ICES Journal of Marine Science, 2026, Vol. 83, Issue 5, fsag058
https://doi.org/10.1093/icesjms/fsag058
Original Article

ICES CIEM International Council for the Exploration of the Sea Conseil International pour l'Exploration de la Mer

OXFORD

The effects of marine protected area designation and associated restrictions on fisheries in EU MPAs

Elliot J. Brown[1,*], Karin Johanna van der Reijden [1,2], Nuno Castro [3,4], Stephen C. Mangi [5], José Carlos Mendoza [6], Oliver Tully [7], Mattias Sköld [8], Tamara Cecilia Vallina[9], Gert Van Hoey [10], Katrien Verlé[10], Robert C. Wakeford[5]

[1]National Institute of Aquatic Resources, The Technical University of Denmark, 2200 Kgs. Lyngby, Denmark
[2]van der Reijden Marine Ecology & Fisheries, Rottevalle, 9221 TD, The Netherlands
[3]AQUALOGUS, Engineering and Environment Lda, Lisbon, Portugal
[4]MARE- Marine and Environmental Sciences Centre /ARNET - Aquatic Research Network, Agência Regional para o Desenvolvimento da Investigação Tecnologia e Inovação (ARDITI), Funchal, 9000, Portugal
[5]MRAG Limited, London, W1J 5PN, United Kingdom
[6]Instituto Español de Oceanografía (IEO-CSIC), Farola del Mar 22, Dársena Pesquera, 38180, Santa Cruz de Tenerife, Spain
[7]Marine Institute, Rinville, Oranmore, Co., Galway, H91 KA07, Republic of Ireland
[8]Department of Aquatic Resources, Swedish University of Agricultural Sciences, 453 30, Lysekil, Sweden
[9]Wageningen Marine Research, Wageningen University and Research, Haringkade 1, 1976 CP, IJmuiden, Netherlands
[10]Flanders Marine Institute of Agriculture, Fisheries and Food, 8400 Oostende, Belgium
*Corresponding author. Section for Ecosystem Based Marine Management, National Institute of Aquatic Resources, The Technical University of Denmark, 2200 Kgs. Lyngby, Denmark. E-mail: elbr@aqua.dtu.dk

Abstract

Marine protected areas (MPAs) are central to European Union (EU) marine policy, yet their effectiveness remains contested, especially with regard to ongoing fishing activities. Using officially reported fisheries effort data from vessel monitoring systems and logbooks, we analysed 781 MPAs across four ICES ecoregions to quantify fishing activity and assess responses to MPA designation and the imposition of fisheries restrictions. We aggregated effort by gear type, i.e. mobile bottom-contacting gears (MBCG) and passive or pelagic gears (PPG). We applied post-hoc before-after-control-impact (BACI) analyses using generalized linear mixed models. A total of 60% of MPAs were fished after designation, including 44% with MBCG and 55% with PPG, with regional variation from 40% in the Celtic Seas to 89% in the Bay of Biscay and Iberian Coast. MPA designation alone did not significantly change fishing effort for any gear category. The imposition of fisheries restrictions reduced effort across all gears, with declines of 30% in MPAs compared to 13% in adjacent controls. Similar reductions were observed for MBCG (31% vs 15%) and PPG (26% vs 9%). MPAs with benthic conservation objectives experienced reductions in MBCG effort (25% vs 9%), but no significant changes in PPG nor total effort. These findings indicate that designation without accompanying restrictions does not alter fishing behaviour. Contrastingly, fishing restrictions can reduce effort in MPAs, albeit with strong site-level variability, and depending on the type of restrictions that are put in place. Our results highlight the need to set clear conservation objectives when designating MPAs and to develop follow-on management measures tailored to these objectives. By leveraging official effort data and BACI designs at a regional scale, this study provides robust evidence to inform policy discussions on the effects of MPAs as fisheries management tools in EU waters.

Keywords Marine Protected Areas, fisheries management, ecosystem based management, before-after-control-impact (BACI), european fisheries

Introduction

Marine Protected Areas as conservation and management tools

The concept of spatially explicit areas for the conservation of marine species and habitats originated from terrestrial conservation efforts of the late nineteenth and early twentieth centuries, where areas were demarcated with the primary intention of providing human experience and enjoyment of nature (Björklund 1974). This conservation focus was adopted globally in an ad-hoc manner. In the marine realm, these efforts were reinforced in the late twentieth century with the adoption of Exclusive Economic Zones (EEZ). This granted coastal states sovereign rights, and corresponding responsibilities, to explore, exploit, conserve, and manage resources in specific maritime zones, including the territorial sea and adjacent continental shelf (Humphreys and Clark 2019). Subsequently, various international policy initiatives have

Received: 2 December 2025. **Revised:** 1 April 2026. **Accepted:** 4 April 2026

incentivized coastal states to increase the number and area of marine protected areas (MPAs) in their jurisdictions (Sala et al. 2018; Campbell and Gray 2019). However, efforts to expand MPA coverage have often led coastal states to prioritize the least contested areas (Devillers et al. 2015, 2020), or to implement only minimal or ineffective protections driven by complex socio-economic trade-offs (Andradi-Brown et al. 2023; Arneth et al. 2023). MPAs may be established with various objectives in mind, ranging from the conservation of physical habitats, vulnerable species, through to unique and culturally significant communities (Grorud-Colvert et al. 2021). They may also be designated with broader goals than conservation objectives alone, including the support of tourism, or even integrating fishing into their objectives and management plans (Aminian-Biquet et al. 2024). As marine management has shifted towards sustainable development and ecosystem-based approaches, the previous focus on nature conservation elements driving the designation of MPAs has changed, incorporating social and economic objectives. This tendency has been shown to lead to improved outcomes across both conservation and social objectives (Oldekop et al. 2016), especially when governance structures are devolved to empower local and stakeholder management (Dawson et al. 2021) or even in partnerships with commercial interests (Brooks et al. 2019). Inclusive governance models can address conservation needs while ensuring that social and economic benefits are equitably distributed among local populations, provided they are properly established in contextually appropriate modes (Armitage et al. 2020; Fidler et al. 2022).

Marine protected areas in Europe

In the European Union (EU), the systematic designation of MPAs began under the Birds Directive (Council of the EU 1979; European Parliament and Council 2009a) and continued primarily under the Habitats Directive (Council of the EU 1992). Special protection areas (SPAs) or special areas of conservation were, and are, established to protect specific habitats, species that are identified as vulnerable, or the habitats that support populations of these vulnerable species. For a long time, these directives were the only EU legislation that enabled coastal states to designate MPAs outside their territorial waters. Consequently, the current international network of MPAs across the EU (Natura2000) remains largely based on these Directives. However, the primary focus of these Directives is terrestrial, with only a few broad-scale marine habitats (often coastal) being included in the Habitats Directive, and the Bird Directive focusing predominantly on waders within its scope of 'marine birds'. In 2008, the EU adopted the Marine Strategy Framework Directive (MSFD; European Parliament and Council 2008), which was the first European Directive to focus entirely on the marine environment (Long 2011). Unlike the Birds and Habitats Directives, which focus on species and habitat protection, the MSFD requires EU Member States to achieve or maintain 'Good Environmental Status' (GES) of marine waters. It mandates an ecosystem-based approach to managing human activities, ensuring sustainable use of marine goods and services while safeguarding the ecological integrity of marine ecosystems (European Parliament and Council 2008). As such, it does not limit the scope of potential MPA objectives, provided they comply with the criteria for a GES. Neither the Habitats Directive, Birds Directive, or MSFD have been very effective in providing for ecosystem-level spatial conservation (Friedrichs et al. 2018; Bastardie et al. 2025). True baseline reference conditions for habitats are generally unavailable, and therefore, compliance with conservation objectives and GES is reported relative to impacted baselines (Aminian-Biquet et al. 2025; Gaget et al. 2025). This lack of appropriate baselines reduces the apparent need for management measures to return to GES, instead, increasing the chances of maintaining conditions closer to the 'status quo'. The more recent nature restoration regulation (European Parliament and Council 2024) is designed to enable true restoration rather than maintenance in this respect. While important for the success of MPAs in achieving their conservation objectives, the appropriate definition of baselines is outside the scope of this article.

Recently, there has been increasing policy pressure to designate more marine space as MPAs to protect or improve biodiversity (European Commission 2020; Convention on Biological Diversity 2022) and restore ecosystem structure and function (European Parliament and Council 2024). This increased interest in MPAs as conservation management tools coincides with increased diversity of activities in marine space, such as offshore wind energy (Thomassen et al. 2025). In turn, allocating marine space to new actors and activities can increase conflict with traditional marine users, particularly with the fisheries sector (Püts et al. 2023; Bastardie et al. 2025), and brings tensions in the Maritime Spatial Planning Directive to bear (European Parliament and European Council 2014; Article 5(2)).

Fisheries and marine protected areas

While EU Member States can establish MPAs throughout their EEZs using various domestic legislative instruments tied to, or independent of, these EU Directives (Birds, Habitats, and MSFD), Member States can only unilaterally impose restrictions on fishing activities within their territorial waters (12 nautical miles), as specified in the Common Fisheries Policy (CFP; European Parliament and Council 2013). The imposition of fisheries restrictions outside territorial waters but within EEZs requires EU legislation (i.e. regionalization) that is supported by all affected Member States, because fisheries management is an EU exclusive competency (European Parliament and Council 2013). This process, within regionalization, may be slow and can be hindered by an individual Member State, thereby limiting its effectiveness. However, because of this centralization of fisheries management to the EU, Member States also share the provisions of the CFP that call for minimising negative environmental impacts (Article 2(3)), establishing protected areas in compliance with other EU law (Article 11), as well as establishing restrictions and prohibitions to protect the targeted species themselves (Article 8); an extension of the typical nature conservation objectives to include fisheries management objectives.

The combination of these Nature Directives, the CFP, and the various transpositions and implementations of these instruments into national law has led to significant areas of the EU's EEZ being designated as MPAs (∼11% by surface area, as of 2022; Aminian-Biquet et al. 2024). However, these often lack clear conservation objectives and the restrictions on human activities that may be needed to achieve them (Relano and Pauly 2023; Aminian-Biquet et al. 2024). As fisheries are regularly considered a major source of pressure in marine ecosystems (Pedreschi et al. 2019; O'Hara et al. 2024; Bornman et al. 2025), especially those employing bottom-contacting gears (Eigaard et al. 2016; McConnaughey et al. 2020; Perry et al. 2022), there has been particular focus on estimating

Table 1 Number of sites (MPA and adjacent control area pairs) with different management status relative to when VMS fishing data were available (2012–2021)

Management status	Baltic Sea	Greater North Sea	Celtic Seas	Bay of Biscay and Iberian Coast	Total
Total sites	387	192	119	83	781
Designation pre-2021	385	191	106	81	763
Designation post-2012, pre-2021	100	63	50	47	260
Restrictions pre-2021	138	94	11	57	300
Restrictions post-2012, pre-2021	109	85	6	30	230
Benthic objectives, restrictions post-2012, pre-2021	44	56	5	16	121

fishing activities in these MPAs. Methods have varied from investigating policies and inferring fishing activities from the absence of restrictions (Roessger et al. 2022; Aminian-Biquet et al. 2024), using surveys of individual or local groups of MPAs (Guidetti et al. 2008), to estimating fishing behaviour based on publicly available position and vessel data (Dureuil et al. 2018; Perry et al. 2022). These earlier studies have estimated both high rates of fishing activity in MPAs and high rates of 'high-risk' fishing, when considering specific benthic conservation objectives.

While all aforementioned studies of fishing activities in MPAs provide valuable insights, two critical methodological gaps remain: First, the use of officially reported fisheries data [e.g. vessel monitoring system (VMS) and Logbook data], to reduce ambiguity or uncertainty about the actual fishing practices and efforts. Second, no attempt to attribute changes in fishing effort in response to MPA designation (or the imposition of fisheries restrictions) using methods supporting causal inference, across the breadth of social and environmental contexts in which EU MPAs are established.

This study addresses these gaps by (i) characterizing the extent and types of fishing activity within EU MPAs since their designation or implementation of fisheries restrictions, using officially reported effort data, and (ii) assessing whether fishing effort or practices change following MPA designation or the introduction of fisheries restrictions through a before-after-control-impact (BACI) analytical framework.

Materials and methods

Marine protected area dataset

A previously aggregated database of MPAs from Feary et al. (2025) was utilized. This database contained records of MPAs designated under the Habitats and Birds Directives, as well as nationally designated MPAs using locally specific legal mechanisms, which were recorded in the Common Database on Designated Areas (CDDA; European Environment Agency 2021). This database of 819 MPAs was geographically restricted to exclude the EU's 'outermost regions', namely the EU parts of Macaronesia (e.g. the Azores, Madeira, and the Canary Islands), where VMS data coverage is known to be deficient (Pham et al. 2013; Moura et al. 2025). A total of 781 MPAs were selected from across four different ICES ecoregions (Table 1). Each MPA in the database was coupled to an adjacent control area. These control areas were defined as a 5 km wide area surrounding each MPA and within the marine realm (Fig. 1). These were selected to represent the fishing activities that occur in the vicinity of the MPA but not too removed from the MPA context (e.g. hydrography, bathymetry, geomorphology, and distance to port). In total, 1562 polygons of MPA and control areas over the four ecoregions were considered.

Of the 781 sites (paired MPA and control areas), fifty had no recorded date of designation or equivalent establishment. More than half of the sites had dates of designation or establishment before VMS data on fishing activities were available (pre-2012; 453), and 18 were designated in the final year of the study. This left 260 sites with at least 1 year of data before and after designation (Table 1). Similarly, 465 sites had no recorded date of any fisheries restrictions being imposed. The types of fisheries restrictions recorded in the database included overall fishing bans, catch limits, effort restrictions, as well as fleet- and gear-specific restrictions, where these are made specific to the MPA boundaries. However, there was large variation in the implementations of these broad categories, undermining their utility in this study. The database also included information on the ecosystem components that were stated targets of conservation objectives. Of those MPAs with restrictions, 86 had restrictions in place before VMS data on fishing activities became available, and 16 were imposed only in 2021 or later. This left 230 sites with both pre- and post-restriction periods, of which 121 specified conservation objectives for benthic habitats (physical and biological components attached to, or buried into, the seafloor's surface; Table 1). The sizes of MPAs varied across regions (Supplementary Fig. 1).

Fisheries datasets

Fisheries data consisted of officially recorded VMS and reported logbook data, which were merged by data providers. The EU made the use of VMS mandatory for vessels longer than 15 meters in 2005, and subsequent regulations expanded this requirement to vessels exceeding 12 meters in 2009 (European Council 2009; Breen et al. 2014). As a consequence, VMS were gradually implemented by nation states. VMS primarily operates through satellite technology, providing data on the geographical positioning, course, and speed of fishing vessels at defined intervals. The gradual implementation of VMS across national states and their fleets means that data from this source has been considered reliable at a broad geographic scale since 2012. VMS data can be linked to logbook records, helping to create a more comprehensive

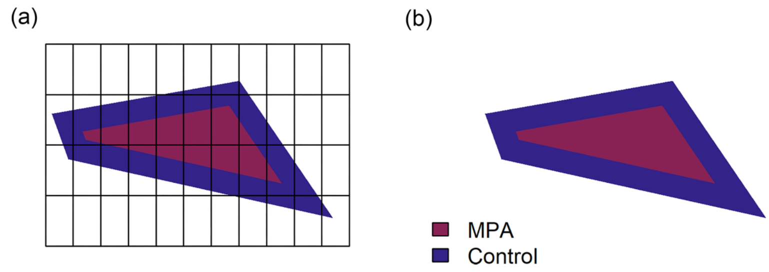

Figure 1 Representations of MPA (inner shaded polygon) and adjacent control area (outer shaded polygon), and how fishing effort data were attributed to these polygons via intersection with overlapping c-squares (a), or directly into the specified polygons (b).

understanding of fishing behaviour and effort distribution across different marine areas (Gerritsen and Lordan 2011).

A data call was issued as part of a European Commission funded project (MAPAFISH) requesting fisheries data from 16 European nation states: 14 from within the EU (Belgium, Denmark, Estonia, Finland, France, Germany, Ireland, Latvia, Lithuania, Netherlands, Poland, Portugal, Spain, Sweden), and two neighbouring, third party countries (Norway and The United Kingdom). The call requested annual estimates of fishing effort in space and time (in absolute fishing hours (h) and vessel power-inclusive fishing hours(kWh) for the period 2012–2021, at *metier* level 6 (Suppl. Mat. 2), based on official reported VMS and logbook data and following ICES guidelines commonly applied to spatial fisheries data reporting (ICES 2019). While no standardized approach for harmonizing effort from passive gears existed, those responsible for organizing the workflow for multinational data collection at this level, determined that this approach provides an appropriate proxy for effort across all gears and across all submitting nations (ICES 2022). Data were requested at two different spatial aggregations, the first applied a grid-based approach Fig. 1a, aligned with standard ICES data calls on fisheries data, aggregating data to cells of 0.05 $\times$ 0.05 degrees of longitude and latitude (a so-called c-square; Rees 2003). The second approach required nation states to intersect the position-based fisheries data with the provided polygons of the MPAs and control areas (Fig. 1b), which we term the direct method. All countries responded to the data call. For three nation states, fisheries data were not available before 2015, 2014, and 2013. In these cases, data were copied, at the metier 6 level, from the first available year backwards (3, 2, and 1 year, respectively).

Fisheries characterization

National records of per-métier fishing activity were aggregated up to coarser gear categories using the International Standard Statistical Classification of Fishing Gear (FAO 2017). These categories were further refined by their interaction with the seafloor, based on additional details from the European Commission's gear reporting codes (Popescu and Breuer 2024) and EU definitions (European Commission 2017), forming a category of mobile bottom-contacting gears (MBCG) and its complement passive or pelagic gears (PPG).

Fisheries data aggregated at the c-square level were summed across countries and coarse gear groups. These were subsequently intersected with the MPA and control polygons to assess fishing activity per MPA and control area. For c-squares that only partially overlapped with a polygon, effort was assumed to be homogeneous within each c-square, with the exact value assigned to an overlapping MPA or control polygon being proportional to this spatial overlap (Fig. 1). For the direct method, fishing data were similarly summed across countries and gear groups, but no spatial intersections were necessary. These processing steps resulted in two datasets of international fishing effort (total, MBCG, and PPG) per MPA and per control: one derived from c-square method and one from the direct method. The estimates derived from these two methods were investigated to identify the most apparently comprehensive dataset, which was then selected for use in subsequent analyses.

To characterize fishing activities within MPAs, the time series per MPA depended on the year in which the management intervention (designation or restriction) occurred. For sites where MPA designation occurred prior to 2012, all years of fishing data were included ($n = 453$). When the year of designation fell within the study period, excluding the final year, only those years following the designated year were included ($n = 260$). MPAs designated in the final year of the fishing data timeseries (2021; $n = 18$) were excluded. When no designation year was known ($n = 50$), only the three most recent years (2018–2021) were included.

A similar selection was performed to characterize fishing activity within MPAs that had fishing regulations in place. All input restrictions (e.g. controls on allowable gear, vessels, or fishing time) specific to the MPA area (i.e. not general restrictions such as catch limits for wider areas), whether permanent or temporally variable, were considered together to investigate any effects across all fishing activities. The year in which fishing regulations came into force was the determining factor. This resulted in 86 MPAs with restrictions before the fisheries time series (characterized by the full time series), 214 restricted during the time series (characterized by a truncated time series), and no MPAs with restrictions in the last year of the fisheries timeseries. No fisheries restrictions were recorded for 465 of the 781 MPAs in this study.

To focus on those MPAs with conservation objectives that included benthic habitats, we retained only MPAs that were designated for physical benthic habitat types or benthic organisms (excluding fish, but including plants and attached macrophytes). Again, for those MPAs matching the conservation objective criteria, that had restrictions imposed prior to 2012, the entire time series of data was utilized ($n = 30$). For MPAs that had restrictions implemented during the period for which we had fisheries data, only those years post-restrictions were included ($n = 100$). For those MPAs that had no recorded year in which their restrictions were implemented ($n = 3$), an average of the last 3 years of fisheries data was used.

Differences in area-standardized mean annual fishing effort (kW.days.km^{-2}) in MPAs between ICES ecoregions were investigated using non-parametric Kruskal-Wallis tests (from the *stats* package of R; R Core Team 2023) and followed by Dunn's test for pairwise comparisons, using Holm's correction for multiple comparisons [as implemented in the R-package *dunn.test* (Dinno 2014)]. Correlations between area-standardized mean annual fishing effort and MPA area were investigated using non-parametric Kendall's rank correlation coefficient (R Core Team 2023).

Impact of marine protected area designation and fisheries restrictions

A BACI approach was used to compare fishing effort in MPAs and their adjacent control areas, before and after a management action (MPA designation or implementation of fisheries restrictions) was taken. These analyses utilized a different subset of the MPA database, where sites were included based on the availability of fisheries data both before and after the management action was taken. For investigating the impact of MPA designation, 260 MPA-control area pairs were available. In contrast, for fisheries restrictions, 230 MPA-control pairs were available, and for fisheries restrictions in MPAs with benthic habitat conservation objectives, 121 MPA-control pairs were available. Nine specific hypotheses were tested:

1. MPA designation changes annual all-gear fishing effort in MPA areas, relative to control areas.
2. MPA designation changes annual MBCG fishing effort in MPA areas, relative to control areas.
3. MPA designation changes annual PPG fishing effort in MPA areas, relative to control areas.
4. Fisheries restrictions change annual all-gear fishing effort in MPA areas, relative to control areas.
5. Fisheries restrictions change annual MBCG fishing effort in MPA areas, relative to control areas.
6. Fisheries restrictions change annual PPG fishing effort in MPA areas, relative to control areas.
7. Fisheries restrictions in MPAs with benthic habitat objectives change annual all-gear fishing effort in MPA areas, relative to control areas.
8. Fisheries restrictions in MPAs with benthic habitat objectives change annual MBCG fishing effort in MPA areas, relative to control areas.
9. Fisheries restrictions in MPAs with benthic habitat objectives change annual PPG fishing effort in MPA areas, relative to control areas.

Two analytical approaches were employed, both utilizing generalized linear mixed models (GLMMs), which were implemented in *glmmTMB* (Brooks et al. 2017). The first approach employed a Time Series BACI (TS-BACI), where annual time series were explicitly modelled such that:

$$g\,(E\,[y]) = \alpha + \beta_1 \cdot BA + \beta_2 \cdot CI + \beta_3 \cdot T + \beta_4 \cdot BA : CI + \beta_5 \cdot BA : T + \beta_6 \cdot CI : T + \beta_7 \cdot BA : CI : T + (1 \mid site)$$

Where g is the link function relating the response to the conditional model, y is one of three different responses: annual fishing effort, annual MBCG fishing effort, or annual PPG fishing effort (all as kW.days). The variable *BA* is a binary indicating whether the observation was made before (reference level) or after the management action (designation or fisheries restrictions) was taken in the MPA. *CI* is a binary representing either control (reference level) or impact (MPA) polygons. T is a numeric variable representing the calendar year, and $(1 \mid site)$ is a random effect of site, where site is an identifier shared by pairs of control and MPA polygons. The parameters α (intercept) and β (coefficients) were estimated during model fitting. The tertiary interaction represents the relative change in trends. In this case, it represents the change in fishing effort trend in the MPA from before to after the management action, relative to the change in the adjacent control area (due to the order of the reference levels in the two binary variables). This interpretation is only valid if the second-order interaction of $CI : T$ is not significant. A significant effect here indicates that the MPA and adjacent control areas exhibited unequal trends in the before period, and thus, the control site is not representative of the trend. In situations where this second-order interaction was significant, the TS-BACI was abandoned, and the second analytical approach was employed. This approach eliminates the time-series component and compares only the relative change in mean annual effort before and after the management action, resulting in only one second-order interaction term. In *glmmTMB*, the equivalent model formulae were:

$$Eff \sim CI * BA * T + (1 \mid site)$$

$$Eff \sim CI * BA + (1 \mid site)$$

Due to the effort data being right-skewed, two distributions were investigated: the log-normal and Tweedie distributions. For the log-normal models, the response was log transformed using $log1p(Eff)$ (where 'log1p' is a more accurate calculation of log(1 + x) for cases where x is small), and the model was fit using a Gaussian distribution with identity link. For the Tweedie distribution, an untransformed response was fit using the Tweedie distribution with a log link function. Because the year in which management interventions were imposed varied by site, and these were rarely in the middle of the timeseries, the numbers of observations before versus after the intervention were unbalanced. To address the unbalanced number of years, we utilized the restricted maximum likelihood estimation function to estimate model parameters, which is known to produce less biased estimates of variance components in unbalanced designs (Bolker et al. 2009; Brooks et al. 2017; Maestrini et al. 2025). The year in which management interventions occurred was included in the 'before' period.

Model fits were investigated using the R package *DHARMa* (Hartig 2024) for standardized residuals. QQ-plots and residuals over covariate groups were visually inspected. Where minor overdispersion was identified, dispersion parameters were allowed to vary with one of the two treatment variables (CI or BA), and these new models were subsequently assessed using standardized residuals. Model selection followed a two-step procedure: (i) goodness-of-fit, assessed via standardized residuals (DHARMa), and (ii) validity of the causal inference assumption, specifically the parallel trends assumption in the TS-BACI design. The significance of the coefficients was determined using non-parametric bootstrapping (1000 iterations of sampling with replacement) to overcome the potentially anti-conservative P-value estimations from the Walt z-tests, as implemented by *glmmTMB* (Bolker et al. 2009). Fixed effects were investigated using *emmeans* (Lenth 2025), while all visualizations were made using *ggplot2* (Wickham 2016), and *gridExtra* (Auguie 2017).

Figure 2 Difference in fishing effort per MPA or control area polygon in days (top) and kWh (bottom), summed over the entire time series (2012–2021). Positive values (purple) represent cases where the direct method recorded higher effort, while negative values (orange & almost imperceptible) represent greater effort from the c-square method.

Results

Method comparison for spatial aggregation of fisheries data to marine protected areas

Directly associating fishing activity with MPA and control polygons (direct method) yielded greater total estimates of fishing effort than the grid-based approach (Fig. 2). Considering the combined gear annual effort (days) per country, there were 10 341 reported estimates of fishing effort retrieved using both methods. The direct method gave larger effort estimates for 62% of the polygons, lower estimates for 37% of the polygons and roughly equal estimates in less than 1% of the polygons. Furthermore, when reports differed between methods, the direct method produced greater magnitudes of difference (means: 495 vs 5 days and medians: 4.3 vs 0.8 days, for the direct and grid-based methods, respectively). We selected the direct method for further analyses, because it appears to deliver a more comprehensive dataset, and is less likely to falsely attribute nearby fishing to the MPA/control polygons via proportional representation in the overlay of c-squares.

Fisheries characterization in marine protected areas

A total of 60% ($n = 458$) of the MPAs in our dataset, for which we had relevant fishing effort data ($n = 763$), were fished (Table 2). Over half of the fished MPAs (51%, $n = 233$) experienced, on average, greater than 5 days of cumulative fishing effort per year. MBCG fishing occurred in 44% ($n = 338$) of all MPAs, of which 51% ($n = 173$) were subject to greater than 5 days of fishing effort per year. PPG fishing occurred in 55% ($n = 421$) of all MPAs, from which 36%

Table 2 Fished and unfished numbers (percentages) of MPAs after their designation, split by eco-region and in total across all four ecoregions.

Region	Total MPAs	All gears		MBCG			PPG		
		Fished	Unfished	Fished	Unfished	Unknown	Fished	Unfished	Unknown
Baltic Sea	385	177 (46%)	208 (54%)	85 (22%)	298 (77%)	2 (1%)	162 (42%)	221 (57%)	2 (1%)
Greater North Sea	191	167 (87%)	24 (13%)	153 (80%)	36 (19%)	2 (1%)	148 (77%)	40 (21%)	3 (2%)
Celtic Seas	106	42 (40%)	64 (60%)	35 (33%)	71 (67%)	0 (0%)	39 (37%)	66 (62%)	1 (1%)
Bay of Biscay and the Iberian Coast	81	72 (89%)	9 (11%)	65 (80%)	10 (12%)	6 (7%)	72 (89%)	9 (11%)	0 (0%)
Totals	763	458 (60%)	305 (40%)	338 (44%)	415 (54%)	10 (1%)	421 (55%)	336 (44%)	6 (1%)

'All gears' represents fishing by any gear type, while 'MBCG' are MPAs fished by MBCG, and 'PPG' are MPAs fished with PPG, specifically ('Unknown' represents fishing activity for which metiers were not properly reported). Percentages are calculated by row, within borders, and relative to Total MPAs.

($n = 150$) were subject to greater than 5 days of fishing effort per year.

The *Bay of Biscay and Iberian Coast* and the *Greater North Sea* eco-regions had the highest rates of fished MPAs (89% and 87%, respectively), as well as the highest rates of MBCG (both 80%) and PPG (89% and 77%, respectively) fishing in MPAs. The *Celtic Seas* had the lowest rates of fished MPAs (40%), and the lowest rates of PPG fished MPAs (37%), but when considering MBCG fishing, the *Baltic Sea* had the lowest rates (22%).

A total of 71% ($n = 212$) of MPAs that had fisheries restrictions imposed ($n = 300$) remained fished (Table 3). More than half (54%, $n = 114$) of these fished MPAs were fished for more than 5 fishing days per year (on average, post-restrictions). More than half (54%, $n = 163$) of MPAs with fisheries restrictions in place were fished with MBCG, with 40% ($n = 84$) of MPAs with fisheries restrictions experiencing more than 5 days of MBCG effort per year (on average, post-restrictions).

The rates of fishing in the *Celtic Seas*' MPAs (91%, $n = 10$) were higher post-restrictions compared to post-designation. In contrast, the rates of fished MPAs post-restrictions for the *Bay of Biscay and Iberian Coast,* and the *Greater North Sea* remained similar to the rates post-designation (89%, $n = 51$% and 86%, $n = 81$, respectively). The *Baltic Sea* had the lowest rates of fished MPAs after fisheries restrictions, for all gears combined (51%, $n = 70$) and for sites fished with MBCG (28%, $n = 38$).

All MPAs that had conservation objectives including components of the benthic habitat and fisheries restrictions imposed were fished by some gear (100%, $n = 133$; Table 4). However, only 49% of these MPAs ($n = 65$) were fished more than 5 days per year, on average, post-restrictions. In the Baltic Sea, this was as low as 13% ($n = 7$). When considering only MBCG gears, we found that 80% ($n = 107$) of benthic conservation MPAs remained fished post-fishing restrictions, but that only 38% ($n = 51$) experienced an average of more than 5 days of effort annually. Again, the Baltic Sea had the highest rates of MPAs unfished by MBCG in this category (37%, $n = 20$), while the other regions continued to have high rates of MBCG fishing (Table 4). However, in the Greater North Sea and the Celtic Seas, only approximately half of these benthic conservation MPAs were fished with MBCG gears for more than 5 days per year, on average (47%, $n = 27$ and 50%, $n = 3$, respectively). Rates of PPG fishing in these benthic conservation MPAs remained high across all regions, post-fishing restrictions, where they remained at 100% ($n = 43$) in the Bay of Biscay and the Iberian Coast, and above 80% for all other regions.

When standardizing these efforts by MPA area (kW.days.km^{-2}) and taking annual means, post-designation fishing effort varied greatly by site, even within ecoregions (Fig. 3, top-left) but differed significantly between ecoregions ($\chi^2 = 236.15$, df = 3, $P < 0.001$). Effort per unit area was significantly higher in the *Greater North Sea* than in the *Bay of Biscay and the Iberian Coast* ($z = 3.04$, $p_{\text{corrected}} = 0.002$), while both were higher than in either the *Baltic Sea* or *Celtic Seas* ($z > 7.30$, $p_{\text{corrected}} < 0.001$, for all pairwise comparisons). Regional differences persisted when looking only at MBCG (Fig. 3, middle left; $\chi^2 = 276.33$, df = 3, $P < 0.001$) or PPG (Fig. 3, bottom left; $\chi^2 = 166.87$, df = 3, $P < 0.001$) fishing effort, independently. Comparing ecoregions pairwise, the *Baltic Sea* had significantly lower MBCG effort across its MPAs relative to all other ecoregions ($z > 2.94$, $p_{\text{corrected}} < 0.003$, for all pairwise comparisons), while the *Celtic Seas* MPAs experienced lower MBCG effort than the *Greater North Sea* and *Bay of Biscay and the Iberian*

Table 3 Fished and unfished numbers (percentages) of MPAs after fisheries restrictions were implemented, split by ecoregion and in total across all four ecoregions.

Region	Total MPAs	All gears		MBCG			PPG		
		Fished	Unfished	Fished	Unfished	Unknown	Fished	Unfished	Unknown
Baltic Sea	138	70 (51%)	68 (49%)	38 (28%)	98 (71%)	2 (1%)	60 (43%)	78 (57%)	0 (0%)
Greater North Sea	94	81 (86%)	13 (14%)	74 (79%)	18 (19%)	2 (2%)	70 (74%)	21 (22%)	3 (3%)
Celtic Seas	11	10 (91%)	1 (9%)	9 (82%)	2 (18%)	0 (0%)	8 (73%)	2 (18%)	1 (9%)
Bay of Biscay and the Iberian Coast	57	51 (89%)	6 (11%)	42 (74%)	8 (14%)	7 (12%)	51 (89%)	6 (11%)	0 (0%)
Totals	300	212 (71%)	88 (29%)	163 (54%)	126 (42%)	11 (4%)	189 (63%)	107 (36%)	4 (1%)

'All gears' represents fishing by any gear type, while 'MBCG' are MPAs fished by MBCG, and 'PPG' are MPAs fished with PPG, specifically ('Unknown' represents fishing activity for which metiers were not properly reported). Percentages are calculated by row, within borders, and relative to the total number of MPAs.

Table 4 Fished and unfished numbers (percentages) of MPAs with benthic habitats as conservation objectives after fisheries restrictions were implemented, split by ecoregion and in total across all four ecoregions.

Region	Total MPAs	All gears		MBCG			PPG		
		Fished	Unfished	Fished	Unfished	Unknown	Fished	Unfished	Unknown
Baltic Sea	54	54 (100%)	0 (0%)	32 (59%)	20 (37%)	2 (4%)	43 (80%)	11 (20%)	0 (0%)
Greater North Sea	57	57 (100%)	0 (0%)	54 (95%)	3 (5%)	0 (0%)	48 (84%)	8 (14%)	1 (2%)
Celtic Seas	6	6 (100%)	0 (0%)	6 (100%)	0 (0%)	0 (0%)	5 (83%)	1 (17%)	0 (0%)
Bay of Biscay and the Iberian Coast	16	16 (100%)	0 (0%)	15 (94%)	0 (0%)	1 (6%)	16 (100%)	0 (0%)	0 (0%)
Totals	133	133 (100%)	0 (0%)	107 (80%)	23 (17%)	3 (2%)	112 (84%)	20 (15%)	1 (1%)

'All gears' represents fishing by any gear type, while 'MBCG' are MPAs fished by MBCG, and 'PPG' are MPAs fished with PPG, specifically ('Unknown' represents fishing activity for which metiers were not properly reported). Percentages are calculated by row, within borders.

Coast ($z = 7.24$, $p_{\text{corrected}} < 0.001$, and $\underline{z} = 6.96$, $p_{\text{corrected}} < 0.001$, respectively). The *Bay of Biscay and the Iberian Coast* had greater PPG fishing effort in MPAs than all other ecoregions ($z > 5.83$, $p_{\text{corrected}} < 0.001$, for all pairwise comparisons). The *Baltic Sea* and the *Celtic Seas* both had significantly lower PPG fishing effort in MPAs compared to the *Greater North Sea* and *Bay of Biscay and the Iberian Coast* ($z > 4.75$, $p_{\text{corrected}} < 0.001$, for all pairwise comparisons), while not significantly different from one another ($z = 0.63$, $p_{\text{corrected}} = 0.264$).

Regional differences in MPA fishing effort (kW.days.km^{-2}) were also found subsequent to the implementation of MPA-specifIc fishing restrictions (Fig. 3, top right; $\chi^2 = 103.04$, df $= 3$, $P < 0.001$). In pairwise tests, the *Baltic Sea* combined effort was significantly lower than those of all other ecoregions ($z > 3.67$, $p_{\text{corrected}} < 0.001$, for all pairwise comparisons), while the *Greater North Sea* was lower than the *Bay of Biscay and the Iberian Coast* ($z = 3.54$, $p_{\text{corrected}} < 0.001$). PPG fishing effort, post-fisheries restrictions, followed the same pattern as for total fishing (Fig. 3, bottom right; $\chi^2 = 81.58$, df $= 3$, $P < 0.001$), where the *Baltic Sea* combined effort was significantly lower than all other ecoregions ($z > 2.42$, $p_{\text{corrected}} < 0.023$, for all pairwise comparisons), while the *Greater North Sea* was lower than the *Bay of Biscay and the Iberian Coast* ($z = 5.56$, $p_{\text{corrected}} < 0.001$). The Baltic Sea, alone, had significantly lower MBCG effort in its MPAs (Fig. 3D; $\chi^2 = 98.87$, df $= 3$, $P < 0.001$; $z > 4.27$, $p_{\text{corrected}} < 0.001$ for all comparisons).

Mean annual, area-standardized effort (kW.days.km^{-2}), post-MPA designation, was found to be correlated with MPA size (Supplementary Fig. 2). While significant, the strength of these correlations were only moderate for total fishing effort ($\tau = 0.487$, $P < 0.001$), MBCG effort ($\tau = 0.502$, $P < 0.001$), and PPG effort ($\tau = 0.405$, $P < 0.001$).

Impact of marine protected area designation

All gear fishing effort

Attempts to attribute change in fishing activity over time (slope) to the designation of MPAs failed (TS-BACI), as we observed deviating trends in the before-period between control and impact (MPA) sites ($\beta_6 = -0.059$, 95% CI $= -0.010 < \beta_6 < -0.013$) (Fig. 4). In MPAs sites, effort declined by 4.7%, whereas control sites showed no trend, in the before period. Therefore, the simpler BACI model was employed to investigate changes in relative means only. The BACI interaction term (log response ratio) indicated that average annual effort at the MPA sites decreased 19.4% less than it did in the control sites from before to after designation, but the effect of this relative difference was found to be non-significant (Fig. 5, top-left; $\beta_4 = 0.099$, 95% CI $= 0.059 < \beta_4 < 0.26$). In summary, we accept the null hypothesis for hypothesis one; namely, that annual all-gear fishing effort in MPA areas remains unchanged, relative to control areas, following MPA designation.

Mobile bottom-contacting gear fishing effort

The TS-BACI model of MBCG effort met the assumption of parallel trends between MPA and control sites in the before period, where the trend at the MPA was not significantly different to that of the control site ($\beta_6 = -0.056$; 95% CI $= -0.105 < \beta_6 < 0.003$). However, there was no significant difference in relative change in the trend of MBCG effort from before to after ($\beta_7 = 0.021$; 95% CI $=$

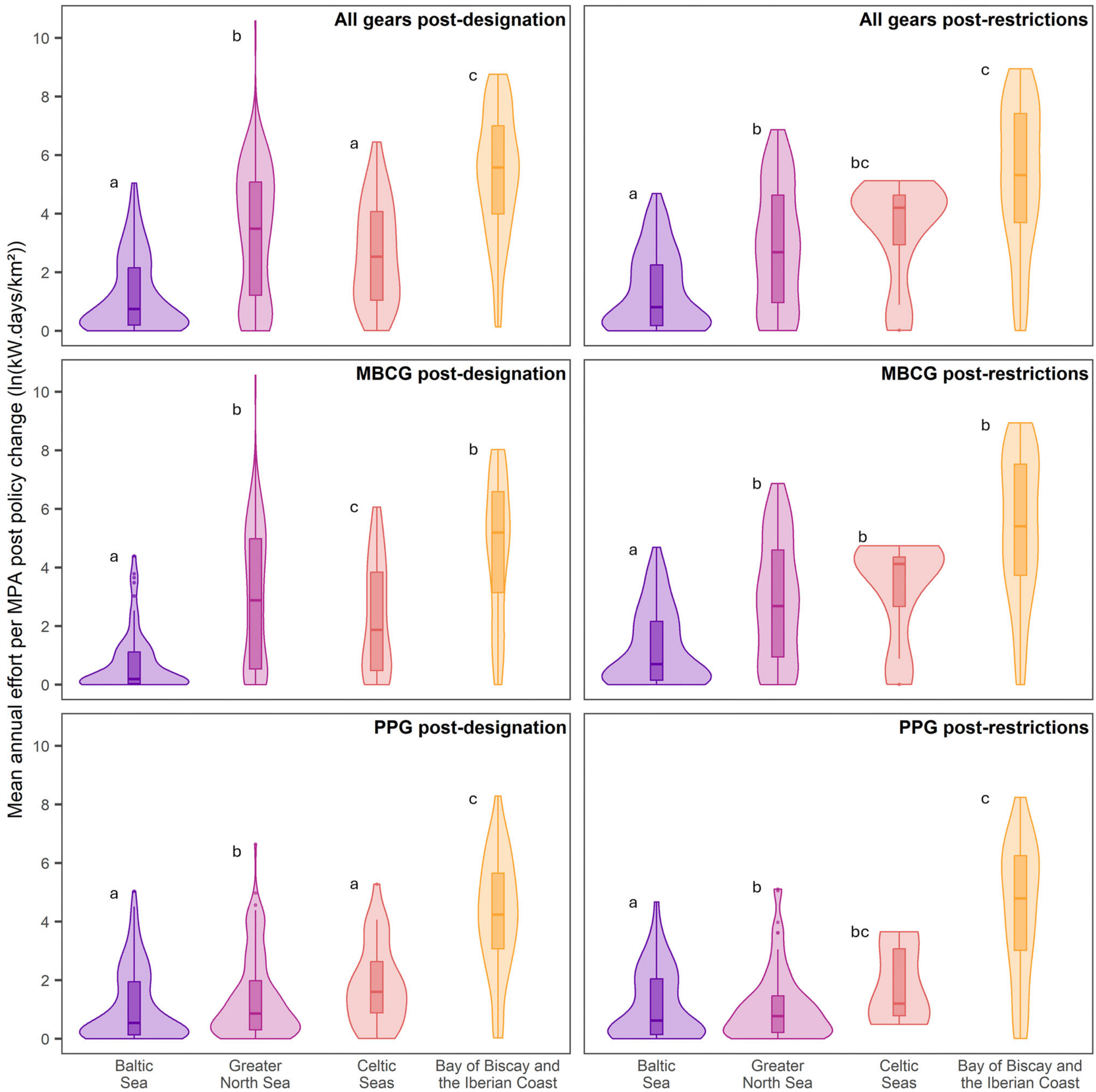

Figure 3 Distributions of mean annual effort (log(kW.days.km^{-2})) of all fishing gears (top), MBCG (middle), and PPG (bottom), across regions in only those periods after MPA designation (left) or post-implementation of fisheries restrictions (right). Note, effort values are log-transformed non-zero values for visualizations. Violins represent the probability density of observations, while the components of the box-plots inside represent: mid-line = median, box extent = upper and lower quartiles, whiskers = outermost observations within 1.5 times the inter-quartile range, from the respective quartiles, and points are observations outside the of said range. Letter annotations represent significant ($P < 0.001$) differences between groups, within each facet, for the full dataset (including zeros, not log-transformed), as described in the main text.

$-0.067 < \beta_7 < 1.02$), nor in the overall relative change in marginal mean annual MBCG effort (classic BACI contrast or log response ratio) between MPAs and controls (Fig. 5, middle-left; $\beta_4 = -42.35$; 95% CI $= -204.3 < \beta_4 < 136$). This can be summarized as having a valid TS-BACI model of MBCG effort, yet detecting no difference in the temporal trends nor the average change in MBCG effort in MPAs, relative to their adjacent control areas. Accordingly, we accept the null hypothesis for hypothesis two: MPA designation does not change the annual MBCG fishing effort in MPA areas, relative to control areas.

PPG fishing effort

Similarly, when investigating only PPG effort, our TS-BACI met the assumption of parallel trends, but did not find relative differences in either the development of temporal trends or the averages between MPAs and their adjacent control areas from before to after designation. The trends in the MPA areas were not significantly different from their control counterparts before designation ($\beta_6 = -0.059$; 95% CI $= -0.112 < \beta_6 < 0.005$), and the relative changes in trends (after vs before) of PPG effort in the MPAs relative to the

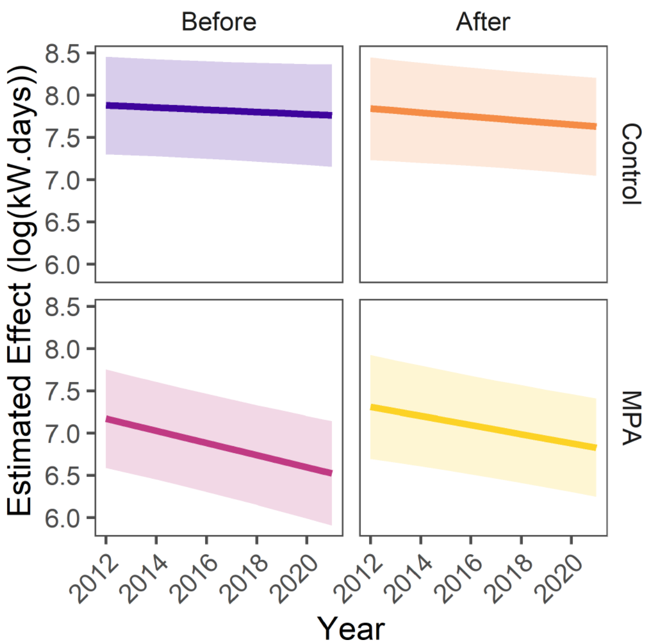

Figure 4 Estimated effects of the TS-BACI interaction term (Before—After and Control—MPA over years) on the linear predictor of the conditional model [log of fishing effort (kW.days)], over the study period (for the case of all gear effort and MPA designation). An illustrative example of the deviation from the assumption of parallel trends between control and impact treatments in the before period.

control sites was also not significantly different ($\beta_7 = 0.037$; 95% CI $= -0.041 < \beta_7 < 0.116$). The log response ratios comparing the change in mean annual PPG effort in MPAs and control from before and after designation were also not significantly different from zero (Fig. 5, bottom left; $\beta_4 = -74.62$; 95% CI $= -233 < \beta_4 < 82.3$). Hence, we again accept the null hypothesis for hypothesis three: MPA designation does not change annual PPG fishing effort in MPA areas, relative to control areas.

Impact of fisheries restrictions

All gear fishing effort

In considering the implementation of fisheries restrictions as the primary intervention, and focusing on all-gear combined effort, we again observed a violation of the parallel trends assumption in the before period (pre-restriction). Specifically, effort trends at MPA sites were, on average, 14% more positive than those at adjacent control sites ($\beta_6 = 0.131$, 95% CI $= 0.08 < \beta_6 < 0.176$). The simplified BACI model demonstrated a 19.4% decrease in effort at MPA sites relative to the mean change in effort at the control sites (Fig. 5, top-centre; $\beta_4 = -0.216$, 95% CI $= -0.378 < \beta_4 < -0.063$). This relative change corresponded to a 13.3% decrease in average annual fishing effort at control sites, and a 30.1% decrease in MPA sites, after fisheries restrictions, relative to before restrictions. Thus, we reject our null hypothesis and accept hypothesis four: Fisheries restrictions change annual all-gear fishing effort in MPA areas, relative to control areas.

Mobile bottom-contacting gear fishing effort

We detected non-parallel trends between changes in MBCG fishing at control and MPA sites, before fisheries regulations were introduced ($\beta_6 = 0.112$, 95% CI $= 0.069 < \beta_6 < 0.156$). The simplified BACI model demonstrated an 18.3% decrease in effort at MPA sites relative to the mean change in effort at the control sites (Fig. 5, middle-centre; $\beta_4 = -0.202$, 95% CI $= -0.395 < \beta_4 <$ -0.063). This relative change corresponded to a 15.3% decrease in average annual MBCG fishing effort at control sites, and a 30.8% decrease in average annual MBCG effort at MPA sites, after fisheries restrictions relative to before restrictions. Thus, we reject the null hypothesis for hypothesis five, and accept that fisheries restrictions change annual MBCG fishing effort in MPA areas, relative to control areas.

PPG fishing effort

As for the MBCG gears, in relation to fisheries restrictions, the TS-BACI for PPG only fishing identified non-parallel trends across control and MPA sites in the before period ($\beta_6 = 0.133$, 95% CI $= 0.085 < \beta_6 < 0.188$). Therefore, we compared the annual average relative differences in PPG fishing effort before and after restrictions using the simplified BACI, and found that the MPA sites had a significantly greater decline in PPG fishing effort (26.4%) relative to the decline in the adjacent control areas (9.1%; Fig. 5, bottom-centre; $\beta_4 = -0.307$, 95% CI $= -0.493 < \beta_4 <$ -0.137). Therefore, we reject our null hypothesis for hypothesis 6 and accept that fisheries restrictions change annual PPG fishing effort in MPA areas, relative to control areas.

Impact of fisheries restrictions in MPAs with benthic habitat conservation objectives

All-gear fishing effort

When focusing our comparison to only those MPAs where the stated conservation objectives included elements of the benthic habitat and which had fisheries restrictions imposed, we found that the parallel trend assumption for the TS-BACI did not hold ($\beta_6 = 0.113$, 95% CI $= 0.062 < \beta_6 < 0.164$). Moving on to the simplified BACI, we found no significant difference in the relative change in all-gear fishing effort after restriction implementation between MPAs and their control areas (Fig. 5, top-right; $\beta_4 = -0.029$, 95% CI = -0.206 $< \beta_4 < 0.143$). Therefore, we accept our null hypothesis for hypothesis 7, that Fisheries restrictions in MPAs with benthic habitat objectives do not change annual all-gear fishing effort in MPA areas, relative to control areas.

Mobile bottom-contacting gear fishing effort

Again, our dataset breaks our assumption of parallel trends for the TS-BACI ($\beta_6 = 0.114$, 95% CI $= 0.047 < \beta_6 < 0.175$), and we reverted to the non-time-series BACI. In MPAs with benthic habitat conservation objectives, the BACI model demonstrated an 18% decrease in MBCG effort at MPA sites relative to control sites, which also declined (9%), corresponding to an overall 25% decline at the MPA site, after the implementation of fisheries restrictions (Fig. 5, middle-right; $\beta_4 = -0.199$, 95% CI $= -0.411 < \beta_4 <$ -0.018). Therefore, we reject the null hypothesis for hypothesis 8, and accept that restrictions in MPAs with benthic habitat objectives change annual MBCG fishing effort in MPA areas, relative to control areas.

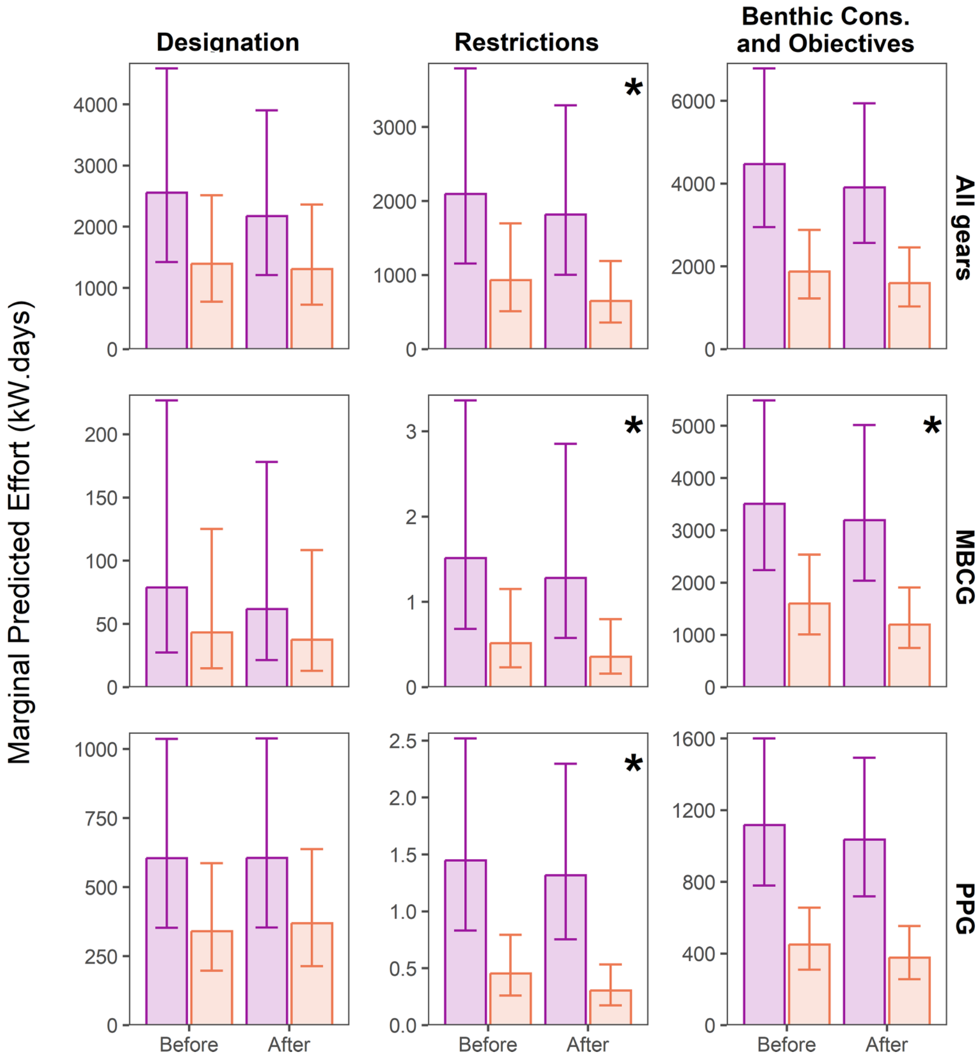

Figure 5 Estimated marginal means from the three retained models addressing the effect of MPA designation (left column), fisheries restrictions (centre column) on annual fishing effort, or the specific effect of fisheries restrictions in MPAs with benthic conservation objectives (right column). Effort is considered from all gears (top), MBCG only (middle), and PPG only (bottom). Error bars represent the 95% confidence intervals of the marginal means, excluding site-level variation. Asterisks indicate significant differences in the relative changes in means between MPAs and their controls (log response ratio). The left column's figures correspond to hypotheses 1–3 and the centre's 4–6, and the right's 7–9 (top down).

PPG fishing effort

In our final hypothesis test, we were again unable to utilise the TS-BACI, due to non-parallel trends in the before period ($\beta_6 = 0.084$, 95% CI $= 0.025 < \beta_6 < 0.143$). When using the simplified BACI model, we find no significant difference in the relative change of PPG fishing effort between MPAs with benthic conservation objectives and their controls, after the implementation of fisheries restrictions (Fig. 5, bottom right; $\beta_6 = -0.104$, 95% CI $= -0.3 < \beta_6 < 0.099$). Therefore, we accept our null hypothesis for hypothesis 9, that restrictions in MPAs with benthic habitat objectives do not change annual PPG fishing effort in MPA areas, relative to control areas.

Before-After-Control-Impact significance vs marginal means

The point predictions of the marginal means (Fig. 5) represent population level estimates, ignoring the site-level variation that is instead incorporated into the confidence intervals around them. Furthermore, these estimated marginal means are the combination of the effects across all fixed-effects values in the models and not the individual fixed effects, the latter of which are what are interpreted in the hypothesis testing above.

Discussion

The EU's northern and western waters have many areas designated as MPAs, however, this designation has no significant impact on the level of fishing activities that occur within these protected areas, overall. Using officially reported fisheries effort data, we have shown that the majority (60%) of these MPAs remain fished after being designated. Furthermore, we show that 44% have MBCG fishing, interacting with the seafloor, while 55% have pelagic and passive gears potentially interacting with other components of the ecosystem. We have also shown that fishing effort, be it with all gears combined, just MBCG, or just PPG, does not change significantly in response to areas being designated as MPAs. While previous studies have attempted to quantify (Dureuil et al. 2018; Perry et al. 2022) or infer (Relano and Pauly 2023) fishing effort in European MPAs, this is, to the best of our knowledge, the first study to either use official effort data, with multi-annual time-series or to apply BACI style analyses at this regional scale. This novelty enables us to approach causal inference when evaluating the effectiveness of current EU MPA implementation as management tools in the context of fisheries.

While MPAs are established with a variety of conservation objectives (Arneth et al. 2023) with diverse legislative and policy mechanisms to identify and regulate them (Grorud-Colvert et al. 2021), fishing activity is regularly considered the largest risk to marine ecosystems (outside of climate change) across different regional contexts (Pedreschi et al. 2019; O'Hara et al. 2024; Bornman et al. 2025). Therefore, it may be surprising that of the EU MPAs considered in this study, fewer than half (42%) were reported to have fisheries restrictions of some kind in place. However, this is not an anomaly nor new knowledge (Aminian-Biquet et al. 2024; Feary et al. 2025), and may be due to management decisions that do not consider some fishing activities in conflict with specific MPA objectives. When we reduced our scope and considered only those MPAs that have had fisheries restrictions imposed, we still found that more than two-thirds (71%) are subject to some fishing activity, more than half (54%) of restricted sites were fished with MBCGs, and nearly two-thirds (63%) with PPG. These findings do not necessarily indicate illegal fishing, as the applied restrictions are often targeted and specific to some gears, times or other technical measures; furthermore, given the complexity of the types of restrictions across jurisdictions, we do not attempt to determine the legality of the reported fishing activities.

However, in contrast to the effect of designating MPAs, we found that fisheries restrictions significantly reduced fishing activities in MPAs where they were implemented, whether we examined all gears combined, or MBCG and PPG fishing independently. Furthermore, when considering MPAs with conservation objectives including benthic habitat components, we found that only MBCG fishing was significantly impacted by fisheries restrictions. This indicates that targeted restrictions for specific objectives (e.g. on MBCG gears for benthic habitats) will not necessarily impact all fishing activities nor overall fishing effort. When considering these effects at the regional scale, we must acknowledge the large site-dependent variation, which indicates a context dependence of this overall significant effect. Therefore, while our BACI models characterize the response of European fisheries to MPA policy, broadly, management actions should be locally relevant (Grorud-Colvert et al. 2021).

Comparison of findings

The rates of fishing activities in MPAs and in fisheries restricted MPAs that we document are equal to or higher than estimates derived from AIS data and models of vessel behaviour in the literature (60% in our studied MPAs vs 59% of MPAs in Dureuil et al. 2018, and 26% of MPAs in Perry et al. 2022). We found higher rates of MPA fishing than were reported in Feary et al. (2025), because we excluded MPAs in Macaronesia, due to concerns about VMS reliability. Furthermore, some regional rates of MPA fishing have changed compared to Feary et al. (2025), due to our more inclusive method, which uses recent annual averages for MPA sites without specified designation dates. When we consider only MBCG fishing, we found lower rates (44% of MPAs fished) than those of Dureuil et al. (2018) for specifically bottom trawling (59% of MPAs fished). However, our estimated rates of MBCG fishing remain higher than Perry et al. (2022), whether we consider MBCG fishing in designated MPAs (44%), fisheries-restricted MPAs (42%), or fisheries-restricted MPAs with benthic habitats components in their conservation objectives (75%). This is despite the latter case more closely aligning with the Perry et al. (2022) definition of high-risk fishing. This discrepancy is likely due to two factors: a more specific pre-selection of relevant MPAs, and the sources of fishing effort data.

Although primarily drawing from the same initial sources (Natura 2000 and Common Database on Designated Areas), the authors of the MAPAFISH database (Feary et al. 2025), from which we draw our list of MPAs, excluded MPAs that were smaller than five square kilometres or with more than 5% of their surface area on land. Three main justifications were made for this pre-selection. The first being scale mismatch between fishing data aggregated to c-square level (0.05×0.05 degrees $= \sim 20$ km^2 throughout western Europe) and the frequency (and distance travelled) of VMS positional recordings. Secondly, the small MPAs that were excluded are of less ecological relevance, given that Feary et al. (2025) retained 89% of the MPA surface area, as well as the relationship between MPA size and efficacy (Edgar et al. 2014). Thirdly, some MPAs in the original databases consisted primarily of coastal marshlands and other predominantly terrestrial sites, which are not relevant to fishing. Finally, the majority of our analyses utilise no selection criteria based on conservation targets set by the MPAs, however, our investigations of MPAs with conservation objectives encompassing benthic habitat components are more closely aligned with Perry et al. (2022), who couple MPAs designated to protect habitats with 'high-risk' fishing.

Furthermore, Perry et al. (2022) utilized data modelled from AIS for vessel location data, which, while a valid data source, also has limitations. Automatic Identification System is only compulsory on vessels greater than 15 metres in length (European Council 2009), while vessel position reporting for fisheries (primarily VMS) is required for vessels above 12 m. While it is often utilized by smaller commercial vessels for safety reasons, the different forms of AIS installed on smaller vessels have caused reliability issues due to connectivity with land-based receivers and satellites (Troupiotis et al. 2022). Furthermore, the data used to inform the analyses by Perry et al. (2022) were model output, which comes with its own inherent uncertainties and biases (Hintzen et al. 2025). So, while the two different sources of fishing effort are complementary, our fisheries dataset may include effort not captured in Perry et al. (2022), further increasing our detection rates of fishing in MPAs.

Regional differences

Regionally, the number of MPAs designated, the number of MPAs with fisheries restrictions, and the level of fishing in these MPAs all varied. The Baltic Sea had relatively low proportions of MPAs with fishing activities, and the MPAs were subject to the lowest concentration of effort. However, the Baltic Sea is generally much less fished than the other ecoregions (Eigaard et al. 2017; ICES

2024;), and it also has a large number of smaller MPAs. This decreases the probability of any given MPA's interactions with fisheries, as indicated by the weak but positive relationship between MPA size and fishing effort per unit area that we found, and is corroborated by the findings of Perry et al. (2022). Similarly, the Celtic Seas had predominantly smaller MPAs, which were also fished significantly less often and with lower average annual effort, post-designation, than the Greater North Sea and the Bay of Biscay and Iberian Coast. This is likely because our MPA dataset did not include MPAs from the United Kingdom, and the contributions from Ireland were predominantly smaller and coastal (but it should be noted that subsequent to our analyses, larger off-shore SPAs have been designated). However, the few sites in the Celtic Seas that had fisheries restrictions imposed continued to experience high rates of fishing with all-gear types, at levels of effort similar to those in other ecoregions. We interpret these patterns of continued high rates of fishing (especially in the Celtic and North Seas) as reflecting two dynamics: first, many MPAs were designated in areas that already had low fishing activity, resulting in low post-designation fishing; second, where MPAs overlapped with fishing grounds, activities were regulated rather than excluded, leading to persistently high fishing effort within these regulated MPAs.

Utility

Knowledge of how current policies and their management implementations affect real-world human activities and impacts is integral to the assessment of these decisions and their development together with new legislation. This reflective approach provides core empirical knowledge that is a valuable complement to scenario-based forecasts, such as in Püts et al. (2023) and Bastardie et al. (2025), which allow decision makers to explore decisions in uncertain futures. Our study confirms that MPA designation alone, does not change fishing activities, as is assumed by the analyses of Aminian-Biquet et al. (2024), who described the lack of regulatory measures within EU MPAs. We also show that fisheries regulations generally reduce fishing activity within Atlantic and Baltic EU MPAs, and that restrictions can be targeted to support conservation objectives by selectively constraining particular gear groups, without affecting the wider fishery. While significant at the scale we have analysed, these reductions in fishing effort also come with large site-specific variation, indicating that in many contexts these regulations do not affect substantial changes in fishing effort, even within the very broad gear types we have analysed (MBCG and PPG). Cases where regulations are ineffectual may be because they are imposed on areas of historically low fishing effort (i.e. for political pragmatism Devillers et al. 2015, 2020). Another possible reason for fisheries regulations being an inconsistent predictor for change in fishing effort may be a lack of surveillance of fishing activities, or enforcement of the restrictions imposed on them (Guidetti et al. 2008; Edgar et al. 2014; Gill et al. 2017). Alternatively, the regulations imposed may be so targeted that they do not alter the effort expended in an MPA, but merely change technical approaches employed in that effort. For these same reasons, studies that infer fishing activity as the merely the opposite or inverse of the implemented restrictions risk overstating potential conflicts between fishing and conservation objectives (Roessger et al. 2022; Aminian-Biquet et al. 2024;).

Limitations

In most of our hypothesis tests, control sites lacked matching effort trends with MPA sites before MPA designation or the imposition of regulations, limiting these tests to comparisons in mean annual effort before and after. This, coupled with the limited (10 year) period over which fisheries data were available, means that some before and after periods were small, with large errors around annual means, making the detection of change more difficult. As the length of time that reliable spatial fisheries data are collected and available increases, and more MPAs are designated or have fisheries regulations enacted, the strength of such analyses will also increase. Therefore, this methodological approach should be considered in future analyses to track changes in the fishery-MPA management interaction on regional and EU scales.

Our use of the gear category 'PPG', was intended as a complement to the more commonly applied 'MBCG'. However, this group includes very different fishing activities such as the use of mobile pelagic gears and passive benthic gears. Furthermore, passive gears may also interact with the seafloor, and the calculation of passive gear effort is limited to a proxy of 'time in area', using the standard ICES spatial fisheries data call methodology.

Our characterizations and our hypothesis tests consider predominantly blunt changes in management, namely the designation of MPAs and the implementation of any fisheries restrictions. While this decision allows us to focus on addressing common misconceptions about the interplay between EU MPAs and fisheries, it prevents deeper analyses of the interactions between conservation objectives, restrictions of human activities (including the diversity of fishing activities), and conservation outcomes. Our findings on changes to MBCG fishing, but not other fishing, in MPAs with benthic habitat conservation objectives are one example of linking MPA objectives to fishing outcomes; however, even within this relationship, we are comparing broad categories of ecosystem components with a broad array of gear types. While such generalizations are acceptable at the regional scale of our hypotheses, more detailed linkages between objectives, restrictions, fishing activities, and conservation outcomes are required to consider impacts at the level of individual MPA or MPA networks.

Future developments

Future work on the interactions between fisheries and MPAs should utilise officially reported fisheries data and endeavour to make explicit extractions of these data directly to MPA polygons, rather than using standardized gridded fisheries data (i.e. c-square data). As European Coastal States along the Atlantic coast, the United Kingdom, and Norway have already produced data to this level, it should continue to be replicable as MPA areas change. To facilitate this, we provide the data call and scripts necessary to combine logbook and VMS data for custom polygons (Supplementary Fig. 2). This approach should be extended to investigate the comparability and complementarity of combining different sources, such as the Global Fishing Watch AIS-based database, other positioning systems, and various forms of catch reporting (e.g. recreational fishing logs and surveys; Støttrup et al. 2018; Skov et al. 2021). Furthermore, to improve upon the BACI style analyses, different methods for establishing per MPA control areas should be investigated. Inspiration may come from observational studies of human health outcomes, where matching is based on subject characteristics (i.e. counterfactual construction and propensity score matching; Steventon et al. 2015; Banack et al. 2024).

As described in the limitations of this study, comparisons of MPA objectives, with the MPA's regulations on human activities (Aminian-Biquet et al. 2024) should be extended to investigate the actual activities occurring in MPAs (i.e. specific fishing types or gears), as opposed to just those that are permitted. The ideas behind the comparisons of this study (MPAs with benthic habitat objectives and MBCG), of 'high risk,' fishing activities within MPAs established for habitat protection (Perry et al. 2022), and the classification of trawling as high-risk to benthic habitats and sensitive by-catch species (Dureuil et al. 2018) should be extended to consider fishing impacts from the diversity of fishing activities on all components of the ecosystem, and their relevance to MPA objectives.

Conclusion

Using official reported fisheries data and BACI analyses, we demonstrate that the designation of MPAs in Europe has no significant effect on the fishing effort taking place within them. Observable changes in fishing effort within MPAs occurred only after the implementation of fisheries regulations in these MPAs. Following the introduction of fisheries regulations, generally, within MPAs, we detected a significant decrease in all types of fishing activities. Following the implementation of fisheries restrictions in MPAs with specific benthic conservation objectives, we detected a significant decrease in only MBCG fishing and not overall fishing. However, these responses varied greatly across sites, indicating a strong context dependence. Thus, we conclude that the designation of MPAs does not change fishing practices without explicit fisheries restrictions in those MPAs. Furthermore, targeted restrictions applied in response to conservation objectives can have an effect on fishing effort of specific gears in MPAs. These results provide more direct evidence of the relationships between MPAs and fisheries, than are currently available, as well as a direct empirical test of these relationships, providing a foundation of support for more forward-looking, scenario-based explorations. Individual MPA objectives should determine the types of activities that are allowable within any given protected area. A more resolved comparison of the types of fishing and the specific MPA objectives is a pre-requisite for evaluating the effectiveness of individual MPAs and their regulations.

Acknowledgements

The authors are grateful to the reviewers for their thorough reviews and valuable suggestions, which substantially improved the manuscript. The authors would like to acknowledge help with establishing the data call from Niels Hintzen and Josefine Egekvist.

Author contributions

Elliot J. Brown (Conceptualization [lead], Data curation [equal], Formal Analysis [lead], Funding acquisition [equal], Investigation [lead], Methodology [lead], Project administration [lead], Resources [lead], Supervision [lead], Validation [equal], Visualization [lead], Writing—original draft [lead], Writing—review & editing [lead]), Karin J. van der Reijden (Data curation [lead], Formal Analysis [equal], Investigation [equal], Methodology [supporting], Validation [equal], Writing—original draft [supporting], Writing—review & editing [equal]), Nuno Castro (Investigation [supporting], Validation [supporting], Writing—review & editing [equal]), Stephen C. Mangi (Investigation [supporting], Resources [supporting], Validation [supporting], Writing—review & editing [equal]), José Carlos Mendoza (Data curation [supporting], Investigation [supporting], Validation [supporting], Writing—review & editing [equal]), Oliver Tully (Data curation [supporting], Investigation [supporting], Validation [supporting], Writing—review & editing [equal]), Mattias Sköld (Data curation [supporting], Investigation [supporting], Validation [supporting], Writing—review & editing [equal]), Tamara Vallina (Data curation [supporting], Investigation [supporting], Validation [supporting], Writing—review & editing [equal]), Gert Van Hoey (Conceptualization [supporting], Data curation [equal], Investigation [supporting], Project administration [equal], Resources [supporting], Supervision [supporting], Validation [supporting], Writing—review & editing [equal]), Katrien Verlé (Data curation [supporting], Investigation [supporting], Validation [supporting], Writing—review & editing [equal]), Robert Wakeford (Data curation [supporting], Funding acquisition [equal], Project administration [supporting], Validation [supporting], Writing—review & editing [equal]).

Supplementary material

Supplementary data is available at *ICES Journal of Marine Science* online.

Conflicts of interest

None declared.

Funding

This work builds on the findings of the specific contracts CINEA/EMFAF/2020/3.2.6-LOT1/SC9 & LOT2/SC10, requested by the European Climate, Infrastructure and Environment Executive Agency (CINEA) and funded through the European Maritime, Aquaculture and Fisheries Fund (EMFAF)

Data availability

The data underlying this article cannot be shared publicly as they were provided under condition of confidentiality, with permission only for the publication of processed results. To improve repeatability, the precise methods used to collect these data from coastal states are shared as Supplementary material, in the form of the data call with links to associated scripts and the privacy agreement.

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Handling editor: Katherine Yates

Received: 2 December 2025. **Revised:** 1 April 2026. **Accepted:** 4 April 2026