 1986).

The food web characteristics were investigated using synthetic ecological indicators. We classified the indicators under four categories, ecosystem theory indicators, fishery indicators, cycling indicators, and information indicators. Ecosystem theory indicators describe ecosystem structure and characteristics. Total system throughput (TST) is the sum of all flows in the food web and composed of total consumption, total export, total respiration, and total flows into detritus (Finn, 1976). In our models, total export only included total catch. Transfer efficiency (TE) is the efficiency of energy flow that passed from a trophic level to the next level (Lindeman, 1942). Its geometric mean for TLII – TLIV is the mean transfer efficiency (Christensen et al., 2005). In addition, sum of all production, calculated total net primary production, net system production, total biomass, total primary production/ total respiration, total primary production/total biomass and total biomass/total throughput, mean trophic level of the community (mTLco), system omnivory index (SOI) that shows the breadth of feeding interactions in the food web (Christensen et al., 2005), and Shannon diversity index (Clarke and Warwick, 2001) were used.

Fishery indices are related to the exploitation status of the ecosystem and included total catch, gross efficiency (catch/net p.p.), the exploitation rates (E) of the functional groups (E=F/Z), the relative primary production required ($\%PPR_c$) to sustain the catches (Christensen and Pauly, 1993; Pauly and Christensen, 1995), cumulative exploitation rate (CumE) (Agnetta et al., 2019), and mean TL of the catch (mTLc) (Pauly et al., 1998). Further, ratios of predatory and demersal fishes to forage fish species, which are expected to decrease in time with exploitation, were calculated to investigate the fishing down the food web effect (Shannon et al., 2009).

The throughput cycled, predatory cycling index (PCI, % throughput excluding detritus), Finn's cycling index (FCI), and Finn's mean path length were calculated to investigate the degree and efficiency of material and energy cycling in the food web. FCI is the recycled part of the TST in an ecosystem and Finn's mean path length (PL) is the average number of groups through which a unit of flow travels. FCI and PL are desired to have higher values in a healthy ecosystem. Further, FCI and Finn's mean path length are related to the recovery time of an ecosystem after perturbations (Finn, 1976).

Ascendency (A), development capacity (C, upper limit of ascendency) and overhead (resilience) (O=C-A) were calculated as information indices. Ascendency shows the organizational status of the network of energy flows in food web. Highly organized food webs would have high ascendency. Ascendency is also linked with resilience of the ecosystem, the ability to withstand perturbations/stress (Ulanowicz, 2004; Heymans and Tomczak, 2016). The sum of ascendency and resilience is the development capacity of an ecosystem. Ascendency can be defined as the ability of a system to recover from stress. In a healthy ecosystem, a balanced degree of ascendency and resilience is desired because a healthy ecosystem is the one that is resilient to stress and has the potential to recover quickly after stress (Costanza and Mageau, 1999).

Mixed trophic impact (MTI) analysis was performed to relatively quantify both direct and indirect impacts of functional groups, including fisheries, on other groups (Ulanowicz and Puccia, 1990). In addition, the index of keystoneness (Power et al., 1996) was used to define important functional groups in maintaining ecosystem structure and function. Keystone groups/species are groups/species that have structuring role in the ecosystem although they have relatively low biomasses (Libralato et al., 2006).

Results

The SoM model was defined with 17 FGs (Table 1) from primary producer and detritus to dolphins. Trophic levels (TL) ranged from 1.0 for the primary producers and detritus group to 4.18 for the dolphins and the fishes with TLs above 4 were turbot (4.09), hake (4.04), bluefish (4.00) and Atlantic bonito (4.02). Among other species, anchovy had the lowest TL (3.00) in relation to its diet that consisted of phytoplankton and zooplankton. The deep-water rose shrimp had the same trophic level with sardine (3.02) (Table 1).

The pedigree index calculated by the model was 0.41 (Table 2). Our functional groups are mostly around the slope line in the PREBAL plots of B, P/B and Q/B and therefore indicated integrity; however, B, P/B and Q/B values of mullets, deep water rose shrimp, thornback ray and picked dogfish may potentially be underestimated and biomass and P/B values of bluefish, Atlantic bonito and European hake and Q/B value of dolphins may potentially be overestimated (Supplementary Figures A2–A4).

Ecosystem description

The food web structure was characterized *via* flow diagrams demonstrating trophic positions and relations between functional groups (Figure 2).

The total system throughput (TST) was 6499 t km^{-2} y^{-1} in the 1990s model and consisted of 42% flows into detritus, 38% exports, 13% consumption, and 7% respiration. Although TST decreased to 4984 t km^{-2} y^{-1} in the 2000s and 4384 t km^{-2} y^{-1} in the 2010s, the proportions of the flows were similar as in the 1990s model (Table 2). From the 1990s to 2010s the relative ratio of system's ascendency increased from 51% to 54% while its overhead (resilience) decreased from 49% down to 46%. Total system biomass (excluding detritus) decreased from 56.2 t km^{-2} in the 1990s to 49.2 in the 2000s (13%) and 36.4 in the 2010s (35%). The mean trophic level of the community (> 2) was 3.27 in the 1990s, 3.20 in the 2000s, and 3.15 in the 2010s (Table 2). The mean TE of the system was highest in the 2000s with 12.6% and 11.3% in the 1990s and 10.6% in the 2010s. The relative throughput cycled considering the parts excluding and including detritus decreased more than 40% from the 1990s to the 2010s. PCI and FCI also decreased throughout the decades indicating a degradation in the material recycling in the ecosystem (Table 2).

The biomass of benthic invertebrates and deep-water rose shrimp increased in the 2000s before declining in the 2010s, the biomass of dolphins have remained constant, and the biomass of all other functional groups decreased over the decades (Table 1). The changes in the biomasses from the 1990s to the 2010s were remarkable in all other functional groups. The highest decrease was observed in hake with 75% from the 1990s to the 2000s and 86% from the 1990s to the 2010s. The decrease from the 2000s to the 2010s was 44% for hake. The cumulative biomass of small pelagic fishes (anchovy, pilchard and horse mackerel), demersal predator fishes (turbot, whiting and hake), medium pelagic predator fishes (bluefish and Atlantic bonito), and shark and skate (picked dogfish + thornback ray) decreased by 17%, 65%, 26% and 18% from the 1990s to the 2000s and 40%, 83%, 63% and 64% from the 1990s to the 2010s, respectively. These results highlighted that significant decreases occurred in the biomasses of the predator fishes. The biomass ratio of medium pelagic predator fishes to small pelagic (forage) fishes decreased from 16% in the 1990s to 10% in the 2010s. Further, the ratio of demersal predator fish to forage fish decreased from 17% to 5% in the 2010s.

The SoM food web was integrated into five integer trophic levels and the energy flows through the trophic levels are shown using Lindeman Spine (Figure 3). Excluding TL I, the highest TST (%), biomass, respiration and flow to detritus were estimated for TL II and TL III where energy was also transferred most efficiently. The grazing flows from primary producers to TL II decreased significantly from 423.1 t km^{-2} y^{-1} in the 1990s to 240.7 t km^{-2} y^{-1}in the 2010s and the primary productivity of the ecosystem decreased significantly from 2948 t km^{-2} y^{-1} in the 1990s to 2204 t km^{-2} y^{-1}and 2020 t km^{-2} y^{-1} in the 2000s and the 2010s, respectively. In addition, flow from detritus to TLII remarkably decreased from the 1990s to the 2010s. However, the proportion of the flows from primary producers (grazing food chain) to flows from detritus (detrital food chain) was around 66% in all three periods indicating the dominance of the grazing food chain in the SoM. Results showed that biomasses and the energy, respiratory and detrital flows corresponding to each TL decreased throughout the periods. The energy transfer efficiencies between trophic levels gradually decreased; however, did not change significantly from the 1990s to the 2010s. A slight decrease from 13% in the 1990s to 11% in the 2010s occurred in the transfer efficiencies of flows from TL III to TL IV. The highest total export by fisheries occurred in TL III and followed by TL IV in all models and the total export in the 2000s model were higher than the other models.

TABLE 2 Ecosystem and fishery indicators for the models.

Parameter	Values			Units
	1990s	2000s	2010s	
Ecosystem theory indices				
Sum of all consumption	832	781	474	t km^{-2} y^{-1}
Sum of all exports	2504	1787	1767	t km^{-2} y^{-1}
Sum of all respiratory flows	449	423	256	t km^{-2} y^{-1}
Sum of all flows into detritus	2713	1994	1888	t km^{-2} y^{-1}
Total system throughput	6499	4984	4384	t km^{-2} y^{-1}
Sum of all production	3164	2406	2143	t km^{-2} y^{-1}
Calculated total net primary production (TPp)	2948	2204	2020	t km^{-2} y^{-1}
Total primary production/total respiration	6.6	5.2	7.9	
Net system production	2499	1781	1764	t km^{-2} y^{-1}
Total primary production/total biomass	52.4	44.8	55.6	
Total biomass/total throughput	0.009	0.010	0.008	
Total biomass (excluding detritus)	56.2	49.2	36.4	t km^{-2}
TE from primary producer	11.6	13.0	10.8	%
TE from detritus	10.9	12.0	10.1	%
TE total	11.3	12.6	10.6	%
Mean TL of the community	1.30	1.31	1.23	
Mean TL of the community (excluding T=1)	2.87	2.73	2.76	
Mean TL of the community (> TL2)	3.27	3.20	3.15	
System omnivory index	0.19	0.19	0.20	
Shannon diversity index	1.90	1.83	1.64	
Pedigree index	0.41	0.41	0.41	
Fishery indices				
Total catch	3.0	4.5	2.6	t km^{-2} y^{-1}
CumE	2.6	3.8	2.6	
Mean TL of the catch	3.3	3.3	3.1	
Gross efficiency (catch/net p.p.)	0.001	0.002	0.001	
PPR to sustain the fishery/TPp	2.9	5.7	2.7	%
Cycling indices				
Throughput cycled (excluding detritus)	38	36	22	t km^{-2} y^{-1}
Predatory cycling index	3.5	3.6	3.5	%
Throughput cycled (including detritus)	109	108	62	t km^{-2} y^{-1}
Finn's cycling index	1.7	2.2	1.4	%
Finn's mean path length	2.2	2.3	2.2	

(Continued)

TABLE 2 Continued

Parameter	Values			Units
	1990s	2000s	2010s	
Information indices				
Ascendency (A)	51	47	54	%
Overhead (O)	49	53	46	%
Capacity (C)	15171	12750	9544	Flowbits
Relative redundancy (R/C)	33	35	31	%
Relative exports (E/C)	23	21	25	%
Flow diversity (C/T)	2.33	2.56	2.18	

Ecological roles of functional groups

Mesozooplankton and benthic invertebrates, small pelagic fishes, particularly anchovy, and top predator fishes, particularly hake and bluefish, had significant direct and indirect impacts throughout the food web (Figure 4). Mesozooplankton had the highest relative total impact and was the keystone species in the 1990s and was followed by anchovy both in terms of keystoneness and relative total impact (Figure 5). Benthic invertebrates came third in the 1990s in terms of keystoneness and relative total impact. However, the trophic dynamics started to change after this decade. In particular, relative total impact of mesozooplankton decreased from 1.0 in the initial decade to 0.93 in the 2000s and 0.9 in the 2010s (Figure 5) and anchovy had the highest relative total impact in the 2010s. However, in all periods mesozooplankton was the keystone group in the ecosystem and followed by anchovy. The keystoneness (and also relative total impact) of bluefish from the 1990s to the 2010s increased significantly from -0.515 to -0.385.

Fishing impacts

The total catch increased from 3 t km^{-2} y^{-1} in the 1990s to 4.5 t km^{-2} y^{-1} in the 2000s and decreased back to 2.6 in the 2010s and therefore the 2000s model produced the highest CumE (3.8), gross efficiency (0.002) and PPRc/TPp (%5.7) that is related to overfishing risk in an ecosystem (Table 2). The exploitation rates

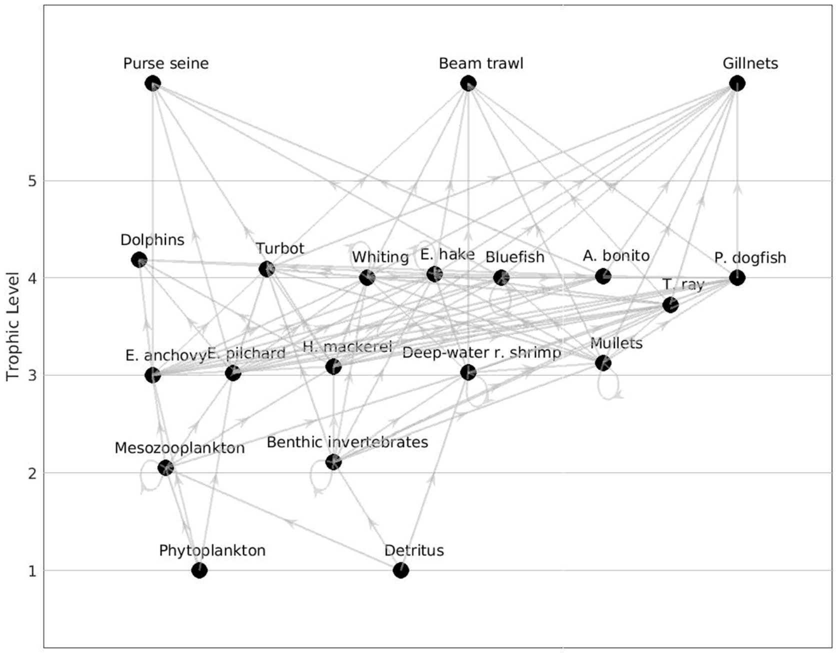

FIGURE 2
Flow diagram of the Sea of Marmara food web model representing the time period 1993 – 1997 (1990s).

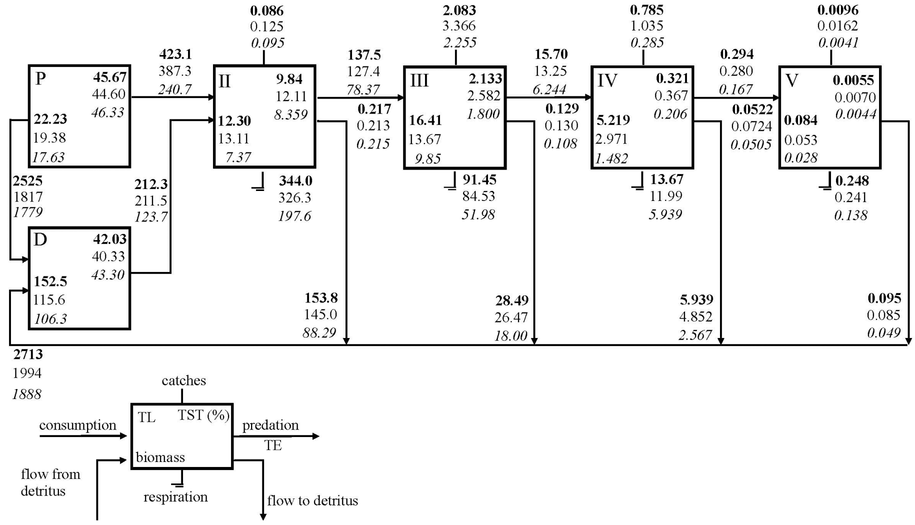

FIGURE 3
Lindeman spine showing trophic flows among integer trophic levels (TL). Primary producer (P); detritus (D); Total system throughput (TST); Transfer efficiency (TE); SoM 1990s model (**bold**); SoM 2000s model (normal); SoM 2010s model (*italic*).

of functional groups were also highest in the 2000s, excluding mullets and whiting which reached the highest values in the 1990s and anchovy and pilchard in 2010s (Table 1).

The gillnet fishery had positive mixed trophic impact on whiting in the 2000s (0.049) and 2010s (0.023) due to indirect effects outcompeting the direct negative effect of exploitation, i.e., exploiting whiting's predators, turbot, thornback ray and picked dogfish. Further, purse seine fishery had negative impact on anchovy in all periods, -0.039 in the 1990s, 0.068 in the 2000s and -0.106 in the 2010s, respectively. Gillnetters had strong negative impacts on bluefish, bonito, turbot, thornback ray, dogfish in all periods (Figure 4). Beam trawlers had negative impacts on thornback ray, picked dogfish and turbot and strong negative impact on deep water rose shrimp in all periods. Gillnetters had indirect positive impact on mullets due to their negative impacts on predators of the group.

Discussion

In this study mass-balance Ecopath models for three consecutive time periods were set up to delineate the ecosystem structure and functioning of the Sea of Marmara since the 1990s using statistical metrics and synthetic ecological indicators. Hence, our study presents the first comprehensive model description of its food web and its transition over three decades. The SoM is an internal sea between Black and the Mediterranean Seas, off the Istanbul metropolitan area with a very intense urbanization and industrialization, as well as with very high ecological and socio-economic importance since centuries. As a nearly enclosed sea with a permanent stratification due to its natural hydrography (Beşiktepe et al., 1994), our modelling study for the SoM ecosystem can be used to demonstrate localized climate change impacts and how stratification in the water body can limit the coupling between the benthic and pelagic environments by preventing nutrients and small particles from crossing these boundaries (Imberger and Patterson, 1989). Benthic-pelagic coupling is defined as the biological processes linking the pelagic environment and deep sediment bottoms below the euphotic zone (Hedberg, 2021). Hence, the permanently-stratified SoM can be an example for similar ecosystems in other parts of the Mediterranean Sea, where changes such as synergistic impacts of fishing and extended or stable stratification in water column (Li et al., 2020) are impending.

State of the SoM ecosystem

The ecosystem theory indices indicated that the SoM is an immature ecosystem at a stage of autotrophic succession, with a very high total primary production, much more above unity that greatly exceeds total respiration (Table 2). SoM has very high primary productivity similar to the Black Sea, because the Black Sea inputs supply most of the nutrients and organic matter *via* its upper layer, and also form a nutricline located at depths of 25-30 m (Ediger and Yilmaz 1996), coinciding with the permanent halocline (Beşiktepe et al., 1994). Very high flow to detritus from primary producers, low material cycling (FCI and PCI), and sharp decrease of TE from TL III to higher TLs showed that there was a high proportion of production which was not utilized within the system and ended up in the detritus

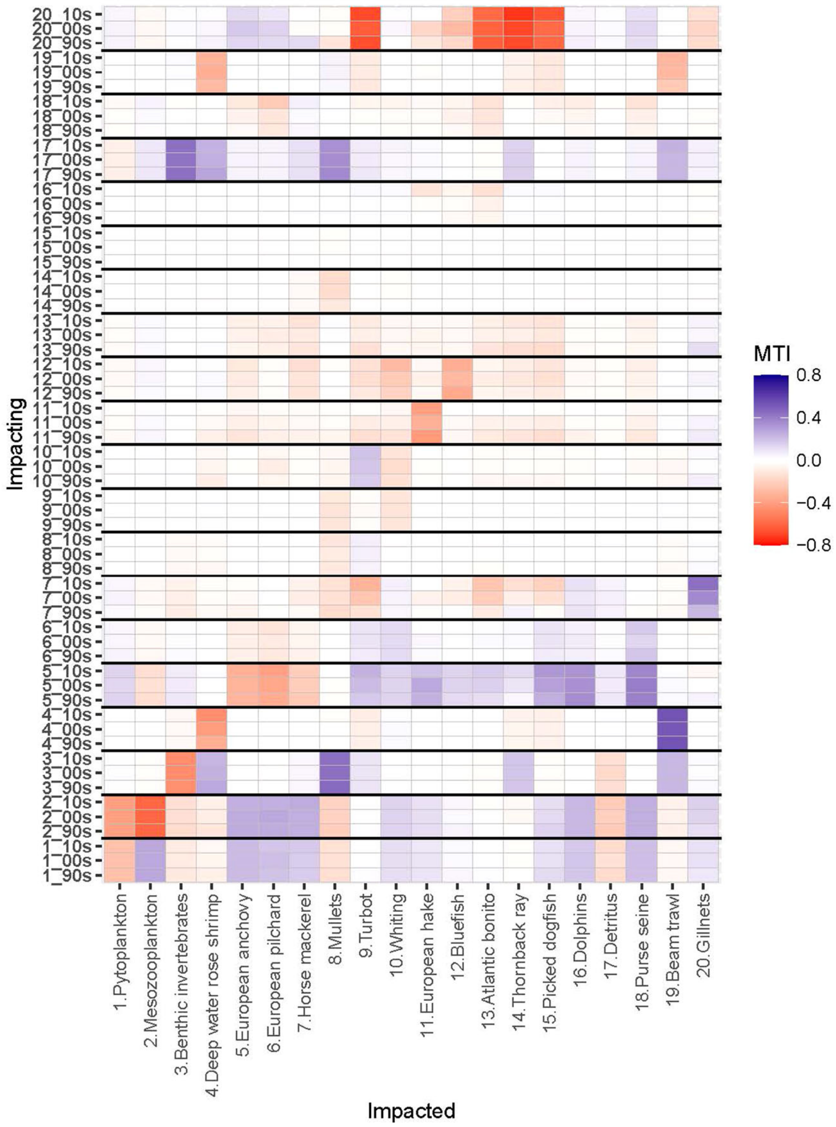

FIGURE 4
Mixed Trophic Impact (MTI) analysis of Sea of Marmara (SoM) food web for the periods of 1993-97 (1990s), 2003-07 (2000s) and 2013-17 (2010s). X-axis shows impacted groups (functional group numbers and names) and Y-axis shows impacting groups (functional group numbers and model periods). SoM 1990s model (90s); SoM 2000s model (00s); SoM 2010s model (10s). Red and blue colours indicate negative and positive impacts, respectively.

compartment. Thus, we concluded that the depth-limited primary productivity cannot reach upper trophic levels efficiently due to intense fishery exploitation of pelagic fishes by industrial fishery, and weak benthic-pelagic coupling due to strong water stratification. Overall, the SoM ecosystem has been dominated by the pelagic compartment, mainly in terms of plankton and one small pelagic fish, anchovy.

Although comparison of ecosystem indices by Ecopath models with different topologies across ecosystems is not recommended (Heymans et al., 2014; Heymans et al., 2016), considering similarities for FGs and aggregations of TLs, some indices that are not affected by the model topology can provide insightful information (Heymans et al., 2016). Therefore, because the SoM is a lesser known part of the Mediterranean basin, some contrasting peculiarities among ecosystems in the Black Sea (Akoglu et al., 2014) and the Mediterranean Sea are worth discussing (Papantoniou et al., 2021; Dimarchopoulou et al., 2022). The total biomass/total throughput ratio was similar to the Black Sea and other parts of the Mediterranean Sea (Akoglu et al., 2014; Papantoniou et al., 2021; Dimarchopoulou et al., 2022). In comparison to the Mediterranean and North Sea ecosystems, the SoM was the largest system after the North Sea in terms of flows (TST~ 4384-6499 t km^{-2} y^{-1}) with an omnivory index of 0.19-0.20 that is

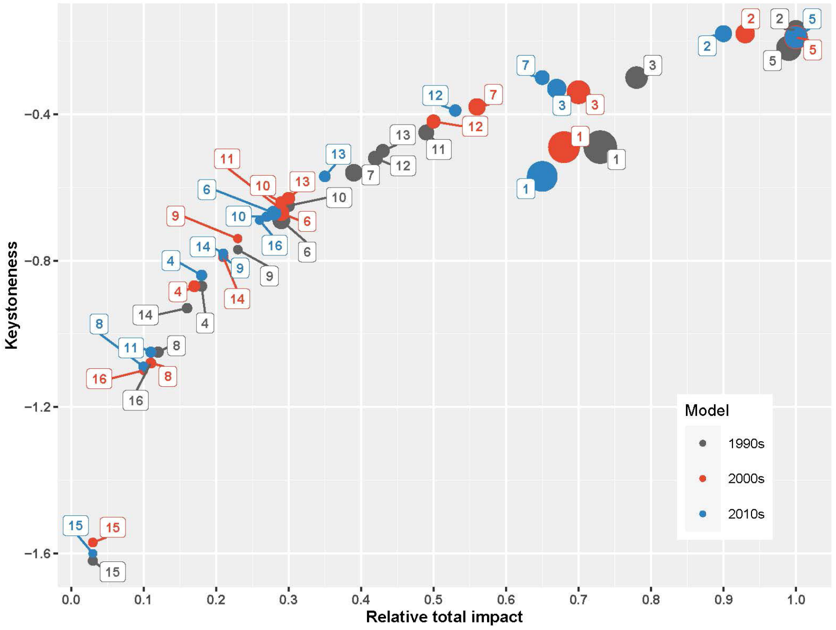

FIGURE 5
The keystoneness and the relative total impact of the functional groups in the SoM models. The size of the circles are proportional to the groups' biomasses. Functional groups correspond to: (1) Phytoplankton, (2) Mesozooplankton, (3) Benthic invertebrates, (4) Deep water rose shrimp, (5) European anchovy, (6) European pilchard, (7) Horse mackerel, (8) Mullets, (9) Turbot, (10) Whiting, (11) European hake, (12) Bluefish, (13) Atlantic bonito, (14) Thornback ray, (15) Picked dogfish, (16) Dolphins, and (17) Detritus.

comparable to the Mediterranean and North Seas (Mackinson 2014; Piroddi et al., 2015), indicating complex food web interactions. The SoM had a greatly higher total primary production/total respiration ratio (5.2-7.9) that is suggestive of autotrophic succession *sensu* Odum (1969), while its net system production was almost double (1764-2499 t km^{-2} y^{-1}) as a sign of its developing ecosystem.

Ecosystem structure and functioning within decades

The comparison among mass-balance Ecopath models for three consecutive time periods of the SoM showed that the energetic capacity (TST) of the ecosystem decreased significantly over the three decades and the decreasing cycling indices indicated an ecosystem continuously becoming fragile to anthropogenic disturbances. Our results highlighted that the size of the entire ecosystem in terms of energy flows and total biomass decreased since the 1990s.

The deterioration of the ecosystem as shown by the ecosystem theory indices has incessantly continued over decades. Under the impact of disturbance, ecosystems react with increased redundancy and flow diversity increases during such transitional periods (Ulanowicz, 1980). As shown by flow diversity (C/T) and relative redundancy (R/C) (Table 2), the 2000s indicated a stress on the ecosystem and could be characterized as a transition period. After this period, in the 2010s, these ratios decreased. Further, the relative ascendency of the system increased and the relative resilience decreased in the 2010s and could be considered an indicator of the fragility of the SoM against stressors. As ecosystems develop into maturity, they tend to increase biomass supported per unit of energy (Christensen, 1995). However, the SoM ecosystem degraded in terms of total biomass supported by its throughput as a further indication of stressed or disturbed conditions in the ecosystem (total biomass/total throughput, Table 2). The decrease in system's throughput by ~33% from the 1990s to the 2010s suggested that some trophodynamic links in the food web did not perform optimally, indicating a deficiency in the energy transfers in the food web. This deficiency was caused by the selective extraction by fisheries, and hence, reduced material/energy cycling (Finn, 1980).

The low throughput cycled and FCI as well as a low overhead in the 2010s model suggested that ecosystem maturity, resilience and stability have been eroded over time (Finn, 1976; Ulanowicz, 1986). A considerable decrease in the grazing flows from TL I to TL II may indicate a decreasing carrying capacity of the ecosystem due to decreased primary productivity that was possibly related to the increasing sea surface temperatures (Akoğlu, 2021). Further, the decreased energy transfers from detritus to TL II indicated a degradation in the ecosystem's maturity *sensu* Odum (1969), as in mature ecosystem there is greater cycling through detritus and the detrital compartments play an important role in nutrient regeneration (Ulanowicz, 1980). The transfer efficiency from TL III to TL IV decreased significantly in the 2010s. This inefficient transfer was mostly attributable to the exploitation of small pelagics and top predators *via* fisheries, thus top predators' biomass was significantly reduced.

Changes in ecological roles of functional groups

The change in mixed trophic interactions and keystone species in the ecosystem represented a strong interplay between bottom-up and wasp-waist controls on the dynamics of the SoM food web. Particularly, mesozooplankton indicated the bottom-up, resource-driven nature of the food web dynamics (Heath et al., 2014); however, anchovy played a bottleneck role in the distribution of energy from lower trophic levels to higher ones by exerting a wasp-waist control on the food web (Bakun, 2006). This is in line with the keystoneness analysis, which showed that the SoM has been under a mixture of two major food web control dynamics, bottom-up and wasp-waist. In all three periods, mesozooplankton was the keystone group and had the highest relative total impact in the 1990s, followed by anchovy. In the 2000s and 2010s, anchovy had the highest relative total impact. Hence, bottom-up controls are dominant in the SoM food web; however, anchovy is a bottleneck in terms of energy transfers from lower trophic levels to higher trophic levels. A collapse of anchovy stocks may bring about a reverse cascading impact up to higher trophic level fish species that may lead to their commercial extinction; hence, collapse of their fisheries. A similar situation occurred in the Black Sea over the second half of the 20th century; however, with major differences. In the Black Sea, first the top-down control by predator species were removed due to the overexploitation of bluefish, bonito and Atlantic mackerel in the 1950s and the 1960s; then later with the onset of eutrophication in the 1970s, and in the 1980s, zooplankton and anchovy gained structuring roles in the Black Sea's food web by exerting bottom-up and wasp-waist controls, respectively (Akoglu et al., 2014). Following the collapse of anchovy stocks in the late 1980s, predatory species also suffered significant decreases in their stocks (Oguz et al., 2012) and a similar situation could be imminent for the SoM ecosystem.

The negative mixed trophic impacts of hake on turbot, thornback ray and picked dogfish in the 1990s and the 2000s were related to indirect effects, i.e. resource competition; however, these negative impacts decreased in the 2010s with the significantly decreased biomass of hake. Bluefish and Atlantic bonito had also negative impacts on the demersal fish species, turbot, whiting, hake, thornback ray and picked dogfish, due to their positive impacts on the gillnetter fleet that targeted these demersal species. Anchovy had an indirect negative mixed trophic impact on European pilchard due to competition. The negative impact of anchovy on horse mackerel, although being its prey, was indirect and due to the direct positive impacts of anchovy on the predators of horse mackerel, namely whiting, hake, thornback ray, picked dogfish, bluefish and Atlantic bonito and dolphins, that outcompeted its direct positive impact; i.e. as prey, of anchovy on horse mackerel. The negative impact of horse mackerel on bluefish and bonito were due to the positive impact of horse mackerel on dolphins, the predators of these two fishes. Dolphins had direct negative impacts on bluefish and bonito as predators.

Fishing impacts on the food web

The cumulative impacts of multiple factors on marine ecosystems have been described using ecosystem modelling (Bentley et al., 2017; Corrales et al., 2017; Salihoglu et al., 2017; Serpetti et al., 2017). It is also suggested that delineation of cumulative impacts can help to build recommendations in the future management options (Corrales et al., 2018). In our study, we identified 30 years of fishing impact that changed ecosystem structure and functioning. The highest fishing pressure was in the 2000s according to fisheries indices. The most significant impact of fisheries was the reduction of higher-trophic-level fish species from the ecosystem, initiating a fishing down the food web impact as indicated by the decreasing biomass ratios of predatory and demersal fish species to forage fish. Further, the increased fisheries exploitation over the decades, i.e., relative exports, decreased the cycling in the food web (E/C and FCI in Table 2).

Model results showed that fisheries' selective extraction of species in the food web decreased the fish biomass in the SoM significantly over three decades. The MTI analysis showed the negative impact of fisheries on the majority of exploited groups. Purse seiners and gillnetters negatively impacted hake and due to heavy exploitation of this species since the 1990s, its biomass decreased significantly. Purse seiners and gillnetters had strong negative impacts on the pelagic predator and demersal predator species, respectively. The degree of negative impacts of beam trawlers and gillnetters on thornback ray and picked dogfish were also alarming as these species have relatively long lifespan and low fecundity. Further, strong negative impact of fishing fleets on turbot, thornback ray and dogfish put the population dynamics of these species at risk and could have pushed their populations towards to the brink of commercial extinction. The eradication of these two top predators from the SoM would be inevitable if the current level of exploitation continues. A similar dire consequence could also be anticipated for turbot. High exploitation levels exerted by fisheries as shown by fishery indices and the decreasing ratios of pelagic and demersal predator fish biomasses to the forage fish biomass, and the decrease in the mean trophic level of the catch from 3.3 in the

1990s to 3.1 in the 2010s indicated a fishing down the food web effect (Pauly et al., 1998; Shannon et al., 2009) in the SoM.

In the Mediterranean and the Black Seas, large predatory fishes were the first to be subject to overfishing, followed by medium-pelagics, demersals, and finally small pelagics (Daskalov, 2008; Froese et al., 2018). The healthy state of small pelagic stocks is of vital importance for the recovery of their predators as the demersal whiting, turbot and rays need large food supplies to ensure population growth (Peck et al., 2021). Making sure that there is sufficient prey biomass left in the system, along with continuous reduction of fishing pressure, is needed in order to ensure the future survival of the predatory fishes in the SoM. Fisheries exploitation had significant responsibility on these changes; however, the effects of other factors that were not explicitly represented in our models such as impacts of pollution, increase in jellyfish biomass and number of new species and changes in primary production dynamics, as indicated by the decreasing primary productivity and phytoplankton biomass in our models, should not be ruled out and future modelling work should consider representing these aspects.

Model skill and data limitations

The Pedigree index calculated by the model pointed out that the SoM model was of medium quality and the inputs used were moderately satisfying (Morissette, 2007). Although most of the diet compositions data were collected at local level and; therefore, had a low degree of uncertainty, the major source of uncertainty in our model was associated with the modeled biomass and fishing mortality estimations for commercial fish groups using the CMSY algorithm (Froese et al., 2017). Since SoM is a data-poor region for fisheries, we best capitalized on results from CMSY as a catch-based stock assessment method to estimate time-series of biomass and fishing mortality values for commercial species. This approach is very helpful to get a 'big picture' view of fisheries, and give an opportunity to estimate necessary inputs in the ecosystem modelling approach for data-poor areas. Although performance analyses on catch-based methods indicate that CMSY performance was more accurate in estimating biomass (Zhou et al., 2018), and produce slightly similar results among other data-limited surplus production method, only a few results were compatible with official stock estimates (Bouch et al., 2021), and its results were strictly dependent on the quality of the prior information which bears a potentially high risk of over- or underestimating stock size (Pons et al., 2020; Sharma et al., 2021). We suggest that biomass and fishing mortality data based on CMSY can now be used in Ecopath modelling and its time dynamic module, Ecosim.

Although published and unpublished biomass data for zooplankton and macrobenthic epi/infaunal groups were available (e.g., Çinar et al., 2020), we avoided using these data as their temporal and spatial coverage were severely limited and would introduce significant uncertainty if used as initial conditions for these groups in the models because of the impossibility of capitalizing on statistical approaches to derive representative values for the whole SoM. This is probably because of the stratified nature of the SoM that creates a substantial barrier between pelagic-benthic ecosystems and hinders to produce data representing the whole ecosystem. Therefore, we relied on Ecopath estimations for their biomasses assuming an EE value of 0.9 for both groups.

Trophic levels of functional groups were equal in the three models, because the models were balanced using the same diet matrix to ensure similar topology. We had to rely on data from adjacent areas and similar ecosystems to parameterize the model when no such information was available from the SoM ecosystem. However, most of the information used to complement missing data was from nearby areas in the Black Sea or Northern Aegean Sea ecosystems and assumed to be representative of the overall characteristics of the populations in the Sea of Marmara because i) some stocks are migratory and shared between the Marmara and the Black Seas, and ii) the SoM is a two-layer stratified system forming a passage between these two seas and could be considered to be formed with the amalgamation of the characteristics of the Black Sea and the Aegean Sea ecosystems to a significant extent.

In the late 2000s, frequent observation of jellyfish blooms together with increases in jellyfish abundances and new species arrivals to the SoM were hypothesized to have significant impacts on the plankton community (Isinibilir et al., 2015) that could also cascade up to the fish community. Our models did not include jellyfishes due to the lack of regional biomass estimation. Jellyfish can affect the pelagic (Colin et al., 2005) and benthic (Sweetman and Chapman, 2015) ecosystems *via* their life history. They are directly related to zooplankton and anchovy in the food web with their wide range of prey (Purcell, 2009) and may have structural roles in the SoM similar to the Black Sea ecosystem (Akoglu et al., 2014). Therefore, the lack of such data is one of the important limitations for current and possibly future SoM models. Another major limitation is to understand the interactions of functional groups with fisheries due to the lack of species and fleet-specific landing and discard data. Such data should be collected and included in the future modelling studies. Our study also highlighted the data needs and the necessity to extend modelling efforts for a better understanding of the changes in the SoM using dynamic temporal modelling, like Ecosim, considering multiple stressors such as climate change, deoxygenation, invasive species and fisheries.

Conclusion

In this study three mass-balance Ecopath models were set up to delineate the ecosystem structure and function of the SoM from the 1990s to the 2010s. The comparison of the models showed that the energetic capacity (total system throughput) of the ecosystem decreased significantly over the three decades and the cycling indices indicated an ecosystem continuously becoming fragile to disturbances, notably fishing that is explicitly considered in this approach. The mixed trophic interactions and keystone species in the ecosystem showed an interplay between bottom-up and wasp-waist controls in the dynamics of the SoM food web. Further, strong negative impacts of fishing fleets on turbot, thornback ray and dogfish put the population dynamics of these species at risk and could push their populations to the brink of commercial extinction in the near future. Fisheries exploitation had significant responsibility on these changes; however, impacts of other factors that were not explicitly represented in our models such as impacts of pollution, sea warming, jellyfish species and changes in primary production dynamics, as indicated by the decreasing primary productivity and phytoplankton biomass in our models, should not be ruled out and future modelling work should consider these aspects.

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

IS, EA, ND: Formal analysis, Visualization, Writing-Original draft preparation. IS, GG, DB: Data curation, Investigation. ND: Supervision, Funding. All authors: Writing-Reviewing and Editing. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.1076399/full#supplementary-material

References

Agnetta, D., Badalamenti, F., Colloca, F., D'Anna, G., Di Lorenzo, M., Fiorentino, F., et al. (2019). Benthic-pelagic coupling mediates interactions in Mediterranean mixed fisheries: An ecosystem modeling approach. *PloS One* 14, e0210659. doi: 10.1371/JOURNAL.PONE.0210659

Akoğlu, E. (2021). Exploring the dynamics of small pelagic fish catches in the Marmara Sea in relation tochanging environmental and bio-optical parameters. *Turk. J. Zool.* 45, 257–265. doi: 10.3906/zoo-2012-23

Akoglu, E., Salihoglu, B., Libralato, S., Oguz, T., and Solidoro, C. (2014). An indicator-based evaluation of Black Sea food web dynamics during 1960–2000. *J. Mar. Syst.* 134, 113–125. doi: 10.1016/J.JMARSYS.2014.02.010

Akyol, O., and Perçin, F. (2006) Tekirdağ İli (Marmara Denizi) Kıyı Balıkçılığı ve Sorunları [The coastal fisheries and problems in Tekirdağ Province (Marmara Sea)] *E.U. J. Fish. Aquat. Sci* 23, 423–426.

Alheit, J., Licandro, P., Coombs, S., Garcia, A., Giráldez, A., Santamaría, M. T. G., et al (2014) Reprint of "Atlantic multidecadal oscillation (AMO) modulates dynamics of small pelagic fishes and ecosystem regime shifts in the eastern north and central atlantic." *J. Mar. Syst.* 133, 88–102 doi: 10.1016/J.JMARSYS.2014.02.005

Allen, K. R. (1971). Relation between production and biomass. *J. Fish. Res. Board. Can.* 28, 1573–1581. doi: 10.1139/f71-236

Altıok, H., Aslan, A., Övez, S., Demirel, N., Yüksek, A., Kıratlı, N., et al. (2014). Influence of the extreme conditions on the water quality and material exchange flux in the strait of Istanbul. *J. Mar. Syst.* 139, 204–216. doi: 10.1016/j.jmarsys.2014.06.005

Audzijonyte, A., and Kuparinen, A. (2016). The role of life histories and trophic interactions in population recovery. *Conservation Biology* 30 (4), 734–743.

Bakun, A. (2006). Wasp-waist populations and marine ecosystem dynamics: Navigating the "predator pit". *topograph. Prog. oceanogr.* 68 (2-4), 271–288. doi: 10.1016/j.pocean.2006.02.004

Bentley, J. W., Serpetti, N., and Heymans, J. J. (2017). Investigating the potential impacts of ocean warming on the Norwegian and barents seas ecosystem using a time-dynamic food-web model. *Ecol. Modell.* 360, 94–107. doi: 10.1016/J.ECOLMODEL.2017.07.002

Beşiktepe, Ş. T., Sur, H. I., Özsoy, E., Latif, M. A., Ouml;ĝuz, T., and Ünlüata, Ü. (1994). The circulation and hydrography of the Marmara Sea. *Prog. Oceanogr.* 34, 285–334. doi: 10.1016/0079-6611(94)90018-3

Beverton, R. J. H., and Holt, S. J. (1957). *On the dynamics of exploited fish populations fishery investigations* (London: Springer-Science-Business-Media B.V).

Bouch, P., Minto, C., and Reid, D. G. (2021). Comparative performance of data-poor CMSY and data-moderate SPiCT stock assessment methods when applied to data-rich, real-world stocks. *ICES J. Mar. Sci.* 78, 264–276. doi: 10.1093/ICESJMS/FSAA220

Christensen, V. (1995). Ecosystem maturity–towards quantification. *Ecol. Model.* 77 (1), 3–32. doi: 10.1016/0304-3800(93)E0073-C

Christensen, V., and Pauly, D. (1993). Flow characteristics of aquatic ecosystems. Trophic models of aquatic ecosystems. *ICLARM Conf. Proc.* 26, 38–352.

Christensen, V., and Walters, C. J. (2004). Ecopath with ecosim: Methods, capabilities and limitations. *Ecol. Modell.* 172, 109–139. doi: 10.1016/j.ecolmodel.2003.09.003

Christensen, V., Walters, C. J., and Pauly, D. (2005). *Ecopath with ecosim: A user's guide* (Vancouver, BC: University of British Columbia, 154).

Çinar, M. E., Bakir, K., Öztürk, B., Doğan, A., Açik, Ş., Kirkim, F., et al. (2020). Spatial distribution pattern of macroinvertebrates associated with the black mussel *Mytilus galloprovincialis* (Mollusca: Bivalvia) in the Sea of marmara. *J. Mar. Syst.* 211, 103402. doi: 10.1016/j.jmarsys.2020.103402

Clarke, K. R., and Warwick, R. M. (2001). Change in marine communities. *approach to Stat. Anal. interpret.* 2, 1–68.

Colin, S. P., Costello, J. H., Graham, W. M., and Higgins, J. (2005). Omnivory by the small cosmopolitan hydromedusa aglaura hemistoma. *Limnol. Oceanogr.* 50, 1264–1268. doi: 10.4319/LO.2005.50.4.1264

Corrales, X., Coll, M., Ofir, E., Heymans, J. J., Steenbeek, J., Goren, M., et al. (2018). Future scenarios of marine resources and ecosystem conditions in the Eastern Mediterranean under the impacts of fishing, alien species and sea warming. *Sci. Rep.* 8, 1–16. doi: 10.1038/s41598-018-32666-x

Corrales, X., Coll, M., Ofir, E., Piroddi, C., Goren, M., Edelist, D., et al. (2017). Hindcasting the dynamics of an Eastern Mediterranean marine ecosystem under the impacts of multiple stressors. *Mar. Ecol. Prog. Ser.* 580, 17–36. doi: 10.3354/MEPS12271

Costanza, R., and Mageau, M. (1999). What is a healthy ecosystem? *Aquat. Ecol.* 31, 105–115. doi: 10.1023/A:1009930313242

Darwall, W. R. T., Allison, E. H., Turner, G. F., and Irvine, K. (2010). Lake of flies, or lake of fish? A trophic model of lake Malawi. *Ecol. Modell.* 221, 713–727. doi: 10.1016/J.ECOLMODEL.2009.11.001

Daskalov, G. M. (2002). Overfishing drives a trophic cascade in the Black Sea. *Mar. Ecol. Prog. Ser.* 225, 53–63. doi: 10.3354/MEPS225053

Daskalov, G. M. (2008). *Overfishing affects more than fish populations: trophic cascades and regime shifts in the Black Sea* (Oxford, UK: Blackwell Publishing Ltd).

Demirel, N., Akoglu, E., Ulman, A., Ertör-Akyazı, P., Gül, G., Bedikoğlu, D., et al. (2023). Uncovering ecological regime shifts in the Sea of marmara and reconsidering management strategies. *Mar. Environ. Res.* 183, 105794. doi: 10.1016/j.marenvres.2022.105794

Demirel, N., Zengin, M., and Ulman, A. (2020). First Large-scale Eastern Mediterranean and Black Sea stock assessment reveals a dramatic decline. *Front. Mar. Sci.* 7. doi: 10.3389/FMARS.2020.00103/BIBTEX

Dimarchopoulou, D., Tsagarakis, K., Sylaios, G., and Tsikliras, A. C. (2022). Ecosystem trophic structure and fishing effort simulations of a major fishing ground in the northeastern Mediterranean Sea (Thermaikos gulf). *Estuar. Coast. Shelf Sci.* 264, 107667. doi: 10.1016/J.ECSS.2021.107667

Ediger, D., and Yilmaz, A. (1996). Characteristics of deep chlorphyll maximum in the northeastern Mediterranean with respect to environmental conditions. *J. Mar. Syst.* 9, 291–303. doi: 10.1016/S0924-7963(96)00044-9

Ergin, M., Bodur, M. N., Ediger, D., Ediger, V., and Yilmaz, A. (1993). Organic carbon distribution in the surface sediments of the Sea of marmara and its control by the inflows from adjacent water masses. *Mar. Chem.* 41, 311–326. doi: 10.1016/0304-4203(93)90263-N

FAO (2022). The state of Mediterranean and Black Sea fisheries 2020. (Rome: General Fisheries Commission for the Mediterranean).

Finn, J. T. (1976). Measures of ecosystem structure and function derived from analysis of flows. *J. Theor. Biol.* 56, 363–380. doi: 10.1016/S0022-5193(76)80080-X

Finn, J. T. (1980). Flow analysis of models of the Hubbard brook ecosystem. *Ecology* 61 (3), 562–571. doi: 10.2307/1937422

Froese, R., Demirel, N., Coro, G., Kleisner, K. M., and Winker, H. (2017). Estimating fisheries reference points from catch and resilience. *Fish. Fish.* 18, 506–526. doi: 10.1111/FAF.12190

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/J.MARPOL.2018.04.018

Halpern, B. S., Frazier, M., Potapenko, J., Casey, K. S., Koenig, K., Longo, C., et al. (2015). Spatial and temporal changes in cumulative human impacts on the world's ocean. *Nat. Commun.* 6, 1–7. doi: 10.1038/ncomms8615

Heath, M. R., Speirs, D. C., and Steele, J. H. (2014). Understanding patterns and processes in models of trophic cascades. *Ecol. Lett.* 17 (1), 101–114. doi: 10.1111/ele.12200

Hedberg, P. (2021). Responses of benthic-pelagic coupling to environmental change. [dissertation]. (Stockholm, Sweden: Stockholm University).

Heymans, J. J., Coll, M., Libralato, S., Morissette, L., and Christensen, V. (2014). Global patterns in ecological indicators of marine food webs: A modelling approach. *PloS One* 9, e95845. doi: 10.1371/JOURNAL.PONE.0095845

Heymans, J. J., Coll, M., Link, J. S., Mackinson, S., Steenbeek, J., Walters, C., et al. (2016). Best practice in ecopath with ecosim food-web models for ecosystem-based management. *Ecol. Modell.* 331, 173–184. doi: 10.1016/J.ECOLMODEL.2015.12.007

Heymans, J. J., and Tomczak, M. T. (2016). Regime shifts in the northern benguela ecosystem: Challenges for management. *Ecol. Modell.* 331, 151–159. doi: 10.1016/J.ECOLMODEL.2015.10.027

Hiddink, J. G., Moranta, J., Balestrini, S., Sciberras, M., Cendrier, M., Bowyer, R., et al. (2016). Bottom trawling affects fish condition through changes in the ratio of prey availability to density of competitors. *J. Appl. Ecol.* 53, 1500–1510. doi: 10.1111/1365-2664.12697

Horbowy, J., and Luzeńczyk, A. (2017). Effects of multispecies and density-dependent factors on MSY reference points: Example of the Baltic Sea sprat. *Can. J. Fish. Aquat. Sci.* 74, 864–870. doi: 10.1139/CJFAS-2016-0220

Hutchings, J. A. (2000). Collapse and recovery of marine fishes. *Nature* 406, 882–885. doi: 10.1038/35022565

Hutchings, J. A., and Reynolds, J. D. (2004). Marine fish population collapses: consequences for recovery and extinction risk. *Bioscience* 54, 297–309. doi: 10.1641/0006-3568(2004)054[0297:MFPCCF]2.0.CO;2

Imberger, J., and Patterson, J. C. (1989). Physical limnology. *Adv. Appl. Mech.* 27, 303–475. doi: 10.1016/S0065-2156(08)70199-6

Isinibilir, M., Yilmaz, I. N., and Demirel, N. (2015). New records of jellyfish species in the Marmara Sea. *Ital. J. Zool.* 82, 425–429. doi: 10.1080/11250003.2015.1040858

İşinibilir, M., and Yılmaz, İ. N. (2016). "Jellyfish species of the Sea of marmara," in *The Sea of marmara; marine biodiversity, fisheries, conservation and governance.* Eds. E. Özsoy, M. N. Çağatay, N. Balkıs, N. Balkıs and B. Öztürk (Turkish Marine Research Foundation Press), 390–400.

Libralato, S., Christensen, V., and Pauly, D. (2006). A method for identifying keystone species in food web models. *Ecol. Modell.* 195, 153–171. doi: 10.1016/J.ECOLMODEL.2005.11.029

Li, G., Cheng, L., Zhu, J., Trenberth, K. E., Mann, M. E., and Abraham, J. P. (2020). Increasing ocean stratification over the past half-century. *Nat. Clim. Change* 10, 1116–1123. doi: 10.1038/s41558-020-00918-2

Lindeman, R. L. (1942). The trophic-dynamic aspect of ecology. *Ecol* 23, 399–417. doi: 10.2307/1930126

Link, J. S. (2010). Adding rigor to ecological network models by evaluating a set of pre-balance diagnostics: A plea for PREBAL. *Ecol. Modell.* 221, 1580–1591. doi: 10.1016/J.ECOLMODEL.2010.03.012

Lynam, C. P., Llope, M., Möllmann, C., Helaouët, P., Bayliss-Brown, G. A., and Stenseth, N. C. (2017). Interaction between top-down and bottom-up control in marine food webs. *Proc. Natl. Acad. Sci. U.S.A.* 114, 1952–1957. doi: 10.1073/pnas.1621037114

Mackinson, S. (2014). Combined analyses reveal environmentally driven changes in the North Sea ecosystem and raise questions regarding what makes an ecosystem model's performance credible?. *Canadian Journal of Fisheries and Aquatic Sciences* 71 (1)31–46. doi: 10.1139/cjfas-2013-0173

McGarvey, R., Punt, A. E., Matthews, J. M., Feenstra, J. E., Gardner, C., Burch, P., et al. (2015). Comparing size-limit and quota policies to increase economicyield in a lobster fishery. *Can. J. Fish. Aquat. Sci.* 72, 1–14. doi: 10.1139/cjfas-2014-0405

Morissette, L. (2007). Complexity, cost and quality of ecosystem models and their impact on resilience : A comparative analysis, with emphasis on marine mammals and the gulf of st. Lawrence (Vancouver: University of British Columbia). doi: 10.14288/1.0074903

Neubauer, P., Jensen, O. P., Hutchings, J. A., and Baum, J. K. (2013). Resilience and recovery of overexploited marine populations. *Science* 340, 347–349. doi: 10.1126/science.1230441

Odum, E. P. (1969). The strategy of ecosystem development. *Science* 164, 262–270. doi: 10.1126/science.164.3877.262

Oguz, T., Akoglu, E., and Salihoglu, B. (2012). Current state of overfishing and its regional differences in the Black Sea. *Ocean. Coast. Manage.* 58, 47–56. doi: 10.1016/J.OCECOAMAN.2011.12.013

Palomares, M. L. D., Froese, R., Derrick, B., Meeuwig, J. J., Noel, S.-L., Tsui, G., et al. (2020). Fishery biomass trends of exploited fish populations in marine ecoregions, climatic zones and ocean basins. *Estuar. Coast. Shelf. Sci.* 243, 106896. doi: 10.1016/j.ecss.2020.106896

Palomares, M. L. D., and Pauly, D. (1998). Predicting food consumption of fish populations as functions of mortality, food type, morphometrics, temperature and salinity. *Mar. Freshw. Res.* 49, 447–453. doi: 10.1071/MF98015

Papantoniou, G., Giannoulaki, M., Stoumboudi, M. T., Lefkaditou, E., and Tsagarakis, K. (2021). Food web interactions in a human dominated Mediterranean coastal ecosystem. *Mar. Environ. Res.* 172, 105507. doi: 10.1016/j.marenvres.2021.105507

Pauly, D. (1980). On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. *ICES J. Mar. Sci.* 39, 175–192. doi: 10.1093/icesjms/39.2.175

Pauly, D., and Christensen, V. (1995). Primary production required to sustain global fisheries. *Nature* 374, 255–257. doi: 10.1038/374255a0

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, J. F. (1998). Fishing down marine food webs. *Science* 279, 860–863. doi: 10.1126/science.279.5352.860

Pauly, D., and Froese, R. (2021). MSY needs no epitaph–but it was abused. *ICES J. Mar. Sci.* 78 (6), 2204–2210. doi: 10.1093/icesjms/fsaa224

Pauly, D., and Zeller, D. (2016). Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining. *Nat. Commun.* 7, 1–9. doi: 10.1038/ncomms10244

Peck, M. A., Alheit, J., Bertrand, A., Catalán, I. A., Garrido, S., Moyano, M., et al. (2021). Small pelagic fish in the new millennium: A bottom-up view of global research effort. *Prog. Oceanogr.* 191, 102494. doi: 10.1016/j.pocean.2020.102494

Piroddi, C., Teixeira, H., Lynam, C. P., Smith, C., Alvarez, M. C., Mazik, K., et al. (2015). Using ecological models to assess ecosystem status in support of the European marine strategy framework directive. *Ecol. Indic.* 58, 175–191. doi: 10.1016/j.ecolind.2015.05.037

Pons, M., Cope, J. M., and Kell, L. T. (2020). Comparing performance of catch-based and length-based stock assessment methods in data-limited fisheries. *Can. J. Fish. Aquat. Sci.* 77, 1026–1037. doi: 10.1139/cjfas-2019-0276

Power, M. E., Tilman, D., Estes, J. A., Menge, B. A., Bond, W. J., Mills, L. S., et al. (1996). Challenges in the quest for keystones: Identifying keystone species is difficult-but essential to understanding how loss of species will affect ecosystems. *Bioscience* 46, 609–620. doi: 10.2307/1312990

Purcell, J. E. (2009). Extension of methods for jellyfish and ctenophore trophic ecology to large-scale research. *Hydrobiologia* 616, 23–50. doi: 10.1007/s10750-008-9585-8

Salihoglu, B., Arkin, S. S., Akoglu, E., and Fach, B. A. (2017). Evolution of future Black Sea fish stocks under changing environmental and climatic conditions. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00339

Serpetti, N., Baudron, A. R., Burrows, M. T., Payne, B. L., Helaouët, P., Fernandes, P. G., et al. (2017). Impact of ocean warming on sustainable fisheries management informs the ecosystem approach to fisheries. *Sci. Rep.* 7, 1–15. doi: 10.1038/s41598-017-13220-7

Shannon, L. J., Coll, M., and Neira, S. (2009). Exploring the dynamics of ecological indicators using food web models fitted to time series of abundance and catch data. *Ecol. Indic.* 9, 1078–1095. doi: 10.1016/j.ecolind.2008.12.007

Sharma, R., Winker, H., Levontin, P., Kell, L., Ovando, D., Palomares, M. L. D., et al. (2021). Assessing the potential of catch-only models to inform on the state of global fisheries and the UN's SDGs. *Sustainability* 13, 6101. doi: 10.3390/su13116101

Sweetman, A. K., and Chapman, A. (2015). First assessment of flux rates of jellyfish carcasses (jelly-falls) to the benthos reveals the importance of gelatinous material for biological c-cycling in jellyfish-dominated ecosystems. *Front. Mar. Sci.* 2. doi: 10.3389/fmars.2015.00047

Tas, S., Kus, D., and Yilmaz, I. N. (2020). Temporal variations in phytoplankton composition in the northeastern Sea of marmara: Potentially toxic species and mucilage event. *Mediterr. Mar. Sci.* 21, 668–683. doi: 10.12681/mms.22562

Tsikliras, A. C., and Froese, R. (2019). "Maximum sustainable yield," in *Encyclopedia of ecology*, *2nd*, vol. 1 . Ed. B. D. Fath (Oxford: Elsevier), 108–115.

Tu, C. Y., Chen, K. T., and Hsieh, C. H. (2018). Fishing and temperature effects on the size structure of exploited fish stocks. *Sci. Rep.* 8, 1–10. doi: 10.1038/s41598-018-25403-x

Ulanowicz, R. E. (1980). An hypothesis on the development of natural communities. *J. Theor. Biol.* 85, 223–245. doi: 10.1016/0022-5193(80)90019-3

Ulanowicz, R. E. (1986). *Growth and development ecosystems phenomenology* (New York: Springer-Verlag).

Ulanowicz, R. E. (2004). Quantitative methods for ecological network analysis. *Comput. Biol. Chem.* 28, 321–339. doi: 10.1016/j.compbiolchem.2004.09.001

Ulanowicz, R. E., and Puccia, C. J. (1990). Mixed trophic impacts in ecosystems. *Coenoses* 5, 7–16.

Ulman, A., Zengin, M., Demirel, N., and Pauly, D. (2020). The lost fish of Turkey: A recent history of disappeared species and commercial fishery extinctions for the Turkish marmara and Black Seas. *Front. Mar. Sci.* 7. doi: 10.3389/fmars.2020.00650

Walters, C., and Kitchell, J. F. (2001). Cultivation/depensation effects on juvenile survival and recruitment: Implications for the theory of fishing. *Can. J. @ Fish. Aqua. Sci.* 58, 39–50. doi: 10.1139/f00-160

Watson, R. A., Cheung, W. W. L., Anticamara, J. A., Sumaila, R. U., Zeller, D., and Pauly, D. (2013). Global marine yield halved as fishing intensity redoubles. *Fish. Fish.* 14, 493–503. doi: 10.1111/J.1467-2979.2012.00483.X

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., et al. (2009). Rebuilding global fisheries. *Science* 325, 578–585. doi: 10.1126/science.1173146

Yazıcı, F. M., İşmen, A., Altınağaç, U., and Ayaz, A. (2006). Marmara Denizi'nde karides algarnasının av kompozisyonu ve hedeflenmeyen av üzerine bir çalışma [A study on the catch composition and bycatch of shrimp beam trawl in the Sea of Marmara]. *E.U. J. Fish. Aquat. Sci.* 23, 269–275.

Yıldız, T., Ulman, A., and Demirel, N. (2020). A comparison of market landings during fish spawning seasons to better understand the effectiveness of the temporal fishery closure in Turkey. *Ocean. Coast. Manage.* 198, 105353. doi: 10.1016/j.ocecoaman.2020.105353

Zengin, M., Güngör, H., Güngör, G., Demirkol, C., İnceoğlu, H., Düz, G., et al. (2017). *Socio-economic structure of Marmara Sea fishery and determination of management strategies (MARBAL)* (Central Fisheries Research Institute), 224. Project No: TAGEM HAYSÜD/2008/09/04/01. Final report

Zhou, S., Punt, A. E., Smith, A. D. M., Ye, Y., Haddon, M., Dichmont, C. M., et al. (2018). An optimized catch-only assessment method for data poor fisheries. *ICES J. Mar. Sci.* 75, 964–976. doi: 10.1093/icesjms/fsx226

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EDITED BY
Athanassios C. Tsikliras,
Aristotle University of Thessaloniki, Greece

REVIEWED BY
Sanja Matic-Skoko,
Institute of Oceanography and Fisheries (IZOR), Croatia
Mohamed Samy-Kamal,
University of Alicante, Spain

*CORRESPONDENCE
Loretta Malvarosa
✉ malvarosa@nisea.eu

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Data availability and participatory approach: the right mix for enhancing Mediterranean fisheries' sustainability

Loretta Malvarosa[1]*, Gualtiero Basilone[2], Pierluigi Carbonara[3], Paolo Carpentieri[1], Maria Cozzolino[1], Maria Cristina Follesa[4], Monica Gambino[1], Vita Gancitano[5], David Parreno Duque[6], Paola Pesci[4], Ilaria Vielmini[7] and Giuseppe Scarcella[8]

[1]NISEA, Fisheries and Aquaculture Economic Research, Salerno, Italy, [2]National Research Council of Italy (CNR)/Institute for the Study of Anthropic Impact and Sustainability in the Marine Environment (IAS), Istituto per lo studio degli Impatti Antropici e Sostenibilità in ambiente marino, Capo Granitola, Italy, [3]COISPA Tecnologia & Ricerca, Bari, Italy, [4]Dipartimento di Scienze della vita e dell'ambiente, Università di Cagliari, Cagliari, Italy, [5]National Research Council of Italy (CNR)/Institute for Biological Resources and Marine Biotechnology (IRBIM), Istituto per le Risorse Biologiche e le Biotecnologie Marine, Mazara del Vallo, Italy, [6]Marine Stewardship Council (MSC) Italia, Milano, Italy, [7]Independent Researcher, Genova, Italy, [8]National Research Council of Italy (CNR)/Institute for Biological Resources and Marine Biotechnology (IRBIM), Istituto per le Risorse Biologiche e le Biotecnologie Marine, Ancona, Italy

A misalignment between the legislation and the effectiveness of Mediterranean fisheries management has emerged due to the status of the stocks (still largely in overfishing) and the discontent of stakeholders regarding management plans and tools that are not always recognized as appropriate to the characteristics of the concerned fisheries. Stakeholders' involvement in management processes is one of the main pillars of the Common Fisheries Policy revision. The literature underlines, in an increasingly urgent manner, the importance of stakeholders fully understanding the contents of management plans and, vice versa, necessary for the successful implementation of policies. Focusing on the path towards sustainability endorsed by the BluFish project, the paper tries to provide an answer about the sustainability of some selected Southern Italian fisheries, by adopting the assessment approach of the Marine Stewardship Council. The assessment approach, based on a set of Performance Indicators and on a well-defined scoring scheme, focuses on three dimensions of sustainability. In addition to the classic assessment of the state of the stocks, there is an extensive screening of the impact of anthropic activity such as fishing on the entire ecosystem, including both the impact on accessory species and on vulnerable habitats and species. The evaluation adopted goes even further, with an approach that also includes the management and governance sphere, also trying to evaluate the level of involvement of the operators in the decision-making process. The paper illustrates that the selected fisheries are not fully sustainable but some of them have excellent potential for improvement even in the short term by identifying and implementing the appropriate action. The most relevant weaknesses identified refer to the low scores obtained for sustainability of stocks, mainly around the Harvest Control Rules (HCR) and the Harvest Strategy indicators, highlighting the importance of improving the management of the assessed stocks. The paper highlights how data and scientific knowledge availability is essential for a detailed mapping and evaluation of fishing activities but also that the path towards more sustainable and responsible fisheries does not work without a strong participation of all the key stakeholders.

KEYWORDS

Mediterranean, fisheries management, sustainability, eco-certification, participatory approach, improvement plans, fisheries, management

Introduction

Approximately 73% of the fish stocks assessed in the Mediterranean Sea are considered to be in overexploitation (FAO, 2022a; FAO, 2022b); however, recent trends show a consistent decrease in stocks assessed in overexploitation, especially since 2012, when this percentage was 83% (FAO, 2022b). This overexploitation is the result of various factors, among which the most important is fleet overcapacity, as emerged, in the last years and especially for Italy, in the framework of STECF (Scientific, Technical and Economic Committee for Fisheries) working groups on balance between fishing opportunities and fishing capacity (STECF, 2020; STECF, 2021; STECF, 2022a). Not marginal is the role of a still lacking monitoring program. Even if improving, data availability on the status of Mediterranean stocks is, indeed, far from being optimal: without taking into account large pelagic species and considering only non-deprecated assessments (e.g. less than three or five years depending on the species) the percentage of stocks for which advice was provided on a quantitative and qualitative (precautionary) basis remained around 25 percent, (FAO, 2022b). According to the European Environmental Agency, the poorness of data collection emerges also for monitoring programs related to the evaluation of the environmental impact of fisheries (EEA, 2020). The poor involvement of the fishing sector in the decision-making process and weak market engagement in promoting the sustainable exploitation of natural resources, might have contributed to further slow the process toward sustainability of the Mediterranean fisheries. Gomez and Lloret (2017) have, indeed, highlighted a misalignment between the rules or legislation and the social practice which does not fully consider local ecological and socio-cultural specificities in the implementation of effective measures or the participation of key stakeholders in fisheries' policies. Stakeholders' involvement in the management processes is one of the pillars of the basic Regulation, the Common Fisheries Policy CFP (EU Reg. No. 1380/2013, Article 3, Principles of good governance, h). The literature on fisheries management underlines, in an increasingly urgent manner, the importance of stakeholders fully understanding the contents of management plans and measures adopted as well as, vice versa, the relevance of fisheries managers correctly interpreting the perceptions of the interested parties involved, which is necessary for the successful implementation of the policies (Garza-Gil et al., 2015; De Vos et al., 2016). Linke and Bruckmeier (2015) describe the importance of the co-management of fisheries developed in Europe through various experimental forms of fishermen participation in the management process, in advisory roles or through delegation and sharing of power. Higher involvement increases transparency and affects positively the interpretation of management measures, the basis for their acceptance by the fishery sector. A greater acceptance leads to higher compliance with and more effective implementation of the measures, allowing the management objectives to be achieved faster (Pita et al., 2012; Malvarosa et al., 2019).

Management authorities are beginning to use certification programs, applied to evaluate the sustainability performance of fisheries throughout the world, to provide a framework for recognizing best practice actions and for identifying and analyzing the challenges regarding the adoption of measures that can achieve improvements in fisheries management in the short and long term (Gozzer-Wuest et al., 2023). In particular, Marine Stewardship Council (MSC) standards are increasingly used before making sweeping adjustments aimed at enhancing efficiencies for all fisheries, not just those seeking certification. This multi-stakeholder, collaborative approach, which is known as the Project Pre-Assessment (PPA) or Fishery Improvement Project (FIP) models, has already been applied in Australia, Indonesia, Mexico, South Africa, Japan, the UK and, most recently, the Mediterranean region, with the aim of promoting the improvement of the sector's management. Through a combination of fisheries mapping and pre-assessments of the fisheries performance against the certification standard, this approach offers governments, fishermen, scientists, market players and local non-governmental organizations the opportunity to collaborate in identifying the most efficient route to make environmental improvements at the most appropriate scale. The main feature of a PPA is that its intended impact extends beyond the immediate project results, aiming at improving the overall fisheries management. Cooperation between fishers, NGOs, research institutes, international agencies, administrations, public institutions, and retailers increases, indeed, the possibility of accessing the necessary resources, expands skills, strengthens the sense of responsibility of the involved actors and paves a more conscious path towards fish sustainability (Anderson et al., 2021), which may eventually lead to certification, creating further benefits for operators. Participatory methods have, indeed, demonstrated their potential to integrate ecosystem-based management in a community-based approach, involving all actors in a proactive manner and "considering fisheries as a human activity socially and culturally rooted in the environment, which would enhance the effective implementation of fisheries policies" (Gomez and Maynou, 2021a).

The first PPA experience in the Mediterranean, the Medfish project, has been carried out in the Western Mediterranean engaging fisheries in France and Spain (http://www.project-medfish.com/). Medfish has led to positive conclusions about the replicability of the PPA approach in other Mediterranean fisheries, identifying that more than half of the performance indicators of the selected fisheries' needed improvements to reach the MSC level. Most of the improvements required were related to the impact of fisheries on the environment and to management/governance aspects of the fisheries concerned and data availability and reliability were identified as transversal weaknesses.

Building on the lessons learnt in Medfish and the main benefits gained from fisheries, its approach has been replicated, fine-tuned and further developed in certain Italian fisheries selected in Southern Italy, including the islands.

This paper highlights the main findings of the first stages of the BluFish pathway, focusing on the a) fisheries' identification and b) their sustainability assessment, the latest carried out according to the MSC principles and standards based on the FAO Code of Conduct for Responsible Fisheries. Feeding on the available public data and on a participatory approach developed during the BluFish project, the paper tries to provide an answer to the question: are Southern Italian fisheries sustainable? If not or not enough, which actions are needed to improve the process toward sustainability? According to the MSC approach, "the sustainability of a fishery can be assessed regardless of its size, geography or the fishing method used" against three main principles: sustainability of the stocks, environmental impact of the fishery, effectiveness of fishery's management.

At the time of writing, over 539 fisheries MSC certified worldwide (MSC, 2022), only three (3) fisheries in the Mediterranean, of which one in Italy, have been deemed to be compliant with MSC standards, thus achieving, after a "full assessment" process, the related certification: e.g. two Bluefin tuna fisheries (Spain and France) and the Venetian striped Venus clam fishery[1]. This proves how difficult it is, for various reasons, for Mediterranean fisheries, to access a sustainability certification. In most cases, the reason lies in the lack of an appropriate management strategy; at times, the lack of appropriate data collection is the reason. The main challenges stem from the multispecies nature of Mediterranean fisheries, where most fleets consist of small-scale vessels often catching mixed stocks with a variety of gears in the course of the same trip (FAO, 2022a). On the other hand, it is demonstrated that wild-caught fish populations targeted by MSC-certified fisheries have higher relative abundance than non-MSC populations (Melnychuk et al., 2022) as well the role of eco-certification as an important tool in addressing IUU fishing (Longo et al., 2021).

The sustainability assessment becomes crucial if we consider that the fishing activities selected for the pre-assessment phase (see the results section for details) are among the most relevant at local and national socio-economic level: the anchovy (ANE, *Engraulis encrasicolus*) fishery is one of the top fishery both in terms of volume and value of landings (4% and 6% of the total, respectively, in 2020); at the same time, the red shrimp (ARS, *Aristaeomorpha foliacea*) and the deep-water rose shrimp (DPS, *Parapenaeus longirostris*) fisheries are the most important, in terms of value: 8% and 7%, respectively. Spiny lobster (SLO, *Palinurus elephas*) is the most valued species (ex-vessel price equal to 44 €/kg in 2020) and, similarly to swordfish (SWO, *Xiphias gladius*), plays a key role for the economy of local coastal communities, representing, as in the case of lobster, an important share of revenues for small-scale vessels. In addition, trawling and purse seine fishing (to which 7 of the 10 selected fisheries refer) are also relevant from a socio-economic point of view, as they generate about 36% of national employment. It is also important to underline that the selected activities mainly refer to the southern Adriatic and Sicily, the most relevant areas, from a productive point of view, for the Italian fishery (STECF, 2022b).

Materials and methods

Scanning and mapping fisheries: the relevance of data collection

The sustainability assessment of the Southern Italian fisheries started with a preliminary scan of all the possible existing fisheries in the coastal areas under analysis - Figure 1.

The "fast scan" (the name of the phase under the BluFish project) built upon the identification of all possible métiers active along the Southern Italian coastline. A métier is defined as "a group of fishing operations targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or the

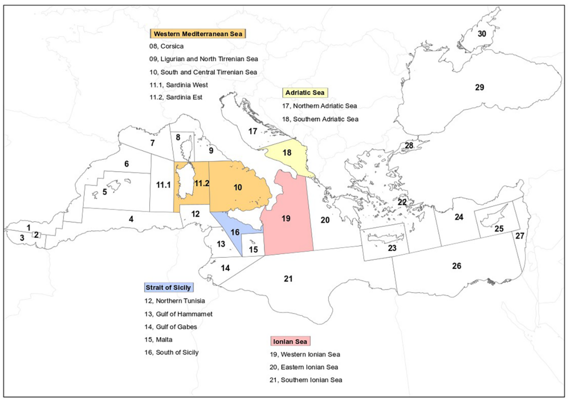

FIGURE 1
Map of the scanned area using the FAO Geographical Subdivision Areas (colored GSAs).

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[1] https://fisheries.msc.org/en/fisheries/sathoan-french-mediterranean-bluefin-tuna-artisanal-longline-and-handline-fishery/@@view; https://fisheries.msc.org/en/fisheries/venetian-wild-harvested-striped-clam-venus-chamelea-gallina/@@view; https://fisheries.msc.org/en/fisheries/jc-mackintosh-greenstick-handline-and-fishing-rod-bluefin-tuna-fishery/@@view

same area and which are characterized by a similar exploitation pattern" (EC Decision 2008/949) [2]. The notion of a métier is therefore closely linked to fishermen's activities, patterns, traditions and gears. Accordingly, each métier involves a set of fishing operations characterized by a combination of fishing gear, target species, area and season, which constitute homogeneous units that supply the main characteristics of a large number of fishing trips in a single variable (González-Álvarez et al., 2016). The list of métiers of the Mediterranean Sea has been identified by the Regional Coordination Meeting for the Mediterranean and the Black Sea (RCM MED&BS, Sete, 2008) and is available on the website of the European Data Collection Framework [3]. For practicality, the level of métier used for the present study was level (5), covering gear type and target assemblage, independently from the mesh size dimension. The "fast scan" substantiated the identification of all combinations of area/métier/species, where the area refers to the geographical sub-area (GSA) as defined by the General Fisheries Commission for the Mediterranean (GFCM) with the Resolution FCM/33/2009/2[4], replying to the need to compile data, monitor and assess fisheries resources in a georeferenced manner, by identifying appropriate boundaries for Mediterranean fishing areas. For each GSA and species, the status of the stock (where available) was reported, using information obtained from various databases (mainly STECF, GFCM, ICCAT.), reporting the information in terms of F/FMSY (F = fishing mortality; FMSY = fishing mortality at the maximum sustainable yield [MSY] level). Biomass reference points were also reported, if available.

The mapping phase was complemented by the identification of the most important fisheries according to two objective criteria:

1. fisheries where the relevant species was a target species and
2. the target species being among the 20 most important species by volume and value of the related GSA.

The first criterion was a scientific method validated by the STECF, which considers the 75% threshold of the cumulative value and volume of landings (STECF, 2015). This approach was originally developed by the STECF in support of the scientific advice to the CFP, in particular to address the EC request for supporting the implementation of the landing obligation regulation and has been employed to identify the main European demersal fisheries in the Mediterranean. By accessing the data on the volume and value of landings collected under the Italian national program for the fishing sector[5], publicly available on the Joint Research Center website[6],the 75% threshold of the cumulative value and volume of landings (sum of the values of the two years 2015–2016, available at the starting time of the study) was used, for each fishery and gear combination in each GSA, to identify the most represented taxa.

Once the "target fisheries" have been classified using the threshold criterion, the top-20 fisheries by volume and value of landings (sum of the values of the two years 2015–2016) have been identified for each GSA.

The sustainability (pre-)assessment

The sustainability assessment of fisheries that carried out during the BluFish path was, actually, a "pre-assessment", according to the MSC terminology. In a standard certification process, the main aim of the (sustainability) pre-assessment of fisheries is to define areas that may require additional data or improvements before a "full assessment" is undertaken. Considering that the ultimate goal of the BluFish path is the creation of an enabling environment for the sustainability and effective management of fish resources, the MSC pre-assessment approach was used to identify areas of improvement to accompany the fisheries, step-by-step, towards a more sustainable state.

MSC defines a standard approach for the pre-assessment of fisheries willing to check if it is possible to conduct a more in-depth sustainability evaluation leading to certification. The pre-assessment follows the MSC standards, consisting of three principles (Principle 1: sustainable target fish stocks, Principle 2: environmental impact of fishing and Principle 3: effective management) and 28 related performance indicators (PIs) - Table 1.

The pre-assessment process involves scoring all PIs using narrative guides focusing on specific aspects of the fishery (called scoring guideposts [SGs]). There are 28 performance indicators that sit under the three principles of the MSC Fisheries Standard. The fishery is assigned a score for each performance indicator, where 60 is the minimum acceptable performance, 80 is global best practice and 100 is state of the art performance. To become certified, the score must be at least 60 for each of the 28 performance indicators. If it scores between 60 and 79 for any of them, the fishery will be required to take appropriate action as a condition of certification, improving the performance so that it scores 80 or above for each indicator within 5 years maximum. Additionally, the fishery must have an aggregate weighted mean score of 80 or higher for each of the three aforementioned principles to be certified.

In some cases, when sufficient quantitative data are not available to score a given PI using the usual set of SGs, a risk-based framework may be used. This tool uses a precautionary approach to estimate aspects such as stock status (Principle 1) and impacts on bycatch and habitats (Principle 2) when conventional data, including reference points derived from analytical stock assessment models, doesn't exist,

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[2] 2008/949/EC: Commission Decision of 6 November 2008 adopting a multiannual Community programme pursuant to Council Regulation (EC) No 199/2008 establishing a Community framework for the collection, management and use of data in the fisheries sector and support for scientific advice regarding the common fisheries policy

[3] https://datacollection.jrc.ec.europa.eu/wordef/fishing-activity-metier.

[4] https://www.fao.org/gfcm/data/maps/gsas/en/

[5] As per EEC 1543/2000, Reg. EEC 199/2008, and EU Reg. 1004/2017, which provide the EU framework for the collection, management and use of data for the fishery sector.

[6] https://datacollection.jrc.ec.europa.eu/data-analysis/fdi

TABLE 1 MSC Performance Indicators (PIs) by Principles and Components.

Principle	Component	Performance Indicator (PI)
1. Sustainable fish stocks	Outcome	1.1.1 Stock status
		1.1.2 Stock rebuilding
	Management	1.2.1 Harvest Strategy
		1.2.2 Harvest control rules and tools
		1.2.3 Information and monitoring
		1.2.4 Assessment of stock status
2. Minimising environmental impacts	Primary species	2.1.1 Outcome
		2.1.2 Management
		2.1.3 Information
	Secondary species	2.2.1 Outcome
		2.2.2 Management
		2.2.3 Information
	ETP species	2.3.1 Outcome
		2.3.2 Management
		2.3.3 Information
	Habitats	2.4.1 Outcome
		2.4.2 Management
		2.4.3 Information
	Ecosystem	2.5.1 Outcome
		2.5.2 Management
		2.5.3 Information
3. Effective management	Governance and Policy	3.1.1 Legal and customary framework
		3.1.2 Consultation, roles and responsibilities
		3.1.3 Long term objectives
	Fishery specific management system	3.2.1 Fishery specific objectives
		3.2.2 Decision making processes
		3.2.3 Compliance and enforcement
		3.2.4 Monitoring & Management performance evaluation

but is not deployed for Principle 3. As the aim of the pre-assessment is not necessarily the certification per se, the results of the scoring were used to identify areas needing improvements in terms of sustainability (i.e. PIs not reaching a score of 60), with the aim of developing an improvement plan of actions.

Each PI has then associated scoring issues that can be scored against SG60, SG80 or SG100. For example, indicator 2.3.2 (ETP species management strategy) had 5 Scoring issues: a) Management Strategy in place (national and international requirements), b) Management Strategy in place (alternative), c) Management Strategy evaluation, d) Management Strategy implementation and e) Review of alternative measures to minimize mortality of Endangered, Threatened and Protected (ETP) species.

To better explain the rationale behind the scoring process, it is worth mentioning that one of the fisheries subjected to assessment scored a SG80 for Scoring issue c) Management Strategy evaluation, as there was an objective basis for confidence that the measures/ strategy would work, based on information directly about the fishery and/or the species involved. However, no detailed quantitative analysis had been carried out to assess the impact of fishery-related mortality on turtles and cetaceans and a quantitative analysis of the effectiveness of the strategy had thus yet to be carried out, not meeting SG 100. On the other hand, another fishery didn't meet the minimum scoring of 60 for this same PI (2.3.2) and Scoring issue - c), Management Strategy evaluation - as there was a lack of management measures that could positively affect some of the shark species identified, not being able to determine that measures were considered likely to work, based on plausible arguments.

The participatory approach: selecting fisheries, identifying improvement actions and checking expectations

The sustainability (pre-)assessment was conducted for a selected number of fisheries, to identify areas that need improvement actions along the path toward sustainability. The feedback from key stakeholders was gathered through a step-by-step consultation process. The selection was carried out while paying particular attention to the following criteria:

- being knowledgeable about the fisheries under evaluation
- possessing expertise in the status and biology of the target stock
- representing or being in key organizations long enough to have observed a before-and-after change
- being able to speak on behalf of the stakeholder group they represent
- representing diverse perspectives regarding the stakeholder type

Based on this, fishermen, scientists, producer associations and Fisheries Local Action Groups' (FLAGs) representatives, representatives of control agencies, etc.…were involved in the process. First of all, key stakeholders were consulted for the selection of fisheries to be assessed in terms of sustainability. This phase was conducted between September and October 2018, through workshops, face-to-face or telephone consultations or attendance in key local events for the fishery sector. The GSAs focus of the BluFish path encompasses fishing ports of different Italian administrative regions of Southern Italy and the islands. Some GSAs are wider, covering more than one administrative region; therefore, multiple consultations were held in some regions. To discuss the selection of GSA 18 fisheries, at the end of September 2018, two workshops were organized in Apulia, in the Adriatic towns of Manfredonia and Bisceglie – two ports characterized by the relevance of demersal trawling fleets. The consultations with stakeholders from Sardinian fisheries (GSA 11) were carried out *via* telephone. Sicilian stakeholders were consulted at the beginning of October 2018 for the three GSAs related to the Sicilian coast (10 for the Northern coast, 16 for the Southern coast and 19 for the Eastern/Ionian coast). These consultations were realized through a workshop organized *ad hoc* in Palermo, with the support of a local key stakeholder, playing a key role in the Sicilian and national fish processing sector. Another consultation was held by gathering stakeholders for a workshop along a well-known event organized yearly in Mazara del Vallo – the Blue Sea Land conference. To cover all the fishing ports of GSA 10, representing the waters of the Southern Tyrrhenian Sea, in person consultations were carried out in Campania, along a national event organized for FLAGs organized in Cetara, a port of the Salerno Gulf, relevant for both small and large pelagic fisheries. Additional stakeholders from Calabria were consulted *via* telephone for both the Tyrrhenian and the Ionian sides.

In all the consultations, the stakeholders were requested to provide their perceptions or opinions on different aspects, among which the most important concern: a) the market of the target species – local, national or foreign? b) if there is interest from the market (and consumers) in sustainable products; c) how and how much does the management system support the identified fishery d) how relevant is the fishery from a socio-economic perspective at the local or regional level (including related industries, such as processing and catering).

In a path towards sustainability, it is also important to test, at a certain stage, the expectations and main concerns of the stakeholders regarding the path itself. Knowledge of stakeholders' expectations, mainly that of fishers, is crucial to understand if, e.g., additional training is required. The perception of sustainability is, indeed, not always unique among stakeholders. It is rather common that sustainability paths carried out under the aegis of certification bodies are often linked directly to the possibility of gaining certification as this means, especially for fishers, new markets, premium prices and higher incomes. Nevertheless, it is also essential to identify the expected concerns of the stakeholders regarding the path, as these could act as obstacles, hindering the concerned fisheries' pathway to sustainability. To this aim, the stakeholders' feedback for fisheries showing an early and likely interest to undertake improvement actions, if needed, was gathered using a questionnaire containing a set of simple questions, 28 in total (Supplementary Material/Annex A), which covered topics ranging from what they expected from the improvement of their fisheries, in terms of benefits, on a personal basis and for the fishing community as a whole to their main concerns about the success of the path. Due to the Covid-19 health restrictions, the questionnaire interviews were performed by a web-based survey, providing respondents with assurance of the confidentiality of the information collected. The number of stakeholders interviewed, by category and by years of experience/involvement in the sector, is reported in Figure 2 (left side).

The participatory approach has played a crucial role also in the post-assessment phase, for the identification of the most appropriate actions that could lead the selected fishery towards a higher level of sustainability overcoming the current, if any, unsustainability of fisheries ("action plan development"). Improvement actions have been identified for the selected fisheries and action plans have been agreed for some of them by means of focus groups (Finch et al., 2014) organized for each selected fishery. The workshops, which were organized physically or virtually between September 2020 and June 2021, tackled the objective of presenting the results of the pre-assessment to the stakeholders and creating a space for the participating groups to give clarifications, provide more information and validate the analysis of the strengths and weaknesses identified during the pre-assessment. A total of 91 individuals were involved in the workshops, including the facilitators (two – the same for each consultation) and action plan requesters on behalf of the local

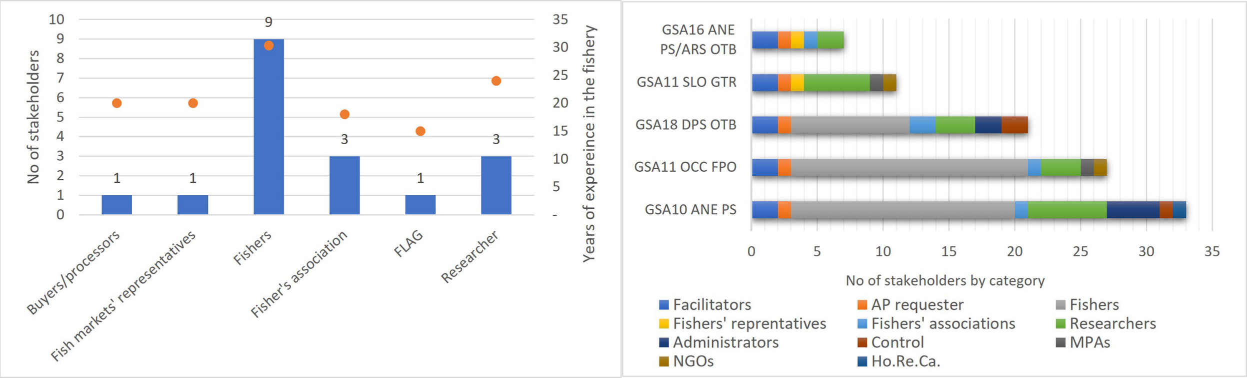

FIGURE 2
Stakeholders consulted in the test of expectations from the BluFish sustainability path (left) and in the action plans' development (right) phases.

fisheries. Most of the participants were fishers (48%) and representatives of cooperatives (2%) or fishers' associations (5%). The research area was also significantly represented (21%). Representatives of the administration were represented by governors of the local administrative regions (7%) or by persons in charge of fisheries control (3%, coast guard). In a few workshops, individuals from fishery ancillary activities were also involved, as they had a high interest in the sustainability path (i.e. processors or Ho.Re.Ca. representatives) - Figure 2, right side.

It is worth noting, however, that authors have not been able to evaluate the final results of this last phase of the BluFish path as some of the Action plans are still in a draft version and those validated by stakeholders are at a very early stage of implementation.

Results

Scanning and selecting fisheries

The scanning methodology resulted in the identification of *2,606* fisheries - combination of GSA/métier/species or Unit of Assessment (UoAs) in the MSC terminology - in Southern and Central Tyrrhenian Sea (GSA 10), Sardinia (GSA 11), Strait of Sicily (GSA 16), Southern Adriatic Sea (GSA 18) and Western Ionian Sea (GSA 19). These fisheries represented 154 species, 12 target assemblages and 16 gears. Demersal and deep-water fisheries represented the vast majority (77%), with bottom trawling dominating (41%) with regard to the gears used. Set gillnets and trammel nets were used in 28% of the UoAs mapped, mainly for demersal fisheries. The use of pots and traps was limited to 2%. Purse seiners were used in 8% of the UoAs for small pelagic fish (Figure 3).

A preliminary examination indicated that official stock assessments (GFCM, STECF, ICCAT) were available only for the target species of 188 (7%) UoAs; alternative sources (Froese et al., 2018) provided further information on 143 additional UoAs. The complete dataset is reported on the Supplementary Material/Annex B.[7] The results of the "fast scan" were then processed by applying the 75% threshold approach illustrated in the methodological section, with the aim of producing the initial list of fisheries to be further "mapped". All combinations of GSA/gear/target assemblage/species were plotted by highlighting the "main" target species (i.e. those falling in the 75% threshold of the cumulative volume and value of landings). The plot for the small pelagic fisheries by purse seines in GSA 11 is illustrated in Figure 4 while all the plots are reported in the Supplementary Material/Annex C. In all the plots, the change in the slope of the cumulative value and volume of landings is also reported to provide detailed information on catch composition.

As illustrated in the methodological section, further sorting was performed by selecting the main fisheries (identified by the 75% threshold approach) falling in the top 20 species by volume and value in the years 2015–2016.

The application of the two aforementioned criteria resulted in the identification of a list of 174 UoAs, distributed differently by GSAs and over different techniques and species. As mentioned in the methodological section, this preliminary list of 174 fisheries represented the base list for the identification of fisheries to be evaluated in terms of sustainability according to the MSC standards for pre-assessment. These 174 fisheries were further screened by checking, with the help of stakeholders, the existence of certain features/attributes. This phase resulted in the identification of 50 fisheries (Figure 5). The list of the 174 fisheries with the results of the two levels of selection (1) objective criteria and 2) stakeholders' consultations) is reported in Supplementary Material/Annex D.

The final selection of the 10 fisheries (Table 2) to be subjected to the sustainability evaluation was driven, in addition to budget

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[7] The BluFish scanning phase coverd also GSAs 17 (Northern Adriatic) and 9 (Ligurian and North Tyrrhenian Sea) but the subsequent phase focused only on GSAs 10, 11, 16, 18 and 19. For more details https://www.msc.org/docs/default-source/it-files/blufish-fast-scan-annex-i-table-uoa01079fc1191c4f60b39884aec309b981.pdf?sfvrsn=9c1bd020_0

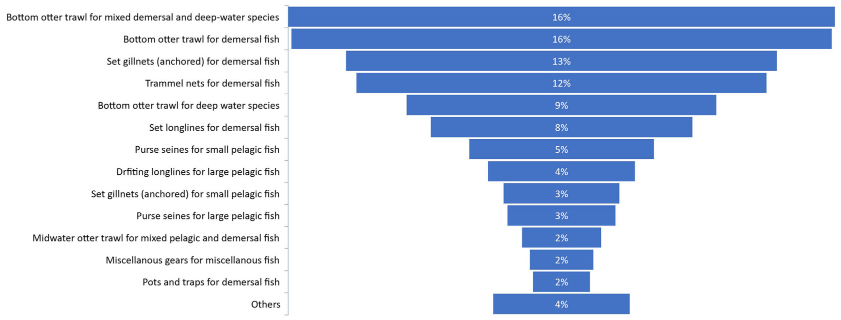

FIGURE 3
Fisheries scanned during the BluFish path in GSAs 10, 11, 16, 18 and 19 by main metiér.

constraints, by the need to investigate the sustainability potential for different fisheries (e.g. cover different ports, areas, techniques, species) and, *condictio sine-qua non*, by the existence, on the side of the sector, of the concrete availability of fishers to be engaged in a (pre-)assessment process.

Sustainability evaluation: the (pre-) assessment results

The pre-assessed fisheries were proven to rely on good practices (around 70% of the PIs scoring >= 80, as shown in Figure 6). Only 4% of the PIs reported a score below 60, representing a critical situation. It is worth noting that the overall number of PIs is not merely equal to the sum of 280 (28 PIs foreseen by the MSC approach times the number of pre-assessed fisheries). The BluFish pre-assessment resulted, indeed, in a total of 273 PIs considering that the PI 1.1.2 related to “stock rebuilding” is scored only if the target stock is overexploited. As a result, this aspect was assessed only for ANE PS GSA18, SWO HAR GSA19 and SLO GTR GSA 11, where a plan was required to rebuild the stock.

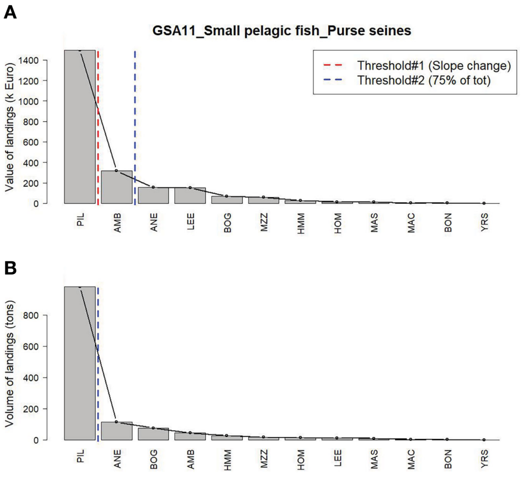

FIGURE 4
Plot of the cumulative value **(A)** and volume **(B)** by species (accoriding to FAO 3 Alpha Codes) for the small pelagic fisheries by purse seines in the GSA 11 (Sardinia).

On examining the synthetic but specific results for each fishery, a very heterogeneous performance was observed in the pre-assessment (Figure 7). By converting the evaluation of each PI in a numerical range from 0 to 1 (with <60 = 0, 60–79 = 0.50 and >= 80 = 1), it can be observed that better scores were recorded for the evaluation of Principle 2 aspects (environmental impact), with the average score for all the fisheries being 0.88. This was followed by Principle 3 (fisheries management), with an average score of 0.75. Principle 1 (stock status) had the lowest scores (0.71), highlighting the urgent need for Mediterranean fisheries to pursue improvement policies in the management of fish stocks.

When specific fisheries are considered, the swordfish fishery with harpoon had the highest score globally (0.93), being a fishery characterized by high selectivity and stocks managed by means of quotas. In contrast, the red shrimp trawl fishery had the lowest score (0.48) because of the lack of information on target species (need to improve the stock assessment) and secondary species and vulnerable species, as well as in terms of governance. In general, the scores recorded for Principle 1 remain at a lower level in comparison to Principles 2 and 3, highlighting the clear need for Mediterranean fisheries to improve their stock management (Figure 8).

Evaluating the expected benefits and concerns of a path towards sustainability

This “check” was performed with some of the fishers/stakeholders involved in two fisheries: the deep-water rose shrimp fishery with bottom trawlers in Southern Adriatic (GSA 18) and the anchovy fishery with purse seiners in the Southern Tyrrhenian Sea (GSA 10). A total of 32 stakeholders were interviewed during the first semester of 2021. The majority (72%) were represented by fishers or their representatives (fishery associations and LAGs); the remainder comprised representatives of marketing or processing activities as well as researchers.

Seven main groups of *expected general benefits* for the fisheries as a whole were identified, with 76 mentions in total (Figure 8). The valorization of the products, along with an increase in market efficiency, was the most mentioned group of benefits (21%) that

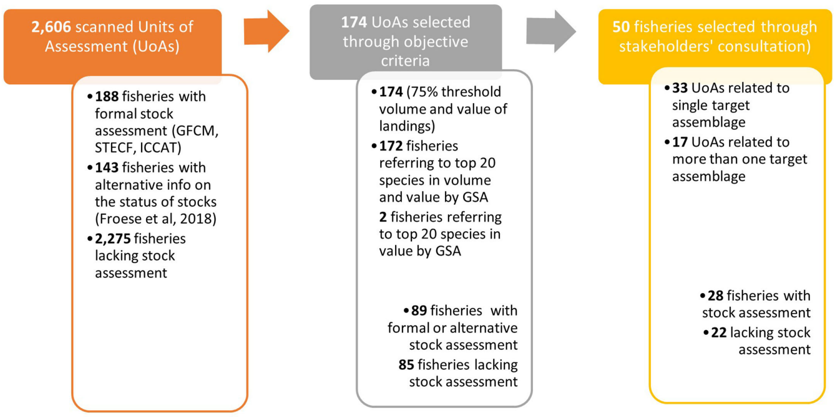

FIGURE 5
Summary of the fisheries' selection process of the BluFish path.

should be achieved, through improvements in traceability, labelling and advertising campaigns aimed at emphasizing the sustainability of the product (Figure 9).

At the second level, the stakeholders expected to achieve a more sustainable fishery (20% of mentions) from the path, especially the stakeholders of the pink shrimp fishery, who expressed a strong need for a concrete and operational path towards sustainability in terms of shared rules for the Adriatic Sea. The improvement of marine resources' management was the second most expected benefit; based on the opinions of all the stakeholders, the management might be improved with more tailored policies for the environment and the conservation of biodiversity. Only 9% of the stakeholders mentioned the improvement of long-term socio-economic sustainability and the adoption of different catch and effort strategies tailored to the specificity of each fishery (e.g. individual effort quota for the shrimp fishery, catch quota for the anchovy fishery). The improvement of the stocks' status was mentioned in 8% of the replies, while another 8% of respondents called for a better governance structure and higher involvement of operators' participation in the decision-making processes. the remaining 5% of responses related to miscellaneous replies.

Nine main groups of *expected personal benefits* were identified, with 61 mentions in total (Figure 10). The mentions (equaling replies) were lower than those for the general benefits, as personal benefits were expected only by fishers – the actors that could gain direct benefits from the path towards sustainability. The potential to earn a higher income was one of the two most expected benefits (with 18% of mentions), The other most mentioned benefit was the potential to gain greater sensitivity and knowledge of what a sustainable fishery is, in both environmental and socioeconomic terms; this was also connected to an improved understanding of the concepts underlying sustainability. Furthermore, 16% of the respondents called for an increase in the product quality (e.g. higher size) and, as a consequence, higher market value (product valorization); they also expected improvements in the traceability process. Another 3% of the replies stated that the main expectation was to gain certification, which is viewed as a valorization tool capable of producing a premium price and, hence, higher earnings.

At the personal level, there was also a strong expectation (15%) of improvements in the current management, highlighting fishers' need for a better comprehension of management tools and the decision-making process behind their implementation. In

TABLE 2 Summary of fisheries selected for the pre-assessment by GSAs, species and gear.

Fishery codification	GSA	Latin name of spp	Gear
GSA10 ANE PS	Southern Tyrrhenian Sea (GSA10)	*Engraulis encrasicolus*	Purse Seine
GSA10 DOL PS		*Coryphaena hippurus*	
GSA11 OCC FPO	Sardinia (GSA11)	*Octopus vulgaris*	Trap
GSA11 LOB GTR		*Palinurus elephas*	Trammel net
GSA16 ANE PS	Strait of Sicily (GSA16)	*Engraulis encrasicolus*	Purse Seine
GSA16 ARS OTB		*Aristaeomorpha foliacea*	Bottom trawl
GSA18 ANE PS	Southern Adriatic (GSA18)	*Engraulis encrasicolus*	Purse Seine
GSA18 DPS OTB		*Parapenaeus longirostris*	Bottom trawl
GSA18 EOI OTB		*Eledone cirrhosa*	
GSA19 SWO HAR	Ionian Sea (GSA 19)	*Xiphias gladius*	Harpoon

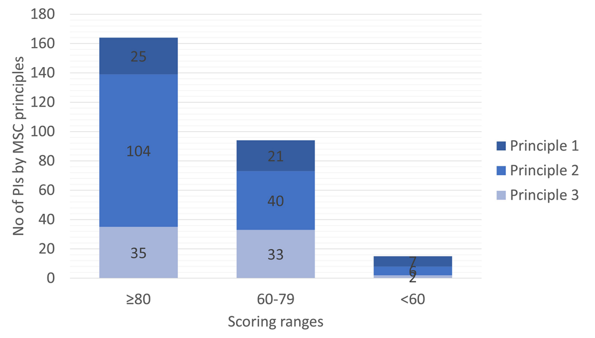

FIGURE 6
Scoring distribution by range and principles.

addition, 8% of fishers called for better working conditions, in reference to the improvement of the social well-being of people living on fishery. Another 5% expected a personal improvement in terms of professionalism. For another 8% of respondents, the opportunity to be more involved in the sustainability path would lead to a careful biological monitoring and scientific data sharing, and therefore to a deeper understanding of the resource status and better cooperation among the management, researchers and the sector.

On the other hand, seventeen main groups of *expected concerns* were identified, with 37 mentions in total (Figure 11). The main problem concerns the real possibility of new fishermen involvement in the process (other than those already actively participating in the path), which would also imply a change in their mindset and, consequently, in fishing practices (19% of mentions). The second most common issue (14%) concerns cooperation with institutional stakeholders, as the fishing sector is still perceived as managed by a top-down approach, with marginal involvement of fishers in the decision-making process (11%). The lack of confidence in ground-up actions was the fourth most cited issue (8%), which was strongly linked to the fear that the path could be an obstacle due to (low) compliance with rules (8%). For 5% of respondents the low level of compliance is due to the excessive degree of stringency of the current rules. Two respondents (5%) also reported concerns about the management of shared resources; this was particularly felt in the Adriatic, whose stocks are under the legislative power of the EU and GFCM. The market capacity to perceive the new brand and, consequently, to accept higher sales prices – in the event that the fishery reaches the certification step – was also mentioned twice. Another set of nine miscellaneous replies should be highlighted, as they concerned the fear that the current crisis in the sector (which is also related to the need for modernizing production and processing activities) could prevent other fishers from undertaking the path. Other concerns, with one mention each, included the need to raise awareness among operators and to conduct more in-depth training for them, the poor quality of the scientific data collected, the proper communication of rules, the efficiency of the management and general skepticism regarding the sustainability approach.

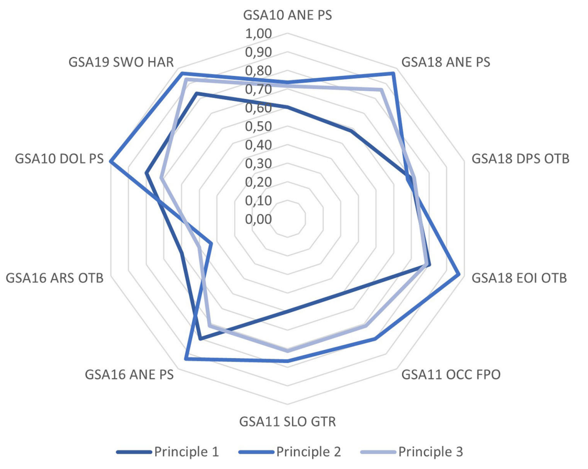

FIGURE 7
Overall scoring by Principle and fisheries.

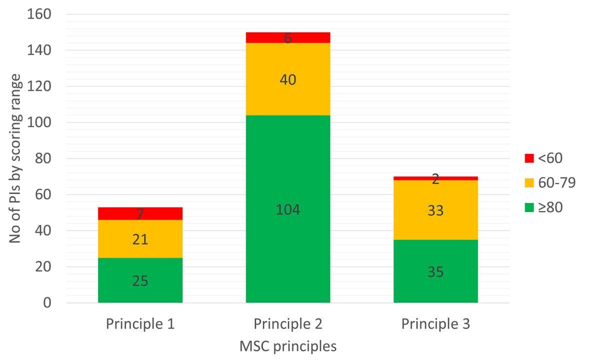

FIGURE 8
Scoring distribution by principles.

Tentative improvement actions

Because of budget and time constraints, improvement actions were identified, and complete action plans were drafted only for 6 fisheries: GSA 10/European anchovy/purse seine, GSA 18/deep water rose shrimp/bottom trawling, GSA 11/common octopus/trap, GSA 11/common spiny lobster/trammel net, GSA 16/European anchovy/purse seine and GSA 16/giant red shrimp/bottom trawling. Only 3 of these action plans have finally been validated by stakeholders and these are, at the time of writing, at a very early stage of implementation[8].

On the 6 action plans drafted, a total of 219 actions were identified to accompany the fisheries towards sustainability. The majority (43%) of areas needing improvements were those aspects related to the "management and status of the stock" (Figure 12). The main actions planned to tackle these gaps concern improvement in the monitoring systems, the revision of the current fishing strategy, of stocks' rebuilding plans (for some

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[8] For details see https://www.msc.org/it/cosa-facciamo/il-nostro-contributo-al-cambiamento/progetto-blufish/sviluppo-piani-dazione

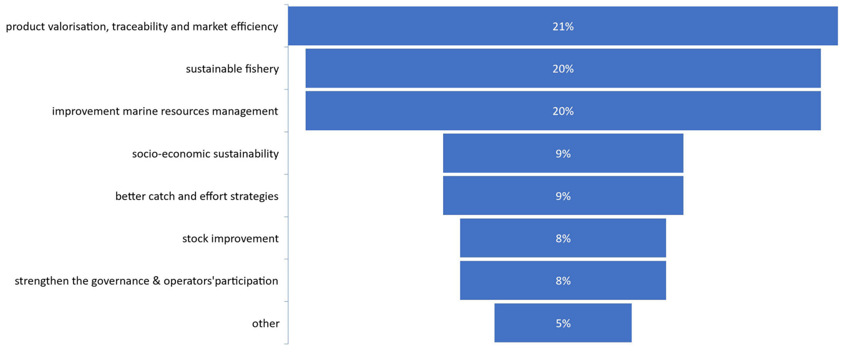

FIGURE 9
Expected general benefits from participating in the BluFish path towards sustainability.

stocks). A weakness common to all fisheries concerns the lack of up-to-date data, in particular for stock assessment as well as "old" or absent management plans (Table 3).

Furthermore, a significant need to improve the sustainability of the selected fisheries emerges with regard to the impact on the environment (34% of the planned actions; Figure 12), conceptualized as the impact of fisheries on all the aquatic resources different from the target species, namely out-of-scope species, ETPs, habitats and the whole ecosystem. As for aspects related to the target species, the action plans foresee update or integration of the current monitoring, the implementation of management strategies also for non-target species and a deeper evaluation of the status of vulnerable species, with the adoption of mitigation measures, if needed.

Several areas needing improvement have also been identified in the governance dimension of the pre-assessed fisheries. For most of them, the lack of sustainability is strongly dependent on inefficiencies in the decision-making process (e.g. lack of a well-defined governance structure, lack of clear monitoring of the achievements of the management plan's objectives and scarce involvement of the sector's representatives in the definition of new management plans) (Figure 12). Actions have been planned for a further development of the structure of governance, to improve or integrate the evaluation and monitoring of existing management plans, for setting roundtables with stakeholders to decide on and promote new proposals on fishing strategies and management plans and, generally, to increase the effectiveness of existing control measures (Table 3).

Discussion

The manuscript describes the approach used for the mapping and the sustainability evaluation of some selected Southern Italian fisheries. The approach was based on merging data-based sources (scanning and mapping of fisheries) with a strong participatory approach, involving stakeholders all along the path. Such approach has allowed to identify fisheries areas of improvements transversally over the three dimensions of sustainability.

The MSC certification process is transparent and open to public scrutiny, encouraging relevant stakeholder input if needed. However, while sometimes experts' judgements may lead to incongruous outputs, using all available knowledge and involving relevant stakeholders in revising the draft scores where necessary allows for improvement in terms of objectivity of the scoring system. Furthermore, the MSC standards always request to apply the precautionary approach when different scores on the same performance indicator emerge during an harmonization process among different UoAs exploiting the same stock. In the framework of the present study the exercise in selecting the UoA was carried out starting from an empirical ranking of the landings followed by multiple interactions with stakeholders to streamline a list of few

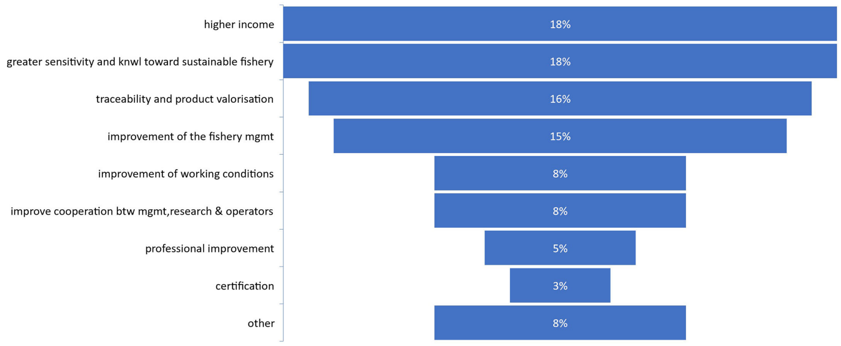

FIGURE 10
Personal benefits from participating in the BluFish path towards sustainability.

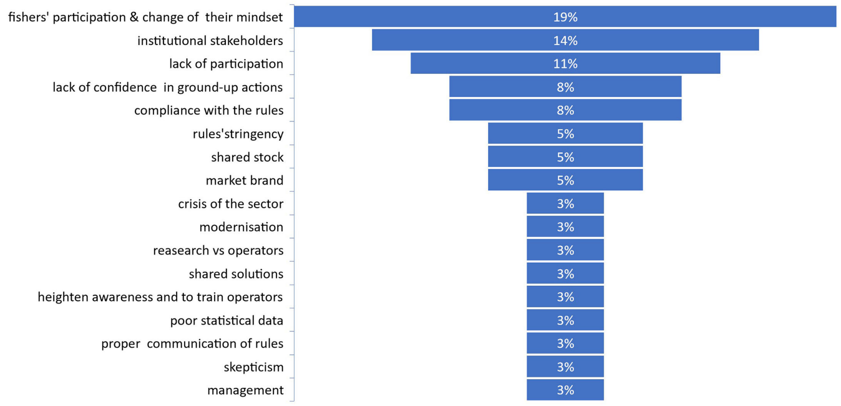

FIGURE 11
Expected concerns from participating in the BluFish path towards sustainability.

UoAs. Also the scoring has been carried out using the most updated knowledge on status of the stocks (both in P1 and P2) and evidence about the performance of the management system both in term measures implemented and availability of information. Also, it is important to recall that the certification was not the objective of the present study while it was the fisheries improvement process through an active participatory approach of the fisheries. The ultimate goal of the fisheries assessment was to indeed discuss the assessments outcomes with the fisheries as to confirm or revise scoring as necessary and - most importantly - to identify concrete bottom-up solutions to overcome the gaps identified in the assessments.

Although an increasing trend is reported in the number of stock assessed in the Mediterranean Sea, data collection to inform the status of stocks not only in relation to fishing mortality still needs improvements: according to FAO (2022b) the scientific advice on the status of resources in relation to biomass is scarcer than advice related to fishing mortality. The results of the scanning show some critical stock abundance information to be missing, as not all the species landed along the Italian coasts are subject to stock assessment. As highlighted in previous sections, a review of the most recent literature (at the time of the scanning phase) for information missing on the status of some Mediterranean stocks was undertaken. Even if it had not been formally validated by the relevant Regional Fisheries Management Organizations (RFMOs) scientific committees at that time, the study by Froese et al. (2018), which examined the status of 397 European stocks using a data-limited approach, was used to fill this gap. The sustainability path described here has contributed to improve the understanding of stocks dynamics through the action plan by: i. identify in the action plan the data needed and the entity(ies) and the fundings available to collect and support the analysis; ii. enabling the fisheries'

FIGURE 12
Distribution of improvement actions by principle and main areas of intervention (fisheries concerned GSA10 ANE PS, GSA18 DPS OTB, GSA11 OCC TRAP, GSA11 SLO GTR, GSA16 ANE PS, GSA16 ARS OTB).

TABLE 3 Main improvement actions needed by areas of intervention.

Area of intervention/MSC principle	Main improvement actions planned
Management and status of the target stocks	➢ New assessment on the status of the identified stock ➢ Revision of existing fishing strategies ➢ Revisions of the current harvest control rules ➢ Implementation of a monitoring program for the collection of data (biological, socio-economic, effort, etc.) on target species ➢ Adoption of new management plans
Management of non-target species, ETP species, habitats and ecosystems	➢ Assessment of the effects of fishing activity on marine habitats ➢ Implementation of monitoring programs for the collection of data on vulnerable species, sensitive habitats and the ecosystem in general ➢ Improvement of the fishing and management strategies for non-target commercial species ➢ Evaluation of the status of vulnerable species and adoption of mitigation measures (if needed)
Governance and decision-making process	➢ Development of structure of governances ➢ Evaluation and monitoring of existing management plans ➢ Roundtables with stakeholders to decide on and promote new proposals on fishing strategies and management plans ➢ Increasing the effectiveness of existing control measures

representatives to make informed management demands including to advocate for updated stock assessments.

For several years, input control has been considered the only possible approach to fishing management in the Mediterranean (Cardinale et al., 2017). However, it has been demonstrated that this strategy has not achieved the objectives of the CFP (Colloca et al., 2017; Vielmini et al., 2017). Other measures, such as output control, bycatch reduction and ecosystem-based management, are necessary to tackle the state of Mediterranean stocks (Cardinale et al., 2017) and ensure the long-term biological and economic sustainability of the fishing sector (Sabatella et al., 2017). However, it is undeniable that output controls are implemented with tools such as TAC, which are not easily applicable in multispecies/multigear fisheries as in Mediterranean context with many vessels and landing sites. This, together with potential quota allocation conflicts, complicates the implementation and control tasks for the management authorities (Bellido et al., 2020). The manuscript reinforces the need for more well-defined harvest strategies. What is particularly missing, indeed, according to the MSC approach, for the pre-assessed fisheries, are proactive, well-defined harvest control rules (HCR) identified and implemented in an effective harvest strategy, which can guarantee healthy state and sustainable management of the stocks in the long run. According to the MSC, a harvest strategy is defined as a combination of elements, including data collection, the provision of scientific advice and the implementation of specific HCRs, ideally working together to achieve the objectives of the standard certification, which are in general agreement with the MSY paradigm. The participatory process described in the present manuscripts aims to actively contribute to pave the way of sustainable fisheries management. Indeed, management related fisheries improvement actions have included, when needed and on the basis of managers and fisheries feedback, the timeline and steps required to develop and implement new harvest strategies.

Recent reports on the achievement of the Good Environmental Status (GES) by EU member states highlight that Italian environmental policies are headed in a good direction regarding the protection of the marine environment; nevertheless, further efforts are still needed (COM[2018] 562 final; SWD[2019] 123 final). This situation is common at the entire EU level, as highlighted by Maes et al. (2021), and is causing most EU countries to be far from attaining GES. According to the European Environmental Agency (EEA, 2020) the condition of biodiversity in the Mediterranean sea is poor or decreasing, for species covered by data collection (e.g. cetaceans, birds, bony fish and sharks and rays). Most importantly, the EEA report highlights a high low coverage of monitoring programs. The results illustrated in the present manuscript are in line with the EEA conclusions. Indeed, one of the major caveats emerging from the pre-assessment is the lack of appropriate data collection and/or availability of enough long time series, if any, on the impact of fisheries on the ecosystem the fisheries operate in. This manuscript contributes to tackle the need to increase the knowledge on the impact of (sometimes highly impacting) fisheries (e.g. bottom trawls) on environment (Hiddink et al., 2023) because improvement actions, identified for the abovementioned 6 fisheries, clearly identifies which are habitats, vulnerable species and non-target species (presence and abundance) that need to be further investigated in order to increase data availability in support of management. It also contributes by clearly highlighting (in the action plans) management gaps, most of the time related to the absence of management strategies whose objectives cover the needs to minimize the environmental impact of the fisheries concerned (e.g. mitigation measures).

The literature recognizes a strong and systematic participatory approach can significantly improve operators' awareness of management measures, paving the way, through a concrete bottom-up approach, for a more sustainable management of fisheries resources in the Mediterranean. On the contrary, a scarce stakeholders involvement in the decision-making process increases transparency and information gaps between policy makers and management users (fishermen), distorting operators' perceptions of management measures (Pita et al., 2012). A misunderstanding of the measures, in particular the link between the measures and the achievement of the objectives, triggers a vicious circle: a lack of knowledge results in low-acceptance, in turn triggering mechanisms of non-compliance that negatively impact the effectiveness of the entire management system (Malvarosa et al., 2019). Maravelias et al. (2018) also indicates that "seeking stakeholder involvement throughout the management process, i.e. proposal preparation, negotiation phase, project kick-off, conception of management scenarios, model parameterization, and evaluation of model outputs", helps in the "identification of realistic and sustainable management measures with high levels of acceptance from stakeholders". The consultations conducted throughout the path have clearly outlined what the turning point for a change in fisheries management in the Mediterranean could be: they expect that the BluFish path would help in reinforcing the relationship between management and fishermen but, on the other side, they highlight room for improvements by indicating the presence of vulnerabilities that could hinder a path toward sustainability: a sector still perceived as top-down managed, lack of confidence in ground-up actions, a high perceived degree of stringency of the current rules with a consequent low perceived compliance with rules represent, for them, real obstacles for the effectiveness of similar path.

As highlighted by Macher et al. (2021), "science/stakeholder/managers partnerships for decision support in fisheries can play an essential role in knowledge integration towards the Ecosystem Based Fishery Management (EBFM)". It is well recognized that the EBFM encapsulates the movement towards a more cooperative and holistic approach to marine resource management (Leslie and McLeod, 2007) and recognizes the combined physical, biological, economic, and social trade-offs affecting the fisheries sector, and the need to address these trade-offs when optimizing fisheries yields from an ecosystem (Link, 2010). In line with these findings, the paper highlights that a concrete path towards sustainability needs to be shaped based on a strong stakeholders' partnership, where all the actors are involved in the co-creation and use of knowledge in support of common decisions. The approach described by the manuscript contributes to the literature as it represents a sort of benchmark for a multi partnerships decision tool. Moreover, in line with Target 4 of the GFCM 2030 Strategy for sustainable fisheries and aquaculture in the Mediterranean and the Black Sea (FAO, 2021), putting fishers at the front in the co-creation of knowledge, the path has highlighted the useful role that they are asked to play, by providing their feedback on *ad-hoc* consultations or, on request, their availability to host observers on board or to directly collect specific data, e.g. on landings, bycatch, etc.…

The capacity of defining, preparing and establishing partnerships between operators, scientists, policymakers and civil society are the strengths behind PPA projects (Gozzer-Wuest et al., 2023). Undertaking a path towards certification can have additional and unanticipated effects, related to social and governance outcomes, i.e. market expansion, collaboration among harvesters and increased trust and cooperation between the industry and managers (Anderson et al., 2021). Furthermore, recent studies demonstrate that r seafood value chains may have impacts on ecosystems, as consumers' choices are drivers of fishing strategies because they exert pressure on product requirements (Gomez and Maynou (2021b), the idea behind the theory of change, at the basis of the MSC approach.

The recent literature on the European value chains (Josupeit, 2016), indicate that, in most cases, the chief problem is related to the distribution channel inefficiency, being most of the time too "crowded" (too many actors from sea-to-fork). Furthermore, according to a recent study on the value chain of two fisheries involved in the BluFish sustainability path (deep-water rose shrimp in Southern Adriatic and anchovies fishery in Southern Tyrrhenian sea), the fresh seafood production is often in the hands of wholesalers and traders, who operate in near-monopoly conditions and thus set prices unilaterally and the presence of several landing sites (which are often not controlled) results in a highly fragmentation of supply, negatively affecting the producers'/fishermen's bargaining power (Malvarosa et al., 2021) and, as a result, the possibility for them of maximizing profits. Pursuing improvements in the fisheries sustainability levels, potentially leading to a sustainability certification can help in the process of product valorization (among the main expected benefits of fishermen consulted, to compensate for declining productivity and revenues), by creating an intrinsic value for landed species thus helping also in increasing the products' placement, both on national and on export markets, especially those markets in which the appeal of eco-labelling for consumers is higher, e.g. Germany (Zander and Feucht, 2017).

In its 2020 report, the European Court of Editor (ECA, 2020) examined whether the use of EMFF, LIFE and Interreg programs contributed to increase environmental sustainability, supporting marine conservation. The EMFF (now substituted by EMFAF) was meant to support fishing and aquaculture activities as well as the Marine Strategy Framework Directive, contributing to protect the marine environment. The report highlights the low level of use of EMFF funds (0.2%) by the 4 Member States visited (among which is Italy, the third recipient of EMFF funds at EU level, the first at Med level according to the report) for projects aimed to limit the impact of fishing on marine environment. Recent studies (Ballesteros et al., 2018) highlight the great potential of these funds when used under an appropriate path, referring to EMFF Measure 4.63, promoting the implementation of community-led local development (CLLD) strategies. According to De Boni et al., (2018), a crucial factor impacting the success of financed projects is, indeed, related to the capacity to establish partnerships between all the actors (operators, scientists, policymakers, and civil society) that can combine growth and sustainability, while respecting the territorial specificities. FLAGs play a key role, as territorial hubs, in promoting competition by increasing the environmental awareness of local operators and fishers (Gambino et al., 2022). In line with these findings, the BluFish path can act as a best practice considering that 2 of the 4 validated action plans are leaded by local action groups (GAL Ponte Lama, for the deep-water rose shrimp fishery in GSA18 and FLAG Approdo di Ulisse, for the anchovies fishery in GSA10) and that the unique in phase of implementation (deep-water rose shrimp GSA 18) has been financed under EMFF 2014/2020 Measure 4.63[9].

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[9] https://www.msc.org/it/news/notizia/2021/10/20/al-via-le-azioni-di-miglioramento-della-pesca-al-gambero-bianco-di-molfetta

To conclude, the manuscript highlights that there are ample margins for the selected fisheries to be deemed sustainable as 68% of the indicators used in the assessment report scores, indeed, as "good practices". At the same time, a number of improvement actions are needed, transversally over the three dimensions represented by management of stocks, environmental impact and governance. A path based on the BluFish approach can really help in leading to increase the effectiveness of management in terms of marine conservation as well as in ensuring the socio-economic sustainability of the sector, in line with the CFP pillars.

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

LM, IV, GS, and DP contributed to the conception and design of the study. LM wrote the first draft of the manuscript and organized the database for the scanning phase. GS performed the scanning phase and coordinated the pre-assesment. PaC coordinated the identification of improvement actions. MG developed the analysis of the expected benefit and concerns. All authors contributed to manuscript revision, read, and approved the submitted version.

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Conflict of interest

This research received funding from the Marine Stewardship Council (MSC), to which DPD is affiliated at the time of writing. IV was affiliated at MSC during the first years of the BluFish project but was independent researcher at the time of the writing. At the time of the final submission, IV is employed in European Commission DG MARE and the information and views set out in this article are those of the author and do not necessarily reflect the official opinion of her employer.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1155762/full#supplementary-material

References

Acocella, I. (2005) *L'uso dei focus groups nella ricerca sociale: vantaggi e svantaggi», quaderni di sociologia [Online]*. Available at: http://journals.openedition.org/qds/1077.

Anderson, C. M., Himes-Cornell, A., Pita, C., Arton, A., Favret, M., Averill, D., et al. (2021). Social and economic outcomes of fisheries certification: Characterizing pathways of change in canned fish markets. *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.791085

Ballesteros, M., Chapela, R., Santiago, J. L., Norte-Navarro, M., Kesicka, A., Pititto, A., et al. (2018). "Research for PECH committee – implementation and impact of key maritime and European fisheries fund measures (EMFF) on the common fisheries policy, and the post-2020 EMFF proposal," in *European Parliament* (Brussels: Policy Department for Structural and Cohesion Policies). Available at: https://www.europarl.europa.eu/cmsdata/194329/Implementation%20and%20impact%20of%20EMFF%20on%20CFP%20study-original.pdf.

Bellido, J. M., Sumaila, U. R., Sánchez-Lizaso, J. L., Palomares, M. L., and Pauly, D. (2020). Input versus output controls as instruments for fisheries management with a focus on Mediterranean fisheries. *Mar. Policy* 118, 103786. doi: 10.1016/j.marpol.2019.103786

Cardinale, M., Osio, G. C., and Scarcella, G. (2017). Mediterranean Sea: A failure of the European fisheries management system. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00072

Colloca, F., Scarcella, G., and Libralato, S. (2017). Recent trends and impacts of fisheries exploitation on Mediterranean stocks and ecosystems. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00244

De Boni, A., Roma, R., and Ottomano Palmisano, G. (2018). Fishery policy in the European Union: A multiple criteria approach for assessing sustainable management of Coastal Development Plans in Southern Italy. *Ocean Coastal Manage* 163, 11–21. doi: 10.1016/j.ocecoaman.2018.05.022

De Vos, B., Döring, R., Aranda, M., Buisman, F. C., Frangoudes, K., Goti, L., et al. (2016). New modes of fisheries governance: implementation of the landing obligation in four European countries. *Mar. Pol.* 64 (2016), 1–8. doi: 10.1016/j.marpol.2015.11.005

ECA (2020) *European Court of Editor, special report 26/2020: Marine environment.* Available at: https://op.europa.eu/webpub/eca/special-reports/marine-environment-26-2020/en/.

EEA (2020). "European Environment agency," in *Marine messages II. navigating the course towards clean, healthy and productive seas through implementation of an ecosystem–based approach*, 2019–2077, ISBN 978-92-9480-197-5. ISSN 1977-8449. doi: 10.2800/71245

FAO (2021). *GFCM 2030 strategy for sustainable fisheries and aquaculture in the Mediterranean and the black Sea* (Rome: Food and Agriculture Organization of the United Nations). doi: 10.4060/cb7562en

FAO (2022a). "The state of world fisheries and aquaculture 2022," in *Towards blue transformation* (Rome: FAO). doi: 10.4060/cc0461en

FAO (2022b). "The state of Mediterranean and black Sea fisheries 2022," in *General fisheries commission for the Mediterranean* (Rome). doi: 10.4060/cc3370en

Finch, L., Lewis, J., and Turley, C. (2014). "Focus groups," in *Qualitative research practice - a guide for social science students and researchers, 2nd ed.* Eds. J. Ritchie, J. Lewis, C. McNaughton Nicholls and R. Ormston (London, Thousand Oaks, New Delhi, Singapore: Sage Publication Ltd), 211–242, ISBN: .

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/j.marpol.2018.04.018

Gambino, M., Accadia, P., Costantini, M., Gomei, M., Malvarosa, L., Sabatella, E. C., et al. (2022). Analysis of the available funds supporting marine activities in some key European Mediterranean countries. *Front. Res. Metr. Anal.* 7. doi: 10.3389/frma.2022.927383

Garza-Gil, M. D., Amigo-Dobaño, L., Surís-Regueiro, J. C., and Varela-Lafuente, M. (2015). Perceptions on incentives for compliance with regulation. the case of Spanish fishermen in the Atlantic. *Fish. Res.* 170, 30–38. doi: 10.1016/j.fishres.2015.05.012

Gomez, S., and Lloret, J. (2017). The small-scale fisheries guidelines as a tool for a marine stewardship: The case of cap de creus marine protected areas. In *The small-scale fisheries guidelines. Global Implementation* (Spain: Springer) pp.401–420. doi: 10.1007/978-3-319-55074-9_19

Gomez, S., and Maynou, F. (2021a). Balancing ecology, economy and culture in fisheries policy: Participatory research in the Western Mediterranean demersal fisheries management plan. *J. Environ. Manage.* 291, 112728. doi: 10.1016/j.jenvman.2021.112728

Gomez, S., and Maynou, F. (2021b). Alternative seafood marketing systems foster transformative processes in Mediterranean fisheries. *Mar. Policy* 127, 104432. doi: 10.1016/j.marpol.2021.104432

González-Álvarez, J., García-de-la-Fuente, L., García-Flórez, L., Fernández-Rueda, M. d. P., and Alcázar-Álvarez, J. L. (2016). Identification and characterization of métiers in multi-species artisanal fisheries. a case study in Northwest Spain. *Natural Resour.* 7, 295–314. doi: 10.4236/nr.2016.76026

Gozzer-Wuest, R., Vinatea Chávez, R. A., Olea Stranger, G., Araya Goncalves, G., Hiriart-Bertrand, L., Labraña-Cornejo, R., et al. (2023). Identifying priority areas for improvement in Chilean fisheries. *Front. Mar. Sci.* 10. doi: 10.3389/fmars.2023.1073397

Hiddink, J. G., Evans, L., Gilmour, F., Lourenço, G., McLennan, S., Quinn, E., et al. (2023). The effect of habitat and fishing-effort data resolution on the outcome of seabed status assessment in bottom trawl fisheries. *Fish. Res.* 259, 106578. doi: 10.1016/j.fishres.2022.106578

Josupeit, H. (2016). *Mercati della pesca artigianale: catena del valore, promozione ed etichettatura small-scale fisheries markets: value chain, promotion and labelling* (Brussels: Parlamento Europeo). Available at: https://www.europarl.europa.eu/RegData/etudes/STUD/2016/573443/IPOL_STU(2016)573443_IT.pdf

Leslie, H. M., and McLeod, K. M. (2007). Confronting the challenges of implementing marine ecosystem-based management. *Front. Ecol. Environ.* 5 (10), 540–548. doi: 10.1890/060093

Link, J. S. (2010). *Ecosystem-based fisheries management: confronting tradeoffs* (Cambridge: Cambridge University Press), ISBN: .

Linke, S., and Bruckmeier, K. (2015). Co-Management in fisheries: experiences and changing approaches in Europe. *Ocean Coast. Manag* 104, 170–181. doi: 10.1016/j.ocecoaman.2014.11.017

Longo, C. S., Buckley, L., Good, S. D., Gorham, T. M., Koerner, L., Lees, S., et al. (2021). A perspective on the role of eco-certification in eliminating illegal, unreported and unregulated fishing. *Front. Ecol. Evol.* 9. doi: 10.3389/fevo.2021.637228

Macher, C, Steins, NA, Ballesteros, M, Kraan, M, Frangoudes, K, Bailly, D, et al. (2021). Towards transdisciplinary decision-support processes in fisheries: experiences and recommendations from a multidisciplinary collective of researchers. *Aquat. Living Resour.* 34, 13. doi: 10.1051/alr/2021010

Maes, J., Teller, A., Erhard, M., Condé, S., Vallecillo, S., Barredo, J. I., et al. (2021). *EU Ecosystem assessment: Summary for policymakers* (Luxembourg: Publications Office of the European Union), ISBN: . EUR 30599 EN. doi: 10.2760/190829

Malvarosa, L., Gambino, M., and Zander, K. (2021). *Analisi preliminare della catena del valore di due specie/prodotti di attività di pesca selezionate dell'Italia meridionale* (The Nisea Press). Available at: http://www.nisea.eu/dir/wp-content/uploads/2022/03/Analisi-ValueChain-Malvarosa.pdf.

Malvarosa, L., Murillas, A., Lehuta, S., Nielsen, J. R., Macher, C., Goti, L., et al. (2019). Sustainability impact assessment (SIA) in fisheries: Implementation in EU fishing regions. *Mar. Policy* 101, 63–79. doi: 10.1016/j.marpol.2018.11.039

Maravelias, C. D., Vasilakopoulos, P., and Kalogirou, S. (2018). Participatory management in a high value small-scale fishery in the Mediterranean Sea. *ICES J. Mar. Sci.* 75, 2097–2106. doi: 10.1093/icesjms/fsy119

Melnychuk, M. C., Veneziano, A., Lees, S., Rasal, J., Koerner, L. M., Hair, P., et al. (2022). Wild-caught fish populations targeted by MSC-certified fisheries have higher relative abundance than non-MSC populations. *Front. Mar. Sci.* 9. doi: 10.3389/fmars.2022.818772

MSC (2022) *The marine stewardship council annual report 2021-22.* Available at: https://www.msc.org/docs/default-source/default-document-library/about-the-msc/msc-annual-report-2021-2022.pdf.

Pita, C., Chuenpagdee, R., and Pierce, G. J. (2012). "Participatory issues in fisheries governance in Europe," In *Management of Environmental Quality: An International Journal*, vol. 23 (4), 347–361. doi: 10.1108/14777831211232209

Sabatella, E. C., Colloca, F., Coppola, G., Fiorentino, F., Gambino, M., Malvarosa, L., et al. (2017). Key economic characteristics of Italian trawl fisheries and management challenges. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00371

Sabatella, E. C., Gambino, M., Malvarosa, L., and Sabatella, R. F. (2021) *La catena del valore e il consumo consapevole di prodotti ittici sostenibili come strategie per fronteggiare la crisi del settore ittico.Sintesi dell'intervento tenuto al blue Sea land, 30 ottobre 2021. NISEA note 2021/1.* Available at: http://www.nisea.eu/dir/wp-content/uploads/2022/03/Analisi-ValueChain-Malvarosa.pdf.

STECF (2015). *Scientific, technical and economic committee for fisheries (STECF) – landing obligation - part 6 (Fisheries targeting demersal species in the Mediterranean Sea) (STECF-15-19) 2015* (Luxembourg: Publications Office of the European Union), ISBN: . doi: 10.2788/65549

STECF (2020). *Scientific, technical and economic committee for fisheries (STECF) – assessment of balance indicators for key fleet segments and review of national reports on member states efforts to achieve balance between fleet capacity and fishing opportunities (STECF-20-11). EUR 28359 EN* (Luxembourg: Publications Office of the European Union). doi: 10.2760/414107

STECF (2021). *Scientific, technical and economic committee for fisheries (STECF) – assessment of balance indicators for key fleet segments and review of national reports on member states efforts to achieve balance between fleet capacity and fishing opportunities (STECF-21-16). EUR 28359 EN* (Luxembourg: Publications Office of the European Union). doi: 10.2760/565659

STECF (2022a). *Scientific, technical and economic committee for fisheries (STECF) – assessment of balance indicators for key fleet segments and review of national reports on member states efforts to achieve balance between fleet capacity and fishing opportunities (STECF-22-15)* (Luxembourg: Publications Office of the European Union).

STECF (2022b). *Scientific, technical and economic committee for fisheries (STECF) - the 2022 annual economic report on the EU fishing fleet (STECF 22-06).* Eds. R. Prellezo, E. Sabatella, J. Virtanen and J. Guillen (Luxembourg: Publications Office of the European Union). doi: 10.2760/120462

Vielmini, I., Perry, A. L., and Cornax, M. J. (2017). Untying the Mediterranean Gordian knot: A twenty first century challenge for fisheries management. *Front. Mar. Sci.* 4. doi: 10.3389/fmars.2017.00195

Zander, K., and Feucht, Y. (2017). Consumers' willingness to pay for sustainable seafood made in Europe. *J. Int. Food Agribusiness Marketing* 3, 251–275. doi: 10.1080/08974438.2017.1413611

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EDITED BY
Nazli Demirel,
Istanbul University, Türkiye

REVIEWED BY
Ekin Akoglu,
Middle East Technical University, Türkiye
Paolo Vassallo,
University of Genoa, Italy

*CORRESPONDENCE
Ioannis Keramidas
keraioan@bio.auth.gr

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Ecotrophic perspective in fisheries management: a review of Ecopath with Ecosim models in European marine ecosystems

Ioannis Keramidas[1]*, Donna Dimarchopoulou[2,3], Eyal Ofir[4], Marco Scotti[5,6], Athanassios C. Tsikliras[1] and Gideon Gal[4]

[1]Laboratory of Ichthyology, Department of Zoology, School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece, [2]Department of Biology, Dalhousie University, Halifax, NS, Canada, [3]Department of Biology, Woods Hole Oceanographic Institution, Falmouth, MA, United States, [4]Kinneret Limnological Laboratory, Israel Oceanographic and Limnological Research, Migdal, Israel, [5]Marine Ecology Research Division, GEOMAR Helmholtz Centre for Ocean Research Kiel, Kiel, Germany, [6]Institute of Biosciences and Bioresources, National Research Council of Italy, Sesto Fiorentino, Firenze, Italy

The aim of this work is to present the food web models developed using the Ecopath with Ecosim (EwE) software tool to describe structure and functioning of various European marine ecosystems (eastern, central and western Mediterranean Sea; Black Sea; Bay of Biscay, Celtic Sea and Iberian coast; Baltic Sea; North Sea; English Channel, Irish Sea and west Scottish Sea; and Norwegian and Barents Seas). A total of 195 Ecopath models based on 168 scientific publications, which report original, updated and modified versions, were reviewed. Seventy models included Ecosim temporal simulations while 28 implemented Ecospace spatiotemporal dynamics. Most of the models and publications referred to the western Mediterranean Sea followed by the English Channel, Irish Sea and west Scottish Sea sub-regions. In the Mediterranean Sea, the western region had the largest number of models and publications, followed by the central and eastern regions; similar trends were observed in previous literature reviews. Most models addressed ecosystem functioning and fisheries-related hypotheses while several investigated the impact of climate change, the presence of alien species, aquaculture, chemical pollution, infrastructure, and energy production. Model complexity (i.e., number of functional groups) increased over time. Main forcing factors considered to run spatial and temporal simulations were trophic interactions, fishery, and primary production. Average scores of ecosystem indicators derived from the Ecopath summary statistics were compared. Uncertainty was also investigated based on the use of the Ecosampler plug-in and the Monte Carlo routine; only one third of the reviewed publications incorporated uncertainty analysis. Only a limited number of the models included the use of the ECOIND plug-in which provides the user with quantitative output of ecological indicators. We assert that the EwE modelling approach is a successful tool which provides a quantitative framework to analyse the structure and dynamics of ecosystems, and to evaluate the potential impacts of different management scenarios.

KEYWORDS

ecopath with ecosim, european marine ecosystems, ecological Indicators, food web modelling, meta - analysis

1 Introduction

Due to the complex nature of interactions within marine ecosystems, it is imperative to view and study them as a whole, complementing single-species assessments with ecosystem considerations (Marshall et al., 2019; Howell et al., 2021). The factors that compose marine ecosystems, as biological entities (fauna and flora) and environmental elements (e.g., nutrients), perform important processes either individually or synergistically (Borja et al., 2014; Armoškaitė et al., 2020). Worldwide, marine ecosystems have been facing a multitude of perturbations that threaten their structure and functioning (Elliott et al., 2015), with most important stressors being related to fisheries (Froese et al., 2018), climate change (Bruno et al., 2018) and non-indigenous species (Galil et al., 2018). Recent assessments demonstrate that marine ecosystems are being degraded or altered (Newton et al., 2020; Korpinen et al., 2021). Climate change has brought about rising temperatures that may trigger regime shift dynamics (Möllmann et al., 2021) while ocean acidification may further weaken the ecological resilience of European seas (Galdies et al., 2020). The introduction of alien species can affect entire ecosystems, especially in regional semi-enclosed seas such as the Baltic Sea (Dobrzycka-Krahel and Medina-Villar, 2020), the Black Sea (Shalovenkov, 2019) and the Mediterranean Sea (Zenetos et al., 2017). Much of the pressure also comes from anthropogenic activities; i.e., exploration and extraction of natural resources, transportation, infrastructure and pollution, including the discarding of fishing gears (Korpinen et al., 2021). In addition to natural resources removals, intensive bottom trawling physically damages the seabed by destroying habitats (Woods and Verones, 2019).

The European continent is characterized by great geomorphological complexity and diversity, both on land and at sea (Costello et al., 2006). A multitude of marine ecosystems exists, ranging from lower latitudes, with the closed and relatively warm Mediterranean Sea, to higher latitudes, with the eutrophic Baltic Sea and the Barents Sea, which is frozen during winter (Heiskanen et al., 2016). Marine ecosystems can be divided into two main categories, the deep seas and the shallow seas (Galparsoro et al., 2012). The deep seas (Mediterranean Sea and Black Sea) were created by geotectonic activity while the shallow seas (Adriatic Sea, Baltic Sea and North Sea) were created by the melting of glaciers covering large areas of land (Galparsoro et al., 2012). Moreover, there are the ecosystems related to the Atlantic Ocean (Bay of Biscay and Celtic Sea) on the western European coasts, which are directly affected by its geophysical and environmental conditions (Holt et al., 2014). European marine ecosystems are also distinguished by the many estuarine wetlands (Newton et al., 2014), which are important to local communities and exploited for aquaculture (Gamito and Erzini, 2005; Díaz López et al., 2008; Izquierdo-Gomez et al., 2016) and wind power plants (Halouani et al., 2020).

Software programs simulating ecosystem dynamics have facilitated the holistic study of marine ecosystems. Ecosystem modelling has provided a means for incorporating management perspectives (Fath et al., 2019); i.e., by implementing scenarios related to fisheries management, climatic and other environmental conditions (e.g., Scotti et al., 2022a), as well as the establishment of marine protected areas (MPAs; Dahood et al., 2020). Ecopath with Ecosim (EwE: Polovina, 1984; Christensen and Walters, 2004a) is the most widely used software tool for modelling marine food webs. It is suitable for examining the impacts of human and environmental stressors on the food web, as it is easy to use and offers a wide range of freely available modules and plug-ins (Pauly et al., 2000). It is based on the static depiction of a food web in a mass-balanced state over a specific time period (Ecopath) and uses differential equations for time simulations (Ecosim: Walters et al., 1997). Further extensions integrate temporal simulations with spatial dynamics and visualization (Ecospace: Walters et al., 1999), and support trophic conversion, configuring the food web as biomass movements from lower to higher trophic levels (EcoTroph: Gascuel and Pauly, 2009).

Ecological Network Analysis (ENA) is a toolkit of matrix manipulation techniques for the modelling of mass-balanced networks (Ulanowicz, 2004). Some ENA algorithms are available in EwE and enable calculating indicators summarizing ecosystem structure and functioning; the ENAtool routine also allows incorporating uncertainty levels in input data (Guesnet et al., 2015). Ecological indicators from ENA can be used to identify the state and health of the ecosystem (Heymans et al., 2016). This routine has been further expanded with a plug-in calculating standardised ecological indicators (ECOIND: Coll and Steenbeek, 2017). Additional plug-ins were established and incorporated into the main software program over the years (1) to assess parameters' uncertainty (Ecosampler: Steenbeek et al., 2018), and (2) to estimate concentrations and densities of contaminants and radioisotopes in aquatic environments (Ecotracer: Walters and Christensen, 2018). Other modules available in the base EwE version include (1) policy optimization of fisheries management (Christensen and Walters, 2004b), (2) spatial optimization of marine protected areas (Christensen et al., 2009), and (3) value chain modelling evaluating the socioeconomic benefits of fisheries (Christensen et al., 2011).

Reviews of EwE applications have previously been completed for the Mediterranean Sea (Coll and Libralato, 2012) and at global scale (Pikitch et al., 2014; Colléter et al., 2015; Coll et al., 2015; Vasslides et al., 2017; Craig and Link, 2023; Stock et al., 2023), highlighting that EwE has the lion's share in food web modelling with an increasing number of models through the years; i.e., over 430 models stored in an open-access database of EwE models called EcoBase (Colléter et al., 2015) and over 500 publications (Coll et al., 2015). The EcoBase initiative (http://ecobase.ecopath.org) is a collaborative project that aims at gathering published EwE models in a digital open-access repository where the models and their metadata are discoverable and easily accessible, with the goal of facilitating further meta-analyses and reviews (Colléter et al., 2015). However, since EcoBase is based on the principle of optional self-uploading by model developers, it does not cover all EwE models exhaustively, especially when it comes to modified versions of original models that examine new research items.

Our review aims to fill the gaps from the EcoBase repository by including all published models to date, original and updated versions, focusing on European seas' ecosystems, which are regulated by common mechanisms and rules for achieving the sustainable management of European fishing fleets and conservation of fish stocks. The EU Marine Strategy Framework Directive (MSFD; EU, 2008) and the revised EU Common Fisheries Policy (CFP; EU, 2013) represent relevant regulations for the management of the European seas. They recognise the need of an integrated understanding of food web characteristics, ecosystem functioning and fisheries management, an intent that can be facilitated by the EwE modelling approach. The main objective of this work was to review all published EwE models across European seas (Ecopath, Ecosim, Ecospace, and EcoTroph), aiming to (1) gather and present all EwE models to provide an overview of their main research topics, (2) discuss the hypotheses and objectives of each model by compiling their metadata, (3) identify the modelled areas, (4) highlight gaps in research, and (5) perform a comparative synthesis by assessing ecosystem health and resilience with ecosystem indicators derived from the models. To facilitate these objectives, we divided the EwE models according to nine major areas: (1) eastern Mediterranean Sea, (2) central Mediterranean Sea, (3) western Mediterranean Sea, (4) Black Sea, (5) Bay of Biscay, Celtic Sea and Iberian coast, (6) Baltic Sea, (7) North Sea, (8) English Channel, Irish Sea and west Scottish Sea, and (9) Norwegian and Barents Seas. The ECOIND plug-in (Coll and Steenbeek, 2017), which is integrated in the EwE software tool to calculate ecological indicators, was also included in the analysis.

2 European areas and overview of published EwE models

This review was based on an overview of all EwE models that have been developed for European marine ecosystems to date. At first, an inventory of the models stored in the EcoBase repository was made and metadata were cross-referenced with the corresponding publications (Tables S1, S2). An exhaustive and thorough search was then conducted using literature search engines (Scopus, Google Scholar and Web of Science) to find all available publications with EwE applications in peer-reviewed journals. This systematic search was executed following the PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses) methodology, which is based on article selection, screening and the extraction of information (Moher et al., 2010), and has been successfully applied in other review exercises (Corrales et al., 2020a). The starting year of eligibility was not defined but a cut-off date of available publications was set as to the first quarter of 2023. For a more efficient realisation of this task, keywords (e.g., "Ecopath", "Ecosim", "Ecospace", and "EcoTroph") were used in correspondence with the nine European areas and their sub-regions. The goal was to identify all subsequent publications that might be based on one original base model modifying or updating it, and that have not yet been included in EcoBase. Values of the indicators derived from the Ecopath summary statistics were compiled to infer ecosystem health and state, and for comparative analysis (Table 1). Finally, analyses of the publications and models were performed in relation to research topic (i.e., ecosystem functioning and ecological roles, fisheries, aquaculture, alien species, environment and climate) and software modules used; cumulative trends of EwE publications over time were highlighted. It is, however, noteworthy that several publications explored more than one topic. Moreover, forcing factors used to drive the spatio-temporal simulations with Ecosim and Ecospace were examined (Vasslides et al., 2017; Stock et al., 2023). Emphasis was also given to publications that used additional analyses based on the ECOIND plug-in to further demonstrate the utility of ecological indicators. These indicators are widely applied by scientists to characterize environmental and fishing impacts related to management (Coll and Steenbeek, 2017). A panel map of the nine European regions and the areas modelled was created to visualize EwE models and modules used. European areas and an overview of the models are described in the following sections.

2.1 Eastern Mediterranean Sea

The Levantine Sea is part of the eastern Mediterranean basin, which is strongly affected by increasing sea temperature and faces the greatest invasion of marine species in the world (Edelist et al., 2013). The warm, saline and ultra-oligotrophic ecosystem of the Levantine Sea results in very low primary production (Varkitzi et al., 2020). This heavily impacted ecosystem is exposed to climate change and multiple pressures and might benefit of using ecological and trophic models to understand its responses to concurrent stress

TABLE 1 Ecological indicators derived from the Ecopath module of the EwE software tool.

Indicator	Description	Behavior	Reference
Total System Throughput (TST)	Sum of all flows (consumption, exports, respiratory flows, flows to detritus)	Higher in systems with more compartments; it increases along the development of an ecosystem towards mature stages	Ulanowicz, 1986; Scotti et al., 2022b
Total primary production (PP)	Sum of primary productivity from all compartments	Higher in eutrophic or upwelling systems	Christensen and Walters, 2004a
Total PP/Total respiration	Ratio indicating maturity	Higher in immature systems (>1), affected with organic pollution	Odum, 1971
Total PP/biomass	Ratio indicating maturity	Declines over time in immature systems	Odum, 1971
Biomass/TST	Ratio indicating maturity	Higher in systems close to maturity	Odum, 1971
Net system production	Difference between total PP and total respiration	Higher in immature systems, closer to zero in mature systems	Christensen and Walters, 2004a
Respiration/biomass	Thermodynamic order function	Increases with biomass	Odum, 1971
Gross efficiency	Sum of fisheries catches to total PP	Higher in upwelling systems (catches of lower trophic level species), lower in systems with underexploited stocks	Marten and Polovina, 1982
Mean Trophic Level of the catch (MTLc)	Fisheries position in the food web	Higher in systems with fisheries targeting apex predators and high trophic-level consumers	Pauly et al., 1998
Connectance	Ratio of actual to possible food web links	Higher in systems with higher number of functional groups	Nee, 1990
System Omnivory index	Distribution of trophic interactions in the food web	Larger than zero in systems where functional groups feed on multiple trophic levels	Pauly et al., 1993
Niche overlap	Index describing response to the distribution of resources	Prey and predator interactions used to identify possible aggregations of functional groups	Pianka, 1973
Particle size distribution	Index of growth (ratio of biomass to weight)	Higher in systems with higher biomass estimates of larger species	Sheldon et al., 1972
Ascendency	Index describing resilience	Decreases in systems with low internal constraints (i.e. with a high resilience) and increases with energy transfer efficiency	Ulanowicz, 1986; Scotti et al., 2022b
Finn Cycling index	Index describing maturity	Higher in systems displaying the capacity to reuse energy/matter	Finn, 1976
Trophic aggregation	Assembly of the Lindeman spine	It enables quantifying the trophic transfer efficiency between discrete trophic levels	Lindeman, 1942; Ulanowicz and Kemp, 1979
Primary production required to sustain fisheries (PPR)	Ratio describing fisheries sustainability	It exhibits higher values in upwelling and shelf ecosystems	Pauly and Christensen, 1995
Mixed trophic impact	Impact of interactions between functional groups	Pairwise summary of net impacts between functional groups in response to biomass changes (it also includes fisheries impact)	Ulanowicz and Puccia, 1990
Keystoneness	Key groups of the system	It helps identifying functional groups with structuring role and considerable ecosystem impact despite their relatively low biomass	Libralato et al., 2006; Valls et al., 2015

Description, behavior and reference are provided for each indicator and those that are part of the included ENA toolkit routine are in bold.

factors and anthropogenic activities (Albouy et al., 2013; Corrales et al., 2018; Shabtay et al., 2018; Grossowicz et al., 2020). A large part of the Levantine Sea, the Israeli Mediterranean coast, was modelled with Ecopath (Corrales et al., 2017a), Ecosim (Corrales et al., 2017b; Corrales et al., 2018) and Ecospace (Ofir et al., submitted)[1] to determine the impacts that alien species, sea warming and fisheries have on the ecosystem. The consequences of alien species and fisheries have also been studied in other parts of the Levantine basin such as the insular shelf ecosystem of Cyprus (Michailidis et al., 2019; Michailidis et al., 2023) and the Gulf of Mersin in Turkey (Saygu et al., 2020). Moreover, EwE was applied to model the effects of aquaculture (Livne et al., 2020), to inform marine spatial planning (Shabtay et al., 2018), and to assess energy- (Grossowicz et al., 2020) and economy-related (Peled et al., 2020; Michael-Bitton et al., 2022) questions.

The Aegean Sea is a relatively closed area in the western part of the eastern Mediterranean Sea, located between Greece and Turkey, with numerous islands and islets (Keramidas et al., 2022). It is one of the most productive areas of the eastern Mediterranean basin, especially its northern part (Tsiaras et al., 2012). It is, however, facing the threats of increasing temperature and alien species

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[1] Ofir, E., Corrales, X., Coll, M., Heymans, J. J., Goren, M., Steenbeek, J., et al. (under review). Evaluation of fisheries management policies in the alien species-rich Eastern Mediterranean under climate change. *Front. Mar. Sci.*

invasion, although at a lesser extent than the Levantine Sea (Katsanevakis et al., 2020). In the north, the inflow of nutrient-rich and lower salinity waters originating from the Black Sea and large rivers (Lykousis et al., 2002) supports large biomasses of commercially important small and medium pelagic fishes. This complex interplay between stress factors and biodiversity stimulated the implementation of trophic models to assess the state of the food web and the impact of fisheries (Tsagarakis et al., 2010; Papapanagiotou et al., 2020; Tsagarakis et al., 2021; Tsagarakis et al., 2022). Ecopath and Ecosim models were also developed at a regional scale to assess trophic interactions and fisheries impacts in Thermaikos Gulf (Dimarchopoulou et al., 2022), Pagasitikos Gulf (Dimarchopoulou et al., 2019), and Saronikos Gulf (Papantoniou et al., 2021). Moreover, an Ecopath model was built to examine the effects that fish overexploitation, due to commercial fisheries in Greece and Turkey, has on the whole Aegean ecosystem (Keramidas et al., 2022).

2.2 Central Mediterranean Sea

The Adriatic Sea is a body of water in the northernmost part of the Mediterranean Sea, which separates the Italian Peninsula from the Balkans (Kourafalou, 1999). It is divided into two basins, the northern being the shallowest and the southern being the deepest (Russo and Artegiani, 1996). The Adriatic's salinity is lower than in other Mediterranean basins because the Adriatic collects a third of the freshwater flowing into the Mediterranean Sea thus acting as a dilution basin (Russo and Artegiani, 1996). The northern and central parts of the Adriatic Sea were studied extensively over past years with Ecopath and Ecosim because of their ecosystem characteristics and commercial fisheries concentration (Coll et al., 2007; Barausse et al., 2009; Coll et al., 2009a; Pranovi and Link, 2009; Coll et al., 2010; Libralato et al., 2010; Libralato et al., 2015; Celić et al., 2018); spatial-explicit versions were constructed using Ecospace (Fouzai et al., 2012; Steenbeek et al., 2013). Two Ecopath models, on the microbenthic loop (Vassallo et al., 2006) and the benthopelagic fauna (Vassallo et al., 2017) were developed in southern Adriatic coasts, while an Ecopath model was built for the Strait of Sicily to evaluate the extent of bentho-pelagic coupling in the food web (Agnetta et al., 2019).

In the eastern Ionian Sea, EwE models were built to investigate ecosystem functioning (Piroddi et al., 2016), fisheries (Piroddi et al., 2010; Piroddi et al., 2011a; Moutopoulos et al., 2013; Moutopoulos et al., 2018) and aquaculture (Piroddi et al., 2011b). Four Ecopath models were constructed for the areas of Salento (northeastern sector) and Calabria (southwestern sector) to characterize ecosystem functioning (Ricci et al., 2019; Ricci et al., 2021), while one model of the Gulf of Taranto was assembled to assess interactions between cetaceans and commercial fisheries (Carlucci et al., 2021). Outside of the European waters, an Ecopath model was developed for the Gulf of Gabes, Tunisia (Hattab et al., 2013). It was extended to implement spatio-temporal simulations (Abdou et al., 2016; Halouani et al., 2016; Abdou et al., 2020) and trophic analysis (Halouani et al., 2015). These models, even though were developed for the northern African coasts, were included in the review because they are located in the central Mediterranean Sea and adjacent to European waters.

2.3 Western Mediterranean Sea

The Balearic Sea is part of the wider north-west Mediterranean Sea and is considered to be the most productive area of the otherwise oligotrophic Mediterranean Sea (D'Ortenzio and Ribera d'Alcalà, 2009). It is a breeding ground for small pelagic fishes such as the European anchovy (*Engraulis encrasicolus*). Moreover, it represents a major fishing ground for commercial fisheries and a highly exploited ecosystem (Papaconstantinou and Farrugio, 2000). The south Catalan Sea is in the northern part of the Balearic archipelago and is the most modelled area of the Mediterranean Sea (Coll and Libralato, 2012). It has been extensively modelled using EwE, either considering a static framework (Ecopath: Coll et al., 2006a; Coll et al., 2006b; Coll et al., 2009b; Navarro et al., 2011) or implementing temporal (Ecosim: Coll et al., 2008a; Coll et al., 2008b; Coll et al., 2012; Forrestal et al., 2012; Coll et al., 2013; Tecchio et al., 2013) and spatiotemporal (Ecospace: Coll et al., 2016a; Pennino et al., 2020) simulations. An Ecopath model that describes food web traits and resilience to fisheries exploitation was developed for the wider area of the northwestern Mediterranean Sea (Corrales et al., 2015). Smaller-scale models were constructed for other regions of the western Mediterranean Sea: the Gulf of Lions (Bănaru et al., 2013; Vilas et al., 2021; Seyer et al., 2023), the Gulf of Alicante (García-Rodríguez et al., 2021), the Santa Pola Bay (Bayle-Sempere et al., 2013; Izquierdo-Gomez et al., 2016), the Cerbère-Banyuls (France), Cap de Creus (Spain) and Medes Islands (Spain) MPAs (Corrales et al., 2020b), and the Port Cros; case studies integrating the EcoTroph plug-in are also available (Valls et al., 2012; Prato et al., 2014). Further east, in the wider areas of the Tyrrhenian and Ligurian seas, EwE models were built focusing on fisheries (Pinnegar and Polunin, 2004; Albouy et al., 2010; Vanalderweireldt et al., 2022), aquaculture (Díaz López et al., 2008),MPAs (Prato et al., 2016) and the microbenthic loop (Fabiano et al., 2004; Vassallo et al., 2012; Vassallo et al., 2013; Vassallo et al., 2022).

2.4 Black Sea

The Black Sea is a marginal sea located at the northeast of the Mediterranean Sea, which is connected with the Aegean Sea through the Sea of Marmara. It is the world's largest body of water with a meromictic basin as the deep waters do not mix with the upper layers that receive oxygen from the atmosphere (Sabatino et al., 2020). As a result, over 90% of the deeper Black Sea volume is anoxic and water circulation is primarily controlled by basin topography and fluvial inputs, which result in a strongly stratified vertical structure (Reeburgh et al., 1991). In the previous decades, significant trophic transformations occurred in the basin, with the most remarkable being the introduction and extreme expansion of the warty comb jelly (*Mnemiopsis leidyi*), an invasive ctenophore from the western Atlantic. In particular, the outburst of *M. leidyi* in the late 1980s was facilitated by the simultaneous action of overfishing, climate and nutrient enrichment (Bodini et al., 2018). The abrupt increase in the size of *M. leidyi* population led to a dramatic drop in commercially important small pelagics like the European anchovy due to competition for the same food sources (mostly copepods) and intraguild predation on its eggs (Kideys, 2002).

Two EwE models were created for the Black Sea, focusing on fishing effects on the food web along a timeline of 30 years, starting from the 1960s (Daskalov, 2002), and assessing the relationships that later linked overfishing to the successful establishment of the warty comb jelly (Gucu, 2002). Moreover, for better understanding food web dynamics, four Ecopath models of the Black Sea were constructed. These models focus on reference periods that match significant structural changes in the food web: (1) the pre-eutrophication period of 1960s, (2) the intense eutrophication phase of 1980s, (3) the regime shift of early-1990s with warty comb jelly displacing the European anchovy, and (4) the post-eutrophication period of the late-1990s with increasing top-down control of another invader, i.e., the brown comb jelly (*Beroe ovata*) over *M. leidyi* (Akoglu et al., 2014). Finally, three Ecopath models were recently developed in the Sea of Marmara, the connecting water body between the Black Sea and the Aegean Sea, investigating fisheries impacts in the food web during the period of three decades (Saygu et al., 2023).

2.5 Bay of Biscay, Celtic Sea and Iberian coast

The Bay of Biscay is a relatively large and deep gulf of the northeast Atlantic Ocean, located south of the Celtic Sea. It is considered a very productive area due to the upwelling characteristics and the drainage waters of large rivers (Borja et al., 2008). It displays a high biodiversity with many habitats and marine species, including endangered and protected species like cetaceans and seabirds (Matear et al., 2019). It faces intense human pressure, including industrialised commercial fisheries from France and Spain, and significant marine transportation (Lorance et al., 2009). EwE models were developed for the Bay of Biscay/Celtic Sea ecosystem (Bentorcha et al., 2017), the Bay of Biscay as a whole (Moullec et al., 2017; Corrales et al., 2022), including a specific focus on its continental shelf only (Lassalle et al., 2011; Lassalle et al., 2012; Lassalle et al., 2014; Le Marchand et al., 2022), and the Celtic Sea (Hernvann et al., 2020). The southern part of the Bay of Biscay, the Cantabrian Sea, is comparatively less studied, with an Ecopath model developed for 1994 to assess fisheries impacts on the food web of the continental shelf (Sánchez and Olaso, 2004). A single Ecopath model was developed describing the trophic network of the Gironde Estuary for the late-1990s/early-2000s (Lobry et al., 2008). The Gironde Estuary is one of the largest and least-polluted estuaries in northern Europe (Sautour and Castel, 1995) and its basin is connected with the Bay of Biscay.

South of the Bay of Biscay and Celtic Sea lies the west coast of the Iberian Peninsula. An Ecopath model was developed for the Gulf of Cadiz, located at the southern part of the west coast, investigating the trophic relationships and fishing impacts in an exploited ecosystem (Torres et al., 2013). Further north, EwE models were constructed for the Portuguese and Galician coasts, focusing on food web structure and ecological roles (Veiga-Malta et al., 2019; Giralt Paradell et al., 2020; Giralt Paradell et al., 2021), small pelagic fisheries (Szalaj et al., 2021) and climate (Szalaj et al., 2022). Smaller parts of the west coast of the Iberian Peninsula and lagoon ecosystems were modelled separately; namely, the Ria Formosa (Gamito and Erzini, 2005), the Mondego Estuary (Patrício and Marques, 2006; Baeta et al., 2011), the Ria de Aveiro (Bueno-Pardo et al., 2018) and the Ria de Arousa (Outeiro et al., 2018). In the southernmost part of the NE Atlantic is the Azores archipelago, composed of nine islands under Portuguese jurisdiction, where an Ecopath model was developed to describe the food web and vulnerabilities of an open-ocean ecosystem (Morato et al., 2016). Finally, a theoretical seamount ecosystem of the NE Atlantic was used as the basis for a trophic model to examine the potential effects of an increase in primary productivity (Morato et al., 2009).

2.6 Baltic Sea

The Baltic Sea is a marginal sea of the northeastern Atlantic Ocean, with limited water exchange between the two water bodies (Barale, 2008). It is considered one of the largest brackish water bodies in the world and its salinity is significantly lower than that of ocean waters (Vuorinen et al., 2015), defining its inherent low biodiversity (Ojaveer et al., 2010). As saltwater is denser than freshwater, the bottom of the Baltic Sea is saltier than the surface (Sohlenius et al., 2001). This condition creates vertical stratification of the water column, i.e., a halocline, which represents a barrier to the exchange of oxygen and nutrients and fosters completely separate marine environments (Väli et al., 2013). The Baltic Sea has been exposed to the simultaneous action of multiple interacting pressures, a feature that qualifies this ecosystem as an ideal site to predict the impact and formulate mitigation of future coastal perturbations (Reusch et al., 2018). As the Baltic proper is subject to most of the human pressures, such as fishing, nutrient pollution and climate change, scientific research and development of ecosystem models in the Baltic Sea have been concentrated mostly there (Korpinen et al., 2022; Reckermann et al., 2022). Several EwE models have been developed in this area, including Ecopath (Sandberg et al., 2000; Harvey et al., 2003; Tomczak et al., 2012; Tomczak et al., 2013), Ecosim (Österblom et al., 2007; Niiranen et al., 2013; Costalago et al., 2019) and Ecospace (Bauer et al., 2018; Bauer et al., 2019) applications, focusing mostly on primary production fluctuations and climate change impacts on the food webs. Recently, an EwE model of the western Baltic Sea was constructed to assess the impact that alternative fisheries management strategies have on the status of commercial stocks, diversity of top predators and fishing yield (Scotti et al., 2022a). At a smaller scale, an Ecopath model was developed for an open coastal area in the southwestern part of the Bothnian Sea to evaluate radioactive fluxes in the food web using the Ecotracer tool (Sandberg et al., 2007). Finally, separate Ecopath models were created for five southeastern Baltic Sea ecosystems to compare the structure of their carbon flow networks: the Puck Bay, the Curonian Lagoon, the Lithuanian coast, the coastal area of the Gulf of Riga and the Pärnu Bay (Tomczak et al., 2009).

2.7 North Sea

The North Sea is a shelf sea on the European continental shelf, connected to the Atlantic Ocean through the English Channel in the south and the Norwegian Sea in the north (Lüdmann et al., 2021). It is a relatively shallow sea, with a mean depth of 90 meters (Lee, 1980). Due to the large human population and high level of industrialization along its shores, the wildlife of the North Sea has suffered from pollution, overhunting, and overfishing, with most of the commercial fisheries stocks considered to be either fully exploited or overexploited (Akbari et al., 2022). The North Sea is Europe's main fishing ground, accounting for over 5% of international commercial fish caught, concentrated mostly in the southern part of the coastal waters (Mackinson et al., 2018). An Ecopath model representing the total area of the North Sea was created for 1981, known as the '*Year of the Stomach*', when more than 55,000 fish were sampled and analysed for their stomach contents through an ICES-coordinated project (Christensen, 1995a). Other EwE models were developed for different periods (Beattie et al., 2002; Mackinson and Daskalov, 2007; Heymans et al., 2011a; Romagnoni et al., 2015) while an Ecosim model of the same area, using multiple environmental drivers, served as the basis for multi-annual plans regarding fisheries and management in the North Sea (Mackinson, 2014). An EwE model was constructed for the southern part of the North Sea to focus on fisheries management (Stäbler et al., 2016; Stäbler et al., 2018); the model was then used as a backbone to develop an Ecospace application integrating habitat preferences (Püts et al., 2020; Püts et al., 2023). In the western part of the North Sea, Ecopath models were developed for the northeastern coast of Scotland (Otogo et al., 2015) and the Eden Estuary (Watson et al., 2020).

2.8 English Channel, Irish Sea and west Scottish Sea

The English Channel is a strait of the Atlantic Ocean that separates southern England from northern France, and is linked to the southern part of the North Sea by the Strait of Dover (Smith, 1989). It is the busiest shipping area in the world (Glegg et al., 2015) and has been exposed to environmental problems caused by accidents involving ships with toxic cargos and oil spills (Vieites et al., 2004). Despite the intense marine traffic, it remains an important ecosystem for marine species that inhabit its numerous bays and estuaries, especially in the western part, including marine mammals such as the grey seal (*Halichoerus grypus*) and the harbour seal (*Phoca vitulina*; Vincent et al., 2017). Separate EwE models were developed for the western part (Araújo et al., 2005; Araújo et al., 2006; Araújo et al., 2008) and the eastern part (Villanueva et al., 2009; Daskalov et al., 2011; Metcalfe et al., 2015). An area of the eastern English Channel, the Bay of Seine, has been extensively modelled with EwE because of its ecological characteristics and multiple anthropogenic disturbances. Several models have been constructed giving insights on food web structure (Rybarczyk and Elkaïm, 2003; Tecchio et al., 2015), energy facilities disturbance (Pezy et al., 2017; Raoux et al., 2017; Raoux et al., 2019; Halouani et al., 2020; Noguès et al., 2022) and climate change (Bourdaud et al., 2021). At a smaller scale, food web models were created for the Bay of Somme (Rybarczyk et al., 2003), the Bay of Mont Saint Michel (Arbach Leloup et al., 2008), the Tamar Estuary (Watson et al., 2020) and the Canche Estuary (Selleslagh et al., 2012)

Further north, the continental shelf of the west coast of Scotland, including the waters around the islands in the Sea of the Hebrides, has been extensively modelled with Ecopath (Alexander et al., 2015), Ecosim (Heymans et al., 2011b; Serpetti et al., 2017; Baudron et al., 2019) and Ecospace (Sayer et al., 2005; Alexander et al., 2016; Serpetti et al., 2021). The Irish Sea is an extensive body of water that separates the islands of Ireland and Great Britain and is linked to the sea of the west coast of Scotland in the north by the North Channel, engulfing numerous estuaries with great biodiversity. However, it is considered to be one of most radioactively contaminated seas in the world with high amounts of nuclear waste, like ^{137}Cs, Pu, and ^{241}Am, discharged daily into it from the Sellafield nuclear site (Ray et al., 2020). This discharge contaminates seawater, sediments and marine life, and several EwE models were developed to address these issues (Tierney et al., 2018; Bentley et al., 2019a; Bentley et al., 2019b; Bentley et al., 2019c; Bentley et al., 2020). Finally, an Ecopath model was built for the Faroe Islands marine ecosystem in 1997 and extended with spatio-temporal simulations to 2007 for modelling spatial fishing restrictions (Zeller and Reinert, 2004).

2.9 Norwegian and Barents Seas

The Norwegian Sea is a marginal sea of the Atlantic Ocean, located northwest of Norway and north of the North Sea. Unlike the majority of the world seas, most of the bottom of the Norwegian Sea is not part of a continental shelf and therefore it is characterised by deep waters (Sætre, 1999). It is in fact a transition zone between Boreal and Arctic conditions, and is thus inhabited by biodiversity of both climatic regions, including many seabirds and marine cetaceans (Jørgensen et al., 2022). In the northeast, its waters are linked to the Barents Sea, which is located in the southern part of the Arctic Ocean (Johannesen et al., 2012). Contrarily to the Norwegian Sea, the Barents Sea is a relatively shallow shelf sea (Gudlaugsson, 1993). The Barents Sea contains the world's largest remaining population of Atlantic cod (*Gadus morhua*) while the Norwegian Sea is one of the most important spawning grounds of the Atlantic herring (*Clupea harengus*), defining them as very important commercial fishing grounds (Ottersen et al., 2014; Pampoulie et al., 2015). EwE models were developed to describe these systems, focussing on the Norwegian Sea and Barents Sea ecosystem as a whole (Dommasnes et al., 2001; Bentley et al., 2017), on the Barents Sea ecosystem (Blanchard et al., 2002; Megrey and Aydin, 2009; Pedersen et al., 2021; Pedersen, 2022) and the Pechora Sea (Larsen et al., 2016). The coastal areas of the Norwegian Sea, known as fjords, are unique and important and have been modelled using EwE to study the Arctic ecosystem functioning (Pedersen et al., 2008; Pedersen et al., 2016), the red king crab (*Paralithodes camtschaticus*) invasion (Pedersen et al., 2018), and ocean warming impacts on kelp forests (Vilas et al., 2020a; Vilas et al., 2020b).

3 Results

Overall, 195 Ecopath models, 70 Ecosim models, 28 Ecospace models and 8 EcoTroph models from the European seas were extracted from 168 peer-reviewed journal publications (Figure 1). Altogether, a total of 301 EwE models were reviewed and analysed, including original models (n=212, 70%), their updated versions with the inclusion of Supplementary Data and information (n=32, 11%), and variant versions with modifications of the original or updated models according to the research subject and needs (n=57, 19%). While the models covered nine different regions, most publications referred to the western Mediterranean Sea (n=31, 18%) while the Black Sea had the fewest (n=4, 2%). More than half of all publications dealt with the investigation of food web structure, functioning and ecological roles (35%), and fisheries (28%; Figure 2). The majority of the models were mass-balanced Ecopath base models, most of which were in the western Mediterranean Sea (n=37, 19%), followed by the Bay of Biscay, Celtic Sea and Iberian coast region (n=35, 18%). In the Mediterranean Sea, the western region was described by most models (n=57, 46%) and publications (n=31, 40%), and is followed by central (n=39, 31% and n=27, 35%) and eastern regions (n=29, 23% and n=19, 25%). The cumulative number of EwE models increased with time, with the highest rate of increase observed for Ecopath models (Figure 3). Almost half of the models were published in the 2010s (49%), with a high number of publications appearing after 2020 (28%). The number of functional groups (FGs) included in the models exhibited similar patterns, with an increasing trend (i.e., higher resolution in food web construction) over time; their number ranged from 8 up to 108, although most of the models (32%) were composed of 31 to 40 FGs (Figure 4). Regarding forcing factors driving spatio-temporal dynamics and used for hindcasting, most of them were related to trophic interactions with the estimation of vulnerabilities (74%), fisheries (74%) and primary production (61%) (Figure 5). The bulk of Ecopath models describes trophic networks in the 1990s (n=58, 30%) and 2000s (n=67, 34%) while no model has a starting year before 1950. Ecosim models had a simulation timeframe reaching up to 150 years, monitoring biomass, fisheries and climate changes over a two-century period while Ecospace displayed a shorter simulation timeframe; i.e., less than 100 years, because of its spatial nature and resulting data and computational requirements, demanding longer simulation runs (Walters et al., 1999).

3.1 Ecosystem indicators and summary statistics

Ecological indicators obtained from Ecopath summary statistics for the models in the nine European marine regions are given as weighted averages for each area, including the minimum and maximum range limits (Table 2). These indicators were scaled and visualised as radar charts for comparisons (Figure 6). The largest average Ecopath modelling area was found for the Norwegian-Barents Sea region (almost 1 million km^2) while the smallest was that of the western Mediterranean Sea region (below 5,000 km^2), indicating the presence of many smaller models. A pattern of increasing modelling area with decreasing latitude was observed. The Norwegian-Barents Sea and the North Sea regions had the highest number of FGs (47 each), followed by the western

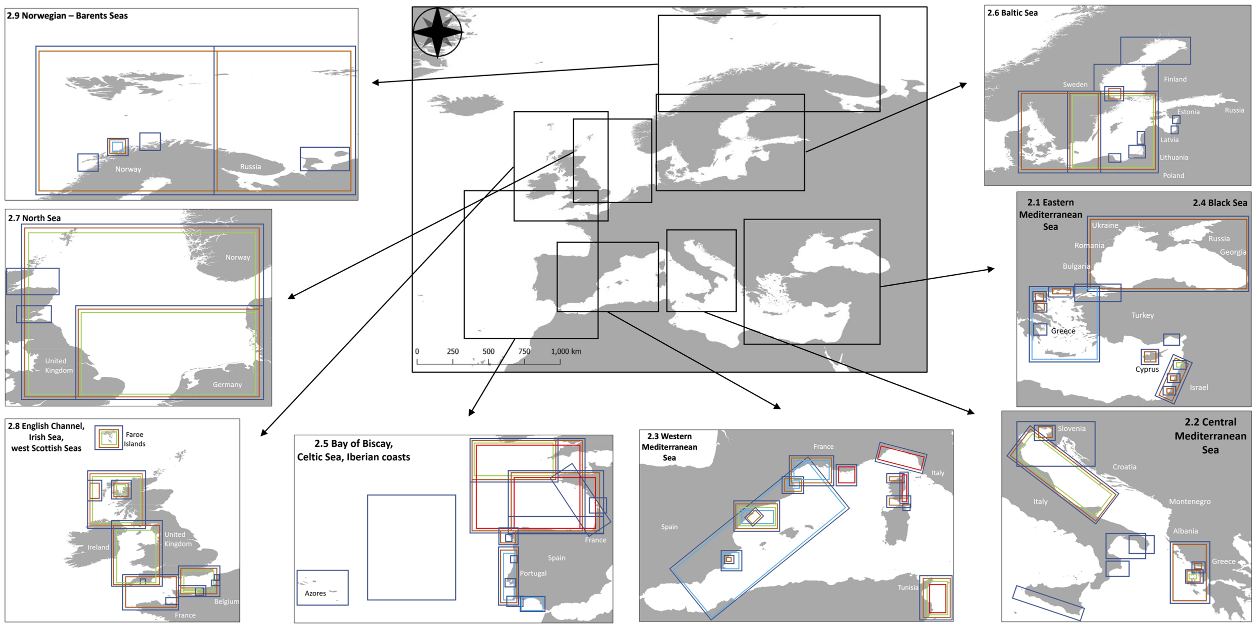

FIGURE 1
Map showing the nine European marine regions considered in this review. Colored rectangles encompass the approximate area modelled with EwE in all nine regions, including Ecopath models (blue rectangles), Ecosim models (orange rectangles), Ecospace models (green rectangles), EcoTroph models (red rectangles), and ECOIND plug-in expansions (light blue rectangles).

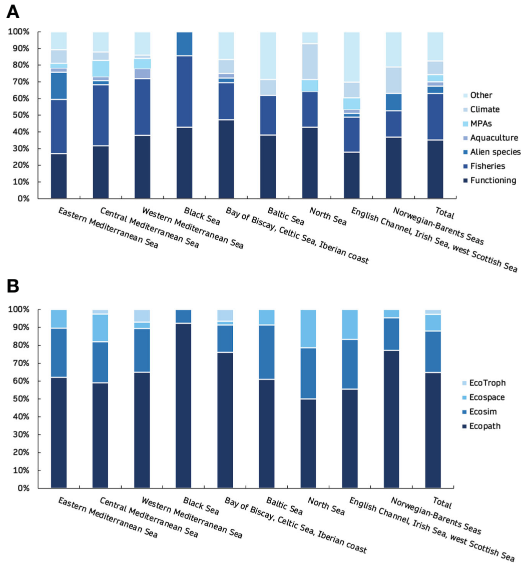

FIGURE 2
Percentage (y-axis) of EwE model publications in peer-reviewed journals classified by topic **(A)** and by EwE module **(B)** for the nine European marine regions.

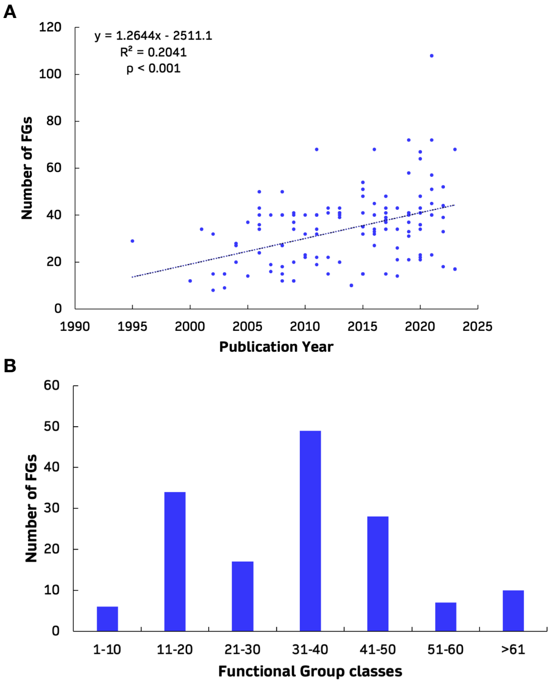

FIGURE 4
Number of functional groups (y-axis) in relation with time of Ecopath models **(A)** and divided by size classes **(B)** in all nine reviewed European marine regions. The dotted line illustrates the trend identified with a linear model, indicating a correlation with highly statistical significance (p value < 0.001).

Mediterranean Sea (41); the Black Sea had the fewest (13) and total average number of FGs was 34.

The Total System Throughput (TST) expresses the sum of all system's flows (total consumption + total export + total respiration + total flows to detritus), representing the size of the system (Ulanowicz, 1986). The North Sea region had the highest average TST (10,285.1 tkm^{-2}year^{-1}) while the eastern Mediterranean Sea had the lowest (1,767.8 tkm^{-2}year^{-1}), a pattern that is in line with expectation as TST depends on the number of flows in a network (Scharler et al., 2015). The Bay of Biscay, the Celtic Sea and Iberian coast had the highest total primary production, followed by other upwelling or eutrophic areas like the North Sea and the west Scottish Sea. Ecosystem maturity can be expressed as the ratio of total primary production over total respiration (Odum, 1971); the less mature ecosystems were in the Baltic Sea region (8.48) while those closest to maturity were in the Norwegian-Barents Sea region (1.01). In less mature systems the production exceeds the respiration for most groups, leading to ratios greater than 1 (Odum, 1971). Average estimates of total biomass (excluding detritus) were higher in the Bay of Biscay, Celtic Sea and Iberian coast models (851.1 tkm^{-2}) while those in the Black Sea had the lowest values (22.7 tkm^{-2}). Regarding commercial fisheries, the ecosystems with most catches (landings plus discards) were observed in the Bay of Biscay, Celtic Sea and Iberian coast region (14.1 tkm^{-2}year^{-1}) following biomass trends, while lowest levels of fish extraction were found in the Norwegian-Barents Sea region (0.08 tkm^{-2}year^{-1}). Mean trophic level of the catch (MTLc) provides clues about the effect of fisheries on the ecosystem; it is calculated as the weighted average trophic level of the species removed, with their relative biomass contribution to the catch used as a scaling factor (Shannon et al., 2014). A decreasing trend of this indicator in time relates to the progressive overexploitation of fish stocks along the trophic chain, from top predators to lower trophic-level consumers (Pauly et al., 1998). The Bay of Biscay, Celtic Sea and Iberian coast

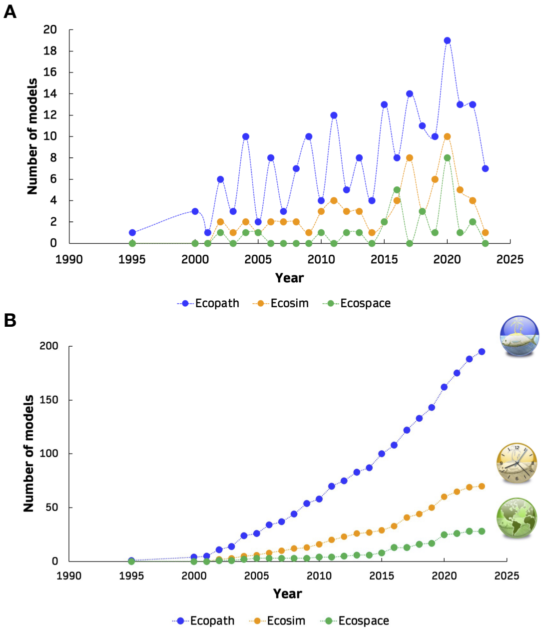

FIGURE 3
Absolute **(A)** and cumulative **(B)** annual number of Ecopath (blue), Ecosim (orange) and Ecospace (green) models in peer-reviewed publications and referring to European waters since 1995.

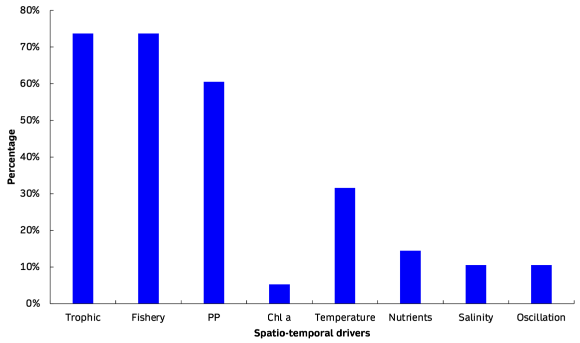

FIGURE 5
Forcing factors used as drivers for spatio-temporal simulations (i.e., Ecosim, Ecospace) in the calibration and fitting procedures; the y-axis informs on their percentage of occurrence.

ecosystems had the highest average MTLc scores (3.59), indicating the presence of large predatory species in the catch while central Mediterranean, English Channel, Irish Sea and west Scottish Sea regions had the lowest average scores (<3), which represents highly exploited fish stocks. The System Omnivory index (SOi) is a measure of the distribution of feeding interactions among trophic levels, developed for complexity and connectivity evaluations in the food web (Pauly et al., 1993). The North Sea region had the highest SOi (0.24), indicating that various trophic groups obtain energy from trophic chains of different lengths. However, all models were characterised by low values compared to temperate ecosystems close to the tropics (Christensen, 1995b).

3.2 Ecological indicators of the ECOIND plug-in

A series of algorithms enabling the calculation of various ecological indicators was added to EwE with the ECOIND plug-in (Coll and Steenbeek, 2017). This plug-in returns standardised indicators incorporating a series of biological traits and features, and provides useful tools for further analysis of the ecosystem, in both static and spatio-temporal dynamics. Due to the relatively recent integration of ECOIND into EwE, it was not widely used in the EwE models reviewed here (Table S3). ECOIND results can be extracted adopting a static (Coll and Steenbeek, 2017; Vilas et al., 2020a; García-Rodríguez et al., 2021; Keramidas et al., 2022) or a spatio-temporal dynamic (Bentley et al., 2019a; Corrales et al., 2020b; Hernvann et al., 2020; Vilas et al., 2020b; Vilas et al., 2021; Le Marchand et al., 2022; Piroddi et al., 2022; Szalaj et al., 2022) perspective.

The ECOIND plug-in can facilitate the comparison between ecosystems with respect to food web structure and functioning (Coll and Steenbeek, 2017). In the latest ECOIND edition, indicators are classified into five categories: (1) biomass, (2) catch, (3) trophic, (4) species, and (5) size-based. Various assessments related to the MSFD have already been successfully implemented with ecological indicators derived from models of various types (e.g., biogeochemical, hydrodynamic, multi-species, end-to-end) in a number of software programs, including EwE (Lynam and Mackinson, 2015; Susini and Todd, 2021), with encouraging results (Piroddi et al., 2015a). There is a multitude of ecological indicators, which can also inform ecosystem-based fisheries management (EBFM), that have been used to successfully address fisheries (Fay et al., 2013; Lassen et al., 2013; Halouani et al., 2019; Link and Watson, 2019) and climate (Shin et al., 2018) issues. As the stressors affecting marine ecosystems increase, so is the need for testing and validating ecological indicators (Rombouts et al., 2013; Ito et al., 2023). A key issue facing the scientific community is the identification of indicators that are the most suitable to effectively describe the status of ecosystems, helping decision-making bodies (Safi et al., 2019). EwE is based on an interactive approach within the scientific community and ECOIND represents a quick and user-friendly tool for assessments based on ecological indicators (Coll et al., 2016b).

4 Discussion

This review of all published Ecopath with Ecosim (EwE) models of European seas aims to serve as an easily accessible and quick guide for researchers working on ecosystem modelling. This centralised aggregation of European EwE models and their metadata highlights the evolution of models and research topics over time and across space. European seas and the NE Atlantic have been extensively modelled, compared to other regions like the Indian and Antarctic Oceans. Also, the Mediterranean Sea has been modelled as a whole ecosystem with two Ecopath models representing the 1950s and the 2000s, sub-divided into four separate sections: western, Adriatic, Ionian and eastern region (Piroddi et al., 2015b). These sub-models were simulated temporally (Piroddi et al., 2017) with Ecosim to demonstrate the effects of climate change and fisheries, and spatiotemporally with Ecospace (Piroddi et al., 2022) to inform ecosystem-based management (EBM). Scientific projects with adequate funding schemes and data availability can support this trend in European regions (Colléter et al., 2015). However, there are still some European marine ecosystems with no existing models; this is the case of several Mediterranean countries (e.g., Malta, Albania and Montenegro) and the southern part of the Norwegian Sea. As many models assessed by this review are not in EcoBase, we strongly encourage the scientific community to use this repository to store metadata of EwE models and take advantage of this large network of users (Colléter et al., 2015). However, the existence of a quality control evaluating the models stored in the EcoBase, similar with the Ecopath pedigree routine, including also Ecosim and Ecospace models (e.g., a quality measure with a traffic light system), would be beneficial for scientists implementing various meta-analyses. Ecopath models are the most numerous because they are a prerequisite for further developing temporal (Ecosim) and spatial (Ecospace) simulations (Christensen and Walters, 2004a). As several Ecopath models have been already developed in the European seas, a good future suggestion can be their temporal or

TABLE 2 Summary statistics of Ecopath models in European areas of interest.

Parameter	Units	2.1 Eastern Mediterranean Sea	2.2 Central Mediterranean Sea	2.3 Western Mediterranean Sea	2.4 Black Sea	2.5 Bay of Biscay, Celtic Sea, Iberian coast	2.6 Baltic Sea	2.7 North Sea	2.8 English Channel, Irish Sea, west Scottish Sea	2.9 Norwegian-Barents Seas
Area modelled	km^2	20253 (1.5-201535)	21382 (405-61000)	4924 (0.6-45547)	103833 (11500-150000)	100148 (8.6-1000000)	103207 (11.5-240000)	411477 (1513.5-570000)	31365 (7.8-190200)	948638 (11.8-3116000)
No of FGs		35 (14-48)	38 (19-72)	41 (12-72)	13 (8-17)	37 (14-52)	15 (12-22)	47 (29-68)	31 (9-51)	47 (27-108)
Sum of all consumption	$tkm^{-2}y^{-1}$	937.9 (185-1456)	1962.7 (348-4679)	3114 (51.4-21339.9)	485.5 (234.4-832)	1672.2 (277.8-7943.45)	589.4 (68-3435)	4905.7 (2333-6227)	1138.1 (44.2-3905)	1252.1 (207.8-2528.5)
Sum of all exports	$tkm^{-2}y^{-1}$	307.5 (41.6-686)	379.2 (1.7-960)	660.4 (21-1348.09)	990.6 (48.6-2504)	1096.3 (1.16-4654)	425.3 (8.3-1375)	317 (105-655)	1702 (84.5-6945)	62.4 (-0.67-179.9)
Sum of all respiratory flows	$tkm^{-2}y^{-1}$	332.3 (96.6-571)	893.7 (174.8-2375.4)	1695.7 (20.2-15151.4)	284.4 (157-449)	852.7 (134.7-2792.8)	310.3 (18-1851)	2039.7 (1501-2658)	488.4 (20.4-1330)	531.5 (106.2-1062.8)
Sum of all flows into detritus	$tkm^{-2}y^{-1}$	624.3 (283.5-1297)	1708.8 (508.9-4605)	1555 (65.8-5723.2)	1138.5 (191.2-2713)	2335.8 (193.6-17057)	564.2 (53-2246)	3023.1 (1348-3867)	2484.1 (296-12837)	882.5 (82.9-1919.1)
Total System Throughput	$tkm^{-2}y^{-1}$	1767.8 (583.3-1521)	4944.5 (1033.5-11792)	7025.2 (157.42784)	2998.8 (681.7-6499)	5956.7 (629-24544)	2408.8 (656-8007)	10285.1 (5837-12786)	5832.3 (1161-25017)	2728.6 (399.6-5638.3)
Sum of all production	$tkm^{-2}y^{-1}$	868.7 (296.4-1521)	1520.9 (599-3303)	2524 (14.83-14006)	1376.9 (286.5-3164)	2816.7 (182-12315)	984.2 (40-3296)	3634.3 (2551-4717.7)	2305.7 (417.1-9371)	486.8 (167.3-1920)
Mean Trophic Level of the Catch		3.21 (2.02-3.47)	2.97 (2.39-3.44)	3.19 (2-3.94)	3.18 (3.03-3.36)	3.59 (3.25-4.08)	3.14 (3.02-3.3)	3.57 (3.42-3.7)	2.89 (2.11-3.61)	3.49 (2.94-4.11)
Gross efficiency		0.004 (0.001-0.01)	0.002 (0.001-0.004)	0.008 (0.0001-0.05)	0.0009 (0.0001-0.002)	0.003 (0.001-0.009)	0.0016 (0.0002-0.0061)	0.0018 (0.0004-0.0027)	0.001 (0.0002-0.003)	0.0005 (0.0004-0.0007)
Calculated total net PP	$tkm^{-2}y^{-1}$	609.8 (240.2-1243)	1288.4 (495.4-2583)	1859.8 (386.7-7818.1)	1272.9 (256.1-2948)	2335.7 (135.9-11605)	1093.6 (28-3222)	2301.3 (2145-2609)	2015.3 (229.3-8275)	289.4 (110.7-1100.4)
Total PP/Total respiration		2.31 (1.1-4.26)	1.82 (1-2.83)	2.91 (0.8-6.84)	4.05 (1.16-7.9)	2.55 (0.56-8.2)	8.48 (1.05-21.35)	1.16 (0.98-1.4)	5.2 (1.09-22.1)	1.01 (1-1.03)

(Continued)

TABLE 2 Continued

Parameter	Units	2.1 Eastern Mediterranean Sea	2.2 Central Mediterranean Sea	2.3 Western Mediterranean Sea	2.4 Black Sea	2.5 Bay of Biscay, Celtic Sea, Iberian coast	2.6 Baltic Sea	2.7 North Sea	2.8 English Channel, Irish Sea, west Scottish Sea	2.9 Norwegian-Barents Seas
Net system production	$tkm^{-2}y^{-1}$	277.6 (28.5-672)	340.1 (6.7-746.2)	654.7 (-20.19-1346.4)	988.6 (48.6-2499)	1823.6 (1.16-9492)	1291.7 (8.3-2973.8)	416.9 (189.8-644)	1714.6 (22.9-6945)	
Total PP/Total biomass		18.7 (8-31.9)	14.7 (8.8-23.2)	18.2 (1.5-34)	83.2 (44.8-132)	19.4 (0.9-67.7)	29.2 (5.8-108.2)	4.1 (3.8-4.7)	27.3 (6.9-100.6)	9.2
Total biomass (excluding detritus)	tkm^{-2}	30.6 (8.7-78)	101.3 (21.3-247.7)	218.4 (3.93-1284)	22.7 (1.9-56.2)	851.1 (12.9-13201	79.4 (0.9-361.7)	556.2 (554-558)	86.6 (4.5-245)	59.2 (13.7-118.8)
Total Catch	$tkm^{-2}y^{-1}$	1.8 (0.4-3.4)	3.5 (0.6-9.5)	6.3 (0.2-36.9)	1.6 (0.03-4.5)	14.1 (3.06-45.1)	0.5 (0.03-1.2)	3.34 (0.8-5.9)	4.9 (1.5-15.9)	0.08 (0.02-0.3)
System Omnivory Index		0.21 (0.16-0.25)	0.23 (0.1-0.36)	0.23 (0.16-0.34)	0.14 (0.07-0.2)	0.15 (0.06-0.27)	0.12 (0.05-0.23)	0.24 (0.22-0.27)	0.15 (0.01-0.33)	0.21 (0.18-0.23
Ecopath Pedigree Index		0.53 (0.11-0.65)	0.58 (0.5-0.66)	0.58 (0.46-0.67)	0.41	0.61 (0.53-0.69)		0.59	0.62 (0.52-0.63)	0.55

Ecosystem indicators are reported as average scores in a weighted scale for each area. For each indicator, minimum and maximum values are in brackets.

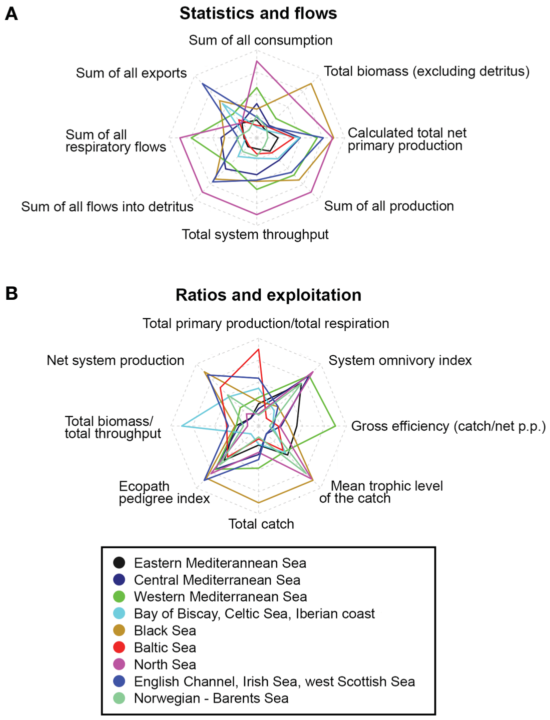

FIGURE 6
Radar charts of ecological indicators and average summary divided in statistics and flows **(A)** and ratios and exploitation **(B)**. Scoring scale ranges from -2 to 2 for Ecopath models of the nine reviewed European marine areas. The units of the indicators are in accordance with Table 2.

spatio-temporal expansion with Ecosim and Ecospace, which are comparatively less developed, recognizing nevertheless the difficulties of obtaining reliable spatial biological and environmental data for validation and hindcasting (Townsend et al., 2014; Colléter et al., 2015).

In line with the need for more effective and holistic considerations in fisheries management, the EBFM approach is key to moving forward. According to Pikitch et al. (2004), the aim of EBFM is threefold: (1) to maintain ecosystem health and resilience, and fisheries sustainability; (2) to take into account all ecosystem components; and (3) to establish a network of indicators to measure ecosystem degradation. EwE can inform on ecosystem-based management (EBM; Piroddi et al., 2022) and more specifically EBFM, with many representative studies (e.g., Guénette et al., 2006; Bourdaud et al., 2016) supporting practical implementations (Smith et al., 2007; Ofir et al., 2022). EBFM takes into consideration the complex dynamics of ecological, fisheries and human interactions in management (Long et al., 2015). EwE models embraced this direction, including aspects of previously ignored ecosystem elements (e.g., species interactions, discards and bycatch) to implement a holistic approach to managing fisheries (Fulton et al., 2014; Trochta et al., 2018). The EwE modelling framework incorporates fisheries, species interactions, environmental drivers, human activities, complexity and uncertainty, all important aspects of EBFM (Collie et al., 2016). In the future, more emphasis can be placed on plug-ins such as EcoTroph and/or ECOIND. EcoTroph can provide in fact more insights into the distribution of biomass and catch among trophic levels, taking into account the whole trophic spectrum, while ECOIND can further expand the framework of ecological analyses.

The assessment of ecosystem summary indicators highlighted some interesting trends. Highest average estimates of total biomass (excluding detritus) and of trophic flows were observed in ecosystems at higher latitudes. Positive correlations between biomass and fishery yields at colder, higher latitude ecosystems have been observed in the literature (Friedland et al., 2012). Such systems appear to have high seasonal productivity, being favoured by advection processes to sustain large shoals of commercially important fishes (Hunt et al., 2016). However, these systems are among the first to be affected by ocean warming (Kjesbu et al., 2014; Lotze et al., 2019). Many publications from the current review (e.g., Bentley et al., 2017; Vilas et al., 2020b; Pedersen et al., 2021) presented models considering the impacts of climate change and ocean warming (Stock et al., 2023). Consequently, the majority of the climate-related publications were focused on higher-latitude ecosystems (63%). In the Mediterranean Sea, chlorophyll a and primary production gradually decline from the western to the eastern basin, according to field and satellite data (e.g., Lavigne et al., 2015; Mattei and Scardi, 2022). As a result, a declining trend of total biomass and ecosystem size (TST) was observed from west to east; the Black Sea displayed instead the lowest estimates of total biomass. As the water moves eastward, it becomes more saline and oligotrophic, which means there is less organic matter and fewer nutrients to support abundance (D'Ortenzio and Ribera d'Alcalà, 2009). The highest number of models on alien species was found for Mediterranean Sea regions, and especially the eastern Mediterranean Sea, an outcome due to the nature of the basin that is one of the most invaded seas in the world (Giakoumi et al., 2019). As expected, upwelling and eutrophic ecosystems (i.e., Bay of Biscay, Celtic Sea, Iberian coast and North Sea) showed the highest total primary production. A similar finding was observed in total catches of these areas, with systems connected to the NE Atlantic Ocean exhibiting the greatest values. However, the Barents Sea represents an exception compared to other North Atlantic ecosystems due to a combination of factors, including lower species diversity, limited accessibility, shorter fishing season due to the sea ice, and underestimations, since unreported landings as well as discards miscalculations are very common (Popov and Zeller, 2018). The waters of NE Atlantic have upwelling characteristics, encouraging the reproduction of small pelagic fishes and the creation of large shoals that drive high fisheries yields and catch (Bjørndal, 2009; Feijó et al., 2018). The Mean Trophic Level of the Catch (MTLc) was highest in the Barents Sea (>4) and generally higher in high latitude regions. These findings are corroborated by the literature on large boreal ecosystems and may be explained by the abundance of large marine predators and elasmobranchs in the catch (Conti and Scardi, 2010). A same pattern was observed in lower latitude ecosystems; for instance, the oligotrophic eastern Mediterranean Sea has intermediate total yields associated with the highest average MTLc in the basin. However, high biomasses of top predators have not been observed in the eastern Mediterranean Sea compared to the central and western parts of the basin, suggesting the MTLc score in the east can be attributed to lower productivity and density of low trophic level organisms (e.g., small pelagics) in the catch (Tsagarakis et al., 2010).

The complexity of EwE models has increased along time, with a higher number of compartments found in most recent works compared to the oldest. Nevertheless, the number of FGs in most of the models ranged from 30 to 50 (Stock et al., 2023), an aspect that acknowledges the challenges posed by increasing model complexity (Green et al., 2005) and software tools' limitations to achieve mass-balance conditions (Villasante et al., 2016). Ecosim and Ecospace are used to implement simulations in time and space, respectively. Their performance, however, depends on the choice of the forcing factors, which influences calibration and fitting procedures (Mackinson et al., 2009). The majority of the models examined in this review incorporated trophic (vulnerabilities), fishery and primary production drivers, due to the automated fitting procedure available in EwE that includes these factors (Scott et al., 2016). Regarding the environmental drivers, most of the models incorporated temperature (e.g., SST, SBT), as most taxa have a strong connection between temperature and metabolic rates such as growth and consumption (Chabot et al., 2016). Also temperature data are the most easily accessible environmental data (Assis et al., 2018), however they play a crucial role in investigating the effects of climate change, confirming the ecosystem-level relevance of ocean warming, especially at higher latitudes (Grimm et al., 2013).

An issue related to the ecosystem indicators and the development of food web models like EwE is the uncertainty analysis (Evans et al., 2013). Ecosystem models are based on three principles (Levins, 1966): (1) generality (the tolerance limits of modelling applications); (2) realism (the degree of correspondence to reality); and (3) accuracy (the level of uncertainty). The latter is an integral part of complex ecosystem modelling (Milner-Gulland and Shea, 2017) and is considered in Ecopath, through the pedigree routine (Christensen and Walters, 2004a) to assess the integrity of input parameters and the quality of the model. However, the pedigree routine is not used by many models (Colléter et al., 2015). Understanding and addressing model uncertainty is the key to implement management scenarios and supporting robust decision making (Refsgaard et al., 2007). According to Rounsevell et al. (2021), uncertainty is divided in three categories: (1) scenario-based, including linguistic variations, alternative forms of implementation and input parameters interpretation; (2) model-based, encompassing structural limitations, input data reliability and error propagation; and (3) decision making-based, involving data interpretation, communication and translation of results. A newer approach to assess uncertainty in EwE was attained with the development of the Ecosampler plug-in (Steenbeek et al., 2018). Ecosampler is based on a Monte Carlo method (Metropolis and Ulam, 1949), built in EwE for automated input parameter optimization (Kavanagh et al., 2004). The Monte Carlo routine scans potential parameters that can ensure preserving the mass-balanced state of the ecosystem, and Ecosampler stores these plausible sets of models in its repository (Steenbeek et al., 2018). However, only 30% of the reviewed publications used the Monte Carlo routine and its results to perform uncertainty analysis on input parameters. Despite having its own limitations (e.g., modification of the base model may lead to invalid stored samples), Ecosampler is an add-on option that aims to account for uncertainty. Although uncertainty is often considered a threat, it can improve predictions obtained from ecological models (Larrosa et al., 2016; Rounsevell et al., 2021) and represents an essential condition for using EwE models to inform and support ecosystem-based management (Heymans et al., 2016).

Model validation, also known as skill assessment, is an essential step in evaluating the performance of coupled trophic/physical models, including EwE models (Stow et al., 2009). As a rule of thumb, some steps can be followed to achieve the best possible fit (Heymans et al., 2016). Ecopath input parameters, including data on biomass and catch, can be cross-checked with those from stock assessments and other sources (i.e., acoustic surveys, trawling samplings), enhancing model credibility (Agnetta et al., 2019; Natugonza et al., 2020). The PREBAL routine incorporated on the EwE software tool ensures a diagnostic check in which biomass and trophic flows are calculated for each group in the food web by balancing the input and output (Link, 2010). In the same way, Ecosim calibration is an essential step in Ecosim modelling to verify that the model accurately represents the temporal dynamics of the food web (Haputhantri et al., 2008; Tomczak et al., 2012). The stepwise fitting procedure is an automated routine for vulnerabilities and primary production drivers, which can be used as a skill assessment metric, based on an optimization algorithm (Mackinson et al., 2009; Scott et al., 2016). Ecospace has a multi-frame nature, as it is designed to capture the interactions between multiple species, fisheries, and the environment across a spatially-explicit domain. Model inter-comparison with other species distribution models (SDMs), like generalised linear models (GLMs), generalised additive models (GAMs) or boosted regression trees (BRTs) and cross-validation (Roberts et al., 2017) are skill assessment metrics used to achieve Ecospace modelling validation (Püts et al., 2020; Steenbeek et al., 2021).Simulating complex marine ecosystems and the impact of stressors acting on them is a challenging feat required to enhance EBFM. Despite existing difficulties in modelling ecosystems, this review reported a marked increase of EwE models in recent years, in all European marine areas. The increase in modelling efforts highlights the embracing of the holistic ecosystem management approach, primarily in vulnerable systems where single-species assessments and management alone have proven to be inadequate (Skern-Mauritzen et al., 2016). According to Craig and Link (2023), EwE modelling approach can support EBFM by following certain criteria, including (1) a clear management objective, (2) an accessible and well-constructed model, (3) a communication channel among scientists and decision-making bodies and (4) a versatile modelling approach (i.e., software modules and addressed topics). Other reviews have also assessed ecosystem models and applications globally, regarding the definition of the best suitable modelling approach according to ecosystems' characteristics and issues (Geary et al., 2020; Perryman et al., 2021). EwE is a successful modelling approach allowing to describe the whole food web state. It enables simulating food web changes in time and space, implementing forecast scenarios that can support management decisions. In the future, other reviews of EwE models, indicators and statistics could be implemented to inform about geographical gaps (e.g., areas with no existing models), possibly assisting global meta-analyses and identification of emerging ecosystem properties.

Author contributions

IK and AT conceived the study. IK collected the data, performed the meta-analysis and wrote the manuscript. DD visualized the graphic illustrations. EO, MS and GG reviewed the manuscript. AT supervised the manuscript and acquired funding. All authors contributed to the article and approved the submitted version.

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Conflict of interest

Publisher’s note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1182921/full#supplementary-material

References

Abdou, K., Halouani, G., Hattab, T., Romdhane, M. S., Ben Rais Lasram, F., and Le Loc’h, F. (2016). Exploring the potential effects of marine protected areas on the ecosystem structure of the gulf of gabes using the ecospace model. *Aquat. Living Resour.* 29 (2), 202. doi: 10.1051/alr/2016014

Abdou, K., Le Loc’h, F., Gascuel, D., Romdhane, M. S., Aubin, J., and Ben Rais Lasram, F. (2020). Combining ecosystem indicators and life cycle assessment for environmental assessment of demersal trawling in Tunisia. *Int. J. Life Cycle Assess.* 25, 105–119. doi: 10.1007/s11367-019-01651-5

Agnetta, D., Badalamenti, F., Colloca, F., D’ Anna, G., Di Lorenzo, G., Fiorentino, F., et al. (2019). Benthic-pelagic coupling mediates interactions in Mediterranean mixed fisheries: an ecosystem modeling approach. *PloS One* 14 (1), e0210659. doi: 10.1371/journal.pone.0210659

Akbari, N., Bjørndal, T., Failler, P., Forse, A., Taylor, M. H., and Drakeford, B. (2022). A multi-criteria framework for the sustainable management of fisheries: a case study of UK’s north Sea Scottish fisheries. *Environ. Manage.* 70, 79–96. doi: 10.1007/s00267-022-01607-w

Akoglu, E., Salihoglu, B., Libralato, S., Oguz, T., and Solidoro, C. (2014). An indicator-based evaluation of black Sea food web dynamics during 1960–2000. *J. Mar. Syst.* 134, 113–125. doi: 10.1016/j.jmarsys.2014.02.010

Albouy, C., Mouillot, D., Rocklin, D., Culioli, J. M., and Le Loc’h, F. (2010). Simulation of the combined effects of artisanal and recreational fisheries on a Mediterranean MPA ecosystem using a trophic model. *Mar. Ecol. Prog. Ser.* 412, 207–221. doi: 10.3354/meps08679

Albouy, C., Velez, L., Coll, M., Colloca, F., Le Loc’h, F., Mouillot, D., et al. (2013). From projected species distribution to food-web structure under climate change. *Global Change Biol.* 20 (3), 730–741. doi: 10.1111/gcb.12467

Alexander, K. A., Heymans, J. J., Magill, S., Tomczak, M. T., Holmes, S. J., and Wilding, T. A. (2015). Investigating the recent decline in gadoid stocks in the west of Scotland shelf ecosystem using a food web model. *ICES J. Mar. Sci.* 72 (2), 436–449. doi: 10.1093/icesjms/fsu149

Alexander, K. A., Meyjes, S. A., and Heymans, J. J. (2016). Spatial ecosystem modelling of marine renewable energy installations: gauging the utility of ecospace. *Ecol. Model.* 331, 115–128. doi: 10.1016/j.ecolmodel.2016.01.016

Araújo, J. N., Mackinson, S., Ellis, J. R., and Hart, P. J. B. (2005). An ecopath model of the western English channel ecosystem with an exploration of its dynamic properties. *Lowestoft: CEFAS Cefas Sci. Ser. Tech. Rep.* 125, 45.

Araújo, J. N., Mackinson, S., Stanford, R. J., and Hart, P. J. B. (2008). Exploring fisheries strategies for the western English channel using an ecosystem model. *Ecol. Model.* 210, 465–477. doi: 10.1016/j.ecolmodel.2007.08.015

Araújo, J. N., Mackinson, S., Stanford, R. J., Sims, D. W., Southward, A. J., Hawkins, S. J., et al. (2006). Modelling food web interactions, variation in plankton production, and fisheries in the western English channel ecosystem. *Mar. Ecol. Prog. Ser.* 309, 175–187. doi: 10.3354/meps309175

Arbach Leloup, F., Desroy, N., Le Mao, P., Pauly, D., and Le Pape, O. (2008). Interactions between a natural food web, shellfish farming and exotic species: the case of the bay of mont saint Michel (France). *Estuarine Coast. Shelf Sci.* 76, 111–120. doi: 10.1016/j.ecss.2007.06.011

Armoškaitė, A., Puriņa, I., Aigars, J., Strāķe, S., Pakalniete, K., Frederiksen, P., et al. (2020). Establishing the links between marine ecosystem components, functions and services: an ecosystem service assessment tool. *Ocean Coast. Manage.* 193, 105229. doi: 10.1016/j.ocecoaman.2020.105229

Assis, J., Tyberghein, L., Bosch, S., Verbruggen, H., Serrão, E. A., and De Clerck, O. (2018). Bio-ORACLE v2.0: extending marine data layers for bioclimatic modelling. *Global Ecol. Biogeogr.* 27 (3), 277–284. doi: 10.1111/geb.12693

Baeta, A., Niquil, N., Marques, J. C., and Patrício, J. (2011). Modelling the effects of eutrophication, mitigation measures and an extreme flood event on estuarine benthic food webs. *Ecol. Model.* 222 (6), 1209–1221. doi: 10.1016/j.ecolmodel.2010.12.010

Bănaru, D., Mellon-Duval, C., Roos, D., Bigot, J. L., Souplet, A., Jadaud, A., et al. (2013). Trophic structure in the gulf of lions marine ecosystem (north-western Mediterranean Sea) and fishing impacts. *J. Mar. Syst.* 111-112, 45–68. doi: 10.1016/j.jmarsys.2012.09.010

Barale, V. (2008). “The European marginal and enclosed seas: an overview” in: *Remote sensing of the European Seas.* Eds. V. Barale and M. Gade (Dordrecht: Springer), 463–474.

Barausse, A., Duci, A., Mazzoldi, C., Artioli, Y., and Palmeri, L. (2009). Trophic network model of the northern Adriatic Sea: analysis of an exploited and eutrophic ecosystem. *Estuarine Coast. Shelf Sci.* 83 (4), 577–590. doi: 10.1016/j.ecss.2009.05.003

Baudron, A. R., Serpetti, N., Fallon, N. G., Heymans, J. J., and Fernandes, P. G. (2019). Can the common fisheries policy achieve good environmental status in exploited ecosystems: the west of Scotland demersal fisheries example. *Fish. Res.* 211, 217–230. doi: 10.1016/j.fishres.2018.10.024

Bauer, B., Gustafsson, B. G., Hyytiäinen, K., Meier, H. E. M., Müller-Karulis, B., Saraiva, S., et al. (2019). Food web and fisheries in the future Baltic Sea. *Ambio* 48, 1337–1349. doi: 10.1007/s13280-019-01229-3

Bauer, B., Meier, H. E. M., Casini, M., Hoff, A., Margoński, P., Orio, A., et al. (2018). Reducing eutrophication increases spatial extent of communities supporting commercial fisheries: a model case study. *ICES J. Mar. Sci.* 75 (4), 1306–1317. doi: 10.1093/icesjms/fsy003

Bayle-Sempere, J. T., Arreguín-Sánchez, F., Sanchez-Jerez, P., Salcido-Guevara, L. A., Fernandez-Jover, D., and Zetina-Rejón, M. J. (2013). Trophic structure and energy fluxes around a Mediterranean fish farm. *Ecol. Model.* 248, 135–147. doi: 10.1016/j.ecolmodel.2012.08.028

Beattie, A., Sumaila, U. R., Christensen, V., and Pauly, D. (2002). A model for the bioeconomic evaluation of marine protected area size and placement in the north Sea. *Natural Res. Model.* 15 (4), 413–437. doi: 10.1111/j.1939-7445.2002.tb00096.x

Bentley, J. W., Hines, J. W., Borrett, S., Serpetti, N., Fox, C., Reid, D. G., et al. (2019a). Diet uncertainty analysis strengthens model-derived indicators of food web structure and function. *Ecol. Indic.* 98, 239–250. doi: 10.1016/j.ecolind.2018.11.008

Bentley, J. W., Hines, D. E., Borrett, S. R., Serpetti, N., Hernandez-Milian, G., Fox, C., et al. (2019c). Combining scientific and fishers' knowledge to co-create indicators of food web structure and function. *ICES J. Mar. Syst.* 76 (7), 2218–2234. doi: 10.1093/icesjms/fsz121

Bentley, J. W., Serpetti, N., Fox, C., Heymans, J. J., and Reid, D. G. (2019b). Fishers' knowledge improves the accuracy of food web model predictions. *ICES J. Mar. Sci.* 76 (4), 897–912. doi: 10.1093/icesjms/fsz003

Bentley, J. W., Serpetti, N., Fox, C. J., Heymans, J. J., and Reid, D. G. (2020). Retrospective analysis of the influence of environmental drivers on commercial stocks and fishing opportunities in the Irish Sea. *Fish. Oceanogr.* 29 (5), 415–435. doi: 10.1111/fog.12486

Bentley, J. W., Serpetti, N., and Heymans, J. J. (2017). Investigating the potential impacts of ocean warming on the Norwegian and barents seas ecosystem using a time-dynamic food-web model. *Ecol. Model.* 360, 94–107. doi: 10.1016/j.ecolmodel.2017.07.002

Bentorcha, A., Gascuel, D., and Guénette, S. (2017). Using trophic models to assess the impact of fishing in the bay of Biscay and the celtic Sea. *Aquat. Living Resour.* 30, 7. doi: 10.1051/alr/2017006

Bjørndal, T. (2009). Overview, roles, and performance of the north East Atlantic fisheries commission (NEAFC). *Mar. Policy* 33 (4), 685–697. doi: 10.1016/j.marpol.2009.01.007

Blanchard, J. L., Pinnegar, J. K., and Mackinson, S. (2002). Exploring marine mammal-fishery interactions using 'Ecopath with ecosim': modelling the barents Sea ecosystem. *Lowestoft: CEFAS Cefas Sci. Ser. Tech. Rep.* 117, 52.

Bodini, A., Rocchi, M., and Scotti, M. (2018). Insights into the ecology of the black Sea through the qualitative loop analysis of the community structure. *Limnol. Oceanogr.* 63 (2), 968–984. doi: 10.1002/lno.10713

Borja, A., Fontán, A., Sáenz, J., and Valencia, V. (2008). Climate, oceanography, and recruitment: the case of the bay of Biscay anchovy (*Engraulis encrasicolus*). *Fish. Oceanogr.* 17 (6), 477–493. doi: 10.1111/j.1365-2419.2008.00494.x

Borja, A., Prins, T. C., Simboura, N., Andersen, J. H., Berg, T., Marques, J.-C., et al. (2014). Tales from a thousand and one ways to integrate marine ecosystem components when assessing the environmental status. *Front. Mar. Sci.* 1, 72. doi: 10.3389/fmars.2014.00072

Bourdaud, P., Ben Rais Lasram, F., Araignous, E., Champagnat, J., Grusd, S., Halouani, G., et al. (2021). Impacts of climate change on the bay of seine ecosystem: forcing a spatio-temporal trophic model with predictions from an ecological niche model. *Fish. Oceanogr.* 30 (5), 471–489. doi: 10.1111/fog.12531

Bourdaud, P., Gascuel, D., Bentorcha, A., and Brind'Amour, A. (2016). New trophic indicators and target values for an ecosystem-based management of fisheries. *Ecol. Indic.* 61 (2), 588–601. doi: 10.1016/j.ecolind.2015.10.010

Bruno, J. F., Bates, A. E., Cacciapaglia, C., Pike, E. P., Amstrup, S. C., Van Hooidonk, R., et al. (2018). Climate change threatens the world's marine protected areas. *Nat. Climate Change* 8, 499–503. doi: 10.1038/s41558-018-0149-2

Bueno-Pardo, J., García-Seoane, E., Sousa, A. I., Coelho, J. P., Morgado, M., Frankenbach, S., et al. (2018). Trophic web structure and ecosystem attributes of a temperate coastal lagoon (Ria de aveiro, Portugal). *Ecol. Model.* 378, 13–25. doi: 10.1016/j.ecolmodel.2018.03.009

Carlucci, R., Capezzuto, F., Cipriano, G., D' Onghia, G., Fanizza, C., Libralato, S., et al. (2021). Assessment of cetacean–fishery interactions in the marine food web of the gulf of taranto (Northern Ionian Sea, central Mediterranean Sea). *Rev. Fish Biol. Fish.* 31, 135–156. doi: 10.1007/s11160-020-09623-x

Celić, I., Libralato, S., Scarcella, G., Raicevich, S., Marčeta, B., and Solidoro, C. (2018). Ecological and economic effects of the landing obligation evaluated using a quantitative ecosystem approach: a Mediterranean case study. *ICES J. Mar. Sci.* 75 (6), 1992–2003. doi: 10.1093/icesjms/fsy069

Chabot, D., Steffensen, J. F., and Farrell, A. P. (2016). The determination of standard metabolic rate in fishes. *J. Fish Biol.* 88 (1), 81–121. doi: 10.1111/jfb.12845

Christensen, V. (1995a). A model of trophic interactions in the north Sea in 1981, the year of the stomach. *Dana* 11 (1), 1–28.

Christensen, V. (1995b). Ecosystem maturity - towards quantification. *Ecol. Model.* 77 (1), 3–32. doi: 10.1016/0304-3800(93)E0073-C

Christensen, V., Ferdaña, Z., and Steenbeek, J. (2009). Spatial optimization of protected area placement incorporating ecological, social and economical criteria. *Ecol. Model.* 220 (19), 2583–2593. doi: 10.1016/j.ecolmodel.2009.06.029

Christensen, V., Steenbeek, J., and Failler, P. (2011). A combined ecosystem and value chain modeling approach for evaluating societal cost and benefit of fishing. *Ecol. Model.* 222 (3), 857–864. doi: 10.1016/j.ecolmodel.2010.09.030

Christensen, V., and Walters, C. J. (2004a). Ecopath with ecosim: methods, capabilities and limitations. *Ecol. Model.* 172 (2-4), 109–139. doi: 10.1016/j.ecolmodel.2003.09.003

Christensen, V., and Walters, C. J. (2004b). Trade-offs in ecosystem-scale optimization of fisheries management policies. *Bull. Mar. Sci.* 74 (3), 549–562.

Coll, M., Akoglu, E., Arreguín-Sánchez, F., Fulton, E. A., Gascuel, D., Heymans, J. J., et al. (2015). Modelling dynamic ecosystems: venturing beyond boundaries with the ecopath approach. *Rev. Fish Biol. Fish.* 25, 413–424. doi: 10.1007/s11160-015-9386-x

Coll, M., Bahamon, N., Sardà, F., Palomera, I., Tudela, S., and Suuronen, P. (2008b). Improved trawl selectivity: effects on the ecosystem in the south Catalan Sea (NW Mediterranean). *Mar. Ecol. Prog. Ser.* 355, 131–147. doi: 10.3354/meps07183

Coll, M., and Libralato, S. (2012). Contributions of food web modelling to the ecosystem approach to marine resource management in the Mediterranean Sea. *Fish Fish.* 13 (1), 60–88. doi: 10.1111/j.1467-2979.2011.00420.x

Coll, M., Navarro, J., Olson, R. J., and Christensen, V. (2012). Assessing the trophic position and ecological role of squids in marine ecosystems by means of food-web models. *Deep-Sea Res. II* 95, 21–36. doi: 10.1016/j.dsr2.2012.08.020

Coll, M., Navarro, J., and Palomera, I. (2013). Ecological role, fishing impact, and management options for the recovery of a Mediterranean endemic skate by means of food web models. *Biol. Conserv.* 157, 108–120. doi: 10.1016/j.biocon.2012.06.029

Coll, M., Palomera, I., and Tudela, S. (2009b). Decadal changes in a NW Mediterranean Sea food web in relation to fishing exploitation. *Ecol. Model.* 220 (17), 2088–2102. doi: 10.1016/j.ecolmodel.2009.04.049

Coll, M., Palomera, I., Tudela, S., and Dowd, M. (2008a). Food-web dynamics in the south Catalan Sea ecosystem (NW Mediterranean) for 1978–2003. *Ecol. Model.* 217, 95–116. doi: 10.1016/j.ecolmodel.2008.06.013

Coll, M., Palomera, I., Tudela, S., and Sardà, F. (2006a). Trophic flows, ecosystem structure and fishing impacts in the south Catalan Sea, northwestern Mediterranean. *J. Mar. Syst.* 59, 63–96. doi: 10.1016/j.jmarsys.2005.09.001

Coll, M., Santojanni, A., Palomera, I., and Arneri, E. (2009a). Food-web changes in the Adriatic Sea over the last three decades. *Mar. Ecol. Prog. Ser.* 381, 17–37. doi: 10.3354/meps07944

Coll, M., Santojanni, A., Palomera, I., and Arneri, E. (2010). Ecosystem assessment of the north-central Adriatic Sea: towards a multivariate reference framework. *Mar. Ecol. Prog. Ser.* 417, 193–210. doi: 10.3354/meps08800

Coll, M., Santojanni, A., Palomera, I., Tudela, S., and Arneri, E. (2007). An ecological model of the northern and central Adriatic Sea: analysis of ecosystem structure and fishing impacts. *J. Mar. Syst.* 67, 119–154. doi: 10.1016/j.jmarsys.2006.10.002

Coll, M., Shannon, L. J., Kleisner, K. M., Juan-Jordá, M. J., Bundy, A., Akoglu, A. G., et al. (2016b). Ecological indicators to capture the effects of fishing on biodiversity and conservation status of marine ecosystems. *Ecol. Indic.* 60, 947–962. doi: 10.1016/j.ecolind.2015.08.048

Coll, M., Shannon, L. J., Moloney, C. L., Palomera, I., and Tudela, S. (2006b). Comparing trophic flows and fishing impacts of a NW Mediterranean ecosystem with coastal upwelling systems by means of standardized models and indicators. *Ecol. Model.* 198, 53–70. doi: 10.1016/j.ecolmodel.2006.04.009

Coll, M., and Steenbeek, J. (2017). Standardized ecological indicators to assess aquatic food webs: the ECOIND software plug-in for ecopath with ecosim models. *Environ. Model. Softw.* 89, 120–130. doi: 10.1016/j.envsoft.2016.12.004

Coll, M., Steenbeek, J., Sole, J., Palomera, I., and Christensen, V. (2016a). Modelling the cumulative spatial–temporal effects of environmental drivers and fishing in a NW Mediterranean marine ecosystem. *Ecol. Model.* 331, 100–114. doi: 10.1016/j.ecolmodel.2016.03.020

Colléter, M., Valls, A., Guitton, J., Gascuel, D., Pauly, D., and Christensen, V. (2015). Global overview of the applications of the ecopath with ecosim modelling approach using the EcoBase models repository. *Ecol. Model.* 302, 42–53. doi: 10.1016/j.ecolmodel.2015.01.025

Collie, J. S., Botsford, L. W., Hastings, A., Kaplan, I. C., Largier, J. L., Livingston, P. A., et al. (2016). Ecosystem models for fisheries management: finding the sweet spot. *Fish Fish.* 17, 101–125. doi: 10.1111/faf.12093

Conti, L., and Scardi, M. (2010). Fisheries yield and primary productivity in large marine ecosystems. *Mar. Ecol. Prog. Ser.* 410, 233–244. doi: 10.3354/meps08630

Corrales, X., Coll, M., Ofir, E., Heymans, J. J., Steenbeek, J., Goren, M., et al. (2018). Future scenarios of marine resources and ecosystem conditions in the Eastern Mediterranean under the impacts of fishing, alien species and sea warming. *Sci. Rep.* 8, 14284. doi: 10.1038/s41598-018-32666-x

Corrales, X., Coll, M., Ofir, E., Piroddi, C., Goren, M., Edelist, D., et al. (2017b). Hindcasting the dynamics of an Eastern Mediterranean marine ecosystem under the impacts of multiple stressors. *Mar. Ecol. Prog. Ser.* 580, 17–36. doi: 10.3354/meps12271

Corrales, X., Coll, M., Tecchio, S., and Bellido, J. M. (2015). Ecosystem structure and fishing impacts in the northwestern Mediterranean Sea using a food web model within a comparative approach. *J. Mar. Syst.* 148, 183–199. doi: 10.1016/j.jmarsys.2015.03.006

Corrales, X., Katsanevakis, S., Coll, M., Heymans, J. J., Piroddi, C., Ofir, E., et al. (2020a). Advances and challenges in modelling the impacts of invasive alien species on aquatic ecosystems. *Biol. Invasions* 22, 907–934. doi: 10.1007/s10530-019-02160-0

Corrales, X., Ofir, E., Coll, M., Goren, M., Edelist, D., Heymans, J. J., et al. (2017a). Modeling the role and impact of alien species and fisheries on the Israeli marine continental shelf ecosystem. *J. Mar. Syst.* 170, 88–102. doi: 10.1016/j.jmarsys.2017.02.004

Corrales, X., Preciado, I., Gascuel, D., Lopez de Gamiz-Zearra, A., Hernvann, P.-Y., Mugerza, E., et al. (2022). Structure and functioning of the bay of Biscay ecosystem: a trophic modelling approach. *Estuarine Coast. Shelf Sci.* 264, 107658. doi: 10.1016/j.ecss.2021.107658

Corrales, X., Vilas, D., Piroddi, C., Steenbeek, J., Claudet, J., Lloret, J., et al. (2020b). Multi-zone marine protected areas assessment of ecosystem and fisheries benefits using multiple ecosystem models. *Ocean Coast. Manage.* 193, 105232. doi: 10.1016/j.ocecoaman.2020.105232

Costalago, D., Bauer, B., Tomczak, M. T., Lundström, K., and Winder, M. (2019). The necessity of a holistic approach when managing marine mammal–fisheries interactions: environment and fisheries impact are stronger than seal predation. *Ambio* 48, 552–564. doi: 10.1007/s13280-018-1131-y

Costello, M. J., Bouchet, P., Emblow, C. S., and Legakis, A. (2006). European Marine biodiversity inventory and taxonomic resources: state of the art and gaps in knowledge. *Mar. Ecol. Prog. Ser.* 316, 257–268. doi: 10.3354/meps316257

Craig, J. K., and Link, J. S. (2023). It is past time to use ecosystem models tactically to support ecosystem-based fisheries management: case studies using ecopath with ecosim in an operational management context. *Fish Fish.* 00, 1–26. doi: 10.1111/faf.12733

D'Ortenzio, F., and Ribera d'Alcalà, M. (2009). On the trophic regimes of the Mediterranean Sea: a satellite analysis. *Biogeosciences* 6, 139–148. doi: 10.5194/bg-6-139-2009

Dahood, A., de Mutsert, K., and Watters, G. M. (2020). Evaluating Antarctic marine protected area scenarios using a dynamic food web model. *Biol. Conserv.* 251, 108766. doi: 10.1016/j.biocon.2020.108766

Daskalov, G. M. (2002). Overfishing drives a trophic cascade in the black Sea. *Mar. Ecol. Prog. Ser.* 225, 53–63. doi: 10.3354/meps225053

Daskalov, G. M., Mackinson, S., and Mulligan, B. (2011). Modelling possible food-web effects of aggregate dredging in the eastern English channel. *Mar. Environ. Prot. Fund (MEPF). Ref No: MEPF.* 8, 37.

Díaz López, B., Bunke, M., and Bernal Shirai, J. A. (2008). Marine aquaculture off Sardinia island (Italy): ecosystem effects evaluated through a trophic mass-balance model. *Ecol. Model.* 212 (3-4), 292–303. doi: 10.1016/j.ecolmodel.2007.10.028

Dimarchopoulou, D., Keramidas, I., Tsagarakis, K., and Tsikliras, A. C. (2019). Ecosystem models and effort simulations of an untrawled gulf in the central Aegean Sea. *Front. Mar. Sci.* 6, 648. doi: 10.3389/fmars.2019.00648

Dimarchopoulou, D., Tsagarakis, K., Sylaios, G., and Tsikliras, A. C. (2022). Ecosystem trophic structure and fishing effort simulations of a major fishing ground in the northeastern Mediterranean Sea (Thermaikos gulf). *Estuarine Coast. Shelf Sci.* 264, 107667. doi: 10.1016/j.ecss.2021.107667

Dobrzycka-Krahel, A., and Medina-Villar, S. (2020). Alien species of Mediterranean origin in the Baltic Sea region: current state and risk assessment. *Environ. Rev.* 28 (3), 339–356. doi: 10.1139/er-2019-0074

Dommasnes, A., Christensen, V., Ellertsen, B., Kvamme, C., Melle, W., Nøttestad, L., et al. (2001). An ecopath model for the Norwegian Sea and barents Sea. *Fish. Impacts North Atlantic Ecosys.: Models Anal.* 9 (4), 213–240.

Edelist, D., Rilov, G., Golani, D., Carlton, J. T., and Spanier, E. (2013). Restructuring the Sea: profound shifts in the world's most invaded marine ecosystem. *Diversity Distributions* 19 (1), 69–77. doi: 10.1111/ddi.12002

Elliott, M., Borja, Á, McQuatters-Gollop, A., Mazik, K., Birchenough, S., Andersen, J. H., et al. (2015). *Force majeure*: will climate change affect our ability to attain good environmental status for marine biodiversity? *Mar. pollut. Bull.* 95 (1), 7–27. doi: 10.1016/j.marpolbul.2015.03.015

EU (2008). Directive 2008/56/EC of the European parliament and of the council establishing a framework for community action in the field of marine environmental policy (Marine strategy framework directive). *Off. J. Eur. Union* 164, 9–40.

EU (2013). Regulation (EU) no 1380/2013 of the European parliament and of the council of 11 December 2013 on the common fisheries policy, amending council regulations (EC) no 1954/2003 and (EC) no 1224/2009 and repealing council regulations (EC) no 2371/2002 and (EC) no 639/2004 and council decision 2004/585/EC. *Off. J. Eur. Union* 354, 22–61.

Evans, M. R., Grimm, V., Johst, K., Knuuttila, T., de Langhe, R., Lessells, C. M., et al. (2013). Do simple models lead to generality in ecology? *Trends Ecol. Evol.* 28 (10), 578–583. doi: 10.1016/j.tree.2013.05.022

Fabiano, M., Vassallo, P., Vezzulli, L., Salvo, V. S., and Marques, J. C. (2004). Temporal and spatial change of exergy and ascendency in different benthic marine ecosystems. *Energy* 29 (11), 1697–1712. doi: 10.1016/j.energy.2004.03.051

Fath, B. D., Asmus, H., Asmus, R., Baird, D., Borrett, S. R., De Jonge, V. N., et al. (2019). Ecological network analysis metrics: the need for an entire ecosystem approach in management and policy. *Ocean Coast. Manage.* 174, 1–14. doi: 10.1016/j.ocecoaman.2019.03.007

Fay, G., Large, S. I., Link, J. S., and Gamble, R. J. (2013). Testing systemic fishing responses with ecosystem indicators. *Ecol. Model.* 265, 45–55. doi: 10.1016/j.ecolmodel.2013.05.016

Feijó, D., Marçalo, A., Bento, T., Barra, J., Marujo, D., Correia, M., et al. (2018). Trends in the activity pattern, fishing yields, catch and landing composition between 2009 and 2013 from onboard observations in the Portuguese purse seine fleet. *Regional Stud. Mar. Sci.* 23, 97–106. doi: 10.1016/j.rsma.2017.12.007

Finn, J. T. (1976). Measures of ecosystem structure and function derived from analysis of flows. *J. Theor. Biol.* 56 (2), 363–380. doi: 10.1016/S0022-5193(76)80080-X

Forrestal, F., Coll, M., Die, D. J., and Christensen, V. (2012). Ecosystem effects of bluefin tuna *Thunnus thynnus thynnus* aquaculture in the NW Mediterranean Sea. *Mar. Ecol. Prog. Ser.* 456, 215–231. doi: 10.3354/meps09700

Fouzai, N., Coll, M., Palomera, I., Santojanni, A., Arneri, E., and Christensen, V. (2012). Fishing management scenarios to rebuild exploited resources and ecosystems of the northern-central Adriatic (Mediterranean Sea). *J. Mar. Syst.* 102-104, 39–51. doi: 10.1016/j.jmarsys.2012.05.003

Friedland, K. D., Stock, C., Drinkwater, K. F., Link, J. S., Leaf, R. T., Shank, B. V., et al. (2012). Pathways between primary production and fisheries yields of Large marine ecosystems. *PloS One* 7 (1), e28945. doi: 10.1371/journal.pone.0028945

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/j.marpol.2018.04.018

Fulton, E. A., Smith, A. D. M., Smith, D. C., and Johnson, P. (2014). An integrated approach is needed for ecosystem based fisheries management: insights from ecosystem-level management strategy evaluation. *PloS One* 9 (1), e84242. doi: 10.1371/journal.pone.0084242

Galdies, C., Bellerby, R., Canu, D., Chen, W., Garcia-Luque, E., Gašparović, B., et al. (2020). European Policies and legislation targeting ocean acidification in european waters - current state. *Mar. Policy* 118, 103947. doi: 10.1016/j.marpol.2020.103947

Galil, B. S., Marchini, A., and Occhipinti-Ambrogi, A. (2018). East Is east and West is west? management of marine bioinvasions in the Mediterranean Sea. *Estuarine Coast. Shelf Sci.* 201, 7–16. doi: 10.1016/j.ecss.2015.12.021

Galparsoro, I., Connor, D. W., Borja, Á, Aish, A., Amorim, P., Bajjouk, T., et al. (2012). Using EUNIS habitat classification for benthic mapping in European seas: present concerns and future needs. *Mar. pollut. Bull.* 64 (12), 2630–2638. doi: 10.1016/j.marpolbul.2012.10.010

Gamito, S., and Erzini, K. (2005). Trophic food web and ecosystem attributes of a water reservoir of the ria Formosa (south Portugal). *Ecol. Model.* 181 (4), 509–520. doi: 10.1016/j.ecolmodel.2004.02.024

García-Rodríguez, E., Coll, M., Vivas, M., Bellido, J. M., Esteban, A., and Torres, M. A. (2021). A food-web comparative modeling approach highlights ecosystem singularities of the gulf of alicante (Western Mediterranean Sea). *J. Sea Res.* 174, 102073. doi: 10.1016/j.seares.2021.102073

Gascuel, D., and Pauly, D. (2009). EcoTroph: modelling marine ecosystem functioning and impact of fishing. *Ecol. Model.* 220, 2885–2898. doi: 10.1016/j.ecolmodel.2009.07.031

Geary, W. L., Bode, M., Doherty, T. S., Fulton, E. A., Nimmo, D. G., Tulloch, A. I. T., et al. (2020). A guide to ecosystem models and their environmental applications. *Nat. Ecol. Evol.* 4, 1459–1471. doi: 10.1038/s41559-020-01298-8

Giakoumi, S., Pey, A., Di Franco, A., Francour, P., Kizilkaya, Z., Arda, Y., et al. (2019). Exploring the relationships between marine protected areas and invasive fish in the world's most invaded sea. *Ecol. Appl.* 29 (1), e1809. doi: 10.1002/eap.1809

Giralt Paradell, O., Díaz López, B., Methion, S., and Rogan, E. (2020). Food-web interactions in a coastal ecosystem influenced by upwelling and terrestrial runoff off north-West Spain. *Mar. Environ. Res.* 157, 104933. doi: 10.1016/j.marenvres.2020.104933

Giralt Paradell, O., Methion, S., Rogan, E., and Díaz López, B. (2021). Modelling ecosystem dynamics to assess the effect of coastal fisheries on cetacean species. *J. Environ. Manage.* 285, 112175. doi: 10.1016/j.jenvman.2021.112175

Glegg, G., Jefferson, R., and Fletcher, S. (2015). Marine governance in the English channel (La manche): linking science and management. *Mar. pollut. Bull.* 95 (2), 707–718. doi: 10.1016/j.marpolbul.2015.02.020

Green, J. L., Hastings, A., Arzberger, P., Ayala, F. J., Cottingham, K. L., Cuddington, K., et al. (2005). Complexity in ecology and conservation: mathematical, statistical, and computational challenges. *BioScience* 55 (6), 501–510. doi: 10.1641/0006-3568(2005)055[0501:CIEACM]2.0.CO;2

Grimm, N. B., Chapin, F. S.III, Bierwagen, B., Gonzalez, P., Groffman, P. M., Luo, Y., et al. (2013). The impacts of climate change on ecosystem structure and function. *Front. Ecol. Environ.* 11 (9), 474–482. doi: 10.1890/120282

Grossowicz, M., Ofir, E., Shabtay, A., Wood, J., Biton, E., Belkin, N., et al. (2020). Modeling the effects of brine outflow from desalination plants on coastal food-webs of the levantine basin (eastern Mediterranean Sea). *Desalination* 496, 114757. doi: 10.1016/j.desal.2020.114757

Gucu, A. C. (2002). Can overfishing be responsible for the successful establishment of *Mnemiopsis leidyi* in the black Sea? *Estuarine Coast. Shelf Sci.* 54, 439–451. doi: 10.1006/ecss.2000.0657

Gudlaugsson, S. T. (1993). Large Impact crater in the barents Sea. *Geology* 21 (4), 291–294. doi: 10.1130/0091-7613(1993)021<0291:LICITB>2.3.CO;2

Guénette, S., Heymans, J. J., Christensen, V., and Trites, A. W. (2006). Ecosystem models show combined effects of fishing, predation, competition, and ocean productivity on steller sea lions (*Eumetopias jubatus*) in Alaska. *Can. J. Fish. Aquat. Sci.* 63 (11), 2495–2517. doi: 10.1139/f06-136

Guesnet, V., Lassalle, G., Chaalali, A., Kearney, K., Saint-Béat, B., Karimi, B., et al. (2015). Incorporating food-web parameter uncertainty into ecopath-derived ecological network indicators. *Ecol. Model.* 313, 29–40. doi: 10.1016/j.ecolmodel.2015.05.036

Halouani, G., Abdou, K., Hattab, T., Romdhane, M. S., Ben Rais Lasram, F., and Le Loc'h, F. (2016). A spatio-temporal ecosystem model to simulate fishing management plans: a case of study in the gulf of gabes (Tunisia). *Mar. Policy* 69, 62–72. doi: 10.1016/j.marpol.2016.04.002

Halouani, G., Gascuel, D., Hattab, T., Ben Rais Lasram, F., Coll, M., Tsagarakis, K., et al. (2015). Fishing impact in Mediterranean ecosystems: an EcoTroph modeling approach. *J. Mar. Syst.* 150, 22–33. doi: 10.1016/j.jmarsys.2015.05.007

Halouani, G., Le Loc'h, F., Shin, Y.-J., Velez, L., Hattab, T., Romdhane, M. S., et al. (2019). An end-to-end model to evaluate the sensitivity of ecosystem indicators to track fishing impacts. *Ecol. Indic.* 98, 121–130. doi: 10.1016/j.ecolind.2018.10.061

Halouani, G., Villanueva, M. C., Raoux, A., Dauvin, J.-C., Ben Rais Lasram, F., Foucher, E., et al. (2020). A spatial food web model to investigate potential spillover effects of a fishery closure in an offshore wind farm. *J. Mar. Syst.* 212, 103434. doi: 10.1016/j.jmarsys.2020.103434

Haputhantri, S. S. K., Villanueva, M. C. S., and Moreau, J. (2008). Trophic interactions in the coastal ecosystem of Sri Lanka: an ECOPATH preliminary approach. *Estuarine Coast. Shelf Sci.* 76 (2), 304–318. doi: 10.1016/j.ecss.2007.07.013

Harvey, C. J., Cox, S. P., Essington, T. E., Hansson, S., and Kitchell, J. F. (2003). An ecosystem model of food web and fisheries interactions in the Baltic Sea. *ICES J. Mar. Sci.* 60, 939–950. doi: 10.1016/S1054-3139(03)00098-5

Hattab, T., Ben Rais Lasram, F., Albouy, C., Romdhane, M. S., Jarboui, O., Halouani, G., et al. (2013). An ecosystem model of an exploited southern Mediterranean shelf region (Gulf of gabes, Tunisia) and a comparison with other Mediterranean ecosystem model properties. *J. Mar. Syst.* 128, 159–174. doi: 10.1016/j.jmarsys.2013.04.017

Heiskanen, A.-S., Berg, T., Uusitalo, L., Teixeira, H., Bruhn, A., Krause-Jensen, D., et al. (2016). Biodiversity in marine ecosystems–European developments toward robust assessments. *Front. Mar. Sci.* 3, 184. doi: 10.3389/fmars.2016.00184

Hernvann, P.-Y., Gascuel, D., Grüss, A., Druon, J.-N., Kopp, D., Perez, I., et al. (2020). The celtic Sea trough time and space: ecosystem modeling to unravel fishing and climate change impacts on food-web structure and dynamics. *Front. Mar. Sci.* 7, 578717. doi: 10.3389/fmars.2020.578717

Heymans, J. J., Coll, M., Link, J. S., Mackinson, S., Steenbeek, J., Walters, C., et al. (2016). Best practice in ecopath with ecosim food-web models for ecosystem-based management. *Ecol. Model.* 331, 173–184. doi: 10.1016/j.ecolmodel.2015.12.007

Heymans, J. J., Howell, K. L., Ayers, M., Burrows, M. T., Gordon, J. D. M., Jones, E. G., et al. (2011b). Do we have enough information to apply the ecosystem approach to management of deep-sea fisheries? an example from the West of Scotland. *ICES J. Mar. Sci.* 68 (2), 265–280. doi: 10.1093/icesjms/fsq065

Heymans, J. J., Mackinson, S., Sumaila, U. R., Dyck, A., and Little, A. (2011a). The impact of subsidies on the ecological sustainability and future profits from north Sea fisheries. *PloS One* 6 (5), e20239. doi: 10.1371/journal.pone.0020239

Holt, J., Allen, J. I., Anderson, T. R., Brewin, R., Butenschön, M., Harle, J., et al. (2014). Challenges in integrative approaches to modelling the marine ecosystems of the north Atlantic: physics to fish and coasts to ocean. *Prog. Oceanogr.* 129 (B), 285–313. doi: 10.1016/j.pocean.2014.04.024

Howell, D., Schueller, A. M., Bentley, J. W., Buchheister, A., Chagaris, D., Cieri, M., et al. (2021). Combining ecosystem and single-species modeling to provide ecosystem-based fisheries management advice within current management systems. *Front. Mar. Sci.* 7, 607831. doi: 10.3389/fmars.2020.607831

Hunt, G. L., Drinkwater, K. F., Arrigo, K., Berge, J., Daly, K. L., Danielson, S., et al. (2016). Advection in polar and sub-polar environments: impacts on high latitude marine ecosystems. *Prog. Oceanogr.* 149, 40–81. doi: 10.1016/j.pocean.2016.10.004

Ito, M., Halouani, G., Cresson, P., Giraldo, C., and Girardin, R. (2023). Detection of fishing pressure using ecological network indicators derived from ecosystem models. *Ecol. Indic.* 147, 110011. doi: 10.1016/j.ecolind.2023.110011

Izquierdo-Gomez, D., Bayle-Sempere, J. T., Arreguín-Sánchez, F., and Sánchez-Jerez, P. (2016). Modeling population dynamics and small-scale fisheries yields of fish farming escapes in Mediterranean coastal areas. *Ecol. Model.* 331, 56–67. doi: 10.1016/j.ecolmodel.2016.01.012

Jørgensen, L. L., Pecuchet, L., Ingvaldsen, R. B., and Primicerio, R. (2022). Benthic transition zones in the Atlantic gateway to a changing Arctic ocean. *Prog. Oceanogr.* 204, 102792. doi: 10.1016/j.pocean.2022.102792

Johannesen, E., Høines, A. S., Dolgov, A. V., and Fossheim, M. (2012). Demersal fish assemblages and spatial diversity patterns in the Arctic-Atlantic transition zone in the barents Sea. *PloS One* 7 (4), e34924. doi: 10.1371/journal.pone.0034924

Katsanevakis, S., Zenetos, A., Corsini-Foka, M., and Tsiamis, K. (2020). "Biological Invasions in the Aegean Sea: Temporal Trends, Pathways, and Impacts" in: *The Handbook of Environmental Chemistry*. (Berlin, Heidelberg: Springer).

Kavanagh, P., Newlands, N., Christensen, V., and Pauly, D. (2004). Automated parameter optimization for Ecopath ecosystem models. *Ecol. Modell.* 172 (2-4), 141–149. doi: 10.1016/j.ecolmodel.2003.09.004

Keramidas, I., Dimarchopoulou, D., and Tsikliras, A. C. (2022). Modelling and assessing the ecosystem of the Aegean Sea, a major hub of the eastern Mediterranean at the intersection of Europe and Asia. *Regional Stud. Mar. Sci.* 56, 102704. doi: 10.1016/j.rsma.2022.102704

Kideys, A. E. (2002). Ecology: fall and rise of the black Sea ecosystem. *Science* 297 (5586), 1482–1484. doi: 10.1126/science.107300

Kjesbu, O. S., Bogstad, B., Devine, J. A., Gjøsæter, H., Howell, D., Ingvaldsen, R. B., et al. (2014). Synergies between climate and management for Atlantic cod fisheries at high latitudes. *Proc. Natl. Acad. Sci. United States America* 111 (9), 3478–3483. doi: 10.1073/pnas.1316342111

Korpinen, S., Laamanen, L., Bergström, L., Nurmi, M., Andersen, J. H., Haapaniemi, J., et al. (2021). Combined effects of human pressures on europe's marine ecosystems. *Ambio* 50, 1325–1336. doi: 10.1007/s13280-020-01482-x

Korpinen, S., Uusitalo, L., Nordström, M. C., Dierking, J., Tomczak, M. T., Haldin, J., et al. (2022). Food web assessments in the Baltic Sea: models bridging the gap between indicators and policy needs. *Ambio* 51, 1687–1697. doi: 10.1007/s13280-021-01692-x

Kourafalou, V. H. (1999). Process studies on the po river plume, north Adriatic Sea. *J. Geophys. Res.* 104 (C12), 29963–29985. doi: 10.1029/1999JC900217

Larrosa, C., Carrasco, L. R., and Milner-Gulland, E. J. (2016). Unintended feedbacks: challenges and opportunities for improving conservation effectiveness. *Conserv. Lett.* 9 (5), 316–326. doi: 10.1111/conl.12240

Larsen, L.-H., Sagerup, K., and Ramsvatn, S. (2016). The mussel path – using the contaminant tracer, ecotracer, in ecopath to model the spread of pollutants in an Arctic marine food web. *Ecol. Model.* 331, 77–85. doi: 10.1016/j.ecolmodel.2015.10.011

Lassalle, G., Chouvelon, T., Bustamante, P., and Niquil, N. (2014). An assessment of the trophic structure of the bay of Biscay continental shelf food web: comparing estimates derived from an ecosystem model and isotopic data. *Prog. Oceanogr.* 120, 205–215. doi: 10.1016/j.pocean.2013.09.002

Lassalle, G., Gascuel, D., Le Loc'h, F., Lobry, J., Pierce, G. J., Ridoux, V., et al. (2012). An ecosystem approach for the assessment of fisheries impacts on marine top predators: the bay of Biscay case study. *ICES J. Mar. Sci.* 69 (6), 925–938. doi: 10.1093/icesjms/fss049

Lassalle, G., Lobry, J., Le Loc'h, F., Bustamante, P., Certain, G., Delmas, D., et al. (2011). Lower trophic levels and detrital biomass control the bay of Biscay continental shelf food web: implications for ecosystem management. *Prog. Oceanogr.* 91, 561–575. doi: 10.1016/j.pocean.2011.09.002

Lassen, H., Pedersen, S. A., Frost, H., and Hoff, A. (2013). Fishery management advice with ecosystem considerations. *ICES J. Mar. Sci.* 70 (2), 471–479. doi: 10.1093/icesjms/fss208

Lavigne, H., D'Ortenzio, F., Ribera d'Alcalà, M., Claustre, H., Sauzède, R., and Gacic, M. (2015). On the vertical distribution of the chlorophyll a concentration in the Mediterranean Sea: a basin-scale and seasonal approach. *Biogeosciences* 12, 5021–5039. doi: 10.5194/bg-12-5021-2015

Lee, A. J. (1980) 24 (B), 467–493.

Le Marchand, M., Ben Rais Lasram, F., Araignous, E., Saint-Béat, B., Lassalle, G., Michelet, N., et al. (2022). Potential combined impacts of climate change and non-indigenous species arrivals on bay of Biscay trophic network structure and functioning. *J. Mar. Syst.* 228, 103704. doi: 10.1016/j.jmarsys.2022.103704

Levins, R. (1966). The strategy of model building in population biology. *Am. Sci.* 54 (4), 421–431.

Libralato, S., Caccin, A., and Pranovi, F. (2015). Modeling species invasions using thermal and trophic niche dynamic sunder climate change. *Front. Mar. Sci.* 2, 29. doi: 10.3389/fmars.2015.00029

Libralato, S., Christensen, V., and Pauly, D. (2006). A method for identifying keystone species in food web models. *Ecol. Modell.* 195 (3-4), 153–171. doi: 10.1016/j.ecolmodel.2005.11.029

Libralato, S., Coll, M., Tempesta, M., Santojanni, A., Spoto, M., Palomera, I., et al. (2010). Food-web traits of protected and exploited areas of the Adriatic Sea. *Biol. Conserv.* 143 (9), 2182–2194. doi: 10.1016/j.biocon.2010.06.002

Lindeman, R. L. (1942). The trophic-dynamic aspect of ecology. *Ecology* 23, 399–418. doi: 10.2307/1930126

Link, J. S. (2010). Adding rigor to ecological network models by evaluating a set of pre-balance diagnostics: a plea for PREBAL. *Ecol. Model.* 221 (12), 1580–1591. doi: 10.1016/j.ecolmodel.2010.03.012

Link, J. S., and Watson, R. A. (2019). Global ecosystem overfishing: clear delineation within real limits to production. *Sci. Adv.* 5 (6), eaav047. doi: 10.1126/sciadv.aav0474

Livne, L., Grossowicz, M., Tchernov, D., and Ayalon, O. (2020). Predicting impacts of offshore monoculture farm expansion in ultra-oligotrophic waters of the levantine basin. *Front. Mar. Sci.* 7, 556. doi: 10.3389/fmars.2020.00556

Lobry, J., David, V., Pasquaud, S., Lepage, M., Sautour, B., and Rochard, E. (2008). Diversity and stability of an estuarine trophic network. *Mar. Ecol. Prog. Ser.* 358, 13–25. doi: 10.3354/meps07294

Long, R. D., Charles, A., and Stephenson, R. L. (2015). Key principles of marine ecosystem-based management. *Mar. Policy* 57, 53–60. doi: 10.1016/j.marpol.2015.01.013

Lorance, P., Bertrand, J. A., Brind'Amour, A., Rochet, M. J., and Trenkel, V. M. (2009). Assessment of impacts from human activities on ecosystem components in the bay of Biscay in the early 1990s. *Aquat. Living Resour.* 22 (4), 409–431. doi: 10.1051/alr/2009049

Lotze, H. K., Tittensor, D. P., Bryndum-Buchholz, A., Eddy, T. D., Cheung, W. W. L., Galbraith, E. D., et al. (2019). Global ensemble projections reveal trophic amplification of ocean biomass declines with climate change. *Proc. Natl. Acad. Sci. United States America* 116 (26), 12907–12912. doi: 10.1073/pnas.1900194116

Lüdmann, T., Saitz, Y. M., Metzing, J., and Emeis, K.-C. (2021). Acoustic backscatter analysis of ground-fishing activity in the German north Sea sector. *Continental Shelf Res.* 212, 104292. doi: 10.1016/j.csr.2020.104292

Lykousis, V., Chronis, G., Tselepides, A., Price, N. B., Theocharis, A., Siokou-Frangou, I., et al. (2002). Major outputs of the recent multidisciplinary biogeochemical researches undertaken in the Aegean Sea. *J. Mar. Syst.* 33-34, 313–334. doi: 10.1016/S0924-7963(02)00064-7

Lynam, C. P., and Mackinson, S. (2015). How will fisheries management measures contribute towards the attainment of good environmental status for the north Sea ecosystem? *Global Ecol. Conserv.* 4, 160–175. doi: 10.1016/j.gecco.2015.06.005

Mackinson, S. (2014). Combined analyses reveal environmentally driven changes in the north Sea ecosystem and raise questions regarding what makes an ecosystem model's performance credible? *Can. J. Fish. Aquat. Sci.* 71 (1), 31–46. doi: 10.1139/cjfas-2013-0173

Mackinson, S., and Daskalov, G. (2007). An ecosystem model of the north Sea for use in research supporting the ecosystem approach to fisheries management: description and parameterisation. *Lowestoft: CEFAS Cefas Sci. Ser. Tech. Rep.* 142, 200.

Mackinson, S., Daskalov, G., Heymans, J. J., Neira, S., Arancibia, H., Zetina-Rejón, M., et al. (2009). Which forcing factors fit? using ecosystem models to investigate the relative influence of fishing and changes in primary productivity on the dynamics of marine ecosystems. *Ecol. Model.* 220 (21), 2972–2987. doi: 10.1016/j.ecolmodel.2008.10.021

Mackinson, S., Platts, M., Garcia, C., and Lynam, C. (2018). Evaluating the fishery and ecological consequences of the proposed north Sea multi-annual plan. *PloS One* 13 (1), e0190015. doi: 10.1371/journal.pone.0190015

Marshall, K. N., Koehn, L. E., Levin, P. S., Essington, T. E., and Jensen, O. P. (2019). Inclusion of ecosystem information in US fish stock assessments suggests progress toward ecosystem-based fisheries management. *ICES J. Mar. Sci.* 76 (1), 1–9. doi: 10.1093/icesjms/fsy152

Marten, G. G., and Polovina, J. J. (1982). "A comparative study of fish yields from various tropical ecosystems," in *ICLARM Conference Proceedings*, Vol. 9, 255–289.

Matear, L., Robbins, J. R., Hale, M., and Potts, J. (2019). Cetacean biodiversity in the bay of Biscay: suggestions for environmental protection derived from citizen science data. *Mar. Policy* 109, 103672. doi: 10.1016/j.marpol.2019.103672

Mattei, F., and Scardi, M. (2022). Mining satellite data for extracting chlorophyll a spatio-temporal patterns in the Mediterranean Sea. *Environ. Model. Softw.* 150, 105353. doi: 10.1016/j.envsoft.2022.105353

Megrey, B. A., and Aydin, K. Y. (2009). A macrodescriptor perspective of ecological attributes for the Bering and barents seas. *Deep Sea Res. Part II: Topical Stud. Oceanogr.* 56 (21-22), 2132–2140. doi: 10.1016/j.dsr2.2008.11.024

Metcalfe, K., Vaz, S., Engelhard, G. H., Villanueva, M. C., Smith, R. J., and Mackinson, S. (2015). Evaluating conservation and fisheries management strategies by linking spatial prioritization software and ecosystem and fisheries modelling tools. *J. Appl. Ecol.* 52, 665–674. doi: 10.1111/1365-2664.12404

Metropolis, N., and Ulam, S. (1949). The Monte Carlo method. *J. Am. Stat. Assoc.* 44 (247), 335–341. doi: 10.1080/01621459.1949.10483310

Michael-Bitton, G., Gal, G., Corrales, X., Ofir, E., Shechter, M., and Zemah-Samir, S. (2022). Economic aspects of fish stock accounting as a renewable marine natural capital: the Eastern Mediterranean continental shelf ecosystem as a case study. *Ecol. Econ.* 200, 107539. doi: 10.1016/j.ecolecon.2022.107539

Michailidis, N., Chartosia, N., and Katsanevakis, S. (2023). Exploring the role of fishing in a heavily bioinvaded shelf ecosystem. *Fish. Res.* 259, 106554. doi: 10.1016/j.fishres.2022.106554

Michailidis, N., Corrales, X., Karachle, P. K., Chartosia, N., Katsanevakis, S., and Sfenthourakis, S. (2019). Modelling the role of alien species and fisheries in an Eastern Mediterranean insular shelf ecosystem. *Ocean Coast. Manage.* 175, 152–171. doi: 10.1016/j.ocecoaman.2019.04.006

Milner-Gulland, E. J., and Shea, K. (2017). Embracing uncertainty in applied ecology. *J. Appl. Ecol.* 54 (6), 2063–2068. doi: 10.1111/1365-2664.12887

Moher, D., Liberati, A., Tetzlaff, J., Altman, D. G.The PRISMA Group (2010). Preferred reporting items for systematic reviews and meta-analyses: the PRISMA statement. *Int. J. Surg.* 8 (5), 336–341. doi: 10.1016/j.ijsu.2010.02.007

Möllmann, C., Cormon, X., Funk, S., Otto, S. A., Schmidt, J. O., Schwermer, H., et al. (2021). Tipping point realized in cod fishery. *Sci. Rep.* 11, 14259. doi: 10.1038/s41598-021-93843-z

Morato, T., Bulman, C., and Pitcher, T. J. (2009). Modelled effects of primary and secondary production enhancement by seamounts on local fish stocks. *Deep Sea Res. Part II: Topical Stud. Oceanogr.* 56 (25), 2713–2719. doi: 10.1016/j.dsr2.2008.12.029

Morato, T., Lemey, E., Menezes, G., Pham, C. K., Brito, J., Soszynski, A., et al. (2016). Food web and ecosystem structure of the open-ocean and deep-sea environments of the Azores, NE Atlantic. *Front. Mar. Sci.* 3, 245. doi: 10.3389/fmars.2016.00245

Moullec, F., Gascuel, D., Bentorcha, K., Guénette, S., and Robert, M. (2017). Trophic models: what do we learn about celtic Sea and bay of Biscay ecosystems? *J. Mar. Syst.* 172, 104–117. doi: 10.1016/j.jmarsys.2017.03.008

Moutopoulos, D. K., Libralato, S., Solidoro, C., and Stergiou, K. I. (2013). Toward an ecosystem approach to fisheries in the Mediterranean Sea: multi-gear/multi-species implications from an ecosystem model of the Greek Ionian Sea. *J. Mar. Syst.* 113-114, 13–28. doi: 10.1016/j.jmarsys.2012.12.002

Moutopoulos, D. K., Tsagarakis, K., and Machias, A. (2018). Assessing ecological and fisheries implications of the EU landing obligation in Eastern Mediterranean. *J. Sea Res.* 141, 99–111. doi: 10.1016/j.seares.2018.08.006

Natugonza, V., Ainsworth, C., Sturludóttir, E., Musinguzi, L., Ogutu-Ohwayo, R., Tomasson, T., et al. (2020). Ecosystem modelling of data-limited fisheries: how reliable are ecopath with ecosim models without historical time series fitting? *J. Great Lakes Res.* 46 (2), 414–428. doi: 10.1016/j.jglr.2020.01.001

Navarro, J., Coll, M., Louzao, M., Palomera, I., Delgado, A., and Forero, M. G. (2011). Comparison of ecosystem modelling and isotopic approach as ecological tools to investigate food webs in the NW Mediterranean Sea. *J. Exp. Mar. Biol. Ecol.* 401, 97–104. doi: 10.1016/j.jembe.2011.02.040

Nee, S. (1990). Community construction. *Trends Ecol. Evol.* 5 (10), 337–339. doi: 10.1016/0169-5347(90)90182-D

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A. C., Colijn, F., et al. (2014). An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. *Estuarine Coast. Shelf Sci.* 140, 95–122. doi: 10.1016/j.ecss.2013.05.023

Newton, A., Icely, J., Cristina, S., Perillo, G. M. E., Turner, R. E., Ashan, D., et al. (2020). Anthropogenic, direct pressures on coastal wetlands. *Front. Ecol. Evol.* 8, 144. doi: 10.3389/fevo.2020.00144

Niiranen, S., Yletyinen, J., Tomczak, M., Blenckner, T., Hjerne, O., MacKenzie, B. R., et al. (2013). Combined effects of global climate change and regional ecosystem drivers on an exploited marine food web. *Global Change Biol.* 19 (11), 3327–3342. doi: 10.1111/gcb.12309

Noguès, Q., Araignous, E., Bourdaud, P., Halouani, G., Raoux, A., Foucher, E., et al. (2022). Spatialized ecological network analysis for ecosystem-based management: effects of climate change, marine renewable energy, and fishing on ecosystem functioning in the bay of seine. *ICES J. Mar. Syst.* 79 (4), 1098–1112. doi: 10.1093/icesjms/fsac026

Odum, E. P. (1971). *Fundamentals of ecology* (Philadelphia: W.B. Saunders Co), 574.

Ofir, E., Silver, T., Steenbeek, J. G., Shachar, N., and Gal, G. (2022). Applying the safe operating space (SOS) approach to sustainable commercial fishing under varying lake levels and littoral zone conditions. *Fisheries*. 48 (3), 107–120. doi: 10.1002/fsh.10869

Ojaveer, H., Jaanus, A., MacKenzie, B. R., Martin, G., Olenin, S., Radziejewska, T., et al. (2010). Status of biodiversity in the Baltic Sea. *PloS One* 5 (9), e12467. doi: 10.1371/journal.pone.0012467

Österblom, H., Hansson, S., Larsson, U., Hjerne, O., Wulff, F., Elmgren, R., et al. (2007). Human-induced trophic cascades and ecological regime shifts in the Baltic Sea. *Ecosystems* 10, 877–889. doi: 10.1007/s10021-007-9069-0

Otogo, G. A., Wangvoralak, S., Pierce, G. J., Hastie, L. C., and Scott, B. (2015). The ecological role of *Loligo forbesii* in the Moray firth ecosystem, northeast Scotland. *Int. J. Mar. Environ. Sci.* 9, 7.

Ottersen, G., Bogstad, B., Yaragina, N. A., Stige, L. C., Vikebø, F. B., and Dalpadado, P. (2014). A review of early life history dynamics of barents Sea cod (*Gadus morhua*). *ICES J. Mar. Sci.* 71 (8), 2064–2087. doi: 10.1093/icesjms/fsu037

Outeiro, L., Byron, C., and Angelini, R. (2018). Ecosystem maturity as a proxy of mussel aquaculture carrying capacity in ria de arousa (NW spain): a food web modeling perspective. *Aquaculture* 496, 270–284. doi: 10.1016/j.aquaculture.2018.06.043

Pampoulie, C., Slotte, A., Óskarsson, G. J., Helyar, S. J., Jónsson, Á, Ólafsdóttir, G., et al. (2015). Stock structure of Atlantic herring *Clupea harengus* in the Norwegian Sea and adjacent waters. *Mar. Ecol. Prog. Ser.* 522, 219–230. doi: 10.3354/meps11114

Papaconstantinou, C., and Farrugio, H. (2000). Fisheries in the Mediterranean. *Mediterr. Mar. Sci.* 1, 5–18. doi: 10.12681/mms.2

Papantoniou, G., Giannoulaki, M., Stoumboudi, M. T., Lefkaditou, E., and Tsagarakis, K. (2021). Food web interactions in a human dominated Mediterranean coastal ecosystem. *Mar. Environ. Res.* 172, 105507. doi: 10.1016/j.marenvres.2021.105507

Papapanagiotou, G., Tsagarakis, K., Koutsidi, M., and Tzanatos, E. (2020). Using traits to build and explain an ecosystem model: ecopath with ecosim modelling of the north Aegean Sea (Eastern Mediterranean). *Estuarine Coast. Shelf Sci.* 236, 106614. doi: 10.1016/j.ecss.2020.106614

Patrício, J., and Marques, J. C. (2006). Mass balanced models of the food web in three areas along a gradient of eutrophication symptoms in the south arm of the mondego estuary (Portugal). *Ecol. Model.* 197 (1-2), 21–34. doi: 10.1016/j.ecolmodel.2006.03.008

Pauly, D., and Christensen, V. (1995). Primary production required to sustain global fisheries. *Nature* 374, 255–257. doi: 10.1038/374255a0

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, J. (1998). Fishing down marine food webs. *Science* 279 (5352), 860–863. doi: 10.1126/science.279.5352.860

Pauly, D., Christensen, V., and Walters, C. (2000). Ecopath, ecosim, and ecospace as tools for evaluating ecosystem impact of fisheries. *ICES J. Mar. Sci.* 57 (3), 697–706. doi: 10.1006/jmsc.2000.0726

Pauly, D., Soriano-Bartz, M. L., and Palomares, M. L. D. (1993). "Improved construction, parametrization and interpretation of steady-state ecosystem models" in *Trophic models of aquatic ecosystems. ICLARM Conf. Proc.* 26, 1–13.

Pedersen, T. (2022). Comparison between trophic positions in the barents Sea estimated from stable isotope data and a mass balance model. *Front. Mar. Sci.* 9, 813977. doi: 10.3389/fmars.2022.813977

Pedersen, T., Fuhrmann, M. M., Lindstrøm, U., Nilssen, E. M., Ivarjord, T., Ramasco, V., et al. (2018). Effects of the invasive red king crab on food web structure and ecosystem properties in an Atlantic fjord. *Mar. Ecol. Prog. Ser.* 596, 13–31. doi: 10.3354/meps12548

Pedersen, T., Mikkelsen, N., Lindstrøm, U., Renaud, P. E., Nascimento, M. C., Blanchet, M.-A., et al. (2021). Overexploitation, recovery, and warming of the barents Sea ecosystem during 1950–2013. *Front. Mar. Sci.* 8, 732637. doi: 10.3389/fmars.2021.732637

Pedersen, T., Nilsen, M., Nilssen, E. M., Berg, E., and Reigstad, E. (2008). Trophic model of a lightly exploited cod-dominated ecosystem. *Ecol. Model.* 214 (2-4), 95–111. doi: 10.1016/j.ecolmodel.2007.12.012

Pedersen, T., Ramsvatn, S., Nilssen, E. M., Nilsen, M., Morissette, L., Ivarjord, T., et al. (2016). Species diversity affects ecosystem structure and mass flows in fjords. *Regional Stud. Mar. Sci.* 3, 205–215. doi: 10.1016/j.rsma.2015.10.007

Peled, Y., Zemah-Shamir, S., Israel, A., Shechter, M., Ofir, E., and Gal, G. (2020). Incorporating insurance value into ecosystem services assessments: mitigation of ecosystem users' welfare uncertainty through biological control. *Ecosystem Serv.* 46, 101192. doi: 10.1016/j.ecoser.2020.101192

Pennino, M. G., Bevilacqua, A. H., Torres, M. A., Bellido, J. M., Sole, J., Steenbeek, J., et al. (2020). Discard ban: a simulation-based approach combining hierarchical Bayesian and food web spatial models. *Mar. Policy* 116, 103703. doi: 10.1016/j.marpol.2019.103703

Perryman, H. A., Hansen, C., Howell, D., and Olsen, E. (2021). A review of applications evaluating fisheries management scenarios through marine ecosystem models. *Rev. Fish. Sci. Aquacult.* 29 (4), 800–835. doi: 10.1080/23308249.2021.1884642

Pezy, J.-P., Raoux, A., Marmin, S., Balay, P., Niquil, N., and Dauvin, J.-C. (2017). Before-after analysis of the trophic network of an experimental dumping site in the eastern part of the bay of seine (English channel). *Mar. pollut. Bull.* 118 (1-2), 101–111. doi: 10.1016/j.marpolbul.2017.02.042

Pianka, E. R. (1973). The structure of lizard communities. *Annu. Rev. Ecol. System.* 4, 53–74. doi: 10.1146/annurev.es.04.110173.000413

Pikitch, E. K., Rountos, K. J., Essington, T. E., Santora, C., Pauly, D., Watson, R., et al. (2014). The global contribution of forage fish to marine fisheries and ecosystems. *Fish Fish.* 15 (1), 43–64. doi: 10.1111/faf.12004

Pikitch, E. K., Santora, C., Babcock, E. A., Bakun, A., Bonfil, R., Conover, D. O., et al. (2004). Ecosystem-based fishery management. *Science* 305 (5682), 346–347. doi: 10.1126/science.1098222

Pinnegar, J. K., and Polunin, N. V. C. (2004). Predicting indirect effects of fishing in Mediterranean rocky littoral communities using a dynamic simulation model. *Ecol. Model.* 172 (2-4), 249–267. doi: 10.1016/j.ecolmodel.2003.09.010

Piroddi, C., Bearzi, G., and Christensen, V. (2010). Effects of local fisheries and ocean productivity on the northeastern Ionian Sea ecosystem. *Ecol. Model.* 221 (11), 1526–1544. doi: 10.1016/j.ecolmodel.2010.03.002

Piroddi, C., Bearzi, G., and Christensen, V. (2011b). Marine open cage aquaculture in the eastern Mediterranean Sea: a new trophic resource for bottlenose dolphins. *Mar. Ecol. Prog. Ser.* 440, 255–266. doi: 10.3354/meps09319

Piroddi, C., Bearzi, G., Gonzalvo, J., and Christensen, V. (2011a). From common to rare: the case of the Mediterranean common dolphin. *Biol. Conserv.* 144 (10), 2490–2498. doi: 10.1016/j.biocon.2011.07.003

Piroddi, C., Coll, M., Liquete, C., Macias, D., Greer, K., Buszowski, J., et al. (2017). Historical changes of the Mediterranean Sea ecosystem: modelling the role and impact of primary productivity and fisheries changes over time. *Sci. Rep.* 7, 44491. doi: 10.1038/srep44491

Piroddi, C., Coll, M., Macias, D., Steenbeek, J., Garcia-Gorriz, E., Mannini, A., et al. (2022). Modelling the Mediterranean Sea ecosystem at high spatial resolution to inform the ecosystem-based management in the region. *Sci. Rep.* 12, 19680. doi: 10.1038/s41598-022-18017-x

Piroddi, C., Coll, M., Steenbeek, J., Macias Moy, D., and Christensen, V. (2015b). Modelling the Mediterranean marine ecosystem as a whole: addressing the challenge of complexity. *Mar. Ecol. Prog. Ser.* 533, 47–65. doi: 10.3354/meps11387

Piroddi, C., Moutopoulos, D. K., Gonzalvo, J., and Libralato, S. (2016). Ecosystem health of a Mediterranean semi-enclosed embayment (Amvrakikos gulf, greece): assessing changes using a modeling approach. *Continental Shelf Res.* 121, 61–73. doi: 10.1016/j.csr.2015.10.007

Piroddi, C., Teixeira, H., Lynam, C. P., Smith, C., Alvarez, M. C., Mazik, K., et al. (2015a). Using ecological models to assess ecosystem status in support of the European marine strategy framework directive. *Ecol. Indic.* 58, 175–191. doi: 10.1016/j.ecolind.2015.05.037

Polovina, J. J. (1984). Model of a coral reef ecosystem: the ECOPATH model and its application to french frigate shoals. *Coral Reefs* 3, 1–11. doi: 10.1007/BF00306135

Popov, S., and Zeller, D. (2018). Reconstructed Russian fisheries catches in the barents Sea: 1950-2014. *Front. Mar. Sci.* 5, 266. doi: 10.3389/fmars.2018.00266

Pranovi, F., and Link, J. S. (2009). Ecosystem exploitation and trophodynamic indicators: a comparison between the northern Adriatic Sea and southern new England. *Prog. Oceanogr.* 81, 149–164. doi: 10.1016/j.pocean.2009.04.008

Prato, G., Barrier, C., Francour, P., Cappanera, V., Markantonatou, V., Guidetti, P., et al. (2016). Assessing interacting impacts of artisanal and recreational fisheries in a small marine protected area (Portofino, NW Mediterranean Sea). *Ecosphere* 7 (12), e01601. doi: 10.1002/ecs2.1601

Prato, G., Gascuel, D., Valls, A., and Francour, P. (2014). Balancing complexity and feasibility in Mediterranean coastal food-web models: uncertainty and constraints. *Mar. Ecol. Prog. Ser.* 512, 71–88. doi: 10.3354/meps10988

Püts, M., Kempf, A., Möllmann, C., and Taylor, M. (2023). Trade-offs between fisheries, offshore wind farms and marine protected areas in the southern north Sea – winners, losers and effective spatial management. *Mar. Policy* 152, 105574. doi: 10.1016/j.marpol.2023.105574

Püts, M., Taylor, M., Núñez-Riboni, I., Steenbeek, J., Stäbler, M., Möllmann, C., et al. (2020). Insights on integrating habitat preferences in process-oriented ecological models – a case study of the southern north Sea. *Ecol. Model.* 431, 109189. doi: 10.1016/j.ecolmodel.2020.109189

Raoux, A., Lassalle, G., Pezy, J.-P., Tecchio, S., Safi, G., Ernande, B., et al. (2019). Measuring sensitivity of two OSPAR indicators for a coastal food web model under offshore wind farm construction. *Ecol. Indic.* 96, 728–738. doi: 10.1016/j.ecolind.2018.07.014

Raoux, A., Tecchio, S., Pezy, J.-P., Lassalle, G., Degraer, S., Wilhelmsson, D., et al. (2017). Benthic and fish aggregation inside an offshore wind farm: which effects on the trophic web functioning? *Ecol. Indic.* 72, 33–46. doi: 10.1016/j.ecolind.2016.07.037

Ray, D., Leary, P., Livens, F., Gray, N., Morris, K., Law, K. A., et al. (2020). Controls on anthropogenic radionuclide distribution in the Sellafield-impacted Eastern Irish Sea. *Sci. Total Environ.* 743, 140765. doi: 10.1016/j.scitotenv.2020.140765

Reckermann, M., Omstedt, A., Soomere, T., Aigars, J., Akhtar, N., Beldowska, M., et al. (2022). Human impacts and their interactions in the Baltic Sea region. *Earth System Dyn.* 13 (1), 1–80. doi: 10.5194/esd-13-1-2022

Reeburgh, W. S., Ward, B. B., Whalen, S. C., Sandbeck, K. A., Kilpatrickt, K. A., and Kerkhof, L. J. (1991). Black Sea methane geochemistry. *Deep Sea Res. Part A. Oceanogr. Res. Papers* 38 (S2), S1189–S1210. doi: 10.1016/S0198-0149(10)80030-5

Refsgaard, J. C., van der Sluijs, J. P., Højberg, A. L., and Vanrolleghem, P. A. (2007). Uncertainty in the environmental modelling process – a framework and guidance. *Environ. Model. Softw.* 22 (11), 1543–1556. doi: 10.1016/j.envsoft.2007.02.004

Reusch, T. B. H., Dierking, J., Andersson, H. C., Bonsdorff, E., Carstensen, J., Casini, M., et al. (2018). The Baltic Sea as a time machine for the future coastal ocean. *Sci. Adv.* 4 (5), eaar8195. doi: 10.1126/sciadv.aar8195

Ricci, P., Libralato, S., Capezzuto, F., D' Onghia, G., Maiorano, P., Sion, L., et al. (2019). Ecosystem functioning of two marine food webs in the north-Western Ionian Sea (Central Mediterranean Sea). *Ecol. Evol.* 9 (18), 10198–10212. doi: 10.1002/ece3.5527

Ricci, P., Sion, L., Capezzuto, F., Cipriano, G., D' Onghia, G., Libralato, S., et al. (2021). Modelling the trophic roles of the demersal Chondrichthyes in the northern Ionian Sea (Central Mediterranean Sea). *Ecol. Model.* 444, 109468. doi: 10.1016/j.ecolmodel.2021.109468

Roberts, D. R., Bahn, V., Ciuti, S., Boyce, M. S., Elith, J., Guillera-Arroita, G., et al. (2017). Cross-validation strategies for data with temporal, spatial, hierarchical, or phylogenetic structure. *Ecography* 40 (8), 913–929. doi: 10.1111/ecog.02881

Romagnoni, G., Mackinson, S., Hong, J., and Eikeset, A. M. (2015). The ecospace model applied to the north Sea: evaluating spatial predictions with fish biomass and fishing effort data. *Ecol. Model.* 300, 50–60. doi: 10.1016/j.ecolmodel.2014.12.016

Rombouts, I., Beaugrand, G., Artigas, L. F., Dauvin, J. C., Gevaert, F., Goberville, E., et al. (2013). Evaluating marine ecosystem health: case studies of indicators using direct observations and modelling methods. *Ecol. Indic.* 24, 353–365. doi: 10.1016/j.ecolind.2012.07.001

Rounsevell, M. D. A., Arneth, A., Brown, C., Cheung, W. W. L., Gimenez, O., Holman, I., et al. (2021). Identifying uncertainties in scenarios and models of socio-ecological systems in support of decision-making. *One Earth* 4 (7), 967–985. doi: 10.1016/j.oneear.2021.06.003

Russo, A., and Artegiani, A. (1996). Adriatic Sea Hydrography. *Scie. Marina* 60, 33–43.

Rybarczyk, H., and Elkaïüm, B. (2003). An analysis of the trophic network of a macrotidal estuary: the seine estuary (Eastern channel, Normandy, France). *Estuarine Coast. Shelf Sci.* 58 (4), 775–791. doi: 10.1016/S0272-7714(03)00184-7

Rybarczyk, H., Elkaïüm, B., Ochs, L., and Loquet, N. (2003). Analysis of the trophic network of a macrotidal ecosystem: the bay of Somme (Eastern channel). *Estuarine Coast. Shelf Sci.* 58 (3), 405–421. doi: 10.1016/S0272-7714(02)00294-9

Sabatino, R., Di Cesare, A., Dzhembekova, N., Fontaneto, D., Eckert, E. M., Corno, G., et al. (2020). Spatial distribution of antibiotic and heavy metal resistance genes in the black Sea. *Mar. pollut. Bull.* 160, 111635. doi: 10.1016/j.marpolbul.2020.111635

Sætre, R. (1999). Features of the central Norwegian shelf circulation. *Continental Shelf Res.* 19 (14), 1809–1831. doi: 10.1016/S0278-4343(99)00041-2

Safi, G., Giebels, D., Arroyo, N. L., Heymans, J. J., Preciado, I., Raoux, A., et al. (2019). Vitamine ENA: a framework for the development of ecosystem-based indicators for decision makers. *Ocean Coast. Manage.* 174, 116–130. doi: 10.1016/j.ocecoaman.2019.03.005

Sánchez, F., and Olaso, I. (2004). Effects of fisheries on the cantabrian Sea shelf ecosystem. *Ecol. Model.* 172, 151–174. doi: 10.1016/j.ecolmodel.2003.09.005

Sandberg, J., Elmgren, R., and Wulff, F. (2000). Carbon flows in Baltic Sea food webs — a re-evaluation using a mass balance approach. *J. Mar. Syst.* 25, 249–260. doi: 10.1016/S0924-7963(00)00019-1

Sandberg, J., Kumbald, L., and Kautsky, U. (2007). Can ECOPATH with ECOSIM enhance models of radionuclide flows in food webs? - an example for ^{14}C in a coastal food web in the Baltic Sea. *J. Environ. Radioactivity* 92, 96–111. doi: 10.1016/j.jenvrad.2006.09.010

Sautour, B., and Castel, J. (1995). Comparative spring distribution of zooplankton in three macrotidal European estuaries. *Hydrobiologia* 311, 139–151. doi: 10.1007/BF00008577

Sayer, M. D. J., Magill, S. H., Pitcher, T. J., Morissette, L., and Ainsworth, C. (2005). Simulation-based investigations of fishery changes as affected by the scale and design of artificial habitats. *J. Fish Biol.* 67 (sB), 218–243. doi: 10.1111/j.0022-1112.2005.00928.x

Saygu, I., Akoglu, E., Gül, G., Bedikoğlu, D., and Demirel, N. (2023). Fisheries impact on the Sea of marmara ecosystem structure and functioning during the last three decades. *Front. Mar. Sci.* 9, 1076399. doi: 10.3389/fmars.2022.1076399

Saygu, I., Heymans, J. J., Fox, C. J., Özbilgin, H., Eryaşar, A. R., and Gökçe, G. (2020). The importance of alien species to the food web and bottom trawl fisheries of the northeastern Mediterranean, a modelling approach. *J. Mar. Syst.* 202, 103253. doi: 10.1016/j.jmarsys.2019.103253

Scharler, U. M., Ulanowicz, R. E., Fogel, M. L., Wooler, M. J., Jacobson-Meyers, M. E., Lovelock, C. E., et al. (2015). Variable nutrient stoichiometry (carbon:nitrogen: phosphorus) across trophic levels determines community and ecosystem properties in an oligotrophic mangrove system. *Oecologia* 179, 863–876. doi: 10.1007/s00442-015-3379-2

Scott, E., Serpetti, N., Steenbeek, J., and Heymans, J. J. (2016). A stepwise fitting procedure for automated fitting of ecopath with ecosim models. *SoftwareX* 5, 25–30. doi: 10.1016/j.softx.2016.02.002

Scotti, M., Bondavalli, C., Rossetti, G., and Bodini, A. (2022b). Flow network indices signal a directional change in ecosystems: evidence from a small mountain lake (Lake Santo, northern Italy). *Ecol. Indic.* 139, 108896. doi: 10.1016/j.ecolind.2022.108896

Scotti, M., Opitz, S., MacNeil, L., Kreutle, A., Pusch, C., and Froese, R. (2022a). Ecosystem-based fisheries management increases catch and carbon sequestration through recovery of exploited stocks: the western Baltic Sea case study. *Front. Mar. Sci.* 9, 879998. doi: 10.3389/fmars.2022.879998

Selleslagh, J., Lobry, J., Amara, R., Brylinski, J.-M., and Boët, P. (2012). Trophic functioning of coastal ecosystems along an anthropogenic pressure gradient: a French case study with emphasis on a small and low impacted estuary. *Estuarine Coast. Shelf Sci.* 112, 73–85. doi: 10.1016/j.ecss.2011.08.004

Serpetti, N., Baudron, A. R., Burrows, M. T., Payne, B. L., Helaouët, P., Fernandes, P. G., et al. (2017). Impact of ocean warming on sustainable fisheries management informs the ecosystem approach to fisheries. *Sci. Rep.* 7, 13438. doi: 10.1038/s41598-017-13220-7

Serpetti, N., Benjamins, S., Brain, S., Collu, M., Harvey, B. J., Heymans, J. J., et al. (2021). Modeling small scale impacts of multi-purpose platforms: an ecosystem approach. *Front. Mar. Sci.* 8, 694013. doi: 10.3389/fmars.2021.694013

Seyer, T., Bănaru, D., Vaz, S., Hattab, T., Labrune, C., Booth, S., et al. (2023). Ecosystem modelling in the northwestern Mediterranean Sea: structure and functioning of a complex system. *J. Mar. Syst.* 240, 103877. doi: 10.1016/j.jmarsys.2023.103877

Shabtay, A., Portman, M. E., Ofir, E., Carmel, Y., and Gal, G. (2018). Using Ecol. Modell. in marine spatial planning to enhance ecosystem-based management. *Mar. Policy* 95, 14–23. doi: 10.1016/j.marpol.2018.06.018

Shalovenkov, N. (2019). "Chapter 31 - alien species invasion: case study of the black Sea," in *Coasts and estuaries*, 547–568.

Shannon, L. J., Coll, M., Bundy, A., Gascuel, D., Heymans, J. J., Kleisner, K., et al. (2014). Trophic level-based indicators to track fishing impacts across marine ecosystems. *Mar. Ecol. Prog. Ser.* 512, 115–140. doi: 10.3354/meps10821

Sheldon, R. W., Prakash, A., and Sutcliffe, Jr. (1972). The size distribution of particles in the ocean. *Limnol. Oceanogr.* 17 (3), 327–340. doi: 10.4319/lo.1972.17.3.0327

Shin, Y.-J., Houle, J. E., Akoglu, E., Blanchard, J. L., Bundy, A., Coll, M., et al. (2018). The specificity of marine ecological indicators to fishing in the face of environmental change: a multi-model evaluation. *Ecol. Indic.* 89, 317–326. doi: 10.1016/j.ecolind.2018.01.010

Skern-Mauritzen, M., Ottersen, G., Handegard, N. O., Huse, G., Dingsør, G. E., Stenseth, N. C., et al. (2016). Ecosystem processes are rarely included in tactical fisheries management. *Fish Fish.* 17 (1), 165–175. doi: 10.1111/faf.12111

Smith, A. J. (1989). The English channel–by geological design or catastrophic accident? *Proc. Geol. Assoc.* 100 (3), 325–337. doi: 10.1016/S0016-7878(89)80052-5

Smith, A. D. M., Fulton, E. J., Hobday, A. J., Smith, D. C., and Shoulder, P. (2007). Scientific tools to support the practical implementation of ecosystem-based fisheries management. *ICES J. Mar. Sci.* 64 (4), 633–639. doi: 10.1093/icesjms/fsm041

Sohlenius, G., Emeis, K.-C., Andrén, E., Andrén, T., and Kohly, A. (2001). Development of anoxia during the Holocene fresh–brackish water transition in the Baltic Sea. *Mar. Geol.* 177 (3-4), 221–242. doi: 10.1016/S0025-3227(01)00174-8

Stäbler, M., Kempf, A., Mackinson, S., Poos, J. J., Garcia, C., and Temming, A. (2016). Combining efforts to make maximum sustainable yields and good environmental status match in a food-web model of the southern north Sea. *Ecol. Model.* 331, 17–30. doi: 10.1016/j.ecolmodel.2016.01.020

Stäbler, M., Kempf, A., and Temming, A. (2018). Assessing the structure and functioning of the southern north Sea ecosystem with a food-web model. *Ocean Coast. Manage.* 165, 280–297. doi: 10.1016/j.ocecoaman.2018.08.017

Steenbeek, J., Buszowski, J., Chagaris, D., Christensen, V., Coll, M., Fulton, E. A., et al. (2021). Making spatial-temporal marine ecosystem modelling better – a perspective. *Environ. Model. Softw.* 145, 105209. doi: 10.1016/j.envsoft.2021.105209

Steenbeek, J., Coll, M., Gurney, L., Mélin, F., Hoepffner, N., Buszowski, J., et al. (2013). Bridging the gap between ecosystem modeling tools and geographic information systems: driving a food web model with external spatial–temporal data. *Ecol. Model.* 263, 139–151. doi: 10.1016/j.ecolmodel.2013.04.027

Steenbeek, J., Corrales, J., Platts, M., and Coll, M. (2018). Ecosampler: a new approach to assessing parameter uncertainty in ecopath with ecosim. *Softw. X* 7, 198–204. doi: 10.1016/j.softx.2018.06.004

Stock, A., Murray, C. C., Gregr, E. J., Steenbeek, J., Woodburn, E., Micheli, F., et al. (2023). Exploring multiple stressor effects with ecopath, ecosim, and ecospace: research designs, modeling techniques, and future directions. *Sci. Total Environ.* 869, 161719. doi: 10.1016/j.scitotenv.2023.161719

Stow, C. A., Jolliff, J., McGillicuddy, Jr. D. J., Doney, S. C., Allen, J. I., Friedrichs, M. A. M., et al. (2009). Skill assessment for coupled biological/physical models of marine systems. *J. Mar. Syst.* 76 (1-2), 4–15. doi: 10.1016/j.jmarsys.2008.03.011

Susini, I., and Todd, V. L. G. (2021). Predictive capacity of ecopath with ecosim: model performance and ecological indicators' response to imprecision. *Environ. Model. Softw.* 143, 105098. doi: 10.1016/j.envsoft.2021.105098

Szalaj, D., Silva, A., Ré, P., and Cabral, H. (2022). Predictions of sardine and the Portuguese continental shelf ecosystem dynamics under future fishing, forced-biomass and SST scenarios. *Mar. pollut. Bull.* 178, 113594. doi: 10.1016/j.marpolbul.2022.113594

Szalaj, D., Torres, M. A., Veiga-Malta, T., Angélico, M. M., Sobrinho-Gonçalves, L., Chaves, C., et al. (2021). Food-web dynamics in the Portuguese continental shelf ecosystem between 1986 and 2017: unravelling drivers of sardine decline. *Estuarine Coast. Shelf Sci.* 251, 107259. doi: 10.1016/j.ecss.2021.107259

Tecchio, S., Coll, M., Christensen, V., Company, J. B., Ramírez-Llodra, E., and Sardà, F. (2013). Food web structure and vulnerability of a deep-sea ecosystem in the NW Mediterranean Sea. *Deep Sea Res. Part I: Oceanogr. Res. Papers* 75, 1–15. doi: 10.1016/j.dsr.2013.01.003

Tecchio, S., Rius, A. T., Dauvin, J.-C., Lobry, J., Lassalle, G., Morin, J., et al. (2015). The mosaic of habitats of the seine estuary: insights from food-web modelling and network analysis. *Ecol. Model.* 312, 91–101. doi: 10.1016/j.ecolmodel.2015.05.026

Tierney, K. M., Heymans, J. J., Muir, G. K. P., Cook, G. T., Buszowski, J., Steenbeek, J., et al. (2018). Modelling marine trophic transfer of radiocarbon (^{14}C) from a nuclear facility. *Environ. Model. Softw.* 102, 138–154. doi: 10.1016/j.envsoft.2018.01.013

Tomczak, M. T., Heymans, J. J., Yletyinen, J., Niiranen, S., Otto, S. A., and Blenckner, T. (2013). Ecological network indicators of ecosystem status and change in the Baltic Sea. *PloS One* 8 (10), e75439. doi: 10.1371/journal.pone.0075439

Tomczak, M. T., Müller-Karulis, B., Järv, L., Kotta, J., Martin, G., Minde, A., et al. (2009). Analysis of trophic networks and carbon flows in south-eastern Baltic coastal ecosystems. *Prog. Oceanogr.* 81, 111–131. doi: 10.1016/j.pocean.2009.04.017

Tomczak, M. T., Niiranen, S., Hjerne, O., and Blenckner, T. (2012). Ecosystem flow dynamics in the Baltic proper–using a multi-trophic dataset as a basis for food–web modelling. *Ecol. Model.* 230, 123–147. doi: 10.1016/j.ecolmodel.2011.12.014

Torres, M. A., Coll, M., Heymans, J. J., Christensen, V., and Sobrino, I. (2013). Food-web structure of and fishing impacts on the gulf of cadiz ecosystem (South-western Spain). *Ecol. Indic.* 265, 26–44. doi: 10.1016/j.ecolmodel.2013.05.019

Townsend, M., Thrush, S. F., Lohrer, A. M., Hewitt, J. E., Lundquist, C. J., Carbines, M., et al. (2014). Overcoming the challenges of data scarcity in mapping marine ecosystem service potential. *Ecosystem Serv.* 8, 44–55. doi: 10.1016/j.ecoser.2014.02.002

Trochta, J. T., Pons, M., Rudd, M. B., Krigbaum, M., Tanz, A., and Hilborn, R. (2018). Ecosystem-based fisheries management: perception on definitions, implementations, and aspirations. *PloS One* 13 (1), e0190467. doi: 10.1371/journal.pone.0190467

Tsagarakis, K., Coll, M., Giannoulaki, M., Somarakis, S., Papaconstantinou, C., and Machias, A. (2010). Food web traits of the north Aegean Sea ecosystem (Eastern Mediterranean) and comparison with other Mediterranean ecosystems. *Estuarine Coast. Shelf Sci.* 88 (2), 233–248. doi: 10.1016/j.ecss.2010.04.007

Tsagarakis, K., Libralato, S., Giannoulaki, M., Touloumis, K., Somarakis, S., Machias, A., et al. (2022). Drivers of the north Aegean Sea ecosystem (Eastern Mediterranean) through time: insights from multidecadal retrospective analysis and future simulations. *Front. Mar. Sci.* 9, 919793. doi: 10.3389/fmars.2022.919793

Tsagarakis, K., Panigada, S., Machias, A., Giannoulaki, M., Foutsi, A., Pierantonio, N., et al. (2021). Trophic interactions in the "small pelagic fish - dolphins - fisheries" triangle: outputs of a modelling approach in the north Aegean Sea (Eastern Mediterranean, Greece). *Ocean Coast. Manage.* 204, 105474. doi: 10.1016/j.ocecoaman.2020.105474

Tsiaras, K. P., Kourafalou, V. H., Raitsos, D. E., Triantafyllou, G., Petihakis, G., and Korres, G. (2012). Inter-annual productivity variability in the north Aegean Sea:

influence of thermohaline circulation during the Eastern Mediterranean transient. *J. Mar. Syst.* 96-97, 72–81. doi: 10.1016/j.jmarsys.2012.02.003

Ulanowicz, R. E. (1986). *Growth and development: ecosystem phenomenology* (New York: Springer Verlag (reprinted by iUniverse, 2000), 203.

Ulanowicz, R. E. (2004). Quantitative methods for ecological network analysis. *Comput. Biol. Chem.* 28 (5-6), 321–339. doi: 10.1016/j.compbiolchem.2004.09.001

Ulanowicz, R. E., and Kemp, W. M. (1979). Toward canonical trophic aggregations. *Am. Nat.* 114, 871–883. doi: 10.1086/283534

Ulanowicz, R. E., and Puccia, C. J. (1990). Mixed trophic impacts in ecosystems. *Coenoses* 5 (1), 7–16.

Väli, G., Meier, H. E. M., and Elken, J. (2013). Simulated halocline variability in the Baltic Sea and its impact on hypoxia during 1961–2007. *J. Geophys. Res.* 118 (12), 6982–7000. doi: 10.1002/2013JC009192

Valls, A., Coll, M., and Christensen, V. (2015). Keystone species: toward an operational concept for marine biodiversity conservation. *Ecol. Monogr.* 85 (1), 29–47. doi: 10.1890/14-0306.1

Valls, A., Gascuel, D., Guénette, S., and Francour, P. (2012). Modeling trophic interactions to assess the effects of a marine protected area: case study in the NW Mediterranean Sea. *Mar. Ecol. Prog. Ser.* 456, 201–214. doi: 10.3354/meps09701

Vanalderweireldt, L., Albouy, C., Le Loc'h, F., Millot, R., Blestel, C., Patrissi, M., et al. (2022). Ecosystem modelling of the Eastern Corsican coast (ECC): case study of one of the least trawled shelves of the Mediterranean Sea. *J. Mar. Syst.* 235, 103798. doi: 10.1016/j.jmarsys.2022.103798

Varkitzi, I., Psarra, S., Assimakopoulou, G., Pavlidou, A., Krasakopoulou, E., Velaoras, D., et al. (2020). Phytoplankton dynamics and bloom formation in the oligotrophic Eastern Mediterranean: field studies in the Aegean, levantine and Ionian seas. *Deep Sea Res. Part II: Topical Stud. Oceanogr.* 171, 104662. doi: 10.1016/j.dsr2.2019.104662

Vassallo, P., Bellardini, D., Castellano, M., Dapueto, G., and Povero, P. (2022). Structure and functionality of the mesozooplankton community in a coastal marine environment: portofino marine protected area (Liguria). *Diversity* 14 (1), 19. doi: 10.3390/d14010019

Vassallo, P., D' Onghia, G., Fabiano, M., Maiorano, P., Lionetti, A., Paoli, C., et al. (2017). A trophic model of the benthopelagic fauna distributed in the Santa maria di leuca cold-water coral province (Mediterranean Sea). *Energy Ecol. Environ.* 2, 114–124. doi: 10.1007/s40974-016-0047-2

Vassallo, P., Fabiano, M., Vezzulli, L., Sandulli, R., Marques, J. C., and Jørgensen, S. E. (2006). Assessing the health of coastal marine ecosystems: a holistic approach based on sediment micro and meio-benthic measures. *Ecol. Indic.* 6 (3), 525–542. doi: 10.1016/j.ecolind.2005.07.003

Vassallo, P., Paoli, C., and Fabiano, M. (2012). Ecosystem level analysis of sandy beaches using thermodynamic and network analyses: a study case in the NW Mediterranean Sea. *Ecol. Indic.* 15 (1), 10–17. doi: 10.1016/j.ecolind.2011.09.016

Vassallo, P., Paoli, C., Schiavon, G., Albertelli, G., and Fabiano, M. (2013). How ecosystems adapt to face disruptive impact? the case of a commercial harbor benthic community. *Ecol. Indic.* 24, 431–438. doi: 10.1016/j.ecolind.2012.07.021

Vasslides, J. M., de Mutsert, K., Christensen, V., and Townsend, H. (2017). Using the ecopath with ecosim modeling approach to understand the effects of watershed-based management actions in coastal ecosystems. *Coast. Manage.* 45 (17), 44–55. doi: 10.1080/08920753.2017.1237241

Veiga-Malta, T., Szalaj, D., Angélico, M. M., Azevedo, M., Farias, I., Garrido, S., et al. (2019). First representation of the trophic structure and functioning of the Portuguese continental shelf ecosystem: insights into the role of sardine. *Mar. Ecol. Prog. Ser.* 617-618, 323–340. doi: 10.3354/meps12724

Vieites, D. R., Nieto-Román, S., Palanca, A., Ferrer, X., and Vences, M. (2004). European Atlantic: The hottest oil spill hotspot worldwide. *Sci. Nat.* 91, 535–538. doi: 10.1007/s00114-004-0572-2

Vilas, D., Coll, M., Corrales, X., Steenbeek, J., Piroddi, C., Macias, D., et al. (2021). Current and potential contributions of the gulf of lion fisheries restricted area to fisheries sustainability in the NW Mediterranean Sea. *Mar. Policy* 123, 104296. doi: 10.1016/j.marpol.2020.104296

Vilas, D., Coll, M., Pedersen, T., Corrales, X., Filbee-Dexter, K., Pedersen, T., et al. (2020a). Kelp-carbon uptake by Arctic deep-sea food webs plays a noticeable role in maintaining ecosystem structural and functional traits. *J. Mar. Syst.* 203, 103268. doi: 10.1016/j.jmarsys.2019.103268

Vilas, D., Coll, M., Pedersen, T., Corrales, X., Filbee-Dexter, K., and Wernberg, T. (2020b). Future trajectories of change for an Arctic deep-sea ecosystem connected to coastal kelp forests. *Restor. Ecol.* 29 (S2), e13327. doi: 10.1111/rec.13327

Villanueva, M. C., Ernande, B., and Mackinson, S. (2009). *Trophic network. channel habitat atlas for marine resource management, final report*. Eds. A. Carpentier, C. S. Martin and S. Vaz (France: IFREMER, Boulogne-sur-mer), 547–562.

Villasante, S., Arreguín-Sánchez, F., Heymans, J. J., Libralato, S., Piroddi, C., Christensen, V., et al. (2016). Modelling marine ecosystems using the ecopath with ecosim food web approach: new insights to address complex dynamics after 30 years of developments. *Ecol. Model.* 331, 1–4. doi: 10.1016/j.ecolmodel.2016.04.017

Vincent, C., Huon, M., Caurant, F., Dabin, W., Deniau, A., Dixneuf, S., et al. (2017). Grey and harbour seals in France: distribution at sea, connectivity and trends in abundance at haulout sites. *Deep Sea Res. Part II: Topical Stud. Oceanogr.* 141, 294–305. doi: 10.1016/j.dsr2.2017.04.004

Vuorinen, I., Hänninen, J., Rajasitla, M., Laine, P., Eklund, J., Montesino-Pouzols, F., et al. (2015). Scenario simulations of future salinity and ecological consequences in the Baltic Sea and adjacent north Sea areas–implications for environmental monitoring. *Ecol. Indic.* 50, 196–205. doi: 10.1016/j.ecolind.2014.10.019

Walters, W. J., and Christensen, V. (2018). Ecotracer: analyzing concentration of contaminants and radioisotopes in an aquatic spatial-dynamic food web model. *J. Environ. Radioactivity* 181, 118–127. doi: 10.1016/j.jenvrad.2017.11.008

Walters, C., Christensen, V., and Pauly, D. (1997). Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Rev. Fish Biol. Fish.* 7, 139–172. doi: 10.1023/A:1018479526149

Walters, C., Pauly, D., and Christensen, V. (1999). Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. *Ecosystems* 2, 539–554. doi: 10.1007/s100219900101

Watson, S. C. L., Beaumont, N. J., Widdicombe, S., and Paterson, D. M. (2020). Comparing the network structure and resilience of two benthic estuarine systems following the implementation of nutrient mitigation actions. *Estuarine Coast. Shelf Sci.* 244, 106059. doi: 10.1016/j.ecss.2018.12.016

Woods, J. S., and Verones, F. (2019). Ecosystem damage from anthropogenic seabed disturbance: a life cycle impact assessment characterisation model. *Sci. Total Environ.* 649, 1481–1490. doi: 10.1016/j.scitotenv.2018.08.304

Zeller, D., and Reinert, J. (2004). Modelling spatial closures and fishing effort restrictions in the faroe islands marine ecosystem. *Ecol. Model.* 172 (2-4), 403–420. doi: 10.1016/j.ecolmodel.2003.09.020

Zenetos, A., Çinar, M. E., Crocetta, F., Golani, D., Rosso, A., Servello, G., et al. (2017). Uncertainties and validation of alien species catalogues: the Mediterranean as an example. *Estuarine Coast. Shelf Sci.* 191, 171–187. doi: 10.1016/j.ecss.2017.03.031

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EDITED BY
Georgios Sylaios,
Democritus University of Thrace, Greece

REVIEWED BY
Bernardo Patti,
National Research Council (CNR), Italy
Gideon Gal,
Israel Oceanographic and Limnological Research, Israel

*CORRESPONDENCE
Paula Sánchez-Zulueta
paula.sanchez.zulueta@ieo.csic.es
María Valls
maria.valls@ieo.csic.es

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Trophic structure and fishing impacts on an oligotrophic ecosystem in the Western Mediterranean: the Balearic Islands

Paula Sánchez-Zulueta[1]*, María Valls[1]*, Beatriz Guijarro[1], María Ángeles Torres[2], María Ángeles Zapata[1], Marta Coll[3], Xavier Corrales[4], Eider Andonegi[4], Marta Díaz-Valdés[5], Enric Massutí[1] and Francesc Ordines[1]

[1]Centre Oceanogràfic de les Balears (COB-IEO), CSIC, Palma, Spain, [2]Centro Oceanográfico de Cádiz (IEO), CSIC, Cádiz, Spain, [3]Institut de Ciències del Mar (ICM), CSIC, Ecopath International Initiative (EII), Barcelona, Spain, [4]AZTI, Marine Research Division, Basque Research and Technology Alliance (BRTA), Sukarrieta, Spain, [5]Marine Research Division, Agrupació Son Navata, Felanitx, Illes Balears, Spain

Within the context of the ecosystem approach to fisheries management, an ecosystem model was developed for the Balearic Islands in the early 2000s, covering from 0 to 800 m of depth. The aim of the study was to describe the structure and functioning of the ecosystem together with the fishing impacts. The results show that the biomass of the primary producers (PP), mainly dominated by *Posidonia oceanica* meadows and red algae beds, represented a high percentage of the biomass (39.20%) in the ecosystem. Most of the trophic flows occurred between PP (mostly benthic) and trophic level II. The mixed trophic impact analysis also highlighted a positive impact of the functional groups (FGs) at the base of the food web. Besides, there were important trophic flows between several demersal FGs, and many groups of the pelagic compartment, confirming the importance of the benthopelagic coupling previously reported. The mean trophic level of the catch (mTLc=2.76) was lower than those reported in adjacent areas due to the discard of benthic producers. By excluding benthic PP, the Balearic Islands showed the highest mTLc (3.44) among all the Mediterranean areas modelled, probably due to differences in the trophic structure, lower fisheries exploitation levels and lower development of the purse seine fleet targeting small pelagic fishes. The mean transfer efficiency (21.80%) was double than the global average (10%), which highlights the oligotrophy of the system, the importance of demersal production which has lower rates of transfer through the food web, and a more efficient use of nutrients in the ecosystem. Sponges, one of the most important groups of benthic invertebrates in the Balearic Archipelago, are known to recycle dissolved organic matter, which is reintroduced into the food web similar to the microbial loop, but the magnitude of this and other processes remains still unknown in this area. The present study is a first step to develop spatio-temporal simulations under different exploitation scenarios and to calculate ecological indicators to assess the state of the marine environment in this region.

KEYWORDS

Balearic Islands, ecopath model, food web, trophic flows, fishing impacts, comparative model approach

1 Introduction

The need to apply an ecosystem approach to fisheries management (EAFM) is globally accepted. This holistic approach allows fisheries assessment and management to take into account the physical, biological and socio-economic complexities of living resources (Pikitch et al., 2004). The complementarity of EAFM to single-species assessment and management takes into account the maximisation and sustainability of the target species yields. However, EAFM approaches are still largely lacking in fisheries assessment and management (Patrick and Link, 2015). This situation is evident in the Mediterranean, where fisheries assessment and management is largely based on the single-species approach of few assessed stocks (GFCM, 2022; STECF, 2022). The current application of the European Multiannual Plan for fisheries exploiting demersal stocks in the Western Mediterranean (MAP; Regulation (EU) 2019/1022 of the Parliament and of the Council) is a clear example of the continued dominance of the single-species approach in Mediterranean fisheries, which uses the Maximum Sustainable Yield of the main target species as the main reference point for the implementation of management measures. This approach, which is perhaps the simplest but not the most effective given the diversity and complexity of the Mediterranean fisheries (Caddy, 1993; Farrugio et al., 1993; Lleonart and Maynou, 2003), may be one of the reasons for the long-term history of generalised overfishing in the area (Colloca et al., 2013).

Key tools for the implementation of the EAFM are marine ecosystem models. In this sense, trophodynamic models have become key tools for the multidimensional assessments required by the EAFM (Longo et al., 2015), as they allow the integration of available information on trophic relationships, interactions with environmental factors and anthropogenic pressures such as fishing. Among them, the Ecopath with Ecosim (EwE) approach (Christensen and Walters, 2004) is one of the most used ecological models worldwide and in the Mediterranean (Coll and Libralato, 2012; Colléter et al., 2015; Keramidas et al., 2023), where ecological models may be particularly important (Caddy, 1993; Lleonart and Maynou, 2003).

In the Western Mediterranean Sea, several Ecopath models have been implemented for the ecosystems of the mainland shore, including coastal, shelfs and deep sea areas (Coll et al., 2006; Coll et al., 2008; Banaru et al., 2013; Tecchio et al., 2013; Corrales et al., 2015; Coll et al., 2021). However, the functioning of the insular ecosystems in their eastern part (i.e. the Balearic Islands) is still unknown, which contrasts with the abundant information on their trophic webs (e.g. Cartes et al., 2008a; Madurell et al., 2008; Fanelli et al., 2009; Valls et al., 2014; Valls et al., 2017).

Due to their specificities, the Balearic Islands are considered as an individualised area for assessment and management purposes in the Western Mediterranean (Quetglas et al., 2012): the Geographical Subarea 5 (GSA 5) of the General Fisheries Commission for the Mediterranean (GFCM). The most important aspects for this differentiation are: (1) Geomorphology: The Balearic Islands are one of the most distant insular areas in the Mediterranean, separated from the mainland by 200 km and by bottom depths up to about 2000 m, except in the Ibiza Channel, the area closest to the mainland, where the maximum depth is about 800 m. The Archipelago is divided into two physiographic provinces, Eivissa-Formentera and Mallorca-Menorca, which are separated by the Mallorca Channel, with maximum depths of more than 600 m (Acosta et al., 2002); (2) Oligotrophy: the waters around the Archipelago, where there is no supply of nutrients from river runoff, are more oligotrophic than the adjacent waters off the Iberian coast and the Gulf of Lions (Estrada, 1996; Bosc et al., 2004); (3) Sediment types: the sediments on the Balearic shelf are mainly biogenic sands and gravels (Acosta et al., 2002), in contrast to the predominance of terrigenous mud on the mainland; (4) The presence of habitats: due to the clear waters in the Archipelago, the algae beds predominate down to a depth of 90 m, overlapping with the shallow shelf bottom trawl fishery, including essential and sensitive habitats such as the *Peyssonnelia* and rhodoliths beds, respectively (Ballesteros, 1992; Ballesteros, 1994; Ordines et al., 2009; Barberá et al., 2012; Ordines, 2015; Ordines et al., 2017); (5) The exploitation state of resources and ecosystems, which is better than in the adjacent mainland area due to a historically relatively less developed fishing fleet, in which the artisanal vessels represent a higher percentage than on the mainland (80% vs 60%, respectively) (Quetglas et al., 2012); and (6) The general lack of interaction with mainland fishing fleets, particularly in Mallorca and Menorca, where the fleets targeting demersal and small pelagic species are entirely local (Quetglas et al., 2012).

This study represents a first step in the characterisation of the Balearic Islands with an ecosystem view, including the Mallorca and Menorca subunit, using an ecosystem model. Our main objectives are: (1) describe the structure and functioning of the ecosystem using the Ecopath food-web model in early 2000s, (2) assess the impact of fishing activity using ecological indicators, and (3) describe the main specificities and differences with other modeled ecosystems in the Western Mediterranean Sea and adjacent waters.

2 Materials and methods

2.1 Study area and period

The Ecopath model developed in this study considered the Mallorca and Menorca subunit of the Balearic Promontory, and represents an average annual situation of the early 2000s (2001–2003; Figure 1). The Balearic Islands model includes the continental shelf and the middle slope (between 0 and 800 m), covering a total area of 15,197 km^2. The islands of Ibiza and Formentera were not included in the model because of their different ecological and oceanographic conditions, but also due to data limitations. The Ibiza and Formentera sub-unit is still a poorly known area, where the international bottom trawl survey in the Mediterranean (MEDITS) started in 2021.

The Balearic Islands are the most distant insular area in the Western Mediterranean, separated from the Iberian Peninsula by a minimum distance of 95 nautical miles and with depths ranging from 800 to almost 2000 m (Acosta et al., 2002). The Archipelago delimits the Balearic sub-basin in the north from the Algerian sub-basin in the south. These sub-basins are characterised by different oceanographic conditions (Lehucher et al., 1995) and are connected by a series of channels that play an important role in the regional circulation, as passages for the exchange of water masses between them. Mesoscale frontal events between Mediterranean and Atlantic waters (Pinot et al., 1995) and the input of cold northern waters into the channels (Fernández de Puelles et al., 2004), may act as external fertilisation mechanisms that increase productivity off the Balearic Islands. The Archipelago has no rivers, and consequently the sediments of its shelf are mainly biogenic sands and gravels, with a high percentage of carbonates (Canals and Ballesteros, 1997). The lack of supply of nutrients from land runoff makes the oligotrophy around the Balearic Islands more pronounced than in the adjacent waters off the Iberian coast and the Gulf of Lions (Estrada, 1996; Bosc et al., 2004).

The main fisheries in the area are those exploited by the bottom trawl fleet and the artisanal fleet (Quetglas et al., 2016b). The bottom trawl fishery is highly multispecific and operates between 50 and 800 m depth, particularly in the continental shelf, targeting more than 60 species of fish, cephalopods and crustaceans (Palmer et al., 2009; Ordines et al., 2014). The main target species in the continental shelf and shelf break are the striped red mullet *Mullus surmuletus* and the European hake *Merluccius merluccius*, in the upper slope are the deep-rose shrimp *Parapenaeus longirostris* and the Norway lobster *Nephrops norvegicus*, and the red shrimp *Aristeus antennatus* in the middle slope. The artisanal fleet operates mainly on the continental shelf, using a variety of gears including trammel nets and gillnets (mainly targeting *M. surmuletus*, the spiny lobster *Palinurus elephas*, and the cuttlefish *Sepia officinalis*), handlines (targeting the squid *Loligo vulgaris*), long lines (mainly targeting *Dentex dentex*, sparid fishes and the scorpionfish *Scorpaena scrofa*), and specific purse seines targeting the dolphinfish *Coryphaena hippurus* and the transparent goby *Aphia minuta* (Quetglas et al., 2016b). The total landings of the commercial fleet were around 4000 t per year, but in addition to these catches there is a very important recreational fishery with catches ranging from 1200 to 2700 t per year (Morales-Nin et al., 2005; Morales-Nin et al., 2008).

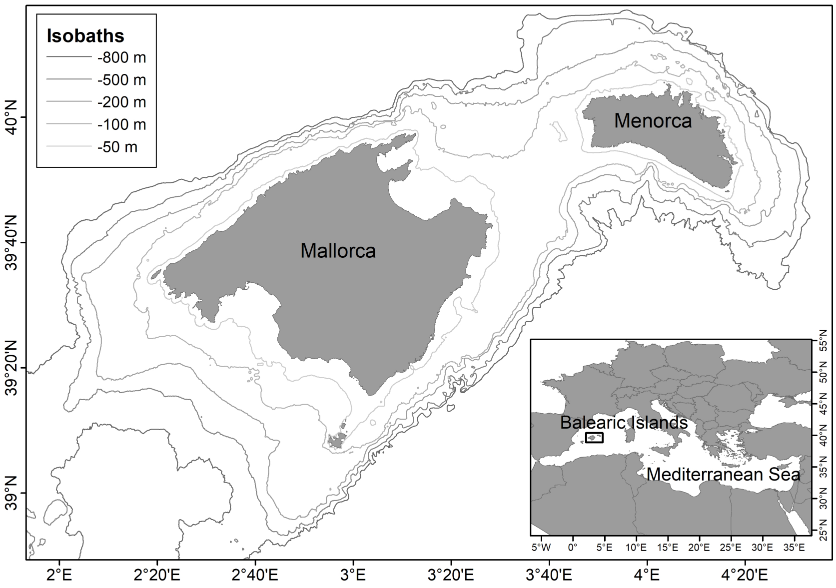

FIGURE 1
Mallorca and Menorca study area, situated in the Western Mediterranean Sea.

2.2 Food-web model description: equations and basic parameters

The food-web model of the Balearic Islands was constructed using the Ecopath with Ecosim (EwE) software version 6.6.8 (Christensen and Walters, 2004; Christensen et al., 2008). We used the static component Ecopath that allowed us to provide a snapshot representation of the food web in terms of biomass and energy flows during the studied period.

Ecopath is based on the mass-balance assumption over the time period and it is parameterized by two master equations, describing the biological production (Eq. 1) and consumption (Eq. 2) of each functional group (thereafter FG).

$$\begin{aligned} Production &= predation\ \ mortality + fishing\ \ mortality \\ &\quad + other\ \ mortality + biomass\ \ accumulation \\ &\quad + net\ \ migration \end{aligned} \tag{1}$$

$$\begin{aligned} Consumption &= production + respiration \\ &\quad + unassimilated\ \ food \end{aligned} \tag{2}$$

Within each FG, three of the four basic parameters are required: biomass of group i (B_i), production/biomass ratio (P/B_i), consumption/biomass ratio (Q/B_i) and the ecotrophic efficiency (EE_i) defined as the fraction of the production that is used in the system (Christensen et al., 2008). The fourth parameter is estimated through the mass-balance routine. Moreover, the catch by fleet and functional group (Y_j) and the diet (DC_{ij}) of all groups are also needed. Further details on the algorithms and equations, and the main advantages and limitations of the approach are described in Christensen and Walters (2004); Christensen et al. (2008) and Heymans et al. (2016).

2.3 Input data and sources

2.3.1 Functional groups

In this model, the ecosystem is defined according to functional groups (FGs) trophically linked to each other. FGs can be constituted by a group of species with similar ecological features (i.e. habitats, feeding habits), a single species, or representing different ontogenetic phases of a species (i.e. juveniles and adults) (Christensen et al., 2008; Heymans et al., 2016). The definition of FGs in the Balearic Islands model was based on the commercial importance of the species, data availability and ecological traits. The main species targeted by the fishery (dolphinfish, hake, anglerfish, mullets, red shrimp, deep-rose shrimp and Norway lobster) were modeled separately. One multi-stanza FG was created for European hake (*Merluccius merluccius*) based on juvenile hake population, i.e. < 25 cm of total length and adult hake, i.e. ≥ 25 cm, with an age of transition of 18 months according to the National Programme of collection, management and use of data in the fisheries sector (Mellon-Duval et al., 2010) (Figure 1A). Overall, up to 994 species were grouped into 45 FGs, including 1 group of dolphins, 1 group of seabirds, 20 groups of fishes, 17 groups of invertebrates, 3 groups of primary producers (benthic primary producers, seagrass and phytoplankton) and 2 groups of detritus (discards and detritus) (Table 1).

2.3.2 Input data

Input parameters (biomass, production, consumption, diet composition and fisheries data) were principally obtained from published literature and unpublished information, mostly taken from local field studies of the Spanish Institute of Oceanography (IEO) (Table 2; Table A.1).

Biomasses were obtained from different sources and methodologies: (1) for demersal and benthopelagic species, distributed between 50 and 800 m depth, biomasses were estimated from information obtained from the BALAR bottom trawl research surveys conducted during spring in 2001, 2002 and 2003, which followed the MEDITS sampling protocol (Bertrand et al., 2002; Spedicato et al., 2019), including the stratified sampling strategy (Massutí and Renones, 2005); (2) for small benthic species (e.g. crabs, shrimps, echinoderms) and algae (e.g. benthic primary producers), biomasses were obtained from beam trawl research surveys in the continental shelf (CANAL, INDEMARES and DRAGONSAL projects; Barberá et al., 2012; Domínguez et al., 2013) and the slope (IDEADOS project: Massutí et al., 2014); (3) for coastal fish species (0-50 m) (Coll et al., 2003; Morey et al., 2003; Mallol and Goñi, 2004; Coll et al., 2007), marine mammals (Forcada et al., 2004), seabirds (Arcos et al., 2009) and sea turtles (Cardona et al., 2005) biomasses were taken from visual census; (4) phytoplankton biomass was calculated through images of chlorophyll-a data *via* satellite (MODIS). Biomass of microzooplankton & mesozooplankton was obtained from data estimated in the study area (Fernandez de Puelles et al., 2003; Fernandez de Puelles et al., 2007); (5) detritus biomass was calculated by the empirical equation of Christensen and Pauly (1993); and (6) for those FGs without biomass data in the area (FGs: medium pelagics, dolphinfish, mesopelagic fishes, suprabenthos and macrozooplankton) biomasses were estimated by mass balancing using the EE default value of 0.95. Annual biomass estimates were weighed to the total area modelled taking into account the area from each stratum and then averaged for the period 2001-2003.

For the majority of FGs, production (P/B) and consumption (Q/B) rates were calculated using empirical equations. P/B parameters were obtained: (1) for target species (adult hake, mullets, red shrimp, white shrimp and Norway lobster) from fish stock assessments GSA 5; (2) for fishes, from the empirical equation of Pauly (1980) or Gascuel et al. (2011); (3) for most invertebrate species (i.e. octopus, cuttlefishes, other shrimps, crabs, echinoderms) was calculated following Brey's model (Brey, 2001); (4) for FGs with no available data (i.e. dolphins, seabirds, loggerhead turtle, suprabenthos, gelatinous plankton, macrozooplankton, microzooplankton & mesozooplankton and benthic primary producers), we used P/B values from other models and corrected for the difference of mean temperature according to Opitz (1996).

TABLE 1 Input data by functional group (FG).

FG		Bi	P/B	Q/B	EE	P/Q	U/Q
1	Dolphins	0.03	0.06	12.31	–	–	0.20
2	Seabirds	0.00	4.65	66.47	–	–	0.20
3	Loggerhead turtle	0.00	0.15	2.27	–	–	0.20
4	Pelagic sharks	0.07	–	2.69	–	0.10	0.20
5	Large pelagics	0.20	–	2.40	–	0.18	0.20
6	Medium pelagics	–	–	5.44	0.95	0.12	0.20
7	Dolphinfish	–	0.45	4.82	0.95	–	0.20
8	Macrocarnivorous fishes (shelf)	0.31	0.45	4.21	–	–	0.20
9	Juvenile Hake	0.03	1.78	9.80	–	–	0.20
10	Adult Hake	0.05	1.69	4.30	–	–	0.20
11	Anglerfish	0.03	0.80	4.00	–	–	0.20
12	Shelf Demersal fishes	1.09	1.70	6.47	–	–	0.20
13	Demersal fishes	1.00	1.59	6.93	–	–	0.20
14	Deep-sea fishes	0.60	1.51	6.34	–	–	0.20
15	Mullets	0.07	0.87	6.30	–	–	0.20
16	Flatfishes	0.29	0.98	8.06	–	–	0.20
17	Mesopelagic fishes	–	1.37	8.24	0.95	–	0.30
18	Horse mackerel	0.48	1.42	8.50	–	–	0.20
19	Sardine & anchovy	0.60	1.83	9.15	–	–	0.30
20	Benthopelagic feeders	2.00	1.37	9.01	–	–	0.30
21	Demersal sharks (shelf)	0.09	0.75	6.18	–	–	0.20
22	Demersal sharks (slope)	0.07	0.65	5.94	–	–	0.20
23	Rays & skates	0.07	0.95	4.17	–	–	0.20
24	Octopus	0.41	1.24	5.27	–	–	0.13
25	Cuttlefishes	0.21	1.48	6.38	–	–	0.20
26	Squids	0.34	1.31	6.29	–	–	0.40
27	Bivalves & gastropods	3.47	1.04	4.26	–	–	0.40
28	Red shrimp	0.04	1.23	9.44	–	–	0.20
29	White shrimp	0.02	2.15	10.03	–	–	0.20
30	Norway lobster	0.03	0.59	6.37	–	–	0.20
31	Lobsters	0.08	1.07	4.51	–	–	0.20
32	Other shrimps	3.33	2.86	12.48	–	–	0.20
33	Crabs (Reptantia)	2.42	2.71	8.98	–	–	0.20
34	Echinodermata	2.62	0.86	2.61	–	–	0.30
35	Other benthic invertebrates	2.44	3.48	12.64	–	–	0.40
36	Annelids	1.54	2.82	10.29	–	–	0.40
37	Suprabenthos	–	–	46.53	0.95	0.30	0.30
38	Gelatinous plankton	0.39	14.13	51.42	–	–	0.20
39	Macrozooplankton	–	20.78	51.88	0.95	–	0.20

(Continued)

TABLE 1 Continued

FG		Bi	P/B	Q/B	EE	P/Q	U/Q
40	Micro & mesozooplankton	5.00	43.42	125.80	–	–	0.20
41	Benthic primary producers	35.09	1.12	–	–	–	–
42	Seagrass	19.82	2.00	–	–	–	–
43	Phytoplankton	5.47	168.26	–	–	–	–
44	Discards	0.84	–	–	–	–	–
45	Detritus	43.14	–	–	–	–	–

Bi, initial biomass (t·km^{-2}); P/B, production/biomass (year^{-1}); Q/B, consumption/biomass (year^{-1}); EE, Ecotrophic Efficiency; P/Q, production/consumption; U/Q, unassimilated food/consumption.

TABLE 2 Modified data input and outputs obtained from the model of the Balearic Islands.

FG		TL	Bf	P/B	Q/B	EE	P/Q	F	M2	M0	F/Z
1	Dolphins	4.63	0.03	0.06	12.31	0.29	0.01	0.02	0.00	0.04	0.29
2	Seabirds	2.94	0.00	4.65	66.47	0.00	0.07	0.00	0.00	4.65	0.00
3	Loggerhead turtle	3.48	0.00	0.15	2.27	0.61	0.07	0.09	0.00	0.06	0.61
4	Pelagic sharks	4.89	0.07	0.27	2.69	0.96	0.10	0.26	0.00	0.01	0.96
5	Large pelagics	4.35	0.20	0.43	2.40	0.99	0.18	0.43	0.00	0.00	1.00
6	Medium pelagics	4.10	0.13	0.65	5.44	0.95	0.12	0.15	0.47	0.03	0.23
7	Dolphinfish	4.38	0.05	0.45	4.82	0.95	0.09	0.34	0.08	0.02	0.76
8	Macrocarnivorous fishes (shelf)	4.55	0.31	0.45	4.21	0.99	0.11	0.04	0.40	0.01	0.09
9	Juvenile Hake	4.16	0.03	1.78	9.80	0.62	0.18	0.36	0.73	0.68	0.20
10	Adult Hake	4.67	0.05	1.69	4.30	0.99	0.39	0.06	1.14	0.08	0.05
11	Anglerfish	4.81	0.03	0.80	4.00	0.30	0.20	0.24	0.00	0.56	0.30
12	Shelf Demersal fishes	3.22	1.09	1.70	6.47	0.99	0.26	0.12	1.56	0.01	0.07
13	Demersal fishes	3.84	1.00	1.59	6.93	0.99	0.23	0.08	1.49	0.02	0.05
14	Deep-sea fishes	3.79	0.60	1.51	6.34	0.99	0.24	0.08	1.41	0.02	0.05
15	Mullets	3.52	0.07	0.87	6.30	0.94	0.14	0.24	0.58	0.05	0.27
16	Flatfishes	3.57	0.29	0.98	8.06	0.96	0.12	0.06	0.88	0.04	0.06
17	Mesopelagic fishes	3.13	3.16	1.37	8.24	0.95	0.17	0.00	1.30	0.07	0.00
18	Horse mackerel	3.46	0.48	1.42	8.50	0.85	0.17	0.09	1.12	0.21	0.06
19	Sardine & anchovy	2.92	0.60	1.83	9.15	0.95	0.20	0.08	1.67	0.08	0.04
20	Benthopelagic feeders	3.36	2.00	1.37	9.01	0.95	0.15	0.04	1.26	0.06	0.03
21	Demersal sharks (shelf)	4.01	0.09	0.75	6.18	0.43	0.12	0.32	0.00	0.42	0.43
22	Demersal sharks (slope)	4.06	0.07	0.65	5.94	0.98	0.11	0.50	0.15	0.01	0.76
23	Rays & skates	4.00	0.07	0.95	4.17	0.78	0.23	0.63	0.11	0.20	0.66
24	Octopus	4.02	0.41	1.24	5.27	0.82	0.23	0.03	0.98	0.22	0.03
25	Cuttlefishes	4.07	0.21	1.48	6.38	0.99	0.23	0.05	1.42	0.01	0.03
26	Squids	4.16	0.34	1.31	6.29	0.96	0.21	0.02	1.24	0.05	0.02
27	Bivalves & gastropods	2.24	3.47	1.04	4.26	0.99	0.24	0.00	1.02	0.02	0.00
28	Red shrimp	3.42	0.04	1.23	9.44	0.98	0.13	0.26	0.94	0.02	0.21

(Continued)

TABLE 2 Continued

FG		TL	Bf	P/B	Q/B	EE	P/Q	F	M2	M0	F/Z
29	White shrimp	3.30	0.02	2.15	10.03	0.62	0.21	0.11	1.22	0.81	0.05
30	Norway lobster	3.45	0.03	0.59	6.37	0.99	0.09	0.04	0.54	0.00	0.07
31	Lobsters	3.31	0.08	1.07	4.51	0.97	0.24	0.09	0.95	0.04	0.08
32	Other shrimps	3.10	3.33	2.86	12.48	0.99	0.23	0.00	2.85	0.01	0.00
33	Crabs (Reptantia)	2.80	2.42	2.71	8.98	0.89	0.30	0.00	2.39	0.31	0.00
34	Echinodermata	2.02	2.62	0.86	2.61	0.48	0.33	0.04	0.37	0.45	0.05
35	Other benthic invertebrates	2.20	2.44	3.48	12.64	0.47	0.28	0.01	1.64	1.83	0.00
36	Annelids	2.04	1.54	2.82	10.29	0.63	0.27	0.00	1.77	1.05	0.00
37	Suprabenthos	2.10	1.42	13.96	46.53	0.95	0.30	–	13.26	0.70	0.00
38	Gelatinous plankton	2.69	0.39	14.13	51.42	0.57	0.27	0.00	8.03	6.10	0.00
39	Macrozooplankton	2.57	1.57	20.78	51.88	0.95	0.40	–	19.74	1.04	0.00
40	Micro & mesozooplankton	2.02	5.00	43.42	125.80	0.56	0.35	–	24.29	19.13	0.00
41	Benthic primary producers	1	35.09	1.12	–	0.14	–	0.01	0.15	0.96	0.01
42	Seagrass	1	19.82	2.00	–	0.00	–	0.00	0.00	2.00	0.00
43	Phytoplankton	1	5.47	168.26	–	0.54	–	–	90.97	77.28	0.00
44	Discards	1	0.84	–	–	0.14	–	–	–	–	–
45	Detritus	1	43.14	–	–	0.36	–	–	–	–	–

TL, trophic level; Bf, final biomass (t·km^{-2}); P/B (Z), production/biomass (year^{-1}); Q/B, consumption/biomass (year^{-1}); EE, ecotrophic efficiency; P/Q, production/consumption. F, fishing mortality (year^{-1}); M2, predation mortality (year^{-1}); M0, other natural mortality (year^{-1}); F/Z, exploitation rate.

Q/B was calculated: (1) for almost all the FGs using the empirical equation of Pauly et al. (1990); (2) for seabirds, consumption was calculated using the empirical equation proposed by Nilsson and Nilsson (1976); (3) in the case of dolphins, consumption was estimated using the empirical equation of Innes et al., 1987 and Trites et al., 1997; (4) for mesopelagic fishes FG, as no information was available, the consumption was calculated assuming their consumption is about 6 times higher than their production from the equation proposed by Christensen et al. (2008); (5) for the juvenile hake, consumption was estimated by the model (as a multistanza group); and (6) for other FGs (i.e. loggerhead turtle, octopus, cuttlefishes, squids, bivalves & gastropods, crabs (reptantia), echinodermata, other benthic invertebrates, annelids, suprabenthos, gelatinous plankton, macrozooplankton and micro-mesozooplankton), as information was not available for this area, the values were adapted from other models, which involved similar species.

The diet composition matrix used in the parameterisation of the model was constructed using local stomach content data of 71 species, including fish, cephalopods and crustaceans, obtained during the MEDITS and IDEADOS research surveys, together with published diet datasets (41 species) from the Balearic Islands (see references in Table A.2 and Table A.1). For those groups for which no trophic information available in the study area, data were collected from the literature. Migratory species (seabirds, marine turtles, pelagic sharks, medium and large pelagic fish, dolphinfish) were taken into account by modelling part of their diet composition as imports into the ecosystem, following Coll et al. (2006).

Bottom trawlers, artisanal boats, purse seiners, surface longliners and recreational boats were included in the model. Official landings were obtained from the daily sales records between 2001 and 2003, provided by OPMALLORCAMAR, the fisheries producers of Mallorca, which commercialises the catches of all the fishing fleets around the Island, and by the Regional Government of the Balearic Islands for the fishing fleets of Menorca. IUU and black market data were compiled from Carreras et al. (2015) and Estimated from Sea Around Us (seaaroundus.org) program datasets. Discards were calculated based on the data collected from sampling developed by scientific observers on board the local fishing fleet between 2001 and 2003. Finally, recreational catches were obtained from studies carried out in the Balearic Islands (Morales-Nin et al., 2005; Cabanellas-Reboredo et al., 2017), also supplemented with data from the Sea Around Us project (Carreras et al., 2015).

2.4 Pre-balancing and balancing analyses

An Ecopath model is considered mass-balanced if the following requirements are accomplished (Christensen and Walters, 2004; Christensen et al., 2008): (1) EE values are less than 1; (2) Gross efficiency of food conversion or P/Q values are between 0.1 - 0.35 with the possible exception of fast growing organisms, such as zooplankton, with larger values, and lower values for top predators; (3) Respiration/biomass ratio (R/B) values are between 1 – 10 year^{-1} for fishes and between 50 – 100 year^{-1} for planktonic organisms; (4)

Respiration/assimilation ratio (R/A) values are lower than 1 (respiration should not exceed food assimilation); (5) Production/respiration ratio (P/R) values should not be higher than 1; and (6) the net efficiency (NE) of food conversion values are lower than 1; moreover, the net efficiency should be higher than the P/Q.

To achieve the balancing of the Balearic Islands model, a manual procedure was applied, following the best practices principles of the approach (Heymans et al., 2016) and a top-down strategy. The main deviations from the initial values were related to the biomass of juvenile hake, octopus, cuttlefishes, squids, bivalves, shrimps (red shrimp, white shrimp and other shrimps), crabs and annelids groups. Therefore, biomass values were readjusted as a condition for achieving a mass balance of the model for these groups with larger EE values. We corrected the estimations of biomass for epifauna (i.e. other shrimps) as beam-trawl sampling could underestimate it by a factor of up to ten (Reiss et al., 2006).

The Prebal diagnostics introduced by Link (2010) are used to ensure that the input data follow general ecological and fisheries principles. These diagnostics guide the balancing procedure, and are applied before the model is balanced and the dynamic application is run (Scotti et al., 2022). The aim is to increase the robustness and the reliability of the results generated. For this model, the Prebal diagnosis (Figure 2A) allowed us to check and visualise B, P/B, Q/B and P/Q per all FGs.

2.5 Model analysis and ecosystem indicators

Ecological indicators were used to analyze fishing impacts based on trophic flow analysis, thermodynamic concepts, information theory and trophodynamic indicators (Christensen and Walters, 2004).

2.5.1 Pedigree index and quality of the model

The Pedigree index (Christensen and Walters, 2004) was used to quantify the uncertainty associated with the input data and to validate the choices made during the balancing process. This index provides an uncertainty value or a confidence interval (CI) for each FG, which is calculated for the overall model. This index varies from 0 (low-quality model, i.e. values estimated from other models) to 1 (high-quality model, i.e. well-sampled and high-precision local input data).

The quality of the model was evaluated by comparing model outputs to independent data. Specifically, trophic levels (TLs) estimated by the model were compared to the δ^{15}N values estimated by stable isotopes values (δ^{15}N) of 130 species belonging to 27 FGs (Figure 2). For those FGs from the model with several δ^{15}N values, they were weighted using biomass proportions of these species in each FG. TLs estimated were plotted against the FGs δ^{15}N values, and a linear regression was performed to fit a model between the two variables and test their correlation. The comparison of TLs estimated by EwE with stable isotopes has been done previously as they are highly correlated (Navarro et al., 2011; Corrales et al., 2017).

2.5.2 Ecosystem properties and trophic flows

The trophic structure of the ecosystem was represented using the Lindeman spine (Lindeman, 1942), proposed by Ulanowicz (1995), in which the system is represented in a linear food chain form where biomasses and trophic flows are represented for each TL, whereas the detritus is separated from the primary producers to show the energy amount that flows through it.

In order to evaluate the status of the ecosystem and its stage of development and maturity, ecological indicators (Table 3) described by the theory of development of Odum (1969) were used: (1) the total system throughput (TST, t·km^{-2}·year^{-1}), which estimates the total flows of the ecosystem (Ulanowicz, 1986; Christensen and Pauly, 1993; Christensen and Walters, 2004), as the sum of consumption, flows to detritus, exports and respiration; (2) the sum of all production (TP, t·km^{-2}·year^{-1}); (3) the total primary production (TPP, t·km^{-2}·year^{-1}), which represents the total primary production of all producers; (4) the ratio of total primary production to total respiration (TPP/TR), which represents the stage of maturity of the ecosystem, where production tends to exceed respiration in immature systems, resulting in a ratio greater than 1 (Odum, 1971); (5) the net system production (TPP-TR, t·km^{-2}·year^{-1}), which represents the difference between total primary production and total respiration, with lower estimates indicating a more mature ecosystem (Christensen and Walters, 2004); (6) the ratio of total primary production to total biomass (TPP/TB), which represents maturity of the ecosystem under study, as in immature ecosystems the production exceeds the biomass; (7) the ratio of total biomass to total system throughput (TB/TST), which represents the degree of maturity, as it increases with the maturity of the ecosystem under study (Odum, 1971); (8) the total biomass (TB, t·km^{-2}), excluding detritus; (9) the transfer efficiency (TE, %), that estimates the fraction of the overall flows at each TL which are either exported out of the ecosystem (e.g by the fishing activity) or transferred to higher trophic TLs as consumption; (10) the Connectance Index (CI), which is the ratio of the number of actual links to the number of possible links in a given food web, roughly proportional to the number of FGs in the system (Nee, 1990); and (11) the System Omnivory Index (SOI), indicates how feeding interactions are distributed across TLs and is defined as the average omnivory index of all consumers weighted by the logarithm of each consumer's food intake (Pauly et al., 1993).

2.5.3 The fishery: exploitation status

The Mixed Trophic Impact (MTI) routine quantifies direct and indirect trophic impacts among all FGs in the ecosystem (Ulanowicz and Puccia, 1990), including fishing fleets. The MTI was also used to quantify the impact of fishing fleets on the FGs. In addition, to evaluate the exploitation status of the fisheries, the following indicators were analysed: (1) total catch (t·km^{-2}·y^{-1}); (2) mean trophic level of the catch (mTLc), which represents the strategy of a fishery in terms of selected food-web components and is calculated as the weighted average of the TL of harvested species (Pauly et al., 1998); (3) the exploitation rate (F/Z, fishing mortality/total mortality); (4) the gross efficiency of the fishery (GE), estimated as the ratio between total catch and total primary

TABLE 3 Characteristics and ecosystem theory indices for the Balearic Islands and other neighboured models.

Statistics and indicators	BI	GA	SCS	NWM	GL	GC	Units
Characteristics of the ecosystem							
Surface	15,197	7,085	4,500	45,547	20,400	7,224	km^2
Study period	2001 – 2003	2011	1994	1999 – 2003	2000 – 2009	2009	year
Number of functional groups	45	45	40	54	40	43	
Number of fleets	5		4	8	7	3	
Depth range	0 - 800	50 - 800	50 - 400	0 - 1000	0 - 2500	15 - 800	m
Ecosystem Theory Indices							
Total system throughput (TST)	2848.79	2565.9	4038.0	3758.0	2995.0	7734.9	t·km^{-2}·year^{-1}
Sum of all consumption (TQ)	1013.78	789.0	851.7	897.3	1480.1	1946.9	t·km^{-2}·year^{-1}
Sum of all exports (TE)	539.25	511.93	1251.9	1088.1	251.7	2233.7	t·km^{-2}·year^{-1}
Sum of all respiratory flows (TR)	460.88	398.2	326.9	279.6	498.7	955.1	t·km^{-2}·year^{-1}
Sum of all flows to detritus (TFD)	834.88	866.8	1607.5	1493.1	764.6	2599.2	t·km^{-2}·year^{-1}
Sum of all production (TP)	1324.18	1095.3	1851.3	1599.9	1572.8	3704.4	t·km^{-2}·year^{-1}
Calculated total net primary production (TPP)	998.8	909.5	1577.0	1366.1	1042.4	3187.7	t·km^{-2}·year^{-1}
Total primary production/total respiration (TPP/TR)	2.2	2.3	4.8	4.9	2.1	3.3	
Net system production (TPP-TR)	537.93	511.3	1250.1	1086.5	543.7	2232.6	t·km^{-2}·year^{-1}
Total primary production/total biomass (TPP/TB)	10.4	31.3	26.7	32.0	15.1	39.8	
Total biomass/total throughput (TB/TST)	0.03	0.01	0.01	0.01	0.02	0.01	
Total biomass (excluding detritus) (TB)	96.09	29.1	58.9	42.7	68.9	80.0	t·km^{-2}
Mean transfer efficiency (mTE)	21.80	13.9	12.2	14.3	19.7	14.9	%
Connectance Index (CI)	0.28		0.20	——	0.15	0.25	
System Omnivory Index (SOI)	0.33	0.26	0.22	0.19	0.21	0.18	
Ecopath pedigree index	0.63	0.55	0.67	0.62	0.67	0.63	
Fishery Status Indices							
Total catch	1.37	1.94	5.36	4.18	2.13	4.55	t·km^{-2}·year^{-1}
Mean trophic level of the catch (mTLc)	2.76	3.16	3.12	3.13	3.24	3.32	
Gross efficiency of the fishery (GE, catch/net PP)	0.001	0.002	0.003	0.003	0.002	0.001	
Primary production required to sustain the fishery (PPR, considering PP)	1.85	16.28	36.70	12.08	13.58	12.97	%
Primary production required to sustain the fishery (PPR, considering PP+detritus)	2.66	22.67	41.99	17.36	——	16.45	%

BI, Balearic Islands; GA, Gulf of Alicante (García-Rodríguez et al., 2021); SCS, South Catalan Sea (Coll et al., 2006); NWM, North-western Mediterranean Sea (Corrales et al., 2015); GL, Gulf of Lions (Banaru et al., 2013); GC, Gulf of Cadiz (Torres et al., 2013).

production; and (5) the primary production required to sustain the fishery (PPR, %), which represents the amount of primary productivity required to sustain commercial fisheries (Christensen and Pauly, 1995) and it is calculated as:

$$PPR = 1/9 \cdot \sum_i [Y_i \cdot (1/TE)^{TL_i - 1}] \qquad (3)$$

where Yi is the catch of a given group (i), TE is the mean transfer efficiency, TLi is the trophic level of group (i) and factor 1/9 is taken as the average conversion coefficient from wet weight to g C.

2.5.4 Comparison with other food-web models in the Western Mediterranean Sea and adjacent waters

Several Ecopath models have been previously constructed in the Mediterranean Sea close to the Balearic Islands, for example: the Catalan Sea (CS) (Coll et al., 2006), the Northwestern Mediterranean Sea (NWM) (Corrales et al., 2015), the Gulf of Lions (GL) (Banaru et al., 2013), and the Gulf of Alicante (GA) (García-Rodríguez et al., 2021); including the Gulf of Cadiz (GC)

(Torres et al., 2013) in the Atlantic Ocean but closely connected to the Mediterranean (Table 3). Although most of these models were developed following similar criteria, comparisons among ecosystem models are difficult due to important differences in the characteristics of the models that can affect results (time period, fishing fleets, depth range and number and species composition of FGs (Heymans et al., 2014). Therefore, we only compared robust indicators following previous studies (Heymans et al., 2014; Corrales et al., 2015; Corrales et al., 2017). The indicators that allowed us to compare neighbouring models were: (1) total system throughput (TST, t·km^{-2}·y^{-1}), which is the sum of all consumption (TQ), sum of all exports (TE), sum of all respiratory flows (TR) and sum of all flows to detritus (TFD); (2) total net primary production (TPP, t·km^{-2}·y^{-1}); (3) total primary production/total respiration (TPP/TR); (4) net system production (TPP-TR, t·km^{-2}·y^{-1}); (5) total primary production/total biomass (TPP/TB); (6) total biomass/total system throughput (TB/TST) and (7) mean transfer efficiency (mTE, %).

3 Results

3.1 Quality and uncertainty of the model

The pedigree index had a value of 0.629 (Table 3), which indicates that the input data used in this model has an adequate quality. Most of the biomass and diet composition data, as well as landings and discards data were collected in the study area. The index values and confidence intervals used in this study are presented in Table A.2. Furthermore, the trophic levels estimated by the Ecopath model were highly and positively correlated with the δ^{15}N values (R^2 = 0.6495, p<0.001; Figure 2). Such correlation is related to the accuracy of the diet information used in the model for 27 out of 40 FGs among consumers.

3.2 Output parameters

The total biomass supported by the ecosystem (excluding detritus and discards) was 96.09 t·km^{-2} (Table 2). This biomass of producers is mainly composed of two FGs: benthic primary producers (36.52%) and seagrass (20.63%). The rest of representative percentages of biomass supported by the ecosystem are structured as follows: phytoplankton (5.69%), microzooplankton & mesozooplankton (5.20%), bivalves & gastropods (3.61%), other shrimps (3.46%) and mesopelagic fishes (3.28%), with other groups having smaller proportions.

Most FGs showed high Ecotrophic Efficiency values (EE > 0.90; Table 2), indicating that the production of each group was widely used within the ecosystem in terms of predation and/or exploitation by fisheries. For fishes, EE were high, except for anglerfish. In contrast, low values of EE (< 0.30) were observed for vulnerable species (dolphins and seabirds), indicating that a small fraction of the production of these groups is being used in the modeled ecosystem (by predation or by fishing). In addition, primary producers (benthic primary producers, seagrass and phytoplankton) also showed low EE values (0.14, 0.002 and 0.54), indicating that a large proportion of their production flows to the detritus compartment. Phytoplankton had the highest flow to

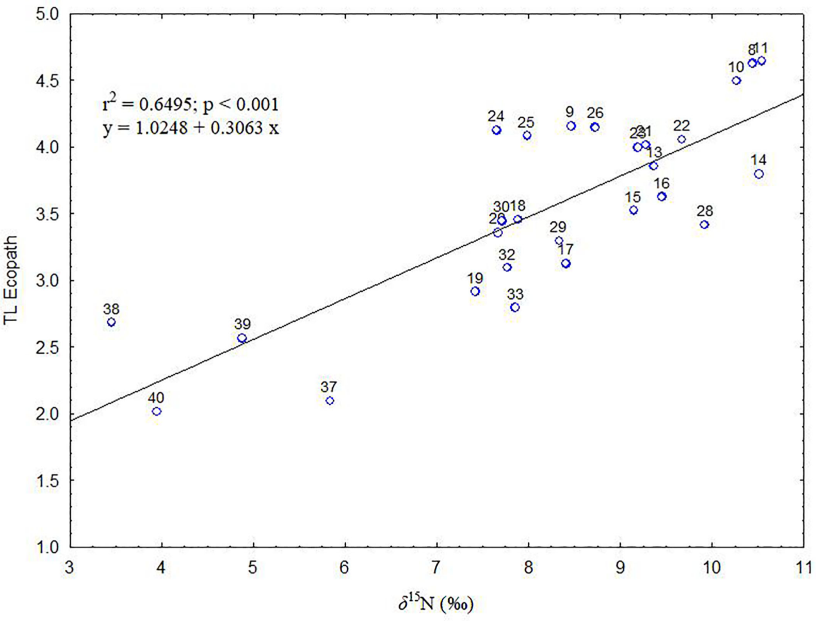

FIGURE 2
Relationship between the trophic levels (TLs) estimated by Ecopath and δ^{15}N values (mean) calculated from stable isotopes analysis in the Balearic Islands. The solid line represents the regression line and the numbers correspond to the functional groups (see Table 1).

detritus (422.5 t·km^{-2}·year^{-1}) (Table A.3) followed by microzooplankton & mesozooplankton (221.4 t·km^{-2}·year^{-1}), and both were much higher compared to seagrass (39.56 t·km^{-2}·year^{-1}) and benthic primary producers (33.77 t·km^{-2}·year^{-1}).

The values of the respiration – assimilation (R/A) (Table A.3) rate ranged from 0.50 for macrozooplankton to 0.99 for dolphins. Most of the highest values were associated with organisms with high TLs, whereas this rate was lower in organisms of lower TLs. The R/B ranged from 0.96 for echinodermata to 57.22 for microzooplankton & mesozooplankton. Most FGs had a ratio lower than 10, except suprabenthos (18.61), macrozooplankton (20.72), gelatinous plankton (27.00), seabirds (48.52) and micro & mesozooplankton. for which P/R values ranged from 0.01 (dolphins) to 1.00 (macrozooplankton). Finally, net efficiency (NE) ranged from 0.006 (dolphins) to 0.50 (macrozooplankton) and was higher than their production/consumption (P/Q) value for all FGs. The values of NE and P/Q were within the expected range (Christensen and Walters, 2004). Most of these high values were associated with fast growing groups, while they were low for dolphins, seabirds or loggerhead turtles.

3.3 Trophic levels and trophic flows

The FGs were included in four trophic levels, ranging from TL=1 for primary producers (benthic primary producers, seagrass and phytoplankton), to TL=4.9 for pelagic sharks. Cephalopods, including octopus, squids and cuttlefishes, together with demersal elasmobranch (rays, skates and sharks), displayed TL>4. For osteichthyes, the TL ranged between 2.9 for sardine and anchovy to TL≥4.5 for anglerfish (4.8), adult hake (4.7) and macrocarnivorous fishes (4.5). Decapod crustaceans encompassed less than one TL (from crabs TL=2.8 to Norway lobster TL=3.5). Plankton (gelatinous plankton, macrozooplankton and microzooplankton & mesozooplankton) and other invertebrates (echinoderms, annelids, suprabenthos and other benthic invertebrates) ranged from TL=2.1 to 2.2.

The trophic flows among the FGs observed from the Ecopath flow diagram (Figure 3) showed several links between some demersal FGs, such as juvenile and adult hake, anglerfish and demersal fishes, and many groups of the pelagic compartment, i.e. mesopelagic fishes, sardine & anchovy, gelatinous plankton, macrozooplankton and microzooplankton & mesozooplankton. The detritus appeared to be an important compartment of the ecosystem and closely linked to the demersal habitat in terms of biomass and production, based on the links of the diagram flow and on the biomass and flow values of this FG, represented in the Lindeman spine.

The Lindeman spine representation (Figure 4) showed that most of the biomasses and trophic flows were between TL I and II. The primary producers had the highest biomass (60.38 t·km^{-2}), followed by detritus (43.98 t·km^{-2}) and TL II (17.78 t·km^{-2}). The biomass located on the primary producers' compartment (TL I) was 43.08%, whereas this percentage was lower in the adjacent modeled areas: 22.7% in North-western Mediterranean, 32.2% in Gulf of Lion and 25.2% in Gulf of Cadiz. Flows obtained from primary producers to detritus (502.5 t·km^{-2}·year^{-1}) were higher than the total flows of upper TLs to detritus (336.8 t·km^{-2}year^{-1}), with the higher contribution of TL II (279.8 t·km^{-2}·year^{-1}). Flow values from primary producers were almost equally distributed between detritus (495.9 t·km^{-2}·year^{-1}) and TL II (502.5 t·km^{-2}·year^{-1}). Respiration was higher for TL II, while the highest Transfer Efficiency (TE), export and catch flows were mainly concentrated between TL III and IV.

3.4 Ecosystem properties

Statistics and ecological indicators obtained are shown in Table 3. The total system throughput (TST) was 2848.79 t·km^{-2}·year^{-1}.

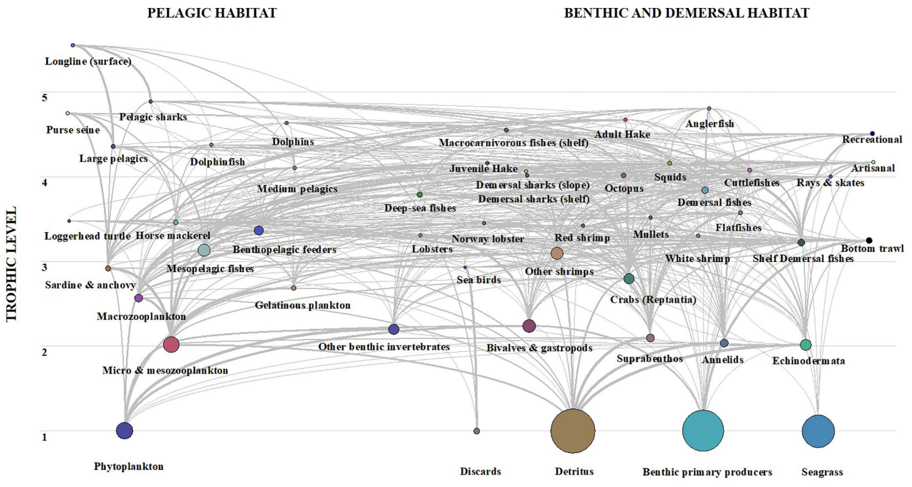

FIGURE 3
Flow diagram of Balearic Islands model separated by pelagic habitat (left) and benthic and demersal habitats (right). The size of each circle is proportional to the biomass of the FG.

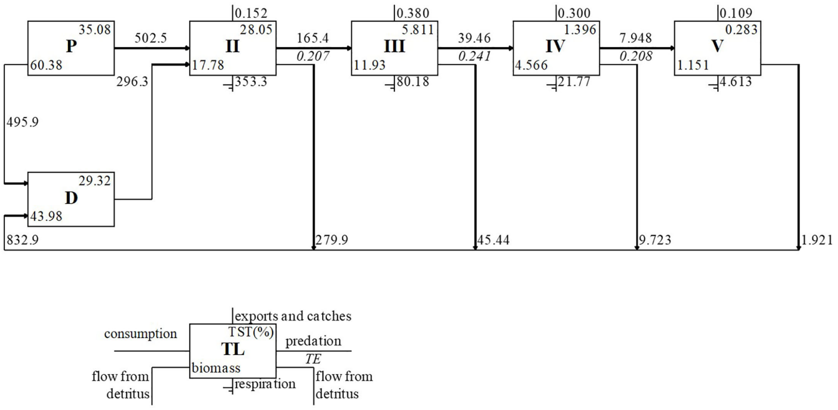

FIGURE 4
Lindeman spine representation of trophic flows from the Balearic Islands model. The flows are represented in t·km^{-2}·year^{-1}.

In terms of flows, the total consumption (35.59%) and the flow to detritus (29.31%) were higher than exports (18.93%) and respiration (16.18%), meaning that a significant quantity of biomass ends up as detritus.

The sum of all production (TP) was 1324.18 t·km^{-2}·year^{-1} and the total primary production (TPP) was 998.80 t·km^{-2}·year^{-1}. The total primary production/total respiration ratio (three times higher for the rest of ecosystems compared except for the GC, which was twice, TPP/TR), close to unity in mature ecosystems, was 2.2 t·km^{-2}·year^{-1} in the Balearic Islands, revealed that the energy produced was two times higher than the total respiration needed to maintain the balance cost. The net system production (TPP-TR) was 537.92 t·km^{-2}·year^{-1}. The total primary production/total biomass (TPP/TB) ratio in the Balearic Islands showed that the productivity was ten and a half times higher than the biomass accumulated within the ecosystem (10.4). The total biomass/total system throughputs (TB/TST) ratio shows the energy needed by the model to sustain the biomass in the Balearic Islands ecosystem and it was 0.03. The Connectance Index (CI) was 0.28 and the System Omnivory Index (SOI) was 0.33. The mean transfer efficiency (mTE) of the system was high (21.80%) and TE values had their maximum between TL III and IV, according to the Lindeman spine (Figure 4).

3.5 Fisheries impacts on the ecosystem

The MTI showed that most FGs had a negative impact on themselves and on their main preys, due to competition for food resources and predation, respectively (Figure 5). For example, adult hake had a negative impact on itself and on juvenile hake. Demersal fishes had a negative impact on horse mackerel and mullets. Octopus had a negative impact on lobsters. FGs at the base of the food web (microzooplankton & mesozooplankton, phytoplankton and detritus) concentrated most of the positive impacts on other FGs. MTI also showed that the bottom trawl and the artisanal fleets had the most widespread impacts on the ecosystem and most of the highest negative impacts on several groups, mainly from the demersal compartments (Figure 5). Bottom trawls had negative impacts on anglerfish, demersal sharks (shelf), demersal sharks (slope), rays & skates and red shrimp. The artisanal fleet had negative impacts on vulnerable groups and/or groups located at the upper part of the TLs, such as dolphins, seabirds, loggerhead turtle, dolphinfish, anglerfish, mullets, rays & skates, octopuses, cuttlefishes and red shrimp. Purse seiners (including tuna purse seiners) and the recreational fishery had negative impacts on some of their target species, large pelagics and dolphinfish, respectively. Longline had a negative impact on pelagic sharks.

The fishing mortality values (F) were higher (F>0.2) for anglerfish, mullets, pelagic sharks, red shrimp, demersal sharks (shelf), dolphinfish, juvenile hake, large pelagic fishes, demersal sharks (slope) and rays & skates, compared to the rest of FGs of the Balearic Islands (Table 2). The majority of FGs showed high predation mortality (M2) and it was the first cause of mortality for most groups of invertebrates (octopus, cuttlefishes, squids, bivalves & gastropods, red shrimp, white shrimp, Norway lobster, lobster, other shrimps, crabs and annelids) and fishes (macrocarnivorous fishes, medium pelagic fishes, mullets, juvenile hake, flatfishes, horse mackerel, adult hake, benthopelagic feeders, mesopelagic fishes, deep-sea fishes, demersal fishes, shelf demersal fishes and sardine & anchovy). Pelagic sharks, large pelagic fishes, dolphinfish, and demersal elasmobranchs (sharks, rays and skates) were the groups with higher fishing mortality than predation mortality (Table 2). The exploitation rates (F/Z) also showed high values >0.5 (i.e. more than 50% of mortality is produced by fishing) for loggerhead turtle, rays & skates, demersal sharks (slope), dolphinfish, pelagic sharks and large pelagic fishes. Some groups showed F/Z values ranging from 0.51 to 0.2: dolphins, demersal sharks (shelf), mullets, anglerfish, medium pelagic fishes, red shrimp and juvenile hake. Other FGs showed lower exploitation rates (Table 2).

The total catch in the model was 1.37 t·km^{-2}·year^{-1} (Table 2) and it was dominated mainly by benthic primary producers (28.18%) and shelf demersal fishes (9.70%). Other FGs such as rays & skates, deep-sea fishes, sardine & anchovy, demersal fishes, large pelagic fishes, benthopelagic feeders and echinoderms showed high landing values too.

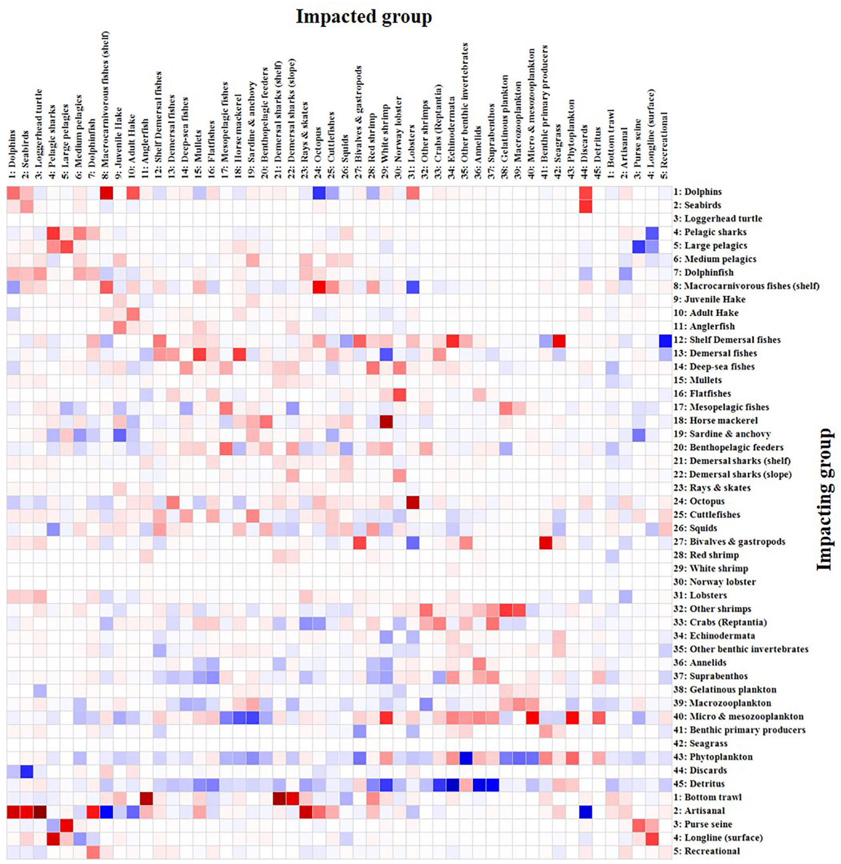

FIGURE 5
Mixed Trophic Impact (MTI) analysis of the Balearic Islands model. The size of the rectangles represents the size of the functional group trophic impact, while blue indicates positive impact and red negative impact. The five fishing fleets are also included.

The mean trophic level of the catch (mTLc) was 2.76. Bottom trawl fleet represented the highest catch in the area (59.85%), but with the lowest trophic level (mTLc = 2.24) (Table 4). On the contrary, the longline (surface) fleet had the highest mTLc (4.55) but the lowest catch. It is important to highlight the weight of the recreational fleet, representing the second highest catch in the area (14.6%) and exploiting species situated at higher trophic level than the artisanal fleet (3.51 *vs* 3.17). When benthic primary producers were excluded from the catches, the mTLc of the bottom trawl fleet increased to 3.31, resulting in a global mTLc=3.44.

Regarding the gross efficiency of the fishery (GE), the model exhibited a low value (0.001) (Table 3). The primary production required to sustain the fishery (PPR, %) was estimated at 1.85% during the period between 2001 and 2003. When this value takes into account the primary production together with the detritus, the value was 2.66% (Table 3).

4 Discussion

The different components of the marine ecosystem of the Balearic Islands have been widely studied, from the shallow continental shelf to deep waters, considering different approaches and including life history and population dynamics of the main

TABLE 4 Catch (t·km^{-2}·year^{-1}), their representative percentage (%) and mean trophic level of the catch (mTLc) for the fleets, with all FGs and excluding primary producers (P): benthic primary producers and seagrass.

Fleets of the Balearic Island	Catch	%	mTLc	Catch (excluding P)	% (excluding P)	mTLc (excluding P)
Bottom trawl	0.82	59.85	2.24	0.44	44.44	3.31
Artisanal	0.18	13.14	3.17	0.18	18.18	3.17
Purse seine	0.13	9.49	3.75	0.13	13.13	3.75
Longline (surface)	0.04	2.92	4.55	0.04	4.04	4.55
Recreational	0.20	14.60	3.51	0.20	20.20	3.51
All fleets	1.37	100	2.76	0.97	100	3.44

target species of fisheries, benthic habitats, vulnerable species and trophic dynamics (e.g. Barberá et al., 2012; Massutí et al., 2014; Valls et al., 2014; Ordines et al., 2015; Quetglas et al., 2016a; Ramírez-Amaro et al., 2020). However, the present study describes, for the first time, the structure, functioning and fishing impacts of the Mallorca and Menorca's marine ecosystem, using a food-web model.

The pedigree index obtained for the Balearic Islands model indicates that the input data used in this model has an adequate quality. In fact, it is located in the upper part of the rank of the models assessed worldwide by Colléter et al. (2015). Further, our results revealed a clear correlation between the TLs estimated by the Ecopath model and the measured stable isotope values of nitrogen (δ^{15}N), validating the assessment of Ecopath output of TL, and highlighting that the diet information used in the model for 27 out of 40 FGs among consumers was accurate (Navarro et al., 2011; Corrales et al., 2017).

Biomasses were mostly estimated from local studies, using a variety of approaches such as underwater visual census in littoral areas and different sampling gears such as bottom trawl, beam trawl, plankton nets at deeper and offshore waters. However, and despite the large amount of scientific knowledge on the marine species and their environment around the Balearic Islands, there are still gaps for some important FGs and a generalized lack of information in terms of biological parameters. For that reason, the biomass of important groups, mainly fish inhabiting the pelagic domain and species at the base of the food web (i.e., medium pelagic fishes, dolphinfish, mesopelagic fishes, suprabenthos and macrozooplankton) had to be estimated by the model. Also, many input parameters such as P/B and Q/B were estimated by the model, calculated with empirical equations or taken from other models.

The mixed trophic impact (MTI) analysis highlighted the positive impact of the FGs found at the base of the food web (detritus, phytoplankton, microzooplankton & mesozooplankton and macrozooplankton). In oligotrophic areas, such as the Balearic Islands, with little supply of terrigenous nutrients and without advective inputs of organic matter *via* submarine canyons, trophic webs are supported to a greater extent by planktonic biomass, rather than by benthic biomass (Madurell et al., 2004; Madurell and Cartes, 2005). In the coastal areas of the Archipelago, stable carbon isotopes provided evidence of the contribution of *Posidonia oceanica* as carbon source for species dwelling seagrass meadows (<35 m depth) (Cardona et al., 2007). However, in deep waters one single carbon source of pelagic origin supports the food webs of the lower shelf and the upper and mid slope (Polunin et al., 2001; Valls et al., 2014). Such energy transfer highlighted the close coupling between surface and benthic production in deep waters. In terms of trophic interactions, the nycthemeral migrating community constitutes an important food source for many abundant demersal species in the Balearic Islands (Cartes et al., 2008b; Cartes et al., 2009; Valls et al., 2011; Valls et al., 2015; Valls et al., 2017), which represents an active transport of carbon and other nutrients to the benthic habitats in the modeled area. Mesopelagic fish and suprabenthos biomass estimates showed a large difference (one order of magnitude higher) compared to other ecosystems modeled using Ecopath in the Mediterranean, which reflects the importance of such groups in the study area. In fact, both FGs play an important role in these nycthemeral migrations and in the formation of the deep scattering layer and the benthic boundary layer in our study area (Olivar et al., 2012; Peña et al., 2014). However, biomass values have to be considered with caution as they were estimated by the model.

4.1 Ecosystem structure and function

The TST (2848.79 t·km^{-2}·year^{-1}) was only compared to the Gulf of Alicante considering the number of FGs (i.e. 45 FGs). Both models reflected similar values in the aggregation of trophic flows. In terms of TST (sum of all consumption, exports, respiratory flows and flows into detritus), the sum of all consumption (35.59%) and flow to detritus (29.31%) were the most important flows in the Balearic Islands model, highlighting the amount of biomass that finishes into the detritus compartment. Trophic flows from primary producers to detritus and to TL II were 50%. The sum of TP (1324.18 t·km^{-2}·year^{-1}) and TPP (998.8 t·km^{-2}·year^{-1}) were similar to the Gulf of Alicante.

In terms of system maturity indices, the TPP/TR ratio and TPP-TR were similar to that found in the Gulf of Lions (Banaru et al., 2013) and the Gulf of Alicante (García-Rodríguez et al., 2021). The TPP/TR ratio (2.2) represents the stage of maturity of the ecosystem (Odum, 1971), and showed that the energy produced was approximately two times greater than the maintenance costs. Lower estimates of TPP/TR and TPP-TR (537.93 t·km^{-2}·year^{-1}) indicated that the Balearic Island ecosystem is slightly closer to maturity (Christensen and Walters, 2004) compared to the neighbouring models. In the case of TPP/TB (10.4), the value was the lowest among published models, reflecting a higher level of biomass accumulation in the system compared to its productivity. TB/TST value (0.03) was 3 times higher when compared to these other ecosystems.

Regarding food-web complexity indices, the Connectance Index (0.28) was similar to that obtained for the Gulf of Cadiz (Torres et al., 2013) and suggested moderate-low inner linkages within the food web. Furthermore, the System Omnivory Index (0.33) was similar to the Gulf of Alicante (García-Rodríguez et al., 2021) and relatively higher than the values obtained in other models. The System Omnivory Index indicated a more complex predator-prey relationships than other neighbouring models, but lower than global averages (Christensen and Pauly, 1995). The Balearic Island ecosystem is in a medium-low stage of development. However, according to several indices values, it could be considered as a slightly more mature ecosystem than the rest of the Mediterranean neighbouring models in comparative terms (Table 3).

The biomass located on the primary producers' compartment (TL I) was higher than in the adjacent modeled areas (North-western Mediterranean, Gulf of Lion and Gulf of Cadiz). The highest Transfer Efficiency (TE) in the present model occurred between the TL III and IV, and both compartments supported the highest and rather similar fishery exploitation. Such export flows contrast with those observed in Mediterranean and Atlantic areas where catches are mainly concentrated in TL II or III.

The mean transfer efficiency (mTE) in the Balearic Islands was 21.80%, which is higher than the average value of 10% reported worldwide (Christensen and Pauly, 1995). Such value can be explained by the oligotrophic character of the Mediterranean, more pronounced in the Balearic Islands (Bosc et al., 2004), compared to the adjacent waters off the Iberian Peninsula and the Gulf of Lions (Estrada, 1996; Bosc et al., 2004). In oligotrophic areas, food webs are known to recycle and use nutrients more efficiently through strong microbial loop (Pomeroy et al., 2007; Armengol et al., 2019).

4.2 The impact of fishing activities in the Balearic Islands

The present model showed lower values of mean trophic level of the catch (mTLc) and catches (t·km^{-2}·year^{-1}), compared to the models developed from adjacent areas. Whereas the mTLc for these models is between 3.12-3.32, in the Balearic Islands model the mTLc is 2.76. Such result was highly influenced by the low TLc of the bottom trawl fleet (2.24), which represents 59.85% of the total catch. The high biomass of benthic primary producers in the shelf bottoms, which constitute 45.9% of the catch, may explain the low TLc of the bottom trawl fleet calculated by the model. By excluding benthic primary producers and seagrass from mTLc calculation, the Balearic Islands showed the highest mTLc (3.44) among all the Mediterranean areas modelled. This may be the result of both differences in the trophic structure and a relatively better status of the ecosystem in the Balearic Islands due to a lower level of development of the fishing fleet, but also due to the scarce representation of purse seiners in this area. Purse-seiners fleet is characterized by large catches of small pelagic fish like sardine and anchovy and consequently has the lowest mTLc in the North-western Mediterranean model (Corrales et al., 2015). In adjacent areas, purse-seiners landings can represent up to 50% of the total landings in contrast to the Balearic Islands where purse-seiners landings do not reach 15% (Quetglas et al., 2012). Moreover, whereas mTLc of purse seiners in the North-western Mediterranean model shows the lowest values among the fleets operating there (2.99; Corrales et al., 2015), in the Balearic Islands its value (3.75) ranks second after longliners. In fact, even excluding algae and seagrass and from the catches, the mTLc of the bottom trawl fleet has a lower value (3.31) than in the adjacent area of the North-western Mediterranean (3.40; Corrales et al., 2015), probably due to the high importance of the middle slope fishery targeting red shrimp in the Balearic Islands, whose landings represented more than 20% of the total biomass landed by this fleet, composed mainly of crustaceans (Quetglas et al., 2012). The longline fleet, which targets large pelagic fish, represented the smallest proportion of the total catch but with the highest mTLc, similar to other longline fleets from the Western Mediterranean (Coll et al., 2006; Corrales et al., 2015). It is noteworthy the higher catches (14.60%) and mTLc (3.51) of the recreational fleet compared to the artisanal fleet in the Balearic Islands. Despite the important catches of the recreational fishery and its increasing trend (Carreras et al., 2015), there is a lack of information and robust data regarding the species targeted by this fleet. Such information is needed to properly account for its potential impact on the exploitation state on some target stocks (Quetglas et al., 2016b).

The Gross Efficiency of the Fishery (GE) for the Balearic Islands (0.001) was higher than the worldwide value (0.0002) calculated by Christensen et al. (2008) from different marine ecosystems, but similar to the values found in neighbouring areas of the Western Mediterranean and the Gulf of Cadiz (Table 3). This value points out the high impact of the fishing activity (Corrales et al., 2015) and the fact that fishing catches are less effective in converting primary production (Keramidas et al., 2022).

The primary production required to sustain the fisheries (PPR, considering PP) for the Balearic Islands ecosystem was 1.85%. This was the lowest value compared to nearby models (12-37%) (Coll et al., 2006; Banaru et al., 2013; Torres et al., 2013; Corrales et al., 2015; García-Rodríguez et al., 2021) and even compared to the usual values (24-35%) of other coastal systems throughout the world (Christensen and Pauly, 1995). This could be due to the high phytoplankton production relative to the low total catch (1.37 t·km^{-2}·year^{-1}). However, our values were similar to models from other oligotrophic areas such as those from the North Aegean Sea (3.45%; Tsagarakis et al., 2010) and the Gulf of Gabes (3.77%; Hattab et al., 2013) (Table 5).

4.3 Limitations of Balearic Islands model

There are still gaps of knowledge on basic biological parameters for many species (target and non-target species) included in this

TABLE 5 Total net primary production (TPP), total catch, Primary production required to sustain the fishery (PPR) from primary producers (a) and from primary producers and detritus (b) of the Balearic Island model (BI) compared to other models of the Aegean Sea area: North Aegean (Tsagarakis et al., 2010), and Central Mediterranean: Gulf of Gabes (Hattab et al., 2013).

Parameter	W Mediterranean	Aegean Sea area	C Mediterranean	Units
	BI	North Aegean	Gulf of Gabes	
TPP	999	535	1258	t·km^{-2}·year^{-1}
Total catch	1.37	2.35	1.72	t·km^{-2}·year^{-1}
PPR (a)	1.85	3.45	3.77	%
PPR (b)	2.66	6.76	7	%

model. In terms of biomass, whereas the data of many FGs was obtained from the MEDITS bottom trawl surveys, developed between 50 and 800 m depth, data from infralittoral bottoms shallower than 50 m depth is scarce and scattered. Biomass of demersal and benthic organisms are also better estimated (studied and sampled) than the biomass for pelagic groups located in the water column. The model had to calculate the biomass for some FGs, such as medium pelagic fishes, dolphinfish, mesopelagic fishes, suprabenthos and macrozooplankton, due to the lack of information. Finally, the biomass of phytoplankton and zooplankton, should be obtained from *in situ* sampling surveys in order to have more accurate data. This lack of information identifies potential targets of future research in the study area.

We gathered local information on the diet of many organisms in the Balearic Islands, obtained from the stomach content analyses carried out during the last decades from samples collected in the MEDITS surveys and previously published data in the area. For some FGs such as dolphins, turtles, pelagic sharks, large pelagic fishes, medium pelagic fishes, sardine & anchovy, bivalves & gastropods, annelids, suprabenthos, gelatinous plankton and zooplankton, the diet composition had to be completed with information from other areas of the Western Mediterranean. In spite of this, some studies developed in this area have revealed that the feeding behaviour of important species such as *Merluccius merluccius* (Rueda et al., 2019) or *Galeus melastomus* (Ordines et al., 2021) show differences even at medium spatial scales. Such results highlight that diet data should be improved with studies developed in the Balearic Islands in future research.

For the majority of FGs, production (P/B) and consumption (Q/B) parameters were calculated using empirical equations. For target species of fisheries developed in the Balearic Islands (adult hake, mullets, red shrimp, white shrimp and Norway lobster), Q/B parameters were obtained from the empirical equation of Pauly et al. (1990), whereas P/B parameters were obtained from the stock assessments models applied to these fisheries. This information should be updated in the future as more empirical data on FGs parameters and assessments are available.

The "other benthic invertebrates" FG considered in the present model is constituted by numerous species with many differences in ecological and biological traits. The role of some of the taxonomic groups inside this FG needs to be better determined and properly addressed in the future. For example, sponges are one of the most important groups of benthic invertebrates in the Balearic Archipelago (Ordines et al., 2017; Massutí et al., 2022). Considering the importance of the sponge loop in recycling dissolved organic carbon making it available for other organisms of the ecosystem (De Goeij et al., 2013), the magnitude of this and other processes such as the microbial loop, currently unknown in the area, should be also properly considered in the model.

Another limitation of the model is to explain how the benthic primary production flows from the infralittoral and circalittoral bottoms, which are mainly dominated by the highly diverse and productive *Posidonia oceanica* meadows and red algae beds, respectively (Ordines and Massutí, 2009; Álvarez et al., 2015), into the whole food web. In fact, both the diet matrix and the MTI analysis showed few possible ways in which the energy and matter of benthic primary producers could influence other FGs. In the model, benthic primary producers and seagrass have high biomass and low EE. According to Heymans et al. (2016), values of EE near to 0 are expected for functional groups, which suffer no predation and are not exploited by the fishery. In the Balearic Islands, benthic primary producers and seagrass are directly consumed by very few species. These peculiarities are supported by isotopic analysis and stomach contents. Further, benthic primary producers EE is higher than seagrass EE because the former can be found at depths up to 90-100 m and are harvested by bottom trawls. Much of the biomass of these FGs goes to detritus. In this sense, the magnitude of the microbial and sponges' loops (De Goeij et al., 2013) which is currently unknown in the area, may be related to the functioning of biomass flows from seagrass and benthic PP to higher trophic levels. On the other hand, benthic primary producers have positive effects in structuring benthic communities of sedimentary bottoms. They act as "Essential Fish Habitats" for the demersal resources and increase their productivity in the Balearic Islands, particularly for the species inhabiting in red algae beds (Ordines and Massutí, 2009; Ordines et al., 2009; Ordines, 2015). These indirect roles and non-trophic interactions can be represented and captured in dynamic versions of the model (Horn et al., 2021).

In fact, the Ecopath model of the Balearic Islands presented in this study will be a first step to develop temporal and spatio-temporal dynamic simulations, with the aim of evaluating the impact of different fisheries management measures, particularly those related to the implementation of the ongoing multiannual plan for the fisheries exploiting demersal stocks in the Western Mediterranean Sea (Regulation (EU) 2019/1022). It can also be useful within the context of the Marine Strategy Framework Directive, which aims to more effectively protect the European marine environment and to achieve a good environmental status, enabling a sustainable use of marine goods and services (Piroddi et al., 2015). For that, it is required the application of an ecosystem-based approach to better link ecosystem components, anthropogenic pressures and impacts on the marine environment as it has been proposed in this study.

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

PZ, MV, BG and FO conceived this work. MD-V, MV, MZ built the reference model and PZ, MV and FO updated the current model, while XC, EA, MT and MC provided support for the development. MV analyzed the stomach contents and provided the diet data. PZ, MV, MZ, BG and FO wrote the manuscript with the contribution of all the authors. All authors contributed to the article and approved the submitted version.

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Conflict of interest

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1166674/full#supplementary-material

References

Acosta, J., Canals, M., López-Martínez, J., Muñoz, A., Herranz, P., Urgeles, R., et al. (2002). The balearic promontory geomorphology (Western Mediterranean): morphostructure and active processes. *Geomorphology* 49, 177–204. doi: 10.1016/S0169-555X(02)00168-X

Álvarez, E., Grau, A. M., Marbà, N., and Carreras, D. (2015) *Praderas de angiospermas marinas de baleares*. Available at: http://hdl.handle.net/10508/10464

Arcos, J. M., Bécares, J., Rodríguez, B., and Ruíz, A. (2009). *Áreas importantes para la conservación de las Aves marinas en españa* (Madrid, Spain: Sociedad Española de Ornitologia (SEO/BirdLife).

Armengol, L., Calbet, A., Franchy, G., Rodríguez-Santos, A., and Hernández-León, S. (2019). Planktonic food web structure and trophic transfer efficiency along a productivity gradient in the tropical and subtropical Atlantic ocean. *Sci. Rep.* 9, 1–19. doi: 10.1038/s41598-019-38507-9

Ballesteros, E. (1992). Els fons rocosos profunds amb osmundaria volubilis (Linné) r. e. Norris a les balears. *Bolletí la Soc d'Història Nat. Balear.* 35, 33–50. Available at: https://www.raco.cat/index.php/BolletiSHNBalears/article/view/168869.

Ballesteros, E. (1994). The deep-water peyssonnelia beds from the Balearic islands (Western Mediterranean). *Mar. Ecol.* 15, 233–253. doi: 10.1111/j.1439-0485.1994.tb00055.x

Banaru, D., Mellon-Duval, C., Roos, D., Bigot, J. L., Souplet, A., Jadaud, A., et al. (2013). Trophic structure in the gulf of lions marine ecosystem (North-Western Mediterranean Sea) and fishing impacts. *J. Mar. Syst.* 111–112, 45–68. doi: 10.1016/j.jmarsys.2012.09.010

Barberá, C., Moranta, J., Ordines, F., Ramón, M., de Mesa, A., Díaz-Valdés, M., et al. (2012). Biodiversity and habitat mapping of menorca channel (Western Mediterranean): implications for conservation. *Biodivers. Conserv.* 21, 701–728. doi: 10.1007/s10531-011-0210-1

Bertrand, J. A., Sola, L. G. I. L. D. E., Papaconstantinou, C., Relini, G., and Souplet, A. (2002). The general specifications of the MEDITS surveys. *Sci. Mar.* 66, 9–17. doi: 10.3989/scimar.2002.66s29

Bosc, E., Bricaud, A., and Antoine, D. (2004). Seasonal and interannual variability in algal biomass and primary production in the Mediterranean Sea, as derived from 4 years of SeaWiFS observations. *Global Biogeochem. Cycles* 18, 1–17. doi: 10.1029/2003GB002034

Brey, T. (2001) *Population dynamics in benthic invertebrates. a virtual handbook*. Available at: http://www.awi-bremerhaven.de/Benthic/Ecosystem/FoodWeb/Handbook/main.html.

Cabanellas-Reboredo, M., Palmer, M., Alós, J., and Morales-nin, B. (2017). Estimating harvest and its uncertainty in heterogeneous recreational fisheries. *Fish. Res.* 188, 100–111. doi: 10.1016/j.fishres.2016.12.008

Caddy, J. F. (1993). Some future perspectives for assessment and management of Mediterranean fisheries. *Sci. Mar.* 57 (2-3), 121–130

Canals, M., and Ballesteros, E. (1997). Production of carbonate particles by phytobenthic communities on the mallorca-menorca shelf, Northwestern Mediterranean Sea. *Deep Sea Res. Part II Top. Stud. Oceanogr.* 44, 611–629. doi: 10.1016/S0967-0645(96)00095-1

Cardona, L., Revelles, M., Carreras, C., San Félix, M., Gazo, M., and Aguilar, A. (2005). Western Mediterranean Immature loggerhead turtles: habitat use in spring and summer assessed through satellite tracking and aerial surveys. *Mar. Biol.* 147, 583–591. doi: 10.1007/s00227-005-1578-9

Cardona, L., Revelles, M., Sales, M., Aguilar, A., and Borrell, A. (2007). Meadows of the seagrass posidonia oceanica are a significant source of organic matter for adjoining ecosystems. *Mar. Ecol. Prog. Ser.* 335, 123–131. doi: 10.3354/meps335123

Carreras, M., Coll, M., Quetglas, A., Massutí, E., Oliver, P., Goñi, R., et al. (2015). Estimates of total fisheries removal for the Balearic Islands, (1950–2010). *Fish. Cent. Res. Rep.* 19, 1–45.

Cartes, J. E., Madurell, T., Fanelli, E., and López-Jurado, J. L. (2008a). Dynamics of suprabenthos-zooplankton communities around the Balearic Islands (Western Mediterranean): influence of environmental variables and effects on the biological cycle of aristeus antennatus. *J. Mar. Syst.* 71, 316–335. doi: 10.1016/j.jmarsys.2006.11.013

Cartes, J. E., Maynou, F., Fanelli, E., Romano, C., Mamouridis, V., and Papiol, V. (2009). The distribution of megabenthic, invertebrate epifauna in the Balearic Basin (Western Mediterranean) between 400 and 2300 m: environmental gradients influencing assemblages composition and biomass trends. *J. Sea Res.* 61, 244–257. doi: 10.1016/j.seares.2009.01.005

Cartes, J. E., Papiol, V., and Guijarro, B. (2008b). The feeding and diet of the deep-sea shrimp aristeus antennatus off the Balearic Islands (Western Mediterranean): influence of environmental factors and relationship with the biological cycle. *Prog. Oceanogr.* 79, 37–54. doi: 10.1016/j.pocean.2008.07.003

Christensen, V., and Pauly, D. (1993). *Trophic models of aquatic ecosystems* (International Center for Living Aquatic Resources Management (ICLARM), the International Council for the Exploration of the Sea, and the Danish International Development Agency: WorldFish).

Christensen, V., and Pauly, D. (1995). Primary production required to sustain global fisheries. *Nature* 374, 255–257. doi: 10.1038/374255a0

Christensen, V., and Walters, C. J. (2004). Ecopath with ecosim: methods, capabilities and limitations. *Ecol. Modell.* 172, 109–139. doi: 10.1016/j.ecolmodel.2003.09.003

Christensen, V., Walters, C., Pauley, D., and Forrest, R. (2008). Ecopath with ecosim version 6: user guide. *Lenfest Ocean Futur. Proj.* 1–235.

Coll, J., García-Rubies, A., and Hereu Fina, B. (2003). Estat actual de les poblacions de peixos vulnerables a la pesca en els fons rocosos de l'àrea d'influència marina del parc natural de sa dragonera (SO de mallorca). *Cent. d'Estudis Avançats Blanes*. 1–46. Available at: http://hdl.handle.net/2445/22608.

Coll, M., and Libralato, S. (2012). Contributions of food web modelling to the ecosystem approach to marine resource management in the Mediterranean Sea. *Fish Fish.* 13, 60–88. doi: 10.1111/j.1467-2979.2011.00420.x

Coll, M., Ortega-Cerdà, M., and Mascarell-Rocher, Y. (2021). Ecological and economic effects of COVID-19 in marine fisheries from the Northwestern Mediterranean Sea. *Biol. Conserv.* 255, 108997. doi: 10.1016/j.biocon.2021.108997

Coll, M., Palomera, I., Tudela, S., and Dowd, M. (2008). Food-web dynamics in the South Catalan Sea ecosystem (NW Mediterranean) for 1978-2003. *Ecol. Modell.* 217, 95–116. doi: 10.1016/j.ecolmodel.2008.06.013

Coll, M., Palomera, I., Tudela, S., and Sardà, F. (2006). Trophic flows, ecosystem structure and fishing impacts in the South Catalan Sea, Northwestern Mediterranean. *J. Mar. Syst.* 59, 63–96. doi: 10.1016/j.jmarsys.2005.09.001

Coll, J., Rubies, A. G., Morey, G., Navarro, O., Martino, S., Munuera, F. R., et al. (2007). "La reserva Marina del nord de menorca com a eina per a la gestió sostenible dels recursos marins: el cas de l'anfós epinephelus marginatus (Lowe 1834)," in *V Jornades del medi ambient de les illes balears: ponències i resums (Societat d'Història natural de les balears)*, 272–273.

Colléter, M., Valls, A., Guitton, J., Gascuel, D., Pauly, D., and Christensen, V. (2015). Global overview of the applications of the ecopath with ecosim modeling approach using the EcoBase models repository. *Ecol. Modell.* 302, 42–53. doi: 10.1016/j.ecolmodel.2015.01.025

Colloca, F., Cardinale, M., Maynou, F., Giannoulaki, M., Scarcella, G., Jenko, K., et al. (2013). Rebuilding Mediterranean fisheries: a new paradigm for ecological sustainability. *Fish Fish.* 14, 89–109. doi: 10.1111/j.1467-2979.2011.00453.x

Corrales, X., Coll, M., Tecchio, S., Bellido, J. M., Fernández, Á.M., and Palomera, I. (2015). Ecosystem structure and fishing impacts in the northwestern Mediterranean Sea using a food web model within a comparative approach. *J. Mar. Syst.* 148, 183–199. doi: 10.1016/j.jmarsys.2015.03.006

Corrales, X., Ofir, E., Coll, M., Goren, M., Edelist, D., Heymans, J. J., et al. (2017). Modeling the role and impact of alien species and fisheries on the Israeli marine continental shelf ecosystem. *J. Mar. Syst.* 170, 88–102. doi: 10.1016/j.jmarsys.2017.02.004

De Goeij, J. M., Van Oevelen, D., Vermeij, M. J. A., Osinga, R., Middelburg, J. J., De Goeij, A. F. P. M., et al. (2013). Surviving in a marine desert: the sponge loop retains resources within coral reefs. *Science* 342, 108–110. doi: 10.1126/science.1241981

Domínguez, M., Fontán, A., Rivera, J., and Ramón, M. (2013). Caracterización del ecosistema bentónico de la plataforma costera del área comprendida entre sa dragonera, cabrera y el cap de ses salines (Mallorca). *Inst. Espanyol d'Oceanografia*.

Estrada, M. (1996). Primary production in the Northwestern Mediterranean. *Sci. Mar.* 60.

Fanelli, E., Cartes, J. E., Rumolo, P., and Sprovieri, M. (2009). Food-web structure and trophodynamics of mesopelagic–suprabenthic bathyal macrofauna of the Algerian basin based on stable isotopes of carbon and nitrogen. *Deep Sea Res. Part I Oceanogr. Res. Pap.* 56, 1504–1520. doi: 10.1016/j.dsr.2009.04.004

Farrugio, H., Oliver, P., and Biagi, F. (1993). An overview of the history, knowledge, recent and future research trends in Mediterranean fisheries *. *Sci. Mar. Trends Res. Mediterr. Fish.* 57, 105–119.

Fernandez de Puelles, M., Grás, D., and Hernández-León, S. (2003). Annual cycle of zooplankton biomass, abundance and species composition in the neritic area of the Balearic Sea, Western Mediterranean. *Mar. Ecol.* 24, 123–139. doi: 10.1046/j.1439-0485.2003.03816.x

Fernandez de Puelles, M. L., Morillas, A., Font, E. A., and Vicente, L. (2007). The spatial zooplankton distribution in the north balear front (Western mediterranean): early spring 2005. *Rapp. Commun. Int. Mer Medit.* 38, 473.

Fernández de Puelles, M. L., Valencia, J., Jansá, J., and Morillas, A. (2004). Hydrographical characteristics and zooplankton distribution in the mallorca channel (Western mediterranean): spring 2001. *ICES J. Mar. Sci.* 61, 654–666. doi: 10.1016/j.icesjms.2004.03.031

Forcada, J., Gazo, M., Aguilar, A., Gonzalvo, J., and Fernández-Contreras, M. (2004). Bottlenose dolphin abundance in the NW Mediterranean: addressing heterogeneity in distribution. *Mar. Ecol. Prog. Ser.* 275, 275–287. doi: 10.3354/meps275275

García-Rodríguez, E., Coll, M., Vivas, M., Bellido, J. M., Esteban, A., and Torres, M.Á. (2021). A food-web comparative modeling approach highlights ecosystem singularities of the gulf of alicante (Western Mediterranean Sea). *J. Sea Res.* 174, 102073. doi: 10.1016/j.seares.2021.102073

Gascuel, D., Guénette, S., and Pauly, D. (2011). The trophic-level-based ecosystem modelling approach: theoretical overview and practical uses. *ICES J. Mar. Sci.* 68, 1403–1416. doi: 10.1093/icesjms/fsr062

GFCM (2022). "General fisheries commission for the Mediterranean," in *Report of the twenty-third session of the scientific advisory committee on fisheries* (Rome, Italy: FAO headquartes), 21–24. doi: 10.4060/cc3109en

Hattab, T., Ben, F., Lasram, R., Albouy, C., Salah, M., Jarboui, O., et al. (2013). An ecosystem model of an exploited southern Mediterranean shelf region (gulf of gabes , Tunisia) and a comparison with other Mediterranean ecosystem model properties. *J. Mar. Syst.* 128, 159–174. doi: 10.1016/j.jmarsys.2013.04.017

Heymans, J. J., Coll, M., Libralato, S., Morissette, L., and Christensen, V. (2014). Global patterns in ecological indicators of marine food webs: a modelling approach. *PloS One* 9, 1–16. doi: 10.1371/journal.pone.0095845

Heymans, J. J., Coll, M., Link, J. S., Mackinson, S., Steenbeek, J., Walters, C., et al. (2016). Best practice in ecopath with ecosim food-web models for ecosystem-based management. *Ecol. Modell.* 331, 173–184. doi: 10.1016/j.ecolmodel.2015.12.007

Horn, S., Coll, M., Asmus, H., and Dolch, T. (2021). Food web models reveal potential ecosystem effects of seagrass recovery in the Northern Wadden Sea. *Restor. Ecol.* 29, 1–13. doi: 10.1111/rec.13328

Innes, S., Lavigne, D. M., Earle, W. M., and Kovacs, K. M. (1987). Feeding rates of seals and whales. *J. Anim. Ecol.* 56 (1), 115–130. doi: 10.2307/4803

Keramidas, I., Dimarchopoulou, D., Eyal, O., Scotti, M., Tsikliras, A. C., and Gal, G. (2023). Ecotrophic perspective in fisheries management : a review of ecopath with ecosim models in European marine ecosystems. *Front. Mar. Sci* 10, 1–22. doi: 10.3389/fmars.2023.1182921

Keramidas, I., Dimarchopoulou, D., and Tsikliras, A. C. (2022). Modelling and assessing the ecosystem of the Aegean Sea, a major hub of the Eastern Mediterranean at the intersection of Europe and Asia. *Reg. Stud. Mar. Sci.* 56, 102704. doi: 10.1016/j.rsma.2022.102704

Lehucher, P. M., Beautier, L., Chartier, M., Martel, F., Mortier, L., Brehmer, P., et al. (1995). Progress from 1989 to 1992 in understanding the circulation of the Western Mediterranean Sea. *Oceanol. Acta* 18, 255–271. Available at: https://archimer.ifremer.fr/doc/00097/20776/.

Lindeman, R. L. (1942). The trophic-dynamic aspect of ecology author (s): Raymond l . lindeman published by : ecological society of America stable URL : http://www.jstor.org/stable/1930126. *Ecology* 23, 399–417. doi: 10.2307/1930126

Link, J. S. (2010). Adding rigor to ecological network models by evaluating a set of pre-balance diagnostics: a plea for PREBAL. *Ecol. Modell.* 221, 1580–1591. doi: 10.1016/j.ecolmodel.2010.03.012

Lleonart, J., and Maynou, F. (2003). Fish stock assessments in the Mediterranean: state of the art. *Sci. Mar.* 67, 37–49. doi: 10.3989/scimar.2003.67s137

Longo, C., Hornborg, S., Bartolino, V., Tomczak, M. T., Ciannelli, L., Libralato, S., et al. (2015). Role of trophic models and indicators in current marine fisheries management. *Mar. Ecol. Prog. Ser.* 538, 257–272. doi: 10.3354/meps11502

Madurell, T., and Cartes, J. E. (2005). Trophodynamics of a deep-sea demersal fish assemblage from the bathyal eastern Ionian Sea (Mediterranean Sea). *Deep Sea Res. Part I Oceanogr. Res. Pap.* 52, 2049–2064. doi: 10.1016/j.dsr.2005.06.013

Madurell, T., Cartes, J. E., and Labropoulou, M. (2004). Changes in the structure of fish assemblages in a bathyal site of the Ionian Sea (eastern Mediterranean). *Fish. Res.* 66, 245–260. doi: 10.1016/S0165-7836(03)00205-4

Madurell, T., Fanelli, E., and Cartes, J. E. (2008). Isotopic composition of carbon and nitrogen of suprabenthic fauna in the NW Balearic Islands (western Mediterranean). *Mar. Syst.* 71, 336–345. doi: 10.1016/j.jmarsys.2007.03.006

Mallol, S., and Goñi, R. (2004). Seguimiento de las pesquerías (Punto Cero) en el ámbito de la futura Reserva Marina de Cala Rajada (Mallorca). Informe Técnico IEO-COB y SGPM. PESCALA I/04-1. Secreta.

Massutí, E., Olivar, M. P., Monserrat, S., Rueda, L., and Oliver, P. (2014). Towards understanding the influence of environmental conditions on demersal resources and ecosystems in the Western Mediterranean: motivations, aims and methods of the IDEADOS project. *J. Mar. Syst.* 138, 3–19. doi: 10.1016/j.jmarsys.2014.01.013

Massutí, E., and Renones, O. (2005). Demersal resource assemblages in the trawl fishing grounds off the Balearic Islands (Western Mediterranean). *Sci. Mar.* 69, 167–181. doi: 10.3989/scimar.2005.69n1167

Massutí, E., Sánchez-Guillamón, O., Farriols, M. T., Palomino, D., Frank, A., Bárcenas, P., et al. (2022). Improving scientific knowledge of mallorca channel seamounts (Western Mediterranean) within the framework of natura 2000 network. *Diversity* 14, 1–71. doi: 10.3390/d14010004

Mellon-Duval, C., De Pontual, H., Métral, L., and Quemener, L. (2010). Growth of European hake (Merluccius merluccius) in the gulf of lions based on conventional tagging. *ICES J. Mar. Sci.* 67, 62–70. doi: 10.1093/icesjms/fsp215

Morales-Nin, B., Moranta, J., Garcia, C., Tugores, P., and Grau, A. M. (2008). “Evaluation of the importance of recreational fisheries on a Mediterranean Island,” in Reconciling fisheries with conservation: Proceedings of the Fourth World Fisheries Congress J. Nielsen , et al. *American Fisheries Society Symposium (AMERICAN FISHERIES SOCIETY).* 49, 1946.

Morales-Nin, B., Moranta, J., García, C., Tugores, M. P., Grau, A. M., Riera, F., et al. (2005). The recreational fishery off Majorca island (western mediterranean): some implications for coastal resource management. *ICES J. Mar. Sci.* 62, 727–739. doi: 10.1016/j.icesjms.2005.01.022

Morey, G., García-Rubies, A., Hereu Fina, B., and Coll, J. (2003). Estat actual de les poblacions de peixos vulnerables a la pesca en els fons rocosos de la futura Reserva Marina de Llevant (NE de Mallorca).

Navarro, J., Coll, M., Louzao, M., Palomera, I., Delgado, A., and Forero, M. G. (2011). Comparison of ecosystem modelling and isotopic approach as ecological tools to investigate food webs in the NW Mediterranean Sea. *J. Exp. Mar. Bio. Ecol.* 401, 97–104. doi: 10.1016/j.jembe.2011.02.040

Nee, S. (1990). Community construction. *Trends Ecol. Evol.* 5, 337–340. doi: 10.1016/0169-5347(90)90182-D

Nilsson, S. G., and Nilsson, I. N. (1976). Numbers, food consumption, and fish predation by birds in lake möckeln, Southern Sweden. *Ornis Scand.* 7, 61–70. doi: 10.2307/3676175

Odum, E. P. (1969). The strategy of ecosystem development: an understanding of ecological succession provides a basis for resolving man’s conflict with nature. *Sci. (80-.).* 164, 262–270. doi: 10.1126/science.164.3877.262

Odum, E. P. (1971). *Fundamentals of ecology* (Saunders Philadelphia: International Council for the Exploration of the Sea (ICES)).

Olivar, M. P., Bernal, A., Molí, B., Peña, M., Balbín, R., Castellón, A., et al. (2012). Vertical distribution, diversity and assemblages of mesopelagic fishes in the western Mediterranean. *Deep Sea Res. Part I Oceanogr. Res. Pap.* 62, 53–69. doi: 10.1016/j.dsr.2011.12.014

Opitz, S. (1996). *Trophic interactions in Caribbean coral reefs* (ICLARM, Manila, Philippines: WorldFish).

Ordines, F. (2015) *Habitats and nekto-benthic communities of the bottom-trawl fishery developed on the continental shelf of the Balearic Islands.* Available at: http://hdl.handle.net/10508/10097.

Ordines, F., Bauzá, M., Sbert, M., Roca, P., Gianotti, M., and Massutí, E. (2015). Red algal beds increase the condition of nekto-benthic fish. *J. Sea Res.* 95, 115–123. doi: 10.1016/j.seares.2014.08.002

Ordines, F., Farriols, M. T., Lleonart, J., Guijarro, B., Quetglas, A., and Massutí, E. (2014). Biology and population dynamics of by-catch fish species of the bottom trawl fishery in the Western Mediterranean. *Mediterr. Mar. Sci.* 15 (3), 613–625. doi: 10.12681/mms.812

Ordines, F., and Massutí, E. (2009). Relationships between macro-epibenthic communities and fish on the shelf grounds of the Western Mediterranean. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 383, 370–383. doi: 10.1002/aqc

Ordines, F., Quetglas, A., Massutí, E., and Moranta, J. (2009). Habitat preferences and life history of the red scorpion fish, scorpaena notata, in the Mediterranean. *Estuar. Coast. Shelf Sci.* 85, 537–546. doi: 10.1016/j.ecss.2009.09.020

Ordines, F., Ramón, M., Rivera, J., Rodríguez-Prieto, C., Farriols, M. T., Guijarro, B., et al. (2017). Why long term trawled red algae beds off Balearic Islands (Western Mediterranean) still persist? *Reg. Stud. Mar. Sci.* 15, 39–49. doi: 10.1016/j.rsma.2017.07.005

Ordines, F., Valls, M., Meléndez, M. J., Ramírez-Amaro, S., López, E., Lloret, J., et al. (2021). Potential factors influencing the condition of demersal sharks in the Mediterranean deep sea ecosystems. *Deep. Res. Part I Oceanogr. Res. Pap.* 176, 103603. doi: 10.1016/j.dsr.2021.103603

Palmer, M., Quetglas, A., Guijarro, B., Moranta, J., Ordines, F., and Massutí, E. (2009). Performance of artificial neural networks and discriminant analysis in predicting fishing tactics from multispecific fisheries. *Can. J. Fish. Aquat. Sci.* 66, 224–237. doi: 10.1139/F08-208

Patrick, W. S., and Link, J. S. (2015). Myths that continue to impede progress in ecosystem-based fisheries management. *Fisheries* 40, 155–160. doi: 10.1080/03632415.2015.1024308

Pauly, D. (1980). On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. *ICES J. Mar. Sci.* 39, 175–192. doi: 10.1093/icesjms/39.2.175

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Jr, F. T. (1998). *Fish. Down Mar. Food Webs* 279, 860–863. doi: 10.1126/science.279.5352.860

Pauly, D., Christensen, V., and Sambilay, J. V. (1990). *Some features of fish food consumption estimates used by ecosystem modelers* (International Council for the Exploration of the Sea (ICES).

Pauly, D., Soriano-Bratz, M. L., and Palomares, M. L. D. (1993). “Improved construction, parameterization and interpretation of steady-state ecosystem models,” in *Trophic models of aquatic ecosystems.* Eds. V. Christensen and D. Pauly (WorldFish), 1–13.

Peña, M., Olivar, M. P., Balbín, R., López-Jurado, J. L., Iglesias, M., and Miquel, J. (2014). Acoustic detection of mesopelagic fishes in scattering layers of the Balearic Sea (Western Mediterranean). *Can. J. Fish. Aquat. Sci.* 71, 1186–1197. doi: 10.1139/cjfas-2013-0331

Pikitch, E. K., Santora, C., Babcock, E. A., Bakun, A., Bonfil, R., Conover, D. O., et al. (2004). Ecosystem-based fishery management. *Sci. (80-.).* 305, 346–347. doi: 10.1126/science.1098222

Pinot, J.-M., Tintoré, J., and Gomis, D. (1995). Multivariate analysis of the surface circulation in the Balearic Sea. *Prog. Oceanogr.* 36, 343–376. doi: 10.1016/0079-6611(96)00003-1

Piroddi, C., Teixeira, H., Lynam, C. P., Smith, C., Alvarez, M. C., Mazik, K., et al. (2015). Using ecological models to assess ecosystem status in support of the European marine strategy framework directive. *Ecol. Indic.* 58, 175–191. doi: 10.1016/j.ecolind.2015.05.037

Polunin, N. V. C., Morales-Nin, B., Pawsey, W. E., Cartes, J. E., Pinnegar, J. K., and Moranta, J. (2001). Feeding relationships in Mediterranean bathyal assemblages elucidated by stable nitrogen and carbon isotope data. *Mar. Ecol. Prog. Ser.* 220, 13–23. doi: 10.3354/meps220013

Pomeroy, L. R., le Williams, P. J. B., Azam, F., and Hobbie, J. E. (2007). The microbial loop. *Oceanography* 20, 28–33. doi: 10.5670/oceanog.2007.45

Quetglas, A., Guijarro, B., Ordines, F., and Massutí, E. (2012). Stock boundaries for fisheries assessment and management in the Mediterranean: the Balearic Islands as a case study. *Sci. Mar.* 76, 17–28. doi: 10.3989/scimar.2012.76n1017

Quetglas, A., Merino, G., González, J., Ordines, F., Garau, A., Grau, A. M., et al. (2016a). *Regional implementation plan for demersal fisheries from the Balearic islands* (Centro Oceanográfico de Baleares: Western Mediterranean).

Quetglas, A., Merino, G., Ordines, F., Guijarro, B., Garau, A., Grau, A. M., et al. (2016b). Assessment and management of western Mediterranean small-scale fisheries. *Ocean Coast. Manage.* 133, 95–104. doi: 10.1016/j.ocecoaman.2016.09.013

Ramírez-Amaro, S., Ordines, F., Esteban, A., García, C., Guijarro, B., Salmerón, F., et al. (2020). The diversity of recent trends for chondrichthyans in the Mediterranean reflects fishing exploitation and a potential evolutionary pressure towards early maturation. *Sci. Rep.* 10, 1–18. doi: 10.1038/s41598-019-56818-9

Reiss, H., Kröncke, I., and Ehrich, S. (2006). Estimating the catching efficiency of a 2-m beam trawl for sampling epifauna by removal experiments. *ICES J. Mar. Sci.* 63, 1453–1464. doi: 10.1016/j.icesjms.2006.06.001

Rueda, L., Valls, M., Hidalgo, M., Guijarro, B., Esteban, A., and Massutí, E. (2019). From trophic ecology to fish condition: contrasting pathways for European hake in the western Mediterranean. *Mar. Ecol. Prog. Ser.* 623, 131–143. doi: 10.3354/meps13025

Scotti, M., Opitz, S., MacNeil, L., Kreutle, A., Pusch, C., and Froese, R. (2022). Ecosystem-based fisheries management increases catch and carbon sequestration through recovery of exploited stocks: the western Baltic Sea case study. *Front. Mar. Sci.* 9. doi: 10.3389/fmars.2022.879998

Sea Around Us. Available at: https://www.seaaroundus.org/.

Spedicato, M. T., Massutí, E., Mérigot, B., Tserpes, G., Relini, G., and Tecnologia, C. (2019). The MEDITS trawl survey specifications in an ecosystem approach to fishery management. *Sci. Mar.* 83, 9–20.

STECF (2022). “Scientific, technical and economic committee for fisheries (STECF)-stock assessments: demersal stocks in the western Mediterranean Sea (STECF-20-09),” in *Publ. off. eur. union, luxemb.* Available at: https://stecf.jrc.ec.europa.eu/documents/43805/1446742/STECF+16-14+-+Methods+for+MED+stock+assessments_JRC102680.pdf.

Tecchio, S., Coll, M., Christensen, V., Company, J. B., Ramírez-Llodra, E., and Sardà, F. (2013). Food web structure and vulnerability of a deep-sea ecosystem in the NW Mediterranean Sea. *Deep. Res. Part I Oceanogr. Res. Pap.* 75, 1–15. doi: 10.1016/j.dsr.2013.01.003

Torres, M.Á., Coll, M., Heymans, J. J., Christensen, V., and Sobrino, I. (2013). Food-web structure of and fishing impacts on the gulf of cadiz ecosystem (South-Western Spain). *Ecol. Modell.* 265, 26–44. doi: 10.1016/J.ECOLMODEL.2013.05.019

Trites, A. W., Christensen, V., and Pauly, D. (1997). Competition between fisheries and marine mammals for prey and primary production in the Pacific Ocean. *J. Northwest Atl. Fish. Sci.* 22, 173–187. doi: 10.2960/J.v22.a14

Tsagarakis, K., Coll, M., Giannoulaki, M., Somarakis, S., Papaconstantinou, C., and Machias, A. (2010). Food-web traits of the north Aegean Sea ecosystem (Eastern

Mediterranean) and comparison with other Mediterranean ecosystems. *Estuar. Coast. Shelf Sci.* 88, 233–248. doi: 10.1016/j.ecss.2010.04.007

Ulanowicz, R. E. (1986). Growth and development: ecosystem phenomenology. *Springer Verlag* 203. doi: 10.1007/978-1-4612-4916-0

Ulanowicz, R. E. (1995). The part-whole relation in ecosystems. *Complex Ecol. Prentice-Hall New Jersey*, 549–560.

Ulanowicz, R. E., and Puccia, C. J. (1990). Mixed trophic impacts in ecosystems. *Coenoses* 5, 7–16. Available at: http://www.jstor.org/stable/43461017

Valls, A., Coll, M., Christensen, V., and Ellison, A. M. (2015). Keystone species: toward an operational concept for marine biodiversity conservation. *Ecol. Monogr.* 85, 29–47. doi: 10.1890/14-0306.1

Valls, M., Quetglas, A., Ordines, F., and Moranta, J. (2011). Feeding ecology of demersal elasmobranchs from the shelf and slope off the Balearic Sea (western Mediterranean). *Sci. Mar.* 75, 633–639. doi: 10.3989/scimar.2011.75n4633

Valls, M., Rueda, L., and Quetglas, A. (2017). Feeding strategies and resource partitioning among elasmobranchs and cephalopods in Mediterranean deep-sea ecosystems. *Deep. Res. Part I Oceanogr. Res. Pap.* 128, 28–41. doi: 10.1016/j.dsr.2017.09.002

Valls, M., Sweeting, C. J., Olivar, M. P., Fernández de Puelles, M. L., Pasqual, C., Polunin, N. V. C., et al. (2014). Structure and dynamics of food webs in the water column on shelf and slope grounds of the Western Mediterranean. *J. Mar. Syst.* 138, 171–181. doi: 10.1016/j.jmarsys.2014.04.002

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EDITED BY
Georgios Sylaios,
Democritus University of Thrace, Greece

REVIEWED BY
Donna Dimarchopoulou,
Dalhousie University, Canada
Konstantinos Tsagarakis,
Hellenic Centre for Marine Research (HCMR), Greece

*CORRESPONDENCE
Eyal Ofir
eyal.ofir@ocean.org.il

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Evaluation of fisheries management policies in the alien species-rich Eastern Mediterranean under climate change

Eyal Ofir[1]*, Xavier Corrales[2], Marta Coll[3,4], Johanna Jacomina Heymans[5,6], Menachem Goren[7], Jeroen Steenbeek[4], Yael Amitai[1], Noam Shachar[1] and Gideon Gal[1]

[1]Kinneret Limnological Laboratory, Israel Oceanographic and Limnological Research, Migdal, Israel, [2]AZTI, Marine Research Division, Basque Research and Technology Alliance (BRTA), Sukarrieta, Spain, [3]Institute of Marine Science (ICM-CSIC), Passeig Marítim de la Barceloneta, Barcelona, Spain, [4]Ecopath International Initiative Research Association, Barcelona, Spain, [5]European Marine Board, Oostende, Belgium, [6]Scottish Association for Marine Science, Argyll, United Kingdom, [7]The Steinhardt Museum of Natural History and School of Zoology, Tel Aviv University, Tel Aviv, Israel

Recent decades have witnessed declines in the amount of fishing catch due to changes in the marine ecosystem of the Eastern Mediterranean Sea. These changes are mainly a consequence of direct human activities as well as global warming and the entry of invasive species. Therefore, there is a need to improve fisheries management so that it accounts for the various stressors and uses of the marine environment beyond fishing, while providing sustainable catches and maintaining a healthy ecosystem. The ability to understand, and sustainably manage, the fishing industry relies on models capable of analyzing and predicting the effects of fishing on the entire ecosystem. In this study, we apply Ecospace, the spatial-temporal component of the Ecopath with Ecosim approach, to study the Israeli continental shelf to evaluate the impact of climate change and alternative management options on the ecosystem. We examine several management alternatives under the severe assumption of the RCP8.5 climate change scenario for the region. Results indicate that under business-as-usual conditions, the biomass of the native species will decrease, the biomass of the invasive species will increase, and there will be a decrease in the fishing catch. In addition, of the management alternatives examined, the alternative of prohibition of fishing in the northern region of Israel along with the establishment of a network of marine nature reserves provides the optimal response for the ecosystem and fisheries. The Achziv Nature Reserve is projected to be successful, improving the biomass of local species and reducing, to some extent, the presence of invasive species. These results are consistent with visual surveys conducted inside and outside the reserve by the Israeli Nature and Parks Authority. Furthermore, simulation results indicate spill-over effects in areas close to nature reserves yielding higher catches in those regions.

KEYWORDS

fishery, management, climate change, fish, invasive species, ecospace

Introduction

Marine ecosystems around the world are changing faster than ever, increasing the challenges resource managers and decision makers face. These ecosystems are highly affected by multiple stressors including climate change, which in turn is affecting and changing the distribution and composition of the species in the ecosystem (Moullec et al., 2016; Tittensor et al., 2021). Moreover, these changes are increasing the complexity of managing the ecosystems and ensuring sustainable management and the maintenance of ecosystem services.

A prime example of an ecosystem subject to a range of natural and anthropogenic stresses is the Eastern Mediterranean Sea, specifically, that of the Israeli continental shelf (ICS). This ecosystem is strongly affected by a large range of stressors, some driven by human activities and others natural. Due to these stressors, the ecosystem has undergone frequent changes (Galil and Zenetos, 2002; van Leeuwen et al., 2022). Many of the changes that have occurred in the ecosystem are a result of two major stressors acting upon it: climate change and the influx of invasive species.

The Eastern Mediterranean Sea region has been recognized for many years as a climate change hotspot—a region in which the climate change is expected to have larger effects than other regions (Lejeusne et al., 2010; Moullec et al., 2016; Givan et al., 2018). According to the latest estimates of the Intergovernmental Panel on Climate Change (IPCC), the expected climate change for the Mediterranean Sea region for the period 2018–2100 with respect to the period 1986–2005, expressed by the most pessimistic RCP (Representative Concentration Pathway) 8.5 scenario, is an increase in atmospheric temperature of up to 4°C and a decrease in rainfall amounts by ca. 10–20% (Hulme, 2016; Bloomfield and Manktelow, 2021; Kikstra et al., 2022). Regional data indicate a multiannual increase in temperature and reinforce projections of rising sea level and rising salinity (Herut, 2021). These changes are significant with far-reaching environmental consequences. For example, the increase in temperature can impact marine ecosystems via different mechanisms such as availability and distribution of species (Wright et al., 2020), and has probably already affected organism resilience, and possibly even the survival of species living at their limit of heat stress (Edelist et al., 2013). Another notable ecological impact of the rise in temperature is the improved conditions for the establishment of invasive species in habitats that until now were too cold for them, as well as the arrival of additional invasive species that benefit from more agreeable conditions thanks to the rise in temperature (Chaikin et al., 2022).

As of 2016, over 800 alien species have been documented in the eastern basin of the Mediterranean Sea (Galil et al., 2017) of which more than half have established and created local populations (Galil et al., 2018). The consequence of the establishment of invasive species populations is displacement of local species (Golani, 2010; Edelist et al., 2013; Corrales et al., 2018). Most of the invasive species in the study area are Eritrean species originating from the Red Sea or the Indian Ocean, reaching the Eastern Mediterranean via Lessepsian migration. The influx of invasive species has led to widespread changes in the faunal composition of the rocky and sandy Israeli coast (Goren et al., 2016; Arndt et al., 2018). For example, the marbled spinefoot (*Siganus rivulatus*) and the dusky spinefoot (*S. luridus*) have changed the ecology of the rocky habitat (Shapiro Goldberg et al., 2021; Escalas et al., 2022). These fish increased the rate of organic nutrient recycling due to their particularly high digestion rates. Furthermore, they comprise approximately one-third of the biomass of this habitat and are an important component of the diet of many predatory fishes, such as groupers (up to 70%) and dominate the angler catch (Arndt et al., 2018; Shapiro Goldberg et al., 2021; Escalas et al., 2022). Similarly, invasive species have had a clear impact on fishing catch composition (Corrales et al., 2017a; Corrales et al., 2017b).

Other anthropogenic processes such as fishing profoundly affect the eastern Mediterranean Sea ecosystem (Piroddi et al., 2022). Israeli fishing in the Mediterranean Sea is mostly coastal and non-selective (Corrales et al., 2017b). According to data from 2010, the number of fishing licenses included a small commercial fleet of ca. 30 trawler boats, a fleet of 519 coastal fishing boats for cast net fishing and longline fishing, and a small fleet of ca. 28 boats for purse seine net fishing (Michael-Bitton et al., 2022). Likewise, there is a growing recreational fishing sector, namely amateur fishing with rods on the beach, or with rods or a floating longline from a boat or kayak (Stambler, 2014; Corrales et al., 2017b). The Israeli coastal fish catch from the Mediterranean Sea has, however, suffered a declining trend in recent years. In 2000, the catch was ca. 3,500 tons and dropped to 2,250 tons in 2009, a decrease of 35% within a decade. In contrast, there was an increase in the number of active boats from 425 in 2000 to 500 in 2009. According to data available from 2000–2009, fishing pressure increased by 55% (Edelist et al., 2013; Stambler, 2014). In addition, Edelist et al. (2013); Edelist et al. (2013) showed that the significant decline in the annual fish catch per boat during this decade was related mainly to purse seine fishing and coastal fishing.

Decision makers have a range of options for implementing policies for sustainable management of commercial fish populations and ecosystems in the area. These options include, among others, fisheries restricted areas (FRAs) and seasonal closures (Petza et al., 2017; Corrales, 2019). The assumption behind restricting fishing areas is to create geographical areas in which fishing cannot take place, in an effort to allow undisturbed marine animal reproduction and survival (Seker, 2015). Placement, the sizes of the protected areas and their connectivity have a significant impact on their success (Edgar et al., 2014; Abdou et al., 2016; Giakoumi et al., 2017), as does their level of enforcement (Horta e Costa et al., 2016). The ecologically most effective form of enforcement is a ban on all fishing activity (no take zones), but social and economic conflicting interests typically require lengthy planning and implementation trajectories of no take zones. Nevertheless, their use has increased rapidly around the world, with the 2030 Agenda for Sustainable Development pushing for 30% of marine areas fully protected by 2030 (Simmons et al., 2021). Some countries have even integrated marine protected areas (MPAs) into the marine spatial planning processes for a maritime zone, to regulate the different uses of the area while also protecting the ecosystem (Frazão Santos et al., 2019).

Implementing a management regime that limits fishing during the reproductive season aims to ensure minimal impact on the fish populations during this critical period (Lavin et al., 2021). The advantage of this method is that for a certain period of the year there is a break in at least part of the fishing pressure on the population; however, this method has several disadvantages. First, it is not always possible to know when the reproductive season will occur due to the unstable nature of water temperature fluctuations. Second, if we allow reproduction for one year, and immediately afterward we catch a critical mass of adult fish, we will have a significant impact on population stability (Sterner, 2012). Moreover, this method relies on the fact that there is not supposed to be any fishing below the reproductive size, in other words, the net mesh size is supposed to be large enough so that the net only catches adult fish; if this is not the case, then those fish that are supposed to reach the following reproductive season will not be successful (Cacaud, 2005; Bagsit et al., 2021).

Fishing management and fishing standards in Israel have undergone several significant changes in recent years. In 2016 the Fishing Division of the Ministry of Agriculture and Rural Development implemented new fishing regulations. Until 2016, there were limited management options in place to control the impact of fisheries. There were only a limited number of small regions closed to some form of fishing along the northern part of the Israeli coastline, with only one closed to all activities (Atlit) and only one nature reserve (Achziv), in which a fishing ban was enforced (Edelist et al., 2013). Fishing surveys at Atlit and diving surveys in, and around, the Achziv Reserve have reported dwindling populations of small commercial species outside the reserves, compared to higher densities of large commercial fish inside the areas (NPA, 2015). The new regulations, implemented in 2016, included restricting fishing to a minimum size for fish of various species, imposing a restriction on fishing areas for trawling, restricting fishing during the reproductive season, restricting the size of the fish catch for recreational fishing, as well as regulating the necessary licenses for recreational fishing (NPA, 2020). In addition, the new fishing regulations closed the northern part of Israel to fishing. Since the implementation of the new regulations, there has been no study of their effectiveness and the consequences for fishing, fish populations, or the ecosystem. Together with the need to examine the effectiveness of the implemented management steps, additional management steps that may contribute to the stability of the ecosystem and fish populations, mainly in sensitive areas, should be tested. These include, for example, creating protected areas and marine nature reserves. The Israel Nature and Parks Authority (NPA) prepared a proposal for establishing six protected areas in which fishing will be banned. To date, of these six reserves, only one is active (Rosh Hanikra), one has been officially declared (Avtach), and the other four are in the planning stages (NPA, 2020).

One of the main ways to examine the possible consequences of management steps on the ecosystem is to use an ecological model and examine scenarios of management options under different environmental conditions (Schuwirth et al., 2019). One of the most widely used approaches to fisheries management modeling is the Ecopath with Ecosim (EwE) approach (Christensen and Walters, 2004; Heymans et al., 2016). The EwE approach consists of three progressive models. The foundation is the Ecopath mass balance model that represents a snapshot of the food web in time and space, it is based mainly on the feeding relationships of all the system components, and includes fishing pressure. Ecopath models provide information about the structure and functioning of the ecosystem, describing the energy flows, and the role of species represented by functional groups (Heymans et al., 2012; Christensen et al., 2014b). Building upon Ecopath, the Ecosim time dynamic model provides a means for introducing temporal changes into the mass balance approach to examine the effect of environmental changes and fisheries on the ecosystem (Walters et al., 1997). Ecosim allows for the exploration of different management scenarios over time (Christensen and Walters, 2005; Coll et al., 2009; Stock et al., 2022) while also provisioning the incorporation of economic and social optimization (Coll et al., 2015). Last, Ecospace, a spatial-temporal explicit model, builds upon Ecosim to dynamically distribute ecosystem components across a 2D grid of a grid cells of the same size (Walters et al., 1999). In every cell, Ecospace executes an Ecosim model whilst also incorporating cell-to-cell biomass transfer, including passive and active movement of all the relevant components in the food web based on a time-dynamic niche model and environmental changes (Christensen et al., 2014a). Within the framework of the new developments in Ecospace, it is possible to incorporate time-varying environmental conditions such as temperature and primary production (Steenbeek et al., 2013; Ofir et al., 2022). The use of this component is not limited to hindcasting, and has been widely used to predict plausible futures using data layers taken from Earth System models (Coll et al., 2016b; Serpetti et al., 2017; Corrales et al., 2018; De Mutsert et al., 2021; Steenbeek, 2021).

In this study, we used an Ecospace model to examine different options for managing fisheries along ICS, while analyzing a number of management alternatives that are in the process of being implemented in Israel. We used the outputs of climate models to capture the potential effects of climate change on the food web and on the management of marine resources and fisheries, and a previously built Ecopath with Ecosim model (Corrales et al., 2017a; Corrales et al., 2017b).

Methods

Study region

The easternmost part of the Mediterranean Sea, known as the Levantine Sea, is almost surrounded on all sides by hot, dry countries that feed it with fresh water at a slower rate than the rate of evaporation from the sea. Therefore, it is saltier than the Western part of the Mediterranean Sea, poorer in nutrients, and has a lower inflow of river-water, and in turn, lower primary production (Reich et al., 2022). As a result, the species richness of the animal and plant communities is much lower than in the Western Mediterranean Sea (Goren et al., 2016). However, since its opening in 1869, the Suez Canal has provided a migration route for invasive species from the Red Sea to the Mediterranean Sea, and consequently, the eastern Mediterranean Sea has become one of the most severely impacted marine ecosystems by biological invasions (Costello et al., 2010; Katsanevakis et al., 2014). A number of the invasive fish species have, however, become an important component of the fishing industry (Corrales et al., 2017b).

Our study region (Figure 1) includes the ICS, to a depth of 200 m and covers an area of 3,725 km^2. It has a moderate slope, and it is mostly sandy, except for the rocky areas located in the north. Although this is a relatively small region, it supports most of Israel's maritime uses, including development of maritime facilities e.g., desalination plants and power plants, two commercial ports and diverse marine activity. In addition, the region serves as the fishing region for the different fishing fleets, which utilize it from the coastline (mostly recreational fishing) to the edge of the continental shelf.

Ecospace model

The Ecospace model was based on the Ecopath, and Ecosim models described in Corrales et al. (2017a); Corrales et al. (2017b). The Ecopath model describes the ecosystem via 39 functional groups (FGs) that included 19 native and 8 invasive species in the year 1994 (Corrales et al., 2017a; Corrales et al., 2017b; Corrales et al., 2019). Based on this Ecopath model, the time dynamic module Ecosim was constructed and fitted to time series of data from 1994 to 2010, considering the combined effect of alien species, fishing activities and changes in sea surface temperature and primary productivity. In addition, the model successfully simulated the arrival and establishment of invasive species that arrived in the ecosystem after 1994 (Corrales et al., 2017a).

The Ecospace model represents the Levantine Sea area via a 1 km^2 grid, with a basemap of 80 columns and 170 rows. This cell size provided a good trade-off between spatial resolution to capture ecosystem dynamics and computational costs. Spatial inputs included bathymetry, relevant classes of substrate, and environmental factors namely water temperature and primary productivity (Table 1). To implicitly capture ecosystem dynamics at depth, temperatures were expressed for two depth ranges: from 0 to 30 m depth for pelagic species, and temperatures near the bottom for benthic species (example in Figure 2). Primary production (PP) was vertically integrated for the upper 30 m. Time series of maps of temperature and PP were downloaded from the Copernicus Mediterranean Sea Physics and Mediterranean Sea Biochemistry reanalyzes at monthly averages (Apicella et al., 2022). These maps were input into the model, at monthly time steps, via the Ecospace spatial-temporal data framework (Steenbeek et al., 2013).

In Ecospace, environmental conditions are applied to the living components through functions that express the preferences for, or tolerances to, varying environmental conditions (Christensen et al., 2014a). For the most sensitive species we created these response functions using information from open sources such as AquaMaps (Kaschner et al., 2016), which were complemented with local expert

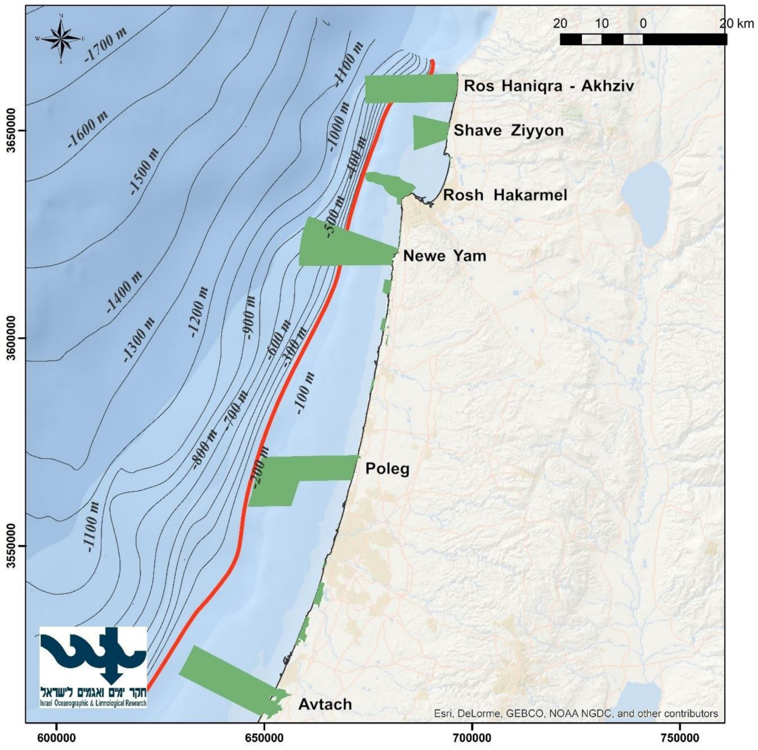

FIGURE 1
A map of the study region. The red line represents the continental shelf down to a depth of 200 m. The green areas represent the MPAs that were used for this paper.

TABLE 1 The environmental variables used in the model of the Eastern Mediterranean Sea.

Variable	Data source	Comments
Depth	IOLR (Israel Oceanographic and Limnological Research) database	
Substrate type (sand/rock)	IOLR (Israel Oceanographic and Limnological Research) database	
Location of fishing ports	Public data	
Temperature layers; monthly intervals	Copernicus: https://resources.marine.Copernicus.eu/products	The information was processed to include temperature data for the upper 30 m and seabed
Primary productivity layers; monthly intervals	Copernicus: https://resources.marine.Copernicus.eu/products	Information was processed to include monthly maps of the upper 30 m.

knowledge. For composite functional groups, response functions were aggregated according to the relative proportion of each species' biomass in the functional group (Appendix 1).

Validation of the model results

To test the results of the model, and to assess its goodness of fit to observed data, we compared our model results to scientific fish surveys. Ecospace was run with temperature and primary productivity (PP) layers of monthly time steps from 1994 (for temperature) and 1999 (for PP) to December 2014 (Apicella et al., 2022). The surveys were conducted in 2012, using trawl nets in two locations, Haifa and Nitzanim, in spring/summer (May, June) and in autumn (October, November) (Goren et al. unpublished data). To enable comparison between the model results and the survey we used the built-in EwE utility that allows extraction of results along transects, assuming that the survey is a straight and narrow line, the model provides results for all groups in the cells included in the transect. We defined three transects at each sample site according to the beginning and end coordinates of the scientific surveys (Figure 3). To calculate the total volume of survey catch, in units of tons per unit area (km^2), we converted the catch using trawling distance and the area of the net opening and created a factor which considers the size of the net, distance and towing time that allowed as to convert the model results to survey data. The comparison between the survey results and the model results was conducted for the relevant (benthic and demersal) functional groups according to the following steps:

1. Classification of the species sampled in the survey to the functional groups of the model
2. Conversion of the number of individuals sampled to wet weight biomass (tons)
3. Division of the results to monthly intervals according to the survey months.

We compared the survey results to the model results and calculated spatial correlations which takes into account the net size, net drag distance and depth between the different groups and

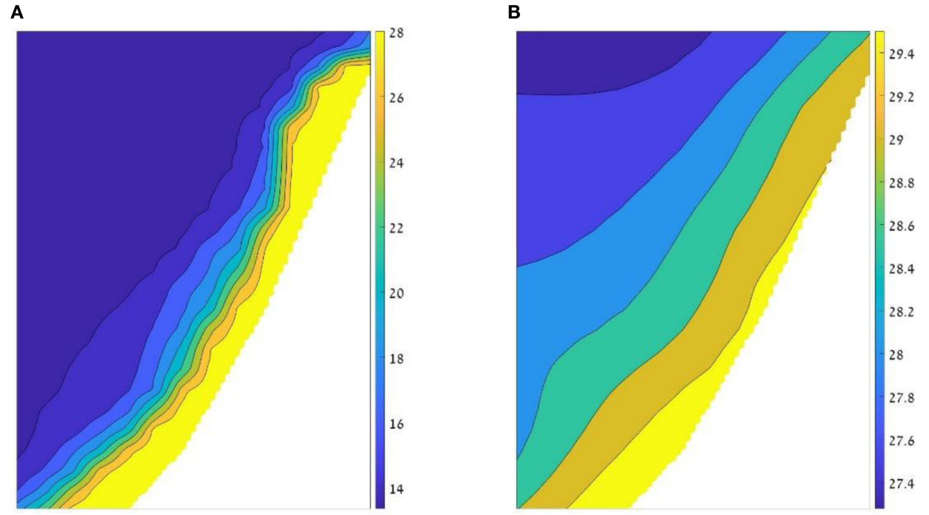

FIGURE 2
Example of the temperatures input layers for the model. The two layers represent August 2014: **(A)** temperature layer of the deep layer, adjacent to the bottom with temperatures ranging between 13.3°C (blue) to 29.9 °C (yellow); and **(B)** temperature layer of the upper 30 m (pelagic zone) in which the values range from 27.2°C to 29.9°C.

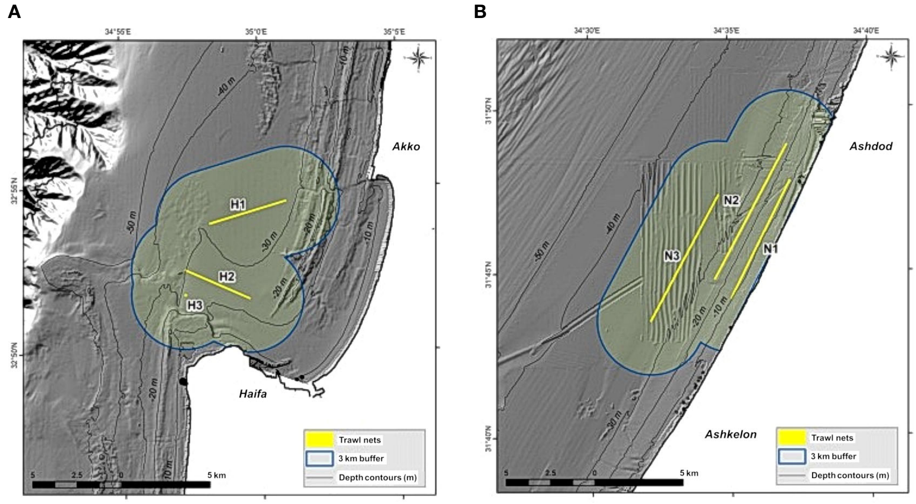

FIGURE 3
Maps of trawl lines and model calculation areas: **(A)** sampling area in Haifa Bay, and, **(B)** sampling area in Nitzanim. The lines represent the sampling transects in the scientific survey; the areas outlined in blue represent the area they cover in the model, representing an area of 1 km^2 on either side of the transects (Kanari et al., 2020).

between the different months of the survey. The comparison was conducted by determining the relationship between the model results and the survey results, and examining seasonal trends in the surveys and in the model by comparing the catch during the different months in relation to the values in June. Thus, we compared, for example, the ratio between the results in October and in June. In addition, we conducted a visual comparison of the results to aid our examination of the differences between the groups.

Scenarios

Finally, several scenarios were developed and tested to examine the plausible consequences of different management interventions on the ecosystem, the fishing community and the catch, under projected climate change conditions.

Climate change scenarios

To examine the future consequences of climate change and the various management alternatives we used the output from the Med-CORDEX modelling framework (Ruti et al., 2016) under the RCP8.5 scenario. Med-CORDEX is a unique framework where research communities make use of atmospheric, land surface, river and oceanic models in regional climate system models to increase the reliability of past and future regional climate information and understand the processes that are responsible for Mediterranean climate variability and trends. We obtained predictions of sea water temperature from three different sources. We adapted predicted water temperatures for the model region to create monthly temperature layers for the period 2006–2100 for the upper 30 m and for the layer adjacent to the bottom. The use of three different sources allowed us to introduce uncertainty into the temperature inputs (Figure 4). Outputs from regional Med-

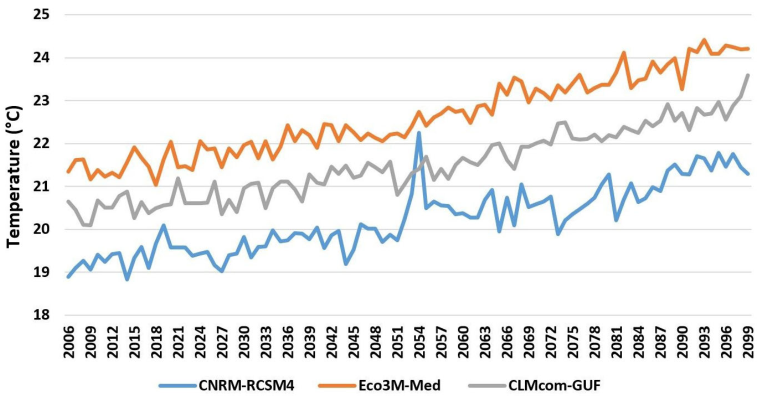

FIGURE 4
Predictions of the three climate models for the RCP8.5 scenario for increasing sea water temperature in the upper 30 m in Israel's continental shelf region for the period 2006-2100.

CORDEX (MC) models were used to drive the Ecospace model under the RCP8.5 emissions scenario (Soto-Navarro et al., 2020):

1. MC1: The first configuration was the coupled CNRM-RCSM4 simulation (Sevault et al., 2014; Darmaraki et al., 2019) that was developed and operated at the Centre National de Recherches Météorologiques (in France). It has an atmospheric resolution of 50 km and an ocean resolution of 9–12 km and 43 ocean vertical levels.
2. MC2: The second configuration was also coupled with a biogeochemical component, the Eco3M-Med (Baklouti et al., 2021). The RCP8.5 projection simulation outcome (Pagès et al., 2020) of net primary production was used in our study.
3. MC3: The third Med-CORDEX ensemble member provided water column temperature evolution under the RCP8.5 climate scenario for this study was the CLMcom-GUF (Model 3) (Bucchignani et al., 2018; Primo et al., 2019) that was developed at the Institute for Atmospheric and Environmental Sciences, Goethe University Frankfurt (Germany), in collaboration with the CLM-Community. It has an atmospheric resolution of 25 km and an oceanic resolution of 12 km. It follows terrain sigma vertical coordinates with 75 layers in the ocean component.

Alternative management options

To examine alternative management options of the ICS region we chose a number of management scenarios, some of which are being assessed by policy makers. The management scenarios were divided into those that restrict fishing for certain periods or regions for different types of fishing fleets, those that examine closure of defined areas to fishing (no-take zones) and their classification as protected areas, and a range of scenarios that combine the different options (Table 2). The following management scenarios were constructed and tested over the period 2006-2100:

1. FM (Fishery management) 1: Closure of the northern part of the maritime region to fishing. The northern region includes expansive rocky areas and unique habitats; therefore, we examined the significance of completely closing it to fishing. The area chosen for closure is greater than that included in the 2016 fishing regulations; the closure stretches from Rosh Hanikra south to Dor Beach.
2. FM2: Closure of a network of nature reserves to fishing (Figure 1). We based our decision on the locations of the nature reserves in the current proposal by Israel's Nature and Park Authority (NPA) to establish a network of six nature reserves of different sizes from north to south (Seker, 2015). We assumed that these nature reserves would be no-take zones.
3. FM3: Seasonal fishing ban during May and June – this scenario is based on the 2016 fishing regulations, in which the Chief Fisheries Officer can order fishing to be stopped during the breeding season. We assumed a complete ban on fishing during May and June.

We further examined a baseline scenario (scenario #1) with no management intervention in the ecosystem. The aim of this baseline scenario was to examine the way the ecosystem fluctuates under the projected future climate change conditions according to the RCP8.5 scenario, with clear increases in water temperature. This scenario did not include changes to fisheries management and served as a basis for comparison with the other scenarios that included fisheries management intervention. All of the scenarios were examined using the model ensemble of the three climate models (MC). All the results of the scenarios were compared to 2020 as a base year since it represents the current period.

The results of the management scenarios presented are an average of the three climate models results. The use of a model ensemble to examine ecological scenarios is becoming more common within the ecosystem modelling community (Jones and Cheung, 2015; Tittensor et al., 2018; Bryndum-Buchholz et al., 2019; Lotze et al., 2019). The use of a model ensemble enables examination of scenarios while accounting for uncertainty associated with the various models and scenarios and increasing confidence in expected outcomes.

TABLE 2 Description of the scenarios examined with the ICS Ecospace model.

SCENARIO NUMBER	TEMPERATURE	NO FISHING IN THE NORTH	MPAS: CLOSING 6 AREAS	SEASONAL CLOSING: MAY- JUNE
1	8.5	No	No	No
2	8.5	Yes	No	No
3	8.5	No	Yes	No
4	8.5	No	No	Yes
COMBINATIONS:				
5	8.5	Yes	Yes	Yes
6	8.5	No	Yes	Yes
7	8.5	Yes	No	Yes
8	8.5	Yes	Yes	No

Analysis of the scenario results

We used a number of indices to examine the scenario results and the effect of climate change and management options on the ecosystem. Some of the indices, such as biomass and catch, and a range of other indices were calculated by the model (Coll and Steenbeek, 2017; Vilas et al., 2020). The indices included general ecological indices that characterize ecosystems, but also specific indices to characterize fishing.

The indices calculated based on the scenarios results included:

1. Marine Trophic Index (MTI) – represents the average trophic level of all components in the catch with TL equal or higher to 3.25 (Kleisner and Pauly, 2011; Coll and Steenbeek, 2017).
2. TL catch – an index that quantifies the trophic level of the catch; it serves as an indicator of the extent of exploitation of the fish resource in the sea. Use of this index is based on the assumption that we fish down the food web (Pauly et al., 1998), meaning, the higher the fishing pressure, the lower the trophic level of the catch (Shannon et al., 2014).
3. Kempton's Q – The Q index is proportional to the inverse slope of the species-abundance curve arranged by trophic levels and is a proxy of ecosystem biodiversity (Coll et al., 2016a).
4. Shannon diversity – an index of species diversity in the ecosystem that enables comparisons over time (Coll et al., 2016a).

Scenario results included annual results for biomass, catch, and the various indicators. The results were compared to the scenario without management intervention to identify the effect (positive or negative) of each management option. In addition, to simplify the analysis of all the information produced by running the scenarios, we grouped the model's FGs into a number of broad groups (fish, local fish, invasive fish, invertebrates, zooplankton, total catch, local species catch, invasive species catch). The division of the broad groups into invasive and local species allowed us to analyze ecosystem fluctuations more clearly and follow the distribution of invasive species and their function in the ecosystem. The layers of information produced by the model were analyzed and visualized at each time step using Matlab, version R2021b. In addition, we created maps of the results in order to examine the spatial distribution of the FGs and broad groups. The results were analyzed using a number of approaches:

1. Temporal dynamic comparison (using annual time steps) of the biomass of the broad groups and calculation of the relationship between the scenario results and the base scenario results.
2. Comparison among scenarios of the average value of each of the indicators throughout the model space. We conducted a relative comparison of the indicators of all scenarios, under the three different climate change drivers.
3. Examination of the contribution of defined protected areas (nature reserves) on the conservation of the local species within the nature reserve.

In order to evaluate the possible benefits of the nature reserves, we calculated the biomass of the different groups in the nature reserve and compared it to the regions outside the protected area and to the scenario without the nature reserves. The examination was done by dividing the cells such that we could isolate the cells comprising the nature reserves and the region next to the nature reserves in the scenario results. We calculated the area of this defined region, and according to its size we calculated the average biomass per km^2 in order to compare different regions. The next step was to compare what exists within, and adjacent to, the Rosh Hanikra Nature Reserve. The choice to present this region stems from the fact that Rosh Hanikra Nature Reserve is an active, monitored, existing nature reserve; thus, we can compare the model results to the insights from the surveys conducted by the NPA in the nature reserve.

Results

Model validation

We compared the survey results with the results produced by the model for a transect analogous to the survey region. Since there is variation between sites and among seasons, we conducted the comparisons for each site separately. Overall, there was a good fit of the biomass estimates for the different groups between the surveys and model results, with differences ranging between -15% to +30% on average over all months and sites. Nevertheless, the results were not uniform and there were differences between sites and among groups.

We obtained a 95% correlation between the survey and model biomass results for the different groups in Haifa, averaged over all months (Figure 5). In most groups, the correlation was very high; however, there were exceptions, such as "earlier invasive dem. fish" for which the model predicted notably higher biomass than was obtained in the survey during May (0.2 *vs.* 0.01 t/km^2, respectively) and June (0.2 *vs.* 0.05 t/km^2, respectively).

For the Nitzanim site, the differences were of +20% on average between the survey and the model results for all groups and all months (Figures 5D-F). These results also showed that the correlation was high for nearly all groups. However, here too, there were exceptions, for example "new invasive dem. fish" which demonstrated greater differences, in October, with 0.5 and 0.08 t/km^2 for the survey and model, respectively, and in November, when the survey results indicated 2-fold higher biomass than model predictions. We also found a difference for "rays and skates" at Nitzanim, in May, with 0.13 and 0.01 t/km^2 for the survey and model, respectively.

Overall, there was a high correlation between the survey and the model results when examining the seasonal fluctuations. For example, for Haifa, the comparison between November and June shows that eight of the nine functional groups exhibited almost perfect correlation between the model and the survey with correlation values (R) of 94%. Similar results were obtained for seven of the nine groups at Nitzanim for the comparison between November and June

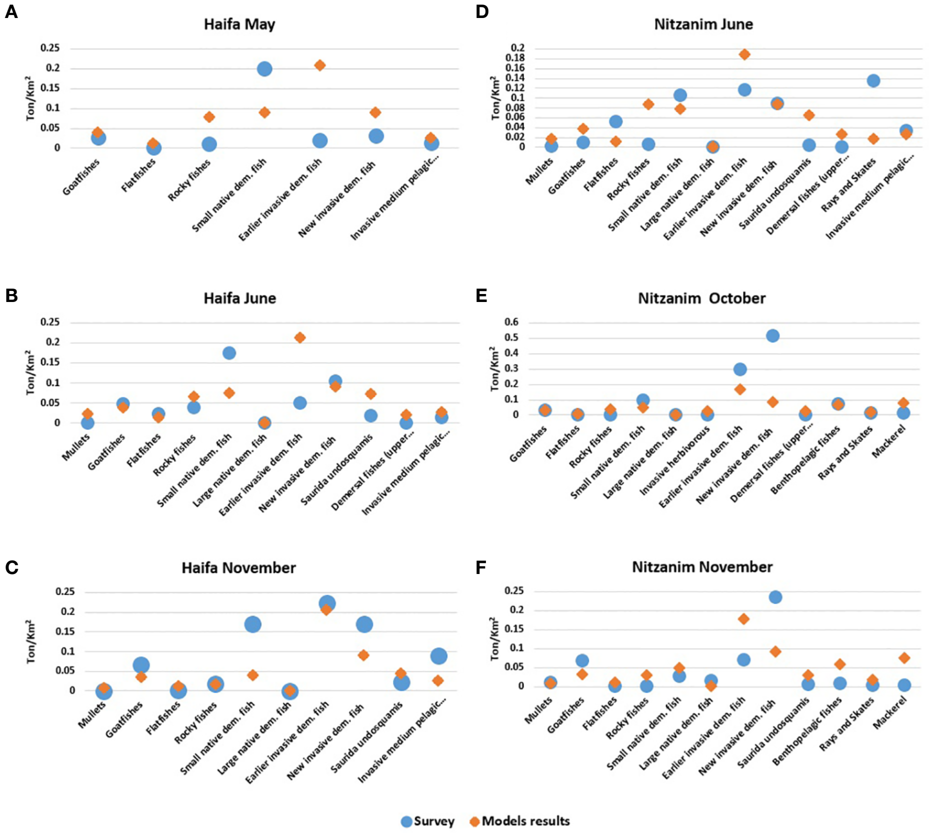

FIGURE 5
Comparison of model results (orange) and survey results (blue). **(A-C)** Haifa: May, June, and November; **(D-F)** Nitzanim: June, October, and November. The numbers represent biomass in t/km^2.

(Figure 6). However, analysis of the outlier groups in May and June in Haifa highlighted the differences in the seasonal variation. The range of seasonal variation in the model results reached 20%, while in the survey results the variation in the extreme cases reached 90% (Figure 6A); this difference in range was observed for five groups. In contrast, there was a large degree of similarity between the model and survey results, in the comparison between November and June in Haifa; eight groups had a large degree of similarity (25-88%) between the November and June results, while only one group exhibited a significant difference of 450% of Rocky fishes (Figure 6A). Even in the case of comparing relationship between November and June in Haifa, to the survey results, only one group had difference between seasons of to 600% (Earlier invasive dem. Fish) while the model results had differences of up to only 60% between seasons (Figure 6B). The results for Nitzanim were similar; the comparison between October and June showed a high correlation (average of 75%) in six groups and differences of up to 600% in the survey results for one group, while the model results demonstrated changes of up to 50% (Figure 6C). The comparison between November and June in Nitzanim showed the same trend with a high correlation (average of 80%) in seven groups and a low correlation (25-16%) in two groups also, the survey results varied between seasons by up to 600%, (Figure 6C) while the model results varied seasonally by up to only 70% (Figure 6D).

Scenario results

Baseline scenario- climate change

In general, outputs from the Ecospace model with the three-climate change driven ocean models resulted in similar trends of change in the food web, though the level of intensity varied between models. Driving Ecospace with the output from MC 1 resulted in the highest biomass results, producing 23% more biomass that the other two models. Driving Ecospace with output from MC 2 and 3 resulted in similar biomass levels (Figure 7).

The results of the baseline climate change scenario (Scenario 1) demonstrated an average decrease in biomass of 16% (from 1.4 to 1.2 ton/km^2) for all the fish in the ecosystem over the course of the scenario (Figure 8A). This result combines a decrease of 37.5% in the biomass of native fish (Figure 8B) and an average increase of 26% in the biomass of invasive fish (Figure 8C). Even the total catch, comprising local species and invasive species, demonstrated a decrease of ca. 11% from 0.3 to 0.27 t/km^2 (Figure 8D).

Climate change scenarios and management steps

In contrast to the baseline scenario, we found a marked increase in the biomass of native fish when we implemented all regulatory measures, namely, limiting fishing during the breeding season, and creating no take zones by networks of nature reserves or by closing

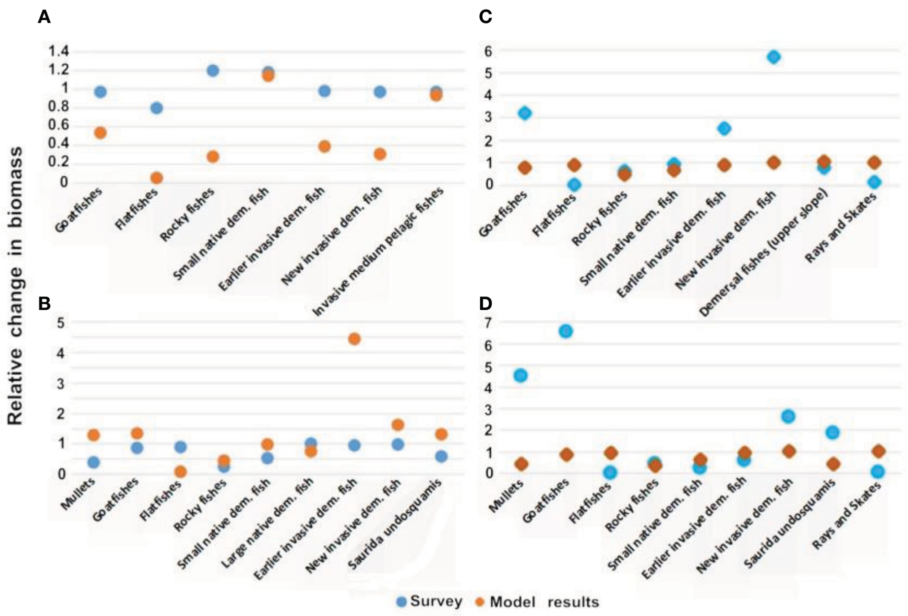

FIGURE 6
Comparison of seasonal differences in the model results and survey data: **(A)** relationship between June and May in Haifa; **(B)** relationship between November and June in Haifa; **(C)** relationship between October and June in Nitzanim; and **(D)** relationship between November and June in Nitzanim. The Y-axis is a relative scale of biomass change. The X-axis are the groups which were found in the month the survey was conducted.

the northern region to fishing (Figure 9). Scenario 5, which represents this option, demonstrated a rapid increase and by 2020, 15 years after the start of the simulation there was a 25% increase in the biomass of native species, with respect to the baseline scenario, and an increase of up to 45% by 2100. In contrast, limiting fishing during the breeding season alone (Scenario 4) led to a moderate (5%) increase in biomass of native species by 2020, and only ca. 2% by 2100 (Figure 9).

The maximum decrease in the biomass of invasive species was achieved when all the management steps were implemented (Scenario 5), namely, limitations to fishing during the breeding season and exclusion of fishing in nature reserves and in the country's north. Under this scenario, the biomass of the invasive species decreased by ca. 28% with respect to the baseline scenario until 2020, and by ca. 23% until 2100. Under the scenario of excluding fishing in the country's north (Scenario 8) the biomass of invasive species decreased by ca. 23% with respect to the baseline scenario until 2020, and by ca. 20% until 2100. The smallest decrease in the biomass of invasive species occurred under the scenario that only limits fishing during the breeding season (Scenario 4), with a decrease of ca. 10% with respect to the baseline scenario until 2020; however, by 2100 the biomass values did not differ from those of the baseline scenario (Figure 10).

Analysis of the expected changes in the catch under different management steps indicate that the management steps will not

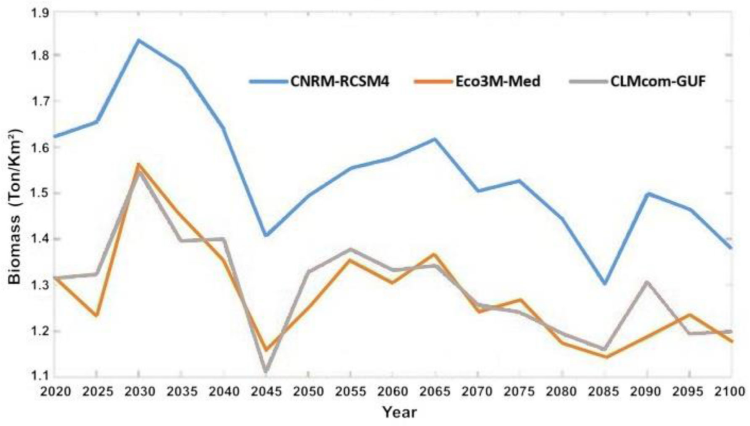

FIGURE 7
Average biomass of fish in ton/km^2 for entire model domain under the three selected ocean models: MC1 (CNRM-RCSM4), MC2 (Eco3M-Med) and MC3 (CLMcom-GUF).

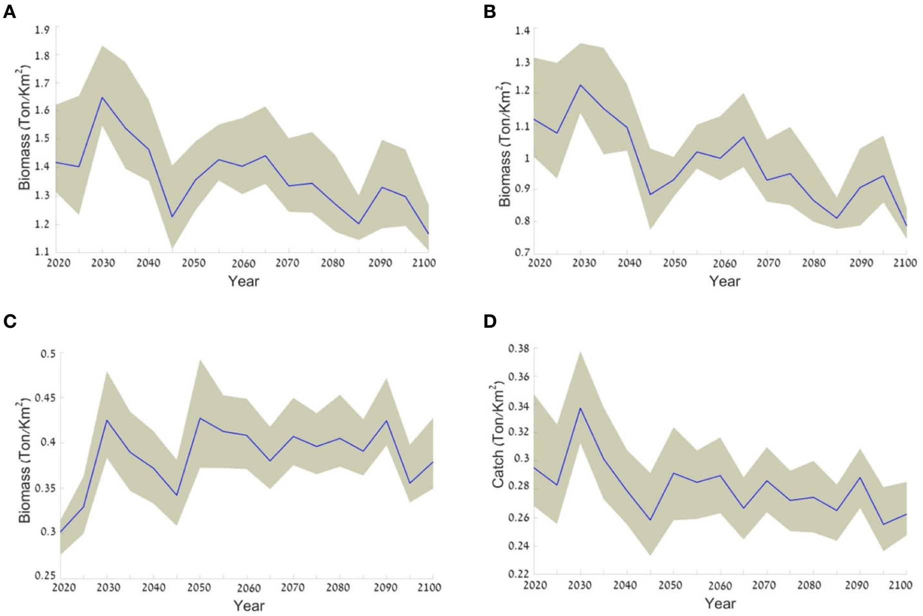

FIGURE 8
Expected changes in the biomass of different groups according to the baseline climate change scenario (Scenario 1). The average is represented by a blue line and the range of values resulting from using the three climate models is represented by the grey area surrounding the average line. The figure presents results for **(A)** all fish groups biomass; **(B)** native fish biomass; **(C)** invasive fish biomass; and **(D)** the total catch. Values are in ton/km^2.

result in an increase in the total catch relative to the baseline scenario for the entire simulation period. Some management steps, however, minimize the impact on the size of the catch and the expected decline is smaller than observed without management measures. While under the base scenario, without any management measures there is about 20% decline in catch by 2020 and about 32% by 2100, there are minimal declines under scenarios 3 and 4. Under both scenario 3, i.e. creation of no-take zones, and scenario 4, i.e. a fishing ban during May and June, the breeding season there is average decline in catch of 5% by 2020 with a variation of 3.4% for scenario 3 and for scenario 4, 5% by 2100 with respect to the baseline scenario. In scenario 5, in which all management steps were implemented the total catch was the lowest with respect to the baseline scenario, with an average decrease of 25% by 2020 and a decrease of 28% by 2100 with variance of 13.2% from 2020 to 2100 (Figure 11).

Effect of management actions on the ecosystem

In all of the scenarios tested, the variation in the Mean Trophic Index (MTI) of the ecosystem was not large, ranging from 3.65 to 3.73, which was lower than the MTI of the baseline scenario. In all scenarios there was a clear trend of an increase in MTI over the period of the scenarios, ranging from 3.63-3.66 in 2020 to 3.68-3.73 in 2100 in the different scenarios (Figure 12A).

Analysis of MTI of the total catch indicates a marked increase from 2020 to 2050, followed by a decrease to 2100. The

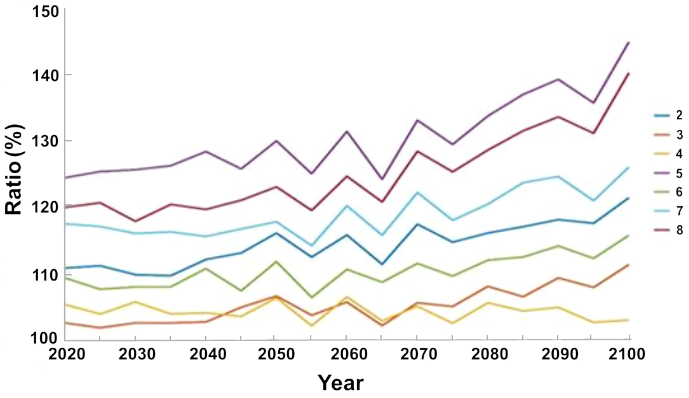

FIGURE 9
Percent variation in the biomass of native species under the different management scenarios (listed in Table 2) with respect to the baseline scenario (year 2020). The results are the average of the three ocean models.

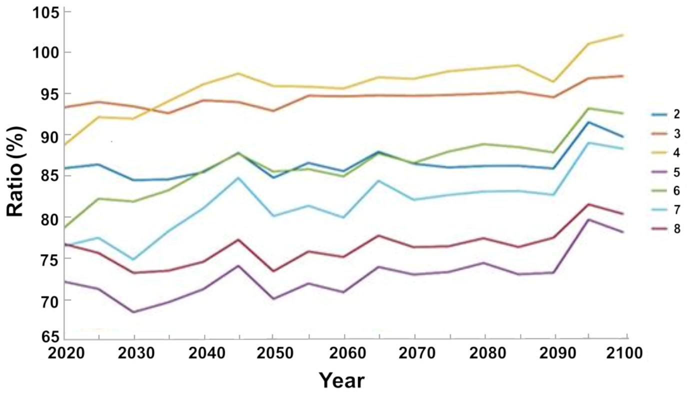

FIGURE 10
Percent variation in the biomass of invasive species under different management scenarios (listed in Table 2) with respect to the baseline scenario that does not include management steps. The results are the average of the three ocean models.

management scenario that excludes fishing in the country's north and in nature reserves (Scenario 5) produced the highest MTI in 2020 and 2050. In contrast, the lowest MTI in the year 2100 occurred in the scenario that only excluded fishing in the country's north (Scenario 2, Figure 12B).

The Kempton's Q results show a small degree of variability between results based on the different ocean model forcing, but a number of patterns did emerge. The lowest diversity, in 2020, when there was no fishing in the north (Scenario 2), i.e. the effect of fishing and climate change on the ecosystem is the lowest for the three examined time periods. And while scenario 2 exhibited low Q values for all time periods it was not the lowest in 2100. Scenario 4, in which seasonal exclusion of fishing was implemented, the index was the highest during all three time periods. In all scenarios, including the baseline scenario, the index increased up to 2050 and then decreased towards 2100. Since this occurs under all scenarios, it appears that up to 2050 the effect of climate change overrides any effects of management scenarios with respect to this index (Figure 12C).

Under all scenarios there was a decrease in the Shannon index over time, with a stronger decrease between 2050 and 2100 (Figure 12D). In 2100, the baseline scenario (Scenario 1), without management steps, had the lowest Shannon index value, while the highest value was found under scenario 7 in which fishing was banned in the country's north and there was a seasonal ban on fishing in general. The results show two notable differences in diversity index values in relation to the other indices. The first, the larger difference in values between 2050 and later periods (2100), the second, was the low variability between the different management scenarios (<0.1) indicating that perhaps the index may not be sensitive enough to the management scenarios examined in this study.

The Achziv (Rosh Hanikra) nature reserve

To examine the benefits of nature reserves to the ecosystem, we chose the Achziv Nature Reserve as a case study. In order to evaluate the effectiveness of the reserve, we used the results from the scenario that included only closures of nature reserves (Scenario 3). These results indicated that within the reserve the fish biomass was higher than outside the reserve. In addition, there was a higher biomass of native species and lower biomass of invasive species

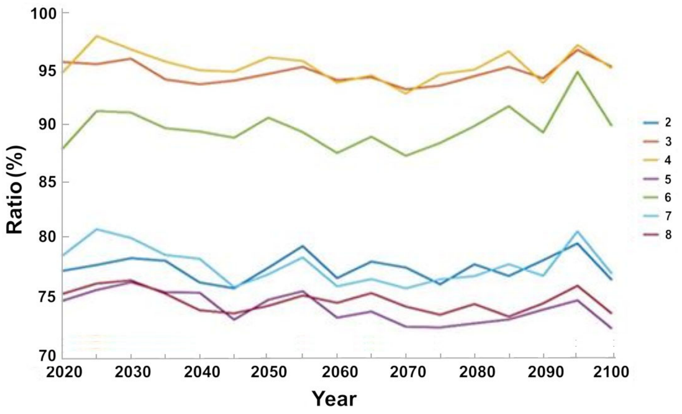

FIGURE 11
The change in the total catch under the different management scenarios with respect to the baseline scenario. The results are the average of the three ocean models.

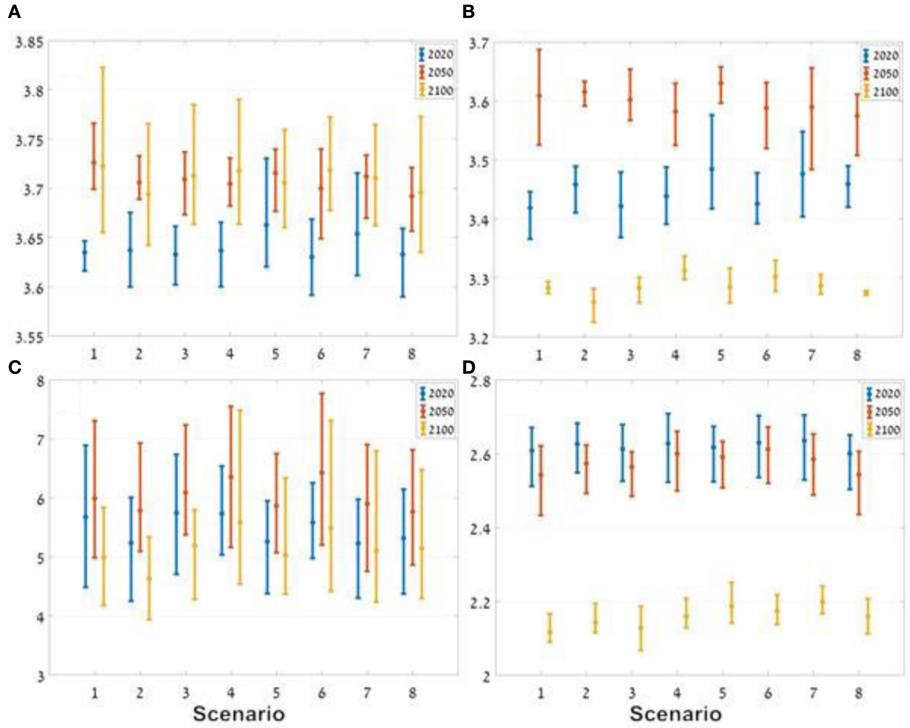

FIGURE 12
Results of the indicators for the three ocean models under the different management scenarios. The horizontal axes represent the different scenarios (1-8) and the vertical axes represent the values of the different indices: **(A)** MTI (mean trophic index); **(B)** total catch; **(C)** Kempton's Q; and **(D)** Shannon diversity index. The vertical lines represent the distribution of the results of the three ocean models around the average (dot). The blue line represents the results for 2020, the red line represents the results for 2050, and the yellow line represents the results for 2100.

within the reserve though the variation in the invasive species group was higher than the native species group and the fish group (Figure 13).

Discussion

There is much uncertainty concerning the severity of the climate change that is projected to take place over the coming decades and the impacts of the possible changes on aquatic ecosystems. In this study, we tested the possible consequences of climate change on the ecosystem of the Eastern Mediterranean based on the more severe IPCC RCP 8.5 scenario (Smith et al., 2014) (Pagès et al., 2020).

In light of the uncertainty in models, extensive use has been made of model ensembles for testing scenarios. In the current study, we implemented the model ensemble approach in order to include the range of uncertainty regarding the future environmental conditions that will affect the ecosystem according to our scenarios. The use of model ensembles, though common in climate studies, is still rare in respect to all aspects of implementation of marine spatial ecological models (Steenbeek et al., 2021). There are, however, a number of examples such as (Bourdaud et al. (2021) who used RCP2.6 and RCP8.5 in order to analyze the impact on species distribution using an ecological model and additional studies at the global scale (Lotze et al., 2019; Coll et al., 2020; Tittensor et al., 2021).

We used the output from three different climate-ocean models (Darmaraki et al., 2019; Pagès et al., 2020) in order to test the impact on the ecosystem. The approach we used allowed us to test a possible response of the ecosystem under a range of future conditions. In this case, similarity in the trends of responses to environmental changes and management steps reinforces our results for drawing conclusions.

The results of the basic climate change scenario (Scenario 1) under RCP8.5 pointed to a number of trends in total biomass of fish, the size of the fish catch, and changes in the biomass ratio between invasive and native species. In this scenario, the fish catch decreased on average by ca. 11% and the average biomass of all the fish in the ecosystem decreased by 16% throughout the period of the scenario. The results are consistent with global projections that show a reduction in the biomass of a variety of species under RCP 8.5 Tittensor et al. (2021). Furthermore, our scenario showed an average increase of 26% in the biomass of invasive species, but a 37.5% decrease in the biomass of native species. These results support the trends identified in other studies conducted in Israel, indicating migration of native species to regions with colder, deeper water, some of which are located outside our model domain (Chaikin et al., 2022). The results also illustrated the impact of climate change not only on biomass and fish catch but also on

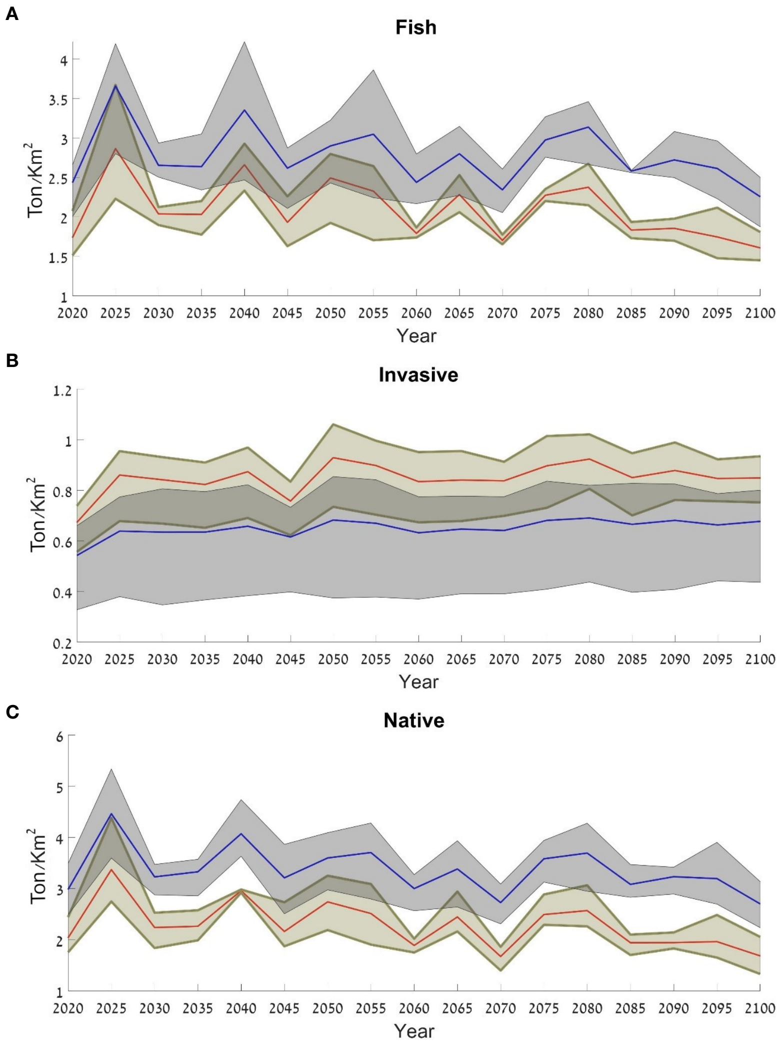

FIGURE 13
Average fish biomass for the three ocean models assuming closure of protected areas (Scenario 3), in and around Achziv Nature Reserve: **(A)** all fish species; **(B)** invasive species; **(C)** native species. The blue line represents the results within the reserve and the red line represents the results outside of the reserve. The results are expressed as t/km^2.

species composition. Higher temperatures, according to the results of the basic scenario, led to an increasing trend in the relative abundance of invasive species in the community, similar to the phenomenon taking place in other regions of the world (Geburzi and McCarthy, 2018). This corresponds to previous studies in our region that show the relationship between the increase in sea water temperature and the biomass of the invasive and local species (Corrales et al., 2018; Corrales, 2019).

The increased water temperature also had an impact on the fish catch. The catch displayed a decrease in size, and also a change in composition, such that it included a large proportion of invasive species and less native species. This is similar to the phenomenon observed in the northern Pacific Ocean where marine heatwaves are worsening the impacts of climate change on 795 fisheries. This is due to the increased ocean temperatures causing changes in the distribution and abundance of fish, leading to economic and ecological consequences for these fisheries (Cheung and Frölicher (2020); Cheung and Frölicher (2020).

Ranking the results of each scenario is a way to analyze the results and attempt to find the management steps that maximize the benefits. Based on the results of the scenarios, we assigned a score to each of the scenarios based on the degree of its contribution to the increase in the biomass of local species, the decrease in the biomass of the invasive species and the changes in catch. Analysis of the scenarios revealed that all management actions improved the state of the native species, caused a decline in the biomass of invasive species and preserved fish catch. Nevertheless, a number of management actions were noticeably more effective than others (Table 3). Scenario 5 provided the best desired results for both the ecosystem and fish catch. This result was achieved by combining all of the management options, namely, combining no-take zones protected from fishing, cessation of fishing in the north and cessation of fishing during May and June in other areas. The significance of this scenario was the creation of a network of protected areas in which the native species can thrive, thus minimizing their displacement by invasive species, as well as complete cessation of fishing for two months to enable the fish populations to survive. However, scenario 5 displayed one of the lowest scores for the fish catch; this stemmed from the creation of no-take zones as well as the complete cessation of fishing for two months.

Another management action that had a positive effect on the fish catch was cessation of fishing during May and June (Scenario 4). The effect of this step on the fish biomass in the long run was not surprising since after cessation of fishing, great effort is made to "make up for lost time", as reported from other locations (Sterner, 2012). Nevertheless, this management step had almost no benefits on the native species; thus, from an ecological perspective it does not contribute to protection of the ecosystem.

It appears that creating no-take zones that enable native species to survive undisturbed as well as steps to restrict fishing (e.g., Scenario 8) facilitated protection of the ecosystem and had some level of benefit to the fish catch. Implementation of this policy requires an effort on the part of the government, since creating no-take zones and restricting fishing to certain times of the year attract great opposition from the fishing community, resulting in conflicts. However, the results of the scenarios clearly show that implementation of such steps is one of the best options to improve the catch and the ecosystem (e.g., Scenarios 3 and 6). These results are in line with reports on the implementation of management steps in other locations around the world (Vilas et al., 2020; Hoppit et al., 2022).

We used ecological indicators to test the consequences of future changes on the ecosystem itself (Susini and Todd, 2021). To understand the impact of management steps on the entire ecosystem, we calculated, for each indicator, the average value over the duration of the scenarios. The advantage of this approach was that it provided a general perspective of the system, it discounts, however, the spatial variation between different locations. Overall, in the baseline scenario all the indicators pointed to a decrease in ecosystem functioning between 2050 and 2100. This may be related to the constant increase in sea water temperature which in some climate scenarios become more intense towards the end of the test period. Rising temperatures are expected to impact ecosystem functioning (Bourdaud et al., 2021).

Examination of MTI showed almost no difference between the management scenarios (despite differences in the proportions of native and invasive species). From this we can conclude that ecologically, there is a phenomenon of displacement of native species by invasive species that take their place in the trophic level of the food web. This is in line with previous studies (Corrales et al., 2017a; Corrales et al., 2017b; Corrales et al., 2018; Corrales, 2019) that indicated displacement, mainly of crabs and shrimps. Nevertheless, the spatial extent of the model is limited to a depth of 200 m, thus, our model cannot detect a situation where native species move to greater depths in search of better thermal conditions or because they have been displaced by invasive species.

Indices of species diversity indicated changes in ecosystem functioning (Goswami et al., 2017). Examination of Kempton's Q, which tests the biodiversity (Ainsworth and Pitcher, 2006; Coll and Steenbeek, 2017) shows that Scenario 2, in which the northern zone was closed to fishing, is the scenario with the lowest biodiversity. This result is not surprising, since stricter restrictions on fishing lead to more biomass across the food web, resulting in lower index values. From a management perspective, we can conclude that closing the northern zone is more effective than creation of no-take zones. However, since this is not the decision makers' sole aim, this step should be considered in the light of other objectives.

Evaluating expected changes in diversity based on the Shannon index highlighted a number of patterns. Firstly, the notable decline in index values between 2050 and 2100. In all management scenarios, this is as a response to climate change. Secondly, the low variability between results based on the three different ocean models, and, thirdly, was the low variability between the different management scenarios. The low variability between the three models and scenarios reflect the displacement phenomenon described in this and other studies (Goren et al., 2016; Corrales et al., 2017a), where invasive species fulfill the same role in the ecosystem as the native species they replace. This insight is reinforced by the fact that the changes in the trophic level of the fish catch are also relatively small; in other words, we also see replacement of native species by invasive species in the fish catch. This has already been taking place for some time and is expected to continue into the future.

One of the most common management steps for protecting fish populations and ecosystems is the declaration of nature reserves (Dimitriadis et al., 2018; Grorud-Colvert et al., 2021). Currently, the option of declaring a number of nature reserves along Israel's Mediterranean coast is being considered, with the aim of protecting native species, unique species and keystone species

TABLE 3 Ranking of the results of the management scenarios, and their impact on native species, invasive species and the fish catch.

Scenario	Native biomass rank	Invasive biomass rank	Total catch rank	Total rank
2	4	4	4	12
3	6	6	2	14
4	7	8	1	16
5	1	1	7	9
6	5	5	3	13
7	3	3	5	11
8	2	2	6	10

Results are ranked from 1 to 8, where 1 is the best result and 8 is the worst with respect to the desired results, i.e., maximum number of native species, minimum number of invasive species and maximum total fish catch. The summary column is the total of the three previous columns (biomass of native species, biomass of invasive species and total fish catch).

(Seker, 2015). However, we must ask whether the reserves will fulfill their role under projected climate change. We chose to focus on the results of the scenarios for Achziv Reserve because it is an active, declared reserve, and many surveys have been performed both inside and outside of the reserve to monitor ecosystem dynamics. The model results indicated that the reserve is fulfilling its role: the total number of fish species and the number of native species inside the reserve exceed the numbers in the area around the reserve, while the number of invasive species is lower inside the reserve than outside of it. These results are in line with the results of the abovementioned surveys (NPA, 2015) and other studies in the Mediterranean (D'Amen and Azzurro, 2019) and in Israel in places where there is continuous monitoring of protected areas compared to what happens outside the protected area (such as the Achziv reserve in the northern part of the territorial waters in Israel). It seems that there is an increase in the amount of local species inside the reserve compared to the area outside (Frid et al., 2022). This observation also indirectly verifies the model.

Analysis of the scenario maps results revealed that in areas adjacent to nature reserves there is an increase in the fish catch (Appendix 1/Figure A3). These results are not surprising and are in line with the phenomenon known as spillover or “fishing the line”, where fishermen wait at the edges of the nature reserves and enjoy an abundant fish catch that comes from the reserve (Kellner et al., 2007; Nillos Kleiven et al., 2019; Grip and Blomqvist, 2020). Examination of the fish catch maps from different years showed that in scenarios with implementation of nature reserves there are several main zones with unusually high fish catches. One is in the northern part of the ICS, between three nature reserves, while the other is in the center of the country, north of a large nature reserve (Appendix 1/Figure A3).

The detection of areas with a large fish catch led to the idea of creating designated fishing zones. Currently, the continental shelf is under a range of pressures from a large number of uses. Fishing zones are usually those left after omitting all of the zones in which fishing cannot take place. As far as we know, the idea of creating designated fishing zones in Israel is innovative and it is based on the idea of Ramírez-Luna and Chuenpagdee (2019); Ramírez-Luna and Chuenpagdee (2019) which indicated the development of such areas for Latin America and the Caribbean for small-scale fishing that can support the preservation of this activity alongside provision of fish from the sea. Establishment of such zones can be integrated into spatial planning maps that restrict the other users and allow only fishing or other activities that do not interfere with fishing. The location of these zones adjacent to nature reserves will allow the fishermen to take advantage of the fish catch that develops inside the reserve and trickles out.

Verifying the results of a spatial ecological model is not trivial. As ecological models have developed, the statistical methods designed to test the quality and accuracy of the model results have also developed. Hipsey et al. (2020); Hipsey et al. (2020) suggested an approach that can be used to validate the results of ecological models and mainly to ensure the appropriate use of a model for scenario testing and application as a management tool. The validation process provides tools designed to test the results and identify situations where model require further development or calibration. The state validation processes, as presented by Hipsey et al. (2020); Hipsey et al. (2020), compares simulated state variables, such as biomass or fish catch, and compare them to the range of expected values in reality, based on monitoring, surveys and sampling. This method is the most common form of model verification, and indicates the level of reliability of the results, particularly those models required for decision-making processes (Link, 2021). The ICS Ecosim model underwent state validation (Corrales et al., 2017a). In this study we implemented spatial state validation based on the surveys conducted in two regions over several seasons. In addition, we perform a restricted form of system-level emergent properties validation (Hipsey et al., 2020) as can be seen in the similar patterns that emerged from studying the impact of the Ackziv nature reserve on the fish communities. Validation of spatial ecosystem models is uncommon and challenging (Steenbeek et al., 2021). And indeed there have been very few cases of validation of Ecospace models (e.g. (Grossowicz et al., 2020); in some studies the results of species distribution maps were compared to distribution models using statistical tools (Coll et al., 2019).

In order to verify the spatial model, we compared the results of the scientific surveys to the results of the model along a defined transect. The aim was to test the reliability of the model both in locations with different characteristics (Haifa and Nitzanim) and during different months (May, June, October and November); thus, the validation was performed across both the spatial and temporal (seasonal) dimensions. The results from our validation demonstrate that the model was successful in simulating most of the functional groups in the model, in both locations, and in each of the months, with average deviations of -15% to 30% between the model results and the survey results. However, in this study, due to a lack of available data, we performed the comparison for only one year. In the future it is imperative to use the same method to compare the results for additional years.

The seasonal comparison between the survey results and the model results was performed with respect to June in both test locations. The results show that all four cases demonstrated a high correlation between the model results and the survey results for most groups; only one group (earlier invasive dem. Fish) presented a noticeable difference between the model results and the survey results. Due to the sampling methodology (trawling) used in the survey, it is possible that the large differences observed between the surveys and model results are a consequence of the sampling method used.

We recognize that the modelling complex contains significant sources of uncertainty that we did not yet address in this study. The eastern Mediterranean will likely experience additional species invasions, and when available, different climate forecasts that exceed the 8.5 emission scenario in severity should be considered. Additionally, assumptions in the EwE parameterization are subject to uncertainty that may affect model predictability. Follow-up studies should systematically assess these sources of uncertainty (Steenbeek et al., 2021) once the technological framework required to perform such assessments is available (Steenbeek et al., In prep).

Climate change is happening now and the impact of rising sea temperature on the ecosystem has already been recorded and demonstrated through a number of studies in our region and in other regions around the world (Herut, 2021; Tuel and Eltahir, 2020). Consequences of the climate changes occurring in our region are reflected in the need for native species to adjust the range of temperatures in which they reside. In addition, invasive species that invaded the region, and that are adapted to the higher range of temperatures in the Red Sea, are successfully colonizing and inhabiting the shallow, warmer waters of the Israeli continental shelf (Arndt et al., 2018). In addition, invasive species that are not currently known to us in the study area may arrive and change the dynamics of the ecosystem and the fishery. These changes cannot be predicted, so it will be necessary to examine the model's products and policy recommendations from time to time and according to the changes in the ecosystem.

By combining an ecological model with climate change scenarios for the purpose of testing different management scenarios, we demonstrate the variability of the ecosystem as a response to different management scenarios, and provide decision makers with tools for selecting the most suitable management measures to mitigate possible climate effects on the ecosystem.

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

EO, and GG conceived the study. XC provided the data for the Ecosapce model. JS, MC, SH helped in model construction and testing. MG provided data and knowledge on the ecosystem. YA, NS, helped in data and spatial layers preparation. All authors contributed to the article, reviewed the manuscript and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1155480/full#supplementary-material

References

Abdou, K., Halouani, G., Hattab, T., Romdhane, M. S., Lasram, F. B. R., and Le Loc'h, F. (2016). Exploring the potential effects of marine protected areas on the ecosystem structure of the gulf of gabes using the ecospace model. *Aquat. Living Resour.* 29, 202. doi: 10.1051/alr/2016014

Ainsworth, C. H., and Pitcher, T. J. (2006). Modifying kempton's species diversity index for use with ecosystem simulation models. *Ecol. Indic.* 6, 623–630. doi: 10.1016/j.ecolind.2005.08.024

Apicella, L., De Martino, M., and Quarati, A. (2022). Copernicus User uptake: from data to applications. *ISPRS Int. J. Geo-Inform.* 11, 121. doi: 10.3390/ijgi11020121

Arndt, E., Givan, O., Edelist, D., Sonin, O., and Belmaker, J. (2018). Shifts in eastern Mediterranean fish communities: abundance changes, trait overlap, and possible competition between native and non-native species. *Fishes* 3, 19. doi: 10.3390/fishes3020019

Bagsit, F. U., Frimpong, E., Asch, R. G., and Monteclaro, H. M. (2021). Effect of a seasonal fishery closure on sardine and mackerel catch in the visayan Sea, Philippines. *Front. Mar. Sci.* 8, 640772. doi: 10.3389/fmars.2021.640772

Baklouti, M., Pagès, R., Alekseenko, E., Guyennon, A., and Grégori, G. (2021). On the benefits of using cell quotas in addition to intracellular elemental ratios in flexiblestoichiometry plankton functional type models. application to the Mediterranean Sea. *Prog. Oceanography* 197, 102634. doi: 10.1016/j.pocean.2021.102634

Bloomfield, E. F., and Manktelow, C. (2021). Climate communication and storytelling. *Climatic Change* 167, 34. doi: 10.1007/s10584-021-03199-6

Bourdaud, P., Ben Rais Lasram, F., Araignous, E., Champagnat, J., Grusd, S., Halouani, G., et al. (2021). Impacts of climate change on the bay of seine ecosystem: forcing a spatio-temporal trophic model with predictions from an ecological niche model. *Fish. Oceanography* 30, 471–489. doi: 10.1111/fog.12531

Bryndum-Buchholz, A., Tittensor, D. P., Blanchard, J. L., Cheung, W. W., Coll, M., Galbraith, E. D., et al. (2019). Twenty-first-century climate change impacts on marine animal biomass and ecosystem structure across ocean basins. *Global Change Biol.* 25, 459–472. doi: 10.1111/gcb.14512

Bucchignani, E., Mercogliano, P., Panitz, H.-J., and Montesarchio, M. (2018). Climate change projections for the middle East–north Africa domain with COSMO-CLM at different spatial resolutions. *Adv. Climate Change Res.* 9, 66–80. doi: 10.1016/j.accre.2018.01.004

Cacaud, P. (2005). *Fisheries laws and regulations in the Mediterranean: a comparative study* (Rome: Food and Agriculture Organization of the United Nations). Available at: https://epub.sub.uni-hamburg.de/epub/volltexte/2011/11880/pdf/752005y5880e00.pdf.

Chaikin, S., Dubiner, S., and Belmaker, J. (2022). Cold-water species deepen to escape warm water temperatures. *Global Ecol. Biogeography* 31, 75–88. doi: 10.1111/geb.13414

Cheung, W. W., and Frölicher, T. L. (2020). Marine heatwaves exacerbate climate change impacts for fisheries in the northeast pacific. *Sci. Rep.* 10, 1–10. doi: 10.1038/s41598-020-63650-z

Christensen, V., Coll, M., Steenbeek, J., Buszowski, J., Chagaris, D., and Walters, C. J. (2014a). Representing variable habitat quality in a spatial food web model. *Ecosystems* 17, 1397–1412. doi: 10.1007/s10021-014-9803-3

Christensen, V., Coll, M., Steenbeek, J., Buszowski, J., Chagaris, D., and Walters, C. J. (2014b). Representing variable habitat quality in a spatial food web model. *Ecosystems* 17, 1397–1412. doi: 10.1007/s10021-014-9803-3

Christensen, V., and Walters, C. J. (2004). Ecopath with ecosim: methods, capabilities and limitations. *Ecol. Model.* 172, 109–139. doi: 10.1016/j.ecolmodel.2003.09.003

Christensen, V., and Walters, C. J. (2005). *Using ecosystem modeling for fisheries management: where are we* Vol. 1000 (University of British Columbia, Fisheries Centre, 2202 Main Mall, Vancouver BC, Canada V6T 1Z4: ICES CM), 19. Available at: https://www.ices.dk/sites/pub/CM%20Doccuments/2005/M/M1905.pdf.

Coll, M., Akoglu, E., Arreguin-Sanchez, F., Fulton, E., Gascuel, D., Heymans, J., et al. (2015). Modelling dynamic ecosystems: venturing beyond boundaries with the ecopath approach. *Rev. Fish. Biol. Fish.* 25, 413–424. doi: 10.1007/s11160-015-9386-x

Coll, M., Bundy, A., and Shannon, L. J. (2009). "Ecosystem modelling using the ecopath with ecosim approach," in *Computers in fisheries research*. Eds. B. A. Megrey and E. Moksness (Van Godewijckstraat 30 3311 GX Dordrecht Netherlands: Springer Netherlands), 225–291. Available at: https://link.springer.com/chapter/10.1007/978-1-4020-8636-6_8.

Coll, M., Pennino, M. G., Steenbeek, J., Solé, J., and Bellido, J. M. (2019). Predicting marine species distributions: complementarity of food-web and Bayesian hierarchical modelling approaches. *Ecol. Model.* 405, 86–101. doi: 10.1016/j.ecolmodel.2019.05.005

Coll, M., Shannon, L., Kleisner, K. M., Juan-Jordá, M., Bundy, A., Akoglu, A., et al. (2016a). Ecological indicators to capture the effects of fishing on biodiversity and conservation status of marine ecosystems. *Ecol. Indic.* 60, 947–962. doi: 10.1016/j.ecolind.2015.08.048

Coll, M., and Steenbeek, J. (2017). Standardized ecological indicators to assess aquatic food webs: the ECOIND software plug-in for ecopath with ecosim models. *Environ. Model. Softw.* 89, 120–130. doi: 10.1016/j.envsoft.2016.12.004

Coll, M., Steenbeek, J., Pennino, M. G., Buszowski, J., Kaschner, K., Lotze, H. K., et al. (2020). Advancing global ecological modeling capabilities to simulate future trajectories of change in marine ecosystems. *Front. Mar. Sci.* 7, 567877. doi: 10.3389/fmars.2020.567877

Coll, M., Steenbeek, J., Sole, J., Palomera, I., and Christensen, V. (2016b). Modelling the cumulative spatial–temporal effects of environmental drivers and fishing in a NW Mediterranean marine ecosystem. *Ecol. Model.* 331, 100–114. doi: 10.1016/j.ecolmodel.2016.03.020

Corrales, X. (2019). *Ecosystem modelling in the Eastern Mediterranean Sea: the cumulative impact of alien species, fishing and climate change on the Israeli marine ecosystem*. Available at: https://link.springer.com/article/10.1007/s10530-019-02160-0.

Corrales, X., Coll, M., Ofir, E., Heymans, J. J., Steenbeek, J., Goren, M., et al. (2018). Future scenarios of marine resources and ecosystem conditions in the Eastern Mediterranean under the impacts of fishing, alien species and sea warming. *Sci. Rep.* 8, 14284. doi: 10.1038/s41598-018-32666-x

Corrales, X., Coll, M., Ofir, E., Piroddi, C., Goren, M., Edelist, D., et al. (2017a). Hindcasting the dynamics of an Eastern Mediterranean marine ecosystem under the impacts of multiple stressors. *Mar. Ecol. Prog. Ser.* 580, 17–36. doi: 10.3354/meps12271

Corrales, X., Katsanevakis, S., Coll, M., Heymans, J. J., Piroddi, C., Ofir, E., et al. (2020). Advances and challenges in modelling the impacts of invasive alien species on aquatic ecosystems. *Biol. Invasions* 22, 907–934. doi: 10.1007/s10530-019-02160-0

Corrales, X., Ofir, E., Coll, M., Goren, M., Edelist, D., Heymans, J., et al. (2017b). Modeling the role and impact of alien species and fisheries on the Israeli marine continental shelf ecosystem. *J. Mar. Syst.* 170, 88–102. doi: 10.1016/j.jmarsys.2017.02.004

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

D'Amen, M., and Azzurro, E. (2019). Lessepsian fish invasion in Mediterranean marine protected areas: a risk assessment under climate change scenarios. *ICES J. Mar. Sci.* 77, 388–397. doi: 10.1093/icesjms/fsz207

Darmaraki, S., Somot, S., Sevault, F., and Nabat, P. (2019). Past variability of Mediterranean Sea marine heatwaves. *Geophysical Res. Lett.* 46, 9813–9823. doi: 10.1029/2019GL082933

De Mutsert, K., Lewis, K. A., White, E. D., and Buszowski, J. (2021). End-to-End modeling reveals species-specific effects of large-scale coastal restoration on living resources facing climate change. *Front. Mar. Sci.* 8, 104. doi: 10.3389/fmars.2021.624532

Dimitriadis, C., Sini, M., Trygonis, V., Gerovasileiou, V., Sourbès, L., and Koutsoubas, D. (2018). Assessment of fish communities in a Mediterranean MPA: can a seasonal no-take zone provide effective protection? *Estuarine Coast. Shelf Sci.* 207, 223–231. doi: 10.1016/j.ecss.2018.04.012

Edelist, D., Rilov, G., Golani, D., Carlton, J. T., and Spanier, E. (2013). Restructuring the s ea: profound shifts in the world's most invaded marine ecosystem. *Diversity Distributions* 19, 69–77. doi: 10.1111/ddi.12002

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., et al. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature* 506, 216–220. doi: 10.1038/nature13022

Escalas, A., Avouac, A., Belmaker, J., Bouvier, T., Clédassou, V., Ferraton, F., et al. (2022). An invasive herbivorous fish (Siganus rivulatus) influences both benthic and planktonic microbes through defecation and nutrient excretion. *Sci. Total Environ.* 838, 156207. doi: 10.1016/j.scitotenv.2022.156207

Frazão Santos, C., Ehler, C. N., Agardy, T., Andrade, F., Orbach, M. K., and Crowder, L. B. (2019). "Chapter 30 - marine spatial planning," in *World seas: an environmental evaluation*, *2nd ed.* Ed. C. Sheppard (London, United Kingdom: Academic Press), 571–592. Available at: https://www.sciencedirect.com/science/article/abs/pii/B9780128050521000334.

Frid, O., Lazarus, M., Malamud, S., Belmaker, J., and Yahel, R. (2022). Effects of marine protected areas on fish communities in a hotspot of climate change and invasion. *Mediterr. Mar. Sci.* 23, 157–190. doi: 10.12681/mms.26423

Galil, B. S., Marchini, A., and Occhipinti-Ambrogi, A. (2018). East Is east and West is west? management of marine bioinvasions in the Mediterranean Sea. *Estuarine Coast. Shelf Sci.* 201, 7–16. doi: 10.1016/j.ecss.2015.12.021

Galil, B., Marchini, A., Occhipinti-Ambrogi, A., and Ojaveer, H. (2017). The enlargement of the Suez canal–erythraean introductions and management challenges. *Manage. Biol. Invasions* 8, 141–152. doi: 10.3391/mbi.2017.8.2.02

Galil, B. S., and Zenetos, A. (2002). "A sea change–exotics in the Eastern Mediterranean Sea," in *Invasive aquatic species of europe. distribution, impacts and management* (Dordrecht Netherlands: Springer), 325–336. https://link.springer.com/chapter/10.1007/978-94-015-9956-6_33

Geburzi, J. C., and McCarthy, M. L. (2018). "How do they do it?–understanding the success of marine invasive species," in *YOUMARES 8–oceans across boundaries: learning from each other* (Cham: Springer), 109–124.

Giakoumi, S., Scianna, C., Plass-Johnson, J., Micheli, F., Grorud-Colvert, K., Thiriet, P., et al. (2017). Ecological effects of full and partial protection in the crowded Mediterranean Sea: a regional meta-analysis. *Sci. Rep.* 7, 1–12. doi: 10.1038/s41598-017-08850-w

Givan, O., Edelist, D., Sonin, O., and Belmaker, J. (2018). Thermal affinity as the dominant factor changing Mediterranean fish abundances. *Global Change Biol.* 24, e80–e89. doi: 10.1111/gcb.13835

Golani, D. (2010). "Colonization of the Mediterranean by red Sea fishes via the Suez canal-lessepsian migration," in *Fish invasions of the Mediterranean Sea: change and renewal.* 145, 188.

Goren, M., Galil, B. S., Diamant, A., Stern, N., and Levitt-Barmats, Y. A. (2016). Invading up the food web? invasive fish in the southeastern Mediterranean Sea. *Mar. Biol.* 163, 180. doi: 10.1007/s00227-016-2950-7

Goswami, M., Bhattacharyya, P., Mukherjee, I., and Tribedi, P. (2017). Functional diversity: an important measure of ecosystem functioning. *Adv. Microbiol.* 7, 82. doi: 10.4236/aim.2017.71007

Grip, K., and Blomqvist, S. (2020). Marine nature conservation and conflicts with fisheries. *Ambio* 49, 1328–1340. doi: 10.1007/s13280-019-01279-7

Grorud-Colvert, K., Sullivan-Stack, J., Roberts, C., Constant, V., Horta e Costa, B., Pike, E. P., et al. (2021). The MPA guide: a framework to achieve global goals for the ocean. *Science* 373, eabf0861. doi: 10.1126/science.abf0861

Grossowicz, M., Ofir, E., Shabtay, A., Wood, J., Biton, E., Belkin, N., et al. (2020). Modeling the effects of brine outflow from desalination plants on coastal food-webs of the levantine basin (eastern Mediterranean Sea). *Desalination* 496, 114757. doi: 10.1016/j.desal.2020.114757

Heymans, J., Coll, M., Libralato, S., and Christensen, V. (2012). "Ecopath theory, modelling and application to coastal ecosystems," in *Treatise on estuarine and coastal science* (London, United Kingdom: Elsevier), 93–113. Available at: https://ricerca.ogs.it/handle/20.500.14083/3743.

Heymans, J. J., Coll, M., Link, J. S., Mackinson, S., Steenbeek, J., Walters, C., et al. (2016). Best practice in ecopath with ecosim food-web models for ecosystem-based management. *Ecol. Model.* 331, 173–184. doi: 10.1016/j.ecolmodel.2015.12.007

Herut, B. (2021). "*The national monitoring program of Israel's Mediterranean Waters – scientific perspectives*". Available at: https://library.oapen.org/handle/20.500.12657/62832.

Hipsey, M. R., Gal, G., Arhonditsis, G. B., Carey, C. C., Elliott, J. A., Frassl, M. A., et al. (2020). A system of metrics for the assessment and improvement of aquatic ecosystem models. *Environ. Model. Softw.* 128, 104697. doi: 10.1016/j.envsoft.2020.104697

Hoppit, G., Schmidt, D. N., Brazier, P., Mieszkowska, N., and Pieraccini, M. (2022). Are marine protected areas an adaptation measure against climate change impacts on coastal ecosystems? a UK case study. *Nature-Based Solutions* 2, 100030. doi: 10.1016/j.nbsj.2022.100030

Horta e Costa, B., Claudet, J., Franco, G., Erzini, K., Caro, A., and Gonçalves, E. J. (2016). A regulation-based classification system for marine protected areas (MPAs). *Mar. Policy* 72, 192–198. doi: 10.1016/j.marpol.2016.06.021

Hulme, M. (2016). *Intergovernmental panel on climate change (IPCC), international encyclopedia of geography*. (U.S.A: Wiley Online Library). 1–7. Available at: https://onlinelibrary.wiley.com/doi/abs/10.1002/9781118786352.wbieg0254.pub2.

Jones, M. C., and Cheung, W. W. (2015). Multi-model ensemble projections of climate change effects on global marine biodiversity. *ICES J. Mar. Sci.* 72, 741–752. doi: 10.1093/icesjms/fsu172

Kanari, M., Tibor, G., Hall, J. K., Ketter, T., Lang, G., and Schattner, U. (2020). Sediment transport mechanisms revealed by quantitative analyses of seafloor morphology: new evidence from multibeam bathymetry of the Israel exclusive economic zone. *Mar. Petroleum Geol.* 114, 104224. doi: 10.1016/j.marpetgeo.2020.104224

Kaschner, A., Kesner-Reyes, K., Garilao, C., Rius-Barile, J., Rees, T., and Froese, R. (2016). *Predicted range maps for aquatic species.* (Paris, France: World Wide Web Electron Publication). Version, 2008. Available at: www.aquamaps.org.

Katsanevakis, S., Coll, M., Piroddi, C., Steenbeek, J., Ben Rais Lasram, F., Zenetos, A., et al. (2014). Invading the Mediterranean Sea: biodiversity patterns shaped by human activities. *Front. Mar. Sci.* 1, 32. doi: 10.3389/fmars.2014.00032

Kellner, J. B., Tetreault, I., Gaines, S. D., and Nisbet, R. M. (2007). Fishing the line near marine reserves in single and multispecies fisheries. *Ecol. Appl.* 17, 1039–1054. doi: 10.1890/05-1845

Kikstra, J. S., Nicholls, Z. R., Smith, C. J., Lewis, J., Lamboll, R. D., Byers, E., et al. (2022). The IPCC sixth assessment report WGIII climate assessment of mitigation pathways: from emissions to global temperatures. *Geosci. Model. Dev.* 15, 9075–9109. doi: 10.5194/gmd-15-9075-2022

Kleisner, K., and Pauly, D. (2011). *The marine trophic index (MTI), the fishing in balance (FIB) index. fisheries centre research reports.* (2202 Main Mall, Vancouver BC V6T1Z4 Canada: Sea Around Us Project, Fisheries Centre, University of British Columbia) 19, 41. Available at: http://www.ecomarres.com/downloads/regional.pdf#page=45.

Lavin, C. P., Jones, G. P., Williamson, D. H., and Harrison, H. B. (2021). Minimum size limits and the reproductive value of numerous, young, mature female fish. *Proc. R. Soc. B.* 288, 20202714. doi: 10.1098/rspb.2020.2714

Lejeusne, C., Chevaldonné, P., Pergent-Martini, C., Boudouresque, C. F., and Pérez, T. (2010). Climate change effects on a miniature ocean: the highly diverse, highly impacted Mediterranean Sea. *Trends Ecol. Evol.* 25, 250–260. doi: 10.1016/j.tree.2009.10.009

Link, J. S. (2021). Evidence of ecosystem overfishing in U.S. large marine ecosystems. *ICES J. Mar. Sci.* 78 (9), 3176–3201. doi: 10.1093/icesjms/fsab185

Lotze, H. K., Tittensor, D. P., Bryndum-Buchholz, A., Eddy, T. D., Cheung, W. W., Galbraith, E. D., et al. (2019). Global ensemble projections reveal trophic amplification of ocean biomass declines with climate change. *Proc. Natl. Acad. Sci.* 116, 12907–12912. doi: 10.1073/pnas.1900194116

Michael-Bitton, G., Gal, G., Corrales, X., Ofir, E., Shechter, M., and Zemah-Shamir, S. (2022). Economic aspects of fish stock accounting as a renewable marine natural capital: the Eastern Mediterranean continental shelf ecosystem as a case study. *Ecol. Econom.* 200, 107539. doi: 10.1016/j.ecolecon.2022.107539

Moullec, F., Lasram, F. B. R., Coll, M., Guilhaumon, F., Halouani, G., Hattab, T., et al. (2016). *Climate change impacts on marine resources: from individual to ecosystem responses* (France, IRD Éditions: IRD Éditions AllEnvi). Available at: https://hal.science/hal-02569960/.

Nillos Kleiven, P. J., Espeland, S. H., Olsen, E. M., Abesamis, R. A., Moland, E., and Kleiven, A. R. (2019). Fishing pressure impacts the abundance gradient of European lobsters across the borders of a newly established marine protected area. *Proc. R. Soc. B.* 286, 20182455. doi: 10.1098/rspb.2018.2455

NPA. (2015). *BioBlitz marine.* Tel Aviv: Nature and Parks Authority Available at: https://mafish.org.il/wp-content/uploads/2018/08/Marine-BioBlitz-Screen.pdf

NPA. (2020). *Fishing regulations in Israel (in Hebrew).* Tel Aviv: Nature and Parks Authority Available at: https://static.parks.org.il/wp-content/uploads/2021/01/%D7%A2%D7%99%D7%A7%D7%A8%D7%99-%D7%AA%D7%A7%D7%A0%D7%95%D7%AA-%D7%94%D7%93%D7%99%D7%92_%D7%93%D7%A6%D7%9E%D7%91%D7%A8-2020-%D7%A0%D7%92%D7%99%D7%A9.pdf.

Ofir, E., Silver, T., Steenbeek, J. G., Shachar, N., and Gal, G. (2022). *Applying the safe operating space (SOS) approach to sustainable commercial fishing under varying lake levels and littoral zone conditions* (U.S.A, Wiley Online Library: Fisheries). https://afspubs.onlinelibrary.wiley.com/doi/abs/10.1002/fsh.10869

Pagès, R., Baklouti, M., Barrier, N., Ayache, M., Sevault, F., Somot, S., et al. (2020). Projected effects of climate-induced changes in hydrodynamics on the biogeochemistry of the Mediterranean Sea under the RCP 8.5 regional climate scenario. *Front. Mar. Sci.* 7, 563615. doi: 10.3389/fmars.2020.563615

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. (1998). Fishing down marine food webs. *Science* 279, 860–863. doi: 10.1126/science.279.5352.860

Petza, D., Maina, I., Koukourouvli, N., Dimarchopoulou, D., Akrivos, D., Kavadas, S., et al. (2017). Where not to fish–reviewing and mapping fisheries restricted areas in the Aegean Sea. *Mediterr. Mar. Sci.* 18, 310–323. doi: 10.12681/mms.2081

Piroddi, C., Coll, M., Macias, D., Steenbeek, J., Garcia-Gorriz, E., Mannini, A., et al. (2022). Modelling the Mediterranean Sea ecosystem at high spatial resolution to inform the ecosystem-based management in the region. *Sci. Rep.* 12, 19680. doi: 10.1038/s41598-022-18017-x

Primo, C., Kelemen, F. D., Feldmann, H., Akhtar, N., and Ahrens, B. (2019). A regional atmosphere–ocean climate system model (CCLMv5. 0clm7-NEMOv3. 3-NEMOv3. 6) over Europe including three marginal seas: on its stability and performance. *Geosci. Model. Dev.* 12, 5077–5095. doi: 10.5194/gmd-12-5077-2019

Ramírez-Luna, V., and Chuenpagdee, R. (2019). "Exclusive fishing zone for small-scale fisheries in northern chocó, Colombia: pre- and post-implementation," in *Viability and sustainability of small-scale fisheries in Latin America and the Caribbean*. Eds. S. Salas, M. J. Barragán-Paladines and R. Chuenpagdee (Cham: Springer International Publishing), 197–217.

Reich, T., Ben-Ezra, T., Belkin, N., Tsemel, A., Aharonovich, D., Roth-Rosenberg, D., et al. (2022). A year in the life of the Eastern Mediterranean: monthly dynamics of phytoplankton and bacterioplankton in an ultra-oligotrophic sea. *Deep Sea Res. Part I.: Oceanographic Res. Papers* 182, 103720. doi: 10.1016/j.dsr.2022.103720

Ruti, P. M., Somot, S., Giorgi, F., Dubois, C., Flaounas, E., Obermann, A., et al. (2016). MED-CORDEX initiative for Mediterranean climate studies. *Bull. Am. Meteorol. Soc.* 97, 1187–1208. doi: 10.1175/BAMS-D-14-00176.1

Schuwirth, N., Borgwardt, F., Domisch, S., Friedrichs, M., Kattwinkel, M., Kneis, D., et al. (2019). How to make ecological models useful for environmental management. *Ecol. Model.* 411, 108784. doi: 10.1016/j.ecolmodel.2019.108784

Seker, Y. (2015). *Israel Marine plan - final report (Technion).* Haifa, Israel: The Technion Available at: https://msp-israel.net.technion.ac.il/en/stage-a-report/israel-marine-plan-final-plan/.

Serpetti, N., Baudron, A. R., Burrows, M., Payne, B. L., Helaouet, P., Fernandes, P. G., et al. (2017). Impact of ocean warming on sustainable fisheries management informs the ecosystem approach to fisheries. *Sci. Rep.* 7, 1–15. doi: 10.1038/s41598-017-13220-7

Sevault, F., Somot, S., Alias, A., Dubois, C., Lebeaupin-Brossier, C., Nabat, P., et al. (2014). A fully coupled Mediterranean regional climate system model: design and evaluation of the ocean component for the 1980–2012 period. *Tellus A.: Dynamic Meteorol. Oceanography* 66, 23967. doi: 10.3402/tellusa.v66.23967%40zela20.2016.68.issue-s2?needAccess=true&role=button

Shannon, L., Coll, M., Bundy, A., Gascuel, D., Heymans, J. J., Kleisner, K., et al. (2014). Trophic level-based indicators to track fishing impacts across marine ecosystems. *Mar. Ecol. Prog. Ser.* 512, 115–140. doi: 10.3354/meps10821

Shapiro Goldberg, D., Rilov, G., Villéger, S., and Belmaker, J. (2021). Predation cues lead to reduced foraging of invasive siganus rivulatus in the Mediterranean. *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.678848

Simmons, B. A., Nolte, C., and McGowan, J. (2021). Delivering on biden's 2030 conservation commitment. *bioRxiv* 2021–02. doi: 10.1101/2021.02.28.433244

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E., et al. (2014). *Climate change 2014: mitigation of climate change. contribution of working group III to the fifth assessment report of the intergovernmental panel on climate change.* 811–922. Switzerland, United Nations: IPCC Fifth Assessment Synthesis Report. Available at: https://reliefweb.int/report/world/climate-change-2014-mitigation-climate-change?gclid=Cj0KCQjwho-lBhC_ARIsAMpgMoe7d1KglleKAtpu_DjK8SzbqoeJ8YSQLcqqgsUFFfx1LL-pKBSEKBIaAudeEALw_wcB.

Soto-Navarro, J., Jordá, G., Amores, A., Cabos, W., Somot, S., Sevault, F., et al. (2020). Evolution of Mediterranean Sea water properties under climate change scenarios in the med-CORDEX ensemble. *Climate Dynamics* 54, 2135–2165. doi: 10.1007/s00382-019-05105-4

Stambler, N. (2014). *Sea the of Glory The Israel of Systems Aquatic the in Change and Stability (In Hebrew).* Available at: https://kotar.cet.ac.il/KotarApp/Viewer.aspx?nBookID=103249208#3.0.5.fitwidth.

Steenbeek, J. (2021). *Ecospace spatial-temporal data framework a brief users' guide. figshare* (Online resource) Spain, Ecopath International Initiative. Available at: https://figshare.com/articles/online_resource/Ecospace_spatial-temporal_data_framework_a_brief_users_guide/13551551.

Steenbeek, J., Buszowski, J., Chagaris, D., Christensen, V., Coll, M., Fulton, E. A., et al. (2021). Making spatial-temporal marine ecosystem modelling better – a perspective. *Environ. Model. Softw.* 145, 105209. doi: 10.1016/j.envsoft.2021.105209

Steenbeek, J., Coll, M., Gurney, L., Mélin, F., Hoepffner, N., Buszowski, J., et al. (2013). Bridging the gap between ecosystem modeling tools and geographic information systems: driving a food web model with external spatial–temporal data. *Ecol. Model.* 263, 139–151. doi: 10.1016/j.ecolmodel.2013.04.027

Steenbeek, J., Ortega, P., Bernardello, R., Christensen, V., Coll, M., Exarchou, E., et al. (In prep). *Making ecosystem modelling operational - using a novel distributed execution framework to systematically explore ecological responses to divergent climate trajectories* *in initialised earth system predictions. earth's future, special issue on past and future of marine ecosystems.*

Sterner, T. (2012). Gernot Wagner: but will the planet notice. *Environ. Resource Econom.* 51, 471–472. doi: 10.1007/s10640-011-9534-2

Stock, A., Murray, C. C., Gregr, E. J., Steenbeek, J., Woodburn, E., Micheli, F., et al. (2022). Exploring multiple stressor effects with ecopath, ecosim, and ecospace: research designs, modeling techniques, and future directions. *Sci. Total Environ.* 869, 1–20. Available at: https://www.sciencedirect.com/science/article/pii/S0048969723003340.

Susini, I., and Todd, V. L. G. (2021). Predictive capacity of Ecopath with Ecosim: Model performance and ecological indicators' response to imprecision. *Environ. Modelling & Software*, 143. doi: 10.1016/j.envsoft.2021.105098

Tittensor, D. P., Eddy, T. D., Lotze, H. K., Galbraith, E. D., Cheung, W., Barange, M., et al. (2018). A protocol for the intercomparison of marine fishery and ecosystem models: fish-MIP v1. 0. *Geosci. Model. Dev.* 11, 1421–1442. doi: 10.5194/gmd-11-1421-2018

Tittensor, D. P., Novaglio, C., Harrison, C. S., Heneghan, R. F., Barrier, N., Bianchi, D., et al. (2021). Next-generation ensemble projections reveal higher climate risks for marine ecosystems. *Nat. Climate Change* 11, 973–981. doi: 10.1038/s41558-021-01173-9

Tuel, A., and Eltahir, E. A. (2020). Why is the Mediterranean a climate change hot spot? *J. Climate* 33, 5829–5843. doi: 10.1175/JCLI-D-19-0910.1

van Leeuwen, S. M., Beecham, J. A., García-García, L. M., and Thorpe, R. (2022). The Mediterranean rhodes gyre: modelled impacts of climate change, acidification and fishing. *Mar. Ecol. Prog. Ser.* 690, 31–50. doi: 10.3354/meps14016

Vilas, D., Coll, M., Corrales, X., Steenbeek, J., Piroddi, C., Calò, A., et al. (2020). The effects of marine protected areas on ecosystem recovery and fisheries using a comparative modelling approach. *Aquat. Conservation: Mar. Freshw. Ecosyst.* 30, 1885–1901. doi: 10.1002/aqc.3368

Walters, C., Christensen, V., and Pauly, D. (1997). Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Rev. Fish. Biol. Fish.* 7, 139–172. doi: 10.1023/A:1018479526149

Walters, C., Pauly, D., and Christensen, V. (1999). Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. *Ecosystems* 2, 539–554. doi: 10.1007/s100219900101

Wright, P., Pinnegar, J., and Fox, C. (2020). Impacts of climate change on fish, relevant to the coastal and marine environment around the UK. *MCCIP Sci. Rev.*, 354–381. doi: 10.14465/2020.arc16.fsh

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EDITED BY
Periklis Kleitou,
Marine and Environmental Research Lab (MER), Cyprus

REVIEWED BY
Othman Jarboui,
Institut National des Sciences et Technologies de la Mer, Tunisia
Stefano Lelli,
General Fisheries Commission for the Mediterranean (GFCM), Italy
Holden Earl Harris,
University of Miami, United States

*CORRESPONDENCE
Athanassios C. Tsikliras
atsik@bio.auth.gr

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Scientific knowledge gaps on the biology of non-fish marine species across European Seas

Luisa R. Abucay[1], Patricia Sorongon-Yap[1], Kathleen Kesner-Reyes[1], Emily C. Capuli[1], Rodolfo B. Reyes Jr[1], Eva Daskalaki[2], Carmen Ferrà[3,4], Giuseppe Scarcella[3], Gianpaolo Coro[5], Francesc Ordines[6], Paula Sánchez-Zulueta[6], Georgi Dakalov[7], Stefania Klayn[7], Liesa Celie[8], Marco Scotti[9,10], David Grémillet[11,12], Charlotte Lambert[13], Gideon Gal[14], Maria Lourdes D. Palomares[15,16], Donna Dimarchopoulou[17,18] and Athanassios C. Tsikliras[2]*

[1]Quantitative Aquatics, Los Banos, Philippines, [2]Laboratory of Ichthyology, School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece, [3]National Research Council - Institute for Biological Resources and Marine Biotechnologies, Ancona, Italy, [4]Department of Biological, Geological and Environmental Sciences (BiGeA), University of Bologna, Bologna, Italy, [5]National Research Council - Institute of Information Science and Technologies "Alessandro Faedo", Pisa, Italy, [6]Centre Oceanogràfic de les Balears (COB-IEO), CSIC, Palma, Spain, [7]Institute of Biodiversity and Ecosystem Research, Bulgarian Academy of Sciences, Sofia, Bulgaria, [8]Flanders Research Institute for Agriculture, Fisheries and Food, Oostende, Belgium, [9]GEOMAR Helmholtz Centre for Ocean Research, Kiel, Germany, [10]Institute of Biosciences and Bioresources, National Research Council of Italy, Sesto Fiorentino, Italy, [11]Centre d'Ecologie Fonctionnelle et Evolutive (CEFE), Univ Montpellier, Centre National de la Recherche Scientifique (CNRS), Ecole Pratique des Hautes Etudes (EPHE), Institut de Recherche pour le Développement (IRD), Montpellier, France, [12]FitzPatrick Institute, Department of Science and Technology (DST)/National Research Foundation (NRF) Excellence Centre at the University of Cape Town, Cape Town, South Africa, [13]Littoral ENvironnement et Société UMR 7266 CNRS-La Rochelle Université, La Rochelle, France, [14]Kinneret Limnological Laboratory, Israel Oceanographic and Limnological Research, Migdal, Israel, [15]Institute for the Oceans and Fisheries, The University of British Columbia, Vancouver, BC, Canada, [16]Sea Around Us, University of British Columbia, Vancouver, BC, Canada, [17]Department of Biology, Dalhousie University, Halifax, NS, Canada, [18]Department of Biology, Woods Hole Oceanographic Institution, Woods Hole, MA, United States

Available information and potential data gaps for non-fish marine organisms (cnidarians, crustaceans, echinoderms, molluscs, sponges, mammals, reptiles, and seabirds) covered by the global database SeaLifeBase were reviewed for eight marine ecosystems (Adriatic Sea, Aegean Sea, Baltic Sea, *Bay of Biscay/Celtic Sea/Iberian Coast*, Black Sea, North Sea, western Mediterranean Sea, Levantine Sea) across European Seas. The review of the SeaLifeBase dataset, which is based on published literature, analyzed information coverage for eight biological characteristics (diet, fecundity, maturity, length-weight relationships, spawning, growth, lifespan, and natural mortality). These characteristics are required for the development of ecosystem and ecological models to evaluate the status of marine resources and related fisheries. Our analyses revealed that information regarding these biological characteristics in the literature was far from complete across all studied areas. The level of available information was nonetheless reasonably good for sea turtles and moderate for marine mammals in some areas (Baltic Sea, *Bay of Biscay/Celtic Sea/Iberian Coast*, Black Sea, North Sea and western Mediterranean Sea). Further, seven of the areas have well-studied species in terms of information coverage for biological characteristics of some commercial species whereas threatened species are generally not well studied. Across areas, the most well-studied species are the cephalopod common cuttlefish (*Sepia officinalis)* and the crustacean Norway lobster (*Nephrops norvegicus*). Overall, the information gap is narrowest for length-weight relationships followed by growth and maturity, and widest for fecundity and natural mortality. Based on these insights, we provide recommendations to prioritize species with insufficient or missing biological data that are common across the studied marine ecosystems and to address data deficiencies.

KEYWORDS

data gaps, marine biodiversity, marine mammals, seabirds, marine reptiles, marine invertebrates, SeaLifeBase, European waters

Introduction

Ecocentric (=ecosystem centered) fisheries management requires detailed knowledge of the structure and functioning of the marine ecosystems, from abiotic data to the status of all ecosystem components (Dimarchopoulou, 2020). This includes the fishing pressure applied on commercial and non-commercial marine populations and their respective biomasses (Tsikliras et al., 2023) and biological information (growth, maturity, spawning, fecundity, mortality, lifespan and diet) of all organisms in an ecosystem because marine organisms respond differently to fishing pressure and population time to recovery depends upon their life-history strategy and ecological traits (Dimarchopoulou et al., 2017). Ecosystem structure is usually described using mass balance ecosystem models (Heymans et al., 2020) while the stock status is derived from age-based or surplus production stock assessments (Tsikliras and Froese, 2019). Ecosystem models and stock assessments are thereby required to examine fisheries management and marine policy scenarios (Piroddi et al., 2022) within the context of environmental (Piroddi et al., 2021), oceanographic (Coll et al., 2019) and climatic change factors (Corrales et al., 2018), whilst also incorporating economic and social parameters (Link, 2010).

The development of ecosystem models demands specific biological data, mainly growth parameters, natural mortality and diet composition per species or functional group of species (Christensen and Walters, 2004), as well as catch data that are available per fleet through official landings statistics (global and regional databases of the Food and Agricultural Organisation of the United Nations: FAO, 2020) and catch datasets (Sea Around Us: Pauly and Zeller, 2016). Similarly, the simpler age-based stock assessments require growth parameters, size at maturity, spawning and natural mortality data (Jardim et al., 2015) while some surplus production models use the maximum intrinsic population growth that is based on several biological characteristics including growth, fecundity, maturity and natural mortality (Froese et al., 2018a; Froese et al., 2018b; Froese et al., 2020).

Among European Seas, the North East (NE) Atlantic Ocean is a marine ecosystem with a long scientific history of investigations across all marine science disciplines (Lotze and Worm, 2009). As a result, long time-series of biological, oceanographic and fisheries data exist, most of which are publicly available. These datasets have supported many ecosystem models (Keramidas et al., 2023) and the official full stock assessments in most marine ecosystems of the NE Atlantic (ICES, 2022). In contrast, the Mediterranean and the Black Seas, despite their longer history of fisheries exploitation (Stergiou et al., 2016) and the early scientific work on biology and fisheries by Aristotle and Oppian, respectively (Deacon, 1997), lack long time series of biological, oceanographic and fisheries data (Fortibuoni et al., 2017, but see Ravier and Fromentin, 2004). Consequently, data-limitations have constrained ecosystem models to specific and well-studied areas of the northern and eastern Mediterranean coastline that are well studied (Adriatic Sea: Barausse et al., 2009; Libralato et al., 2015; Catalan coast: Coll et al., 2008; Coll et al., 2009; Aegean Sea: Dimarchopoulou et al., 2019; Dimarchopoulou et al., 2022; Keramidas et al., 2022; Levantine Sea: Corrales et al., 2017; Shabtay et al., 2018; Corrales et al., 2019; Ofir et al., 2023). This also restricted full analytical stock assessments to a proportionally low number of exploited stocks (Piroddi et al., 2020) despite the increasing efforts of the Expert Working Groups (EWG) of the Scientific, Technical and Economic Committee for Fisheries (STECF) of the European Union and the General Fisheries Commission for the Mediterranean Sea (GFCM) of the FAO. However, the most important issues in Mediterranean fisheries are the north-south gradient in marine research and data, with more scientific output along the northern Mediterranean coastline (Stergiou and Tsikliras, 2006), and that valuable datasets are not openly available (McManamay and Utz, 2014). This is because some people/institutions (and countries, which were historically amongst the most scientifically advanced) do not believe in open science – a policy priority for the European Commission – even when supported by public funds (Damalas et al., 2018).

Robust and adaptive fisheries management policies require understanding their key sources of uncertainty, such as knowledge gaps in biology of marine species (Link et al., 2012). A recent update on the gaps in the biological knowledge of Mediterranean marine fishes (Daskalaki et al., 2022) indicated that efforts were made to reduce these gaps in knowledge across the Mediterranean Sea compared to previous records (Dimarchopoulou et al., 2017). This is especially true for threatened species fishes such as sharks and rays (Tsikliras and Dimarchopoulou, 2021) as well as for alien species that rapidly colonized the Mediterranean during the last decades (Katsanevakis et al., 2014). Filling the gaps in ecological and biological knowledge and assessing anthropogenic impacts marine ecosystems are prerequisites for developing robust ecosystem models (Heymans et al., 2020) hence for promoting effective ecocentric management (Claudet et al., 2019).

The principal aim of the present work was to review available information on key biological characteristics (diet, fecundity, maturity, length-weight relationships, spawning, growth, lifespan, and natural mortality) of non-fish marine species across European Seas. This allowed a gap analysis and a comparison of the availability of biological data across areas and taxonomic groups, leading to recommendations to reduce knowledge gaps (if and where required). Thus, future research will have a baseline to prioritize species of special interest based on specific criteria such as conservation status. This review covers the non-fish marine organisms belonging to eight taxonomic groups occurring in the European Seas. Fish species will be covered in a separate publication that will follow the same methodology and spatial coverage and will expand the review of the Mediterranean marine fishes (Daskalaki et al., 2022) to European Seas.

Materials and methods

This review was based on information that was extracted from the literature and captured in SeaLifeBase (www.sealifebase.org; Palomares and Pauly, 2021, consulted in December 2021) for eight marine ecosystems (Adriatic Sea, Aegean Sea, Baltic Sea, *Bay of Biscay/Celtic Sea/Iberian Coast*, Black Sea, Levantine Sea, North Sea, and western Mediterranean Sea) (Figure 1). SeaLifeBase is a global biodiversity information system on non-fishes that covers a wide range of information on taxonomy, biology, trophic ecology, life history and uses (Palomares and Pauly, 2021). The extensiveness of information in the database has catered to a diversity of stakeholders (scientists, researchers, policy-makers, fisheries managers, donors, conservationists, teachers, and students) for various applications targeting sustainable fisheries management (Froese et al., 2018a), ecosystem modelling (Grüss et al., 2019), biodiversity conservation (Stasolla et al., 2021) and environmental protection (Jâms et al., 2020).

Eight major groups of marine non-fish species were considered, including cnidarians (corals, jellyfishes, other cnidarians like hydrozoans, hydroids, anemones, and sea pens), crustaceans (decapods, other malacostraca like shrimps, amphipods, isopods, copepods and ostracods), echinoderms (sea cucumbers, sea urchins, starfishes, brittle stars, crinoids, basket stars), molluscs (bivalves, cephalopods, gastropods, chitons, solenogasters and tusk shells), sponges, mammals (dolphins, whales, seals), reptiles (sea turtles), and seabirds.

The working species lists for the review were drawn from a combination of ecosystem, country, and FAO area assignments in SeaLifeBase that approximate the areas covering each of the eight study areas (Palomares and Pauly, 2021). Thus, the species lists for the Adriatic Sea and the Aegean Sea come from the SeaLifeBase faunal records under the Adriatic Sea and Aegean Sea marine ecoregions, whereas those for the Baltic Sea, Black Sea, and North Sea come from the faunal records under the Large Marine Ecosystem (LME) units of the same name (Figure 1). The species list for the western Mediterranean Sea area combines faunal records for the Balearic Islands, Tyrrhenian Sea, Sardinia Island, Corsica Island, as well as marine records for the Mediterranean coasts of Spain and France, i.e., excluding the southern Mediterranean coastline (Figure 1). The list for the Levantine Sea consolidates records from the Levantine Sea ecosystem, marine records for Cyprus, Syria, Lebanon as well as from the side of Israel, Egypt, and Turkey (excluding the Aegean Sea) in FAO area 37. The *Bay of Biscay/Celtic Sea/Iberian Coast* combines faunal lists for two LMEs, namely, Celtic-Biscay Shelf and Iberian Coast, and thus includes species in an area that extends from the Gulf of Cadiz and north to the Outer Hebrides (Figure 1). An assessment of the representativeness of the species coverage in each site, however, was not within the scope of this work.

The review of biological information covered in SeaLifeBase largely follows the approach of two recent gap analyses reviews on the biology of fishes in the Mediterranean Sea (Dimarchopoulou et al., 2017; Daskalaki et al., 2022).

Eight categories of biological characteristics were examined and include corresponding records in SeaLifeBase: Diet (D) covered diet composition, prey items, and feeding preferences; fecundity (F) included absolute and relative number of oocytes produced per

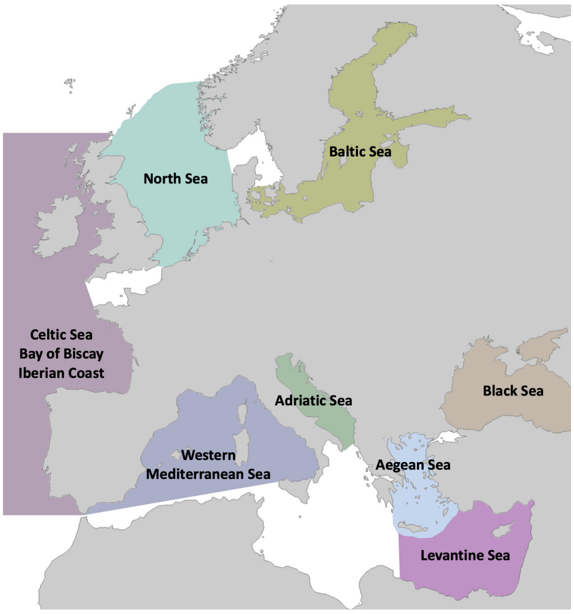

FIGURE 1
Map of the marine ecosystems that were reviewed.

female; maturity covered length/size at first maturity (L_m); spawning (S) looked at onset and duration of spawning (i.e., spawning period); mortality (M) considered the rate of natural mortality regardless of the estimation method; life span (t_{max}) covered maximum age; growth (G) refers to the growth parameters asymptotic length (L_∞) and the rate at which it is approached (K), while length-weight relationships (LWR) considered the slope and intercept of the LWR function (Dimarchopoulou et al., 2017; Daskalaki et al., 2022). Gaps arise from the difference between the level of current knowledge and that of desired knowledge. For the present review, well-studied species were identified as those with available information for at least six out of eight of the biological characteristics described above, and which have 30 or more records available in SeaLifeBase. The desired knowledge for an area was defined as the area with at least half of the reported species being well-studied. The least-studied species were those that do not meet the above criteria (Table 1). Overall, the number of unique references for data on biological characteristics from the literature captured in SeaLifeBase total 637 records (Palomares and Pauly, 2021).

Species were categorized as threatened/non-threatened according to the species conservation status in SeaLifeBase which follows the IUCN Red List of Threatened Species (version 2021-1) considering the global classification of species (EX: Extinct; EW: Extinct in the Wild; CR: Critically Endangered; EN: Endangered; VU: Vulnerable; NT: Near Threatened; LC: Least Concern; DD: Data deficient; NE: Not Evaluated). For this review, species categorized as threatened included only those flagged as Critically Endangered (CR), Endangered (EN), Vulnerable (VU) and Near-Threatened (NT).

This review assesses how much information on biological traits is available for species, examining species with and without biological information, identifying where data gaps are smallest and widest, and identifying the most and least-studied species. Where a species list is short, full details for the species are included in the table, otherwise the list is summarized according to Order/Family and species count. An overall assessment of information that follows a basic traffic light classification of Good, Moderate and Poor information coverage and the criteria for each category are also provided (Table 1).

Recommendations for filling data gaps are provided in two levels. The first consists of specific recommendations on species and aims at addressing deficiencies or missing information on biological characteristics. The second focuses on filling the gaps with respect to biological characteristics, and particularly on prioritizing species with insufficient or missing biological data that are common to most areas.

Results

Adriatic Sea

A total of 359 non-fish species were recorded for the Adriatic Sea in SeaLifeBase, including 349 native, five endemic and five introduced species that belong to 168 Families, 68 Orders and 15 Classes. Regarding the number of biological characteristics studied, there is no information for 248 species (69%). There are 48 species (13%) with information for only one characteristic (mostly on length-weight relationships), while two species (1%) have studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the Adriatic Sea, from most-studied to least-studied, are: length-weight relationships (97 species, 27%), followed by growth (52 species, 14%), size at maturity (33 species, 9%), spawning (22 species, 6%), maximum age (22 species, 6%), natural mortality (16 species, 4%), fecundity (11 species, 3%), and diet (6 species, 2%) (Figure 3).

Five species are included in the IUCN Red List and are listed under the categories CR [Noble pen shell (*Pinna nobilis*)], VU [Leatherback turtle (*Dermochelys coriacea*), Common spiny lobster (*Palinurus elephas*), Horned grebe (*Podiceps auritus*)], and NT [Dalmatian pelican (*Pelecanus crispus*)]. These five species have a relatively small gap for one biological characteristic, growth (3 species, 60% studied). This is followed by larger gaps regarding six biological characteristics: length-weight (2 species, 40% studied), maturity (2 species, 40% studied), lifespan (2 species, 40% studied), fecundity (1 species, 20% studied), spawning (1 species, 20% studied), and natural mortality (1 species, 20% studied). The largest data gap is for diet, where no information was available for any of the threatened species (Figure 3).

The most-studied species of the Adriatic Sea make up about 2% (6 species) of non-fish species reported from the area. These species belong to two Classes and six Families (Table 2). The six most-studied Adriatic species in terms of biological characteristics are the Blue crab (*Callinectes sapidus*), which is an introduced species, and Common cuttlefish (*Sepia officinalis*), each with available information on eight biological characteristics. Deep-water rose shrimp (*Parapenaeus longirostris*), Giant red shrimp (*Aristaeomorpha foliacea*), Spottail mantis shrimp (*Squilla mantis*) and Common spiny lobster (*Palinurus elephas*) have seven studied biological characteristics. Out of these six species, only the Common spiny lobster is included in the IUCN Red List as VU (Table 2).

The least-studied species make up about 98% (353 species) of non-fish species reported in the Adriatic Sea (15 Classes and 164 Families) (Table S1) including four IUCN Red List species (Table 2).

TABLE 1 Criteria for comparing the knowledge level of areas based on the number of studied biological characteristics and the available number of records for each characteristic.

Literature coverage	Good	Moderate	Poor
Biological characteristics	At least 50% of species have data for 6 to 8 characteristics	At least 50% of species have data for 3 to 8 characteristics	More than 50% of species only have data for 2 or less characteristics, or no data at all

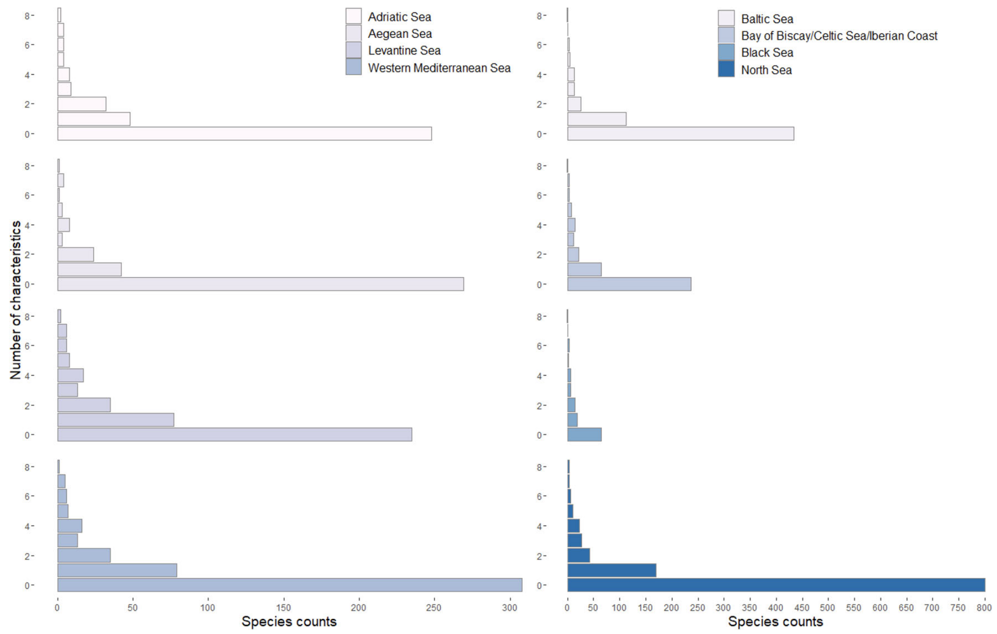

FIGURE 2

Species counts by number of biological characteristics (0 to 8) studied for non-fish marine organisms in the Adriatic Sea.

Aegean Sea

A total of 355 non-fish species were recorded for the Aegean Sea in SeaLifeBase, including 347 native, six endemic and two introduced species, and belong to 166 Families, 61 Orders and 15 Classes. Regarding the number of biological characteristics studied, there is no information for 269 species (76%). There are 42 species (12%) with information for only one characteristic (mostly length-weight relationships), while for one species all eight biological characteristics are available (Figure 2).

The individual biological characteristics of the non-fish species of the Aegean Sea, from most-studied to least-studied, are length-weight relationships (77 species, 22%), growth (39 species, 11%), maturity (19 species, 5%), spawning (17 species, 5%), lifespan (13 species, 4%), natural mortality (11 species, 3%), fecundity (7 species, 2%), and diet (5 species, 1%) (Figure 2).

Eight species are threatened and listed under the IUCN Red List categories as endangered (EN) and vulnerable (VU), and have large gaps regarding all biological characteristics: length-weight (2 species, 25%), maturity (2 species, 25%), growth (2 species, 25%), fecundity (1 species, 13%), diet (1 species, 13%), spawning (1 species, 13%), natural mortality (1 species, 13%) and lifespan (1 species, 13%) (Figure 3).

The most-studied species of the Aegean Sea make up about 1% (5 species) of non-fish species reported from the area and cover two Classes and five Families (Table 3). The five most studied Aegean species in terms of biological characteristics are the Common cuttlefish (*Sepia officinalis*) with available information on eight biological characteristics, as well as the Deep-water rose shrimp (*Parapenaeus longirostris*), Giant red shrimp (*Aristaeomorpha foliacea*), Spottail mantis shrimp (*Squilla mantis*) and Common spiny lobster (*Palinurus elephas*) that have seven studied biological characteristics. Of these, only the Common spiny lobster is included in the IUCN Red List as VU (Table 3).

The least-studied species make up about 99% (350 species) of the non-fish species reported in the Aegean Sea, covering 15 Classes and 162 Families (Table S2). Seven of the least-studied species are included in the IUCN Red List. These include the Mediterranean monk seal (*Monachus monachus*), listed as EN, with available information on one biological characteristic, the Leatherback turtle (*Dermochelys coriacea*), listed as VU, with three studied biological characteristics, as well as the coral (*Crassophyllum thessalonicae*) and the sea anemone (*Paranemonia vouliagmeniensis*) listed as CR. The Pink sea fan (*Eunicella verrucosa*), Horned grebe (*Podiceps auritus*), and, Levantine shearwater (*Puffinus yelkouan*) are listed as VU, and have no available biological information (Table 3).

Baltic Sea

A total of 606 non-fish species were retrieved for the Baltic Sea from SeaLifeBase, including 595 native and 11 introduced species. The species of the area belong to 263 Families, 75 Orders and 23 Classes. Regarding the number of biological characteristics studied, there is no information for 434 species (72%). There are 112 species (18%) with information on one biological characteristic (mostly on length-weight relationships) while one species has studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the Baltic Sea, from most-studied to least-studied, are length-weight relationships (140 species, 23%), growth (46 species, 8%), diet (29 species, 5%), lifespan (29 species, 5%), maturity (24 species,

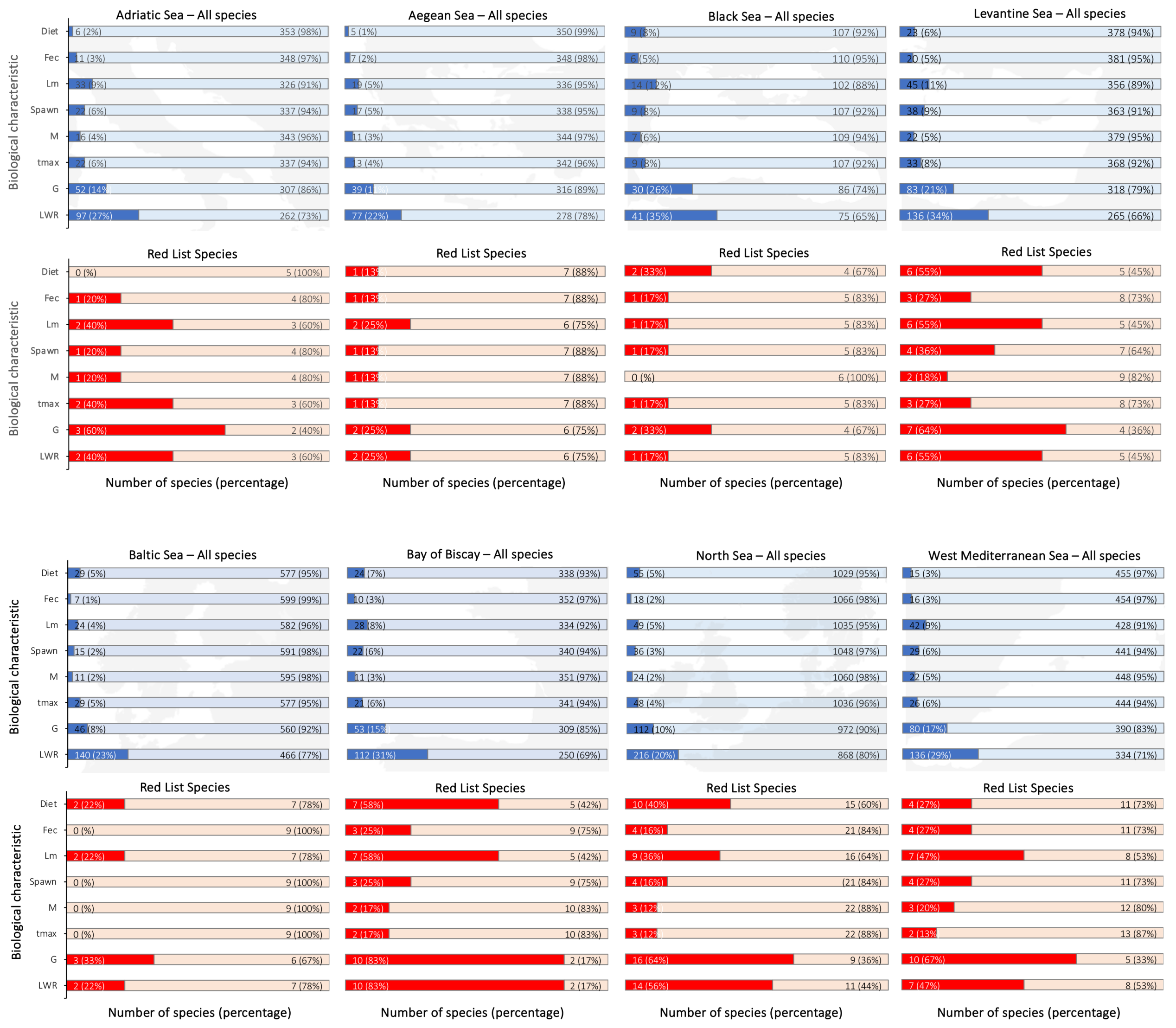

FIGURE 3
(top) Percentage of non-fish species in the Adriatic Sea, Aegean Sea, Black Sea, Levantine Sea Baltic Sea, *Bay of Biscay/Celtic Sea/Iberian Coast, North Sea and Western Mediterranean Sea* with (dark color) and without (light color) information on biological characteristics: feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR), (bottom) the same percentages calculated for species under IUCN Red List categories near threatened (NT), vulnerable (VU), endangered (EN) and critically endangered (CR).

4%), spawning (15 species, 2%), natural mortality (11 species, 2%) and fecundity (7 species, 1%) (Figure 3). Out of all 606 non-fish species reported from the Baltic Sea there are no species that could be considered as well-studied (Table S3). All the Baltic species range from having none to moderately sufficient information on their biological characteristics.

Eight species are listed in IUCN Red List, and have large gaps regarding four biological characteristics: growth (3 species, 33% studied), length-weight relationships (2 species, 22% studied), maturity, (2 species, 22% studied) and diet (2 species, 22% studied). Species listed as VU are Leatherback turtle (*Dermochelys coriacea*), Sperm whale (*Physeter macrocephalus*), Long-tailed duck (*Clangula hyemalis*), Velvet scoter (*Melanitta fusca*), Horned grebe (*Podiceps auritus*), and Steller's eider (*Polysticta stelleri*)] all of which have no available biological information. Those listed as NT include Eurasian river otter (*Lutra lutra*) with information on one biological characteristic, and Common eider (*Somateria mollissima*) with no biological information available (Table 4). There is no available record on these species regarding fecundity, spawning, natural mortality, and lifespan (Figure 3).

Bay of Biscay/Celtic Sea/Iberian Coast

A total of 362 non-fish species from the *Bay of Biscay/Celtic Sea/Iberian Coast* have records in SeaLifeBase, including 356 native and six introduced species. The species within this area belong to 206 Families, 78 Orders and 22 Classes. Regarding the number of biological characteristics studied, there is no information for 236 species (65%), whereas there are 65 species (18%) with information for only one

TABLE 2 List of the most- and least-studied non-fish species in the Adriatic Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec.) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Most-studied								
Malacostraca	Portunidae	*Callinectes sapidus*	Blue crab	introduced	NE	8/8	68	2 Diet, 1 Fec, 13 L_m, 2 Spawn, 5 M, 1 t_{max}, 19 G, 25 LWR
Cephalopoda	Sepiidae	*Sepia officinalis*	Common cuttlefish	native	LC	8/8	30	2 Diet, 1 Fec, 4 L_m, 1 Spawn, 3 M, 1 t_{max}, 8 G, 10 LWR
Malacostraca	Penaeidae	*Parapenaeus longirostris*	Deep-water rose shrimp	native	NE	7/8	60	1 Fec, 6 L_m, 1 Spawn, 3 M, 1 t_{max}, 36 G, 12 LWR
Malacostraca	Aristeidae	*Aristaeomorpha foliacea*	Giant red shrimp	native	NE	7/8	59	2 Diet, 8 L_m, 1 Spawn, 8 M, 2 t_{max}, 28 G, 10 LWR
Malacostraca	Squillidae	*Squilla mantis*	Spottail mantis shrimp	native	NE	7/8	40	1 Fec, 2 L_m, 3 Spawn, 2 M, 1 t_{max}, 8 G, 23 LWR
Malacostraca	Palinuridae	*Palinurus elephas*	Common spiny lobster	native	VU	7/8	32	1 Fec, 7 L_m, 1 Spawn, 2 M, 1 t_{max}, 12 G, 8 LWR
Least-studied								
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Bivalvia	Pinnidae	*Pinna nobilis*	Noble pen shell	native	CR	2/8	12	6 t_{max}, 6 G
Aves	Pelecanidae	*Pelecanus crispus*	Dalmatian pelican	native	NT	0/8	0	–
Aves	Podicipedidae	*Podiceps auritus*	Horned grebe	native	VU	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; NE, not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

characteristic (mostly on length-weight relationships), and one species has studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the *Bay of Biscay/Celtic Sea/Iberian Coast*, from most studied to least studied, are length-weight relationships (112 species, 31%), growth (53 species, 15%), maturity (28 species, 8%), diet (24 species, 7%), spawning (22 species, 6%), lifespan (21 species, 6%), natural mortality (11 species, 3%) and fecundity (10 species, 3%) (Figure 3).

Ten species listed under the IUCN Red List have relatively smaller gaps regarding four biological characteristics: length-weight (10 species, 83%), growth (10 species, 83%), diet (7 species, 58%), and maturity (7 species, 58%). Larger gaps were observed in fecundity (3 species, 25%), spawning (3 species, 25%), lifespan (2 species, 17%) and natural mortality (2 species, 17%) (Figure 3). These species are Hawksbill turtle (*Eretmochelys imbricata*; CR), Kemp's ridley turtle (*Lepidochelys kempii*; CR), North Atlantic right whale (*Eubalaena glacialis*; CR), Sei whale (*Balaenoptera borealis*; EN), Blue whale (*Balaenoptera musculus*; EN), Loggerhead turtle (*Caretta*; VU), Leatherback turtle (*Dermochelys coriacea*; VU), Fin whale (*Balaenoptera physalus*; VU), Hooded seal (*Cystophora cristata*; VU), and Cape Verde petrel (*Pterodroma feae*; NT).

The most studied non-fish species in the *Bay of Biscay/Celtic Sea/Iberian Coast* make up about 1% (4 species) and cover three Classes and four Families (Table 5). The four most studied species from the area in terms of biological characteristics were the Common cuttlefish (*Sepia officinalis*) with information on eight biological characteristics, and the Green sea turtle (*Chelonia mydas*), Giant red shrimp (*Aristaeomorpha foliacea*) and Spottail mantis shrimp (*Squilla mantis*) which have seven studied biological characteristics. Out of these species, only the Green sea turtle is included in the endangered list of IUCN (Table 5).

The least studied species make up about 99% (358 species) of non-fish species reported in the *Bay of Biscay/Celtic Sea/Iberian Coast*, covering 22 Classes and 205 Families (Table S4) including the ten species of the IUCN Red List (Table 5).

Black Sea

A total of 97 non-fish species recorded from the Black Sea in SeaLifeBase, including seven introduced species. The species of the area belong to 57 Families, 40 Orders and 13 Classes. Regarding the number of biological characteristics studied, there is no information for 58 species (56%). There are 19 species (16%) with information on one biological characteristic (mostly on length-weight relationships), whereas one species (1%) has studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the Black Sea, from most studied to least studied, are length-weight relationships (41 species, 35%), growth (30 species, 26%),

TABLE 3 List of the most- and least-studied non-fish species in the Aegean Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec	No. of records per characteristic
Most-studied								
Cephalopoda	Sepiidae	*Sepia officinalis*	Common cuttlefish	native	LC	8/8	30	2 Diet, 1 Fec, 4 L_m, 1 Spawn, 3 M, 1 t_{max}, 8 G, 10 LWR
Malacostraca	Penaeidae	*Parapenaeus longirostris*	Deep-water rose shrimp	native	NE	7/8	60	1 Fec, 6 L_m, 1 Spawn, 3 M, 1 t_{max}, 36 G, 12 LWR
Malacostraca	Aristeidae	*Aristaeomorpha foliacea*	Giant red shrimp	native	NE	7/8	59	2 Diet, 8 L_m, 1 Spawn, 8 M, 2 t_{max}, 28 G, 10 LWR
Malacostraca	Squillidae	*Squilla mantis*	Spottail mantis shrimp	native	NE	7/8	40	1 Fec, 2 L_m, 3 Spawn, 2 M, 1 t_{max}, 8 G, 23 LWR
Malacostraca	Palinuridae	*Palinurus elephas*	Common spiny lobster	native	VU	7/8	32	1 Fec, 7 L_m, 1 Spawn, 2 M, 1 t_{max}, 12 G, 8 LWR
Least-studied								
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Mammalia	Phocidae	*Monachus monachus*	Mediterranean monk seal	native	EN	1/8	1	1 Diet
Anthozoa	Pennatulidae	*Crassophyllum thessalonicae*		native	EN	0/8	0	–
Anthozoa	Gorgoniidae	*Eunicella verrucosa*	Pink sea fan	native	VU	0/8	0	–
Anthozoa	Actiniidae	*Paranemonia vouliagmeniensis*		native	EN	0/8	0	–
Aves	Podicipedidae	*Podiceps auritus*	Horned grebe	native	VU	0/8	0	–
Aves	Procellariidae	*Puffinus yelkouan*	Levantine shearwater	native	VU	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; NE, not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

maturity (14 species, 12%), spawning (9 species, 8%), lifespan (9 species, 8%), diet (9 species, 8%), natural mortality (7 species, 6%), and fecundity (6 species, 5%) (Figure 3).

Five species are listed under the categories CR, endangered EN and vulnerable VU of the IUCN Red List and have large gaps regarding seven biological characteristics: diet (2 species, 33%), growth (2 species, 33%), fecundity (1 species, 17%), maturity (1 species, 17%), spawning (1 species, 17%), lifespan (1 species, 17%) and length-weight relationships (1 species, 17%). The widest information gap refers to natural mortality, where no biological information is available (Figure 3).

The only well-studied non-fish species in the Black Sea is the Blue crab (*Callinectes sapidus*), an introduced species, having information for all eight biological characteristics (Table 6). The least studied species make up about 96% (93 species) of non-fish species reported in the Black Sea, covering 13 Classes and 57 Families (Table S5). The least-studied species reported from the Black Sea are 28 species in total, covering seven Classes and 23 Families belonging to five taxonomic groups (Table 6). Amongst these species, the Mediterranean monk seal (*Monachus monachus*) with one biological characteristic and the Loggerhead turtle (*Caretta caretta*), with six biological characteristics, are listed as endangered (EN) and vulnerable (VU), respectively (Table 6). The former is considered extinct in the Black Sea and the latter is reported in occasional sightings, without an established population. Three species that are listed as VU [Velvet scoter (*Melanitta fusca*), Horned grebe (*Podiceps auritus*), and the Levantine shearwater (*Puffinus yelkouan*)], have no available biological information (Table 6).

Levantine Sea

A total of 401 non-fish species recorded from the Levantine Sea are found in SeaLifeBase, including 388 native, one endemic and 12 introduced species that belong to 172 Families, 60 Orders and 17 Classes. Regarding the number of biological characteristics studied, there is no information for 235 species (59%). There are 77 species

TABLE 4 List of the most- and least-studied non-fish species in the Baltic Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Most-studied								
-								
Least-studied								
Mammalia	Physeteridae	*Physeter macrocephalus*	Sperm whale	native	VU	4/8	34	15 Diet, 2 L_m, 3 G, 14 LWR
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Mammalia	Mustelidae	*Lutra lutra*	Eurasian river otter	native	NT	1/8	1	1 Diet
Aves	Anatidae	*Clangula hyemalis*	Long-tailed duck	native	VU	0/8	0	–
Aves	Anatidae	*Melanitta fusca*	Velvet scoter	native	VU	0/8	0	–
Aves	Podicipedidae	*Podiceps auritus*	Horned grebe	native	VU	0/8	0	–
Aves	Anatidae	*Polysticta stelleri*	Steller's eider	native	VU	0/8	0	–
Aves	Anatidae	*Somateria mollissima*	Common eider	native	NT	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; N.E., not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

(19%) with information on one biological characteristic (mostly on length-weight relationships), while two species have studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the Levantine Sea, from most studied to least studied, are length-weight relationships (136 species, 34%), growth (83 species, 21%), maturity (45 species, 11%), spawning (38 species, 9%), lifespan (33 species, 8%), diet (23 species, 6%), natural mortality (22 species, 5%) and fecundity (20 species, 5%) (Figure 3).

Eight species are listed under the IUCN Red List categories near threatened (NT), vulnerable (VU), endangered (EN) and critically endangered (CR) and have smaller gaps regarding four biological characteristics: growth (7 species, 87%), diet (6 species, 75%), maturity (6 species, 75%) and length-weight relationships (6 species, 75%). Larger gaps were observed in spawning (4 species, 50%), fecundity (3 species, 33%), lifespan (3 species, 33%) and natural mortality (2 species, 22%) (Figure 3). These species are the Mediterranean monk seal (*Monachus monachus*; EN), Loggerhead turtle (*Caretta caretta*; VU), Sperm whale (*Physeter macrocephalus*; VU), Leatherback turtle (*Dermochelys coriacea*; VU), Levantine shearwater (*Puffinus yelkouan*; VU), Armenian gull (*Larus armenicus*; NT), False killer whale (*Pseudorca crassidens*; NT), and Noble pen shell (*Pinna nobilis*; CR).

The well-studied species of the Levantine Sea make up about 2% (8 species) of non-fish species reported from the area and cover three Classes and seven Families (Table 7). The eight most studied Levantine species in terms of biological characteristics are: the alien Blue crab (*Callinectes sapidus*) and Common cuttlefish (*Sepia officinalis*), both having eight biological characteristics, the Speckled shrimp (*Metapenaeus monoceros*), which is another introduced species, Green sea turtle (*Chelonia mydas*), Deep-water rose shrimp (*Parapenaeus longirostris*), Giant red shrimp (*Aristaeomorpha foliacea*), Spottail mantis shrimp (*Squilla mantis*) and Common spiny lobster (*Palinurus elephas*), all of which have seven biological characteristics. Out of these eight species only two are included in the IUCN Red List, the Green sea turtle as EN and the Common spiny lobster as VU (Table 7).

The least studied species make up about 98% (393 species) of non-fish species reported in the Levantine Sea, covering 17 Classes and 170 Families (Table S6). Including the eight species reported in the IUCN Red List (Table 7).

North Sea

A total of 1,084 non-fish species were recorded from the North Sea in SeaLifeBase, including 1043 native species and 41 introduced ones. These species belong to 389 Families, 101 Orders and 24 Classes. There is no information on biological characteristics for 800 species (74%). Furthermore, there are 170 species (16%) with information on one biological characteristic (mostly on length-weight relationships), and three species have studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the North Sea, from most studied to least studied, are length-weight relationships (216 species, 20%), growth (112 species, 10%), diet (55 species, 5%), maturity (49 species, 5%), lifespan (48 species, 4%), spawning (36 species, 3%), natural mortality (24 species, 2%) and fecundity (18 species, 2%) (Figure 3).

Twenty-two species are listed under the categories near threatened (NT), vulnerable (VU), endangered (EN) and critically endangered (CR) of the IUCN Red List. These species show smaller gaps regarding two biological characteristics: growth (16 species, 73%) and length-weight relationships (14 species, 64%), but larger gaps were observed in

TABLE 5 List of the most- and least-studied non-fish species in the *Bay of Biscay/Celtic Sea/Iberian Coast* based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec.) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Most-Studied								
Cephalopoda	Sepiidae	*Sepia officinalis*	Common cuttlefish	native	LC	8/8	30	2 Diet, 1 Fec, 4 L_m, 1 Spawn, 3 M, 1 t_{max}, 8 G, 10 LWR
Reptilia	Cheloniidae	*Chelonia mydas*	Green sea turtle	native	EN	7/8	79	12 Diet, 2 Fec, 2 L_m, 33 Spawn, 2 M, 23 G, 5 LWR
Malacostraca	Aristeidae	*Aristaeomorpha foliacea*	Giant red shrimp	native	NE	7/8	59	2 Diet, 8 L_m, 1 Spawn, 8 M, 2 t_{max}, 28 G, 10 LWR
Malacostraca	Squillidae	*Squilla mantis*	Spottail mantis shrimp	native	NE	7/8	40	1 Fec, 2 L_m, 3 Spawn, 2 M, 1 t_{max}, 8 G, 23 LWR
Least-studied								
Reptilia	Cheloniidae	*Eretmochelys imbricata*	Hawksbill turtle	native	CR	6/8	74	4 Diet, 10 Fec, 7 L_m, 39 Spawn, 10 G, 4 LWR
Reptilia	Cheloniidae	*Caretta caretta*	Loggerhead turtle	native	VU	6/8	38	4 Diet, 10 Fec, 1 L_m, 9 Spawn, 12 G, 2 LWR
Reptilia	Cheloniidae	*Lepidochelys kempii*	Kemp's ridley turtle	native	CR	5/8	28	7 Diet, 2 L_m, 2 M, 15 G, 2 LWR
Mammalia	Balaenopteridae	*Balaenoptera borealis*	Sei whale	native	EN	4/8	10	1 Diet, 2 L_m, 1 t_{max}, 6 LWR
Mammalia	Balaenidae	*Eubalaena glacialis*	North Atlantic right whale	native	CR	4/8	6	2 L_m, 1 t_{max}, 1 G, 2 LWR
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Mammalia	Balaenopteridae	*Balaenoptera physalus*	Fin whale	native	VU	3/8	16	2 Diet, 2 G, 12 LWR
Mammalia	Balaenopteridae	*Balaenoptera musculus*	Blue whale	native	EN	3/8	12	1 Diet, 2 G, 9 LWR
Mammalia	Phocidae	*Cystophora cristata*	Hooded seal	native	VU	2/8	4	2 G, 2 LWR
Aves	Procellariidae	*Pterodroma feae*	Cape Verde petrel	native	NT	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; N.E., not evaluated; NT, near threatened; VU, vulnerable; CR, critic7ally endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

diet (10 species, 45%), maturity (9 species, 41%), fecundity (4 species, 18%), spawning (4 species, 18%), natural mortality (3 species, 14%) and lifespan (3 species, 14%) (Figure 3). The twenty-two species that are included in IUCN Red List are Hawksbill turtle (*Eretmochelys imbricata*; CR), Kemp's ridley turtle (*Lepidochelys kempii*; CR), North Atlantic right whale (*Eubalaena glacialis*; CR), Sei whale (*Balaenoptera borealis*; EN), Blue whale (*Balaenoptera musculus*; EN), Loggerhead turtle (*Caretta*; VU), Sperm whale (*Physeter macrocephalus*; VU), Leatherback turtle (*Dermochelys coriacea*; VU), Fin whale (*Balaenoptera physalus*; VU), Walrus (*Odobenus rosmarus*; VU), Hooded seal (*Cystophora cristata*; VU), North Atlantic bottlenose whale (*Hyperoodon ampullatus*; NT), Balearic shearwater (*Puffinus mauretanicus*; CR), Black-legged kittiwake (*Rissa tridactyla*; VU), Atlantic puffin (*Fratercula arctica*; VU), Velvet scoter (*Melanitta fusca*; VU), Horned grebe (*Podiceps auritus*; VU), Steller's eider (*Polysticta stelleri*; VU), Starlet anemone (*Nematostella vectensis*; VU), Sooty shearwater (*Puffinus griseus*; NT), Red knot (*Calidris canutus*; NT) and Common eider (*Somateria mollissima*; NT).

The most studied species of the North Sea make up about 0.5% (5 species) of non-fish species reported in the area and cover four Classes and five Families (Table 8). These five species are the alien Blue crab (*Callinectes sapidus* and Common cuttlefish (*Sepia officinalis*), with information on eight biological characteristics, the Japanese carpet shell (*Ruditapes philippinarum*), which is another introduced species, Green sea turtle (*Chelonia mydas*) and Common spiny lobster (*Palinurus elephas*) all have information on seven biological characteristics. Out of these five species, the Green sea turtle is listed as EN and the Common spiny lobster as VU (Table 8).

The least studied species make up about 99.5% (1079 species) of non-fish species reported in the North Sea and cover 24 Classes and 387 Families (Table S7) including the 22 species reported in the IUCN Red List (Table 8).

TABLE 6 List of the most- and least-studied non-fish species in the Black Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Most-studied								
Malacostraca	Portunidae	*Callinectes sapidus*	Blue crab	Introduced	NE	8/8	68	2 Diet, 1 Fec, 13 L_m, 2 Spawn, 5 M, 1 t_{max}, 19 G, 25 LWR
Least-studied								
Reptilia	Cheloniidae	*Caretta caretta*	Loggerhead turtle	native	VU	6/8	38	4 Diet, 10 Fec, 1 L_m, 9 Spawn, 12 G, 2 LWR
Mammalia	Phocidae	*Monachus monachus*	Mediterranean monk seal	native	EN	1/8	1	1 Diet
Aves	Anatidae	*Melanitta fusca*	Velvet scoter	native	VU	0/8	0	–
Aves	Podicipedidae	*Podiceps auritus*	Horned grebe	native	VU	0/8	0	–
Aves	Procellariidae	*Puffinus yelkouan*	Levantine shearwater	native	VU	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; N.E., not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

Western Mediterranean Sea

A total of 470 non-fish species are recorded from the Western Mediterranean Sea in SeaLifeBase, including 462 native, two endemic and six introduced species, belonging to 210 Families, 73 Orders and 19 Classes. Regarding the number of biological characteristics studied, there is no information for 308 species (66%). There are 79 species (17%) with information on one biological characteristic (mostly on length-weight relationships), while only one species has studies for all eight biological characteristics (Figure 2).

The individual biological characteristics of the non-fish species of the Western Mediterranean Sea, from most studied to least studied, are: length-weight relationships (136 species, 29%), followed by growth (80 species, 17%), maturity (42 species, 9%), spawning (29 species, 6%), lifespan (26 species, 6%), natural mortality (22 species, 5%), fecundity (16 species, 3%) and diet (15 species, 3%) (Figure 3).

Thirteen species are included in the IUCN Red List under the categories near threatened (NT), vulnerable (VU), endangered (EN) and critically endangered (CR). These species have a small gap regarding one biological characteristic (growth) with information available for 10 species (67%). Larger gaps were observed for the rest biological characteristics: length-weight relationships (7 species, 47%), maturity (7 species, 47%), diet (4 species, 27%), fecundity (4 species, 27%), spawning (4 species, 27% studied), natural mortality (3 species, 20%), and lifespan (2 species, 13%) (Figure 3). The thirteen included in the IUCN Red List are Hawksbill turtle (*Eretmochelys imbricata*; CR), Kemp's ridley turtle (*Lepidochelys kempii*; CR), Loggerhead turtle (*Caretta caretta*; VU), Leatherback turtle (*Dermochelys coriacea*; VU), Olive ridley turtle (*Lepidochelys olivacea*; VU), Noble pen shell (*Pinna nobilis*; CR), Balearic shearwater (*Puffinus mauretanicus*; CR), Audouin's gull (*Larus audouinii*; VU), Velvet scoter (*Melanitta fusca*; VU), Horned grebe (*Podiceps auritus*; VU), Levantine shearwater (*Puffinus yelkouan*; VU), Sooty shearwater (*Puffinus griseus*; NT) and Common eider (*Somateria mollissima*; NT).

The most studied species of the western Mediterranean Sea make up about 1% (6 species) of non-fish species reported from the area and cover three Classes and six Families (Table 9). The six most studied western Mediterranean species in terms of biological characteristics were the Common cuttlefish (*Sepia officinalis*), the only species with eight studied biological characteristics, while the Green sea turtle (*Chelonia mydas*), Deep-water rose shrimp (*Parapenaeus longirostris*), Giant red shrimp (*Aristaeomorpha foliacea*), Spottail mantis shrimp (*Squilla mantis*) and Common spiny lobster (*Palinurus elephas*) have seven biological characteristics studied. Out of these six species, only the Common spiny lobster is included in the list of IUCN as VU (Table 9).

The least-studied species make up about 98% (464 species) of non-fishes reported in the western Mediterranean Sea, cover 19 Classes and 210 Families (Table S8). Of the least-studied species, there are thirteen species that are included in the IUCN Red List (Table 9).

Discussion

Common patterns

The general pattern, observed across all studied ecosystems, is that data availability on biological characteristics of non-fish marine organisms are rather poor, with only two taxonomic groups (sea turtles and marine mammals) appearing to have been adequately studied across most study areas. There is moderately good

TABLE 7 List of the most- and least-studied non-fish species in the Levantine Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec.) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common_name	Status	IUCN	No. Char	No. Rec.	No. of records per characteristic
Most-studied								
Malacostraca	Portunidae	*Callinectes sapidus*	Blue crab	introduced	NE	8/8	68	2 Diet, 1 Fec, 13 L_m, 2 Spawn, 5 M, 1 t_{max}, 19 G, 25 LWR
Cephalopoda	Sepiidae	*Sepia officinalis*	Common cuttlefish	native	LC	8/8	30	2 Diet, 1 Fec, 4 L_m, 1 Spawn, 3 M, 1 t_{max}, 8 G, 10 LWR
Malacostraca	Penaeidae	*Metapenaeus monoceros*	Speckled shrimp	introduced	NE	7/8	106	2 Fec, 10 L_m, 8 Spawn, 31 M, 2 t_{max}, 34 G, 19 LWR
Reptilia	Cheloniidae	*Chelonia mydas*	Green sea turtle	native	EN	7/8	79	12 Diet, 2 Fec, 2 L_m, 33 Spawn, 2 M, 23 G, 5 LWR
Malacostraca	Penaeidae	*Parapenaeus longirostris*	Deep-water rose shrimp	native	NE	7/8	60	1 Fec, 6 L_m, 1 Spawn, 3 M, 1 t_{max}, 36 G, 12 LWR
Malacostraca	Aristeidae	*Aristaeomorpha foliacea*	Giant red shrimp	native	NE	7/8	59	2 Diet, 8 L_m, 1 Spawn, 8 M, 2 t_{max}, 28 G, 10 LWR
Malacostraca	Squillidae	*Squilla mantis*	Spottail mantis shrimp	native	NE	7/8	40	1 Fec, 2 L_m, 3 Spawn, 2 M, 1 t_{max}, 8 G, 23 LWR
Malacostraca	Palinuridae	*Palinurus elephas*	Common spiny lobster	native	VU	7/8	32	1 Fec, 7 L_m, 1 Spawn, 2 M, 1 t_{max}, 12 G, 8 LWR
Least-studied								
Reptilia	Cheloniidae	*Caretta caretta*	Loggerhead turtle	native	VU	6/8	38	4 Diet, 10 Fec, 1 L_m, 9 Spawn,12 G, 2 LWR
Mammalia	Delphinidae	*Pseudorca crassidens*	False killer whale	native	NT	6/8	26	1 Diet, 2 L_m, 1 Spawn, 2 t_{max}, 16 G, 4 LWR
Mammalia	Physeteridae	*Physeter macrocephalus*	Sperm whale	native	VU	4/8	34	15 Diet, 2 L_m, 3 G, 14 LWR
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Bivalvia	Pinnidae	*Pinna nobilis*	Noble pen shell	native	CR	2/8	12	6 t_{max}, 6 G
Mammalia	Phocidae	*Monachus monachus*	Mediterranean monk seal	native	EN	1/8	1	1 Diet
Aves	Laridae	*Larus armenicus*	Armenian gull	native	NT	0/8	0	–
Aves	Procellariidae	*Puffinus yelkouan*	Levantine shearwater	native	VU	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; NE, not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

information coverage for sea turtles in the Celtic Sea/Bay of Biscay/Iberian coast, Black Sea, Levantine Sea, North Sea and western Mediterranean Sea, whereas information coverage for marine mammals is moderate for the Baltic Sea, Celtic Sea/Bay of Biscay/Iberian coast, Black Sea, Levantine Sea, North Sea and western Mediterranean Sea (Table 10). Data on biological characteristics was lowest for cnidarians, whereas there were no available biological information on sponges; the latter group of organisms being globally understudied in terms of biological characteristics (Bell et al., 2015).

Because of the low total number of species recorded compared to the other areas, the Black Sea appears to be among the better studied areas together with the Bay of Biscay/Celtic Sea/Iberian Coast, the Levantine Sea and the western Mediterranean Sea. The Adriatic Sea is the area with the most data gaps compared to the other study areas. This area, for instance, does not have information available on any of the present marine mammal species (Lotze et al., 2011). Likewise, there are no data reported on the biological characteristics of sponges in the Black Sea, where their checklist has been recently updated, at least for part of this area (Topaloglu and Alper, 2014). The difference in species composition among areas has certainly contributed to the number of species studied and the extent of the available information.

There are seven well-studied species across the reviewed ecosystems: Common cuttlefish (*Sepia officinalis*), Giant red shrimp (*Aristaeomorpha foliacea*), Spottail mantis shrimp (*Squilla*

TABLE 8 List of the most- and least-studied non-fish species in the North Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Most-studied								
Malacostraca	Portunidae	*Callinectes sapidus*	Blue crab	Introduced	NE	8/8	68	2 Diet, 1 Fec, 13 L_m, 2 Spawn, 5 M, 1 t_{max}, 19 G, 25 LWR
Cephalopoda	Sepiidae	*Sepia officinalis*	Common cuttlefish	native	LC	8/8	30	2 Diet, 1 Fec, 4 L_m, 1 Spawn, 3 M, 1 t_{max}, 8 G, 10 LWR
Reptilia	Cheloniidae	*Chelonia mydas*	Green sea turtle	native	EN	7/8	79	12 Diet, 2 Fec, 2 L_m, 33 Spawn, 2 M, 23 G, 5 LWR
Bivalvia	Veneridae	*Ruditapes philippinarum*	Japanese carpet shell	introduced	NE	7/8	62	1 Fec, 4 L_m, 4 Spawn, 2 M, 3 t_{max}, 5 G, 43 LWR
Malacostraca	Palinuridae	*Palinurus elephas*	Common spiny lobster	native	VU	7/8	32	1 Fec, 7 L_m, 1 Spawn, 2 M, 1 t_{max}, 12 G, 8 LWR
Least-studied								
Reptilia	Cheloniidae	*Eretmochelys imbricata*	Hawksbill turtle	native	CR	6/8	74	4 Diet, 10 Fec, 7 L_m, 39 Spawn, 10 G, 4 LWR
Reptilia	Cheloniidae	*Caretta caretta*	Loggerhead turtle	native	VU	6/8	38	4 Diet, 10 Fec, 1 L_m, 9 Spawn, 12 G, 2 LWR
Reptilia	Cheloniidae	*Lepidochelys kempii*	Kemp's ridley turtle	native	CR	5/8	28	7 Diet, 2 L_m, 2 M, 15 G, 2 LWR
Mammalia	Physeteridae	*Physeter macrocephalus*	Sperm whale	native	VU	4/8	34	15 Diet, 2 L_m, 3 G, 14 LWR
Mammalia	Balaenopteridae	*Balaenoptera borealis*	Sei whale	native	EN	4/8	10	1 Diet, 2 L_m, 1 t_{max}, 6 LWR
Mammalia	Balaenidae	*Eubalaena glacialis*	North Atlantic right whale	native	CR	4/8	6	2 L_m, 1 t_{max}, 1 G, 2 LWR
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Mammalia	Balaenopteridae	*Balaenoptera physalus*	Fin whale	native	VU	3/8	16	2 Diet, 2 G, 12 LWR
Mammalia	Balaenopteridae	*Balaenoptera musculus*	Blue whale	native	EN	3/8	12	1 Diet, 2 G, 9 LWR
Aves	Laridae	*Rissa tridactyla*	Black-legged kittiwake	native	VU	2/8	32	2 Diet, 30 G
Mammalia	Odobenidae	*Odobenus rosmarus*	Walrus	native	VU	2/8	11	9 G, 2 LWR
Mammalia	Phocidae	*Cystophora cristata*	Hooded seal	native	VU	2/8	4	2 G, 2 LWR
Mammalia	Ziphiidae	*Hyperoodon ampullatus*	North Atlantic bottlenose whale	native	NT	2/8	3	2 Diet, 1 LWR
Aves	Alcidae	*Fratercula arctica*	Atlantic puffin	native	VU	1/8	87	87 G
Aves	Procellariidae	*Puffinus griseus*	Sooty shearwater	native	NT	1/8	2	2 G
Aves	Scolopacidae	*Calidris canutus*	Red knot	native	NT	0/8	0	
Aves	Anatidae	*Melanitta fusca*	Velvet scoter	native	VU	0/8	0	
Anthozoa	Edwardsiidae	*Nematostella vectensis*	Starlet anemone	introduced	VU	0/8	0	

(Continued)

TABLE 8 Continued

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Aves	Podicipedidae	*Podiceps auritus*	Horned grebe	native	VU	0/8	0	
Aves	Anatidae	*Polysticta stelleri*	Steller's eider	native	VU	0/8	0	
Aves	Procellariidae	*Puffinus mauretanicus*	Balearic shearwater	native	CR	0/8	0	
Aves	Anatidae	*Somateria mollissima*	Common eider	native	NT	0/8	0	

The status of the species in the area (Status) as origin categories (native, endemic, introduced) and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; NE, not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied (5 species, 5 Families, 4 Classes).

mantis), Common spiny lobster (*Palinurus elephas*), Blue crab (*Callinectes sapidus*), Green sea turtle (*Chelonia mydas*) and Deep-water rose shrimp (*Parapenaeus longirostris*). These species have good coverage of biological information and sufficient data records for use in ecosystem assessments and modelling. Of these, the most common well-studied species include the Common cuttlefish (*Sepia officinalis*), which is reported within six sites and is highly commercial (Pereira et al., 2019). The current absence of data for the studied taxonomic groups may be due to actual absence of real data (i.e., lack of studies on non-fish marine organisms) or time-lagged entering of research publications in SeaLifeBase or that source of information has not been considered (for instance, grey literature or local journals).

In terms of biological characteristics, the information gap for all species is largest for fecundity, natural mortality and diet, with the better studied characteristics being length-weight relationships (LWR) followed by spawning, lifespan, maturity and growth. The most-studied characteristic (LWR) is common and well-studied across areas but the least-studied ones differ between the Atlantic and the Mediterranean areas. Consequently, research priorities and survey data availability often differ (e.g., Ugland, 1976), as well as to scientific tradition and historical data records (Lotze and Worm, 2009) that are generally scarce in the Mediterranean (Stergiou and Tsikliras, 2006; Fortibuoni et al., 2017). It is worth mentioning here that LWR is the most common even though it is not a trait that is measured for many non-fish taxonomic groups such as marine mammals, reptiles and seabirds. Contrary to LWR that are easier to collect and compute, the sample collection and laboratory work required to determine the diet and fecundity of specimens are costly and time-consuming and require technological equipment and advanced expertise (Dimarchopoulou et al., 2017). In contrast, natural mortality can be easily calculated using existing datasets based on the many known empirical equations that are available (constant across ages/sizes: Pauly, 1980; Then et al., 2015; size/age-based: Chen and Watanabe, 1989; Gislason et al., 2010) without any extra cost or sampling that would be required if other methods were selected (tagging: Krause et al., 2020; length-based and age structured models: Lorenzen, 2022). Therefore, it is strongly recommended, at least for decapod crustaceans and cephalopods, to report maximum age in every study in which growth parameters are determined and, if possible, to calculate and report natural mortality.

For the species that are exploited such as many crustaceans and cephalopods, commercially targeted species are indeed better-studied compared to by-catch and discarded ones that are generally neglected (Baran, 2002). The biological information of the former is more complete due to historically more intensive sampling effort across the studied marine ecosystems because of their economic importance to the fisheries and frequent assessments (Dimarchopoulou et al., 2017). For non-commercial groups, there is generally less information on threatened species compared to those with high commercial value, as it has recently reported for fishes (Dimarchopoulou et al., 2017; Daskalaki et al., 2022) due to the low accessibility of deep-water non-fish marine species that are generally less sampled in routine surveys that rarely extend to deep waters (Sardà et al., 2004). Therefore, the study of threatened, deep-water and non-indigenous species should be prioritized over the well-studied commercial species similar to recommendations for marine fishes (Daskalaki et al., 2022). When threatened species cannot be sampled with non-destructive methods, such as underwater censuses or tagging experiments, it is suggested that if dead after capture, the specimens should be exhaustively studied across their biological characteristics to ensure the maximum economy of sampling (Dimarchopoulou et al., 2017). The study of threatened species should be a priority as they are all good candidates for field data collection. However, the existing gaps on species that are routinely sampled during scientific surveys should also be considered by scientists. The basic characteristics of a species (measurement of length and weight) should always be recorded even from single individuals in the market (in the case of crustaceans and cephalopods) or stranded individuals in the case of marine mammals and reptiles (see the importance of single specimen characteristics for sharks in Tsikliras and Dimarchopoulou, 2021).

Better research coverage on the diets of all marine organisms would greatly benefit future ecosystem models and improve future versions of the current ones (Dimarchopoulou et al., 2017), while more studies on growth, mortality, maturity and spawning of exploited populations will improve the quality of stock assessments within the framework of STECF and GFCM. This, in turn, will reduce uncertainty on the outcome of stock assessment and ecosystem models and will eventually lead to improvements in ecosystem based fisheries management, especially in the

TABLE 9 List of the most- and least-studied non-fish species in the western Mediterranean Sea based on the number of studied biological characteristics (No. Char.) and the number of records (No. Rec.) per characteristic (feeding preferences (Diet), fecundity (Fec), maturity (L_m), spawning (Spawn), mortality (M), lifespan (t_{max}), growth (G), and length-weight relationships (LWR).

Class	Family	Scientific name	Common name	Status	IUCN	No. Char.	No. Rec.	No. of records per characteristic
Most-studied								
Cephalopoda	Sepiidae	*Sepia officinalis*	Common cuttlefish	native	LC	8/8	30	2 Diet, 1 Fec, 4 L_m, 1 Spawn, 3 M, 1 t_{max}, 8 G, 10 LWR
Reptilia	Cheloniidae	*Chelonia mydas*	Green sea turtle	native	EN	7/8	79	12 Diet, 2 Fec, 2 L_m, 33 Spawn, 2 M, 23 G, 5 LWR
Malacostraca	Penaeidae	*Parapenaeus longirostris*	Deep-water rose shrimp	native	NE	7/8	60	1 Fec, 6 L_m, 1 Spawn, 3 M, 1 t_{max}, 36 G, 12 LWR
Malacostraca	Aristeidae	*Aristaeomorpha foliacea*	Giant red shrimp	native	NE	7/8	59	2 Diet, 8 L_m, 1 Spawn, 8 M, 2 t_{max}, 28 G, 10 LWR
Malacostraca	Squillidae	*Squilla mantis*	Spottail mantis shrimp	native	NE	7/8	40	1 Fec, 2 L_m, 3 Spawn, 2 M, 1 t_{max}, 8 G, 23 LWR
Malacostraca	Palinuridae	*Palinurus elephas*	Common spiny lobster	native	VU	7/8	32	1 Fec, 7 L_m, 1 Spawn, 2 M, 1 t_{max}, 12 G, 8 LWR
Least-studied								
Reptilia	Cheloniidae	*Eretmochelys imbricata*	Hawksbill turtle	native	CR	6/8	74	4 Diet, 10 Fec, 7 L_m, 39 Spawn, 10 G, 4 LWR
Reptilia	Cheloniidae	*Caretta caretta*	Loggerhead turtle	native	VU	6/8	38	4 Diet, 10 Fec, 1 L_m, 9 Spawn, 12 G, 2 LWR
Reptilia	Cheloniidae	*Lepidochelys kempii*	Kemp's ridley turtle	native	CR	5/8	28	7 Diet, 2 L_m, 2 M, 15 G, 2 LWR
Reptilia	Dermochelyidae	*Dermochelys coriacea*	Leatherback turtle	native	VU	3/8	19	2 L_m, 7 G, 10 LWR
Reptilia	Cheloniidae	*Lepidochelys olivacea*	Olive ridley turtle	native	VU	3/8	7	1 L_m, 3 G, 3 LWR
Bivalvia	Pinnidae	*Pinna nobilis*	Noble pen shell	native	CR	2/8	12	6 t_{max}, 6 G
Aves	Laridae	*Larus audouinii*	Audouin's gull	native	VU	1/8	5	5 G
Aves	Procellariidae	*Puffinus griseus*	Sooty shearwater	native	NT	1/8	2	2 G
Aves	Anatidae	*Melanitta fusca*	Velvet scoter	native	VU	0/8	0	–
Aves	Podicipedidae	*Podiceps auritus*	Horned grebe	native	VU	0/8	0	–
Aves	Procellariidae	*Puffinus mauretanicus*	Balearic shearwater	native	CR	0/8	0	–
Aves	Procellariidae	*Puffinus yelkouan*	Levantine shearwater	native	VU	0/8	0	–
Aves	Anatidae	*Somateria mollissima*	Common eider	native	NT	0/8	0	–

The status of the species in the area (Status) as origin categories (native, endemic, introduced), and the status as IUCN Red List categories (LC, least concern; EN, endangered; DD, data deficient; NE, not evaluated; NT, near threatened; VU, vulnerable; CR, critically endangered) are also included. Only species with information on at least 7 biological characteristics and at least 30 records available are considered as well studied.

Mediterranean and the Black Sea (Rodriguez-Perez et al., 2023). In many areas that experience an influx of non-indigenous species, such as the eastern Mediterranean Sea (Galil et al., 2015), the study of non-indigenous species biology should also be prioritized. Their biological characteristics in the new habitats/areas should be compared to those in native range, aiming to identify the potential effects of alien species on local populations, habitats and communities (Daskalaki et al., 2022).

TABLE 10 Comparison regarding the status of the studied groups in European Seas and adjacent waters.

Species groups	Adriatic		Aegean		Baltic Sea		Bay of Biscay Celtic Sea Iberian Coast		Black Sea		Levantine Sea		North Sea		Western Mediterranean Sea	
	N	Status	N	Status	N	Status	N	Status	N	Status	N	Status	N	Status	N	Status
Cnidarians	100	Poor	72	Poor	74	Poor	50	Poor	8	Poor	19	Poor	149	Poor	82	Poor
Sponges	4	Poor	1	Poor	8	Poor	40	Poor	–	–	1	Poor	22	Poor	7	Poor
Echinoderms	68	Poor	67	Poor	35	Poor	19	Poor	1	Poor	8	Poor	58	Poor	16	Poor
Molluscs	86	Poor	90	Poor	205	Poor	120	Poor	51	Poor	258	Poor	314	Poor	219	Poor
Crustaceans	95	Poor	121	Poor	250	Poor	113	Poor	43	Poor	88	Poor	455	Poor	119	Poor
Seabirds	5	Poor	2	Poor	20	Poor	4	Poor	7	Poor	12	Poor	54	Poor	18	Poor
Sea turtles	1	Moderate	1	Moderate	1	Moderate	5	Good	1	Good	3	Good	5	Good	6	Good
Marine mammals	–	–	1	Poor	13	Moderate	11	Moderate	5	Moderate	12	Moderate	27	Moderate	3	Moderate
Total number of species	359		355		606		362		116		401		1084		470	
Species with 1 characteristic	48		42		112		65		19		77		170		79	
Species with 8 characteristics	2		1		1		1		1		2		3		1	
	N	%	N	%	N	%	N	%	N	%	N	%	N	%	N	%
LWR	97	27	77	22	140	23	112	31	41	35	136	34	216	20	136	29
Growth	52	14	39	11	46	8	53	15	30	26	83	21	112	10	80	17
Mortality	16	4	11	3	11	2	11	3	7	6	22	5	24	2	22	5
Lifespan	22	6	13	4	29	5	21	6	9	8	33	8	48	4	26	6
Maturity	33	9	19	5	24	4	28	8	14	12	45	11	49	5	42	9
Spawning	22	6	17	5	15	2	22	6	9	8	38	9	36	3	29	6
Fecundity	11	3	7	2	7	1	10	3	6	5	20	5	18	2	16	3
Diet	6	2	5	1	29	5	24	7	9	8	23	6	55	5	15	3

Adriatic Sea

In the Adriatic Sea, there is currently poor biological information coverage for non-fish species, with relatively more information being available for sea turtles compared to crustaceans, echinoderms and cnidarians, while no data exists for marine mammals (Table 10). Only sea turtles qualify as moderate data coverage with all other areas being data poor (Table 10). In general, the gap is widest for information on diet and fecundity. The information coverage for the Adriatic is good for only a few well-studied species of crustaceans and one cephalopod that are commercially important.

Compared to other Mediterranean areas, the Adriatic Sea is an overall well-studied ecosystem in terms of stock assessments (Froese et al., 2018b) and ecosystem models (Barausse et al., 2009), with important contributions on the effects of fishing (Coll et al., 2007), filling gaps in survey datasets (Coro et al., 2022), the effect of COVID-19 on fish stocks (Scarcella et al., 2022) and fisheries in general, especially in the western part of the sea (Lotze et al., 2011). There are even some historical data available for large marine animals (Lotze and Worm, 2009). Non-indigenous species have also been extensively studied in terms of their effect on the food web dynamics (Libralato et al., 2010; Libralato et al., 2015). It appears that the data collected from scientific surveys on non-fish marine organisms either remain unpublished, or do not include the biological characteristics covered in this review. Furthermore, they have potentially not yet been included in SeaLifeBase.

Aegean Sea

Within the Aegean Sea, Sea turtles have better coverage, in terms of biological characteristics, compared to all other groups, (Table 10). However, similarly to the Adriatic, only sea turtles qualify as moderately studied while in all other areas are poorly studied (Table 10). The knowledge gap is widest for information on diet followed by fecundity. Good biological information is available for a few well-studied species of crustaceans and one cephalopod, all being commercially exploited.

Official stock assessments are generally scarce in the Aegean Sea and cover only a handful of commercial stocks owing to several years missing from data collection framework (Tsikliras et al., 2021). Nevertheless, over 100 fish and invertebrate Aegean Sea stocks have been recently assessed using data-poor methods (Froese et al., 2018b; Tsikliras et al., 2021). Several EwE ecosystem models have been recently developed for parts of the Aegean Sea (Thracian Sea: Tsagarakis et al., 2010; Pagassitikos Gulf: Dimarchopoulou et al., 2019; Thermaikos Gulf: Dimarchopoulou et al., 2022) and a recent one for the entire Aegean Sea (Keramidas et al., 2022) along with temporal simulations (Papantoniou et al., 2021; Dimarchopoulou et al., 2022), while spatial models are still not available (Keramidas et al., 2023). Besides overfishing, which is considered the main driver of exploited marine populations in the Mediterranean Sea (Dimarchopoulou et al., 2021), the direct (sea warming and species distribution shifts) and indirect (entering and expansion of alien species) effects of climate change are major issues in the eastern Mediterranean Sea that concern the scientific community (Cherif et al., 2020).

Baltic Sea

Although there are no well-studied species in the Baltic Sea, marine mammals and echinoderms have higher counts of species with studied biological characteristics compared to molluscs, crustaceans, seabirds, and cnidarians (Table 10). The biological information coverage is moderate for sea turtles and marine mammals and poor for all other groups (Table 10). The knowledge gap is widest for information on fecundity and natural mortality and narrowest for length-weight relationships. An introduced species, Harris mud crab (*Rhithropanopeus harrisii*) is considered as a near well-studied species, having eight biological characteristics and 17 records available.

The Baltic Sea is a well-studied ecosystem (Feistel et al., 2008) with many stock assessments available (Froese et al., 2018b; Froese et al., 2021) and ecosystem models using various approaches (Österblom et al., 2007; Bauer et al., 2019) that geographically cover basins of the entire sea (see Scotti et al., 2022 and references in their Table S2). Status and dynamics of several ecosystem components have been studied in the Baltic Sea including hypotheses on alien species (Dobrzycka-Krahel and Medina-Villar, 2020) and their effect on ecosystem services (Ojaveer et al., 2023), eutrophication (Bauer et al., 2019), fisheries (Scotti et al., 2022) but also climate change (Niiranen et al., 2013) and grey seal (*Halichoerus grypus*) interaction with fisheries (Costalago et al., 2019). The study of non-indigenous species, which are numerous in the Baltic Sea (Reusch et al., 2018), and their effects on marine ecosystems should be prioritized. With such a wealth of biotic and abiotic information on the ecosystems of the Baltic Sea ecosystems, with long-term datasets of many marine groups available since the 1950s and some expeditions dating back to 1850s (Ojaveer et al., 2010), it is surprising that the basic biological characteristics for many marine organisms supporting ecosystem models and assessments have not been published. The possibility that this information is published but has not yet been scrutinized by SeaLifeBase is also a potential explanation especially in case of local or not yet digitized journals.

Bay of Biscay/Celtic Sea/Iberian Coast

In the *Bay of Biscay/Celtic Sea/Iberian Coast* (combined), vertebrates (marine mammals, sea turtles and seabirds) have higher counts of species with biological information compared to invertebrates (echinoderms, molluscs, crustaceans and cnidarians) (Table 10). The biological information coverage is good for sea turtles, moderate for marine mammals and poor for all other groups (Table 10). The knowledge gap is widest for information on fecundity and natural mortality and narrowest for length-weight relationships. In the *Bay of Biscay/Celtic Sea/Iberian Coast* area, there is good biological information for a few well-studied species of crustaceans, one cephalopod (all commercial) and one species of sea turtle.

The three combined areas of the NE Atlantic (*Celtic Sea, Bay of Biscay, Iberian Coast) are all* high biodiversity areas with many habitats and marine species, including endangered and protected species like cetaceans (Laran et al., 2017; Spitz et al., 2018) and seabirds (Morley et al., 2016). The *Celtic Sea and Bay of Biscay* are often considered as a single ecosystem in modelling studies (Moullec et al., 2017*).* They *are all rich in terms of scientific output* (Borja et al., 2011)*, with many ecosystem models developed, simulated* (Lassalle et al., 2011; Corrales et al., 2022) *and compared (*Moullec et al., 2017*). Several ecological hypotheses have been examined based on ecosystem and ecological models* (Le Marchand et al., 2020). The number of stocks that have been assessed covers the majority of commercial fisheries (Guénette and Gascuel, 2012; Froese et al., 2018a; Froese et al., 2021).

Black Sea

Marine mammals and molluscs have higher counts of species with biological information compared to crustaceans and cnidarians in the Black Sea (Table 10). The biological information coverage is good for sea turtles, moderate for marine mammals and poor for all other groups (Table 10). The biological knowledge gap is widest for information on fecundity and natural mortality and narrowest for length-weight relationships. An introduced crustacean, Blue crab (*Callinectes sapidus*) has sufficient information for eight biological characteristics and is considered the best studied organism in the Black Sea of those with records in SeaLifeBase.

The Black Sea together with the Mediterranean marine ecosystems are rather poorly studied compared to the NE Atlantic ones (Güneroğlu et al., 2019). However, the Black Sea ecosystem structure (Akoglu et al., 2014) and fisheries (Prodanov et al., 1997; Daskalov, 2002; Gucu, 2002) are relatively well studied in certain parts of the sea. The effect of non-indigenous species on the populations and ecosystem of the Black Sea (Shiganova, 1998) and the overall status of the ecosystem after anthropogenic impacts has been evaluated (Zaitsev, 1992; Kideys, 2002; Daskalov et al., 2017). Furthermore, some of the commercial fish and invertebrate Black Sea stocks have been assessed using catch-based (Tsikliras et al., 2015) and other data-limited methods (Froese et al., 2018a; Daskalov et al., 2020; Demirel et al., 2020).

Levantine Sea

In the Levantine Sea, marine mammals, sea turtles, seabirds and crustaceans have higher counts of species with information on biological characteristics compared to echinoderms, molluscs and cnidarians (Table 10). The biological information coverage is good for sea turtles, moderate for marine mammals and poor for all other groups (Table 10). In general, the knowledge gap is widest for information on fecundity and natural mortality. Overall, current coverage on biological information is good for a few well-studied species of crustaceans, one sea turtle and one cephalopod.

Due to its proximity to the Suez Canal, the Levantine Sea is the first to receive the non-indigenous species migrating into the Mediterranean Sea from the Red Sea, which have altered the biodiversity of the area (Galil et al., 2015). *Despite the existence of local journals with long publishing history in the area (Israel Journal of Ecology & Evolution/* Israel Journal of Zoology, *published since 1963;* Turkish Journal of Zoology, *published since 1977;* Egyptian *Journal of Aquatic Biology and Fisheries*, published since 1997) and some recent attempts (e.g., *Syrian Journal of Agricultural Research* since 2014), the amount of data on the biology of non-fish marine organisms is rather limited (Stergiou and Tsikliras, 2006) and the Syrian coast has been characterized as one of the least-studied areas for marine mammals in the Mediterranean Sea (Saad and Mahfoud, 2022). *Although the extent of scientific surveys is rather limited and historical biological data are generally lacking or concentrated in specific countries* (Tsikliras et al., 2010)*, there has been an* increase of scientific output in the Levantine Sea during the last decades (Tsikliras and Stergiou, 2014). Despite the data deficiencies and the lack of long time series, EwE ecosystem models have been developed to examine the effect of non-indigenous species, climate change and other anthropogenic affects in Israel (Corrales et al., 2017; Corrales et al., 2018; Shabtay et al., 2018; Grossowicz et al., 2020; Ofir et al., 2023) in addition to bioeconomic models (Peled et al., 2020; Michael-Bitton et al., 2022) and non-indigenous species in Cyprus (Michailidis et al., 2019), including lionfish *Pterois miles* (Savva et al., 2020) and silver-cheeked toadfish *Lagocephalus sceleratus* (Ulman et al., 2021).

North Sea

Similarly to the *Bay of Biscay/Celtic Sea/Iberian Coast* (combined), marine vertebrates (sea turtles, marine mammals and seabirds) have higher counts of species with information on biological characteristics compared to the marine invertebrates (echinoderms, molluscs, crustaceans and cnidarians) (Table 10). Biological coverage is good for sea turtles, moderate for marine mammals and poor for all other groups (Table 10). In general, the knowledge gap is widest for information on fecundity and natural mortality, and narrowest for length-weight relationships. Overall, the current coverage on biological information is good for a few well-studied species of crustaceans, sea turtle, cephalopod and Harris mud crab (*Rhithropanopeus harrisii*), an introduced non-commercial species, is considered a near well-studied species, having eight biological characteristics and 17 records available.

The North Sea marine ecosystem is one of the most biotically-rich and productive seas in Europe (Quante et al., 2016) and has been well studied for many decades with respect to ecosystem structure (Stäbler et al., 2018), effects of fishing and climate (Heath, 2005), system dynamics (Luczak et al., 2012), and regime shifts (Beaugrand, 2004), as well as ecological models (Fransz et al., 1991). There are several ecosystem models available for the North Sea (Burkhard et al., 2011; Mackinson et al., 2018), including temporal (Mackinson et al., 2009), spatial (Püts et al., 2020) and bioeconomic (Beattie et al., 2002) models, while the majority of commercial fish and invertebrate stocks are being regularly assessed (Froese et al., 2021).

Western Mediterranean Sea

Marine mammals and sea turtles have higher counts of species with studied biological characteristics compared to seabirds, crustaceans, molluscs, echinoderms and cnidarians in the western Mediterranean Sea with good biological information coverage for sea turtles, moderate for marine mammals and poor for all other groups (Table 10). In general, the knowledge gap is widest for information on diet followed by fecundity. Overall, the current coverage on biological information is better for a few well-studied species of crustaceans, one sea turtle and one cephalopod.

Parts of the western Mediterranean Sea, especially the northern coastline, have been well studied in terms of ecosystem modelling (Catalan Sea: Coll et al., 2006; Coll et al., 2008; Gulf of Lions: Vilas et al., 2021), even in deep waters (Tecchio et al., 2013), and invertebrate stock assessments (Froese et al., 2018a). Specific aspects of the biology of many marine taxonomic groups have been studied in various parts of the area (feeding/crustaceans: Cartes et al., 2002; maturity/cephalopods: Quetglas et al., 2010; cnidarians/growth and spawning: Rosa et al., 2013). This is partly due to the presence of scientific journals in the area with long publication history (e.g., *Scientia Marina* published since 1955 as *Investigación Pesquera*) devoted to the biology of marine organisms and of course due to the long and consistent scientific tradition of western Mediterranean countries in marine sciences. It should be noted here that the southern Mediterranean countries have a long scientific tradition in fisheries and marine biology (Stergiou and Tsikliras, 2006) and have produced significant scientific output on the biology of marine populations for over a century (Tsikliras et al., 2010).

Priority areas for future research

In order to reduce knowledge gaps on the biology of non-fish marine species across European Seas, future research should focus on species with insufficient or missing biological data that are common to the majority of the studied areas such as sea turtles, monk seal and seabirds; more effort is generally required for the Adriatic and the Aegean Seas. Invertebrate species with low or no commercial value that are often collected in scientific surveys and/or as by-catch in commercial fisheries should not be overlooked. Long-lived species should be prioritized in order to understand their biology and potential threats to their populations other than fishing. In areas invaded by non-indigenous species, such as the eastern Mediterranean Sea, research should be focused on the study of life-history characteristics of these species in their new environment and a comparison with their habits in their native distribution. Besides overfishing and incidental fishing, climate change is one of the major threats to marine life and the response of marine populations to climate effects is directly related to their population characteristics and thermal preferences. Knowledge of the latter, which today is known only for a small proportion of marine species, will improve species distribution models and the understanding of climate effects. Threatened species that are listed under the IUCN categories should be prioritized through focused research and use of any possible data available, including strandings and incidental catches (without harming the animal if still alive). This approach offers an expedient strategy in addressing the gap between current and desired knowledge with respect to biological characteristics through focused field studies. Despite the number of scientific publications that investigate the welfare of charismatic rather than non-charismatic species (Hosey et al., 2020) the gaps of biological knowledge in charismatic species are still wide and should be addressed.

Author contributions

LA, PS-Y, KK-R, EC, and RR extracted and analyzed the data and drafted the manuscript with contributions from all other authors. All authors contributed to the article and approved the submitted version.

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Conflict of interest

Publisher's note

Supplementary material

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References

Akoglu, E., Salihoglu, B., Libralato, S., Oguz, T., and Solidoro, C. (2014). An indicator-based evaluation of Black Sea food web dynamics during 1960–2000. *J. Mar. Syst.* 134, 113–125. doi: 10.1016/j.jmarsys.2014.02.010

Baran, E. (2002). "The importance of non-commercial fish," in *Fisheries and Aquaculture: Towards Sustainable Aquatic Living Resources Management. UNESCO Encyclopedia of Life Support Systems.* Ed. P. Safran (Oxford: EOLSS Publishers), 1±7.

Barausse, A., Duci, A., Mazzoldi, C., Artioli, Y., and Palmeri, L. (2009). Trophic network model of the Northern Adriatic Sea: Analysis of an exploited and eutrophic ecosystem. *Estuarine Coast. Shelf Sci.* 83, 577–590. doi: 10.1016/j.ecss.2009.05.003

Bauer, B., Horbowy, J., Rahikainen, M., Kulatska, N., Müller-Karulis, B., Tomczak, M. T., et al. (2019). Model uncertainty and simulated multispecies fisheries management advice in the Baltic Sea. *PloS One* 14, e0211320. doi: 10.1371/journal.pone.0211320

Beattie, A., Sumaila, U. R., Christensen, V., and Pauly, D. (2002). A model for the bioeconomic evaluation of marine protected area size and placement in the North Sea. *Nat. Resour. Model* 15, 413–437. doi: 10.1111/j.1939-7445.2002.tb00096.x

Beaugrand, G. (2004). The North Sea regime shift: Evidence, causes, mechanisms and consequences. *Prog. Oceanogr.* 60, 245–262. doi: 10.1016/j.pocean.2004.02.018

Bell, J. J., McGrath, E., Biggerstaff, A., Bates, T., Cárdenas, C. A., and Bennett, H. (2015). Global conservation status of sponges. *Conserv. Biol.* 29, 42–53. doi: 10.1111/cobi.12447

Borja, Á, Galparsoro, I., Irigoien, X., Iriondo, A., Menchaca, I., Muxika, I., et al. (2011). Implementation of the European Marine Strategy Framework Directive: a methodological approach for the assessment of environmental status, from the Basque Country (Bay of Biscay). *Mar. Pollut. Bull.* 62, 889–904. doi: 10.1016/j.marpolbul.2011.03.031

Burkhard, B., Opitz, S., Lenhart, H., Ahrendt, K., Garthe, S., Mendel, B., et al. (2011). Ecosystem based modeling and indication of ecological integrity in the German North Sea—Case study offshore wind parks. *Ecol. Indic.* 11, 168–174. doi: 10.1016/j.ecolind.2009.07.004

Cartes, J. E., Abelló, P., Lloris, D., Carbonell, A., Torres, P., Maynou, F., et al. (2002). Feeding guilds of western Mediterranean demersal fish and crustaceans: an analysis based in a spring survey. *Scientia Marina* 66 (S2), 209–220. doi: 10.3989/scimar.2002.66s2209

Chen, S., and Watanabe, S. (1989). Age dependence of natural mortality coefficient in fish population dynamics. *Nippon Suisan Gakkaishi* 55, 205–208. doi: 10.2331/suisan.55.205

Cherif, S., Doblas-Miranda, E., Lionello, P., Borrego, C., Giorgi, F., Iglesias, A., et al. (2020). "Drivers of change," in *Climate and Environmental Change in the Mediterranean Basin – Current Situation and Risks for the Future. First Mediterranean Assessment Report*. Eds. W. Cramer, J. Guiot and K. Marini (Marseille, France: Union for the Mediterranean, Plan Bleu, UNEP/MAP), 59–180.

Christensen, V., and Walters, C. (2004). Ecopath with Ecosim: methods, capabilities and limitations. *Ecol. Model.* 172, 109–139. doi: 10.1016/j.ecolmodel.2003.09.003

Claudet, J., Bopp, L., Cheung, W. W. L., Devillers, R., Escobar-Briones, E., Haugan, P., et al. (2019). A roadmap for using the UN decade of ocean science for sustainable development in support of science, policy, and action. *One Earth* 2, 34–42. doi: 10.1016/j.oneear.2019.10.012

Coll, M., Grazia Pennino, M., Steenbeek, J., Sole, J., and Bellido, J. M. (2019). Predicting marine species distributions: Complementarity of food-web and Bayesian hierarchical modelling approaches. *Ecol. Model.* 405, 86–101. doi: 10.1016/j.ecolmodel.2019.05.005

Coll, M., Palomera, I., and Tudela, S. (2009). Decadal changes in a NW Mediterranean Sea food web in relation to fishing exploitation. *Ecol. Model.* 220, 2088–2102. doi: 10.1016/j.ecolmodel.2009.04.049

Coll, M., Palomera, I., Tudela, S., and Dowd, M. (2008). Food-web dynamics in the South Catalan Sea ecosystem (NW Mediterranean) for 1978–2003. *Ecol. Model.* 217, 95–116. doi: 10.1016/j.ecolmodel.2008.06.013

Coll, M., Palomera, I., Tudela, S., and Sardà, F. (2006). Trophic flows, ecosystem structure and fishing impacts in the South Catalan Sea, Northwestern Mediterranean. *J. Mar. Syst.* 59, 63–96. doi: 10.1016/j.jmarsys.2005.09.001

Coll, M., Santojanni, A., Palomera, I., Tudela, S., and Arneri, E. (2007). An ecological model of the Northern and Central Adriatic Sea: Analysis of ecosystem structure and fishing impacts. *J. Mar. Syst.* 67, 119–154. doi: 10.1016/j.jmarsys.2006.10.002

Coro, G., Bove, P., Armelloni, E. N., Masnadi, F., Scanu, M., and Scarcella, G. (2022). Filling Gaps in Trawl Surveys at Sea through Spatiotemporal and Environmental Modelling. *Front. Mar. Sci.* 9, 919339. doi: 10.3389/fmars.2022.919339

Corrales, X., Coll, M., Ofir, E., Heymans, J. J., Steenbeek, J., Goren, M., et al. (2018). Future scenarios of marine resources and ecosystem conditions in the Eastern Mediterranean under the impacts of fishing, alien species and sea warming. *Sci. Rep.* 8, 14284. doi: 10.1038/s41598-018-32666-x

Corrales, X., Coll, M, Ofir, E, Piroddi, C., Goren, M., Edelist, D., et al. (2017). Hindcasting the dynamics of an Eastern Mediterranean marine ecosystem under the impacts of multiple stressors. *Mar. Ecol. Prog. Ser.* 580, 17–36. doi: 10.3354/meps12271

Corrales, X., Katsanevakis, S., Coll, M., Heymans, J. J., Piroddi, C., Ofir, E., et al. (2019). Advances and challenges in modelling the impacts of invasive alien species on aquatic ecosystems. *Biol. Invasions* 22, 907–934. doi: 10.1007/s10530-019-02160-0

Corrales, X., Preciado, I., Gascuel, D., Lopez de Gamiz-Zearra, A., Hernvann, P.-Y., Mugerza, E., et al. (2022). Structure and functioning of the Bay of Biscay ecosystem: A trophic modelling approach. *Estuarine Coast. Shelf Sci.* 26, 107658. doi: 10.1016/j.ecss.2021.107658

Costalago, D., Bauer, B., Tomczak, M. T., Lundström, K., and Winder, M. (2019). The necessity of a holistic approach when managing marine mammal–fisheries interactions: Environment and fisheries impact are stronger than seal predation. *Ambio* 48, 552–564. doi: 10.1007/s13280-018-1131-y

Damalas, D., Kalyvioti, G., Sabatella, E. C., and Stergiou, K. I. (2018). Open data in the life sciences: the 'Selfish Scientist Paradox'. *Ethics Sci. Environ. Politics* 18, 27–36. doi: 10.3354/esep00182

Daskalaki, E., Koufalis, E., Dimarchopoulou, D., and Tsikliras, A. C. (2022). Scientific progress made towards bridging the knowledge gap in the biology of Mediterranean marine fishes. *PloS One* 17, e0277383. doi: 10.1371/journal.pone.0277383

Daskalov, G. (2002). Overfishing drives a trophic cascade in the Black Sea. *MEPS* 225, 53–63. doi: 10.3354/meps225053

Daskalov, G. M., Boicenco, L., Grishin, A. N., Lazar, L., Mihneva, V., Shlyakhov, V. A., et al. (2017). Architecture of collapse: regime shift and recovery in an hierarchically structured marine ecosystem. *Global Change Biol.* 23, 1486–1498. doi: 10.1111/gcb.13508

Daskalov, G. M., Demirel, N., Ulman, A., Georgieva, Y., and Zengin, M. (2020). Stock dynamics and predator–prey effects of Atlantic bonito and bluefish as top predators in the Black Sea. *ICES J. Mar. Sci.* 77, 2995–3005. doi: 10.1093/icesjms/fsaa182

Deacon, M. (1997). *Scientists and the Sea 1650–1900: A Study of Marine Science. 2nd ed* (England: Routledge).

Demirel, N., Zengin, M., and Ulman, A. (2020). First large-scale eastern Mediterranean and Black Sea stock assessment reveals a dramatic decline. *Front. Mar. Sci.* 7, 103. doi: 10.3389/fmars.2020.00103

Dimarchopoulou, D. (2020). Ecosystem approach to fisheries management in the Aegean Sea (School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece).

Dimarchopoulou, D., Keramidas, I., Sylaios, G., and Tsikliras, A. C. (2021). Ecotrophic effects of fishing across the Mediterranean Sea. *Water* 13, 482. doi: 10.3390/w13040482

Dimarchopoulou, D., Stergiou, K. I., and Tsikliras, A. C. (2017). Gap analysis on the biology of Mediterranean marine fishes. *PloS One* 12, e0175949. doi: 10.1371/journal.pone.0175949

Dimarchopoulou, D., Tsagarakis, K., Keramidas, I, and Tsikliras, A. C. (2019). Ecosystem models and effort simulations of an untrawled gulf in the central Aegean Sea. *Front. Mar. Sci.* 6, 648. doi: 10.3389/fmars.2019.00648

Dimarchopoulou, D., Tsagarakis, K., Sylaios, G., and Tsikliras, A. C. (2022). Ecosystem trophic structure and fishing effort simulations of a major fishing ground in the northeastern Mediterranean Sea (Thermaikos Gulf). *Estuarine Coast. Shelf Sci.* 264, 107667. doi: 10.1016/j.ecss.2021.107667

Dobrzycka-Krahel, A., and Medina-Villar, S. (2020). Alien species of Mediterranean origin in the Baltic Sea Region: current state and risk assessment. *Environ. Rev.* 28 (3), 339–356. doi: 10.1139/er-2019-0074

FAO (2020). *The State of World Fisheries and Aquaculture 2020* (Rome: Sustainability in action).

Feistel, R., Nausch, G., and Wasmund, N. (2008). *State and Evolution of the Baltic Sea.* (Wiley Interscience), 1952–2005.

Fortibuoni, T., Libralato, S., Arneri, E., Giovanardi, O., Solidoro, C., and Raicevich, S. (2017). Fish and fishery historical data since the 19th century in the Adriatic Sea, Mediterranean. *Sci. Data* 4, 170104. doi: 10.1038/sdata.2017.104

Fransz, H. G., Mommaerts, J. P., and Radach, G. (1991). Ecological modelling of the North sea. *Netherlands J. Sea Res.* 28, 67–140. doi: 10.1016/0077-7579(91)90005-L

Froese, R., Tsikliras, A. C., Scarcella, G., and Gascuel, D. (2021). Progress towards ending overfishing in the Northeast Atlantic. *Mar. Policy* 125, 104282. doi: 10.1016/j.marpol.2020.104282

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2020). Estimating stock status from relative abundance and resilience. *ICES J. Mar. Sci.* 77, 527–538. doi: 10.1093/icesjms/fsz230

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018b). A new approach for estimating stock status from length frequency data. *ICES J. Mar. Sci.* 75, 2004–2015. doi: 10.1093/icesjms/fsy078

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018a). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/j.marpol.2018.04.018

Galil, B. S., Boero, F., Campbell, M. L., Carlton, J. T., Cook, E., Fraschetti, S., et al. (2015). 'Double trouble': the expansion of the Suez Canal and marine bioinvasions in the Mediterranean Sea. *Biol. Invasions* 17, 973–976. doi: 10.1007/s10530-014-0778-y

Gislason, H., Daan, N., Rice, J. C., and Pope, J. G. (2010). Size, growth, temperature and the natural mortality of marine fish. *Fish Fisheries* 11, 149–158. doi: 10.1111/j.1467-2979.2009.00350.x

Grossowicz, M., Ofir, E., Shabtay, A., Wood, J., Biton, E., Belkin, N., et al. (2020). Modeling the effects of brine outflow from desalination plants on coastal food-webs of the Levantine basin (eastern Mediterranean Sea). *Desalination* 496, 114757. doi: 10.1016/j.desal.2020.114757

Grüss, A., Palomares, M. L. D., Poelen, J. H., Barile, J. R., Aldemita, C. D., Ortiz, S. R., et al. (2019). Building bridges between global information systems on marine organisms and ecosystem models. *Ecol. Model.* 398, 1–19. doi: 10.1016/j.ecolmodel.2019.01.023

Gucu, A. C. (2002). Can Overfishing be Responsible for the Successful Establishment of *Mnemiopsis leidyi* in the Black Sea? *Estuarine Coast. Shelf Sci.* 54, 439–451. doi: 10.1006/ecss.2000.0657

Guénette, S., and Gascuel, D. (2012). Shifting baselines in European fisheries: The case of the Celtic Sea and Bay of Biscay. *Ocean Coast. Manage.* 70, 10–21. doi: 10.1016/j.ocecoaman.2012.06.010

Güneroğlu, A., Samsun, O., Feyzioğlu, M., and Dihkan, M. (2019). “Chapter 21 - The Black Sea—The Past, Present, and Future Status,” in *Coasts and Estuaries*, vol. 2019 . Eds. E. Wolanski, J. W. Day, M. Elliott and R. Ramachandran., 363–375.

Heath, M. R. (2005). Changes in the structure and function of the North Sea fish foodweb 1973–2000, and the impacts of fishing and climate. *ICES J. Mar. Sci.* 62, 847–868. doi: 10.1016/j.icesjms.2005.01.023

Heymans, J. J., Bundy, A., Christensen, V., Coll, M., de Mutsert, K., Fulton, E. A., et al. (2020). The Ocean Decade: A true ecosystem modelling challenge. *Front. Mar. Sci.* 7, 554573. doi: 10.3389/fmars.2020.554573

Hosey, G., Melfi, V., and Ward, S. J. (2020). “Problematic Animals in the Zoo: The Issue of Charismatic Megafauna,” in *Problematic Wildlife II*. Eds. F. Angelici and L. Rossi (Cham: Springer).

ICES (2022). *ICES Stock Information Database* (Copenhagen, Denmark). Available at: https://sid.ices.dk (Accessed 17 Nov 2022).

Jâms, I. B., Windsor, F. M., Poudevigne-Durance, T., Ormerod, S. J., and Durance, I. (2020). Estimating the size distribution of plastics ingested by animals. *Nat. Commun.* 11, 1594. doi: 10.1038/s41467-020-15406-6

Jardim, E., Millar, C. P., Mosqueira, I., Scott, F., Osio, G. C., Ferretti, M., et al. (2015). What if stock assessment is as simple as a linear model? The a4a initiative. *ICES J. Mar. Sci.* 72, 232–236. doi: 10.1093/icesjms/fsu050

Katsanevakis, S., Coll, M., Piroddi, C., Steenbeek, S., Ben Rais Lasram, F., Zenetos, A., et al. (2014). Invading the Mediterranean Sea: biodiversity patterns shaped by human activities. *Front. Mar. Sci.* 1, 32. doi: 10.3389/fmars.2014.00032

Keramidas, I., Dimarchopoulou, D., Ofir, E., Tsikliras, A. C., and Gal, G. (2023). Ecotrophic perspective in fisheries management: A review on EwE models in European marine ecosystems. *Front. Mar. Sci.* 10, 1182921. doi: 10.3389/fmars.2023.1182921

Keramidas, I., Dimarchopoulou, D., and Tsikliras, A. C. (2022). Modelling and assessing the ecosystem of the Aegean Sea, a major hub of the eastern Mediterranean at the intersection of Europe and Asia. *Regional Stud. Mar. Sci.* 56, 102704. doi: 10.1016/j.rsma.2022.102704

Kideys, A. E. (2002). Fall and rise of the Black Sea ecosystem. *Science* 297, 1482–1484. doi: 10.1126/science.1073002

Krause, J. R., Hightower, J. E., Poland, S. J., and Buckel, J. A. (2020). An integrated tagging and catch-curve model reveals high and seasonally-varying natural mortality for a fish population at low stock biomass. *Fisheries Res.* 232, 105725. doi: 10.1016/j.fishres.2020.105725

Laran, S., Authier, M., Blanck, A., Dorémus, G., Falchetto, H., Monestiez, P., et al. (2017). Seasonal distribution and abundance of cetaceans within French waters – Part II: the Bay of Biscay and the English Channel. *Deep Sea Res. Part II* 141, 31–40. doi: 10.1016/j.dsr2.2016.12.012

Lassalle, G., Lobry, J., Le Lo’h, F., Bustamante, P., Certain, G., Delmas, D., et al. (2011). Lower trophic levels and detrital biomass control the Bay of Biscay continental shelf food web: implications for ecosystem management. *Prog. Oceanogr.* 91, 561–575. doi: 10.1016/j.pocean.2011.09.002

Le Marchand, M., Hattab, T., Niquil, N., Albouy, C., Le Loc’h, F., and Lasram, F. B. R. (2020). Climate change in the Bay of Biscay: Changes in spatial biodiversity patterns could be driven by the arrivals of southern species. *Mar. Ecol. Prog. Ser.* 647, 17–31. doi: 10.3354/meps13401

Libralato, S., Caccin, A., and Pranovi, F. (2015). Modeling species invasions using thermal and trophic niche dynamics under climate change. *Front. Mar. Sci.* 2, 29. doi: 10.3389/fmars.2015.00029

Libralato, S., Coll, M., Tempesta, M., Santojanni, A., Spoto, M., Palomera, I., et al. (2010). Food-web traits of protected and exploited areas of the Adriatic Sea. *Biol. Conserv.* 143 (9), 2182–2194. doi: 10.1016/j.biocon.2010.06.002

Link, J. S. (2010). *Ecosystem-Based Fisheries Management: Confronting Tradeoffs* (Cambridge, UK: Cambridge University Press).

Link, J. S., Ihde, T. F., Harvey, C. J., Gaichas, S. K., Field, J. C., Brodziak, J. K. T., et al. (2012). Dealing with uncertainty in ecosystem models: The paradox of use for living marine resource management. *Prog. Oceanogr.* 102, 102–114. doi: 10.1016/j.pocean.2012.03.008

Lorenzen, K. (2022). Size- and age-dependent natural mortality in fish populations: Biology, models, implications, and a generalized length-inverse mortality paradigm. *Fisheries Res.* 255, 106454. doi: 10.1016/j.fishres.2022.106454

Lotze, H. K., Coll, M., and Dunne, J. A. (2011). Historical changes in marine resources, food-web structure and ecosystem functioning in the Adriatic sea, mediterranean. *Ecosystems* 14, 198–222. doi: 10.1007/s10021-010-9404-8

Lotze, H. K., and Worm, B. (2009). Historical baselines for large marine animals. *Trends Ecol. Evol.* 24, 254–262. doi: 10.1016/j.tree.2008.12.004

Luczak, C., Beaugrand, G., Lindley, J. A., Dewarumez, J.-M., Dubois, P. J., and Kirby, R. R. (2012). North Sea ecosystem change from swimming crabs to seagulls. *Biol. Lett.* 8, 821–824. doi: 10.1098/rsbl.2012.0474

Mackinson, S., Deas, B., Beveridge, D., and Casey, J. (2009). Mixed-fishery or ecosystem conundrum? Multispecies considerations inform thinking on long-term management of North Sea demersal stocks. *Can. J. Fish. Aquat. Sci.* 66, 1107–1129. doi: 10.1139/F09-057

Mackinson, S., Platts, M., Garcia, C., and Lynam, C. (2018). Evaluating the fishery and ecological consequences of the proposed North Sea multi-annual plan. *PloS One* 13 (1), e0190015. doi: 10.1371/journal.pone.0190015

McManamay, R. A., and Utz, R. M. (2014). Open-Access databases as unprecedented resources and drivers of cultural change in fisheries science. *Fisheries* 39, 417–425. doi: 10.1080/03632415.2014.946128

Michael-Bitton, G., Gal, G., Corrales, X., Ofir, E., Mordechai, S., and Zemah-Shamir, S. (2022). Economic aspects of fish stock accounting as a renewable marine natural capital: The Eastern Mediterranean continental shelf ecosystem as a case study. *Ecol. Economics* 200, 107539. doi: 10.1016/j.ecolecon.2022.107539

Michailidis, N., Corrales, X., Karachle, P. K., Chartosia, N., Katsanevakis, S., and Sfenthourakis, S. (2019). Modelling the role of alien species and fisheries in an Eastern Mediterranean insular shelf ecosystem. *Ocean Coast. Manage.* 175, 152–171. doi: 10.1016/j.ocecoaman.2019.04.006

Morley, T. I., Fayet, A. L., Jessop, H., Veron, P., Veron, M., Clark, J., et al. (2016). The seabird wreck in the Bay of Biscay and South-Western Approaches in 2014: A review of reported mortality. *Seabird* 29, 22–38.

Moullec, F., Gascuel, D., Bentorcha, K., Guénette, S., and Robert, M. (2017). Trophic models: What do we learn about Celtic Sea and Bay of Biscay ecosystems? *J. Mar. Syst.* 172, 104–117. doi: 10.1016/j.jmarsys.2017.03.008

Niiranen, S., Yletyinen, J., Tomczak, M., Blenckner, T., Hjerne, O., MacKenzie, B. R., et al. (2013). Combined effects of global climate change and regional ecosystem drivers on an exploited marine food web. *Global Change Biol.* 19 (11), 3327–3342. doi: 10.1111/gcb.12309

Ofir, E., Ribas, X., Coll, M., Heymans, J. J., Goren, M., Steenbeek, J. G., et al. (2023). Evaluation of fisheries management policies in the alien species-rich Eastern Mediterranean under climate change. *Front. Mar. Sci.* 10, 1155480. doi: 10.3389/fmars.2023.1155480

Ojaveer, H., Einberg, H., Lehtiniemi, M., Outinen, O., Zaiko, A., Jelmert, A., et al. (2023). Quantifying impacts of human pressures on ecosystem services: Effects of widespread non-indigenous species in the Baltic Sea. *Sci. Total Environ.* 858, 159975. doi: 10.1016/j.scitotenv.2022.159975

Ojaveer, H., Jaanus, A., MacKenzie, B. R., Martin, G., Olenin, S., Radziejewska, T., et al. (2010). Status of biodiversity in the Baltic sea. *PloS One* 5, e12467. doi: 10.1371/journal.pone.0012467

Österblom, H., Hansson, S., Larsson, U., Hjerne, O., Wuff, F., Elmgren, R., et al. (2007). Human-induced trophic cascades and ecological regime shifts in the Baltic sea. *Ecosystems* 10, 877–889. doi: 10.1007/s10021-007-9069-0

Palomares, M. L. D., and Pauly, D. (2021). *SeaLifeBase* (World Wide Web electronic publication). Available at: www.sealifebase.org.

Papantoniou, G., Giannoulaki, M., Stoumboudi, M. T., Lefkaditou, E., and Tsagarakis, K. (2021). Food web interactions in a human dominated Mediterranean coastal ecosystem. *Mar. Environ. Res.* 172, 105507. doi: 10.1016/j.marenvres.2021.105507

Pauly, D. (1980). On the interrelationships between natural mortality, growth parameters, and mean environmental temperatures in 175 fish stocks. *J. du Conseil Int. pour l’Exploration la Mer* 39, 175–192. doi: 10.1093/icesjms/39.2.175

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

Peled, Y., Zemah-Shamir, S., Israel, A., Shechter, M., Ofir, E., and Gal, G. (2020). Incorporating insurance value into ecosystem services assessments: mitigation of ecosystem users’ welfare uncertainty through biological control. *Ecosyst. Serv.* 46, 101192. doi: 10.1016/j.ecoser.2020.101192

Pereira, F., Vasconcelos, P., Moreno, A., and Gaspar, M. B. (2019). Catches of Sepia officinalis in the small-scale cuttlefish trap fishery off the Algarve coast (southern Portugal). *Fisheries Res.* 214, 117–125. doi: 10.1016/j.fishres.2019.01.022

Piroddi, C., Akoglu, E., Andonegi, E., Bentley, J. W., Celic, I., Coll, M., et al. (2021). Effects of nutrient management scenarios on marine food webs: a Pan-European Assessment in support of the Marine Strategy Framework Directive. *Front. Mar. Sci.* 7, 596797. doi: 10.3389/fmars.2021.596797

Piroddi, C., Coll, M., Macias, D., Steenbeek, J., Garcia-Gorriz, E., Mannini, A., et al. (2022). Modelling the Mediterranean Sea ecosystem at high spatial resolution to inform the ecosystem-based management in the region. *Sci. Rep.* 12, 19680. doi: 10.1038/s41598-022-18017-x

Piroddi, C., Colloca, F., and Tsikliras, A. C. (2020). The living marine resources in the Mediterranean Sea Large Marine Ecosystem. *Environ. Dev.* 36, 100555. doi: 10.1016/j.envdev.2020.100555

Prodanov, K., Mikhailov, K., Daskalov, D., Maxim, K., Ozdamar, E., Shlyakhov, V., et al. (1997). Environmental management of fish resources in the Black Sea and their rational exploitation. *Stud. Reviews 68 GFCM* 1997, 178.

Püts, M., Taylor, M., Núñez-Riboni, I., Steenbeek, J., Stäbler, M., Möllmann, C., et al. (2020). Insights on integrating habitat preferences in process-oriented ecological models – a case study of the southern North Sea. *Ecol. Model.* 431, 109189. doi: 10.1016/j.ecolmodel.2020.109189

Quante, M., Colijn, F., Bakker, J. P., Härdtle, W., Heinrich, H., Lefebvre, C., et al. (2016). *Introduction to the assessment—Characteristics of the region*. (Springer). doi: 10.1007/978-3-319-39745-0_1

Quetglas, A., de Mesa, A., Ordines, F., and Grau, A. (2010). Life history of the deep-sea cephalopod family Histioteuthidae in the western Mediterranean. *Deep Sea Res. Part I: Oceanographic Res. Papers* 57, 999–1008. doi: 10.1016/j.dsr.2010.04.008

Ravier, C., and Fromentin, J.-M. (2004). Are the long-term fluctuations in Atlantic bluefin tuna (Thunnus thynnus) population related to environmental changes? *Fish. Oceanogr.* 13, 145–160. doi: 10.1111/j.1365-2419.2004.00284.x

Reusch, T. B. H., Dierking, J., Andersson, H. C., Bonsdorff, E., Carstensen, J., Casini, M., et al. (2018). The baltic sea as a time machine for the future coastal ocean. *Sci. Adv.* 4, eaar8195. doi: 10.1126/sciadv.aar8195

Rodriguez-Perez, A., Tsikliras, A. C., Gal, G., Steenbeek, J., Falk-Andersson, J., and Heymans, S. J. J. (2023). Using ecosystem models to inform ecosystem-based fisheries management in Europe: a review of the policy landscape and related stakeholder needs. *Front. Mar. Sci.* 10, 1196329. doi: 10.3389/fmars.2023.1196329

Rosa, S., Pansera, M., Granata, A., and Guglielmo, L. (2013). Interannual variability, growth, reproduction and feeding of Pelagia noctiluca (Cnidaria: Scyphozoa) in the Straits of Messina (Central Mediterranean Sea): Linkages with temperature and diet. *J. Mar. Syst.* 111–112, 97–107. doi: 10.1016/j.jmarsys.2012.10.001

Saad, A., and Mahfoud, I. (2022). "Marine mammals in Syria," in *Marine mammals*. Ed. H. A. E. Kaoud (London: IntechOpen).

Sardà, F., Calafat, A., Flexas, M. M., Tselepides, A., Canals, M., Espino, M., et al. (2004). An introduction to Mediterranean deep-sea biology. *Scientia Marina* 68 (S3), 7–38. doi: 10.3989/scimar.2004.68s37

Savva, I., Chartosia, N., Antoniou, C., Kleitou, P., Georgiou, A., Stern, N., et al. (2020). They are here to stay: the biology and ecology of lionfish (Pterois miles) in the Mediterranean Sea. *J. Fish Biol.* 97, 148–162. doi: 10.1111/jfb.14340

Scarcella, G., Angelini, S., Armelloni, E. N., Costantini, I., De Felice, A., Guicciardi, S., et al. (2022). The potential effects of COVID-19 lockdown and the following restrictions on the status of eight target stocks in the Adriatic Sea. *Front. Mar. Sci.* 9, 920974. doi: 10.3389/fmars.2022.920974

Scotti, M., Opitz, S., MacNeil, L., Kreutle, A., Pusch, C., and Froese, R. (2022). Ecosystem-based fisheries management increases catch and carbon sequestration through recovery of exploited stocks: The western Baltic Sea case study. *Front. Mar. Sci.* 9, 879998. doi: 10.3389/fmars.2022.879998

Shabtay, A., Portman, M., Ofir, E., Carmel, Y., and Gal, G. (2018). Using ecological modelling in marine spatial planning to enhance ecosystem-based management. *Mar. Policy* 95, 14–23. doi: 10.1016/j.marpol.2018.06.018

Shiganova, T. A. (1998). Invasion of the Black Sea by the ctenophore *Mnemiopsis leidyi* and recent changes in pelagic community structure. *Fish. Oceanogr.* 7, 305–310. doi: 10.1046/j.1365-2419.1998.00080.x

Spitz, J., Ridoux, V., Trites, A. W., Laran, S., and Authier, M. (2018). Prey consumption by cetaceans reveals the importance of energy-rich food webs in the Bay of Biscay. *Prog. Oceanogr.* 166, 148–158. doi: 10.1016/j.pocean.2017.09.013

Stäbler, M., Kempf, A., and Temming, A. (2018). Assessing the structure and functioning of the southern North Sea ecosystem with a food-web model. *Ocean Coast. Manage.* 165, 280–297. doi: 10.1016/j.ocecoaman.2018.08.017

Stasolla, G., Tricarico, E., and Vilizzi, L. (2021). Risk screening of the potential invasiveness of non-native marine crustacean decapods and barnacles in the Mediterranean Sea. *Hydrobiologia* 848, 1997–2009. doi: 10.1007/s10750-020-04432-6

Stergiou, K. I., Somarakis, S., Triantafyllou, G., Tsiaras, K. P., Giannoulaki, M., Petihakis, G., et al. (2016). Trends in productivity and biomass yields in the Mediterranean Sea Large Marine Ecosystem during climate change. *Environ. Dev.* 17, 57–74. doi: 10.1016/j.envdev.2015.09.001

Stergiou, K. I., and Tsikliras, A. C. (2006). Underrepresentation of regional ecological research output by bibliometric indices. *Ethics Sci. Environ. Politics* 2006, 15–17. doi: 10.3354/esep006015

Tecchio, S., Coll, M., Christensen, V., Company, J. B., Ramírez-Llodra, E., and Sardà, F. (2013). Food web structure and vulnerability of a deep-sea ecosystem in the NW Mediterranean Sea. *Deep Sea Res. Part I: Oceanographic Res. Papers* 75, 1–15. doi: 10.1016/j.dsr.2013.01.003

Then, A. Y., Hoenig, J. M., Hall, N. G., and Hewitt, D. A. (2015). Evaluating the predictive performance of empirical estimators of natural mortality rate using information on over 200 fish species. *ICES J. Mar. Sci.* 72, 82–92. doi: 10.1093/icesjms/fsu136

Topaloglu, B., and Alper, E. (2014). Updated checklist of sponges (Porifera) along the coasts of Turkey. *Turkish J. Zool.* 38, 665–676. doi: 10.3906/zoo-1405-79

Tsagarakis, K., Coll, M., Giannoulaki, M., Somarakis, S., Papaconstantinou, C., and Machias, A. (2010). Food web traits of the North Aegean Sea ecosystem (Eastern Mediterranean) and comparison with other Mediterranean ecosystems. *Estuarine Coast. Shelf Sci.* 88 (2), 233–248. doi: 10.1016/j.ecss.2010.04.007

Tsikliras, A. C., Antonopoulou, E., and Stergiou, K. I. (2010). Spawning period of Mediterranean marine fishes. *Rev. Fish Biol. Fisheries* 20, 499–538. doi: 10.1007/s11160-010-9158-6

Tsikliras, A. C., Coro, G., Daskalov, G., Gremillet, D., Scotti, M., and Sylaios, G. (2023). A need for a paradigm shift from anthropocentric to ecocentric fisheries management. *Front. Mar. Sci.* 10

Tsikliras, A. C., and Dimarchopoulou, D. (2021). Filling in knowledge gaps: length-weight relations of 46 uncommon sharks and rays in the Mediterranean Sea. *Acta Ichthyologica Piscatoria* 51, 249–255. doi: 10.3897/aiep.51.e65858

Tsikliras, A. C., Dinouli, A., Tsiros, V.-Z., and Tsalkou, E. (2015). The Mediterranean and Black Sea fisheries at risk from overexploitation. *PloS One* 10, e0121188. doi: 10.1371/journal.pone.0121188

Tsikliras, A. C., and Froese, R. (2019). "Maximum sustainable yield," in *Encyclopedia of ecology*, *2nd ed.*, vol. 1 . Ed. B. Fath (Oxford: Elsevier), 108–115.

Tsikliras, A. C., and Stergiou, K. I. (2014). Size at maturity of Mediterranean marine fishes. *Rev. Fish Biol. Fisheries* 24, 219–268. doi: 10.1007/s11160-013-9330-x

Tsikliras, A. C., Touloumis, K., Pardalou, A., Adamidou, A., Keramidas, I., Orfanidis, G., et al. (2021). Status of 74 non-commercial fish and invertebrate stocks in the Aegean Sea using abundance and resilience. *Front. Mar. Sci.* 7, 578601. doi: 10.3389/fmars.2020.578601

Ugland, K. (1976). Population studies on eastern North Atlantic minke whales, Balaenoptera acutorostrata Lacepede. *Rep. Int. Whaling Commission (Sci. Rep.)* 26, 366–381.

Ulman, A., Yildiz, T., Demirel, N., Canak, O., Yemişken, E., and Pauly, D. (2021). The biology and ecology of the invasive silver-cheeked toadfish (Lagocephalus sceleratus), with emphasis on the Eastern Mediterranean. *NeoBiota* 68, 145–175. doi: 10.3897/neobiota.68.71767

Vilas, D., Coll, M., Corrales, X., Steenbeek, J., Piroddi, C., Macias, D., et al. (2021). Current and potential contributions of the Gulf of Lion Fisheries Restricted Area to fisheries sustainability in the NW Mediterranean Sea. *Mar. Policy* 123, 104296. doi: 10.1016/j.marpol.2020.104296

Zaitsev, Y. P. (1992). Recent changes in the trophic structure of the Black Sea. *Fish. Oceanogr.* 1, 180–189. doi: 10.1111/j.1365-2419.1992.tb00036.x

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Using ecosystem models to inform ecosystem-based fisheries management in Europe: a review of the policy landscape and related stakeholder needs

Ana Rodriguez-Perez[1], Athanassios C. Tsikliras[2], Gideon Gal[3], Jeroen Steenbeek[4], Jannike Falk-Andersson[5] and Johanna J. Heymans[1]*

[1]European Marine Board, Oostende, Belgium, [2]Laboratory of Ichthyology, School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece, [3]Kinneret Limnological Laboratory, Israel Oceanographic and Limnological Research, Migdal, Israel, [4]Ecopath International Initiative, Barcelona, Spain, [5]Norwegian Institute for Water Research, Oslo, Norway

EDITED BY
Marco Scotti,
National Research Council (CNR), Italy

REVIEWED BY
Yunne-Jai Shin,
Institut de Recherche pour le Développement (IRD), France
Howard Townsend,
National Marine Fisheries Service (NOAA), United States

*CORRESPONDENCE
Johanna J. Heymans
sheilaheymans@yahoo.com

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The need to implement an ecosystem-based fisheries management (EBFM) is enshrined in numerous regulations and strategies, at both global and European level. In practice, it is challenging to implement EBFM because it requires a complex evaluation of interlinked management effects and environmental and climate forcing on multi-species interactions, habitat status and human activities. Ecosystem models are one of the most critical research tools to inform EBFM, because they can integrate a wide variety of data, examine multiple and complex ecosystem interactions, and can make forecasts based on specific management scenarios. However, despite clear progress in marine ecosystem modelling, many models do not address policy goals and targets, which hinders uptake in policy. In this paper, we review the global and European policies and implementing bodies which directly or indirectly have a repercussion on the implementation of EBFM. Moreover, we highlight specific stakeholder needs related to the implementation of EBFM in European waters, which ecosystem models could help address. We review the policy commitments that drive these needs and the concerns raised by stakeholders during a survey and dedicated workshop. Key topics of concern were effects of climate change; bycatch; protected areas/fisheries restricted areas; and reducing the impacts of trawling. Stakeholders also provided specific questions related to these topics which ecosystem models could help address. Scenario and data results visualizations, as well as specific barriers in using the results of ecosystem models for decision-making are also discussed. A close involvement of stakeholders in scenario development and in designing graphical outputs is important, and can help overcome some of the main barriers that can hinder uptake of models and scenarios, including a lack of understanding of the benefits and limits of ecosystem models; insufficient involvement and interaction with stakeholders; and inadequate characterization of uncertainties.

KEYWORDS

ecosystem models, ecosystem-based fisheries management, policy, implementation, stakeholder needs

1 Introduction

Fishing impacts marine organisms and ecosystems directly and indirectly (Jennings and Kaiser, 1998), affecting biodiversity, habitats and food web structure and functions, from local populations and communities to entire ecosystems (Jackson et al., 2001). Unsustainable fishing, which has been documented to occur globally (Pauly et al., 1998; Myers and Worm, 2003), results in declining catches and exploitation beyond safe biological limits (Christensen et al., 2003; Froese et al., 2018), and has altered ecosystem structure and function (Moullec et al., 2023). The traditional single species fisheries management and single stock advice neglects explicit multispecies interactions (Vinther et al., 2004), lacks environmental/climatic forcing and does not implicitly assess the socioeconomic impact of fishing (Dolan et al., 2016). Thus, a pressing need for managing fisheries in the context of an ecosystem (an ecosystem-based fisheries management, EBFM) emerged with a more systemic and multi-sector perspective. Applying EBFM requires moving from traditional single-species management to a more complex approach, which includes evaluating the interlinked effects of management and environmental forcing on multi-species interactions, habitat status and human activities (Garcia, 2003). EBFM has to acknowledge the effects of fishing on the whole ecosystem, including the human socio-ecologic system, and it should also help maintain resilient and sustain ecosystem services in the face of changing climate (Fu et al., 2013).

Numerous European and global policies require the implementation of EBFM. For instance, in the EU, the Common Fisheries Policy (CFP) explicitly states that an *"ecosystem-based approach to fisheries management needs to be implemented"* and both the Marine Strategy Framework Directive (MSFD) and the Marine Spatial Planning Directive (MSPD) endorse an ecosystem-based approach to management. Internationally, the UN Food and Agriculture Organisation (FAO) strongly promotes an ecosystem approach to fisheries and has produced numerous publications as guidelines (e.g. Garcia et al., 2003; FAO, 2005; FAO, 2008; Carocci et al., 2009; Staples and Funge-Smith, 2009). However, in practice implementation of EBFM requires interdisciplinarity, including applied science, modelling, and analysis of diverse streams of information, making it difficult to implement (Townsend et al., 2019).

Ecosystem models are a way of representing whole ecological systems, and are also able to integrate economic and social data (Heymans et al., 2018; Steenbeek et al., 2021a). They integrate a wide stream of information that can be used for testing the ecological, economic and social consequences of implementing specific management scenarios. These management simulations can be both retrospective (i.e. hindcasts) or for future scenarios. Ecosystem models have therefore been highlighted as one of the most critical research tools to inform EBFM (Townsend et al., 2019). However, despite clear capability and progress in marine ecosystem modelling, many models are designed to answer scientific, not policy questions, which hinders uptake in policy (Heymans et al., 2018). Ecosystem models that are designed to address policy questions need to be linked to policy goals and targets (e.g. Ofir et al., 2022). In order to inform EBFM, it is therefore important to understand the relevant policy landscape and the needs of related stakeholders.

One of the principal aims of the EcoScope project (EcoScope, 2021) is to use ecosystem modelling as a tool to assist in the implementation of EBFM and, within the ecosystem modelling framework and in parallel with research questions, to co-design the modelling scenarios with the relevant stakeholders in order to address policy questions. Ecosystem models will be available through an interactive platform, allowing users to run modelling scenarios and obtain easy to understand results. In the frame of the project, a survey and a foresight workshop were conducted with the involvement of key stakeholders to understand the main needs, challenges and barriers in implementing EBFM through the help of ecosystem modelling.

The aim of this paper is to distil critical policy-related needs relevant to EBFM, that can be addressed using ecosystem modelling. To this end, the paper has three main sections: (1) a review of the global and European policy landscape (including policies and implementing bodies) which directly or indirectly have a repercussion on the implementation of EBFM; (2) a review of stakeholder needs for implementing EBFM with the help of ecosystem modelling, including a review of the policy commitments that drive these needs and the concerns raised by stakeholders during a survey and dedicated workshop; and (3) a discussion on the main barriers and enablers that hinder or support the uptake of ecosystem model results in decision making.

2 EBFM policy landscape

2.1 International policy landscape

Internationally, the United Nation Convention on the Law of the Sea (UNCLOS) (UN General Assembly, 1982) is the framework under which all activities in the ocean must be carried out, including the conservation and sustainable use of marine resources. It sets limits to various maritime zones (i.e. territorial waters, Exclusive Economic Zone, continental shelf and high seas) and recognises the rights of coastal states to control fish harvests in adjacent waters. EU fishing activities take place under the framework of UNCLOS and the rights and duties of states with respect to the use of ocean space and resources are defined therein.

In 1995, UNCLOS was supplemented by the UN Fish Stocks Agreement (UNFSA) (UN General Assembly, 1995) on highly migratory and straddling fish stocks. UNFSA establishes a set of rights and obligations for States to conserve and manage fish stocks and associated species, as well as to protect biodiversity in the marine environment. Regional Fisheries Management Organisations (RFMOs) are the mechanism through which States should cooperate internationally to fulfil their obligation to manage and conserve fish stocks in the high seas. RFMOs are made up of countries that share a practical and/or financial interest in managing and conserving fish stocks in a particular region. While some RFMOs have a purely advisory role, most set catch and fishing effort limits, technical measures and control regulations. Examples of RFMOs that set fishing regulations are the General Fisheries Commission for the Mediterranean (GFCM) and the International Commission for the Conservation of Atlantic Tunas (ICCAT) (see section 2.2.2 for more details). RFMO member countries must adopt management measures implemented by RFMOs and must transpose these measures into law (if not already covered), applicable to all vessels using their countries flag (Popescu, 2019).

The main UN body relevant for EBFM is the Food and Agriculture Organisation (FAO), which leads international efforts to defeat hunger and aims to make fisheries more productive and sustainable. FAO plays a leading role in international fisheries policy, including through the Committee of Fisheries (COFI), which review and address issues and challenges related to fisheries. COFI has fostered the development and adoption of several binding- and non-binding agreements, such as the International Plans of Action (IPOA). These action plans are implemented in close collaboration with intergovernmental organisations (e.g. CITES, CMS, IUCN and other NGOs) and include: Reducing Incidental Catch of Seabirds in Longline Fisheries; Conservation and Management of Sharks; Management of Fishing Capacity; and Prevent, Deter, and Eliminate Illegal, Unreported and Unregulated Fishing (IUU) (FAO, 1999; FAO, 2001). FAO strongly promotes the ecosystem approach to fisheries, including the application of modelling tools, and has produced a number of publications on the topic (Garcia et al., 2003; FAO, 2005; Plagányi, 2007; FAO, 2008; Carocci et al., 2009; Staples and Funge-Smith, 2009).

The Convention on Biological Diversity (CBD, 1992) is an international treaty for the conservation and sustainable use of biological diversity, and for fair and equitable sharing of the benefits arising from utilising genetic resources. Member countries implement CBD objectives through National Biodiversity Strategies and Action Plans (NBSAPs). The ecosystem approach, adopted as the primary framework for action since 1995, is a central principle in the implementation of the CBD. In 2022, the Kunming-Montreal Global Biodiversity Framework (CBD, 2022) replaced the Aichi Biodiversity Targets for 2011-2020. This new Global Biodiversity Framework includes four goals and 23 targets for 2030. The most relevant targets for EBFM are: having restoration completed or underway on at least 30% of degraded marine and coastal ecosystems (target 2), protecting at least 30% of coastal areas and the ocean (target 3), and ensuring that the harvest of wild species is done using an ecosystem-approach, preventing overexploitation and minimizing impacts on non-target species and ecosystems (target 5). In March 2023, the UN High Seas Treaty to protect the ocean in areas beyond national jurisdiction (also known as the Biodiversity Beyond National Jurisdiction – BBNJ - agreement) was finalised. This treaty provides the legal framework to establish large-scale marine protected areas (MPAs) on the high seas, which will be necessary to meet the global commitment of the Kunming-Montreal Global Biodiversity Agreement.

The Bonn Convention on Migratory Species (CMS, 1979) is a UN treaty for the conservation and sustainable use of migratory animals and their habitats. CMS brings together the countries through which migratory species pass and lays the foundation for internationally coordinated conservation measures. The arrangements under CMS range from legally binding Agreements to less-formal instruments, such as Memoranda of Understanding (MoU). To date, 19 international MoUs and 7 Agreements have been signed under CMS, of which the following three are particularly relevant for EBFM: Memorandum of Understanding on the Conservation of Migratory Sharks (MoU Sharks), Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS) and Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic Area (ACCOBAMS).

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES, 1973) aims to ensure that international trade of wild animals and plants does not threaten the survival of these species in the wild. Although CITES was signed in 1973, marine species have only recently been included in CITES Appendices, including several shark and sea cucumber species. CITES has been criticised for insufficiently regulating marine fish species, yet it could be a relevant and appropriate instrument for promoting sound marine fisheries management (Vincent et al., 2014). To support the implementation of CITES in the fisheries context, FAO published a handbook in 2020 on implementing CITES through national fisheries legal framework (Nakamura and Kuemlangan, 2020) in collaboration with the CITES Secretariat.

2.2 European policy landscape

The European Union (EU) is advancing towards the goal of managing fisheries under an ecosystem approach (Ramirez-Monsalve et al., 2021). But similar to the international policy landscape, there is separation between environmental and fisheries regulations and advisory bodies. This dichotomous policy landscape has been criticised as an impediment in implementing EBFM (Ramirez-Monsalve et al., 2021). The following section provides an overview of this divided EU policy landscape.

2.2.1 Fisheries regulations and advisory bodies

The European Commission (EC), founded in 1958 as the executive branch of the European Union (EU), promotes the general interest of the EU by proposing and enforcing legislation as well as by implementing policies and the EU budget. The EC is divided into departments (Directorates-General, DGs) that handle a set of specific responsibilities. The most relevant DGs for EBFM implementation are the Directorate-General for Maritime Affairs and Fisheries (DG MARE), the Directorate-General for Environment (DG ENV), and the Directorate-General for Research and Innovation (DG RTD). The responsibilities of DG MARE include to: (i) ensure that the ocean resources are used sustainably and that coastal communities and the fishing sector have a prosperous future; (ii) promote maritime policies and stimulate a sustainable blue economy; and (iii) promote ocean governance at an international level. DG ENV has the mandate to protect, preserve and improve Europe's environment for present and future generations. It develops and carries out the Commission's policies on the environment, including on the marine environment. DG RTD is responsible for EU policy on research, science, and innovation, and funds science and research, including on EBFM, under the EU framework programmes for research and innovation, of which the most recent is called Horizon Europe.

The Common Fisheries Policy (CFP) of the EU lays out the rules for sustainably managing European fishing fleets and conserving fish stocks. Under the CFP, all European fishing fleets have equal access to EU waters and fishing grounds, and the EU has "exclusive competence" for the conservation of marine biological resources. This means that only the EU is able to legislate and adopt binding regulations concerning the common fisheries resources. Member States cannot self-legislate on these matters and the legislation and regulations implemented by the EU through the CFP are directly applicable in Member States. The CFP applies to management of fisheries in EU waters and to international EU fisheries relations and bilateral fisheries agreements signed with third party countries (Popescu, 2019). Since its introduction in 1970, the CFP has since been reformed several times. The 1983 reform introduced the "quota" system of catch limits shared among Member States (i.e., Total Allowable Catches). The 1992 reform endeavoured to remedy the serious imbalance between fleet capacity and catch potential, but the measures did not halt overfishing (Breuer, 2022). The latest 2013 reform (Regulation EU 1380/2013, 2013) introduced the target to achieve exploitation of all stocks at Maximum Sustainable Yield (MSY) by 2020, and to implement an ecosystem-based approach to fisheries management (Regulation EU 1380/2013, article 2). EBFM is defined in this regulation as: "*an integrated approach to managing fisheries within ecologically meaningful boundaries which seeks to manage the use of natural resources, taking account of fishing and other human activities, while preserving both the biological wealth and the biological processes necessary to safeguard the composition, structure and functioning of the habitats of the ecosystem affected, by taking into account the knowledge and uncertainties regarding biotic, abiotic and human components of ecosystems*" (Regulation EU 1380/2013, article 4). The 2013 reform made the adoption of Multiannual Management Plans (MAPs)[1] a priority and these plans stress the need to implement EBFM. This reform also introduced landing obligations, fleet capacity ceilings and the regionalisation of decision-making. The landing obligation, phased in by 2019, aims to end the practice of discarding fish back into the sea, and the fleet capacity ceilings aim to ensure a balance between fishing capacity and fishing opportunities over time. The regionalisation of decision-making enables Member States to adopt conservation measures based on joint recommendations to the EC. Joint recommendations have to be submitted by all Member States with a management interest in the area (Reg.1380/2013, article 11) through so-called Member State Regional Groups (see below for more details). Although the 2013 reform has improved the status of some stocks, it has failed to meet the goal of ending overfishing by 2020 (Froese et al., 2021).

The latest CFP reform also brought an overhaul of the technical measures, which had accumulated over time to form a complicated regulatory structure. The new Technical Measures Regulation (Regulation EU 2019/1241, 2019) are a set of rules stipulating how, where and when fishers may fish. These can differ from one basin to another, in accordance with regional conditions. The measures include regulations on minimum landing sizes, minimum mesh sizes, specifications for design and use of gears, and closed areas and seasons. The technical measures regulation aims to de-centralise the management of technical features to the region.

Catch quotas are the main mechanism used to regulate fisheries in the North East Atlantic. In contrast, the main strategy in the Mediterranean is the control of fishing effort and setting specific technical measures (e.g. gear regulation, establishment of a minimum conservation reference size, and selective closure of areas and seasons) (Cardinale et al., 2017). In this context, the Mediterranean Regulation (EC Council Regulation 1967/2006, 2006) provides a set of additional technical measures for the Mediterranean, including provisions on fisheries restriction in protected habitats (e.g. prohibition to fish above seagrass beds with gears that can damage the beds), establishing protected areas, restricting certain fishing activities (such as explosives and toxic substances), and establishing minimum mesh sizes and minimum conservation sizes of marine organisms.

Several actors relevant for EBFM are expected to provide advice in the EU to ensure that fisheries management measures are founded on scientific advice (Figure 1). The Scientific, Technical and Economic Committee for Fisheries (STECF) is a group of

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[1] To date, four multiannual management plans have been adopted on: (i) stocks of cod, herring and sprat in the Baltic Sea (Regulation EU 2016/1139, 2016); (ii) demersal stocks in the North Sea (Regulation EU 2018/973, 2018), (iii) stocks in Western Waters (Regulation EU 2019/472, 2019), and (iv) stocks in the western Mediterranean Sea (Regulation EU 2019/1022, 2019).

fisheries experts appointed by DG MARE for three years, who provide advice to the EC on fisheries management. The EC is the only body which can request advice from STECF and STECF also provides the EC opinions on its own initiative. The EC can consult STECF on any matter relating to marine and fisheries biology, fishing gear technology, fisheries economics, fisheries governance, ecosystem effects of fisheries, or similar topics. In many cases STECF will convene an expert working group to carry out technical analysis and compile an evidence report from which the STECF plenary can draw its advice. Where necessary, STECF also consults and collaborates with other bodies (Montana et al., 2020; Ramirez-Monsalve et al., 2021).

The Joint Research Centre (JRC) is the EC's science and knowledge service, which employs scientists to carry out research to provide independent scientific advice and to support implementation of EU policy, such as the CFP. The JRC acts as the secretariat of STECF and coordinates its scientific advice process by collecting, quality-checking, and analysing the fisheries data from EU Member States and making them available to STECF.

The International Council for the Exploration of the Sea (ICES) is a key EC scientific advisory body, which supports the implementation of the CFP in the North-East Atlantic, North Sea and Baltic Sea. ICES provides scientific assessments and advice upon request to the EC and public authorities, including national governments and Regional Sea Conventions (see section 2.2.2) for the stocks of the North-East Atlantic and the Baltic Sea on: (i) fishing quotas or fishing opportunities; (ii) fisheries overviews and advice on mixed fisheries, multi-species interactions, and by-catch issues; and (iii) ecosystem overviews, where primary pressures from anthropogenic activities are identified and assessed for each of the ICES ecoregions. The last two components represent the scientific basis for ecosystem-based decisions in ICES. ICES advice is produced through a four-stage framework, of request formulation, knowledge synthesis, peer review, and advice production (see ICES, 2020b for more details). The advice is provided on a client-contractor basis, where the client pays for the service (Montana et al., 2020; Ramirez-Monsalve et al., 2021).

In the Mediterranean and the Black Sea, scientific stock assessments and advice are provided by two Regional Fisheries Management Organisations (RFMOs): The General Fisheries Commission for the Mediterranean (GFCM), established under FAO, and the International Commission for the Conservation of Atlantic Tunas (ICCAT). GCFM's main objective is to ensure the conservation and sustainable use of living marine resources in the Mediterranean and in the Black Sea. GFCM's Scientific Advisory Committee of Fisheries (SAC) is responsible for assessing all commercial species (except for tuna or tuna-like species) in the Mediterranean and the Black Sea, and providing scientific stock assessment advice to STECF. Efforts to include EBFM aspects within the scientific advice provided by SAC are reflected in its SAC Subcommittee on Marine Environment and Ecosystem (SCMEE) (e.g. see SCMEE, 2005) to implement EBFM within the GFCM geographical area. GFCM has also created a series of working groups to address environmental aspects associated with fishing, including: impacts on elasmobranch, monk seal, red coral and sea turtles; minimising impacts of longline fishing on seabirds;

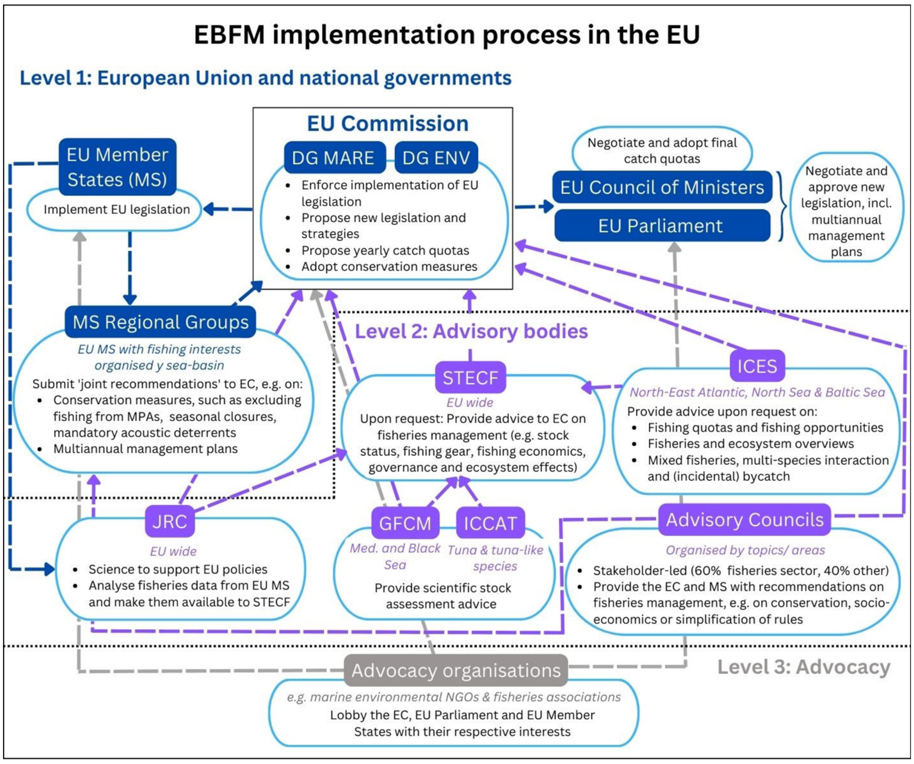

FIGURE 1
EBFM implementation process in the EU, including the decision-making process and the influencing power of the different bodies.

and implementation of Marine Strategy Framework Directive (MSFD) indicators, MPAs, and vulnerable marine ecosystems (Ramirez-Monsalve et al., 2021).

The International Commission for the Conservation of Atlantic Tunas (ICCAT) compiles fisheries statistics from its members and other entities, coordinates research, including stock assessments and develops scientific-based management advice. Scientific assessments of all tuna or tuna-like species in EU waters is provided to STECF by the Standing Committee on Research and Statistics (SCRS) of ICCAT. ICCAT has been developing the scientific foundations for EBFM since 2005, with a focus on developing an EBFM understanding and EBFM tools (Ramirez-Monsalve et al., 2021). ICCAT has also described the status and trends of selected ecosystem indicators, and has reviewed five tuna RFMOs in terms of their application of EBFM. However, the application of the EBFM is still “patchy” in ICCAT, with challenges relating to the understanding of the EBFM concept and the requirements for its implementation (Ramirez-Monsalve et al., 2021).

The two final players in the CFP advisory landscape are Member States Regional Groups and Advisory Councils, which since the 2013 CFP reform have been given greater control to influence fisheries management. Member States Regional Groups (MSRGs) are EU Member States that are organised by sea basin to cooperate and submit joint recommendations (e.g., for conservation measures or multiannual management plans). The joint recommendation procedure enables Member States with fishing interests in an area to collaborate for proposing management measures, such as excluding fishing from an MPA or implementing seasonal closures. Joint recommendations have to be accompanied by relevant information, including the rationale of measures, scientific evidence in support and details on practical implementation and enforcement. While the EC has the final decision-making power on whether to adopt the proposed measures, submitting a joint recommendation is a pre-requisite for adopting any conservation measures. In practice, this means that to implement any conservation measure, all Member States that fish in that area must agree on management measures, which can take many years and therefore frequently hinders implementation of effective management measures. Since MSRGs operate at the scale of regional marine ecosystems and they are the ones submitting joint recommendations, they are very relevant for implementing EBFM. However, they have no legal requirements for transparency and stakeholder involvement, and some MSRGs have been criticised for not sufficiently integrating the advice provided by the Advisory Councils (Ramirez-Monsalve et al., 2021).

Advisory Councils (ACs) are stakeholder-led organisations that provide the EC with recommendations on fisheries management matters related to the CFP. Each AC has a special focus, for instance regional seas ACs (e.g. the Mediterranean AC MEDAC, the Baltic Sea AC BSAC, the Black Sea AC BLSAC, and the North Sea AC NSAC) and topic-related AC (e.g. Pelagic stocks AC). Recommendations of ACs include advice on conservation and socio-economic aspects of management, as well as advice on simplification of rules. ACs are composed of 60% of fisheries sector representatives and 40% of other interested groups, such as environmental organisations and consumer groups. They are considered an important mechanism for the implementation of EBFM in Europe, because they provide experienced-based information and a platform to discuss social, economic and ecological outcomes for fisheries (Ramirez-Monsalve et al., 2016). Before 2013, ACs provided their advice directly to the EC, but since the 2013 CFP reform ACs provide their advice mainly to the MSRGs (Ramirez-Monsalve et al., 2021).

2.2.2 Environmental and maritime legislation and bodies

Other than the CFP the marine environmental legislation of the EU is composed mainly of directives, which are not automatically applicable to the Member States, but require transposition into national law. These directives must become law in the Member States by a certain, specified deadline. Therefore, for each of the directives mentioned below, equivalent national level legislation exists in the EU Member States.

The Marine Strategy Framework Directive (MSFD) (Directive 2008/56/EC, 2008) is Europe’s most holistic directive on protecting the marine environment. After the CFP, it is also the second most important European Directive in the context of EBFM. The MSFD was established in 2008 and has the goal of achieving good environmental status (GES) in European Waters, with an original deadline of 2020. The MSFD stipulates that GES is to be achieved through an ecosystem approach to the management of human activities (article 3). The directive sets out 11 descriptors (Figure 2), which describe what the environment will look like when GES has been achieved. Four of these descriptors (D) are directly associated with EBFM, namely: D1 - biodiversity is maintained; D3 - the population of commercial fish species is healthy; D4 - elements of food webs ensure long-term abundance and reproduction; and D6 - the seafloor integrity ensures functioning of the ecosystem. The Directive also stipulates that a coherent and representative network of protected areas must be created. In order to achieve GES, each Member State is required to develop a national Marine Strategy, i.e., a strategy for its marine waters. These Marine Strategies must be kept up to date and reviewed every six years.

The Water Framework Directive (WFD) (Directive 2000/60/EC, 2000) is closely linked to the MSFD. It sets the goal of achieving Good Ecological Status (GecS, Figure 3) and Good Chemical Status, for all EU surface and groundwaters. The WFD applies to rivers, lakes, estuaries, groundwater, and coastal marine waters. For the marine environment, the WFD specifically covers marine territorial waters (12 nautical miles) for aspects of chemical quality, and marine coastal waters (up to 1 nautical mile) for aspects of ecological quality. Similar to the MSFD, Member states prepare River Basin Management Plans that require the implementation of measures to contribute to the achievement of Good Ecological Status and Good Chemical Status of water bodies by 2027. These plans are implemented and reviewed on a six-year cycle. The actions taken in these plans aim to reduce marine pollution from land-based sources and to protect ecosystems in coastal and estuarine waters, which are vital habitats for many marine species.

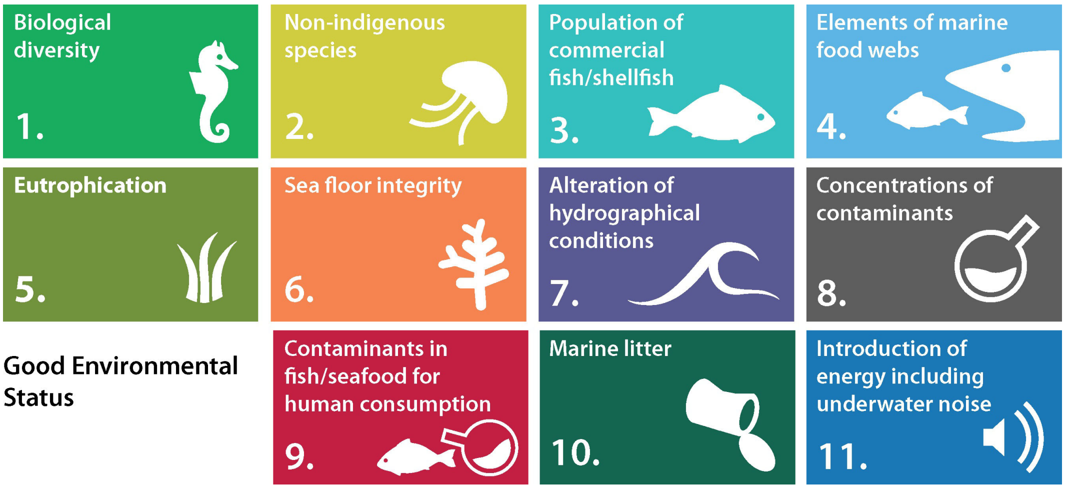

FIGURE 2
The 11 qualitative descriptors for determining good environmental status as presented in the MSFD (Directive 2008/56/EC). Credit: OSPAR Commission (2017).

The Birds Directive and Habitats Directive aim to achieve Favourable Conservation Status (Figure 3) of habitats and species listed in the directives. This includes all seabird species that occur in the EU (under the Birds Directive, Directive 2009/147/EC, 2009) and the habitats and species listed in the Habitats Directives (Council Directive 92/43/EEC, 1992), including nine broad marine habitats, all cetaceans and several marine turtle species. To protect these species and habitats Member States must designates Special Protection Areas (SPAs) (for birds) and Sites of Community Importance (SCIs)/Special Areas of Conservation (SACs) (for species and habitats listed in the Habitats Directive). The SPAs designated under the Birds Directive and the SCIs and SACs designated under the Habitats Directive together make up the Natura 2000 network. The Natura 2000 network includes more than 3,000 marine Natura 2000 sites, which cover almost 10% of the EU marine area (European Commission, 2018). Reporting under the Habitats and Birds Directives requires Member States to monitor the habitats and species listed in the Annexes and send reports to the Commission every six years.

ASCOBANS and ACCOBAMS are agreements under the Convention on Migratory Species (CMS, see section 2.1), for the protection of small cetaceans, such as dolphins and porpoises. ASCOBANS promotes cooperation between countries to achieve a favourable conservation status of small cetaceans in the Baltic, North-East Atlantic, Irish and North Seas, while ACCOBAMS does the same in the Black Sea, Mediterranean Sea, and contiguous Atlantic area. These agreements link to the need of strictly protecting cetaceans in the EU to achieve and maintain a “favourable conservation status” as prescribed in the Habitats Directive.

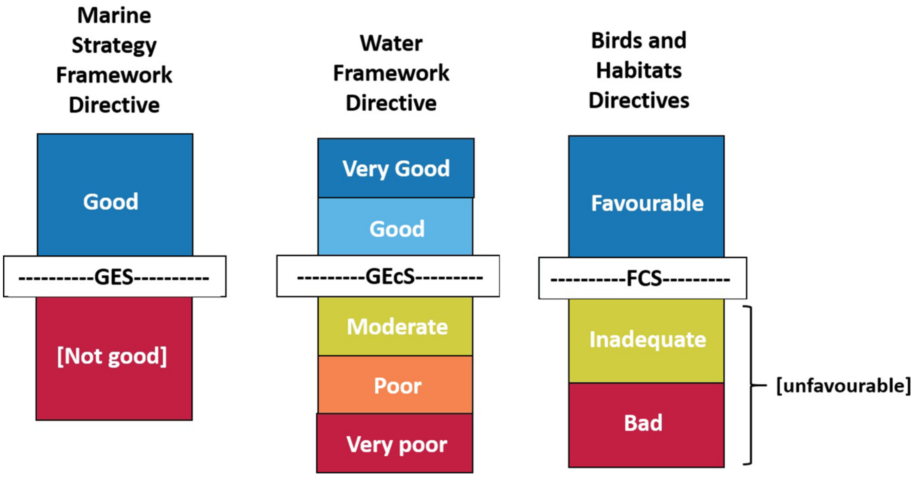

FIGURE 3
Ecosystem status classification according to the Marine Strategy Framework Directive (MSFD), the Water Framework Directive (WFD), and the Birds and Habitats Directives. GES, Good Environmental Status; GEcS, Good Ecological Status; FCS, Favourable Conservation Status. Image redrawn after European Commission (2022).

The Marine Spatial Planning Directive (MSPD) (Directive 2014/89/EU, 2014) was adopted as part of the Integrated Maritime Policy (IMP) and establishes a common framework for maritime spatial planning in the EU. The directive places the legal requirement for all EU Member States with coastal seas to develop and implement Marine Spatial Plans by 2021. The MSPD aims to promote the sustainable development and co-existence of maritime activities and to balance this development with the need to protect the marine environment. The MSPD requires that an ecosystem-based approach is implemented, and that the collective pressure of all activities must be kept within levels compatible with achieving Good Environmental Status (Dir. 2014/89/EU, preamble). Moreover, Member States shall consider economic, social, and environmental aspects when developing their Marine Spatial Plans (Dir. 2014/89/EU, article 5). To promote the ecosystem-based approach to marine spatial planning, the EC has prepared a guidance for its implementation (Ruskule et al., 2021).

The final players in the European marine environmental landscape are the Regional Sea Conventions (RSCs). RSCs are intergovernmental organisations that aim to protect the marine environment and bring together Member States and neighbouring countries that share marine waters to coordinate the implementation of legal requirements of EU marine environmental policies (particularly the MSFD). The RSCs provide a platform to improve regional and cross-regional coherence of national implementation and use the ecosystem approach as a guiding principle. RSCs are relevant for EBFM because they oversee environmental action in regional marine ecosystems. However, their mandate does not include fisheries, and thus their advice is not fully integrated in the EU EBFM advice landscape and mostly arrives through different channels (Ramirez-Monsalve et al., 2021). In Europe, the four RSCs are: the Oslo-Paris Convention for the Protection of the Marine Environment in the North-East Atlantic (OSPAR Convention) implemented by the OSPAR Commission (OSPAR, 1992), the Convention for the Protection of the Marine Environment in the Baltic Sea Area (Helsinki Convention) implemented by the Baltic Marine Environment Protection Commission or HELCOM (HELCOM, 1992), the Barcelona Convention for the Mediterranean Sea, implemented by the United Nations Environmental Program Mediterranean Action Plan (UNEP-MAP, 1995), and the Bucharest Convention for the Black Sea, implemented by the Black Sea Commission (Black Sea Commission, 1992). The main objective of these RSCs is to preserve the marine environment by, for instance, tackling biodiversity loss, reducing pollution and setting up networks of MPAs. Their actions has resulted in a number of improvements in the regional seas (e.g. HELCOM, 2021).

2.2.3 Overarching recent EU strategies

The European Green Deal (COM/2019/6 final, 2019) aims to reach zero net emissions of greenhouse gases in the EU by 2050 and to protect, conserve, and enhance EU's environment, among others. To reach climate neutrality by 2050, the EC adopted a new European Climate Law (Regulation EU 2021/1119, 2021) in 2021, which sets the target of reducing net greenhouse gas emissions by at least 55% by 2030 compared to 1990 levels and achieving climate neutrality (i.e. net-zero greenhouse gas emissions) by 2050. The EU's blue economy is fundamental to meeting the objectives of the EU Green Deal. To fully embed the blue economy into the Green Deal, the Commission adopted in 2021 a new approach for a Sustainable Blue Economy, the Sustainable Blue Economy Strategy (European Commission, 2021). This agenda aims to put sustainability at the forefront (i.e. transition from "Blue Growth" to "Sustainable Blue Economy") and stresses the importance of applying an ecosystem-based management approach to human activities (including fisheries, renewable energy and marine spatial planning).

The EU Biodiversity Strategy 2030 (COM/2020/380 final, 2020) is a core part of the European Green Deal. The strategy is a holistic and long-term plan to protect nature and reverse the degradation of ecosystems, and was a precursor to several of the commitments made under the 2022 Kunming-Montreal CBD Global Biodiversity Framework (see section 2.1). A core commitment under the Biodiversity Strategy is the expansion of protected areas to cover 30% of land and 30% of the sea. Moreover, one third of these protected areas, i.e., 10% on land and 10% at sea, must be strictly protected. Strict protection is defined as leaving natural processes essentially undisturbed to respect the areas' ecological requirements. The Biodiversity Strategy also sets ambitious restoration targets, including the development of a new Nature Restoration Law (COM(2022) 304 final, 2022), which, if approved, will require Member States to cover at least 20% of the EU's land and sea areas with nature restoration measures by 2030, and eventually extend these to all ecosystems in need of restoration by 2050. The measures adopted under the Biodiversity Strategy and the Nature Restoration Law aim to strengthen the protection of the marine ecosystems and to restore them to achieve GES. The Biodiversity Strategy also stresses the need for an ecosystem-based approach to the management of human activities at sea. For fisheries, it sets the targets to maintain or reduce fishing mortality to or under MSY levels; eliminate or reduce bycatch, particularly for sea mammals, turtles and birds that are threatened with extinction or in bad status; and to tackle practices that damage the seabed. In line with these commitments and as part of the Biodiversity Strategy, the EC recently published a new Action Plan to protect and restore marine ecosystems for sustainable and resilient fisheries (Action Plan for fisheries) (European Commission, 2023). The plan sets out concrete measures that Member States have to implement to achieve the objectives of (i) keeping fish stocks at sustainable levels; (ii) reducing the impact of fishing on the seabed; and (iii) minimising fisheries impacts on sensitive species. The measures include to gradually phase out mobile fishing in protected areas by 2030; adopt national measures or submit joint recommendations to minimise by-catch on selected species (including harbour porpoise, common dolphin, and several shark and ray species); and develop threshold for maximum allowable mortality rate for species that are at risk of incidental by-catch in the corresponding regions (including species of birds, mammals, reptiles and non-commercially-exploited species of fish and cephalopods) and adopting management measures to implement these thresholds. By 2024, the Commission will review the progress in implementing the Biodiversity Strategy and the Action Plan and will evaluate whether further actions, such as legislative proposals, are needed.

The main EBFM-relevant international and European policies and bodies presented in this review are pictured in Figures 4, 5. Figure 4 shows their foundation or implementation year and Figure 5 provides a summary of the landscape showing interlinkages, main objectives and relevance of the different legislations, strategies and conventions to EBFM in Europe.

3 Stakeholder needs to implement EBFM

3.1 Policy needs that can be addressed with ecosystem models

As seen in the review, the need to implement an ecosystem-based approach is enshrined in numerous policies and strategies, and various directives, strategies and bodies contribute to its implementation. The following sections will discuss specific EBFM needs that arise from some of the directives described above, which ecosystem modelling can help address (see Table 1 for a summary). While "ecosystem model" is a broad term, in this document the use of ecosystem model refers to temporally and/or spatially dynamic models that simulate the marine food-web or the entire ecosystem by incorporating physical, chemical and biological (i.e. food web) processes under influence of natural and anthropogenic stressors (Figure 6). Because models can differ in their structures and functioning, not all ecosystem models can address all EBFM policy needs equally. For example, not all ecosystem models can address spatial issues, such as MPAs, and species interactions in the models can be based on functional groups, trophic levels or size classes, making MSY hard to address. For a comprehensive overview and assessment of what different ecosystem models can be used for see Chust et al. (2022) and Craig and Link (2023).

The current CFP regulation observes that an EBFM needs to be implemented, and this requires advice on biotic, abiotic, social and economic components (Ramirez-Monsalve et al., 2021). In order to be able to provide sound advice and implement an EBFM, the following needs have been identified by the EC (European Commission, 2008). First, there is a need for long-term predictions. This is because multiple and often conflicting interests need to be reconciled in the process. While there may be short-term contradictions between social and ecological objectives, such contradictions largely disappear in the long-term, making long-term predictions essential. Second, there is a need to include the effects of climate change in the predictions because it is essential that fisheries should be conducted in a way which is robust to environmental change. Exploitation of fish stocks should therefore always allow for resilience to climate change. Third, there is a need to base management on the predictions of the diverse ecosystem effects of fisheries and of management measures, i.e., a need for predicting the consequences of diverse scenarios. This includes the description of ecosystems and their structure, processes and functions using all available knowledge.

The EC's advice (European Commission, 2008) also elaborates on several issues that need to be addressed to ensure an EBFM. These include reducing fishing pressure to MSY; protecting sensitive species and sensitive habitats; and taking measures to prevent distortions in the food web and ensure that natural ecosystem processes are not disrupted [e.g., dependence of seabird colonies on sand eels for breeding success or the importance of herring for other predators (Furness, 2002; Read and Brownstein, 2003)]. Finally, the document highlights the importance of expanding the current assessment of the status

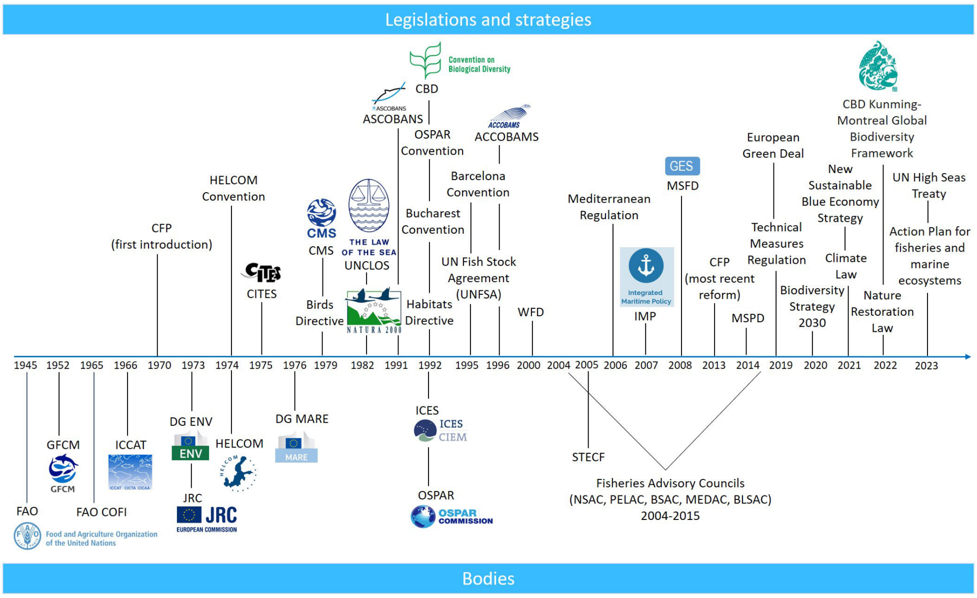

FIGURE 4
A timeline of the main legislations and bodies that directly or indirectly have a repercussion on implementing EBFM in European seas.

TABLE 1 Summary of identified EBFM needs that can be addressed through ecosystem modelling, specifically with Ecopath with Ecosim (EwE) and its spatial component, Ecospace (where relevant).

Identified EBFM need	References	Can be addressed through modelling (temporal and/or spatial component)
MSY *Maintaining fishing pressure at MSY or less; applying MSY to mixed fisheries*	CFP, Biodiversity Strategy; Action Plan for fisheries, ICES Science Plan	Yes (temporal component)
Incidental by-catch *Identifying areas of highest incidental by-catch and assessing effects on populations and ecosystems*	Birds and Habitats Directive, Biodiversity Strategy, Action Plan for fisheries, ASCOBANS and ACCOBAMS Species Action Plans	Yes (temporal and spatial components)
MPAs *Finding the most suitable areas for the 30/10% targets*	Biodiversity Strategy, Birds and Habitats Directive, MSFD, CFP, Action Plan for fisheries, GFCM 2030 Strategy	Yes (temporal and spatial components)
Protecting sensitive/endangered species *Defining maximum allowable mortality, and finding key areas to protect important life-stages*	Birds and Habitats Directive, MSFD, Biodiversity Strategy, Action Plan for fisheries, GFCM 2030 Strategy	Yes (temporal and spatial components)
Reducing seabed impacts *Selecting the most suitable areas to implement the MSFD threshold values*	MSFD, Biodiversity Strategy, Action Plan for fisheries, GFCM 2030 Strategy	Yes (temporal and spatial components)
Effects of climate change *Integrating effects in forecasts of management scenarios*	CFP, MSFD, Birds and Habitats Directive, Biodiversity Strategy, Action Plan for fisheries, ICES Science Plan, GFCM 2030 Strategy	Yes (temporal and spatial components)
Ensuring natural ecosystem processes are not disturbed	CFP, MSFD, Birds and Habitats Directive, ACCOBAMS Conservation Plan for the Common dolphin	Yes (temporal and spatial components)
Marine Spatial Planning	MSPD, MSFD, Birds and Habitats Directive, EU Strategy on Offshore Renewable Energy, Biodiversity Strategy	Yes (temporal and spatial component)
Long-term predictions of management scenarios	CFP, MSFD, Birds and Habitats Directive, Biodiversity Strategy, Action Plan for fisheries; GFCM 2030 Strategy	Yes (temporal and spatial component)
Biodiversity indicators	MSFD, Biodiversity Strategy	Yes (temporal and spatial components)
Regionalisation *Regional groups can influence fisheries management through Advisory Councils and Member States Regional Groups*	CFP	No, but ecosystem modellers can engage with the local stakeholders to co-design relevant management scenarios

and trends of fish stocks to include the impact of fishing on ecosystems.

The 11 descriptors of the MSFD (Figure 2) represent environmental targets that Member States have to achieve to attain “Good Environmental Status” or GES. One approach for testing scenarios that will allow meeting GES is through the application of ecosystem models. Ecosystem models can be used to explore the short- and long-term effectiveness of scenarios for meeting the descriptors relevant to EBFM by: (1) using biodiversity indices outputs to assess biological diversity; (2) using traditional fisheries management indices to assess the health of commercial fish stock populations; (3) assessing the integrity of food webs; and (4) assessing which areas would be most suited to implement the upcoming threshold values of maximum allowable extent of seabed disturbance. The MSFD report on the first implementation cycle (2012-2017) (COM/2020/259 final) also points to specific issues that need to be improved. For instance, the report highlights that EU’s marine waters are still facing overfishing and unsustainable fishing practices, that there has been a steep reduction of elasmobranchs (40% decline) in the Mediterranean Sea, and that a high proportion of Europe’s seabed (79% of the coastal seabed and 43% of the shelf/slope) is physically disturbed, mainly due to bottom trawling.

The Birds Directive and Habitats Directive require the strict protection of all species listed in Annex I of the Birds Directive and Annex IV of the Habitats Directives. This includes numerous seabirds, all cetaceans, as well as five marine turtle species[2]. However, several of these strictly protected species are susceptible to incidental catch. At least 29 seabird species listed in Annex I of the Birds Directives were found to be susceptible to bycatch (STECF, 2020), and incidental catch is a high concern for the strictly protected Harbour porpoise (*Phocoena phocoena*) in the Baltic Sea and the Common dolphin (*Delphinus delphis*) in the Bay of Biscay. This has led environmental NGOs to request

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[2] Marine turtles requiring strict protection under the Habitats Directive: *Caretta caretta, Chelonia mydas, Lepidochelys kempii, Eretmochelys imbricate, Dermochelys coriacea*

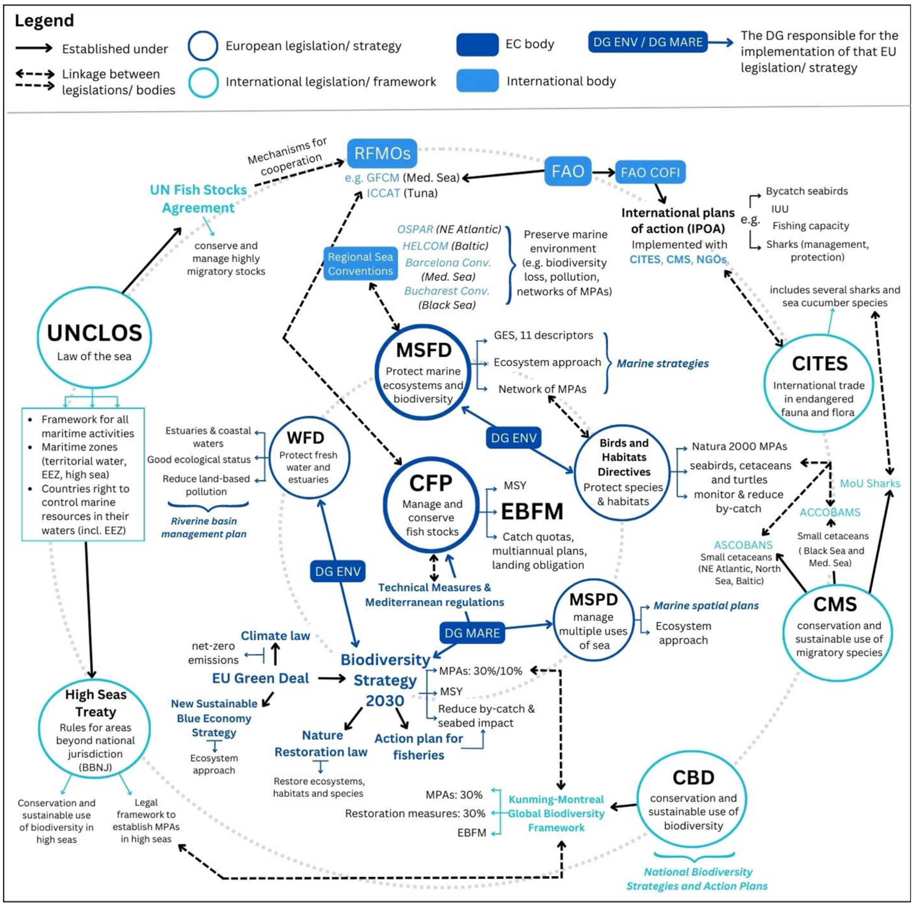

FIGURE 5
EBFM policy landscape, including interlinkages and main objectives of legislations, conventions and strategies relevant to EBFM in the EU. The CFP is depicted in the middle as the most relevant European legislation, surrounded by other European environmental directives and strategies (with the MSFD highlighted in bold because it is the second most relevant legislation for implementing EBFM in the EU). The outer circle represents international conventions and bodies, which are directly or indirectly linked to the implementation of EBFM in Europe.

action[3], and the Commission has started Infringement procedures (European Commission, 2020b), urging the implicated countries to reduce bycatch. Finding solutions to significantly reduce incidental catch of strictly protected species is thus a significant need of the Commission, which has also asked advice from ICES on this matter (ICES, 2020a). Reporting under the Habitats and Birds Directives also showed that the conservation status for most marine habitats and species is either bad or poor (i.e., unfavourable-bad or unfavourable-inadequate in the nomenclature of the Directives; see Figure 3) (European Environmental Agency, 2020), indicating a need for more efficient conservation measures.

The Biodiversity Strategy 2030 and the EU Action Plan for fisheries contain specific actions and commitments, which ecosystem modelling could help inform. For instance, the spatial component of ecosystem models could help to find the most suitable areas to designate the 30% protection and 10% strict protection, which is a need of Member States and the Commission. Similarly, ecosystem modelling can help inform the Action Plan objectives of developing thresholds for maximum allowable mortality rate for the species/species groups listed in the Action Plan and can help to evaluate the effectiveness of new measures that could be applied to help reduce their incidental by-catch to a level that allows species recovery and conservation. Substantially reducing the negative impacts on the seabed, particularly from bottom-contacting gears is another important commitment, and ecosystem modelling can provide advice on which areas would benefit most from a reduction of seabed impacts, as well as in finding the best trade-offs between improving seabed integrity versus minimising the resulting economic impacts on fisheries. Ecosystem models are also well placed to help meet the Biodiversity Strategy 2030 targets of maintaining or reducing fishing mortality to or under MSY levels by helping to inform on the species interactions and trophic cascading effects of these single species measures on other species in the ecosystem. Furthermore, the use of ecosystem models allows testing the likely impact of management measures on fishing mortality, under environmental variability and change, as has

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[3] NGOs call on the EC to take action over huge amounts of cetacean deaths: https://seas-at-risk.org/press-releases/groups-call-on-the-european-commission-to-take-action-over-huge-number-of-cetacean-deaths/

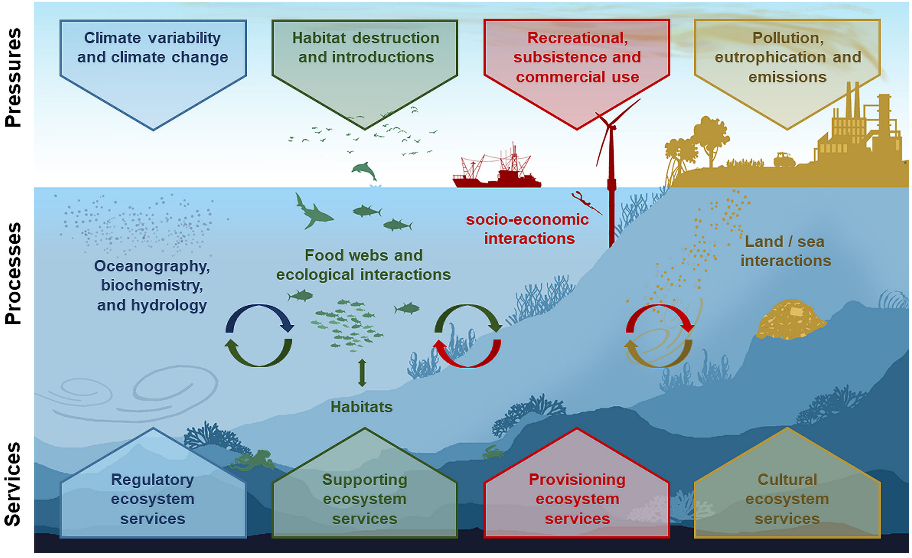

FIGURE 6
The range of interconnected pressures, processes and ecosystem services that complex spatial-temporal marine ecosystem models may consider. Image from Steenbeek et al., 2021a (CC BY-NC-ND 4.0).

been shown by the work undertaken by Bentley et al. (2021) on refining fisheries advice with stock-specific ecosystem information. The Biodiversity Strategy 2030 commitments and targets are key guiding principles for the EU policy landscape for the next 10 years, and the temporal and spatial components of ecosystem modelling, while incorporating ongoing environmental and anthropogenic changes, could make a significant contribution to help implementing these goals.

The legally-binding restoration targets for marine ecosystem in the Nature Restoration Law are another key commitment, which Member States will have to implement (once the law is approved). Under this law, Member States will have to put restoration measures on at least 30% of the area of each habitat type that is not in good condition by 2030, on at least 60% by 2040, and on at least 90% by 2050. Since the targets also include passive restoration (i.e., removing pressures), ecosystem models can inform how best to meet these targets and the likely timeline for habitat and species improvement. For the marine environment, restoration measures have to be put in place in the following habitats: seagrass beds, macroalgal forests, shellfish beds, maerl beds, sponge, coral and coralligenous beds, vents and seeps, and soft sediments above 1000 meters of depth (Annex II of the draft regulation), as well as for the species listed in Annex III of the draft regulation, including many shark and ray species.

To reach the European Climate Law targets of reducing net greenhouse gas emissions by at least 55% by 2030 and achieving climate neutrality by 2050, the EU Strategy on Offshore Renewable Energy (European Commission, 2020a) sets the targets of increasing Europe's offshore wind capacity five-fold by 2030 and 25-fold by 2050. Achieving these targets will require a significant expansion of Marine Renewable Energy. The spatial component of ecosystem models used in combination with marine spatial planning tools will be crucial to help evaluate where best to place offshore Marine Renewable Energy devices, and to predict the immediate and longer-term impacts that the placement of these areas will have on ecosystems and other uses of the ocean.

Finally, strategic research agendas, strategic plans and species action plans reflect key needs of organisations and partnerships that are relevant for policy and which ecosystem modelling can help address. For instance, the ICES Science Plan “Marine ecosystem and sustainability science for the 2020s and beyond” (ICES, 2019) presents seven interrelated scientific priorities for ICES, and identifies the need of “further understanding and operationalising the EBFM and MSY concept, including their application in mixed, multispecies and mesopelagic fisheries” and “improving ICES' capacity to provide ecosystem-based advice”. The GFCM 2030 Strategy (FAO, 2021), provides the most up to date goals and objectives of GFCM and includes the targets to: (i) provide advice on alternative management options for key fisheries; (ii) establish effective area-based measures to reduce impacts on vulnerable species, sensitive habitats and essential fish habitats and meet international spatial conservation targets; (iii) determine the fishing footprint of bottom contact fisheries and their potential interactions with essential and vulnerable habitats; and (iv) implement an adaptation strategy to address the potential effects of climate change and non-indigenous species on fisheries and the marine environment. The species action plans of ASCOBANS[4] and ACCOBAMS[5] include management actions to improve the conservation status of small cetacean populations and identify incidental catch as an essential priority for the Harbour porpoise in the North Sea and the Baltic Sea (ASCOBANS, 2009) and for the Common dolphin in the North-East Atlantic (ASCOBANS, 2019).

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[4] https://www.ascobans.org/en/documents/action-plans

[5] https://accobams.org/species_/conservation-plans/

The species action plans include the following targets: identify the highest-risk fisheries in terms of activities and spatial extent, include Harbour porpoise and Common dolphin in ecosystem models (covering temporal and spatial components), and manage fishing of small epipelagic fish stocks in a way that the energetic needs of Common dolphin are accounted for (ACCOBAMS, 2004).

3.2 Stakeholder needs related to ecosystem modelling reported during a dedicated survey and workshop

The needs of stakeholders in relation to using ecosystem model results to inform the implementation of EBFM were further gauged through a stakeholder survey and a foresight workshop performed as part of the EcoScope project[6]. Stakeholders were selected to represent the main organisations relevant for implementing and advising on EBFM policies in Europe (see EBFM policy landscape section), as well as other important European organisations with interest in EBFM and ecosystem modelling, such as NGOs, fisheries scientific associations and ocean data aggregators. Representatives from the following organisations were invited to participate in the survey and workshop: DG MARE, DG ENV, DG RTD, JRC, GFCM, STECF, ICES, FAO, the Mediterranean Advisory Council (MEDAC), the Baltic Sea Advisory Council (BSAC), the North Sea Advisory Council (NSAC), the European Fisheries and Aquaculture Research Organizations (EFARO), the Fisheries and Aquaculture strategic group of DG RTD (SCAR-FISH), the NGOs World Wide Fund for Nature (WWF), OCEANA and Birdlife, the European Global Ocean Observing System (EuroGOOS), the European Marine Observation and Data Network (EMODnet), and the fisheries organisations Europeche and Low impact fishers of Europe. All organisations were represented at the workshop, except for FAO, WWF, EMODnet and the two fisheries organisations, which were not able to attend. In total, 24 stakeholders participated in the workshop and 18 individuals responded to the survey. The request to fill in the survey was sent together with the workshop invitations to all invited stakeholders. Since the survey was anonymous it was not possible to identify which organisations participated in the survey, although at least one individual that was not able to attend the workshop answered the survey (as indicated by email). It is likely that most of the individuals attending the workshop also answered the survey and thus the 18 survey respondents were probably mostly a subset of the workshop participants.

The aim of the survey was to obtain feedback on (i) the main EBFM policy commitments, general topics and specific questions for which stakeholders thought that ecosystem modelling can help provide answers, (ii) the preferred output format of the ecosystem model results, and (iii) specific limitations or barriers in using the results of ecosystem models to advice on or implement EBFM (see all survey questions in Annex I and a detailed report of the answers in the EcoScope deliverable D.8.1: Report of stakeholder survey, footnote 6). The workshop also gauged the ecosystem modelling needs of these stakeholders, but had a wider and more targeted focus on informing and discussing the development of the EcoScope tools directly with the relevant EcoScope consortium members. The EcoScope tools include EwE ecosystem models for eight European case study areas, but have also other components, such as an interactive platform to visualise relevant data and the results of ecosystem models[7] and a scoring system to evaluate the implications of the different management scenarios[8]. During the workshop, the foreseen EcoScope tools were presented in detail and this was followed by (1) breakout sessions, in which hypothetical scenarios were used as a starting point to obtain targeted feedback on the tools, (2) plenary sessions, in which a rapporteur from each of the breakout rooms reported back on the key messages emerging from each scenario for the design and outputs of the EcoScope tool, (3) plenary discussions and (4) 'deep-dive sessions', in which topics that emerged during the workshop meriting more attention were discussed in more detail in breakout rooms (detailed information on the methods used during the workshop and the feedback obtained is provided in the EcoScope deliverable D.8.3: Report on First Foresight Workshop, footnote 6).

This section will provide a summary of stakeholder needs reported during the survey and workshop in relation to using ecosystem models to inform EBFM in Europe. Most of the relevant feedback was obtained during the survey, because it had a strong focus on the needs and barriers for using ecosystem models to inform EBFM implementation, but the workshop also provided relevant insights, which are included in the summary.

3.2.1 Relevant policy commitments

The key policy commitments, for which stakeholders indicated that ecosystem modelling can help provide answers (survey questions 12 and 13, Annex I) were the MSFD, the CFP and the Biodiversity Strategy 2030. For the MSFD, achieving Good Environmental Status and descriptors D1 (biodiversity is maintained), D3 (the population of commercial fish species is healthy), D4 (elements of food webs ensure long-term abundance and reproduction) and D6 (the sea floor integrity ensures functioning of the ecosystem) were highlighted as priorities. For the CFP, implementing an EBFM, exploiting all stocks at or below MSY, and establishing fish stock recovery areas were the highest priorities. For the Biodiversity Strategy 2030, implementing the protected areas target of 30% protection and 10% strict protection were seen as highly relevant, as well as the commitments under the Nature Restoration Law and the (at the moment of the survey) upcoming Action Plan for Fisheries and the Marine Environment.

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[6] For more information see EcoScope deliverables D.8.1: Report of stakeholder survey and D.8.3: Report on First Foresight Workshop: https://ecoscopium.eu/deliverables

[7] https://ecoscopium.eu/ecoscope-platform

[8] https://ecoscopium.eu/ecoscope-toolbox

3.2.2 Overarching topics of concern and specific questions

To identify key topics and questions, which ecosystem modelling could help address, participants were asked to select the five most relevant overarching topics out of 14 pre-selected ones (survey question 13, Annex I). These topics mainly focused on ecological aspects, except for one topic related to area use, and one on the profitability of the fisheries (Figure 7). In addition, the survey respondents were asked to provide specific questions related to those topics for which they need answers (see question 14 in Annex I and Table 2 for a summary of the answers).

The most relevant EBFM topics highlighted by the stakeholders were effects of climate change, bycatch, protected areas/fisheries restricted areas, and biodiversity indicators. Area use and species distribution were also ranked quite highly, while few respondents weighted fisheries profitability, invasive species and fisheries sustainability indicators strongly (Figure 7). The most highly ranked topics, as well as the specific questions related to those topics, for which the respondents indicated that they would like ecosystem models to help provide answers (Table 2), reflect needs regarding the implementation of key policy commitments (see Table 1). Issues related to fishing quotas and protected areas were related to biological sustainability issues, not socio-economic aspects. Economic concerns were raised with respect to trade-offs in area use, and under the topic of “socio-economic aspects”, while social concerns (such as human well-being and equity) were not specifically mentioned. The latter might be because there are no legal requirements in these policies to implement social aspects, and social aspects generally fall under “socio-economics”, which tends to focus on economic implications as a proxy for well-being.

The stakeholders stressed, both during the survey and workshop, that models need to be tailored to specific issues and cannot be generic if they are to inform policy implementation. There was a strong support of having bespoke scenarios run by experts, which meet the specific needs of stakeholders, and being involved in the scenario development of ecosystem models was a key request of the stakeholders.

3.2.3 Scenario and data results visualisation

The preferred output formats of ecosystem modelling results were simple plots and summaries, visual spatial graphs and infographics (as noted both during the survey and workshop). In addition, workshop participants observed that short-term versus long-term effects (including seasonal variations), socio-economic effects, (cumulative) ecological impacts, indirect impacts and historical values are important information for them. The workshop participants further indicated that it would be useful to have indicators on biomass and catches of the target species and other relevant species, biodiversity indicators, economic indicators for fishers (e.g. profit) and MSFD-related indicators on e.g. biodiversity, food webs and sea bottom impact to inform EBFM implementation.

Overall, two main target audiences were identified by the workshop participants: (1) stakeholders that want summary results with simple plots and numbers that are very clear and easy to understand (e.g. politicians, fishers, etc.), and (2) stakeholders that need more details and the possibility to dig further and understand the background (e.g. advisory bodies, advocacy groups, etc.). Therefore, the participants suggested to combine simple outputs with the possibility to see more details and understand how those results came to be.

3.2.4 Limitations or barriers in using the results of ecosystem models

Although stakeholders were positive about the use of ecosystem models as a tool for managers in meeting EU policy requirements, some concerns emerged. The three main barriers identified during the survey were: reliability of the model’s results, insufficient data and having enough trust in the model outputs. There was a general concern about the reliability and realism of the model and their forecasts, including the accuracy of the models due to limited understanding of some ecological processes and data scarcity. Insufficient data and the quality of input data were key concerns. A lack of data was also seen as one of the main impediments in implementing an EBFM. Many stakeholders voiced concerns about model limitations, given the

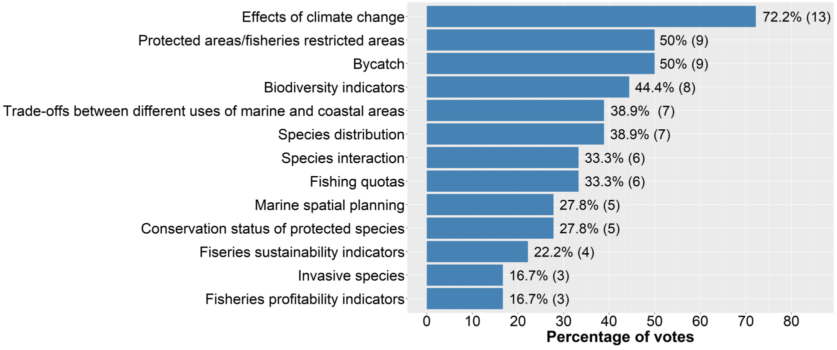


FIGURE 7

EBFM topics ordered by relevance as voted in a stakeholder survey by 18 respondents, each selecting the 5 most relevant topics. Percent values represent the percentage of responders voting for a topic with the number of voters given in brackets.

TABLE 2 EBFM related questions listed by stakeholders during the survey, which ecosystem modelling could help address.

Topic	Specific question
Effects of climate change	What will be the impacts on fish stocks (e.g. distribution and productivity)?
	How will it impact marine species distribution?
	How will the distribution of forage fish change and what are the impacts on marine sensitive species (specifically seabirds during the breeding season)?
	Will cumulative changes lead to a regime shift?
Bycatch	What are the population impacts of specific incidental bycatch levels on marine sensitive species (e.g. harbour porpoise in the Baltic and common dolphin in the Bay of Biscay)?
	What is the "allowed" incidental bycatch of a protected/sensitive species (and the species that these species depends on) that will allow recovery or sustaining healthy levels?
	What is the incidental bycatch impact of fisheries on the status of protected species (under the Birds and Habitats Directive) now and in the near future and how can different management scenarios change this?
	What are the impacts of bycatch on the ecosystem?
	What are the best gear modification options to minimize capture of juveniles/vulnerable species?
Protected areas/fisheries restricted areas	Which areas, across a certain region, should be protected to harness maximum positive effects?
	What are the most valuable ecosystems to designate protected areas and strictly protected areas and how do they overlap with areas important for fishing (and other uses)?
	How would the closure of Bay *x* to fishery *y* effect the species diversity/abundance in *z* years?
Biodiversity indicators	What is the threshold of good environmental status for marine biodiversity?
	What would be the effect of reductions of "charismatic species" (relevant to MSFD D1)?
	What are the best ecosystem based indicators for biodiversity, in relation to the Biodiversity Strategy targets?
Trade-offs between different uses of marine and coastal areas	What is the effect of reducing trawling (or other fishing techniques) in all marine protected areas versus in *x*% of a marine area on (a) economic performance of fisheries and (b) restoring biodiversity?
	What are the impacts of closure of *x*% of bottom trawling?
	What are the trade-offs of the impact of preserving seabed habitats or areas of higher sensitive species occurrence (through "strictly protected" MPAs) vs. impact on economic activities, fishing in particular?
Fishing quotas	Which fishing quotas are really sustainable (e.g. considering impacts of climate change, interspecies interactions and ecosystem resilience to stressors)?
	Are current quotas (also FMSY; BMSY) sustainable in an ecosystem context - also in light of future climate change?
	What is the fishing mortality that allows a harvested species to develop its role in the ecosystem (e.g. predator prey-interactions, etc.)?
	What is the exploitation rate that ensures that all species in a mixed fishery are maintained at "healthy" levels?
	How would *x* percent reduction in quota of species *y* change its biomass in *z* years? Would it cause changes in abundance of other species?
Invasive species	What would be the overall economic and ecological impact of restricting commercial fishery for an already settled invasive species?
Socio-economic aspects	What is the effect of possible management scenarios on medium and long-term profitability of fisheries?

complexity of ecological systems and questioned if these systems can be adequately described by models. The respondents also questioned whether models can properly quantify uncertainty in a whole ecosystem scenario. Uncertainty, credibility, and assumptions made were also key concerns expressed in the workshop. Thus, trust in the models was seen as a key barrier and, both in the survey and workshop, it was suggested to better understand and communicate the limitations and uncertainties of the models to increase trust in them.

4 Discussion

Using ecosystem models in support of EBFM requires an understanding of which policy processes and stakeholder needs could be addressed through ecosystem models (Townsend et al., 2019). This paper provided a detailed overview of the relevant policies, related policy commitments and specific questions, which ecosystem modelling could help address. The stakeholder feedback reflected the needs identified in the document's policy analysis.

Effective application and uptake of scenarios and models in policy and decision-making not only requires understanding the main topics and questions of interest, but also a close involvement of policy makers, practitioners and other relevant stakeholders throughout the entire process of model and scenario development (IPBES, 2016; Heymans et al., 2018). To this end, the paper also summarised the main European and international bodies that influence the implementation of EBFM. This is important knowledge to identify the relevant stakeholders to co-design the models and scenarios. A close involvement of relevant stakeholders throughout the entire process of model and scenario development was also a specific request of stakeholders participating in the workshop, and it has been highlighted as one of the most important recommendations for the update of multispecies models in fisheries management in a recent paper (Karp et al., 2023).

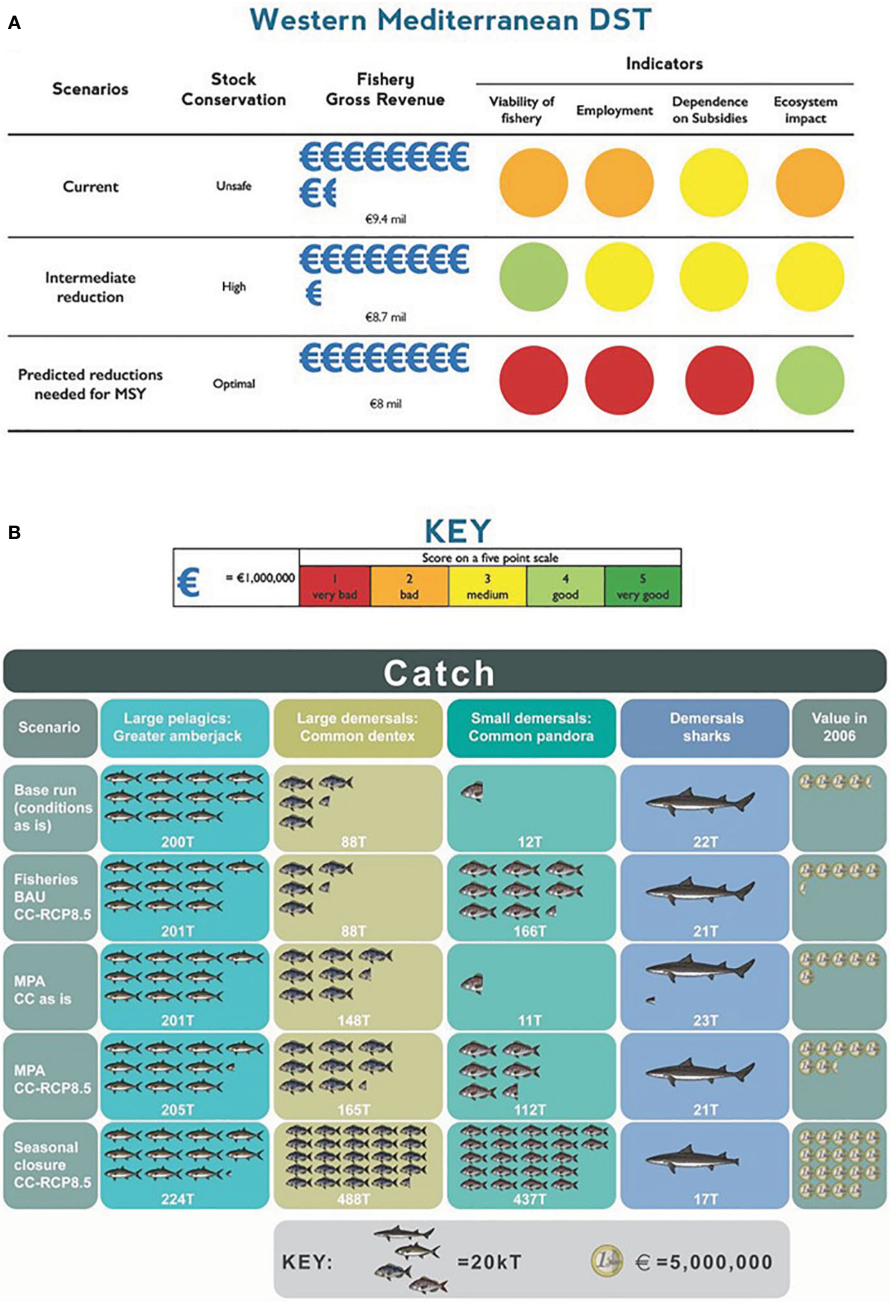
A good example of stakeholder involvement is the regional implementation plan developed for the Balearic Islands in the framework of the EU Myfish project (Myfish, 2012). This study was a first step toward the application of an EBFM in the Balearic Islands by developing a harvest strategy with defined objectives, targets, limits, and clear management control rules aimed at optimizing socioeconomic and ecological objectives in the framework of the CFP. Different management scenarios designed to achieve that goal were modelled for the main demersal commercial fisheries from the study area, the bottom trawl and small-scale fisheries. Throughout the process there was strong involvement of relevant stakeholders through meetings and constant feedback. The management scenarios were agreed with stakeholders, and local stakeholders were involved in how to best present the model results from the selected management scenarios (Quetglas et al., 2017). Another example of good, early and often stakeholder engagement is the WKIRISH work undertaken through ICES, where the stakeholders requested the use of ecosystem models, and then were engaged in the construction and valuation of these models throughout the 3-year process (Bentley et al., 2019a; Bentley et al., 2019b; Bentley et al., 2021).

Simple plots and summary infographics, which were one of the preferred outputs of the stakeholders in this study, will be important for the uptake of complex modelling results by a non-specialist audience. Data visualisation is a powerful method for improving communication of complex scientific outputs and well-designed data visualisations are particularly useful with certain audiences (Bannister et al., 2021). However, presenting the effects and trade-offs of different modelled management scenarios in a simple and understandable way is challenging especially when uncertainty is included, as is highlighted by Bannister et al. (2021). One way of improving data visualisation is by using decision support tables (Levontin et al., 2017; Figure 8). These graphical tables are designed to convey the outcomes of implementing different modelled management scenarios in a simple way (Quetglas et al., 2017). In order to inform policy implementation and management, it is important to design *a priori* the decision support tables with a strong involvement of the targeted stakeholders. This will ensure that the main output variables needed for their decision-making process are reflected. Moreover, it is also important to remember that EBFM stakeholders are not one homogenic group, and as a result the information they require and might want to explore varies greatly - as was also highlighted by the workshop participants. For instance, some stakeholders may want to see the direct output from the model including quantification of errors and uncertainty (e.g. scientific advisory bodies), while another stakeholder might only want to see an infographic or an index value demonstrating, for example, the percent change in catch or state of the ecosystem under a certain management scenario (e.g. decision makers). Therefore, ecosystem model results might need to be provided at a number of levels, and the involvement of the relevant stakeholders is key.

An emergent path of research focuses on enabling the operation of ecosystem models by users that are not ecological experts, but who require ecological insights in decision making and planning processes. This can be done explicitly by including indicator dashboards in ecosystem modelling software (e.g., Coll and Steenbeek, 2017), or implicitly through integrating ecosystem models into planning and decision support software tools (e.g. Santos et al., 2020; Steenbeek et al., 2021b). In this latter pathway, the integrated ecosystem model responds to planning and decision-making explorations by providing meaningful data, graphs, maps and virtual 3D environments related to ecological concerns - without requiring explicit understanding of the ecosystem modelling approach used (Steenbeek et al., 2020). This emergent path of research coincides with unprecedented scientific developments (Stock et al., 2023) and associated technical challenges (Steenbeek et al., 2021a).

One of the main barriers in the uptake of ecosystem modelling results is trust in the models. During the EcoScope survey and workshop, reliability of model forecasts, insufficient data, model limitations and uncertainty were the main concerns of stakeholders. These concerns closely align with insights from previous studies, which found that communicating model limitations and uncertainty is vital if the models are to be used in decision-making. Large gaps of appropriate data was also found to be a significant barrier, because this will often reduce the statistical power of models and limit their ability to predict (IPBES, 2016; Heymans et al., 2018; Heymans et al., 2020). However, when data are lacking or are fragmented ecosystem models may be the best approach to overcome missing data (Regev et al., 2023). One way to address this problem might be the inclusion of stakeholders' knowledge into the models, as was undertaken in the WKIRISH project. However, this is a lengthy and ongoing process that cannot be undertaken in a short period of time. Stakeholder trust is something that can only be built through long term engagement and mutual respect (Bentley et al., 2019b).

Uncertainty associated with model outcome is related to a number of factors ranging from poor model calibration/validation data and input data, to a lack of information on critical model parameters, through to unknown futures (Beck, 1987; Gal et al., 2014). These sources of uncertainty and their impact on decision making have led to new approaches such as robust decision making under deep uncertainty (Lempert, 2003). To further increase challenges, model uncertainty is often poorly evaluated and reported (Steenbeek et al., 2021a), and this can lead to serious misconceptions regarding the



A

Western Mediterranean DST

Scenarios	Stock Conservation	Fishery Gross Revenue	Indicators: Viability of fishery	Indicators: Employment	Indicators: Dependence on Subsidies	Indicators: Ecosystem impact
Current	Unsafe	€€€€€€€€€€ €9.4 mil	🟠	🟠	🟡	🟠
Intermediate reduction	High	€€€€€€€€€ €8.7 mil	🟢	🟡	🟡	🟡
Predicted reductions needed for MSY	Optimal	€€€€€€€€ €8 mil	🔴	🔴	🔴	🟢

B

KEY

€ = €1,000,000	Score on a five point scale				
	🟥 1 very bad	🟧 2 bad	🟨 3 medium	🟩 4 good	🟩 5 very good

Catch

Scenario	Large pelagics: Greater amberjack	Large demersals: Common dentex	Small demersals: Common pandora	Demersals sharks	Value in 2006
Base run (conditions as is)	200T	88T	12T	22T	
Fisheries BAU CC-RCP8.5	201T	88T	166T	21T	
MPA CC as is	201T	148T	11T	23T	
MPA CC-RCP8.5	205T	165T	112T	21T	
Seasonal closure CC-RCP8.5	224T	488T	437T	17T	

KEY: =20kT € =5,000,000

FIGURE 8
Examples of visual decision support tables that can be used to communicate ecosystem modelling scenario results in a simple manner. **(A)** Decision support table reflecting the model results of different management scenarios for the main commercial bottom trawl fisheries of the Balearic Sea (Myfish, 2012). **(B)** Decision support table presented as part of the EcoScope workshop, inspired by the graphics produced by Quetglas et al., 2017 (image credit: Gideon Gal).

confidence level with which results can be used in decision-making (IPBES, 2016). Reporting uncertainty increases the confidence to use model outputs for decision-making as well as the credibility of models (Heymans et al., 2018). Recommendations from the stakeholders participating in the workshop on how to address and communicate uncertainty were: (i) using a range of possible values, instead of final numbers; (ii) focusing on trends, rather than a specific value, as these are easier to communicate and have less uncertainty; and (iii) labelling the certainty of the results, rather than the uncertainty. A good example for the latter is the IPCC calibrated language, where results are labelled with "*very high confidence* ", "*high confidence*", "*medium confidence*", "*low confidence*" and "*very low confidence*" (IPCC, 2018). In addition, it was suggested to clearly convey by whom/how the models had been validated to increase trust in their outputs.

Overall, the main barriers impeding the widespread use of ecosystem models and scenario testing in decision-making are: (i) a lack of understanding of the benefits and limits of these tools for assessment and decision support among decision-makers; (ii) insufficient involvement of, and interactions between, scientists, stakeholders and policymakers in developing scenarios and models to assist policy design and implementation; and (iii) inadequate characterization of uncertainties derived from data constraints, problems in system understanding and representation or low system predictability (IPBES, 2016).

5 Conclusion

In conclusion, implementing EBFM is complex due to the many aspects that have to be considered, such as multi-species interactions, environmental/climate forcing, habitat status, human activities and stakeholder acceptance. Ecosystem models are able to predict the effects of management decisions on some of these interrelated variables and can therefore make an important contribution to an effective implementation of EBFM. This paper provided an overview of the global and European policy commitments that are driving the implementation of EBFM in Europe, and associated stakeholder needs relevant to ecosystem modelling. The most relevant topics were effects of climate change, bycatch, protected areas/fisheries restricted areas, and reducing the impacts of trawling. These topics reflect main European policy commitments, such as the MSFD, the CFP and the Biodiversity Strategy 2030, with its associated Nature Restoration Law and Action Plan for Fisheries and the Marine Environment. Uptake of ecosystem models in policy requires that models address specific policy needs, such as the ones presented in this paper, and deliver outputs that are easily interpreted by policy makers and can be adjusted to the management capabilities and legislation while communicating the degree of certainty (or uncertainty) in the model projections. Moreover barriers, such as insufficient trust in the models, have to be overcome. To ensure the relevance of model results to policy implementation, it is important that stakeholders are involved throughout the process of scenario development, and that the results of the models are presented – in consultation with the relevant stakeholders – in a way that is understandable to them, and which allows them to comprehend the limitations of the results. Specific recommendations on how to increase confidence in using model outputs for decision-making were to present a range of values instead of final numbers, focus on trends rather than specific values, and to label the certainty (e.g. medium confidence) instead of the uncertainty of the results.

Author contributions

SH, ARP and AT conceived the original idea. ARP, SH, GG, JS, JFA and AT organised and performed the stakeholder engagement events. ARP wrote the initial manuscript. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The reviewer YJS declared a past co-authorship with the author JS to the handling editor.

Publisher's note

References

ACCOBAMS (2004) *Conservation Plan for short-beaked common dolphins in the Mediterranean Sea*. Available at: https://accobams.org/species_/conservation-plans/.

ASCOBANS (2009) *ASCOBANS Conservation Plan for Harbour Porpoises (Phocoena phocoena L.) in the North Sea*. Available at: https://www.ascobans.org/en/documents/actionplans/North-Sea-Conservation-Plan.

ASCOBANS (2019) *ASCOBANS Species Action Plan for North-East Atlantic Common Dolphin*. Available at: https://www.ascobans.org/en/document/ascobans-species-action-plan-north-east-atlantic-common-dolphin.

Bannister, H. J., Blackwell, P. G., Hyder, K., and Webb, T. J. (2021). Improving the visual communication of environmental model projections. *Sci. Rep.* 11, 19157. doi: 10.1038/s41598-021-98290-4

Beck, M. B. (1987). Water quality modeling: a review of the analysis of uncertainty. *Water Resour. Res.* 23, 1393–1442. doi: 10.1029/WR023i008p01393

Bentley, J. W., Hines, D. E., Borrett, S. R., Serpetti, N., Hernandez-Milian, G., Fox, C., et al. (2019a). Combining scientific and fishers' knowledge to co-create indicators of food web structure and function. *ICES J. Mar. Sci.* 76, 2218–2234. doi: 10.1093/icesjms/fsz121

Bentley, J. W., Lundy, M. G., Howell, D., Beggs, S. E., Bundy, A., De Castro, F., et al. (2021). Refining fisheries advice with stock-specific ecosystem information. *Front. Mar. Sci.* 8. doi: 10.3389/fmars.2021.602072

Bentley, J. W., Serpetti, N., Fox, C., Heymans, J. J., and Reid, D. G. (2019b). Fishers' knowledge improves the accuracy of food web model predictions. *ICES J. Mar. Sci.* 76, 897–912. doi: 10.1093/icesjms/fsz003

Black Sea Commission (1992) *The Convention on the Protection of the Black Sea Against Pollution (Bucharest Convention)*. Available at: http://www.blacksea-commission.org/_convention.asp.

Breuer, M. E. G. (2022) *The Common Fisheries Policy: Origins and Development*. Available at: https://www.europarl.europa.eu/ftu/pdf/en/FTU_3.3.1.pdf.

Cardinale, M., Osio, G. C., and Scarcella, G. (2017). Mediterranean Sea: a failure of the European fisheries management system. *Front. Mar. Sci.* 4, 72. doi: 10.3389/fmars.2017.00072

Carocci, F., Bianchi, G., Eastwood, P., and Meaden, G. (2009). Geographic Information Systems to support the ecosystem approach to fisheries. *FAO Fish. Aquac. Pap.* 532, 120. Available at: http://www.fao.org/docrep/012/i1213e/i1213e.pdf

CBD (1992) *Convention on Biological Diversity*. Available at: https://www.cbd.int/convention/text/.

CBD (2022) *Kunming-Montreal Global Biodiversity Framework Press Release*. Available at: https://prod.drupal.www.infra.cbd.int/sites/default/files/2022-12/221219-CBD-PressRelease-COP15-Final.pdf.

Christensen, V., Guenette, S., Heymans, J. J., Walters, C. J., Watson, R., Zeller, D., et al. (2003). Hundred-year decline of North Atlantic predatory fishes. *Fish Fish.* 4, 1–24. doi: 10.1046/j.1467-2979.2003.00103.x

Chust, G., Corrales, X., Gonzáles, F., Villarino, E., Chifflet, M., Fernandes, J. A., et al. (2022). *Marine biodiversity modelling study*. Open Access. (Luxemburg: Publications Office of the European Union) 141 pp. doi: 10.2777/213731

CITES (1973) *Convention on International Trade in Endangered Species of Wild Fauna and Flora*. Available at: https://cites.org/eng/disc/text.php.

CMS (1979) *Convention on the Conservation of Migratory Species of Wild Animals*. Available at: https://www.cms.int/en/convention-text.

Coll, M., and Steenbeek, J. (2017). Standardized ecological indicators to assess aquatic food webs: The ECOIND software plug-in for Ecopath with Ecosim models. *Environ. Model. Software* 89, 120–130. doi: 10.1016/j.envsoft.2016.12.004

COM/2019/6 final (2019) *Communication from the Commission to the European Parliament, the European Council, the Council, the European Economic and Social Committee and the Commitee of the Regions: The European Green Deal*. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM%3A2019%3A640%3AFIN.

COM/2020/380 final (2020) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: EU Biodiversity Strategy for 2030 Bringing nature back into our lives. COM/2020/380 Final*. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A52020DC0380.

COM(2022) 304 final (2022) *Proposal for a Regulation of the European Parliament and of the Council on nature restoration. COM(2022) 304 Final*. Available at: https://environment.ec.europa.eu/publications/nature-restoration-law_en.

Council Directive 92/43/EEC (1992). Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Off. J. Eur. Union* 11, 0114. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A31992L0043

Craig, J. K., and Link, J. S. (2023). It is past time to use ecosystem models tactically to support ecosystem-based fisheries management: Case studies using Ecopath with Ecosim in an operational management context. *Fish Fish.* 24, 381–406. doi: 10.1111/faf.12733

Directive 2000/60/EC (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. *Off. J. Eur. Union* 327, 0001–0037. Available at: https://eur-lex.europa.eu/eli/dir/2000/60/oj

Directive 2008/56/EC (2008). Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). *Off. J. Eur. Union* 164, 19. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32008L0056

Directive 2009/147/EC (2009). Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (Codified version). *Off. J. Eur. Union* 20, 7. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32009L0147

Directive 2014/89/EU (2014). Directive 2014/89/EU of the European Parliament and of the Council of 23 July 2014 establishing a framework for maritime spatial planning. *Off. J. Eur. Union* 257, 135. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32014L0089

Dolan, T. E., Patrick, W. S., and Link, J. S. (2016). Delineating the continuum of marine ecosystem-based management: a US fisheries reference point perspective. *ICES J. Mar. Sci.* 73, 1042–1050. doi: 10.1093/icesjms/fsv242

EC Council Regulation 1967/2006 (2006). Council Regulation (EC) No 1967/2006 of 21 December 2006 concerning management measures for the sustainable exploitation of fishery resources in the Mediterranean Sea. *Off. J. Eur. Union* 409, 11. Available at: https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX%3A32006R1967

EcoScope (2021). Ecocentric management for sustainable fisheries and healthy marine ecosystems. (European Commission's Horizon 2020 Research and Innovation programme). Available at: https://ecoscopium.eu/.

European Commission (2008) *Communication from the Commission to the Council and the European Parliament: The role of the CFP in implementing an ecosystem approach to marine management. COM(2008) 187 Final*. Available at: https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2008:0187:FIN:EN:PDF.

European Commission (2018) *Natura 2000 in the Marine Environment*. Available at: https://ec.europa.eu/environment/nature/natura2000/marine/index_en.htm#:~:text.

European Commission (2020a) *Communication from the Commission to the European Parliament, the council, the European Economic and Social Committee and the Committee for the Regions: An EU Strategy to harness the potential of offshore renewable energy for a climate neutral future. COM/2020/741 Final*. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM%3A2020%3A741%3AFIN.

European Commission (2020b) *July infringement package: key decisions*. Available at: https://ec.europa.eu/commission/presscorner/detail/en/INF_20_1212.

European Commission (2021) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on a new approach for a sustainable blue economy in the EU Transforming the EU's Blue Economy for a Sust. COM/2021/240 Final*. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM:2021:240:FIN.

European Commission (2022) *GES under MSFD and WFD: similarities and differences*. Available at: https://slideplayer.com/slide/16562551/.

European Commission (2023) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committe of the Regions. EU Action Plan: Protecting and restoring marine ecosystems for sustainable and resilient fisheries. COM/2023/102 Final*. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52023DC0102.

European Environmental Agency (2020) *State of nature in the EU. Results from reporting under the nature directives 2013-2018*. Available at: https://www.eea.europa.eu/publications/state-of-nature-in-the-eu-2020.

FAO (1999). *IPOA - International Plan of Action for reducing incidental catch of seabirds in longline fisheries. International Plan of Action for the conservation and management of sharks. International Plan of Action for the management of fishing capacity* (Rome: FAO). Available at: https://www.fao.org/publications/card/en/c/26edca5c-aadf-5ace-979e-56482168ad1c/.

FAO (2001). *International Plan of Action to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated Fishing* (Rome: FAO). Available at: https://www.fao.org/documents/card/en/c/71be21c9-8406-5f66-ac68-1e74604464e7.

FAO (2005). *Putting into practice the ecosystem approach to fisheries* (Rome: FAO). Available at: https://www.fao.org/publications/card/en/c/80f97581-8f96-5313-9db7-1792e912730e.

FAO (2008). Fisheries management. 2. The ecosystem approach to fisheries. 2.1 Best practices in ecosystem modelling for informing an ecosystem approach to fisheries. *FAO Fisheries Technical Guidelines for Responsible Fisheries*. (Rome: FAO) 4 (Suppl 2), 78.

FAO (2021). *GFCM 2030 Strategy for sustainable fisheries and aquaculture in the Mediterranean and the Black Sea* (Rome).doi: 10.4060/cb7562en.

Froese, R., Winker, H., Coro, G., Demirel, N., Tsikliras, A. C., Dimarchopoulou, D., et al. (2018). Status and rebuilding of European fisheries. *Mar. Policy* 93, 159–170. doi: 10.1016/j.marpol.2018.04.018

Froese, R., Tsikliras, A. C., Scarcella, G., and Gascuel, D. (2021). Progress towards ending overfishing in the Northeast Atlantic. *Mar. Policy* 125, 104282. doi: 10.1016/j.marpol.2020.104282

Fu, C., Perry, R. I., Shin, Y.-J., Schweigert, J., and Liu, H. (2013). An ecosystem modelling framework for incorporating climate regime shifts into fisheries management. *Prog. Oceanogr.* 115, 53–64. doi: 10.1016/j.pocean.2013.03.003

Furness, R. W. (2002). Management implications of interactions between fisheries and sandeel-dependent seabirds and seals in the North Sea. *ICES J. Mar. Sci.* 59, 261–269. doi: 10.1006/jmsc.2001.1155

Gal, G., Makler-Pick, V., and Shachar, N. (2014). Dealing with uncertainty in ecosystem model scenarios: application of the single-model ensemble approach. *Environ. Model. Software* 61, 360–370. doi: 10.1016/j.envsoft.2014.05.015

Garcia, S. M. (2003). *The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook* (Food & Agriculture Org). Available at: https://www.fao.org/documents/card/en?details=701c6108-e79d-57bb-87d8-eeb85effb2cd/

Garcia, S. M., Zerbi, A., Aliaume, C., Do Chi, T., and Lasserre, G. (2003). Ecosystem approach to fisheries: issue, terminology, principles, institutional foundations, implementation and outlook. *FAO Fish. Tech. Pap.* 443, 71.

HELCOM (2021) *Baltic Sea Action Plan 2021 update*. Available at: https://helcom.fi/baltic-sea-action-plan/#:~:text=TheBalticSeaActionPlan,Seainahealthystate.

HELCOM. (1992). Convention on the protection of the marine environment of the Baltic sea area, 1992 (Helsinki Convention). Available at: https://helcom.fi/about-us/convention/

Heymans, J. J., Bundy, A., Christensen, V., Coll, M., de Mutsert, K., Fulton, E. A., et al. (2020). The ocean decade: a true ecosystem modeling challenge. *Front. Mar. Sci.* 766. doi: 10.3389/fmars.2020.554573

Heymans, S. J. J., Skogen, M. D., Schrum, C., and Solidoro, C. (2018). *Enhancing Europe's Capability in Marine Ecosystem Modelling for Societal Benefit*. K. E. Larkin, J. Coopman, A. Muñiz Piniella, P. Kellett, C. Simon, C. Rundt, et al [Eds.] (Ostend, Belgium: Future Science Brief 4 of the European Marine Board), 32 pp.

ICES (2020a). *EU request on emergency measures to prevent bycatch of common dolphin (Delphinus delphis) and Baltic Proper harbour porpoise (Phocoena phocoena) in the Northeast Atlantic* (ICES Spec. Req. Advice). Available at: https://ices-library.figshare.com/articles/report/EU_request_on_emergency_measures_to_prevent_bycatch_of_common_dolphin_Delphinus_delphis_and_Baltic_Proper_harbour_porpoise_Phocoena_phocoena_in_the_Northeast_Atlantic/18636260.

ICES (2020b) *Guide to ICES advisory framework and principles. In Report of the ICES Advisory Committee 2020. ICES Advice 2020, section 1.1*. doi: 10.17895/ices.advice.7648.

ICES. (2019). Science plan of the International Council for the Exploration of the Sea. doi: 10.17895/ices.pub.5469

IPBES (2017). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services.* S. Ferrier, K. N. Ninan, P. Leadley, R. Alkemade, L. A. Acosta, H. R. Akcakaya, et al (eds.). (Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), 348 pages. doi: 10.5281/zenodo.3235428

IPCC (2018). *Summary for Policymakers. In: Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response*. Eds. V. Masson-Delmotte, P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P. R. Shukla, et al (Cambridge, UK and New York, NY, USA: Cambridge University Press), 3–24. doi: 10.1017/9781009157940.001.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., et al. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293, 629–637. doi: 10.1126/science.1059199

Jennings, S., and Kaiser, M. J. (1998). "The effects of fishing on marine ecosystems," in *Advances in marine biology* (Elsevier) 34, 201–352. Available at: https://www.sciencedirect.com/science/article/abs/pii/S0065288108602126

Karp, M. A., Link, J. S., Grezlik, M., Cadrin, S., Fay, G., Lynch, P., et al. (2023). Increasing the uptake of multispecies models in fisheries management. *ICES J. Mar. Sci.* 80(2), 243–257. doi: 10.1093/icesjms/fsad001

Lempert, R. J. (2003). *Shaping the next one hundred years: new methods for quantitative, long-term policy analysis.* (RAND Corporation)

Levontin, P., Baranowski, P., Leach, A. W., Bailey, A., Mumford, J. D., Quetglas, A., et al. (2017). On the role of visualisation in fisheries management. *Mar. Policy* 78, 114–121. doi: 10.1016/j.marpol.2017.01.018

Montana, J. (2020). Scientific advice for fisheries management in the European Union: transnational science diplomacy in practice. In: M. Young, T. Flink and E. Dall (eds.) *Science Diplomacy in the Making: Case-based insights from the S4D4C project.*

Moullec, F., Barrier, N., Guilhaumon, F., Peck, M. A., Ulses, C., and Shin, Y.-J. (2023). Rebuilding Mediterranean marine resources under climate change. *Mar. Ecol. Prog. Ser.* 708, 1–20. doi: 10.3354/meps14269

Myers, R. A., and Worm, B. (2003). Rapid worldwide depletion of predatory fish communities. *Nature* 423, 280–283. doi: 10.1038/nature01610

Myfish (2012). Maximising yield of fisheries while balancing ecosystem, economic and social concerns (European Union's Seventh Framework Programme). Available at: https://www.myfishproject.eu/.

Nakamura, J. N., and Kuemlangan, B. (2020). *Implementing the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) through national fisheries legal frameworks: a study and a guide. Legal Guide No. 4* (Rome: FAO). Available at: https://www.fao.org/documents/card/fr/c/cb1906en/.

Ofir, E., Silver, T., Steenbeek, J. G., Shachar, N., and Gal, G. (2022). Applying the Safe Operating Space (SOS) approach to sustainable commercial fishing under varying lake levels and littoral zone conditions. *Fisheries.* 48 (3), 107–120. doi: 10.1002/fsh.10869

OSPAR (1992) *Convention for the protection of the marine environment of the North-East Atlantic (OSPAR Convention).* Available at: https://www.ospar.org/convention/text.

OSPAR Commission (2017) *The 11 qualitative descriptors for determining good environmental status as presented in the Marine Strategy Framework Directive (MSFD).* Available at: https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/introduction/policy-context/.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. Jr (1998). Fishing down marine food webs. *Science* 279, 860–863. doi: 10.1126/science.279.5352.860

Plagányi, É.E. (2007). *Models for an ecosystem approach to fisheries.* (Rome, Italy: Food and Agricultural Organization), 108pp. (FAO Fisheries Technical Paper. No. 477). doi: 10.25607/OBP-392

Popescu, I. (2019) *Fisheries. Briefing: EU policies – Delivering for citizens.* Available at: https://www.europarl.europa.eu/RegData/etudes/BRIE/2019/633150/EPRS_BRI(2019)633150_EN.pdf.

Quetglas, A., Merino, G., González, J., Ordines, F., Garau, A., Grau, A. M., et al. (2017). Harvest strategies for an ecosystem approach to fisheries management in western Mediterranean demersal fisheries. *Front. Mar. Sci.* 4, 106. doi: 10.3389/fmars.2017.00106

Ramirez-Monsalve, P., Nielsen, K. N., Ballesteros, M., Kirkfeldt, T. S., Dickey-Collas, M., Delaney, A., et al. (2021). Pulling mechanisms and pushing strategies: how to improve Ecosystem Approach Fisheries Management advice within the European Union's Common Fisheries Policy. *Fish. Res.* 233, 105751. doi: 10.1016/j.fishres.2020.105751

Ramirez-Monsalve, P., Raakjær, J., Nielsen, K. N., Santiago, J. L., Ballesteros, M., Laksá, U., et al. (2016). Ecosystem Approach to Fisheries Management (EAFM) in the EU–Current science–policy–society interfaces and emerging requirements. *Mar. Policy* 66, 83–92. doi: 10.1016/j.marpol.2015.12.030

Read, A. J., and Brownstein, C. R. (2003). Considering other consumers: fisheries, predators, and Atlantic herring in the Gulf of Maine. *Conserv. Ecol.* 7 (1). https://www.jstor.org/stable/26271914.

Regev, S., Carmel, Y., and Gal, G. (2023). Using high level validation to increase lake ecosystem model reliability. *Environ. Model. Software* 162, 105637. doi: 10.1016/j.envsoft.2023.105637

Regulation EU 1380/2013 (2013). Regulation (EU) No 1380/2013 of the European parliament and of the council of 11 December 2013 on the common fisheries policy, (2013). *Off. J. Eur. Union* 354, 22. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32013R1380

Regulation EU 2016/1139 (2016). Regulation (EU) 2016/1139 of the European Parliament and of the Council of 6 July 2016 establishing a multiannual plan for the stocks of cod, herring and sprat in the Baltic Sea and the fisheries exploiting those stocks. *Off. J. Eur. Union* 191, 1. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=uriserv:OJ.L_.2016.191.01.0001.01.ENG&toc=OJ:L:2016:191:TOC

Regulation EU 2018/973 (2018). Regulation (EU) 2018/973 of the European Parliament and of the Council of 4 July 2018 establishing a multiannual plan for demersal stocks in the North Sea and the fisheries exploiting those stocks, specifying details of the implementation of the landing o. *Off. J. Eur. Union* 179, 1. Available at: https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX:32018R0973

Regulation EU 2019/1022 (2019). Regulation (EU) 2019/1022 of the European Parliament and of the Council of 20 June 2019 establishing a multiannual plan for the fisheries exploiting demersal stocks in the western Mediterranean Sea. *Off. J. Eur. Union* 172, 1. Available at: https://eur-lex.europa.eu/eli/reg/2019/1022/oj

Regulation EU 2019/1241 (2019). Regulation (EU) 2019/1241 of the European Parliament and of the Council of 20 June 2019 on the conservation of fisheries resources and the protection of marine ecosystems through technical measures. *Off. J. Eur. Union* 198, 105. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32019R1241

Regulation EU 2019/472 (2019). Regulation (EU) 2019/472 of the European Parliament and of the Council of 19 March 2019 establishing a multiannual plan for stocks fished in the Western Waters and adjacent waters, and for fisheries exploiting those stocks. *Off. J. Eur. Union* 83, 1. Available at: https://eur-lex.europa.eu/eli/reg/2019/472/oj

Regulation EU 2021/1119 (2021). Regulation (EU) 2021/1119 of the European Parliament and of the Council of 30 June 2021 establishing the framework for achieving climate neutrality and amending Regulations (EC) No 401/2009 and (EU) 2018/1999 ('European Climate Law'). *Off. J. Eur. Union* 243, 1. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32021R1119

European Commission, European Climate and Infrastructure and Environment Executive Agency, Ruskule, A., Oulès, L., Zamparutti, T., et al (2021). *Guidelines for implementing an ecosystem-based approach in maritime spatial planning: including a method for the evaluation, monitoring and review of EBA in MSP* (Publications Office). doi: 10.2926/84261

Santos, C. P., Warmelink, H., Boode, W., de Groot, P., Hutchinson, K., Gancalves, M., et al. (2020). A digital game-based simulation platform for integrated maritime spatial planning: design challenges and technical innovations. *J. Ocean Technol.* 15, 78–100.

SCMEE (2005) *SCMEE Transversal Workshop on Ecosystem Approach to Fisheries.* Available at: https://gfcmsitestorage.blob.core.windows.net/documents/web/ftp/fi/document/gfcm/sac8/dma6e.pdf.

Staples, D., and Funge-Smith, S. (2009). *Ecosystem approach to fisheries and aquaculture: implementing the FAO code of conduct for responsible fisheries* (Bangkok, Thailand: FAO Regional Office for Asia and the Pacific). Available at: https://www.fao.org/3/i0964e/i0964e00.pdf.

STECF (2020). *Scientific, Technical and Economic Committee for Fisheries (STECF) – Review of technical measures (part 1) (STECF-20-02).* H. Polet, A. Rindorf, C. Konrad and P. Vasilakopoulos editor(s), (Luxembourg: EUR 28359 EN, Publications Office of the European Union). doi: 10.2760/734593,JRC123092

Steenbeek, J., Buszowski, J., Chagaris, D., Christensen, V., Coll, M., Fulton, E. A., et al. (2021a). Making spatial-temporal marine ecosystem modelling better–a perspective. *Environ. Model. Software* 145, 105209. doi: 10.1016/j.envsoft.2021.105209

Steenbeek, J., Felinto, D., Pan, M., Buszowski, J., and Christensen, V. (2021b). Using gaming technology to explore and visualize management impacts on marine ecosystems. *Front. Mar. Sci.* 8, 619541. doi: 10.3389/fmars.2021.619541

Steenbeek, J., Romagnoni, G., Bentley, J., Heymans, J. J., Serpetti, N., Gonçalves, M., et al. (2020). Combining ecosystem modeling with serious gaming in support of transboundary maritime spatial planning. *Ecol. Soc* 25. doi: 10.5751/ES-11580-250221

Stock, A., Murray, C. C., Gregr, E., Steenbeek, J., Woodburn, E., Micheli, F., et al. (2023). Exploring multiple stressor effects with Ecopath, Ecosim, and Ecospace: Research designs, modeling techniques, and future directions. *Sci. Total Environ.* 869, 161719. doi: 10.1016/j.scitotenv.2023.161719

Townsend, H., Harvey, C. J., DeReynier, Y., Davis, D., Zador, S. G., Gaichas, S., et al. (2019). Progress on implementing ecosystem-based fisheries management in the United States through the use of ecosystem models and analysis. *Front. Mar. Sci.* 6, 641. doi: 10.3389/fmars.2019.00641

UNEP-MAP. (1995). Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean (Barcelona Convention). Available at: https://www.unep.org/unepmap/who-we-are/barcelona-convention-and-protocols.

UN General Assembly (1982) *United Nations Convention on the Law of the Sea of 10 December 1982.* Available at: https://www.un.org/depts/los/convention_agreements/convention_overview_convention.htm (Accessed December 20, 2022).

UN General Assembly (1995) *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.* Available at: https://www.un.org/depts/los/convention_agreements/convention_overview_fish_stocks.htm (Accessed December 20, 2022).

Vincent, A. C. J., Sadovy de Mitcheson, Y. J., Fowler, S. L., and Lieberman, S. (2014). The role of CITES in the conservation of marine fishes subject to international trade. *Fish Fish.* 15, 563–592. doi: 10.1111/faf.12035

Vinther, M., Reeves, S. A., and Patterson, K. R. (2004). From single-species advice to mixed-species management: taking the next step. *ICES J. Mar. Sci.* 61, 1398–1409. doi: 10.1016/j.icesjms.2004.08.018

Annex I – stakeholder survey

Section 1: Context

1. Which category does the organisation you work for belong to? 1= Policy/Regulatory; 2= Advisory/Scientific; 3= Other

2. Which of the following (if any) inform your work? Please check only the most relevant (up to 5) 1 = Common Fisheries Policy; 2 = Marine Strategy Framework Directive; 3 = Habitats and Birds Directive; 4 = Marine Spatial Planning Directive; 5 = EU Biodiversity Strategy 2030; 6 = EU Green Deal; 7 = Water Framework Directive; 8 = Invasive Alien Species Regulation; 9 = Blue Economy Strategy; 10 = Other

2b. If 10, Please specify ___________

3. To what extent do you implement/regulate/advise on ecosystem-based fisheries management? (Scale: Never/Almost never, Occasionally, Regularly/Very often)

4. How would you rate your capacity/expertise to implement/regulate/advise on ecosystem-based fisheries management? (Scale: Low, Medium, High)

Section 2: Ecosystem-based fisheries management needs

5. What are the main challenges/barriers you face when implementing/regulating/advising on ecosystem-based fisheries management? Please list the three most significant challenges/barriers.

a. __________

b. __________

c. __________

6. What do you foresee will be the main future challenges when implementing/regulating/advising on ecosystem-based fisheries management? Please list the three most significant potential future challenges or risks.

a. __________

b. __________

c. __________

7. In order to better implement/regulate/advise on ecosystem-based fisheries management, what are the main questions you need answers to? Please list the three most significant questions.

a. __________

b. __________

c. __________

Section 3: Ecosystem Modelling

Ecosystem models can be used to test a wide range of management scenarios over time and space, and observe the influence of these decisions on the ecosystem and the Blue Economy. For instance, one could use the models to address questions such as:

- What would be the impact of a new Fisheries Restricted Area on the fisher's catch and the marine ecosystem over the next 5 years?

- How do different Total Allowable Catches influence the targeted stock population, the wider marine ecosystem and the fisher's profitability?

- Which of four potential marine areas would benefit most from strict protection, and what would be the long-term impact on the marine ecosystem and the fisher's catch?

- What would be the influence of invasive species on the marine ecosystem, the influence of aquaculture and different placing of cages, etc.?

This 5 min video illustrates how ecosystem models can be used as a decision support system in the Israeli Mediterranean.

8. To what extent do you think robust ecosystem modelling could help to address the most significant questions you identified in the previous question (Q7)? (Scale: Not at all, a little, significantly)

9. Do you foresee any specific limitations or barriers in using the results of ecosystem models to advise/implement on ecosystem-based fisheries management? Please specify __________

10. In which specific situations would you use the results generated by ecosystem modelling in the context of ecosystem-based fisheries management? Please provide examples __________

11. In what form would the model results be most useful to you and why? __________

Section 4: Ecosystem modelling and EU policy

12. Do you think ecosystems modelling can help meet existing EU policy requirements? Please mention specific directives and requirements within the directives __________

13. Do you think ecosystems modelling can help meet planned policy requirements (e.g. upcoming action plans, new directives, etc.)? Please specify __________

Ecosystem modelling needs

14. Which of the following topics are most relevant for you? Please check the top 5

1 = fishing quotas; 2 = protected areas/fisheries restricted areas; 3 = bycatch; 4 = invasive species; 5 = marine spatial planning; 6 = effects of climate change; 7 = species distribution; 8 = conservation status of protected species; 9 = fisheries sustainability indicators; 10 = biodiversity indicators; 11 = fisheries' profitability indicators; 12 = species interaction; 13 = trade-offs between different uses of marine and coastal areas; 14 = others

14b. If 14, Please specify __________

15. From the topics you selected as most relevant for you (Q14), what specific questions and issues (scenarios) would you like ecosystem models to help you with? For instance: if bycatch is a main topic of interest, a specific question could be: what is the effect of x level of bycatch on a protected species and the wider ecosystem over the next 5 years. Or if fishing quota is a main topic of interest, a specific question could be: given projected climate change will current quotas allow sustaining commercial fish populations in the future? Please list up to 5 questions you would most like ecosystem models to address.

a. __________

b. __________

c. __________

d. __________

e. __________

Section 5: Other comments

16. Please provide any other comments you may have related to ecosystem-based fisheries management needs that could be met with additional data or models. __________

17. Is there anything else you would like to mention? __________

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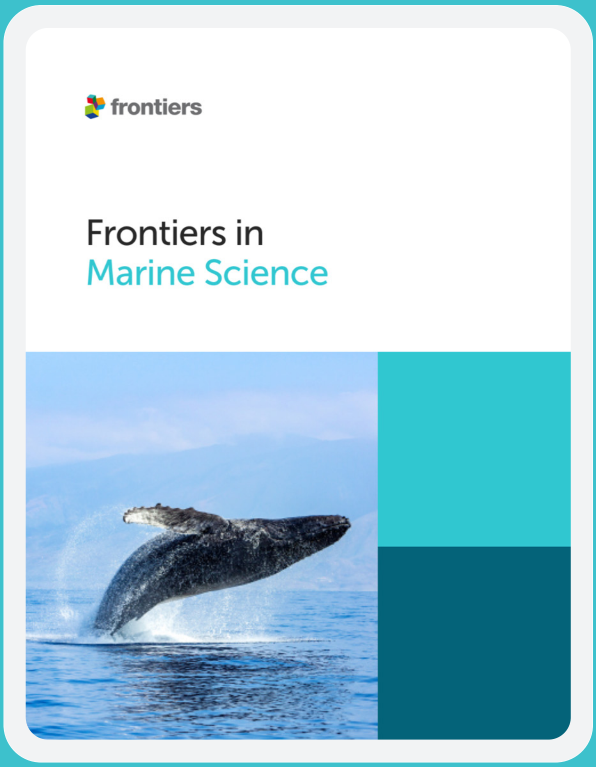
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Avenue du Tribunal-Fédéral 34
1005 Lausanne, Switzerland
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