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The value of bottom trawling in Europe

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Corresponding Author:	Katherine Millage National Geographic Society UNITED STATES
First Author:	Katherine Millage
Order of Authors:	Katherine Millage
	Juan Mayorga
	Sara Orofino
	Trisha Atwood
	Alan Friedlander
	Louise Teh
	Maria Palomares
	U. Sumaila
	Enric Sala
Abstract:	Commercial bottom trawl and dredge fisheries are active across much of Europe, with more than half of the seabed area trawled every year in some regions. These fisheries remain contentious; significant ecological and economic damages have been well documented. Yet, they remain a source of food and provide jobs and economic revenue. Considering recent pushes to reduce bottom trawling effort in European waters, we assess how the social costs associated with this practice might compare to the benefits it provides. We find that society may be losing out to the private sector. This finding is driven primarily by climate impacts from the release of organic carbon from seabed sediments following disturbance by bottom-trawling. Despite uncertainty in how trawl-induced carbon remineralization is parameterized, the associated climate costs to society outweigh the benefits in most scenarios. Further, we show that bottom trawling occurs in a significant portion of Marine Protected Areas (MPAs) across Europe. We argue that phasing out bottom trawling in MPAs could yield meaningful net benefits if done correctly.
Response to Reviewers:	

Title

The value of bottom trawling in Europe

Authors

Katherine D Millage[a,b*], Juan Mayorga[a,b], Sara Orofino[b1], Trisha B. Atwood[a,c], Alan M. Friedlander[a,d], Louise S.L. Teh[e], Maria L.D. Palomares[e], U. Rashid Sumaila[e,f,g], Enric Sala[a]

Affiliations

[a] Pristine Seas, National Geographic Society, Washington, DC, 20036, USA

[b] Environmental Markets Lab, University of California, Santa Barbara, Santa Barbara, CA, United States

[c] Department of Watershed Sciences and the Ecology Center, Utah State University, Logan, UT, United States

[d] Hawai'i Institute of Marine Biology, University of Hawai'i, Kaneohe, HI 96744, USA

[e] Institute for the Oceans and Fisheries, University of British Columbia, Vancouver, V6T 1Z4, BC, Canada

[f] School of Public Policy and Global Affairs, University of British Columbia, Vancouver, V6T 1Z2, BC, Canada

[g] Department of Agricultural Economics and Rural Development, University of Pretoria, Pretoria, South Africa

*Corresponding author. Email: kmillage@ngs.org

[1] Present address: California Oceans Program, The Nature Conservancy, Sacramento, CA 95811, USA

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Authors declare that they have no competing interests.

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All data, code, and materials used in the analysis are available at https://github.com/pristine-seas/net-benefits-bottom-trawling.

Supplementary material

Materials and Methods

Figs. S1 to S20

Tables S1 to S9

Supplementary Data File 1

Title

The value of bottom trawling in Europe

Abstract

Commercial bottom trawl and dredge fisheries are active across much of Europe, with more than half of the seabed area trawled every year in some regions. These fisheries remain contentious; significant ecological and economic damages have been well documented. Yet, they remain a source of food and provide jobs and economic revenue. Considering recent pushes to reduce bottom trawling effort in European waters, we assess how the social costs associated with this practice might compare to the benefits it provides. We find that society may be losing out to the private sector. This finding is driven primarily by climate impacts from the release of organic carbon from seabed sediments following disturbance by bottom-trawling. Despite uncertainty in how trawl-induced carbon remineralization is parameterized, the associated climate costs to society outweigh the benefits in most scenarios. Further, we show that bottom trawling occurs in a significant portion of Marine Protected Areas (MPAs) across Europe. We argue that phasing out bottom trawling in MPAs could yield meaningful net benefits if done correctly.

Keywords

Marine conservation, fisheries management, bottom trawling, CO_2 emissions, societal value

Highlights

- Average annual net value of bottom trawling in Europe is negative (–€2.07 to –€15.97 billion per year)
- CO_2 emissions from disturbed seafloor sediment yield the largest quantified societal cost (–€4.87 to –€18.15 billion per year); societal costs not quantified here include the loss of benthic habitat, the direct and indirect impacts to other fisheries, and pollution from lost fishing gear
- The value of protein for human consumption yields the largest quantified societal benefit (€2.46 billion per year); societal benefits not quantified here include downstream employment benefits in the seafood processing and distribution sectors
- Reducing bottom trawling effort across Europe could yield meaningful net benefits
- 23.2% of European bottom trawling effort (in hours spent fishing) occurred within the boundaries of MPAs

1. Introduction

Bottom trawling has been a common fishing practice across much of the continental shelf and upper slope around Europe for centuries, targeting a range of bottom dwelling fishes, crustaceans, and bivalves (Eigaard et al., 2017). But the ecological impacts of bottom trawling (Hiddink et al., 2017; Kaiser et al., 2006, 2002; NRC, 2002), large carbon emissions (Atwood et al., 2024; Sala et al., 2021), and need for government subsidies to prop up unprofitable operations (Sumaila et al., 2010a; Sumaila et al., 2019) have led many to question whether continuing the practice is in society's best interest. Here we explore this question by providing a first estimate of the net value to society from bottom trawling in European waters, explore overlap between bottom trawling activity and marine protected areas (MPAs), and discuss potential policy pathways and implications associated with reducing bottom trawling effort.

Significant ecological damages associated with bottom trawling are well documented globally; for example, reductions in habitat complexity (Kaiser et al., 2002), permanent changes in the composition of seabed communities (Kaiser et al., 2006; Pitcher et al., 2022), and reduced productivity (Hiddink et al., 2017). Many bottom trawl fisheries are nonselective, yielding average bycatch (i.e., catch of non-target species) rates that range between 31-55% of the total catch (Lucchetti et al., 2021; Steadman et al., 2021). The discarding of undersized or non-target species also remains a problem in some bottom trawl fisheries (Tsagarakis et al., 2014), with observed discard rates of 30% or more (Diamond and Beukers-Stewart, 2011; Lucchetti et al., 2021; Machias et al., 2001). An estimated 60% of global discards come from trawl fisheries (Gilman et al., 2020). These practices can introduce uncertainty in stock assessments, have serious implications for biodiversity and the structure of marine communities, and also are a source of economic loss for the fishery (or other fisheries) (Condie et al., 2014).

The climate impacts associated with active gears such as bottom trawls and dredges are also becoming more apparent. These fisheries emit large quantities of greenhouse gasses and other pollutants such as CO_2 and NO_X as byproducts of burning diesel oil (Coello et al., 2015; Parker et al., 2018) and disturbing sedimentary carbon (Andersen et al., 2024; Atwood et al., 2024; Luisetti et al., 2019; Sala et al., 2021). The economic costs to society of atmospheric CO_2 emissions are well documented (Tol, 2023), and recent studies suggest the annual emissions resulting from disturbance of sediments by bottom trawlers could equate to ~10% of annual global emissions from land-use change (Atwood et al., 2024). Further, the direct economic costs associated with bottom trawl fisheries are significant. Governments spend hundreds of millions of dollars annually to manage these fisheries (Carvalho et al., 2021), in addition to providing the industry with large subsidies (at the taxpayer's expense) to offset the costs of fuel and other operating expenses (Sumaila et al., 2019). Studies suggest that some bottom trawl fisheries would not be profitable without government subsidies offsetting operating expenses (Sala et al., 2018).

Yet, the benefits derived from bottom trawl fisheries must be taken into account (Kaiser et al., 2016). Approximately 26% of wild-caught fish and shellfish globally come from bottom trawl and dredge fisheries (Steadman et al., 2021), accounting for millions of dollars in profits annually. These fisheries support extensive food (and non-food) production systems and provide public benefits in the form of employment, employing an estimated 58.5 million people worldwide in 2020 (FAO, 2022). Indeed, not all bottom trawl fisheries are equal in their impacts–there exist large differences between the impacts of different types of bottom trawl fisheries. For instance, fuel efficiency and benthic macrofauna depletion have been found to be quite different between otter trawl and dredge fisheries, due in part to differences in target species distributions and habitat type preferences (Parker et al., 2018; Rijnsdorp et al., 2020). Further, gear innovations, adherence to strict data collection standards, and increased observer coverage have helped to stabilize overfished marine wildlife populations and reduce pressure on seabed habitats in some bottom trawl fisheries (Hilborn et al., 2021; Steadman et al., 2021).

While the ecological and economic impacts of bottom trawling are well quantified, it remains unclear how the costs of these fisheries compare to their benefits. What are the trade-offs between extraction and conservation? How might reductions in bottom trawling affect these trade-offs? Our intent in exploring such questions here is to help facilitate a more informed discourse about the future of this fishing practice. Although bottom trawling is pervasive across continental shelves worldwide (Amoroso et al., 2018), here we focus on Europe as a case study because it is a data-rich region and the footprint of bottom trawling in this area is one of the most intense and extensive globally (Amoroso et al., 2018; Eigaard et al., 2017). Recent announcements calling for greater restrictions on bottom trawling in Europe–particularly in MPAs–make these questions of utmost policy-relevance (McVeigh, 2024a; McVeigh and Smith, 2024; Struna, 2024).

2. Materials and methods

2.1. European bottom trawling fleet

We define the European bottom trawling fleet as otter trawlers, beam trawlers, and dredge fishing vessels flagged to the 27 EU Member states, the Faroe Islands, the United Kingdom, Norway, Svalbard and Jan Mayen, and Iceland. Only known fishing vessels classified by Global Fishing Watch (GFW) with Automatic Identification System (AIS)-predicted fishing effort in the study area (Fig. S1) between 2016 - 2021 are included. We estimate fishing effort for each vessel-year at a 0.01 x 0.01 degree resolution in units of hours and kilowatt-hours using a neural net model of AIS-inferred fishing effort developed by GFW (Kroodsma et al., 2018)(Table S5).

Vessel monitoring system (VMS) data (and logbook reports of fishing effort)–used by member countries of the EU, the United Kingdom, Norway, and Iceland to track their vessels and to derive official fishery statistics–are not publicly available across the study area. Nonetheless, the use of AIS by commercial fishing vessels in Europe is more widespread than in any other part of the world (Paolo et al., 2024); comparisons between VMS and AIS data for bottom trawlers in important European fishing grounds (e.g., the North Sea) are quite consistent in recent years (Zhang et al., 2024). Therefore, the use of AIS-derived effort in this analysis likely results in fewer omissions of vessels than it would for similar analyses in other parts of the world. The EU requires all fishing vessels of at least 12 m in length to be equipped with a VMS device, and all fishing vessels of at least 15 m to maintain continuous use of an AIS device. It is therefore likely that smaller trawlers were unintentionally omitted from our vessel sample (particularly in the Mediterranean where there are a greater number of small trawlers). We note that this omission likely results in an underestimate of the components which were calculated based on individual vessel-level activity (i.e., CO_2 emissions from fuel and disturbed sedimentary carbon). Most other components in our estimation of current net benefits would be largely unaffected; we allocated reported totals by flag state among all vessels in our sample such that the total magnitude would remain unchanged if the number of vessels were increased.

We make every effort to exclude midwater trawlers from the vessel sample, though we recognize this is not always a straightforward characterization as many vessels utilize different gear types throughout the year. GFW characterizes the gear type of fishing vessels broadly (e.g., “trawlers”, “dredge fishing”), but we refine these characterizations using official registry data. We classify all vessels in our sample into three gear groups based on their primary gear: otter trawls (OT), beam trawls (BT), and dredges (TD) (Tables S1-S2).

Vessels are characterized in our model by flag state, gear group, overall length, gross tonnage, and engine power (Tables S3-S4). When available, GFW obtains characteristics from official vessel registries, and fills gaps using machine learning and regression models (Kroodsma et al., 2018). We obtain estimates of auxiliary engine power from official registries where available, and fill gaps using machine learning models following the approach of Sala et al. (2018)(SM Section 2.2).

We estimate the design speed, specific fuel consumption (SFC), gear width, and gear penetration depth of each vessel following the approach of Sala et al. (2018). We estimate design speed in knots for each vessel as a function of main engine power (*KW*):

$$d = 10.4818 + (0.0012 * \ KW_{main}) - (3.84710 * 10^{-8} * KW_{main}^2)$$

Most trawlers have design speeds between 7 - 15 knots; in order to prevent low estimated design speeds from unduly influencing our results, we assume the minimum design speed for vessels in our sample to be 5 knots. The average design speed across all vessels in our sample is 10.8 knots. The SFC of a vessel reflects the efficiency of the engine and varies with engine type, type of fuel used, engine age, vessel size, and type of activity (European Environment Agency, 2023). We estimate the SFC (g/kWh) of each vessel based on length (Sala et al., 2018)(SM Section 2.3).

Gear width is estimated using vessel-size footprint relationships derived from Eigaard et al. (2016) who studied the gear configurations of trawlers in 13 countries. Based on their general findings, we take a whole-gear approach (WGA) to estimating gear width and penetration depth (van der Reijden et al., 2025) and define the total width, *W*, of the gear footprint (m) as as functions of main engine power (*KW*) and total length (*L*):

$$W_{OT} = 10.6608 * KW^{0.2921}$$

$$W_{BT} = 0.6601 * KW^{0.5078}$$

$$W_{TD} = 0.3142 * L^{1.2454}$$

For many Mediterranean otter trawlers that were able to be matched to official registries, we instead use the following relationship from Eigaard et al. (2016) to define gear width:

$$W_{OT-MED} = 6.6371 * L^{0.7706}$$

We use the average penetration depths of each gear type from Hiddink et al. (2017): 2.44 cm for otter trawls, 2.72 cm for beam trawls, and 5.47 cm for dredges. Though detailed vessel-type information was available from European registries for some of the vessels in our sample, it was not available for all, which precluded the use of a gear component approach (GCA). A GCA approach would allow for finer-scale distinctions to be made regarding the seabed impacts of different fishery segments, but may have also resulted in higher estimated fishing impacts as was found by a study comparing WGA and GCA approaches in three Danish otter trawl fisheries (van der Reijden et al., 2025).

2.2. Estimation of current net value

We estimate private costs and benefits accruing to the fishing industry, as well as public costs and benefits accruing to society associated with bottom trawling in Europe for our vessel sample. We define total net benefits for vessel *i* in year *t* ($U_{i,t}$) as:

$$U_{i,t} = \Pi_{i,t} + \Gamma_{i,t} + \Phi_{i,t} - D_{i,t} - S_{i,t} - \Omega_{i,t}$$

where $\Pi_{i,t}$ are fishing profits, $\Gamma_{i,t}$ is the value of direct employment (i.e., fishers/vessel crew), $\Phi_{i,t}$ is the value of protein provided for direct human consumption, $D_{i,t}$ is the value of discarded catch, $S_{i,t}$ are subsidies (the cost of which are borne by the taxpayer), and $\Omega_{i,t}$ is the value of carbon released (by way of fuel emissions and disturbed sediment carbon being released back into the atmosphere).

All private benefits and costs accruing to the fishing industry are captured within the calculation of fishing profits, $\Pi_{i,t}$, which includes revenues, subsidies that directly offset operating costs, fuel costs, labor costs, and other operating costs. We assume all remaining benefits and costs accrue to society, though some may be captured more locally (i.e., within Europe), while others have more global implications.

2.2.1. Fishing profits

We calculate fisheries profits as revenues minus costs, plus the value of subsidies provided directly to the fishing industry to offset costs. For each vessel *i* in year *t*, profits are calculated as:

$$\Pi_{i,t} = R_{i,t} - C_{i,t}^{total} + (E_{i,t} * s_f^{op})$$

where $R_{i,t}$ is revenue, $C_{i,t}^{total}$ are total vessel operating costs, and $E_{i,t} * s_f^{op}$ are subsidies provided by the government that directly offset operating costs. s_f^{op} is the average rate of subsidization for flag state *f* per unit effort ($/kWh) for bottom trawlers, only considering subsidies that directly offset operating costs (see Section 2.2.5. for more about subsidies).

We use landed value data from the Sea Around Us (SAU) research initiative (Zeller et al., 2016) for 2016 - 2019 to estimate bottom trawl revenues for all relevant flag states fishing within the study area (SM Section 3.1). We allocate revenues proportionally to each vessel in our sample based on effort by flag state and EEZ (Sala et al., 2018). SAU estimates of catches and revenues are typically greater than official statistics published by national fisheries agencies or regional fisheries management organizations because these reconstructed catches aim to fill gaps where catches may have been un- or under-reported. We note that the use of SAU data in this study likely inflates the value of estimated private benefits.

Following the approach of Sala et al. (2018), we define total operating costs as:

$$C_{i,t}^{total} = \frac{C_{i,t}^{fuel} + C_{i,t}^{labor}}{\varsigma_f^{costs}}$$

where $C_{i,t}^{fuel}$ are total fuel costs for vessel i in year *t*, $C_{i,t}^{labor}$ are total labor costs for vessel *i* in year *t*, and ς_f^{costs} is the average fraction that fuel and labor make up of total fishing costs for bottom trawlers flagged to state *f*.

We estimate fuel costs following the European Environmental Agency's method for estimating emissions from the shipping industry (Coello et al., 2015; European Environment Agency, 2023). For each AIS position, we calculate fuel consumption (Fig. S6A) for both the main and auxiliary engines as a function of the engine power of the vessel (in kilowatts), the specific fuel consumption (SFC, in grams per kilowatt-hour), and the load factor (expressed as a percentage) which represents the engine loading relative to its maximum continuous rate. The load factor (LF) of vessel *i* at any given position *j* can be estimated from the cubed ratio of a vessel's instantaneous speed ($\varrho_{i,j}$) at that position and the design speed of the vessel (d_i):

$$LF_{i,j} = L_{max} * \left(\frac{\frac{\varrho_{i,j}^3}{d_i} + \frac{L_{min}}{L_{max} - L_{min}}}{1 + \frac{L_{min}}{L_{max} - L_{min}}} \right)$$

We assume this to be bounded between a minimum load (L_{min} = 0.2) when engines are idling to a maximum load (L_{max} = 0.9) when vessels are operating at design speed. However, since we are dealing with trawlers, we need to account for high loading factors at relatively low speeds when vessels are towing gear in the water, so we only use the above formulation for non-fishing activity, and assume the load factor of trawlers to be equal to 0.75 when the vessel is fishing (Coello et al., 2015). We assume the load factors of the auxiliary engine to be 0.5 and 0.3 while fishing and cruising, respectively (Sala et al., 2018).

Fuel costs (Fig. S6B) are calculated for the main engine and auxiliary engine using the time spent in the AIS position, fuel consumption, and the annual global average price of fuel (SM Section 3.2). Average annual fuel prices are calculated from reported daily average prices of marine gas oil (MGO) from Bunker Index (Fig. S5) for Europe, the Middle East, and Africa.

We estimate labor costs (Fig. S7) and the average fraction of total costs made up by fuel and labor using data from the EU, Iceland, and Norway (European Commission et al., 2022; Fiskeridirektoratet , 2021; Statistics Iceland, 2024). These data provide estimates of different types of costs for trawlers by flag, vessel-size class, and gear. We calculate annual average labor costs per vessel, per kW, and per GT by flag, size class, and gear group (where available) and use these rates to estimate labor costs for each vessel-year. We also use these data to estimate the average fraction of costs made up by fuel and labor, and use these fractions to estimate total operating costs for each vessel-year (SM Section 3.2).

2.2.2. Employment

Employment is an essential component of a functioning economy and directly contributes to human well-being making it an important indicator for decision-makers. We assume the value of crew employment to be equal to the wages paid to those fishing. Though this represents a cost borne to vessel owners/operators and has an impact on their profits, it is also a benefit provided by the fishery to society. Our approach of using wages as a direct proxy for the public value of employment assumes that the wages being paid accurately reflect the marginal productivity of the labor. This approach may undervalue employment in cases where work is unpaid or when major imperfections exist in the labor market. In practice, there is always going to be some distortion in real-world labor markets, but we believe this to be a reasonable assumption for the European bottom trawling industry (e.g., a single employer does not dominate hiring; unions don’t control wages above the competitive equilibrium). We recognize that this approach may yield a higher social benefit than would alternative approaches such as estimating the true social opportunity cost of employing a worker (Boardman et al., 2018) or calculating the contribution of commercial bottom-trawl fisheries to public fiscal resources (Dustmann and Frattini, 2014).

Employment may provide other indirect benefits that are not captured here (e.g., jobs in one industry often create demand for jobs in other related industries; employed workers spend their wages in local shops which creates demand for jobs in unrelated industries). Our approach also does not capture the non-market societal benefits associated with employment (e.g., life satisfaction, well-being). These types of positive externalities associated with employment mean that the social value of employment may actually be greater than the wages paid.

2.2.3. Protein supply

Some portion of the fish harvested by bottom trawlers goes to direct human consumption and thus represents a source of protein–in addition to other valuable nutrients not considered here–for the population that would need to be replaced. We therefore consider the food-security value of the protein provided for direct human consumption (Fig. S8) as a public benefit provided by bottom trawling to society. We utilize the concept of nutrition-equivalent replacement value to calculate the economic cost of replacing the protein provided to society by European bottom trawlers. This approach estimates how much it would cost at today’s prices to replace the nutrient contribution to the human diet from bottom trawl catches using other foods. Suitable replacement foods are those that provide the missing nutrients at the lowest cost without introducing barriers to access or acceptability (Cifelli et al., 2020; Drewnowski, 2010). We therefore calculate the value of protein provided by vessel *i* in year *t* as:

$$\Phi_{i,t} = H_{i,t} * \varsigma_f^{hc} * \varpi_f * p_t^{protein}$$

where $H_{i,t}$ is the total harvest from vessel *i* in time *t* (mt), ς_f^{hc} is the average fraction of harvest from flag state *f* going towards direct human consumption, ϖ_f is the average protein content of fish harvested by bottom trawlers flagged to state *f* (g protein/mt harvest) , and $p_t^{protein}$ is the market price of a substitutable protein source available domestically ($/g protein). We use data from SAU to estimate bottom trawl catches and the fraction of catches going towards human consumption for each flag state fishing within the study area (Fig. S4). We then obtain the protein content of different species from *rfishbase* (Boettiger et al., 2012). In order to identify the "best" replacement of protein, we use prices of substitutable animal proteins from the Agri-food Data Portal of the European Commission to identify the animal products readily available in each European country that could provide the missing protein for the lowest cost (Cifelli et al., 2020) (SM Section 3.3).

2.2.4. Discards

Some portion of the fish caught by bottom trawlers ends up being discarded or returned to the sea. SAU estimates discards as a component of reconstructed catches for each EEZ area, and we use these to estimate the magnitude of discards associated with bottom trawling by flag state across our study area (Fig. S3C).

Discards are often viewed as a waste of fishery resources, particularly in cases when other fisheries operating in the same region target and retain the discarded species. For example, species discarded in industrial fisheries may be targeted by artisanal fisheries or juveniles discarded in one fishery may be targeted by another fishery as adults. In such cases, the value of discards might be quantified as the potential loss to the other fisheries (Diamond and Beukers-Stewart, 2011), but this is often difficult to discern in practice. Discards may also reflect a source of uncertainty for the managers of these other fisheries. High rates of discarding may also have an ecological effect, negatively influencing the biodiversity and community structure of an area. On the flip side, some studies have shown discarding is a more economically efficient strategy for fishers, and banning discards can cause severe economic losses for the fishery (Prellezo and Villasante, 2023).

We conservatively estimate the value of discards from vessel *i* in year *t* as:

$$D_{i,t} = d_{i,t} * p_t^{ex-25th}$$

where $d_{i,t}$ is the total magnitude of discards from vessel *i* in time *t* (mt) and $p_t^{ex-25th}$ is the 25th percentile of the landings prices for species harvested by fleet *f* ($/mt). In this way, we are assuming the economic value of discards from bottom trawling to be only a fraction of that of the same magnitude of landed catches of comparable species, but not zero (Fig. S3D).

2.3.5. Subsidies

Fisheries subsidies are classified in this data based on the scheme applied to multiple iterations of global fisheries subsidies estimates made by Sumaila et al. (2019, 2016, 2010a). We assume that only the portion of total subsidies with the potential to be capacity enhancing ("bad" subsidies) directly offset operating costs for vessels and factor into the calculation of fisheries profits. However, we assume that the total cost of fisheries subsidies are borne by taxpayers.

We use estimates of fisheries subsidies provided by each flag state to industrial fisheries from Schuhbauer et al. (Schuhbauer et al., 2020), scaled to only include the fraction being provided to bottom trawlers within our study area. These calculations, made by SAU, were made based on the fraction of each flag-states' total landed value coming from bottom trawl fishing in the study area. We then calculate rates of subsidization by flag state and use these rates to estimate vessel-specific subsidies (Fig. S3E). This approach assumes that rates of subsidization remain constant by flag state across years, but the magnitude of subsidies provided each year will differ.

2.2.6. CO_2 emissions

We estimate two carbon costs associated with bottom trawling: the value of carbon lost via CO_2 emissions from burning fuel (gasoline or diesel), and that from disturbed sedimentary carbon being remineralized into aqueous CO_2 and then released back into the atmosphere via aqueous-atmospheric gas transfer.

We assume the total value of carbon released into the atmosphere as a result of the bottom trawling activities of vessel *i* is equal to:

$$\Omega_{i,t} = CE_{i,t} + CD_{i,t}$$

where $CE_{i,t}$ is the total value of carbon emitted from burning fuel by vessel *i* in year *t* and $CD_{i,t}$ is the total value of organic carbon stored in ocean sediments that is remineralized into aqueous CO_2 by way of trawling and then transferred to the atmosphere.

From our formulation of fuel costs, described previously, we can estimate the amount of fuel consumed (mt) by each vessel (Fig. S6A). The total value of carbon emitted from burning fuel for vessel *i* is then given by:

$$CE_{i,t} = FC_{i,t} * \nu * \left(\frac{p_t^{CO_2}}{(1+d)^t}\right)$$

where ν is the emissions factor, $p_t^{CO_2}$ is the social cost of carbon in time t (\$/mt of CO_2), and *d* is the discount rate. We assume the emissions factor to be constant at 3.17 mt of CO_2 emitted per mt of fuel consumed (Corbett et al., 2009; Greer et al., 2019)(Fig. S6C).

We estimate the value of disturbed sediment carbon that makes it way into the atmosphere as CO_2 as:

$$CD_{i,t} = m * \tau * I_{i,t} * \eta * \left(\frac{p_t^{CO_2}}{(1+d)^t}\right)$$

where *m* is the estimated amount of carbon stored in the first meter of sediment (mt), τ is the depletion factor accounting for historical depletion of the carbon store as a result of bottom trawling and other disruptive activities (Atwood et al., 2024), $I_{i,t}$ is the fraction of carbon that is released back into the atmosphere as a result of the trawling activity of vessel *i* in time *t*, η is the ratio of the weight of C relative to that of CO_2 (3.67 mt of CO_2 equals 1 mt of carbon), $p_t^{CO_2}$ is the social cost of carbon in time *t* (\$/mt of CO_2), and *d* is the discount rate (5%).

The fraction of carbon released back into the atmosphere from vessel *i* is estimated as (Sala et al., 2021):

$$I_{i,t} = SVR_{i,t} * \delta_{crd} * \delta_{lab} * (1 - e^{-kt}) * a$$

where $SVR_{i,t}$ (swept volume ratio) is the fraction of sedimentary carbon disturbed by bottom trawling of vessel *i* in time *t*, δ_{crd} is the proportion of carbon that resettles after disturbance, δ_{lab} is the proportion of carbon that is labile, *k* is the first-order degradation rate constant, *t* represents time (1 year throughout our model), and *a* is the fraction of remineralized C that will be transferred from the ocean to the atmosphere. We estimate swept volume ratio as:

$$SVR_{i,t} = \nu_i * \left(\frac{\frac{E_{i,t}}{KW_i} * dist_i * W_i}{A}\right)$$

$\frac{E_{i,t}}{KW_i}$ represents the total time fished by vessel *i* in time *t* (hours), $dist_i$ is the distance fished per hour (m/h), W_i is the width of gear of vessel *i* (m), *A* is the total area of the fishing ground (m^2), and ν_i is the penetration depth of the gear used by vessel *i* as a fraction of the first meter of sediment.

We utilize estimates of the amount of carbon stored in the first meter of sediment in each pixel (Fig. S10) and historical levels of depletion (Fig. S12) from Atwood et al. (2020). The global dataset of organic sedimentary carbon stocks from Atwood et al. (2020) has gaps for parts in Europe as a result of missing data for predictor variables. Some of these missing values are located in areas of particularly high trawling intensity (e.g., the EEZ areas of Belgium and the Netherlands). To prevent these missing data from resulting in large omissions of estimated CO_2 emissions, we interpolate missing values for organic carbon stocks using a moving-window average with a 150 x 150 pixel grid size (Fig. S11).

We assume the fraction of carbon in each cell that resettles in the same cell after trawling to be constant at 0.87 based on previous studies quantifying lost sediment loads following disruptive activities (Atwood et al., 2024; Durrieu De Madron et al., 2005; Sala et al., 2021). For the remaining carbon, we use sediment type as a proxy for estimating the labile fraction of carbon in each pixel following the approach of Sala et al. (2021). Pixels where more than 50% of the area is made up "fine" sediments (muds or silts) are assigned a labile fraction of 0.7; pixels where more than 50% of the area is made up of "coarse" sediments (gravel) are assigned a fraction of 0.286; the remaining pixels with a "sandy" makeup (combinations of other sediment types) are assigned a fraction of 0.04 (Fig. S13). The labile organic carbon in sediments is the portion which has been shown to be most susceptible to trawling impact (De Borger et al., 2021); our model only considers the labile fraction of carbon in the resuspended fraction of sediment to be capable of being remineralized.

The model used here is a regionally scaled version of that used by Atwood et al. (2024) and we use the same first order degradation rate constants for each pixel as previous studies (Atwood et al., 2024; Sala et al., 2021). These values were based on oceanic regions using the best available values from the literature as follows: Atlantic = 1.00, Mediterranean = 12.3, Arctic = 0.275 (Fig. S14). We recognize there is uncertainty associated with this parameter due to limited empirical data. Some studies have suggested that the first-order degradation rates utilized by previous studies (Atwood et al., 2024; Sala et al., 2021) are too high. Though we have intended to be conservative in our parameter choices given the best available scientific estimates, we acknowledge there is current disagreement in the literature about the best values for certain parameters, in part because the processes surrounding the storage and trawling-induced remineralization of organic carbon in benthic sediments are complex and empirical validation is lacking (Zhang et al., 2024).

Plankton is a key source of sedimentary organic carbon (Middelburg, 2018); upon being deposited on the ocean floor, the organic material is not immediately available for use by benthic flora, fauna and microbial communities. Natural remineralization of organic carbon is largely controlled by exposure to oxygen (Hartnett et al., 1998). Thus, the primary mechanism by which trawling remineralizes organic carbon is because the physical resuspension of sediment (O'Neill and Summerbell, 2011) can increase exposure time to oxygen (as well as preventing the deposition of new material)(Middelburg, 2018). However, trawling may also increase lateral transport of resuspended organic matter near the seafloor, which could actually yield small increases in sedimentary carbon storage in other areas (Mengual et al., 2019). The role played by benthic flora and fauna in meditating storage of organic carbon in sediments is also important–these organisms aid in both the remineralization of and preservation of sedimentary organic carbon (e.g., by way of bioturbation and bio-irrigation)(Epstein et al., 2022). Severe impacts of bottom-trawling activity on the complexity, biomass, and productivity of benthic faunal communities are well documented (Hiddink et al., 2017; Kaiser et al., 2006, 2002; Pitcher et al., 2022), but some studies have shown the impacts of trawling on benthic faunal communities to partially offset losses of organic carbon from direct physical disturbances (e.g., by way of increased photosynthesis)(Epstein et al., 2022; Zhang et al., 2024). However, other studies have suggested that the effects on primary productivity may be negative (Tiwari et al., 2025).

There is general consensus in the literature that bottom-trawling changes the natural flux of organic carbon stored in marine sediments. However, studies have yielded mixed estimates of the directionality and magnitude of these effects, largely reflecting the spatial variability and complexity of the process (Epstein et al., 2022). In a review of published studies measuring changes in the organic carbon content of seabed sediments due to mobile demersal fishing activities, 29% of studies reported decreases in organic carbon in fished sites compared to unfished control sites, 10% reported increases in organic carbon in fished sites, and the remaining 61% reported findings of no significant effect (Epstein et al., 2022). The authors of this study note that these findings are not exhaustive–more research is needed–and the findings must be interpreted with care since most reports were qualitative (or semiquantitative) and they did not set quality or validity standards for inclusion of studies in their review. Many of the studies reporting findings of no significant effect shared certain characteristics: they were lacking controls (i.e., comparing sites of different fishing intensities rather than fished to unfished sites), were conducted in sandy environments which have less organic carbon, and only considered shallower sediment depths (Epstein et al., 2022). Altogether, the study concluded that these factors would have made it difficult to detect impacts from mobile demersal fishing gears.

Given the complexity of mechanisms driving bottom-trawling effects on storage of organic carbon in ocean sediments and the large number of unknowns that still exist, we can not model all of the (often conflicting) pathways that drive this process across Europe. Recognizing that there are many nuances that will not be captured by our model, we explore how our results might change if the first order degradation rates were significantly lower. To be extra conservative, we consider reducing this parameter by a factor of 10 (Table S9). This makes the full range of k values considered here to be comparable to those used in other studies (ranging from 10^{-3} to 3 yr^{-1})(Zhang et al., 2024).

It is also important to consider that not all remineralized carbon will make it back to the atmosphere. Based on the findings of Atwood et al. (Atwood et al., 2024), we set the fraction of remineralized carbon that will be transferred from the ocean to the atmosphere to be 0.55. CO_2 emissions from disturbances to the sediment would likely make it to the atmosphere over a 9 year horizon (Atwood et al., 2024), but we attribute them here to the year in which the trawling activity occurred.

2.3. Projections of future net benefits

We explore potential outcomes associated with changes in bottom trawling effort based on the following framework that describes net benefits as a function of the fishing effort (*E)* in time *t* and the stock biomass (*B*) in time *t*:

$$U_t = \Pi_t(E_t, B_t) + \Gamma_t(E_t) + \Phi_t(E_t, B_t) - D_t(E_t, B_t) - S_t(E_t) - \Omega_t(E_t)$$

We assume fishing profits, protein supply, and discards to be functions of effort (*E*) and stock biomass (*B*). Direct employment, subsidies, and the value of emitted carbon are assumed to be functions of effort (*E*).

Our projections of future net benefits associated with European bottom trawl fisheries couple all components of the framework used to estimate current net benefits (described in Section 2.3) with a simple fisheries production model. We use the estimation of current net benefits and average characteristics of all vessels in our sample to parameterize this model (SM Section 4). For all future projections, we use a discount rate of 5%.

2.3.1. Stock growth

We use a Pella-Tomlinson production model to describe the underlying population dynamics for stock biomass in discrete time, defined as:

$$B_{t+1} = B_t + \frac{\phi + 1}{\phi} g B_t \left(1 - \left(\frac{B_t}{K}\right)^{\phi}\right) - H_t$$

where B is stock biomass (mt), $g \in (0,1)$ characterizes population growth rate, K is the carrying capacity (maximum population size for growth to be positive, mt), and H_t is total harvest across all vessels in time

t. We assume the ratio of stock biomass providing the maximum sustainable yield (B_{msy}) relative to the carrying capacity to be equal to 0.4 (Thorson et al., 2012), which corresponds to $\phi = 0.188$.

2.3.2. Parameterization

We parameterize the stock production model for an aggregate "trawlfish" stock, defined by the characteristics of the main species comprising bottom trawl landings in the study area. SAU estimates bottom trawl catches of more than 500 different species (or species groups) by flag states in our vessel sample within the study area between 2016-2019 (Zeller et al., 2016). Nonetheless, the catches of many species are trivial. By weight, the top 10 species (or species groups) represented in the SAU bottom trawl catches included in this analysis made up 67.67% of all catches between 2016-2019. Considering the top 20, 30, and 50 species (or species groups) accounts for 81.10%, 85.3%, and 91.17% of all catches by weight respectively. We use the characteristics of the top 50 species (or species groups) by catch weight to define the aggregate stock.

Biological parameters can have significant impacts on the outcomes of simple stock projection models like the one used here. Since we are using an aggregate stock, composed of many species with different growth rates, biomasses, and carrying capacities, we consider multiple scenarios and explore the effects of how uncertainty in these parameters may impact our results (Fig. S17). In cases where the stock is faster growing and/or closer to its carrying capacity (B~K), we find net economic benefits (encompassing all public and private costs and benefits) to generally be greater (SM Section 4).

2.4. Spatial overlap between bottom trawling and MPAs

We explore spatial overlap between bottom trawling activity and marine protected areas in our study area. Using the AIS-inferred estimates of fishing effort aggregated at a 0.01 x 0.01 degree resolution for all vessels in our sample between 2016 - 2021, we calculate the portion of effort that occurred within the boundaries of European MPAs.

Following the approach of Rechberger et al. (2025), we combine the 2024 version of the World Database of Protected Areas (WDPA), with existing MPA assessments (Pike et al., 2024), and protection information from Protected Seas (ProtectedSeas, 2024) to identify relevant MPAs and their protection level. In order to be included in this analysis, a protected area must encompass a marine area that falls (at least in part) within the study area. We remove terrestrial protected areas, Other Effective area-based Conservation Measures (OECMs), and overlapping polygons from this dataset. We then clean the remaining protected areas and recalculate the marine area of each MPA. Any zones smaller than 0.001 km^2 and those where less than 1% of the total area is marine are then removed.

The MPA classification system of Rechberger et al. (2025) utilizes classifications based on the level of protection offered against abatable activities made by the MPA Guide (Grorud-Colvert et al., 2021). "Incompatible" means that the zone offers no protection against abatable activities (e.g., allows mining or intensive commercial fishing), "minimal" means that allowed activities may have a high total impact, but that the MPA still meets IUCN criteria, "light" means that allowed activities may have a moderate impact, "high" means that allowed activities may have minimal impact, and "full" means there is no impact from extractive or destructive activities (Grorud-Colvert et al., 2021). "Unassessed" areas are those for which no classification information is available (often due to recent implementation and/or incomplete management plan documentation detailing allowed activities). "Unprotected" areas are those not classified as an MPA.

3. Results and discussion

3.1. Characterizing net value

We find the average net value associated with bottom trawling activity in Europe between 2016 - 2021 to be negative: between –€15.97 and –€2.07 billion per year (Fig. 1C). Calculating net value by beneficiary (public vs. private) demonstrates a large disparity (Fig. 1B): a net benefit accrues to the private sector (€0.18 billion per year on average) while a net cost is borne by the public (–€16.15 to –€2.25 billion per year). While annual fishing revenues (€4.41 billion on average), the value of protein produced for human consumption (€2.46 billion), and the value of direct fisheries employment (€1.78 billion) are significant, the associated costs are substantial. Average annual government subsidies supporting bottom trawl fisheries (€1.17 billion) and the lost value from discarded catches (€220 million) represent considerable expenses. However, we find the value of atmospheric CO_2 emissions from disturbed sedimentary carbon to be the largest cost falling on society, ranging between –€18.15 billion and –€4.87 billion annually depending on the social cost of carbon.

Our framework likely underestimates costs in a few ways: (a) we only include industrial trawlers carrying an Automated Identification System (AIS) device which primarily affects the magnitude of estimated CO_2 emissions from both fuel and sediment, (b) we do not include the economic costs arising from the loss of benthic habitat, and (c) we do not quantify the direct and indirect impacts to other fisheries in the region, many of which are less economically advantaged than industrial bottom trawlers. Our framework likely underestimates benefits in a few ways: (a) we do not include downstream public and private benefits associated with the seafood processing, distribution, and retail sectors, and (b) we do not include the replacement value of protein that goes towards feeding aquaculture (which may ultimately go towards human consumption). The relative magnitudes of these omitted costs and benefits are more difficult to discern. Many studies have suggested that the costs arising from benthic habitat destruction or impacts to other fisheries could be significant (NRC, 2002). Estimates from the U.S. suggest that the value added from downstream seafood industries (excluding imports) could be more than double that accruing directly to commercial harvesters (NOAA, 2022). However, the contribution of imports to the value of similar industries in Europe is much higher (AIPCE-CEP, 2024), which would in turn significantly lower the portion of this benefit that could be directly attributed to domestic fisheries.

There are limitations associated with our method of estimating atmospheric CO_2 emissions from disturbed sedimentary carbon. The complex landscape of processes dictating changes in stocks of sedimentary organic carbon are influenced by benthic flora, fauna, and microbes, the physical composition of sediments, and the physical, biological, and chemical attributes of the surrounding water column (Epstein et al., 2022). Many of these processes are thought to vary at a fine-scale, but there remain many unknowns. Our finding that trawl-induced CO_2 emissions from disturbed sedimentary carbon were on average 112.4 million mt annually between 2016 - 2021 does not directly account for the impact of trawling on benthic fauna.

Other studies have highlighted the uncertainty associated with certain parameters used in our type of model–much of the current debate centers around the assumed first-order degradation rate, a value for which relatively few empirical estimates are available (Zhang et al., 2024). Review of 78 empirical studies of ocean carbon first-order degradation rate shows that the majority (78%) of measured first-order reactivity rates range between 0.01 and 100 (SM Section 3.6). The average first-order reactivity rate reported across all studies is 2.97 (Fig. S20). The values used in our model range between 0.275 and 12.3 (Fig., S14). Nonetheless, it has been suggested that the values used in our model should be lower (Hiddink et al., 2023; Khedri et al., 2025; Zhang et al., 2024). If we reduce this parameter in our model by a factor of 10 in line with those suggestions, that would reduce the CO_2 emissions estimated by our model by a factor of 2.9 (38.4 million mt on average, Table S9). Even under such a scenario, we find the value of these emissions to be significant: between –€6.20 billion and –€1.66 billion annually, comparable to total fishing revenues.

The larger source of variability in our results surrounding the value of trawl-induced CO_2 emissions from sediment stems from uncertainty associated with the ***cost*** that we place on atmospheric CO_2 emissions. We use the social cost of carbon (SCC) here to estimate the economic costs associated with CO_2 emissions from bottom trawling. The SCC is a monetary estimate of the economic damages–resulting from changes in productivity of production systems, damages associated with sea level rise, and declines in human health and labor productivity–associated with emitting one additional ton of CO_2 into the atmosphere. Estimates of the SCC vary widely depending on the assumptions put into the models used to calculate them: notably the types of damages considered, the assumed discount rate, and the scale of damages being considered. Higher estimates tend to be generated based on the assumption that damages from CO_2 emissions originating in one location affect the whole world, while lower estimates have assumed that damages are localized. Higher estimates also tend to place a higher weight on the future, thus raising the estimated economic damage and the SCC. Estimates of the SCC have been increasing through time as we have gained a better understanding of the full scale of the impacts resulting from atmospheric CO_2 emissions (Rennert et al., 2022; Tol, 2023).

We feel that there are a number of factors justifying the use of a higher cost placed on atmospheric CO_2 emissions. The price of CO_2 under the world's largest carbon market–the EU Emissions Trading System (ETS)–was between €95.2–109.9 per tonne in March 2023 and carbon taxes in some European countries have ranged upward of €120 per tonne. Further, there is growing evidence that markets and taxes tend to undervalue CO_2 emissions and the SCC should be even higher (Rennert et al., 2022; Tol, 2023).

Given the variability in estimates of the SCC, we conservatively use €43 and €161 per metric tonne as low and high end estimates of the value of atmospheric CO_2 released from bottom trawling. If we value CO_2 emissions on the low end of this range, our results would suggest that bottom trawling in Europe yields a relatively neutral net benefit (slightly negative assuming a higher first-order reactivity rate; slightly positive assuming a lower first-order reactivity rate). In either case, the majority of costs still fall on society, while the majority of benefits are captured by the fishing industry. Under a reality where CO_2 emissions are valued closer to the top end of the range in SCC we consider–which we believe to be more realistic based on the values of carbon markets and carbon taxes–the costs of bottom trawling outweigh the benefits. This outcome remains unchanged with a lower first-order degradation rate.

3.2. Projections of future value: Might less be more?

Using our characterization of current net value, we simulate how changes in fishing effort might impact the balance of costs and benefits in the future. We find that reductions in bottom trawling effort could yield greater net benefits as compared to the status quo across a wide range of potential effort scenarios (Fig. 1D). In cases where the climate impacts of bottom trawling are valued more (i.e., a higher SCC), the significant public costs arising from CO_2 emissions nearly always outweigh private benefits, suggesting the optimal level of bottom trawling would be nearly nil. Even if we value CO_2 emissions conservatively (i.e., a lower SCC), we still see gains from reducing bottom trawling effort. We find that permanently reducing bottom trawling effort in aggregate across Europe by more than 50% could maximize net benefits under this scenario (Fig. 1D), yet significant (albeit lesser) costs still accrue to society.

These results, although simplistic, underscore the potential for transitions away from bottom trawling to yield meaningful climate benefits. Of course, achieving such outcomes is contingent on there being no activity leakage–that is, the effort to be reduced should be permanently eliminated (as assumed in our simulations) and must not be allowed to relocate elsewhere. Reducing bottom trawling effort will likely impact the private benefits accruing to the fishing industry in the short term, but long-term benefits to the fishing industry are certainly possible, especially for fisheries targeting stocks that are already overfished or are slower growing (Fig. S17).

3.3. Bottom trawling in MPAs

The World Database of Protected Areas includes more than 6,000 implemented and designated marine protected areas in European (EU, UK, Norway and Iceland) exclusive economic zones, encompassing a total area of over 900,000 square kilometers (Fig. 2A). We estimate that the footprint of bottom trawling encompasses 25.6% (~1.87 million km^2) of the European EEZ area considered here (Fig. 3), but MPAs offering strong protections against bottom trawling ("fully" or "highly" protected, see Section 2.4 for more on MPA classifications) only encompass 0.07% (~5,000 km^2) of the study area (Fig. 2).

We find that on average, 13.04% of all trawling effort (kWh) each year (Fig. 3) between 2016-2021 occurred within the boundaries of implemented or designated MPAs in our study area. In terms of time spent fishing (hours), the figure is higher: 23.17%. However, this figure varies greatly by country–more than 25% of the annual trawling effort in the EEZs of Belgium, Bulgaria, France, Germany, Guernsey, Netherlands, Romania, and Spain occurred in MPAs (Fig. 4B). Additionally, we find trawling intensities to be similar inside of MPAs in many countries, as compared to unprotected areas where trawling occurs (Fig. 4C). These results are not surprising, as a recent study found bottom trawling effort to be greater in many MPAs in northern Europe than in nearby unprotected areas (Dureuil et al., 2018).

This finding supports recent policy shifts in Europe seeking to limit bottom trawling in MPAs. In 2022, the UK introduced laws banning bottom-towed gears in four marine protected areas (MPAs), and another law in 2024 that would restrict bottom trawling in an additional 13 MPAs (McVeigh, 2024b). In 2023, the European Commission presented a proposal that would phase out bottom trawling in MPAs across the EU by 2030. In 2024, Greece became the first European country to announce its commitment to ban bottom trawling in all MPAs within its waters by 2030 (McVeigh and Smith, 2024). Only a few months later, Sweden announced its intention to also ban bottom trawling in Swedish territorial waters (12 nm from shore) (Struna, 2024). Many have been skeptical as to whether these actions would actually affect bottom trawling effort in a meaningful way–protected areas are generally imagined to confer protection against damage or harm arising from human activities. Yet many MPAs in Europe are only minimally protected (Grorud-Colvert et al., 2021), and as evidenced here, there is still a great deal of overlap between the footprint of European bottom trawlers and MPAs.

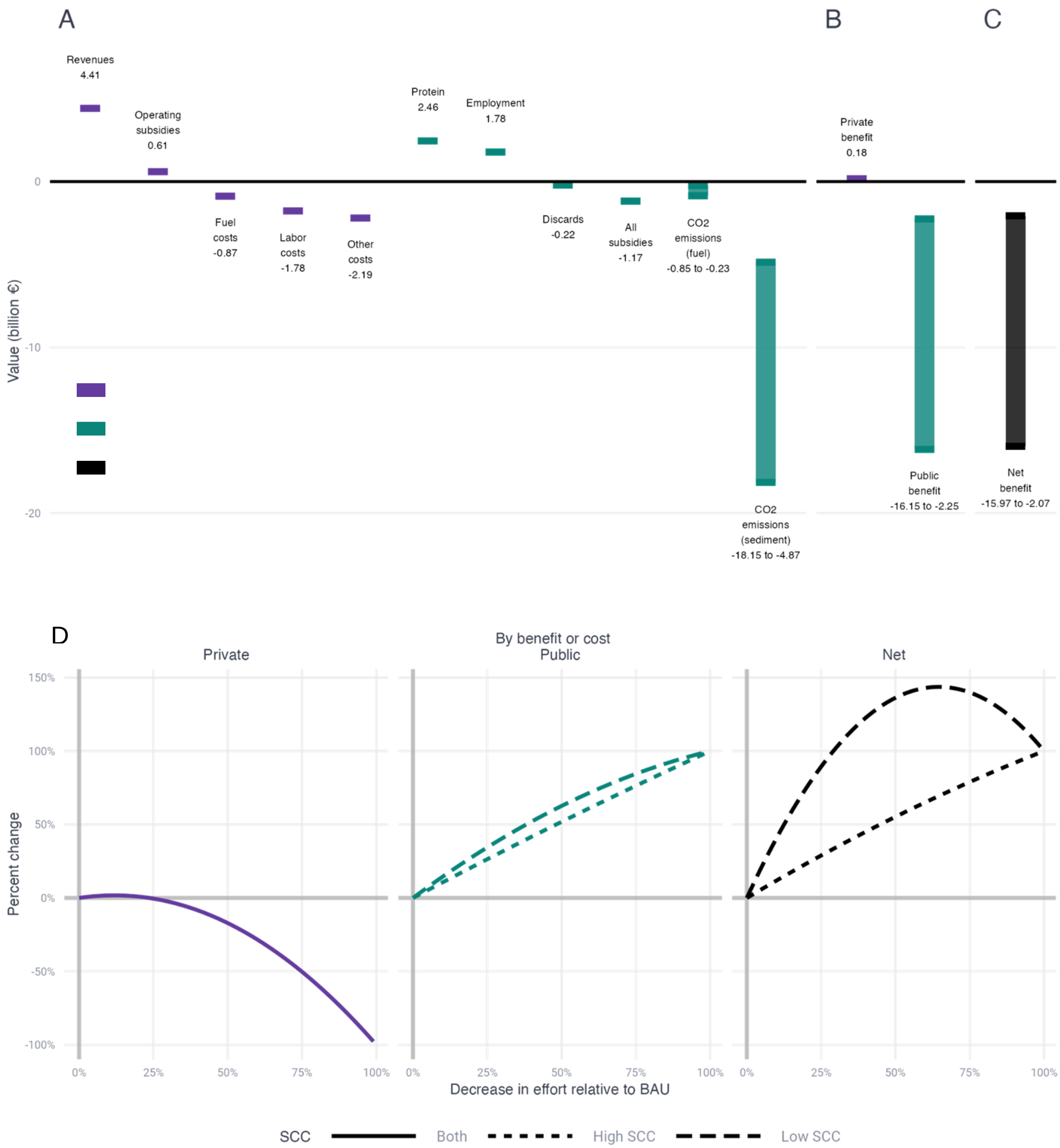

Fig. 1. Value of benefits and costs associated with bottom trawl fishing in Europe (2016-2021). The height of the bars (A-C) indicates the average magnitude of each cost or benefit annually (in billions of Euros). Bars are colored (A-B) based on the beneficiary. Ranges associated with CO_2 emissions (A-C) stem from high (€161/mt) versus low (€43/mt) assumed social costs of carbon. (D) Simulated annual net benefits (2050) are shown by beneficiary as a function of effort relative to a business as usual (BAU) scenario where effort continues unchanged. The different line types (D) depict the value placed on CO_2 emissions. SCC: Social cost of carbon.

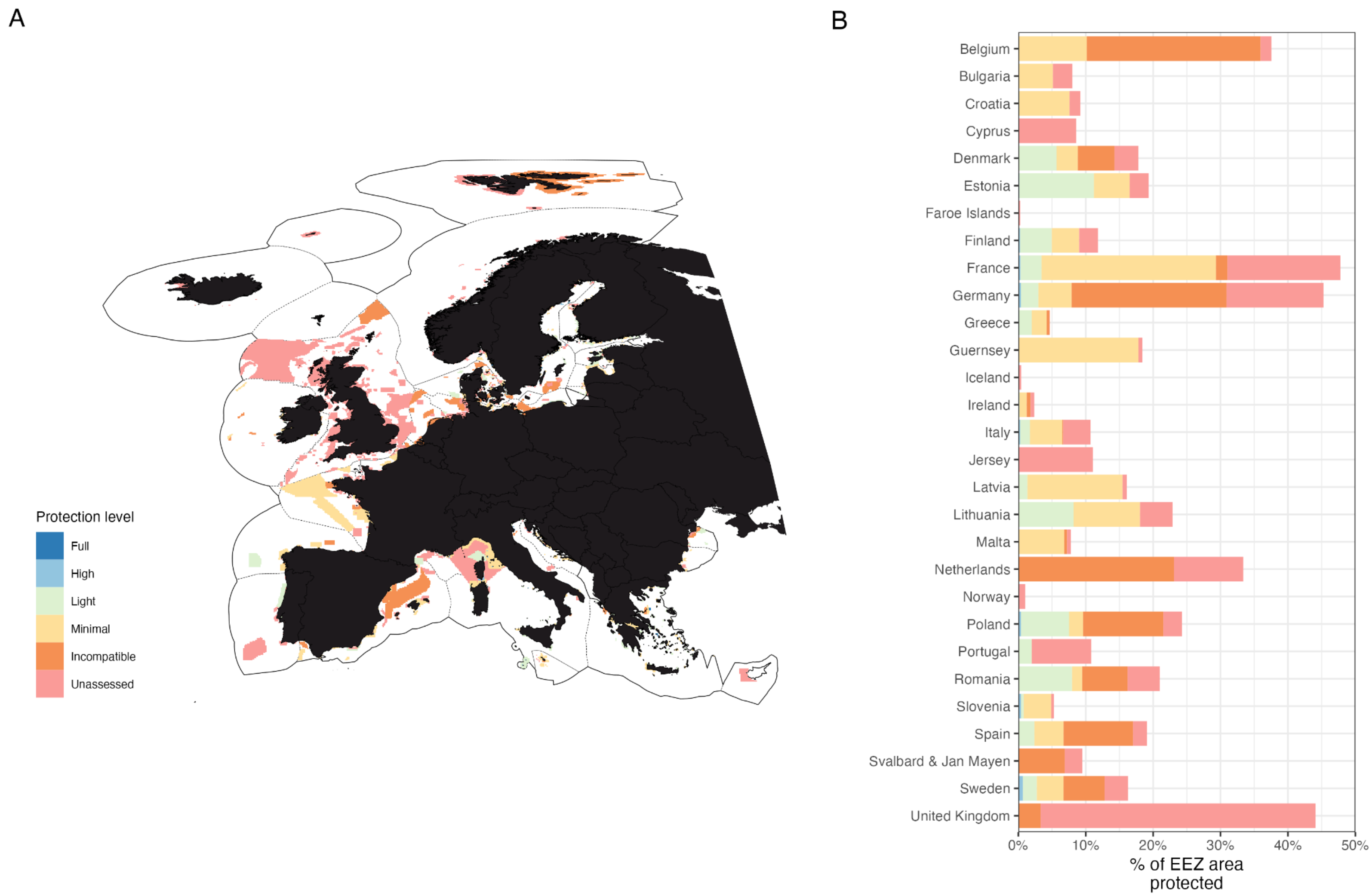

Fig. 2. Protections offered by European MPAs. (A) MPAs considered here classified based on their level of protection offered. (B) Percent of each EEZ area protected by MPAs of different classifications. MPA boundaries are from the World Database of Protected Areas (2024) and classifications are from Rechberger et al. (2025) based on MPA Guide.

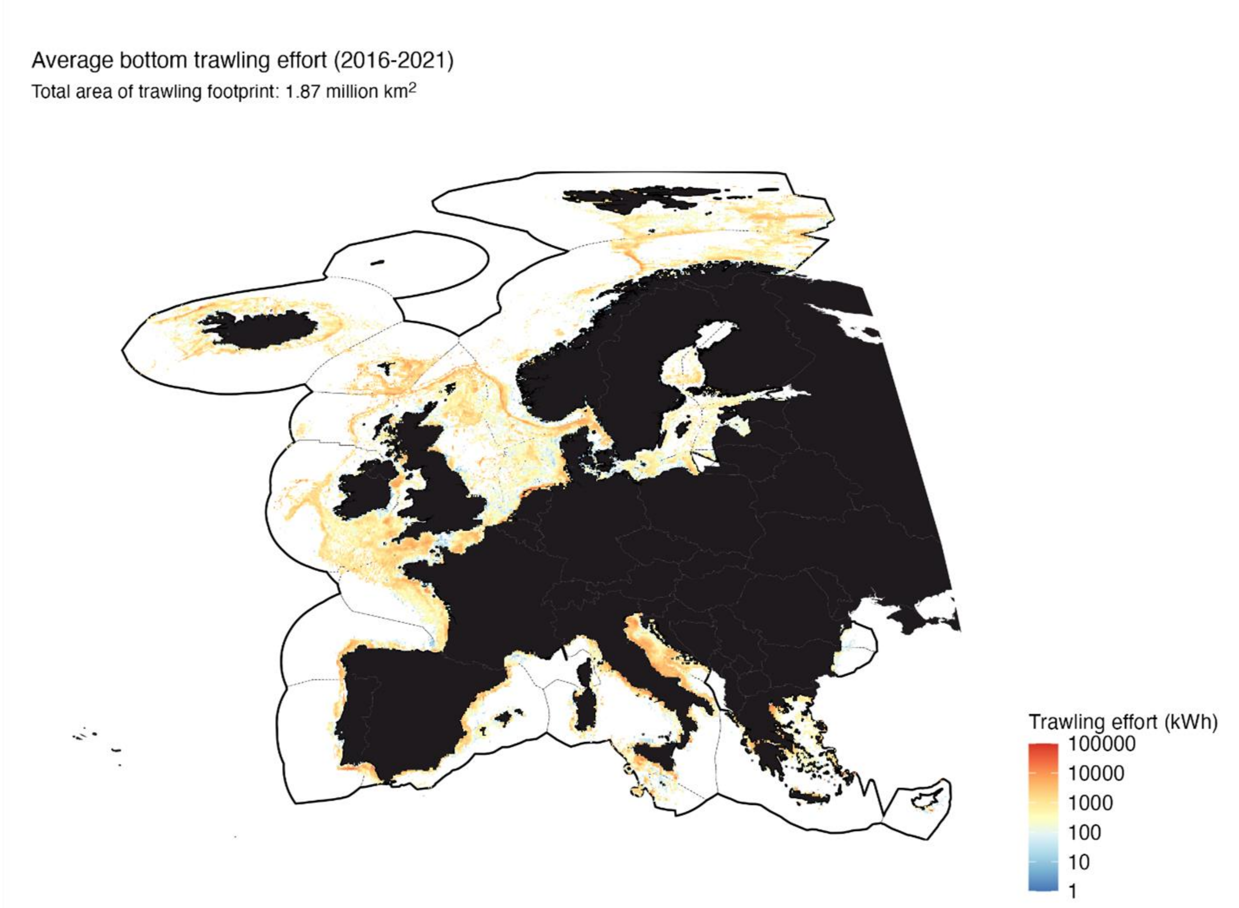

Fig. 3. Bottom trawling effort in Europe. Average annual trawling effort (kWh) between 2016 - 2021 aggregated by 0.01 x 0.01 degree. Effort is from Global Fishing Watch (Kroodsma et al., 2018).

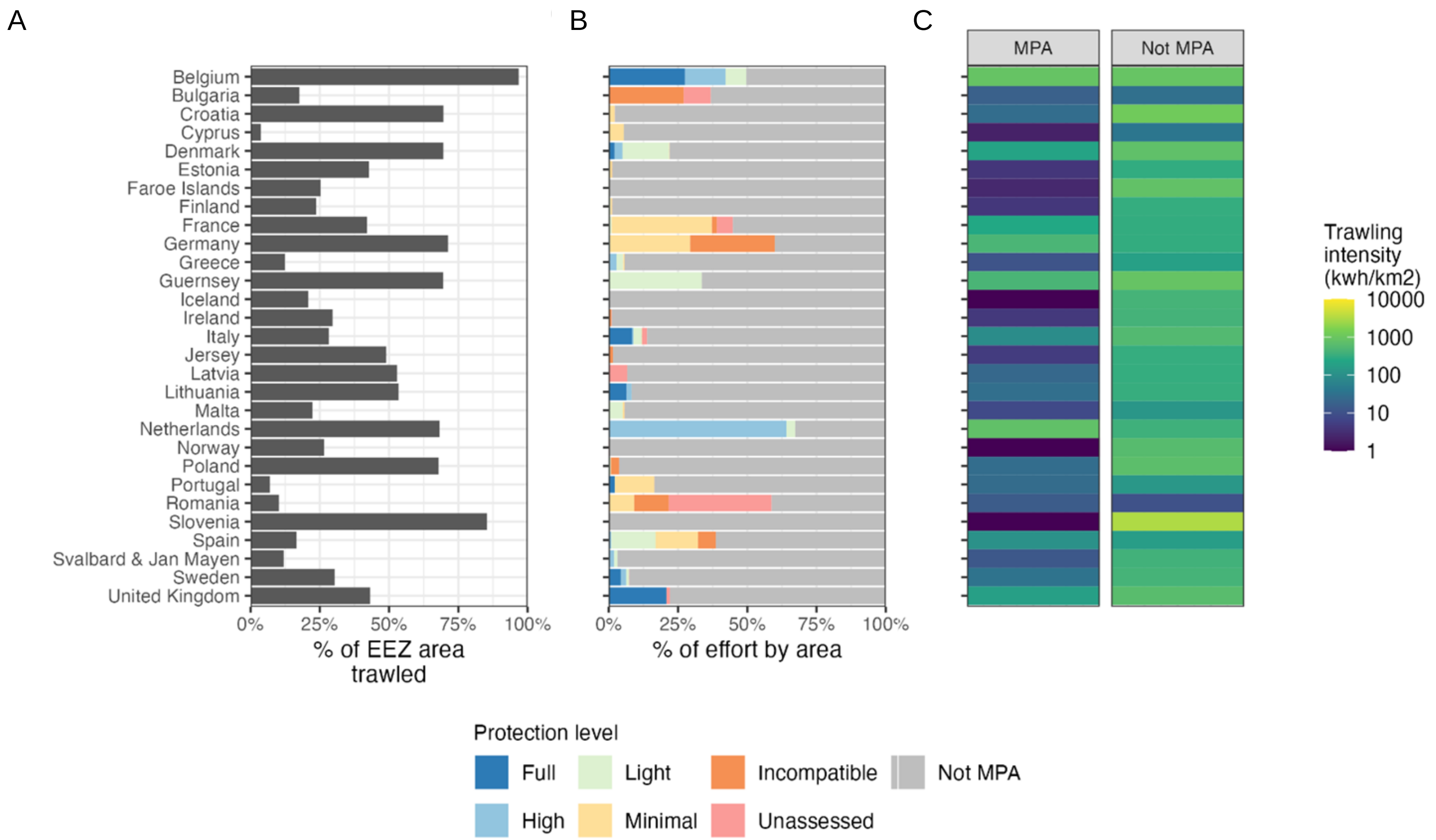

Fig. 4. Bottom trawling activity in European EEZs. (A) Percent of total EEZ area trawled on average between 2016 - 2021. (B) Percent of total trawling effort by protection status of area. (C) Trawling intensities (kWh/km^2) by area. MPA boundaries are from the World Database of Protected Areas (2024) and classifications are from Rechberger et al. (2025) based on MPA Guide.

4. Conclusions

Our analysis demonstrates that bottom trawling in European waters could generate substantial net costs to society (up to €16 billion annually), primarily through potential climate impacts associated with trawl-induced release of organic carbon from seafloor sediments. While our cost-benefit framework has limitations—notably the uncertainty regarding trawl-induced CO_2 emissions from organic sedimentary carbon (Atwood et al., 2024; Epstein et al., 2022; Middelburg, 2018; Zhang et al., 2024)—the potential magnitude of identified costs warrants serious consideration by policymakers. Further, the omission of well-documented but unquantified costs in this framework (e.g., benthic habitat loss, direct and indirect impacts to other fisheries, pollution from lost fishing gear) is a major limitation (Cau et al., 2017; Hiddink et al., 2017; Kaiser et al., 2006, 2002; NRC, 2002).

Area-based management offers a promising approach to reduce bottom trawling effort and achieve meaningful climate benefits. Eliminating bottom trawling from MPAs could deliver immediate emission and conservation benefits while allowing ecosystem recovery to enhance adjacent fisheries productivity. Targeting carbon-rich sediment areas that contribute minimally to food security could further reduce emissions while preserving economically important fishing grounds. However, realizing these climate benefits requires proper implementation to ensure that bottom trawling effort is not simply displaced to other areas.

Effort-based management approaches offer complementary pathways to area-based restrictions for reducing bottom trawling impacts. Effective effort reduction through measures such as seasonal closures, license limitations, quotas, vessel buyback programs, or trip limits (often used in combination) can significantly decrease both the spatial footprint and intensity of trawling activities (McConnaughey et al., 2020). However, effort-based measures require careful design to prevent unintended consequences, as reductions in fishing time may be offset by increased vessel capacity, technological improvements, or disproportionate compression into sensitive areas, potentially negating conservation benefits (Eigaard et al., 2014). The success of effort-based approaches is particularly important in fisheries with overcapacity, where gear modifications and area closures alone may be insufficient to achieve meaningful habitat protection (McConnaughey et al., 2020).

When there is limited capacity to target species individually–such as is the case with many bottom trawl fisheries–trade-offs between food production, conservation objectives, and profitability are inevitable. Undeniably, there will be short-term costs associated with transitions away from bottom trawling. Europe's existing €1.17 billion in annual bottom trawling subsidies provides a ready funding source for industry transition. Redirecting these harmful subsidies toward license buyouts, vessel decommissioning, and fisher retraining programs offers an economically efficient pathway forward (Steadman et al., 2021; Sumaila et al., 2010b). A more comprehensive accounting of greenhouse gas emissions from the bottom trawling industry, including those resulting from the degradation of long-term carbon stocks in marine sediments, would help to ensure the industry is fully accountable for all its emissions. Incorporating sediment carbon emissions into the EU Emissions Trading System would create additional financing mechanisms, enabling carbon credit sales for permanent trawling reductions while incentivizing climate-friendly fishing practices.

Employment and food security considerations must be carefully managed during any transition away from bottom trawling. While the industry employs approximately 20,000 people directly on bottom trawlers in the EU, small-scale fisheries provide three times more jobs and may benefit from reduced trawling pressure that currently impacts their operations negatively (European Commission et al., 2022). Given that bottom trawling provides only 2% of Europe's animal protein, food security disruption should be minimal compared to other regions more dependent on marine ecosystems (Selig et al., 2019). However, short-term adjustments will be necessary as some catches currently support aquaculture feed systems, requiring careful assessment that alternative protein sources—whether from other fisheries, aquaculture, or terrestrial systems—do not generate greater environmental costs than those being eliminated (McConnaughey et al., 2020).

The scale of required transition support can be contextualized within existing policy frameworks: Europe's annual subsidies to bottom trawling are comparable to the sector's direct employment value, suggesting substantial resources are available for retraining programs and alternative livelihood development. Policymakers should consider whether redirecting these resources could help to foster more sustainable fishing methods that would provide greater employment per euro invested and greater net social benefits. With Europe committed to achieving net-zero emissions by 2050 and halting biodiversity loss by 2030, the transition away from bottom trawling in protected areas could represent a critical opportunity to align fisheries policy with broader environmental commitments while generating substantial economic benefits for society.

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