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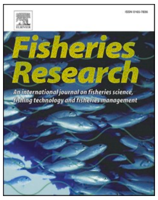

Full Length Article

Assessing the spatial economic effects of marine conservation and offshore wind development on fisheries: A Bayesian network framework

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ABSTRACT

Integrating economic considerations into spatial decision-making remains a central challenge in Marine Spatial Planning. However, frameworks specifically designed to analyse the economic implications of fisheries management under competing maritime uses remain scarce. This study develops and applies a novel spatially explicit Bayesian network to assess the direct economic effects of alternative management scenarios on fisheries. By modelling conditional dependencies among fishing effort, costs, and revenues, it enables uncertainty propagation and scenario-based inference under data-constrained conditions. The framework is applied to real-world management scenarios from the Italian Marine Spatial Plan for the Adriatic Sea, focusing on the establishment of new Marine Protected Areas and offshore wind farms. Results indicate that planned Marine Protected Areas overlap with valuable areas for small-scale fisheries, highlighting the need for management strategies that balance conservation and socioeconomic objectives. In contrast, large-scale fisheries show greater potential direct economic effects under the planned Natura 2000 areas, indicating the need for proactive stakeholder engagement. Offshore wind farm development is projected to generate no direct economic effects on small-scale fisheries, whereas large-scale fisheries are estimated to experience greater direct economic effects due to the loss of accessible fishing grounds. Overall, the framework advances spatial economic assessment within Marine Spatial Planning by enabling the identification of potential conflicts, supporting stakeholder engagement, and strengthening the evaluation of trade-offs among competing maritime uses.

1. Introduction

Marine Spatial Planning (MSP) is a comprehensive process to manage human activities across time and space to achieve ecological, economic, and social objectives (Frazão Santos et al., 2019; Ehler and Douvere, 2009). The increasing pressures on marine ecosystems, driven by long-term established and emerging human activities, underscore the need for adaptive strategies that respond to evolving socio-ecological conditions and stakeholder needs (Navarro et al., 2022; Gacutan et al., 2019). In this context, MSP has emerged as a response to the limitations of traditional, sectoral management approaches, and the need to integrate ecosystem-based principles, adaptive management, and participatory processes (Galparsoro et al., 2025).

Over the past decade, MSP has become a key policy framework for managing competing uses of marine space while supporting the sustainable development of maritime activities (Li and Jay, 2020). Within the European Union (EU), the MSP process was operationalised through the MSP Directive (MSPD; 2014/89/EC), which established a legal framework requiring Member States to develop transparent maritime plans that promote sustainable maritime economic growth alongside the responsible use of marine and coastal resources (Friess and Grémaud--Colombier, 2021). The MSPD underpins the EU sustainability agenda by aligning maritime planning with conservation and blue economy objectives. The Biodiversity Strategy for 2030 (COM/2020/380) calls for an expanded network of Marine Protected Areas (MPAs) covering at least 30% of European seas, including one-third under strict protection (European Commission, 2020a). Within the broader blue economy agenda, notably reflected in the Blue Growth Strategy (European

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Commission, 2017), the EU Offshore Renewable Energy Strategy (COM/2020/741) supports the large-scale deployment of offshore wind farms (OWFs), targeting 86–89 GW by 2030 and 355–366 GW by 2050 (European Commission, 2024).

Overall, the growing establishment of conservation areas and OWFs, increasingly recognised as key solutions to global challenges (i.e., climate change, biodiversity loss, and sustainable energy production), entails trade-offs and opportunity costs, with MSP serving as a central governance tool for the sustainable development of marine sectors (European Commission, 2020b). Such developments and the resulting demand for sea space are expected to increase marine traffic and necessitate the identification of new sea routes designed to protect natural resources while coexisting with human activities (Galparsoro et al., 2025; Gourgiotis et al., 2024; Yim et al., 2024). In this regard, the European Commission has underscored the importance of incorporating spatial analyses of economic activities when making MSP decisions to address both the ecological and socioeconomic effects of plans (European Commission, 2020b).

Among maritime sectors, fisheries are especially sensitive to spatial allocation decisions due to their direct dependence on access to marine space and resources (Vigo et al., 2024; Said and Trouillet, 2020; Coccoli et al., 2018). Fisheries are commonly categorised according to vessel characteristics and gear type as small-scale fisheries (SSF) or large-scale fisheries (LSF). SSF includes fishing activities carried out by fishing vessels under 12 m in length using passive gears, while LSF comprises vessels over 12 m using passive gears and all vessels operating with towed gears, both exhibiting distinct operational patterns, adaptive capacities, and economic structures (STECF, 2023; EMFF 1198/2006). Spatial restrictions may therefore generate uneven economic effects across fisheries, potentially threatening their economic viability.

Persistent data limitations characterise the EU fishery sector, particularly within SSF, constraining the assessment of socioeconomic responses and ultimately hindering the development of evidence-based management measures (Murillas-Maza et al., 2021; Goti-Aralucea, 2019). Several analyses examining fisheries in the context of MSP have been conducted (Galparsoro et al., 2024; White et al., 2024; Bonsu et al., 2024; Vigo et al., 2024; Stelzenmüller et al., 2024; Iwona et al., 2021). However, the spatial assessment of economic effects associated with or derived from the application of MSP remains underexplored, with few frameworks and applications (Murillas-Maza et al., 2023; Surís-Regueiro et al., 2021). Previous studies have employed bioeconomic simulations or opportunity-cost analyses to estimate fisheries' economic responses to spatial restrictions (Scheld et al., 2022; van de Geer et al., 2013). Moreover, Kruse et al. (2024) employed the Bayesian Network (BN) approach to provide spatially explicit estimates of adaptive capacity and profitability within fisheries' socio-ecological systems under different management scenarios. OWF development has been linked to fisheries exclusion and effort displacement, resulting in revenue reductions and increased operational costs (Scheld et al., 2022), whereas MPA expansion may lead to the loss of traditional fishing grounds, longer travel distances, and the need for strategic adaptation by fishers (van de Geer et al., 2013). Fisheries social-ecological systems facing expanding spatial constraints are therefore often required to reorganise their activities to remain economically viable (Kruse et al., 2024). Building on this literature, conservation measures and OWFs are expected to generate heterogeneous spatial economic effects across fisheries, reflecting differences in their dependence on specific fishing grounds.

In this context, this study addresses a gap in the spatial economic assessment of fisheries within MSP contexts by developing a BN framework to estimate the direct economic effects of conservation and OWFs on fisheries using the gross value added (GVA) as an indicator for such assessments. The BN enables the explicit representation of conditional dependencies among economic variables, allowing uncertainty to propagate through interdependent variables, thereby supporting scenario-based inference under limited data availability (Pascoe, 2018). The developed framework is applied to a real-world planning context in the Italian Exclusive Economic Zone of the Adriatic Sea (GSA17) to analyse its relevance for management and decision-making. Specifically, this study pursues three objectives: (i) to develop a conceptual model designed to analyse the integrated performance of fisheries and their underlying system structure; (ii) to operationalise the conceptual model into a spatially explicit model using a BN approach; and (iii) to demonstrate the suitability of the proposed BN for projecting the direct economic effects of spatial allocation of MPAs and OWFs on fisheries. Overall, this study advances the development of new approaches and methods for better informing decision-making processes towards sustainable marine and fishery management within the context of MSP.

2. Methods

2.1. Study area

The study area encompasses the Italian Exclusive Economic Zone (EEZ) of the Adriatic Sea (GSA17), covering 36,055 km^2 (Fig. 1). The northern part features a broad, shallow continental shelf, with an average depth of 35 m, representing the largest continental shelf in the Mediterranean Sea. In contrast, the central basin reaches greater depths, with the Pomo/Jabuka Pit reaching 270 m (Russo et al., 2020). High population density along the coast, together with intensive fishing, tourism, and maritime activities, contributes significantly to cumulative environmental pressures and impacts (Bastardie et al., 2017). Additionally, the 19 seaports in the area, each handling over a million tonnes of cargo annually, further exacerbate pollution (Giani et al., 2019). The GSA17 is recognised as one of the most productive fishing areas in the Mediterranean Sea and, consequently, one of the most exploited European basins (Russo et al., 2020; Grati et al., 2013).

A total of 2941 fishing vessels, including all fleet segments, operate in the area, accounting for 24.4% of the national fleet. The SSF, employing polyvalent passive gears (PGP), plays a key socioeconomic role by linking fishery resources with local traditions and values, thus fostering social cohesion within local communities (Cavraro et al., 2023). LSF operates across the continental shelf, targeting demersal species and reflecting the inherently multi-species structure of Adriatic fisheries (Bastardie et al., 2017). The GSA17, which is predominantly characterised by sandy, muddy, and alluvial seabed, is highly suitable for trawl fisheries, although it remains prohibited within three nautical miles of the coast. Key fishing activities include bottom trawling (DTS) and *rapido* trawling (TBB) for demersal species, and dredgers (DRB) for harvesting clams. Trawling hotspots show an intensive fishing pressure, with significant areas experiencing intensities exceeding 10 fishing events per year (Eigaard et al., 2017). Fisheries management in the study area relies on a range of management instruments aimed at protecting biodiversity, including effort reduction, technical measures (e.g., minimum landing sizes and mesh limitations) and seasonal or permanent closures. The Italian strategy for implementing the EMFAF 21–27 objectives in the case study area includes a reduction in the number of vessels for purse seiners (PS) and pelagic trawlers (TM) by 20% and by 10% for TBB and DTS (MASAF, 2023). The area hosts three established MPAs (i.e., Miramare, Torre del Cerrano, and the Tremiti Islands) and five MPAs under the Natura 2000 network. Protecting biologically important habitats and nursery areas for commercial species represents a major management priority. Under the Italian MSP, two new MPAs (i.e., Monte Conero and Costa del Piceno) along with two MPAs under the Natura 2000 network have been preliminarily identified by the national authorities as areas for designation (MIT, 2024). Moreover, in accordance with the EU Marine Renewable Energy Strategy, nine concessions for fixed and floating OWFs are currently at the planning stages (4COFFSHORE, 2024).

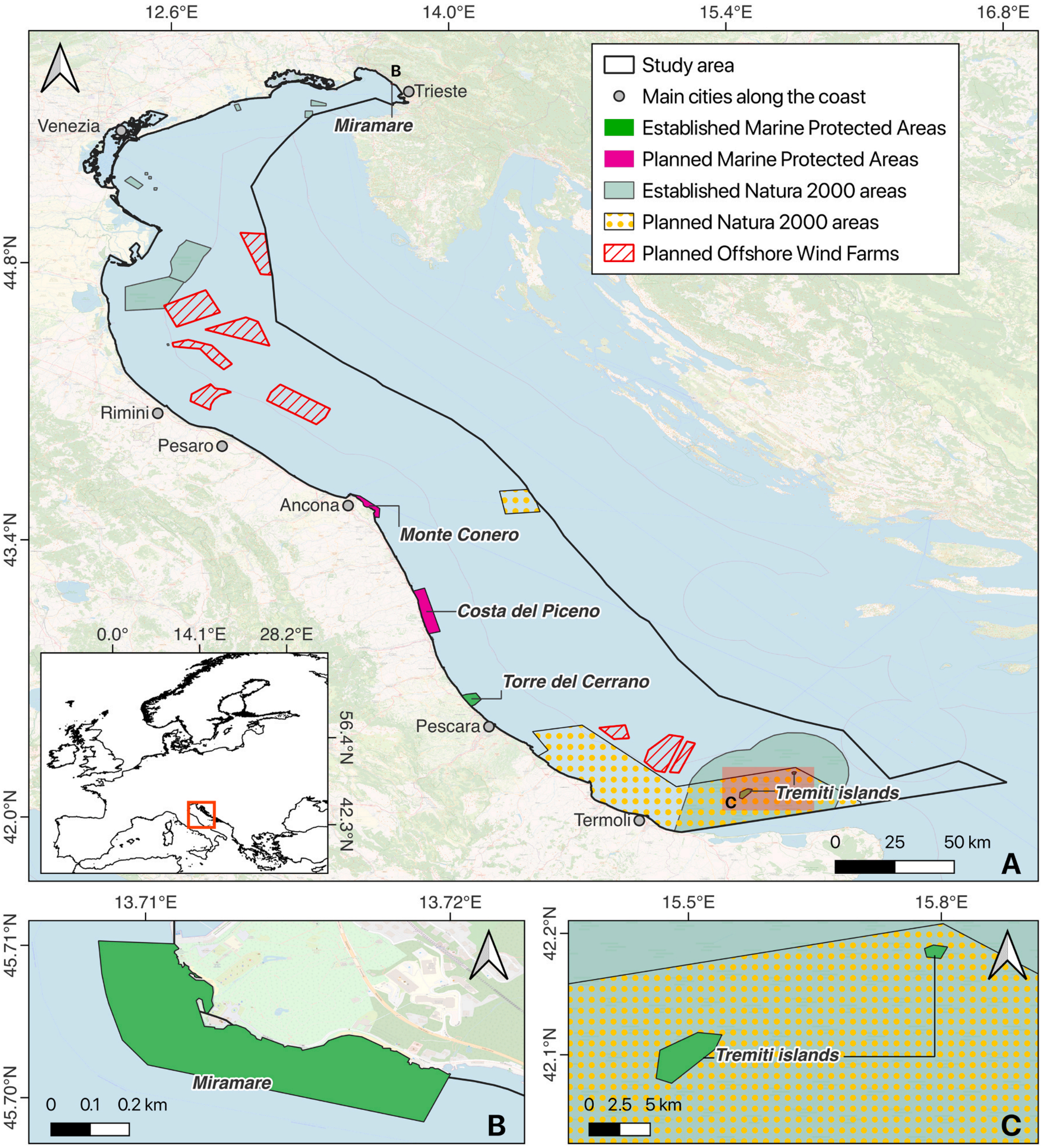

Fig. 1. Study area location with existing and planned marine protected areas (MPAs) and offshore wind farms (OWFs): (A) full extent; (B-C) enlarged views of selected MPAs.

2.2. Data collection and processing

2.2.1. Geospatial data

Geospatial data were collected from different sources (Table 1). The latest available data on LSF and SSF fishing effort until 2024 were used to characterise the spatial distribution of fishing activities. SSF fishing effort data were gathered from a participatory mapping study by Grati et al. (2022), which provided spatial information based on the number of operations recorded in 2019. For LSF, fishing effort data were sourced from the Italian MSP repository (MIT, 2024), which provided spatial information on the average number of hours fished annually over the period 2006–2018 derived from Vessel Monitoring System (VMS) data. The main fishing ports (n = 28) were retrieved from the EU Fleet Register (2024) (Fig. 2). Polygon shapefiles for planned OWFs were derived from 4COFFSHORE (2024), while planned MPAs and Natura 2000 were sourced from the Italian MSP repository (MIT, 2024).

Geospatial data were processed using the Quantum Geographic Information System (QGIS). A 1 × 1 km grid, consisting of 37,183 grid cells, was applied across the entire study area to serve as a reference for subsequent analyses. The shortest distance from each grid cell to its

Table 1
Geospatial data used for analysing fishing activities and spatial planning in the study area.

Spatial information layer	Geometry type	Source
Main fishing ports	Points	EU Fleet Register (2024)
Planned Offshore Wind Farms	Polygons	4COFFSHORE (2024)
Planned Marine Protected Areas	Polygons	Italian Ministry of Infrastructure and Transport (2024)
Planned Marine Protected Areas under Natura 2000	Polygons	Italian Ministry of Infrastructure and Transport (2024)
Small-scale fisheries effort	Polygons	Grati et al. (2022)
Large-scale fisheries effort	Polygons	Italian Ministry of Infrastructure and Transport (2024)

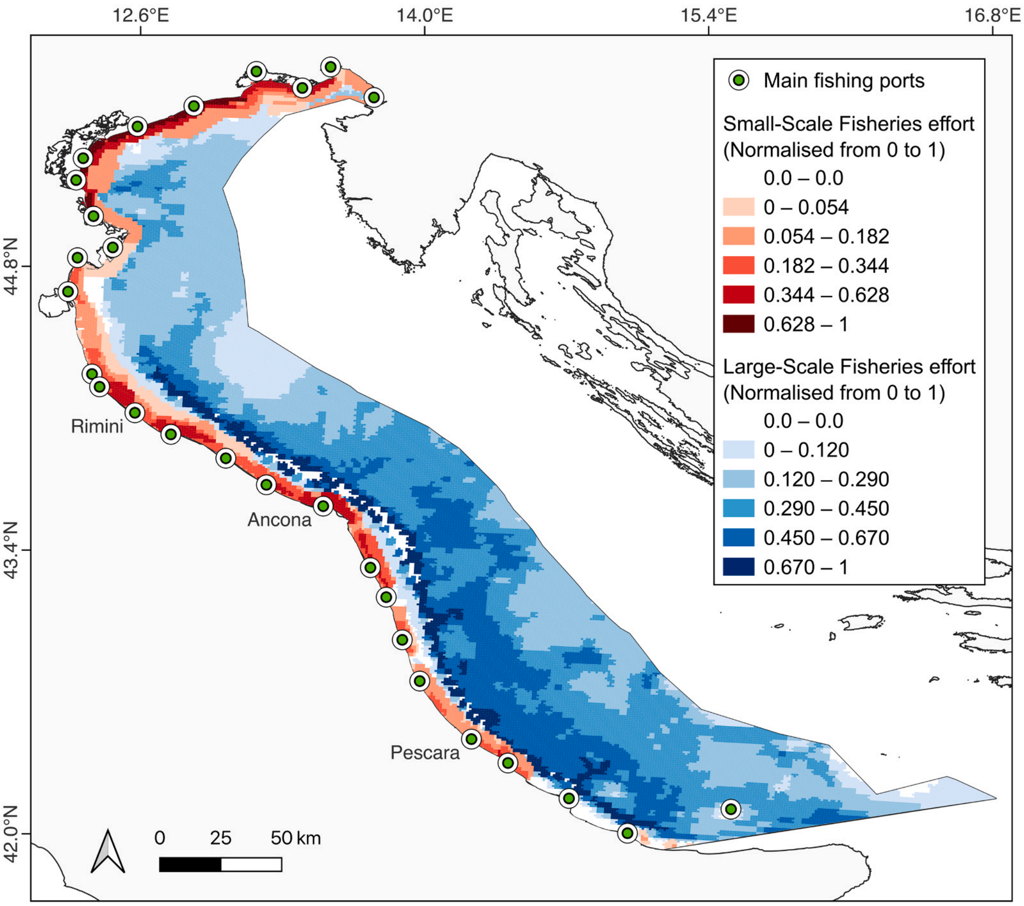

Fig. 2. Spatial distribution of fishing effort and main fishing ports.

nearest port was calculated. SSF and LSF fishing effort values were normalised from 0 to 1 and assigned to each grid cell by merging the values based on location, ensuring consistent spatial resolution (Fig. 2). Finally, the shapefiles of the planned OWFs and MPAs were overlaid onto the grid cells.

2.2.2. Operational and economic data

Data on operational and economic variables used to calculate the GVA at the fishery level were collected from the STECF (2023) database for European fleets' clusters defined by country, gear and vessel length class. These clusters are therefore associated with SSF and LSF operating within the study area (MASAF, 2023), whose description is shown in Table 2.

The STECF database provides inflation-adjusted information on fisheries expenditures and income. The variables, including the number of days at sea (DAS), are reported as annual sums for national fleets across the supra-region. To calculate the aggregate expenditures and income for SSF and LSF, each economic variable was first divided by the number of DAS for each fishing segment, defined by gear type and vessel length class (STECF, 2023), then multiplied by the corresponding number of vessels, and finally summed for each fishery. The average fuel price was calculated as the mean ratio of energy cost (€) to fuel consumption (litres), while the average market price of first sale landings was calculated by dividing the aggregate gross value landings (€) by the aggregate live weight landings (kilograms). The year 2019 was selected to align with the temporal scope of the spatially explicit fishing effort, ensuring consistency in associating economic values with the recorded effort. Moreover, it represents the last fully validated pre-pandemic year of fishing activity, as 2020–2022 were substantially affected by COVID-19-related disruptions that altered typical spatial fishing patterns, and post-pandemic economic datasets at the required resolution are not yet fully validated or publicly available. The aggregate economic values at fishery level for SSF and LSF within the study area are outlined in Table 3.

Table 2
Fisheries operating within the study area by gear employed, vessel length, and number of vessels. Source: MASAF (2023).

Fisheries	Gear	Vessel's length (metres)	Number of vessels
Small-scale	Polyvalent passive gears	0–6	478
		6–12	807
Large-scale	Polyvalent passive gears	12–18	43
	Dredgers	12–18	585
	Demersal trawlers and/or demersal seiners	6–12	24
		12–18	219
		18–24	167
		24–40	37
	Purse seiners	6–12	3
		12–18	2
		24–40	9
		> 40	1
	Beam trawlers	12–18	10
		18–24	26
		24–40	25
	Pelagic trawlers	12–18	27
		18–24	18
		24–40	35

2.2.3. Spatialisation of economic variables

The spatialisation of the economic variables was carried out by considering their influencing factors, defined according to the Annual

Table 3
Summary of aggregate economic values within the study area for Large-scale fisheries (LSF) and Small-scale fisheries (SSF), reference year 2019. DAS: Days at sea.

Variable name	Unit	Large-scale fisheries	Small-scale fisheries
Repair and maintenance costs	€/DAS	118,045	11,525
Energy costs	€/DAS	345,398	26,235
Other variable costs	€/DAS	131,914	15,865
Other non-variable costs	€/DAS	95,122	14,716
Other income	€/DAS	18,165	16,011
Gross value landings	€/DAS	2088,366	247,292
Market price	€/kg	3.54	6.37
Fuel price	€/litre	0.59	0.78

Economic Report nowcast methodology (STECF, 2023) and expert knowledge (Table 4).

Repair and maintenance costs include both fixed and variable components depending on fleet characteristics, which cannot be readily distinguished from annual data (Pascoe et al., 2015). Following previous studies (Cao et al., 2021; Duy et al., 2015), Repair and maintenance costs were treated as fixed, assuming they are influenced by the physical characteristics of the vessels and the technology employed. Similarly, Other non-variable costs and Other income, which are influenced by regulations (e.g., resolutions, orders), are not dependent on fishing activity. Therefore, for the variables (i) Repair and maintenance costs, (ii) Other non-variable costs, and (iii) Other income, the aggregate values for the study area were distributed equally across the grid cells according to Eq. (1).

$$z_{k,i} = \begin{cases} \dfrac{\sum z_{k,i}}{N_{k,\ \ active}}, \&Effort_{k,\ \ i} > 0 \\ 0, \&Effort_{k,\ \ i} = 0 \end{cases} \tag{1}$$

Where $z_{k,i}$ represents the values of (i) Repair and maintenance costs, or (ii) Other non-variable costs, or (iii) Other income for the fishery k (i.e., SSF or LSF) at the grid cell i. $N_{k,active}$ is the number of active grid cells where the fishing effort ($Effort_{k,i}$) for the fishery k is greater than zero.

The variable Energy costs, which is influenced by the distance from ports, and the fishing effort (Pelletier et al., 2014), was spatialised using Eq. (2):

$$Energy\_cost_{k,i} = \frac{Effort_{k,i} \times Port\_distance_i}{\sum (Effort_{k,i} \times Port\_distance_i)} \times \sum Energy\_cost_{\ k,i} \tag{2}$$

Where $Energy\_cost_{k,i}$ represents the Energy cost of the fishery k at the grid cell i, $Effort_{k,i}$ is the fishing effort of the fishery k at the grid cell i, and $Port\_distance_i$ is the distance of the grid cell i from the nearest port.

Other variable costs and Gross value landings, which are influenced by fishing effort (Cao et al., 2021; Daurès et al., 2013), were spatialised based on fishing effort (Eq. (3)).

Table 4
Key factors influencing the values of the economic variables for the fishery sector.

Variable group	Variable name	Key factor(s) influencing the variable's value
Expenditure	Repair and maintenance costs	Fishing technology
Expenditure	Energy costs	Fishing effort; Fishing technology; Distance from port; Fuel price; Days at sea
Expenditure	Other variable costs	Fishing effort; Fish harvested
Expenditure	Other non-variable costs	Regulations
Income	Other income	Regulations
	Gross value landings	Fishing effort; Market price

$$y_{k,i} = \frac{Effort_{k,i}}{\sum Effort_{k,\ \ i}} \times \sum y_{\ k,i} \tag{3}$$

Where $y_{k,i}$ is the (i) Other variable costs or (i) Gross value landings of the fishery k at the grid cell i, and $Effort_{k,i}$ is the fishing effort of the fishery k at the grid cell i.

All variables were mapped in QGIS as point vectors and merged into the analysis grid, which included the following attributes: cell ID, fishery (SSF or LSF), Fishing effort, Distance from port, Gross value landings, Other income, Energy costs, Other variable costs, Market price and Fuel price. The variables Repair and maintenance costs and Other non-variable costs were grouped into a single variable, named fixed costs. Three datasets were created to represent SSF, LSF, and the combined fisheries (i.e., SSF and LSF) (Table 5). For the SSF and LSF datasets, the spatial distribution of economic variables was determined by mapping values across 37,183 grid cells in the study area that exhibited fishing effort greater than zero for the respective fishery. In the combined fisheries dataset, where both SSF and LSF co-occurred within the same grid cells (5% of the study area), income and expenses were aggregated, and the average values for market price and fuel price were calculated and spatialised accordingly. Finally, GVA was calculated for each grid cell.

2.2.4. *Model development*

2.2.4.1. Development of a conceptual model of fisheries economics. A conceptual model representing the fishery system was developed through a structured expert-elicitation process grounded in the stock-based dimensions of the DAPSI(W)R(M) framework, which includes drivers, activities, pressures, impacts (on welfare), and responses (Elliott et al., 2017) (Fig. 3). This approach enabled the consideration of socio-economic interactions characterising fisheries, using a systematic approach to analyse their integrated dynamics and underlying system structure. The conceptual model was defined through a series of workshops involving five experts in the field of fisheries economics, marine ecology, and MSP. Participants included academic researchers and experienced MSP practitioners. The sessions combined iterative discussion to identify key system components, define causal linkages, and validate the conceptual structure of the model.

The main driving factors, including the employees, the first-sale market price of fish, and fuel price, were linked to two human activities carried out in the study area, encompassing both SSF and LSF, producing pressures in terms of fishing effort. GVA was employed as a socioeconomic indicator of impacts linked to the response component, where management measures (i.e., establishment of MPAs and Natura 2000, reductions in fishing effort) were identified as strategies to mitigate or address the ecological, social, and economic effects driven by activities and pressure components.

2.2.4.2. Development of the Bayesian network. A BN approach was adopted to estimate the direct economic effects of the spatial allocation of MPAs and OWFs on predominant fisheries. Spatial resolution input and output data were used when applying the BN using a given grid cell merged with different spatial management scenarios. The BN provides a probabilistic representation of spatial economic effects under uncertainty, enabling scenario-based inference in data-constrained contexts. The BN methodology, based on probability theory, is well suited for modelling complex systems with interdependent variables to achieve spatial management objectives under uncertain future conditions (Bromley et al., 2005). BNs have been recognised as a tool to support MSP (Maldonado et al., 2022; Pınarbaşı et al., 2019; Stelzenmüller et al., 2010) and to assess the effects of different spatial management scenarios on fisheries (Kruse et al., 2024; Breen et al., 2020; Coccoli et al., 2018).

A BN represents a system through a network of nodes connected by directed edges that capture causal interrelationships and conditional

Table 5
Descriptive statistics of the datasets. Dataset A includes both Small-scale fisheries (SSF) and Large-scale fisheries (LSF), whereas Dataset B considers only LSF, and Dataset C focuses exclusively on SSF.

Dataset	Descriptive statistics	Port distance (metres)	Fishing effort (Normalised into 0–1)	Other income (€/DAS)	Fixed costs (€/DAS)	Energy costs (€/DAS)	Other variable costs (€/DAS)	Gross value landings (€/DAS)	Market price (€/kg)	Fuel price (€/litre)	Gross value added (€/DAS)
SSF and LSF (A)	Min.	115.6	0	0	0	0	0	0	0	0	-5.149
	Median	29,744.9	0.292	0.565	6.633	9.678	3.751	59.300	3.540	0.590	39.787
	Mean	32,800.7	0.309	0.919	6.439	9.995	3.974	62.820	3.683	0.582	43.327
	Max.	108,094.7	1.209	3.661	11.707	44.085	15.642	246.820	6.370	0.780	216.573
LSF (B)	Min.	115.6	0	0	0	0	0	0	0	0	-5.149
	Median	29,744.9	0.272	0.565	6.633	9.201	3.471	54.950	3.540	0.590	35.576
	Mean	32,800.7	0.278	0.489	5.733	9.289	3.548	56.160	3.059	0.510	38.083
	Max.	108,094.7	1	0.565	6.633	44.085	12.760	202.010	3.540	0.590	173.392
SSF (C)	Min.	115.6	0	0	0	0	0	0	0	0	-1.750
	Median	29,744.9	0	0	0	0	0	0	0	0	0
	Mean	32,800.7	0.031	0.431	0.706	0.706	0.427	6.651	0.886	0.109	5.243
	Max.	108,094.7	1	3.096	5.074	43.595	13.637	212.558	6.370	0.780	168.649

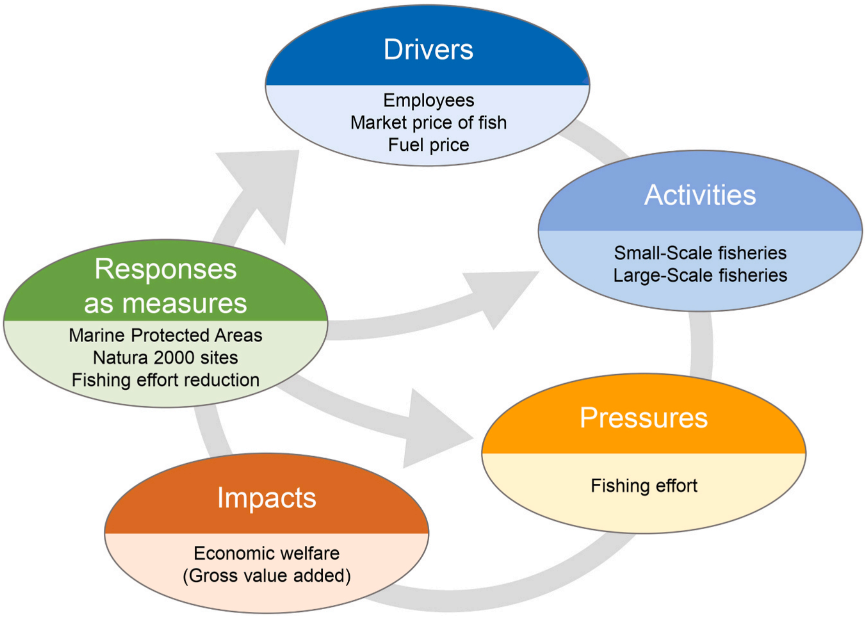

Fig. 3. Conceptual model of fisheries economics considering Drivers, Activities, Pressures, Impacts on human Welfare, and Responses as Measures, adapted from the DAPSI(W)R(M) framework (Elliott et al. 2017).

dependencies among variables (Pearl, 1988). The strength of these relationships is quantified through Conditional Probability Tables (CPTs), which specify the degree of belief, expressed as the probability of a variable assuming a particular state, given the states of its parent nodes (Chen and Pollino, 2012). Decision nodes, in addition to deterministic nodes (i.e., nature nodes), can be incorporated into a BN to model the ability to choose among alternative scenarios. Specifically, decision nodes represent discrete actions that decision-makers can select to compare outcomes among competing approaches. The expected values of all deterministic nodes for each alternative decision enable the identification of the effects of competing strategies, based on the probability structure of the model (Penman et al., 2020). After defining the conceptual model, the influence diagram to depict the relationships of variables within the conceptual model was developed and implemented in Netica software (Norsys Software Corp.; https://www.norsys.com/). The modelling software is based on Bayes' Theorem and machine learning algorithms (Norsys), a subset of Artificial intelligence, to determine the relationship between the selected decision and the operational and economic variables in the geographic condition in which they took place. Continuous variables were discretised using Jenks natural breaks, which maximises variance between bins and minimises variance within bins, ensuring statistically robust classification (Coccoli et al., 2018; Gonzalez-Redin et al., 2016). The resulting bins were used as the states for each node, with nodes scaled into three states, plus a zero state. Nodes presenting negative values were assigned an additional state. In the spatial BN, each node corresponds to a GIS attribute, and each grid cell in the study area was assigned a unique ID to ensure spatially explicit outputs. Three BN models were developed to analyse the outputs for each fishery and their combined effects according to the datasets created. Model A included both SSF and LSF, Model B included only LSF, and Model C included only SSF. Finally, after defining the network structure, the models were trained on the data.

2.2.4.3. Scenario definition and projection. The BN models were run for four real-world MSP scenarios, each defined by the selection of decision nodes representing alternative decision actions associated with management measures planned for the study area. In the "Base scenario", the models were run across all grid cells, without selecting any decision node. Additional scenarios considered areas planned to be closed to fishing for OWF development and associated operational and safety requirements, alongside the assumption that fishing activities would be prohibited in the planned MPAs and Natura 2000. Specifically, the "MPA scenario" focused on grid cells overlapping the planned MPAs, while the "N2K scenario" considered grid cells overlapping the planned MPAs under Natura 2000. Finally, the "OWF scenario" analysed grid cells overlapping the planned OWFs. Table 6 provides an overview of the defined scenarios, outlining the number of grid cells involved in the management measure, their areas, and their relative importance within the overall study area.

A scenario-based approach was adopted in which each management measure was implemented and analysed independently to isolate their individual direct economic effects. This approach supports a transparent ex-ante screening of spatial dependencies and potential trade-offs to inform iterative planning processes. Cell-level GVA projections were calculated for each scenario and expressed as a percentage of the total GVA under the Base scenario. This approach enabled the estimation of the fisheries' dependence on areas subject to management measures in terms of GVA, providing the potential direct economic effect of implementing fishing closures.

3. Results

3.1. *Influence diagram*

The influence diagram incorporates the stock-based dimensions of the DAPSI(W)R(M) framework to integrate fishing effort from predominant fisheries with economic variables across different management measures. The model's structure (Fig. 4) consists of 14 elements and 15 links, representing the economic variables driving the GVA dynamics in the fishery sector and their influencing factors. Three primary management measures (i.e., MPAs, Natura 2000 and OWFs) affect fishing effort, influencing Other variable costs, Gross value landings, and Energy costs. Gross value landings are influenced by the Market price of fish, while Energy costs are affected by Fuel price and Port distance. All economic variables are integrated into the model's output GVA, which is identified as the main management target. By capturing these interactions, the proposed influence diagram enables a systematic assessment of interactions between management measures and fishing activities, providing estimates of total variations in GVA.

3.2. *Bayesian network results*

Three BN models were developed (Supplementary Material, Table S1). Model A considered the combined effects of both SSF and LSF, consisting of 3 decision nodes, 14 nature nodes and 19 links, 7 decision conditions and 6376 conditional probabilities (Fig. 5A). Models B and C were developed to represent LSF and SSF separately, consisting of 3 decision nodes, 13 nature nodes, 17 links, 7 decision conditions and 1500 conditional probabilities (Fig. 5B, C).

3.2.1. *Characterisation of fishing activity*

The Base scenario characterised the spatial distribution of fishing activity across the study area. Results from Model A, combining LSF and SSF, indicated that the overall fishing activity (fishing effort > 0) occurred in 84.8% of the case study area. When analysing the fisheries separately, the probability states of the fishing effort node in models B and C revealed LSF activity in 77.4% and SSF activity in 41.7% of the study area, respectively. Specifically, for LSF, medium fishing effort (0.159 – 0.44) was observed in 29% of the study area, and high (0.44 – 1) in 31.1%. In model C (Fig. 5C), the probability states of the SSF fishing effort showed no fishing activity (zero state) in 58.3% of cases, with medium (0.159 – 0.545) and high (0.545 – max) fishing effort recorded in 16.6% and 10.4%, respectively. LSF incurred significantly higher expenses, primarily driven by energy costs. Variations in the average fuel prices were observed across LSF and SSF, reflecting differences in technical efficiency across fisheries and price variability captured in the reference year (Chassot et al., 2021; Pelletier et al., 2014). Regarding income, LSF exhibited a higher total value of landings, whereas SSF focused on catches with a higher average unit market price. The three most landed species for SSF included *Changeable nassa*, *Common cuttlefish*, and *Mullets nei*, while LSF primarily targeted *European pilchard*, *Striped venus*, and *European anchovy*. Differences in fishing effort distribution, expenses, and revenues between SSF and LSF are reflected in their overall economic output, as captured by the results of the models. Under the baseline scenario, the GVA node indicated positive economic outcome (GVA>0) for LSF in 78.3% of grid cells, whereas SSF exhibited a zero GVA state in 51.8% of grid cells. Positive GVA values in SSF were recorded in 36.6% of cases, with probabilities of medium GVA (25.96 – 90.63) or higher at 24.4%. In Model A, considering both fisheries, the GVA fell within the medium state (28.16 – 77.05) in 39.6% of cases. The probabilities of the negative state of the target nodes GVA across the models were recorded at 12.9% for LSF, 11.6% for SSF and at 9.23% considering both fisheries.

Table 6

Overview of the MPA, N2K, and OWF scenarios, reporting the number of grid cells included in each management measure, together with their total area and relative importance within the overall study area. The MPA scenario refers to grid cells overlapping the planned Marine Protected Areas (MPAs), the N2K scenario to grid cells overlapping the planned MPAs under Natura 2000, and the OWF scenario to grid cells overlapping the planned offshore wind farms (OWFs).

Scenario	No. of grid cells	Area (km^2)	% of the study area
MPA	210	206.38	0.57%
N2K	3737	4090.87	11.35%
OWF	1754	1407.50	3.90%

3.2.2. *Scenario-based analysis and projections*

The BN models were applied to each scenario, focusing on the grid cells overlapping areas subject to management measures integrated into MSP scenarios (Tables S2, S3 and S4). Probability states of the node GVA indicated the magnitude of fisheries' dependence on areas in terms of GVA, providing the potential direct economic effect of assuming closures to fishing (Fig. 6).

Under the MPA scenario, for LSF (Model B), 79% of the designated area fell into a zero-fishing effort state, while 21% was categorised into a low-fishing effort state (0 – 0.159) with no higher values captured. Correspondingly, GVA was recorded as negative in 22.1% of the area, at zero state in 18%, and low in 26.4%. The results of the model indicated that the economic effect for LSF of establishing the proposed MPAs was projected to be limited. Conversely, the SSF (Model C) fishing activity was observed in 87.6% of the MPA-designated area, with the highest probability (50%) falling into the medium state (0.159 – 0.545). GVA for SSF was evenly distributed at approximately 19% each across the low, medium and high states, while negative values (min – 0) and the zero state were recorded in 17.4% and 25.9% of the area, respectively. When analysing the combined effects on SSF and LSF under the MPA scenario (Model A), the aggregate GVA across 12.1% of the area planned for MPA establishment fell into a negative state probability and 7.95% into a zero-state probability. Positive states of GVA were recorded in 80% of the MPA-designated area, with low state (0 – 28.16) and medium state (28.16 – 77.05) at 27.1% and 33.1%, respectively.

Under the Natura 2000 scenario, LSF exhibited significant fishing activity, with medium fishing effort (0.159 – 0.44) recorded in 43.6% of the area and high (0.44 – 1) in 37.6% of cases. GVA for LSF fell into the medium state (24.25 – 69.47) in approximately 42% of the area and high (69.47 – max) in 27.7%. SSF activity, however, was negligible, with no fishing effort captured in 96.7% of the Natura 2000 planned area and GVA at zero state in 73% of the area. When both fisheries were considered, the medium GVA state (28.16 – 77.05) was captured in 49.2% of the area and high (77.05 – max) in 21.9%.

In the OWF scenario, LSF fishing effort fell into the medium state (0.159 – 0.44) in 64.3% of the designated area and low (0 – 0.159) in

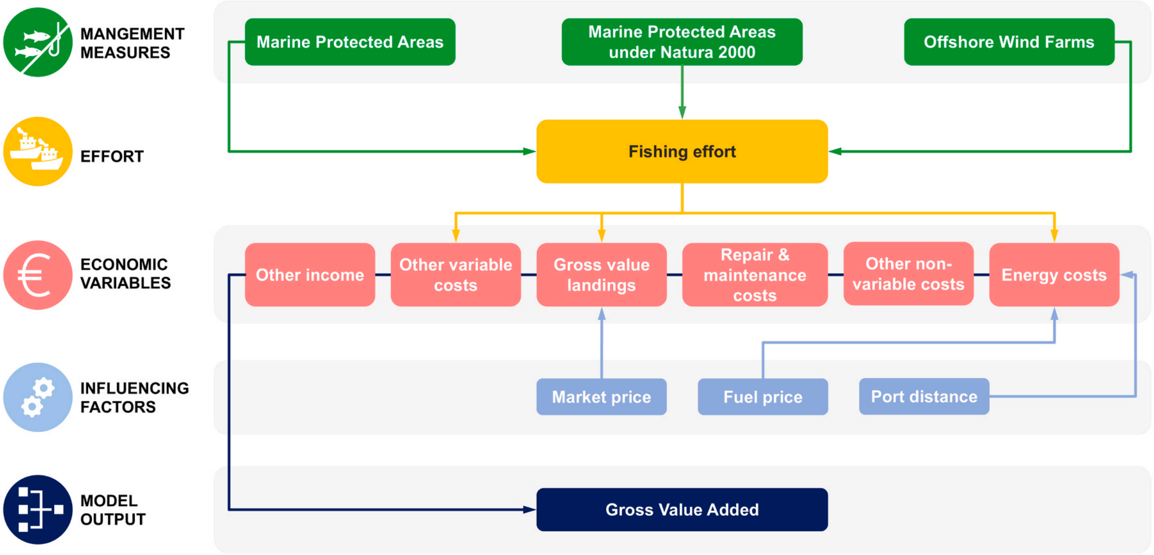

Fig. 4. Influence diagram representing the interrelationships between key economic variables, fishing effort, and scenarios involving management measures.

18.8%. Medium GVA (24.25 – 69.47) was captured in 49% of the area, while low (0 – 24.25) in 21.1% and high (69.47 – max) in 14.6%. For SSF, no fishing effort was recorded in 97.8% of the OWFs planned area and zero GVA in 74% of cases. Considering both fisheries (Model A), GVA fell mostly at the low state (0 – 28.16) in 27% of the area and at the medium state (28.16 – 77.05) at 48.4%.

Projections of GVA for each cell of the analysis grid were derived based on the most probable GVA predicted for each scenario run. The model provided a 'snapshot' of the economic activities operating in the study area based on single-year data layers. The predicted values were then expressed as a percentage of the total GVA of the entire study area under the Base scenario, enabling the estimation of the direct economic effect of scenarios on predominant fisheries (Table 7).

In the MPA scenario, the results indicated a potential direct economic effect on SSF, with 4% of their GVA affected, highlighting the economic relevance of the areas to be designated as MPAs for SSF. No direct economic effects were projected for LSF, indicating that these fisheries operated outside the planned MPAs and had minimal reliance on their resources. In the N2K scenario, the model projected a 1% direct economic effect on GVA for SSF, while LSF experienced a significant potential direct economic effect of 15.3%, reflecting their high dependence on areas likely to be designated as MPAs under Natura 2000 implementation. In the OWF scenario, a 4.8% direct economic effect on GVA was projected for LSF, indicating that the OWFs overlap with valuable areas for industrial fisheries, thereby affecting economic returns. In contrast, SSF was found to be unaffected by OWF deployment.

4. Discussion

4.1. Modelling framework and methodological contribution

The implementation of MSP requires comprehensive consideration of key sectors and their spatial requirements (Kruse et al., 2024). A key challenge for policymakers and practitioners lies in overcoming barriers to stakeholder engagement in addressing the fishery sector within MSP processes (Psuty et al., 2020). In this regard, this study developed a spatially explicit modelling framework based on BNs to integrate fisheries data with management decisions, enabling the evaluation of alternative spatial scenarios and their associated direct economic effects.

The advantage of the use of BN compared with other modelling frameworks involves considering multiple management measures with limited information (Pascoe, 2018). By incorporating decision nodes within the influence diagram, the proposed model informs the structured comparison of planning alternatives while explicitly accounting for conditional dependencies among economic variables. This capability enhances transparency in scenario evaluation and facilitates the communication of potential economic implications to decision-makers and stakeholders (Penman et al., 2020).

Within the influence diagram, scenarios cascade through the system by influencing fishing effort in areas affected by the spatial management measure and, consequently, the associated economic variables. Resulting shifts in revenue and operational costs generate downstream economic effects (Willis-Norton et al., 2024), providing actionable insights for balancing conservation and economic objectives. Consistent with the study objectives, fishing effort is treated as a conditioning variable to estimate direct economic effects under fixed behavioural assumptions. The adoption of GVA represents a defining feature of the framework, as it captures the economic contribution generated by fisheries across space, enabling the identification of economically relevant fishing grounds and supporting cross-sectoral comparisons to assess trade-offs among competing maritime uses. Such an approach is well suited for managing potential conflicts and supporting Blue Growth objectives within MSP (Gambino et al., 2024). Finally, the replicability of the modelling approach is strengthened by the proposed spatialisation methodology, allowing its application in other FAO areas covered by the STECF database.

4.2. Direct economic effects and management implications

The empirical application of the model in the Italian EEZ of the GSA17 revealed marked differences in the projected economic effects across fisheries, underscoring the importance of analysing SSF and LSF separately, as spatial management measures influence each fishery differently (Su et al., 2024; Murillas-Maza et al., 2021). The findings align with previous studies on OWF development and MPA expansion, highlighting the role of spatial restrictions in shaping fisheries' economic performance (Scheld et al., 2022; van de Geer et al., 2013). Planned MPAs were projected to generate no direct economic effects for LSF, whereas SSF exhibited greater dependence on fishing grounds located within the planned MPAs, despite their limited spatial extent (approximately 0.6% of the study area). Given the implementation of the EU Biodiversity Strategy, certain fishing grounds are expected to be closed to fisheries by 2030. As closures are expected to be implemented within the designated MPAs, a worst-case scenario wherein all declared

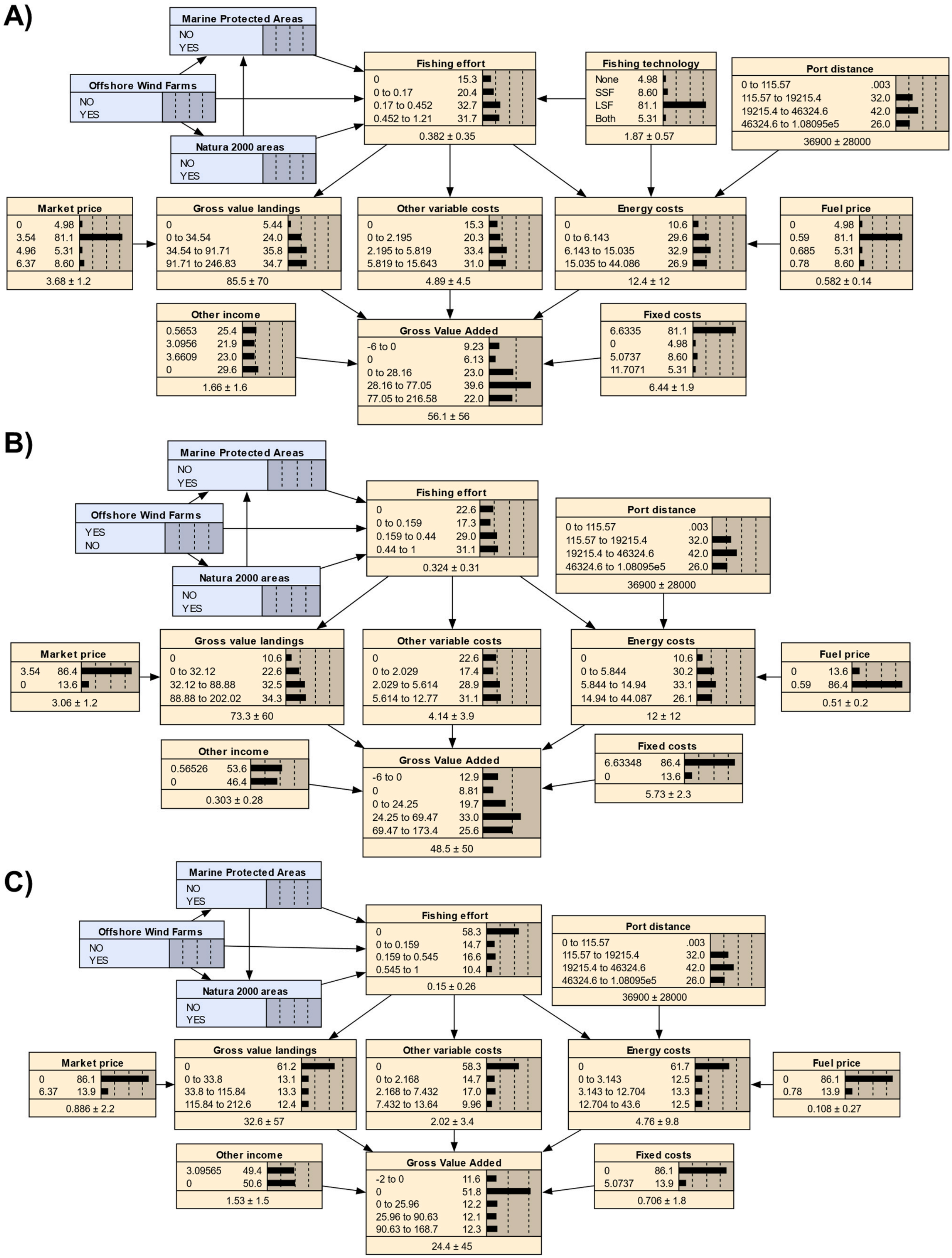

Fig. 5. Trained Bayesian network for the Base Scenario: (A) combined Small-scale and Large-scale fisheries, (B) Large-scale fisheries, and (C) Small-scale fisheries. Node states and discretisation intervals are detailed in Supplementary Material (Tables S2, S3 and S4).

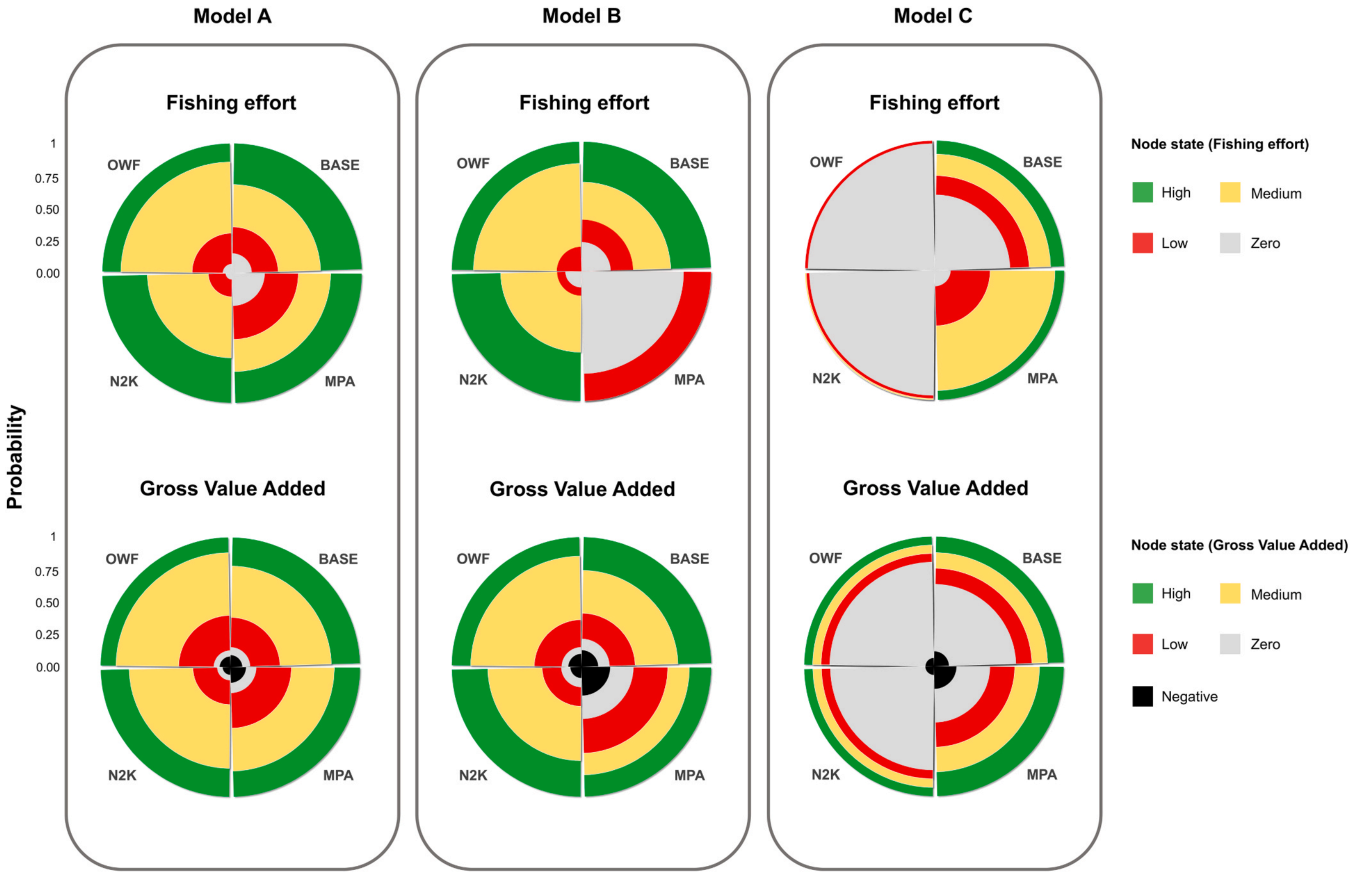

Fig. 6. Probabilities of the nodes fishing effort and gross value added (GVA) under the scenarios considered in the analysis. Model A combined Small-scale (SSF) and Large-scale fisheries (LSF), Model B focused on LSF, and Model C on SSF. In the "Base scenario" the models were run across all grid cells in the case study area, without selecting any decision node. The "MPA scenario" focused on grid cells overlapping the planned Marine Protected Areas (MPAs), the "N2K scenario" considered grid cells overlapping the planned MPAs under Natura 2000, and the "OWF scenario" analysed grid cells overlapping the planned offshore wind farms (OWFs). The names of the node state (i.e., negative, low, medium, and high) referring to the discretization intervals are detailed in Supplementary Material (Tables S2, S3 and S4).

Table 7

Bayesian Network (BN) estimates of the direct economic effects on predominant fisheries under the MPA, N2K, and OWF scenarios. The MPA scenario refers to grid cells overlapping the planned Marine Protected Areas (MPAs), the N2K scenario to grid cells overlapping the planned MPAs under Natura 2000, and the OWF scenario to grid cells overlapping the planned offshore wind farms (OWFs). Model B represents Large-scale fisheries (LSF), whereas Model C represents Small-scale fisheries (SSF).

Model	Scenarios		
	MPA	N2K	OWF
(B) Large-scale fisheries	0%	15.3%	4.8%
(C) Small-scale fisheries	4%	1%	0%

MPAs would be closed to fishing was considered. The resulting economic effects highlight the spatial dependence of SSF and position their economic viability as a key consideration in MPA design and localised management (Weigel et al., 2015). Given the reliance of SSF on higher-value target species, gear selectivity emerges as a critical lever for sustaining economic viability while supporting sustainable resource use. Selective fishing practices reduce bycatch and the capture of undersized species, contributing to long-term stock productivity (European Commission, 2023) and improving market returns (Coppa et al., 2021; Zampardi et al., 2024). Integrating zoning plans with differentiated protection levels, including highly protected buffer areas where access is restricted to selective gear, could reduce overall fishing pressure while enhancing ecological resilience and supporting long-term economic benefits for SSF (Horta e Costa et al., 2022).

A general gap in information regarding conservation objectives and monitoring activities within Adriatic Natura 2000 areas has been documented, highlighting the need to strengthen governance effectiveness and stakeholder engagement (Gianni et al., 2022). The greater projected direct economic effects on LSF within the planned MPAs under Natura 2000, likely reflecting their broader spatial extent, strengthen the evidence base for informed management planning (European Commission, 2018). Accordingly, proactive stakeholder consultation and careful planning remain essential to prevent conflicts (Zaucha and Kreiner, 2021; Jones et al., 2016), including the early definition of protection goals and the establishment of monitoring frameworks (Vrooman et al., 2022). Active fisher participation may further enhance regulatory compliance and support community-based management approaches (Vindigni et al., 2020; Gonzalvo et al., 2011).

The OWF scenario highlights critical trade-offs between offshore energy development and fisheries-related opportunity costs (Yates et al., 2015). By reducing accessible fishing grounds through safety zones and avoidance behaviour, OWFs effectively function as de facto exclusion areas, even in the absence of formal prohibitions (Fitkov-Norris et al., 2025). Such spatial restrictions are particularly relevant for mobile gear fleets, including trawlers and seiners, which exhibit marked reductions in fishing effort in response to OWF development (Fitkov-Norris et al., 2025). Resulting effort displacement towards alternative fishing grounds or harbours may increase operational costs and intensify spatial competition, with potential implications for fishing efficiency and conflict dynamics (Szostek et al., 2025; Stelzenmüller et al., 2022; Kafas et al., 2018). Within this context, the projected economic effects on LSF underscore the need for cross-sectoral coordination and co-design approaches to minimise spatial conflicts (Stelzenmüller et al., 2021). Strategic siting of OWFs outside traditional SSF grounds may reduce spatial conflicts and support the economic viability of small-scale fisheries. The spatial analysis considered turbine locations and did not explicitly incorporate interactions with associated infrastructure (e.g., export and inter-array cables) or broader ecosystem responses, which may influence future spatial dynamics and cumulative effects (Watson et al., 2025). Although OWFs may generate local ecological effects, including increased benthic biomass and local biodiversity due to artificial substrate introduction (Thomassen et al., 2025), the extent to which co-location translates into population-level fisheries gains remains uncertain and highly context-dependent (Fitkov-Norris et al., 2025; Lloret, 2025). OWF development has been shown to result in spatial redistribution of biomass, modifications in predator–prey dynamics, and scenario-dependent changes in target species biomass and catches (Couce Montero et al., 2025). Accordingly, the emerging “Not in Marine Protected Areas” (NIMPA) perspective cautions against framing OWFs as conservation instruments and instead supports their strategic placement as complementary components within MSP frameworks (Fitkov-Norris et al., 2025; Lloret, 2025). Explicit consideration of cumulative spatial constraints arising from overlapping management measures is therefore essential for regional planning (Kruse et al., 2024). Where co-location is feasible, integrated planning approaches may optimise space use and have the potential to reduce planning costs (Schupp et al., 2021; Yates et al., 2015). The projected direct economic effects provide an empirical basis for policy discussions on mitigation strategies, including compensation frameworks where appropriate. However, the model assesses potential direct economic effects under fixed behavioural assumptions and does not explicitly capture adaptive responses, indirect impacts, or heterogeneity in vessel-level financial consequences linked to differences in operating leverage. Therefore, additional vessel-level analyses would be required to design compensation schemes that adequately reflect such financial heterogeneity. Compensations may help address co-location risks and additional liabilities associated with fishing within OWF areas (Bonsu et al., 2024), while supporting displaced fisheries facing longer travel times or increased competition in available fishing grounds (Van Hoey et al., 2021; Kafas et al., 2018).

4.3. Limitations and future research directions

The findings should be interpreted considering several data-related limitations. Spatially explicit maps were available only for aggregated fishing effort, constraining the representation of gear-specific dynamics and potentially masking intra-fleet heterogeneity. While the SSF and LSF categorisation provides a widely adopted structure, it may obscure organisational diversity within fleet segments (e.g., differences between dredgers and pelagic trawlers). Further disaggregation could enhance future analyses where data availability permits (Kinds et al., 2021). Additionally, the absence of multi-year spatial fishing effort data prevented the linkage between annual economic variables and interannual effort variability, limiting the ability to associate species-specific market prices with fishery-level temporal dynamics. This constraint reflects a broader challenge in fisheries management regarding the availability and quality of spatially explicit data (Murillas-Maza et al., 2023). Consequently, the model provides a static assessment of direct economic effects for the reference year (2019) and may not fully capture current fisheries conditions in the study area. The framework is, however, designed to be iteratively updated as new data become available, supporting adaptive MSP processes. Extending the framework towards a dynamic BN would enable time-series evaluation of system dynamics (Gacutan et al., 2019), including climate-sensitive drivers such as fuel and market prices that are expected to influence the future trajectory of fishing industries (Hamon et al., 2021).

Beyond data-related constraints, social dimensions were not explicitly incorporated, given the focus on direct spatial economic effects. The model should therefore be interpreted as one component of a broader evidence base to be complemented by socio-cultural analyses.

While conservation measures and OWFs are likely to be implemented concurrently, isolating individual effects within the present analysis improves causal attribution and minimises bias associated with overlapping spatial interventions. Future research could extend the framework by integrating cumulative spatial interactions to assess the combined economic effects of concurrent management measures on key fisheries. The framework can further support a transition from a stock-based to an ecosystem-based approach by adopting a fisheries socio-ecological perspective that accounts for natural capital dependencies and ecosystem service provision (Gacutan et al., 2019).

5. Conclusions

This study develops a spatially explicit Bayesian network framework to estimate the direct economic effects of alternative spatial management scenarios on fisheries. The applied framework proves to be useful for practitioners and policymakers conducting spatial economic analyses, serving as a tool to inform decision-making processes for sustainable fisheries management. By assessing the fisheries’ dependence on areas subject to management measures and quantifying the potential direct economic effect of implementing fishing closures, the proposed approach helps identify areas of potential conflict and facilitates stakeholder discussions throughout MSP processes.

A key methodological advancement lies in the spatial allocation of economic variables combined with the explicit modelling of conditional dependencies among revenues, costs, and fishing effort, allowing uncertainty propagation and supporting scenario-based inference. The integration of GVA as a management target in the analysis enables the valuation of fishing grounds based on their contribution to the national economy, thereby improving the assessment of trade-offs between planning objectives and the economic viability of the fishery sector. In this regard, the model can inform planners and practitioners in identifying planning solutions to balance conservation and economic objectives effectively while preventing fisheries from incurring the overall costs associated with the spatial allocation of MPAs and OWFs. Moreover, the framework is designed to assess economic effects at the fishery level, including both LSF and SSF. This is particularly relevant for SSF, as the model highlights their locally estimated economic contribution, supports their inclusion in MSP processes, and helps balance conservation efforts with the livelihoods of coastal communities. Additionally, the model’s results can inform the design of adaptive management strategies, ensuring the alignment of ecological, economic, and social objectives. This study establishes a methodological foundation for advancing spatial economic assessments of fisheries within MSP and provides a basis for future research integrating temporal dynamics and natural capital dependencies.

CRediT authorship contribution statement

Ibon Galparsoro: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Investigation, Formal analysis, Conceptualization. **Andrea Mattia Pacifico:** Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Giulio Malorgio:** Writing – review & editing, Supervision, Methodology, Data curation, Conceptualization, Validation. **Arantza Murillas:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Investigation, Conceptualization, Formal analysis. **Luca Mulazzani:** Writing – review & editing, Supervision, Investigation, Methodology, Visualization.

Declaration of Competing Interest

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

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Data avail ability

Data will be made available on request.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.fishres.2026.107745.

Data availability

Data will be made available on request.

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