

MAPPING OF MARINE PROTECTED AREAS AND THEIR ASSOCIATED FISHING ACTIVITIES (MAPAFISH)

Baltic and North Seas, Atlantic EU Western Waters and Outermost Regions

Final Report Annex 5: Case Studies

European Maritime, Aquaculture and Fisheries Fund (EMFAF)



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Contact: CINEA EMFAF CONTRACTS

E-mail: CINEA-EMFAF-CONTRACTS@ec.europa.eu

European Commission
B-1049 Brussels
BELGIUM

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Waters and EU Outermost Regions**

Final Report Annex 5: Case Studies

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ANNEX 5: CASE STUDY REPORTS

In total, nine case studies in the Baltic Sea, North Sea, Atlantic EU Western Waters and some outermost regions (Madeira and Canary Islands) were selected to gather more information on the response of the fishing activities to MPA designation and implementation (Table A1).

The selection of case studies is well-spread in the different sea basins, cover an array of Member States, but also vary in the type of species (e.g. fish, crustacea, bivalves) and level of MPA protection (e.g. no-take, multi-zone, with buffer areas). In the case studies, the fisheries response to MPAs are analysed based on qualitative and/or quantitative analytical approaches, using available data and/or a stakeholder survey.

Table A1. Case studies on the assessment of potential spatial redistribution (displacement) of fishing activities in response to MPA designation and implementation.

Nr	Case Studies	Country	Regional Sea
1	The North Sea Coastal Zone	The Netherlands	North Sea
2	The Flemish Banks	Belgium	North Sea
3	The Nida-Perkone Marine Protected Area	Latvia	Baltic Sea
4	Adler Grund og Rønne Bank	Denmark	Baltic Sea
5	The Bratten Offshore Natura 2000 Area	Sweden	Skagerrak
6	Dundalk Bay	Ireland	Celtic Seas
7	The Madeira Archipelago	Portugal	Macaronesia
8	The Professor Luiz Saldanha Marine Park	Portugal	Iberian Coast
9	La Palma Island and La Graciosa Island	Spain	Macaronesia

Case Study Report

The North Sea Coastal Zone The Netherlands – North Sea



A shrimp trawler in (ANP) front of the port of Lauwersoog. Photo by Koen Suyk

Mapping of marine protected areas and their associated fishing activities

Tamara Vallina

***Wageningen Marine Research, Wageningen University and
Research
The Netherlands***

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LIST OF ABBREVIATIONS

Term	Description
BACI	Before-After Control-Impact
BPI	Bathymetry Position Index
CPUE	Catch Per Unit of Effort
EU	European Union
EMODnet	The European Marine Observation and Data Network
GLMM	Generalised Linear Mixed Model
ICES	International Council for the Exploration of the Sea
ICES WGSFD	ICES Working Group on Spatial Fisheries data
MPA	Marine Protected Area
NL	Netherlands
NSCZ	North Sea Coastal Zone
RVO	The Netherlands Enterprise Agency
TAC	Total Allowable Catch
TBS	Shrimp Beam Trawlers
VIBEG	Visserij in beschermde gebieden akkoord (NL) Fisheries In Protected Areas Agreement (EN)
VMS	Vessel Monitoring System
WMR	Wageningen Marine Research

1. EXECUTIVE SUMMARY

The brown shrimp (*Crangon crangon*) fishery represents one of the largest in the coastal zones of the Netherlands. The fishery primarily utilises bottom otter- and beam-trawls and focuses predominantly in the North Sea Coastal Zone (NSCZ). Covering approximately 1,445 km², this area encompasses shallow waters comprising sandbanks, mudflats, salt meadows and shifting dunes, designated for various habitats and diverse animal species under Natura 2000. The Fisheries In Protected Areas Agreement (*Visserij in beschermde gebieden akkoord*: VIBEG) was established in 2011 to balance nature conservation and fisheries within Natura 2000 sites. As a result of this agreement, five separate areas totalling approximately 144 km² within the NSCZ were granted a fully protected status within the marine protected area (MPA), making them inaccessible for shrimp fisheries.

A primary concern when implementing fishing restrictions in specific areas involves the potential relocation of fishing activity to other sites. Hence, this study's principal aim was to evaluate the potential spatial redistribution of fishing activities following the implementation of the fully protected areas in the NSCZ. The research addressed the following key questions: (i) how does the overall fishing effort change for affected fishermen compared to other fishermen subsequent to the implementation of the fully protected areas?; (ii) which factors influence the relocation of fishermen within the NSCZ? By using statistical models, a before-after, control-impact (BACI) analysis was employed to examine both of these questions.

Although the fully protected areas were formally closed to shrimp fishers in 2013, restrictions were only adhered to from 2017 onwards. From 2017, there was a notable reduction in fishing effort in the shrimp-fishery sector, not only from fishers that had previously fished in the protected areas (the hindered group), but also from other shrimp fishers (the unhindered control group). Hence, reduction in fishing effort cannot be attributed solely to closure of the fully protected areas in the NSCZ. Despite this, the decline in fishing activity was most pronounced within the fully protected areas. Virtually no fishing activity persisted in the fully protected areas, signifying that the restrictions had the intended impact on fishing efforts. Based on these findings, we can rebut the concern that implementing fishing restrictions in specific locations resulted in a direct displacement of fishing activity to alternative areas in a one-to-one manner.

After 2017, a minor shift in visited habitats occurred, showing that restrictions did not result in fishers substantially extending travel distances. However, fishers slightly increased the use of areas of high gravel percentages. Visual representations indicated that reallocation predominantly occurred around the central fully protected areas, with reduced effort observed further around the outer peripheries of the Wadden islands.

This study illustrates that the establishment of fully protected areas led to a significant reduction in fishing effort within those zones, without direct displacement of fishing activities to other areas, and an overall decrease in fishing effort in the entire area studied. These findings argue against the concerns typically associated with the implementation of fully protected areas. However, further investigation is needed to ascertain the specific reasons behind the reduction in fishing activity and whether limitations in space were the leading factor leading to the reduction of fishing effort. Understanding these factors is crucial for determining whether future establishments of fully protected areas lead to similar outcomes.

2. BACKGROUND

2.1. Dutch brown shrimp fishery

The Dutch brown shrimp (*Crangon crangon*) fishery is one of the largest fisheries in the coastal zones of the Netherlands, with recent estimates (2014) showing that 199 Dutch-flagged vessels caught brown shrimp. In total, these cutters landed 19 815 tonnes of brown shrimp with a turnover of EUR 65.7 million. The auctions around the Wadden Sea in the north of the Netherlands (i.e. Harlingen, Den Oever, Zoutkamp, Lauwersoog) accounted for 82 % of the total landings of brown shrimp within the Dutch fishery (Turenhout et al., 2015). In addition to the Dutch shrimp fleet, the German and Danish shrimp fleets also operate substantial fisheries in the North Sea and Wadden Sea, while the English and Belgian shrimp fleets undertake limited brown shrimp fisheries in the coastal zones of England and Belgium, respectively. It can be assumed that annually, the Dutch shrimp fleet accounts for 40–50 % of total landings of brown shrimp within Europe (STECF, 2015).

2.1.1. Problems associated with shrimp fisheries

Fishing with otter- and beam-trawls accounts for the major part of shrimp production, as these gears target species that form large schools over the bottom. Trawling physically disturbs the seabed by dragging fishing gear over the seabed to catch bottom-dwelling fish and benthic invertebrates. The direct impacts are the physical disturbance on the habitat and on the organisms living on and in the seabed (Eigaard et al., 2017; Tulp et al., 2020). Apart from the brown shrimp, a shrimp fisher also catches other species. In most cases, this bycatch is of no interest to the fishers. Bycatch and discard of non-target species are particular problems of shrimp trawling, which decreases survival of ecologically and commercially important fish (Bauer, 2020). The shrimp sector is known for its large bycatch of benthos, flatfish (in particular plaice) and undersized shrimp. The bycatch of this fishery is high partly because it uses the smallest mesh size (20 mm) of the entire Dutch fishing industry (Jongbloed et al., 2015; Catchpole et al., 2008).

2.2. Fishery in protected areas

The North Sea Coastal Zone (NSCZ) covers the entire northern coastal strip from North Holland Bergen to the Ems above Rottum. The NSCZ is approximately 1,445 km², with shallow waters, including sandbanks, mudflats, salt meadows and developing shifting dunes (Ramsar, 2022). The area is designated for several habitat types and many animal species under Natura 2000. The Netherlands has implemented MPAs in order to restore and conserve the environmental status and condition of the North Sea, including seafloor communities (European Environment Agency, 2018). In 2009/2010, the NSCZ was designated as a Natura 2000 area. This makes the area part of a European network of nature areas subject to special protection. The potential ecological effects of the brown shrimp fishery in the shallow shelf of the North Sea and the Wadden Sea are currently heavily debated, especially because the fishery operates for a large part in designated Natura 2000 areas. The protection of the NSCZ is expected to be further supported by the recent European Union (EU) marine Action Plan (Ministry of Infrastructure and Water Management, 2022).

To enable a combination of nature conservation and fisheries within the Natura 2000 areas in the North Sea, the Fisheries In Protected Areas Agreement (Visserij in beschermde gebieden akkoord, VIBEG) was concluded in 2011. The aim of VIBEG is for the fishing

sectors to be involved and contribute to the achievement of the conservation targets for the Natura 2000 areas Vlake van de Raan and NSCZ so that a healthy and resilient ecosystem can develop in these areas, with the secondary objectives of maintaining favourable prospects for professional and sport fishing. This would be achieved through closure of areas and by impact reduction of shrimp fishing. The core of the agreements involves spatial zoning of areas for different types of fishery, allowing various types of fishery in some of these zones, sometimes limited to a specific season (Ministerie van Economische Zaken (Ministry of Economic Affairs), 2013). This report will further focus on the areas permanently closed to shrimp fishing. Specific area names and sizes are given in Figure 1 and Table 1.

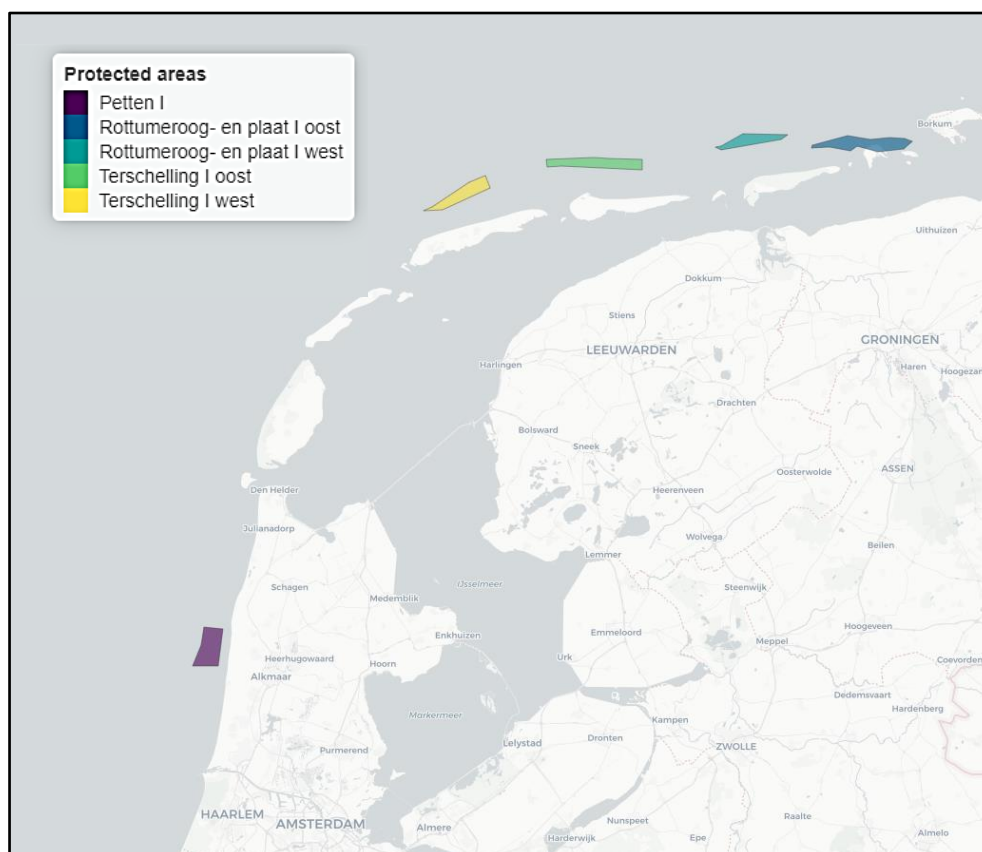


Figure 1. Overview of shrimp-fishery protected areas in the Natura 2000 area, the North Sea Coastal Zone.

Table 1. Overview of sizes of shrimp-fishery protected areas.

Area	Size
Petten I	30.8 km ²
Rottumeroog – en plaat I West	25.4 km ²
Rottumeroog – en plaat I Oost	34.0 km ²
Terschelling I West	18.7 km ²
Terschelling I Oost	34.9 km ²
Total	143.8 km²

2.2.1. Fishery agreement

Officially, shrimp fishing has not been permitted in zone 1 areas since 2013 (Figure 1). Initially, the majority of shrimp fishers disagreed with the closure and did not adhere to the rules. However, when the first fines were handed out for not following legal spatial fishing restrictions in 2016, adherence to the restrictions in the areas appeared instantaneously (Figure 2). This was verified by looking at fishing effort in protected areas over the years and months. In January 2017, there was a sudden drop in fishing effort in the protected areas, corresponding with the time the first fines were imposed. In conclusion, although areas were legally closed in 2013, January 2017 was considered the cut-off point for adherence to the closure of the fishing areas.

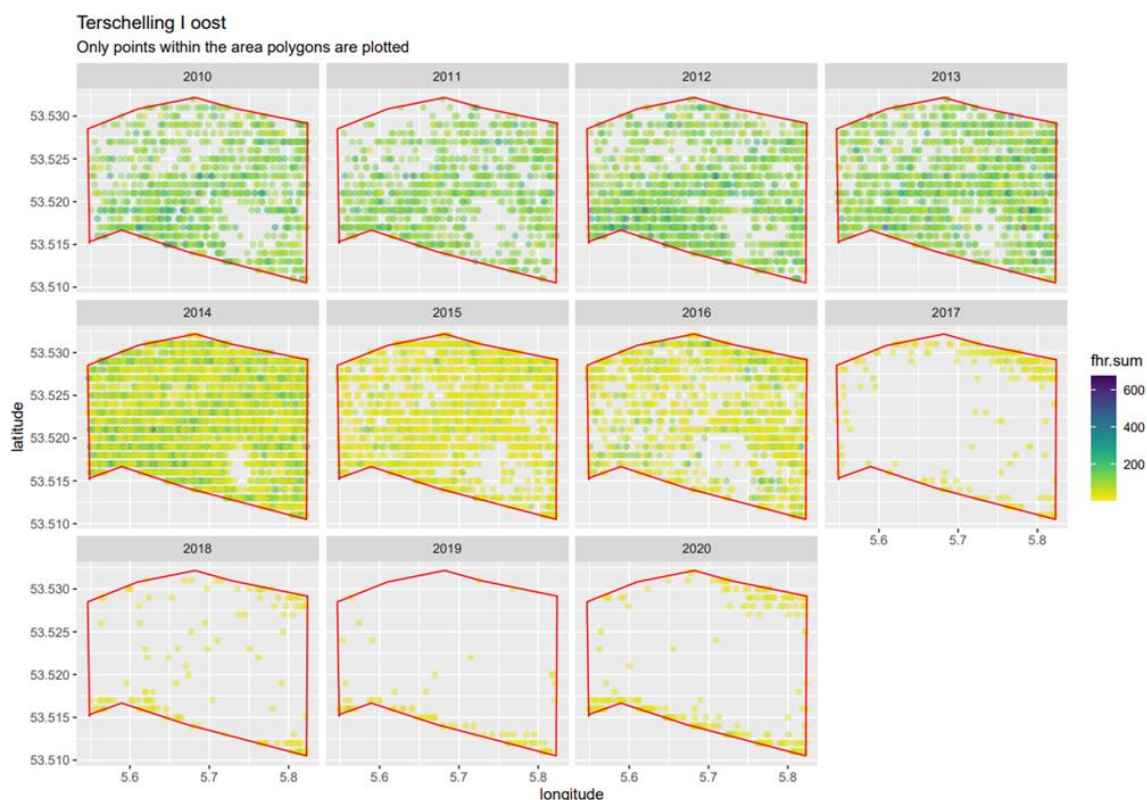


Figure 2. Overview of shrimp-fishery activity in one of the protected areas (Terschelling I Oost) in the NSCZ over time, to establish when fishing regulations were adhered to by fishers.

NB: The variable shown is fishing hours (fhr) at a specific location, with the colour corresponding to the cumulative hours that year in that specific location (fhr.sum in the legend) – darker colours indicate more fishing hours. Only observations within the area polygon are plotted. From 2017, there is a sudden drop in overall fishing activity in the restricted area, here shown as fishing hours. Created by Wilkes (2022).

2.2.2. Fishing displacement

A major concern of implementing fishing restrictions in specific locations is the potential for fishing activity to be displaced to alternative locations (Rijnsdorp et al., 2001; Dinmore et al., 2003). Such displaced fishing activity may have serious unintended consequences, both for the ecology and the fishery sector. For fisheries, it may have economic effects,

because steaming to and from the fishing grounds may take longer, new fishing grounds have to be explored, and competition for resources may increase. Ensuring that displacement of fishing activity does not negate the ecological benefits gained from MPAs is essential. In an earlier theoretical modelling study, the potential effects of displacement in the North Sea were modelled (Greenstreet et al., 2009). Fishing effort redistribution was modelled using international landings and fishing landing and effort data and the analysis of catch per unit effort (CPUE) data. MPAs on their own appeared unlikely to achieve significant regional-scale ecosystem benefits because local gains are largely negated by fishing effort displacement into the remainder of the North Sea. However, in combination with appropriate total allowable catch (TAC) reductions, the effectiveness of MPAs could be enhanced (Greenstreet et al., 2009).

3. AIMS AND OBJECTIVES

Understanding the spatial-temporal patterns of fishing effort re-allocation around MPAs is essential for assessing their effectiveness (Forcada et al., 2010). Instead of a literature review and modelling potential effects, we want to analyse what has happened based on empirical data. The main objective of this report is to assess the potential spatial reallocation of fishing activities in response to MPA implementation in the NSCZ. A BACI observational experiment was set up for this case study. Physical and biological features associated with fisheries activities were examined. Then changes in fishing effort before and after the time at which the most substantial legal protections came into force were assessed and focused in and around the protected areas during these two time periods. The main questions that will be answered in this research are: what happens to the overall fishing effort for the affected fishermen (in comparison to other fishermen) after the implementation of the fully protected areas, and what variables predict where the fishermen move to (reallocation) in the NSCZ?

4. METHODOLOGY

4.1. Potential variables for displacement

There are many variables identified that might play a role in fisheries displacement after closing a fishing area (Slijkerman and Tamis 2015). In this study, general variables that influence overall fishing behaviour, like socioeconomic conditions and market conditions, will not be studied. The analysis will focus on variables influencing reallocation. The variables that are considered in this analysis will be further discussed below.

4.1.1. Logistics

Fishing ground closure

The enforcement of fishing regulations, in this case the fully protected areas, will influence where fishermen fish. It is expected that the effects of closing fishing grounds will be greater for fishers that fished in those grounds before compared to fishers that did not.

Distance to nearest harbour / landing port (costs, including fuel price)

The cost of fishing, including fuel and labour, will influence where fishermen fish. Costs will increase as the distance to the nearest landing port or harbour increases. However, because fishing effort is aggregated and fishermen may land in ports other than the nearest port, mean distance to the shore will be included in the reallocation analysis.

4.1.2. Ecological suitability

Preferred habitat of target species (/water depth)

The European Marine Observation and Data Network (*EMODnet*) habitat data has been used to build maps of benthic habitats in the case study. However, the NSCZ has only one dominant habitat type (mostly sand), which does not fully reflect the variability in suitability for the brown shrimp. Therefore, we will also focus on depth and slope. The Bathymetry Position Index (BPI) is a measure that represents the depth and slope of the seafloor in a given location (Monk et al., 2010). The BPI is used in oceanography and marine ecology to characterise the physical structure of the seafloor. BPI values are calculated by combining bathymetry (the measurement of ocean depth) with the slope of the seafloor. BPI values can highlight areas of high ecological significance or areas that may be important for conserving biodiversity. Therefore, we will include BPI values as indicators of ecological suitability for the brown shrimp in this analysis.

BPI values can be positive or negative depending on the orientation of the seafloor slope. Higher BPI values indicate steeper slopes that are often associated with areas of the seafloor that have abundant and diverse marine life, such as seamounts, ridges, and upwelling zones, where currents bring up deep, nutrient-rich water that supports a rich tapestry of marine life. BPI values closer to zero, on the other hand, are often associated with areas of the seafloor that are relatively featureless and have lower levels of biodiversity, such as abyssal plains or areas with a gently sloping seafloor.

4.1.3. The data

The study area and MPAs

This study focuses on the NSCZ. A distance of 14 km from the coastline was used to delineate the study area as this area accounts for around 97 % of all fishing effort in the NL exclusive economic zone (Wilkes, 2022,

Figure 3). The study area had to be bordered to ensure that the number grid cells with a 0-hour fishing effort was not too large, as this causes challenges in the analysis. Coordinates of the protected areas were provided by the Ministerie van Economische Zaken (Ministry of Economic Affairs) (2013).

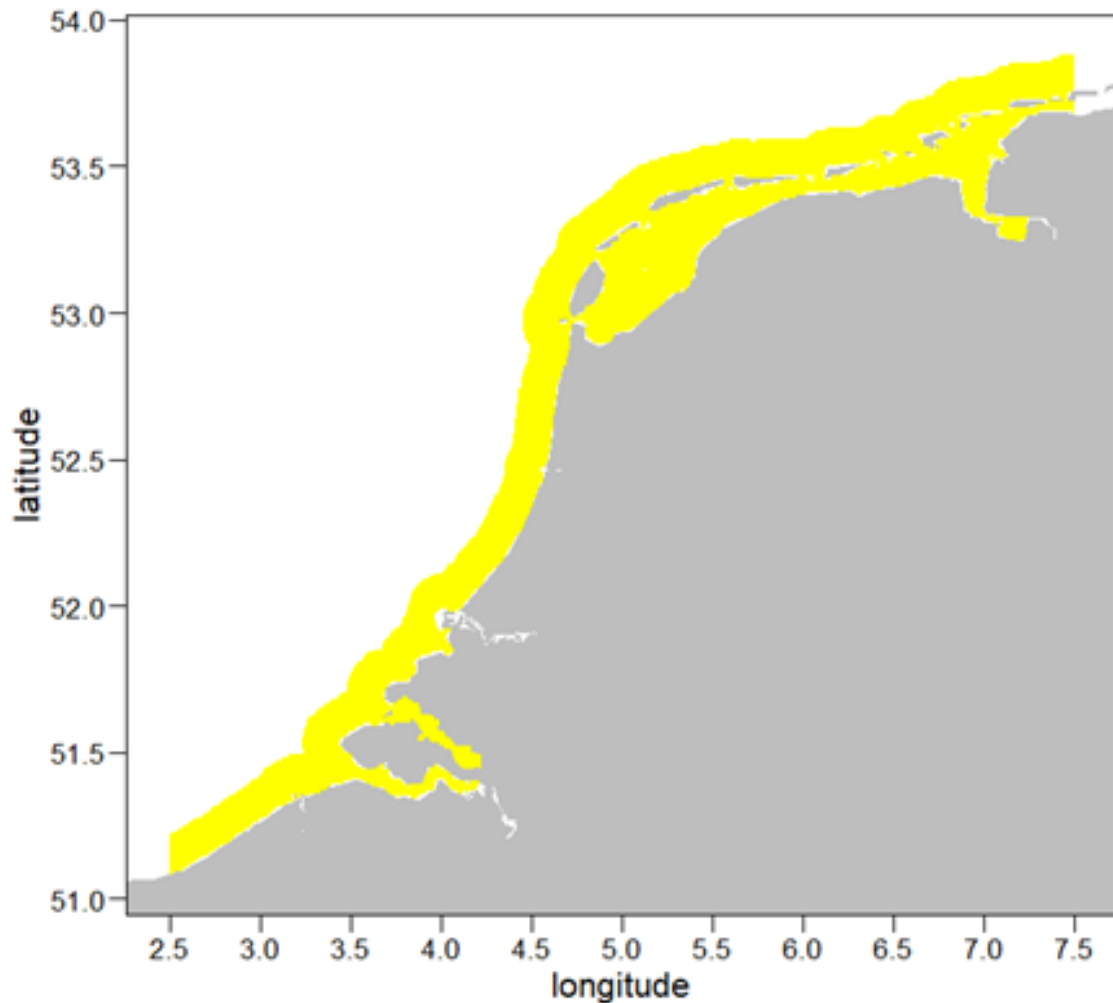


Figure 3. Map of the 14 km offshore delineation of the study area highlighted in yellow. (Wilkes, 2022)

Vessel Monitoring System and logbook data

Since 2005, all fishing vessels over 15 m have had to be equipped with a vessel monitoring system (VMS). In 2012, this became obligatory for all vessels over 12 m. A VMS transponder sends a signal at regular intervals to a satellite providing information on the vessel's ID, position, time and date, direction and speed. Hence, VMS is a valuable data source for studying the spatial and temporal distribution of fishing fleet. Wageningen Marine Research (WMR) receives VMS and logbook data from the Netherlands Enterprise Agency (RVO) for scientific purposes. We used VMS and logbook data for the time period 1 January 2012 to 31 December 2021. The VMS is coupled to fisheries logbooks, which contain information on when fishermen leave the harbour, what gear has been used for fishing, catch composition, and a rough estimate of the location of the catches for every 24 hours.

Habitat data

A dataset with readily available information was provided by the International Council for the Exploration of the Sea Working Group on Spatial Fisheries Data (ICES WGSFD, 2021). See Table 2 for relevant variables.

Table 2. Overview of relevant habitat variables in dataset provided by ICES

Variable name	Description
C-square	Identifier of 0.05 degree C-square grid cell
Distance to coast	Distance to coast for centre of C-square location
BPI5	Bathymetry position index (range of 5 km)
Mud percentage	Percentage of mud inside a C-square
Sand percentage	Percentage of sand inside a C-square
Gravel percentage	Percentage of gravel inside a C-square
Rock 50 cm	Percentage of rock content (minimum 50 cm) inside a C-square

The BPI is a measure of where a referenced location is relative to the locations surrounding it. It classifies landscape structure (e.g. valleys, plains, hill tops) based on the change in slope position (Walbridge et al., 2018). This allows for the identification of underwater sand ridges, troughs and relative flat areas.

4.1.4. Data processing

First, VMS and logbook datasets were cleaned following steps described in Hintzen (2022). They were then linked using the unique vessel identifier and date-time stamp in both datasets available. Fishing expeditions using bottom gear types known to fish shrimp (metiers with the code 'TBS', which is specific for beam trawl for shrimp fisheries), were selected. Lastly, cumulative fishing hours were aggregated by 0.05 degree C-square, and separated for year (and therewith, before or after restrictions), protected area (yes/no) and fishing group (hindered or unhindered).

Fishing groups indicate whether a specific vessel used to fish in the protected areas before the establishment of the protected areas. Two groups were distinguished: (i) vessels that spent at least 1 % of their fishing effort trying to catch shrimp in at least one of the protected areas before 2017 (these vessels have been hindered by restrictions, and are therefore referred to as 'hindered'); (ii) vessels that spent less than 1 % of their total fishing effort within the protected areas throughout the study period before 2017 (these vessels have not been hindered to the same extent, and are referred to as 'unhindered', Figure 4). In summary, there were 80 vessels in the hindered group and 150 vessels in the unhindered group.

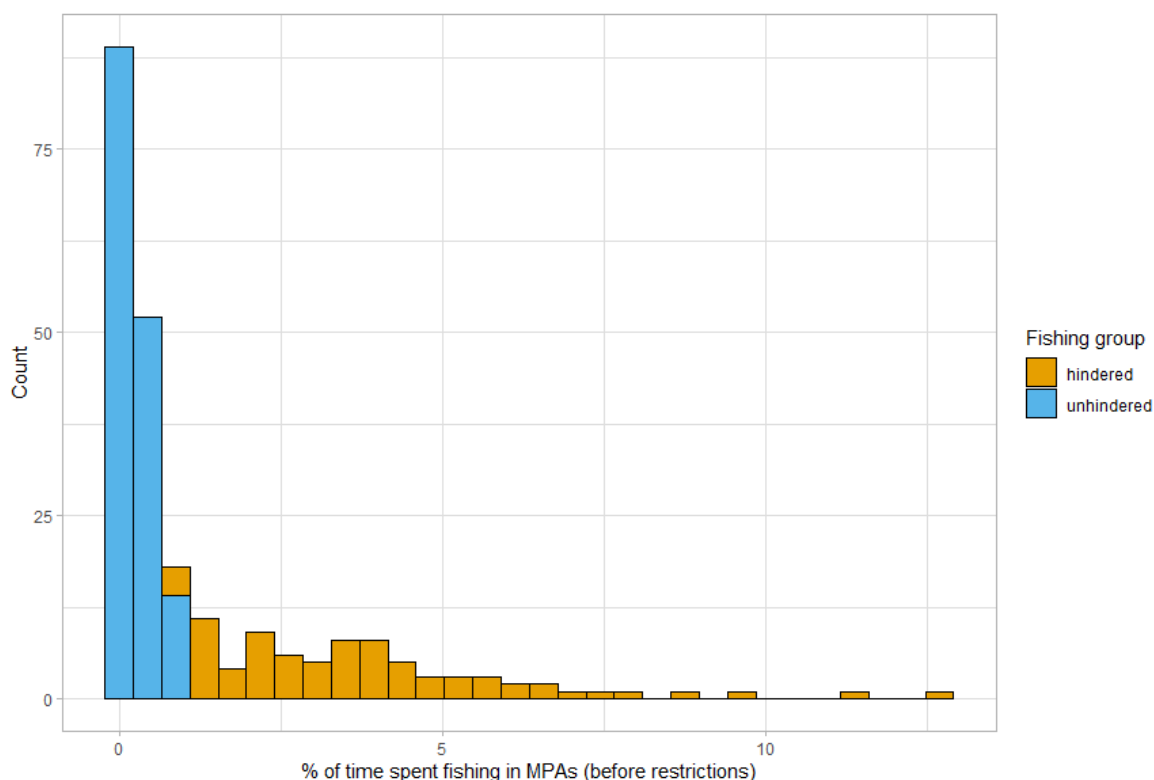


Figure 4. A histogram of the mean time spent fishing in the MPAs per fishing vessel before restrictions were adhered to.

NB: Two groups were distinguished and used as the control and impact group. In this analysis, the unhindered group, in blue on the left of the plot spent less than 1 % of their fishing time in the protected areas before restrictions were adhered to. The hindered group, in orange on the right, spent 1 % or more of their fishing time in these areas.

4.2. Statistical analysis

The aim is to conduct a post-hoc BACI observational experiment. In the current dataset, the following BACI set up was designed:

- Before: before the restrictions were adhered to (pre-2017);
- After: after the restrictions were adhered to (post-2017);
- Control: the unhindered fishing group;
- Impact: the hindered fishing group.

The response variable is the fishing effort (in cumulative fishing days per year) per 0.05 degree c-square per year. The specific analysis will be described per theme.

4.2.1. Changes in fishing effort before and after restrictions were adhered to

To assess changes in fishing effort before and after spatial restrictions were adhered to, a generalised linear mixed model (GLMM) with a negative binomial error distribution was fitted. The model was formulated with fishing effort as response variable, and an interaction of the period (before or after restrictions) and fishing group (hindered or unhindered) as the predictors (eq. 1). This will indicate whether an overall reduction in effort has occurred since the enforcement of protected areas, and whether this reduction

is specific to the hindered fishing group or whether there is a reduction across the entire fishery sector.

$$\text{Fishing effort} \sim \text{Period (before vs after)} \times \text{Fishing group (hindered vs unhindered)} \quad (\text{eq. 1})$$

4.2.2. Changes in fishing effort within and outside the protected areas before and after restrictions

To assess changes in fishing effort within and outside the protected areas before and after spatial restrictions were adhered to, a GLMM with a negative binomial error distribution was fitted. The model was formulated with fishing effort as response variable, and an interaction of the area (open or restricted i.e. one of the five protected areas) and period (before or after restrictions) as the predictors (eq. 2). This will show whether an overall reduction in effort has occurred since the enforcement of protected areas, and whether this reduction is specific to the protected areas or not.

$$\text{Fishing effort} \sim \text{Area (one of the five MPAs vs open area)} \times \text{Period (before vs after)} \quad (\text{eq. 2})$$

4.2.3. Changes in habitat use by fisheries in response to the protected areas

There are multiple indicators for habitat. As mud percentage and sand percentage were almost completely collinear ($r=.99$), mud percentage is dropped from further analysis. First, two GLMMs with binomial error structure were fitted, with fishing effort as response variable, and distance to coast, BPI5, sand percentage, gravel percentage, and 50 cm rock percentage as predictor variables, all in interaction with the period (before or after restrictions were adhered to (eq. 3)). Comparing the coefficients of these models will give us insights into changes in the main habitat-predicting variables for fishing effort before and after restrictions, and tell us which changes are significant.

$$\begin{aligned} \text{Fishing effort} \sim & \text{Period} \times \text{Distance to coast} + \\ & \text{Period} \times \text{BPI5} + \\ & \text{Period} \times \text{Sand percentage} + \\ & \text{Period} \times \text{Gravel percentage} + \\ & \text{Period} \times \text{Rock 50 cm} \end{aligned} \quad (\text{eq. 3})$$

4.2.4. Visualisation spatial reallocation fisheries in response to the MPAs

A fishery effort map was created for mean effort per year before and after regulations were adhered to, as well as a map of absolute change of mean effort per year per 0.05 degree C-square. Analyses were carried out in R (R version 3.6.3, R Core Team, 2020 and R studio version 1.3.959 (RStudio Team, 2020)). The data processing used the VMStools R-package (Hintzen et al., 2012) and all its dependencies.

5. RESULTS

All models significantly diverged from a null model (same model structure without predictor variables, compared with a likelihood ratio test). No model had dispersion problems (dispersion parameters ranged between 0.88 and 1.86).

5.1. Changes in fishing effort before and after restrictions were adhered to

The results of the negative binomial GLMM assessing fishing effort for the fisher groups in relation to the period are shown in Table 3. The period after regulations were adhered to had a negative effect on fishing effort (estimate = 0.418, $p < 0.001$). In general, the unhindered fishing group fished more than the hindered fishing group (estimate = 0.426, $p < 0.001$), and this difference was consistent before and after regulations (estimate = -0.006, not significant (N.S.), Figure 5), meaning that the reduction in effort was not statistically stronger for one particular group. Therefore, protection of the areas did not significantly reduce fishing effort for the hindered group only.

Table 3. Summary of coefficients of the model assessing fishing effort as a response to period and fishing group

Term	Estimate	Std error	z-value	P-value
Intercept	2.430	0.032	74.771	-
Period (after)	-0.418	0.049	-8.524	< 0.001
Fishing group (unhindered)	0.426	0.045	9.547	< 0.001
Period * Fishing group	-0.006	0.062	-0.102	0.919

NB: The p-value of the intercept is not shown because of the limited interpretation.

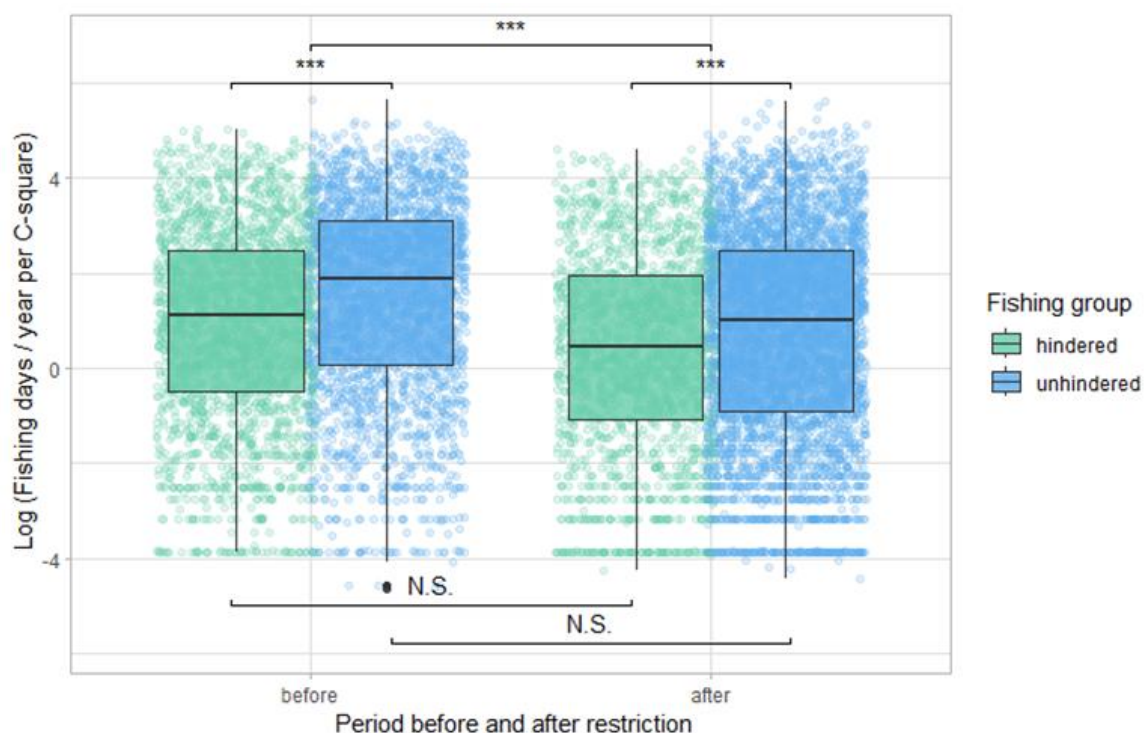


Figure 5. A boxplot comparing fishing days per year per C-square (here displayed as a log function) between fishing groups (hindered vs unhindered; before adherence to regulations and after adherence to regulations).

NB: The lines extending vertically from the boxes indicate the range of the data. The boxes represent the interquartile range, which covers the middle 50 % of the data, from the first quartile (Q1) to the third quartile (Q3). The horizontal line inside the box represents the median value of the data of that group.

The results of the negative binomial GLMM assessing fishing effort within and outside the protected areas in relation to the period are shown in Table 4. There is a significant interaction between the period before and after the MPA designation, when fishing effort decreased strongly in protected areas (estimate=-1.491, $p < 0.001$, Figure 6). As the interactive effect is significant, the individual predictors will not be considered.

Table 4. Summary of coefficients of the model assessing fishing effort as a response to period and protected areas

Term	Estimate	Std error	z-value	P-value
Intercept	8.823	0.209	42.186	-
Period (after)	0.025	0.304	0.081	0.935
MPA (yes)	-3.970	0.297	-13.365	< 0.001
Period (after) * MPA (yes)	-1.491	0.435	-3.428	0.001

NB: The p-value of the intercept is not shown because of the limited interpretation.

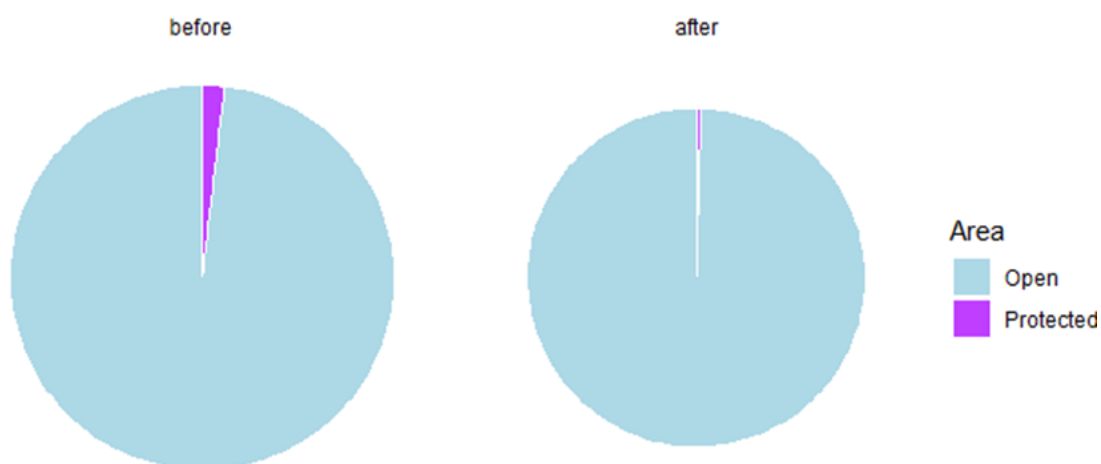


Figure 6. Pie plots comparing fishing days per year per 0.05 degrees C-square before adherence to regulations and after adherence to regulations.

NB: The cut-out (purple) shows the share of time spent in the now protected areas. The overall size of the circle represents the total time spent fishing per year.

5.1.1. Changes in habitat use by fisheries in response to the MPA

The results of the negative binomial GLMM assessing predicted fishing effort by habitat variables in relation to the period are shown in Table 5. Distance to the coast has a negligible effect (estimate = < 0.001 , $p < 0.001$), indicating a biologically insignificant impact. Sand has a significantly negative effect (estimate = -1.491 , $p < 0.001$), suggesting that, overall, higher sand percentages correspond to decreased fishing effort. The only significant interactive effect is that of the percentage of gravel and period (estimate = -1.491 , $p < 0.001$, Figure 7). After regulations, the effect of avoiding locations with high gravel content became less pronounced. For more details of the general relations between the habitat descriptive variables and fishing hours, see Annex 1.

Table 5. Summary of coefficients of the model assessing fishing effort in relation to habitat descriptive variables

Term	Estimate	Std error	z-value	P-value
Intercept	4.303	0.327	13.158	-
Period (after)	-1.112	0.429	-2.591	0.010
Distance to coast	0.001	0.000	-15.871	< 0.001
BPI5	0.025	0.011	2.355	0.019
Sand %	-0.010	0.004	-2.815	0.005
Gravel %	-0.312	0.024	-13.218	< 0.001
Rock % (50 cm)	-0.220	0.624	-0.353	0.724
Period (after) * Distance to coast	0.000	0.000	0.762	0.446
Period (after) * BPI5	0.019	0.014	1.341	0.180
Period (after) * Sand %	0.007	0.005	1.447	0.148
Period (after) * Gravel %	0.156	0.031	4.988	< 0.001
Period (after) * Rock % (50 cm)	-0.402	0.821	-0.489	0.625

NB: The p-value of the intercept is not shown because of the limited interpretation. As the interaction has a significant effect, it is not relevant to interpret individual variables.

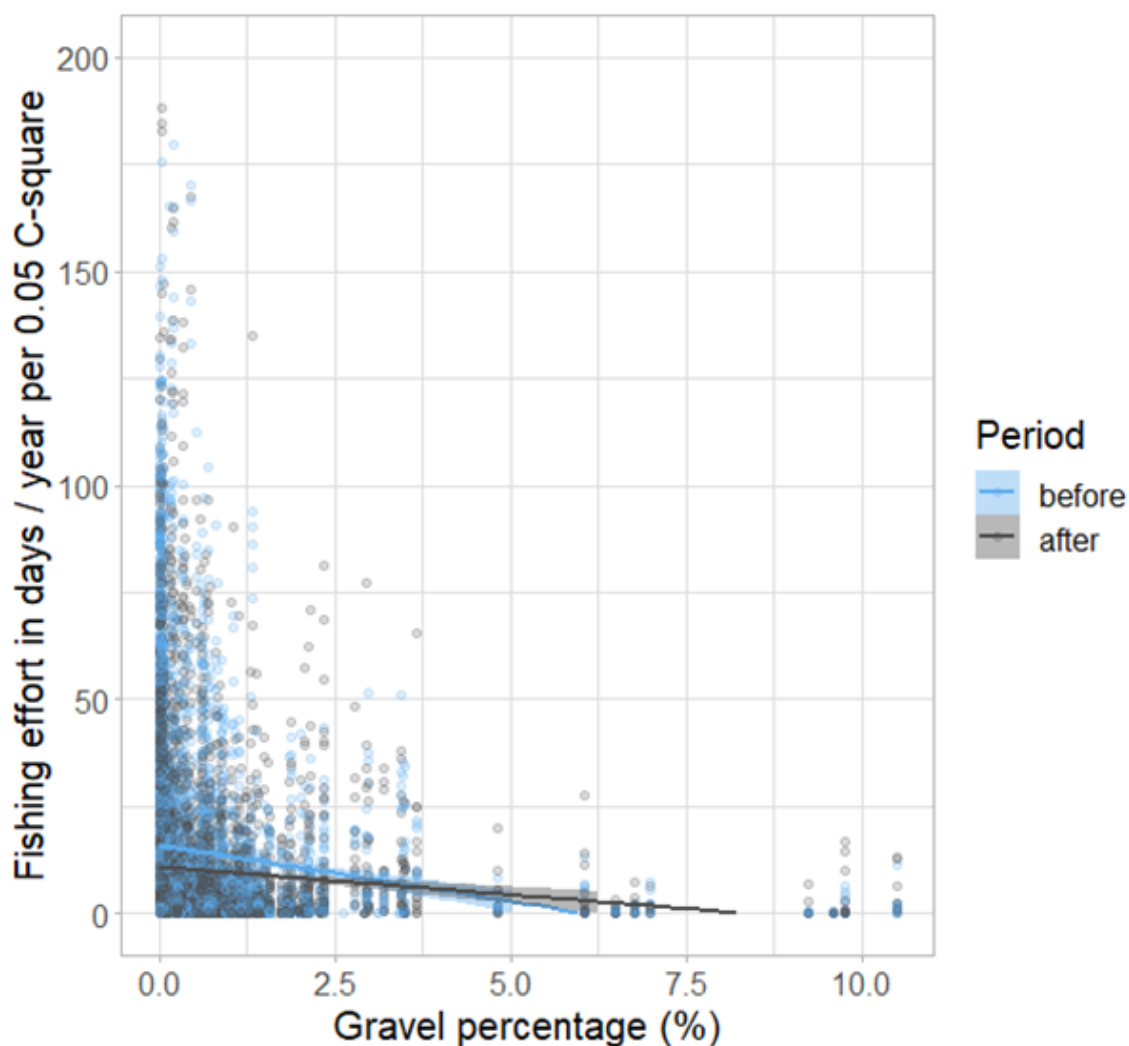


Figure 7. Observations for fishing effort in days per year per 0.05 degree C-square (Y-axis) and the percentage of gravel content in that 0.05 degree C-square (X-axis).

NB: The coloured lines show the relationship between gravel and effort for the two periods.

5.1.2. Visualisation of the spatial reallocation of fisheries effort in response to the designation of protected areas

Before versus after

The visualisation of fishing effort per year before and after regulations are shown in Figure 8. The plots are cropped to the most important areas (surrounding the protected areas), but a plot of the entire coastline of the Netherlands can be found in Annex 2. Differences between the periods are not very apparent. There seems to be overall less activity surrounding the protected areas. To better visualise the differences, change per 0.05 degree C-square is shown in Figure 9 (and Annex 3 for the full coastline). Although the changes per C-square are the main focus of this study, it is important to have an understanding of the absolute amount of fishing to put the absolute differences into perspective.

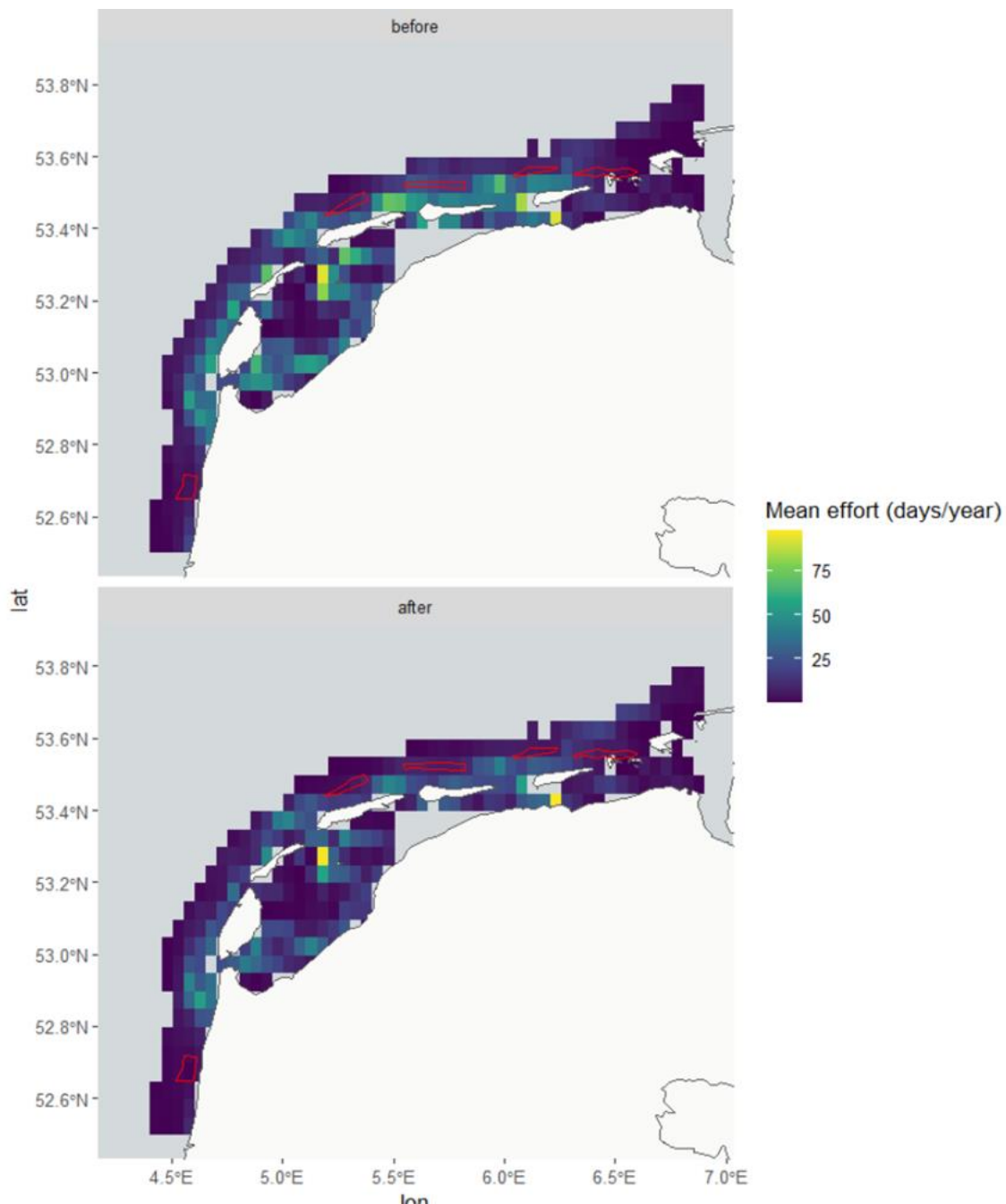


Figure 8. Maps comparing fishing days per year per 0.05 degrees C-square between periods before adherence to regulations and after adherence to regulation.

NB: The map shows the north-west of the Netherlands, where all protected areas are located. The coloured blocks indicate the effort in days per year, with lighter colours indicating more days. The protected areas are delineated in red.

Absolute change in effort

The visualisation of the mean difference in fishing effort per year before and after regulations is shown in Figure 9. A map of the entire coastline of the Netherlands can be found in Annex 3.

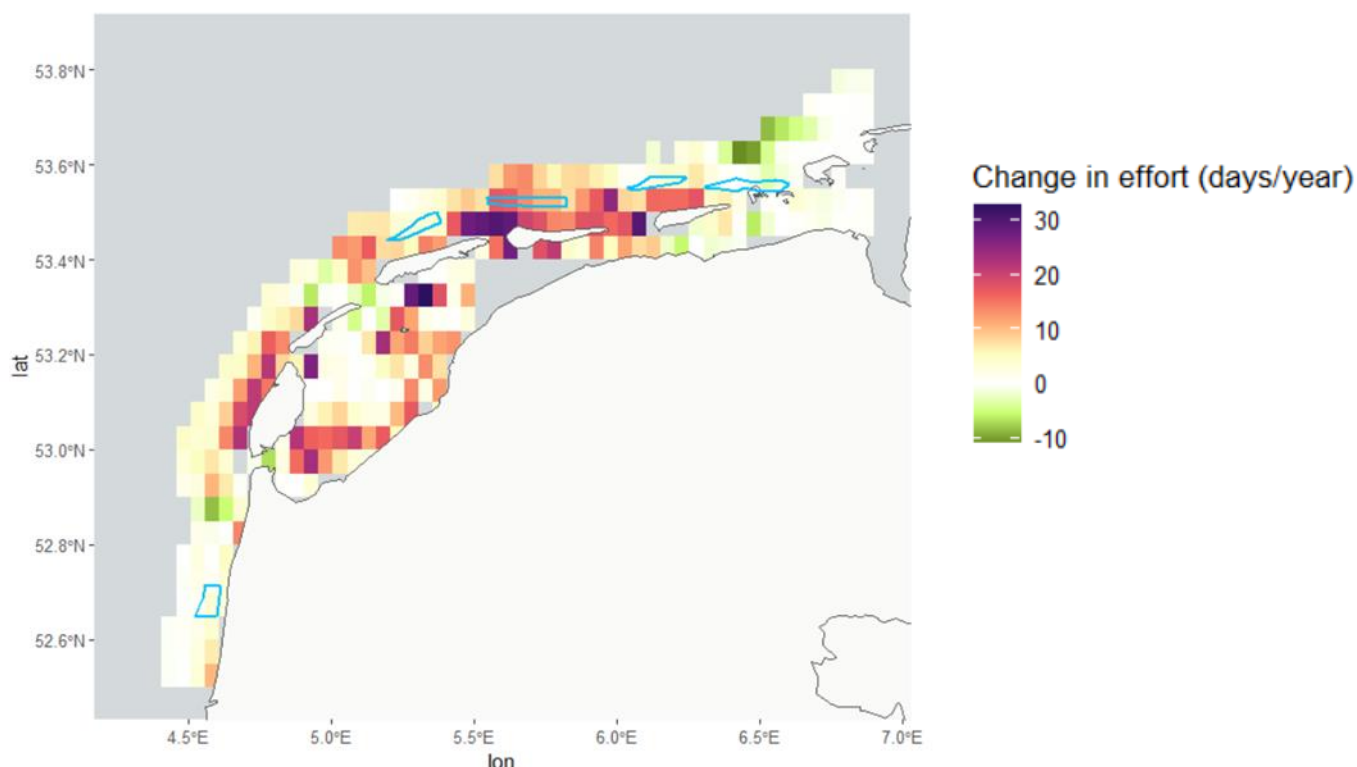


Figure 9. Map visualising the mean absolute change in fishing effort in days per year per 0.05 degrees C-Square in the periods before and after adherence to regulations.

NB: The map shows the north-west of the Netherlands, where all protected areas are located. The coloured blocks indicate the delta effort in days per year: for the yellow–purple blocks, the darker the colour, the greater the number of days; shades of green indicate a decrease in fishing effort – the darker the colour, the greater the reduction in the number of days. The protected areas are delineated in blue.

6. DISCUSSION

The main objective of this case study was to assess the potential spatial reallocation of fishing activities in response to MPA implementation in the NSCZ.

First, we examined whether there were changes in the total fishing effort before and after 2017, which is the time we assumed fishing restrictions were adhered to. **On the whole, there was significantly less fishing effort in the study area in the period after the protected areas were established.** However, this was not significantly different for the hindered group and the unhindered group. This could mean several things. The first is that the restriction hindered the entire shrimp fishing sector, not only the fishers that fished in the now-protected areas before the restrictions. This could be because of increased competition in space, as the same number of fishers now have to fish in a smaller area. The second is that the restrictions resulted in reduced motivation to stay in the fisheries sector. A third possibility is that the entire fishing sector has been reduced as a result of other factors not studied here, such as general fishing law, the availability of shrimp, etc.

We also compared fishing effort within and outside the protected areas before and after 2017. As expected, there was a significant interaction between the area (protected or not) and the period (before or after 2017). After the regulations were adhered to, **there was**

significantly less fishing activity in both the open and restricted areas, but the effect was strongest in the areas where restrictions applied. This tells us that the greatest reduction of fishing effort occurs in the protected areas, and fishing restrictions are having the desired effect. Post-2017, there was almost no fishing effort in the protected areas, and this reduction in **effort was not compensated for in areas outside the MPAs**, as effort here also decreased. **With these findings, we can rebut the concern that implementing fishing restrictions in specific locations leads to a 1 to 1 displacement of fishing activity to alternative locations** (Rijnsdorp et al., 2001; Dinmore et al., 2003).

Lastly, we assessed changes in habitat use by fisheries in response to the MPAs. There was a significant interactive effect of period and gravel percentage. For both periods, the tendency to visit locations decreases with higher gravel content. However, **after restrictions** were adhered to, these negative effects became less pronounced, and **fishers visited areas with higher gravel percentages more frequently**. This could be because of space limitations as fishers no longer have the option to avoid plots with high gravel contents. Further inspection of the habitat preferences (Annex 1) shows that BPI has a significant positive effect, and that there is no difference between before and after restrictions. Distance to the coast has no significant effect, very likely because of our hard cut-off boundary at 14 km of the shore, thereby removing all vessels that do travel further from further analysis. It is however a positive sign that the **fishers do not travel significantly further after the establishment of the protected areas**. The effect of sand content was biologically negligible, most probably because the sand content along the Dutch coast is so uniform.

The visualisation of the change in fished areas revealed that most of the effort displacement stayed within the Northern Wadden area. This could be because most of the harbouring places for shrimp fishers are located along the Wadden Sea coastline (e.g. Lauwersoog, Den Helder, Oudeschild). This could be studied further by analysing individual movement tracks (e.g. by methodologies of Bastardie et al., 2010). The C-squares with the most increase in fishing effort after regulations are positioned in the centre of the Wadden area, with the highest values in the north-eastern part of Terschelling and around Ameland (the third and fourth Wadden Sea islands). This is quite remarkable because this area closely borders two protected areas. This phenomenon can be explained in several ways. First, it could be that this was a very preferred area in general, and because of the spatial limitations, the effort is now aggregated in the spaces between the restricted areas. However, the areas with reduced effort are not nearby, making this explanation of a small shift in space quite unlikely. Second, the fishers might be attracted towards the border of these central protected areas as they might expect or notice fishery spillover effects. Third, a slight decrease in fishing activity bordering the German Wadden is observed. Although not studied here, it could be that the increasing fuel prices discourage fishermen from travelling far from the harbours, which could lead to the observation of lessened fishing activity on the Dutch borders.

There are several points of improvement for future research into displacement effects. Rather than looking at absolute differences in fishing pressure in days, it might be interesting to look at how fishing pressure changes spatially, corrected for the overall reduction in fishing hours. This might better represent the spatial preferences of the fishermen and excludes other reasons why fishers may choose to fish less. In this analysis of fishing pressure, total time spent on the water was accumulated, and no distinction was made between when the vessels were travelling towards their fishing locations and when they were fishing. There are ways to distinguish between these two behaviours, based on the vessel's speed profiles (Hintzen et al., 2012); in future, this would be a more refined

approach. Lastly, our preferred indicator of fishing effort was in absolute time spent in the water, but there are many other ways to assess fishing effort. One could look at the total biomass gathered (kg) or at the area swept (km²), which might be more representative of the ecological consequences of fishing and displaced fishing. Future research could look at other indicators of the ecological consequences of displacement of fishing effort, and further look into area swept and total biomass taken out. It is also important to study what happened to the revenues of these fishermen, to better understand the consequences on the fishers. Lastly, it would be of great value of to see whether potential ecological/fisheries spillover has an effect on the choices of fishermen.

7. CONCLUSIONS

The main objective of this report was to assess the potential spatial reallocation of fishing activities in response to MPA implementation in the NSCZ. **The establishment of fully protected areas significantly reduced the fishing effort** for the hindered group and the unhindered control group. Therefore, the reduction in fishing effort cannot be explained fully by the closing of the areas. Although there was significantly less fishing activity in both the open and restricted areas after adherence to restrictions, the effect was strongest in the areas where restrictions applied. Post-2017, and the introduction of fines, there was almost no fishing effort in the fully protected areas, which tells us that the restrictions are having the desired effect in terms of fishing effort. After the implementation of the restrictions, there appeared to be a minor change in the habitats visited and in fisher preferences. The lack of effect of distance to the coast informs us that **fishers did not have to travel significantly further after the restrictions**. However, **fishers did visit areas with higher gravel percentages more frequently after the establishment of the MPAs**. The visualisations of changes in fishing areas before and after restrictions revealed that reallocation mostly happened around the most central MPAs, and that effort decreased furthest around the far peripheries of the Wadden islands.

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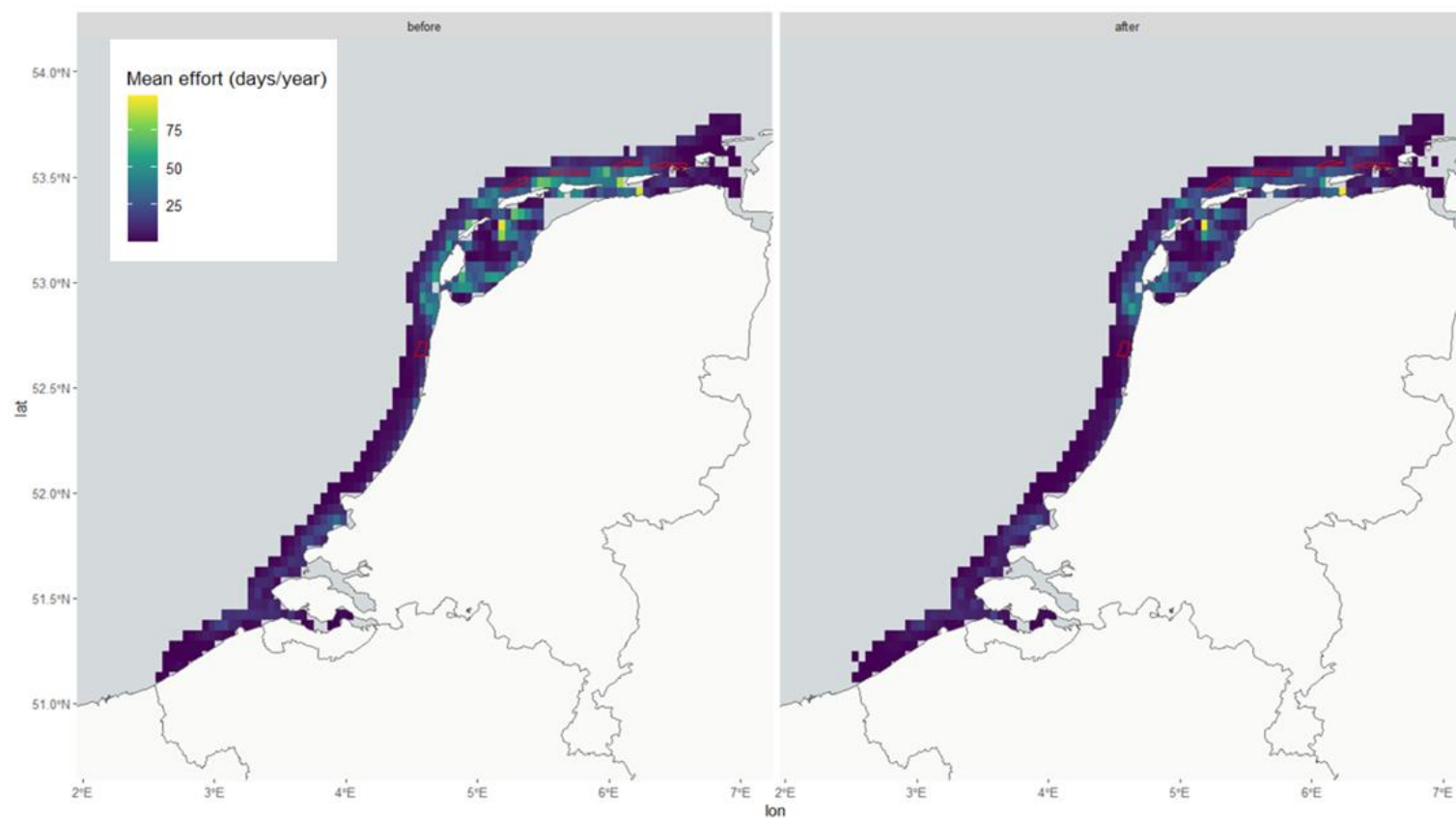
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APPENDIX 1: COEFFICIENTS OF STATISTICAL MODEL ASSESSING HABITAT DESCRIBING VARIABLES IN RELATION TO FISHING EFFORT

Term	Estimate	Std error	z-value	P-value
Intercept	3.869	0.212	18.217	-
Period (after)	-0.409	0.032	-12.610	< 0.001
Distance to coast	0.000	0.000	-23.035	< 0.001
BPI5	0.036	0.007	5.127	< 0.001
Sand %	-0.006	0.002	-2.556	0.011
Gravel %	-0.315	0.021	-14.996	< 0.001
Period (after) * Gravel %	0.149	0.027	5.519	< 0.001

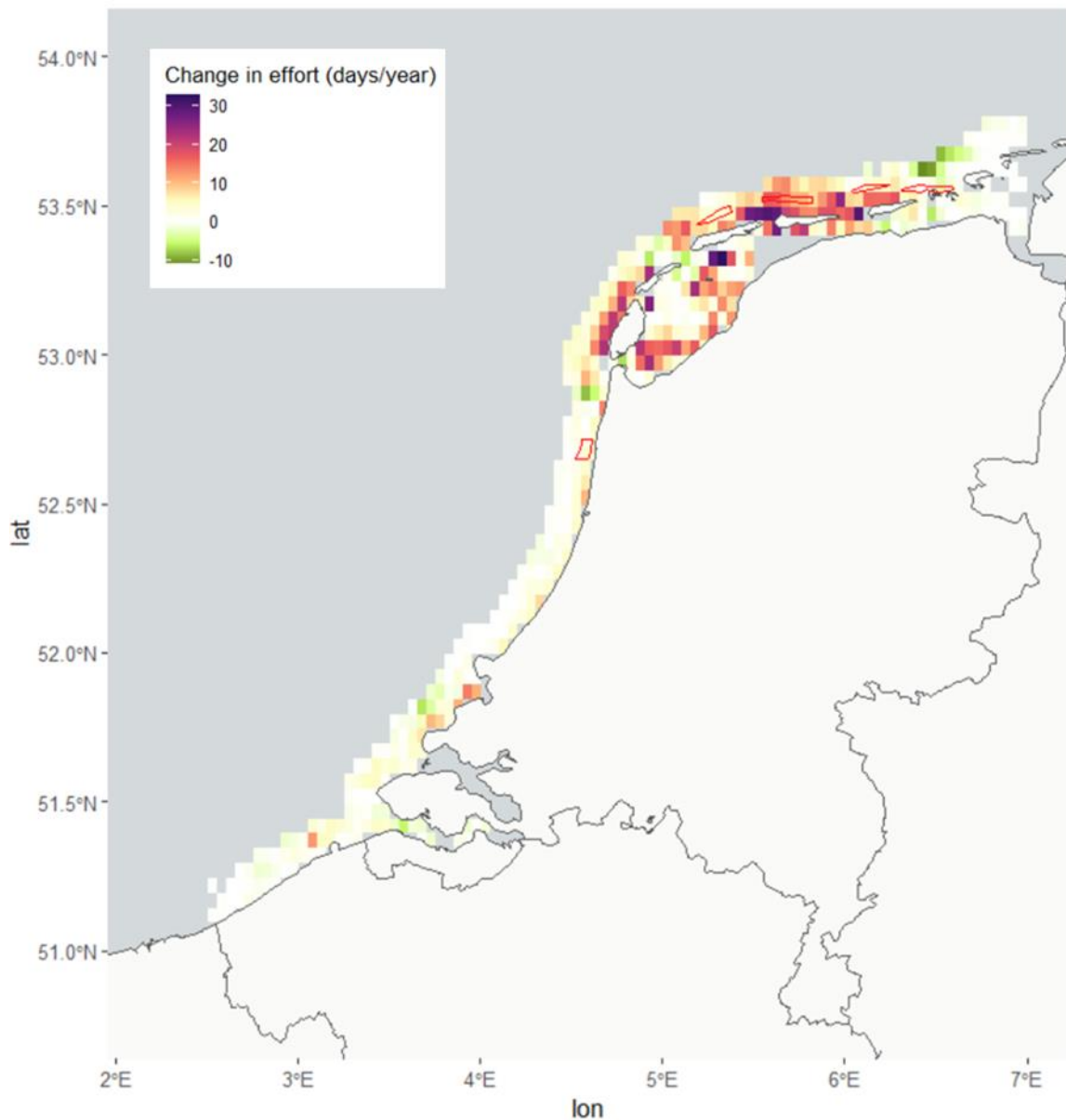
Summary of coefficients of the model assessing fishing effort in relation to habitat descriptive variables. Showing the variable name (term), followed by the estimate, standard error (Std error), z-value and p-value. The p-value of the intercept is not shown because of the limited interpretation. The individual predictor variables (without interaction) describe the general spatial preference of fishers, regardless of whether this was before or after restrictions. It was already known that after restrictions, there was less fishing effort in general (*estimate* = -0.409, $p < 0.001$). Distance to the coast does not influence fishing pressure, as the estimate of this effect is zero. BPI5 significantly predicts fishing pressure, with more complex floor structures leading to higher fishing effort. Sand content also significantly predicts fishing effort, but this effect is biologically negligible. It was previously seen, and consistent in this model, that in the period after the restrictions, the effort was displaced to locations with higher gravel content.

APPENDIX 2: FISHING PRESSURE BEFORE AND AFTER MPA IMPLEMENTATION



Maps comparing fishing days per year per 0.05 degrees C-square between periods before adherence to regulations (left plot) and after adherence to regulation (right plot). The map shows the Netherlands, where the fully protected areas are located in the north and west around the Wadden islands (delineated in red). The coloured blocks indicate the effort in days per year, with lighter colours indicating more days.

APPENDIX 3: THE CHANGE OF FISHING EFFORT BEFORE AND AFTER MPA IMPLEMENTATION



Map visualising the mean change in fishing effort in days per year per 0.05 degrees C-square in the periods before and after adherence to regulations. The map shows the Netherlands, where fully protected areas are located in the north and west around the Wadden islands (delineated in red). The coloured blocks indicate the delta effort in days per year: for the yellow–purple blocks, the darker the colour, the greater the number of days; shades of green indicate a decrease in fishing effort – the darker the colour, the greater the reduction in the number of days.

Case Study Report

The Flemish Banks Belgium – North Sea



Coastal beam trawler fisher (Belgium waters). Photo by ILVO

Mapping of marine protected areas and their associated fishing activities

Katrien Verlé, Ellen Pecceu, Gert Van Hoey

***Flemish Research Institute of Agriculture, Fishery and Food
Belgium***

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LIST OF ABBREVIATIONS

Term	Description
BPNS	Belgian Part of the North Sea
CFP	Common Fisheries Policy
EU	European Union
GT	Gross Tonnage
kW	Kilowatt
LIFSN	Low Impact Fishery Southern North Sea
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive
MSP	Marine Spatial Plan
nm	Nautical Mile
SAC	Special Area of Conservation
TAC	Total Allowable Catch
VMS	Vessel Monitoring System

1. EXECUTIVE SUMMARY

An area in the south-west part of the Belgian North Sea called 'Vlaamse Banken' or Flemish Banks was designated as a Special Area of Conservation (SAC) under the Habitats Directive in the framework of Natura 2000 in 2012. The area was allocated mainly to protect sandbanks and reefs. Conservation objectives are defined in the SAC, but no real management plan is in force yet. Further, there are currently no fishing restrictions in the Flemish Banks for commercial fisheries, despite previous and ongoing efforts to implement management measures to preserve sea-floor integrity.

The main objective of this case study is to assess spatial reallocation of fishing activities in response to the MPA implementation process over the period 2007–2022. The specific objective is to evaluate whether the management of fisheries in and around the Flemish Banks area has had an influence on fishing activity. Currently, there are three management areas defined where fishery restrictions will be implemented (by 2025). This is done based on fishery activity data using vessel monitoring systems (VMS) of all countries active in the area. The Netherlands and Belgian fishery are the most active within the area, executing mainly beam trawl fishery.

Since the designation of the Flemish Banks as a SAC in 2012, findings show a small decline but no relative change in total fishing effort. Indeed, the share of effort and landings in the Flemish Banks compared to the Belgian Part of the North Sea (BPNS) was relatively stable overall and particularly for the Netherlands and Belgium, over the entire time period investigated. Similar patterns could be observed when considering important gears (beam trawl, shrimp trawl), with exception to some increase in the coastal fishery activities mainly outside the SAC. In general, it appears that fishers did not adjust their behaviour neither following the designation nor when fishing measures are pending. The area holds important fishing grounds, therefore announcing possible measures and closures is not sufficient to lead to behavioural changes. Possible alternative fishing grounds are limited and declining due to other uses of space in the vicinity. The BPNS is also intensely and increasingly used for other activities such as a zone for renewable energy and area for sustainable aquaculture, and sand extraction areas. On the other hand, no increases in fishing effort were observed in the Flemish Banks relative to the BPNS, indicating that there are no claims on the area either.

This study indicated that the Flemish Banks MPA Natura 2000 area is an important fishing area, so the implementation of the management areas might have consequences for future fishery activities within the BPNS. For future work, it can be worthwhile to test possible fishery displacement effects in the BPNS based on fishery displacement models.

2. BACKGROUND

An area in the south-west of the Belgian part of the North Sea called 'Vlaamse Banken', or Flemish Banks, was designated as a Special Area of Conservation (SAC) under the Habitats Directive in the framework of Natura 2000. Its designation was based on scientific advice (Degraer et al., 2009). The area was allocated mainly to protect two habitat types (Pecceu et al., 2021). The first is classified as 'sandbanks which are slightly covered by sea water all the time' (habitat type 1110). The second is classified as 'Reefs' (habitat type 1170) and include gravel beds and polychaete worm (*Lanice conchilega*) biogenic aggregations that occur within sandbank systems (Pecceu et al., 2021). The

designation of this area also aims to protect harbour porpoises and common and grey seals (Verhalle and Van de Velde, 2020).

Specific conservation objectives were adopted in 2017 and revised in 2021 by the Federal Government's Department of Environment. These were aligned with the environmental targets set under the Marine Strategy Framework Directive (MSFD) (Pecceu et al., 2021). However, negotiation regarding measures and closures for bottom-disturbing activities (e.g. fishing) is ongoing.

2.1. The Flemish Banks

In 2005, the Trapegeer-Stroombank SAC (181 km²) was designated. The SAC was extended in 2012 to cover 1,100 km² or about 30 % of the territorial waters and was redesignated as a marine protected area (MPA) named 'Vlaamse Banken' (Figure 1) (Pecceu et al., 2016). The MPA includes the port of Nieuwpoort and borders the port of Ostend. Both ports harbour commercial and recreational fishing vessels (Scherrens 2022; Verleye et al., 2022). Within this area, multiple other activities are permitted under the Marine Spatial Plan (MSP), including: a zone for renewable energy and area for sustainable aquaculture, sand extraction areas, zones for commercial and industrial activities, shipping routes, an area for military exercises and a dredging dumping site (Figure 1) (Verhalle and Van de Velde, 2020).

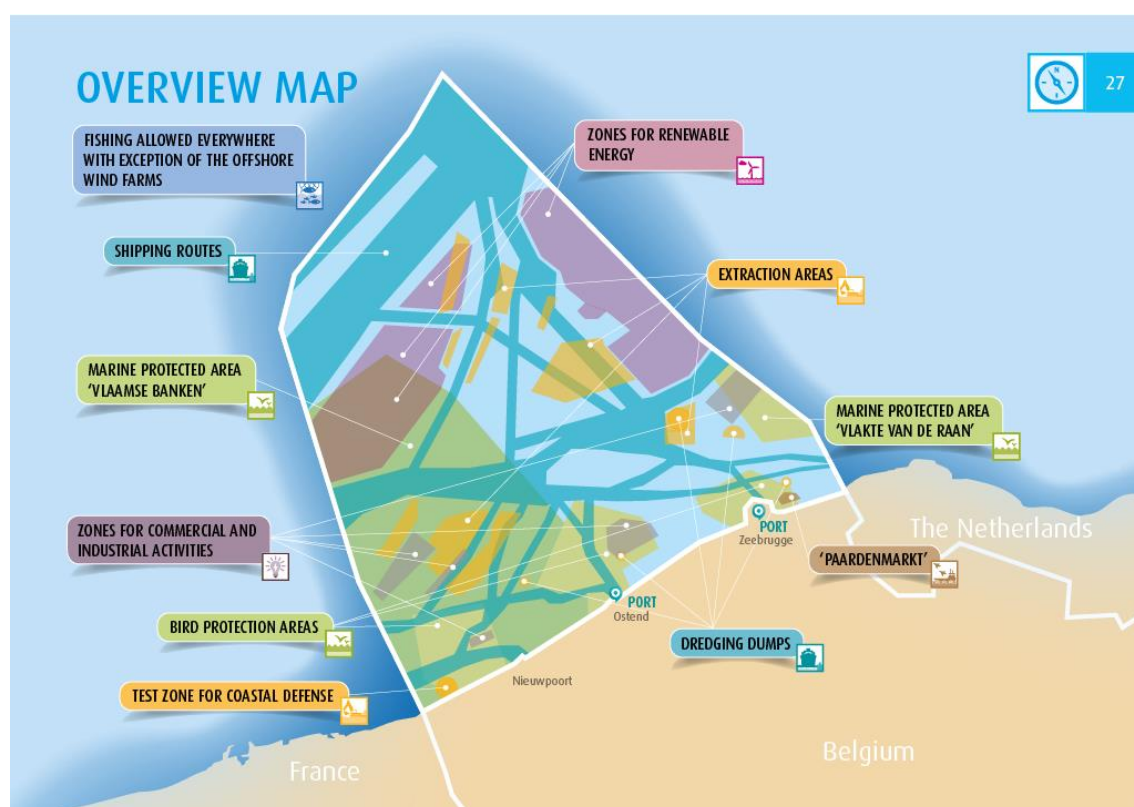


Figure 1. Activities designated in the Belgian part of the North Sea under MSP 2020–2026. The area in green on the south-western side is the 'Flemish Banks' MPA (Verhalle and Van de Velde, 2020).

2.2. MPA designation process

In 1999, Belgium started its MPA designation process in the Belgian part of the North Sea (BPNS). Several attempts failed because of a lack of scientific involvement and a lack of stakeholder/user feedback during the consultation processes; however, stakeholder involvement did increase over time.

Between 2003 and 2005, bilateral consultations were held with stakeholders on the designation of MPAs in general. A consultation on management measures was held behind closed doors. The information was explicitly stated to be confidential and there was no interaction between stakeholders. The nature of the stakeholder involvement and the impact of their input on the decision-making process remained unclear (Pecceu et al., 2016).

In 2005, the first successful MPA designations were realised as part of the European Natura 2000 network. Amongst these areas was the SAC 'Trapegeer-Stroombank' later extended and renamed 'Flemish Banks' in 2012 (Pecceu et al., 2016).

During the designation process of the Flemish Banks (between 2008 and 2011), public hearings were held to inform all who had an interest, explaining the process and its scientific basis. This led to the designation of the MPA in 2012 as a Natura 2000 site, later formalised within the process of the MSP. A roadmap for the management measures was also presented. In addition, four information meetings were held. From 2012 onwards, stakeholder involvement was also more formalised within the process of the MSP, with formal and informal bilateral stakeholder consultations on management measures in the Flemish Banks. However, the process continued to focus on bilateral consultations rather than negotiations amongst stakeholders, thereby leading to limited transparency and openness (Pecceu et al., 2016).

The first six-year MSP ran from 2014 to 2020 (Pecceu et al., 2016). The current MSP covers 2020 to 2026 (Verhalle and Van de Velde, 2020).

2.3. Management measures and subzone designations

There has already been a lengthy process to define management measures and designate subzones with fishery restrictions within the BPNS, as reflected in the changes seen in the two MSP maps (Figure 2).

In 2009, policy plans were formulated for 'Trapegeer-Stroombank'. The plans included very few measures to protect the area and fishing was still permitted. Wind farms were prohibited; however, because there was already a large area designated for renewable energy, this was not an issue for the energy industry (Pecceu et al., 2016).

The first management measures for the Flemish Banks were approved in Belgian legislation in 2014. These had implications for both commercial and recreational fisheries, with the delimitation of four subzones with fishing restriction for bottom-disturbing gears (Figure 2) (Pecceu et al., 2016). The aim was to preserve sea-floor integrity. Exceptions were made for fishing on foot or on horseback (Van de Velde et al., 2015). Recreational trawling was essentially forbidden throughout the Flemish Banks. However, exceptions were made for recreational trawlers that had been active for more than three years (Van de Velde et al., 2015). This is still valid under the current MSP (Royal Decree, 2019). Recreational angling remained permitted throughout the MPA.

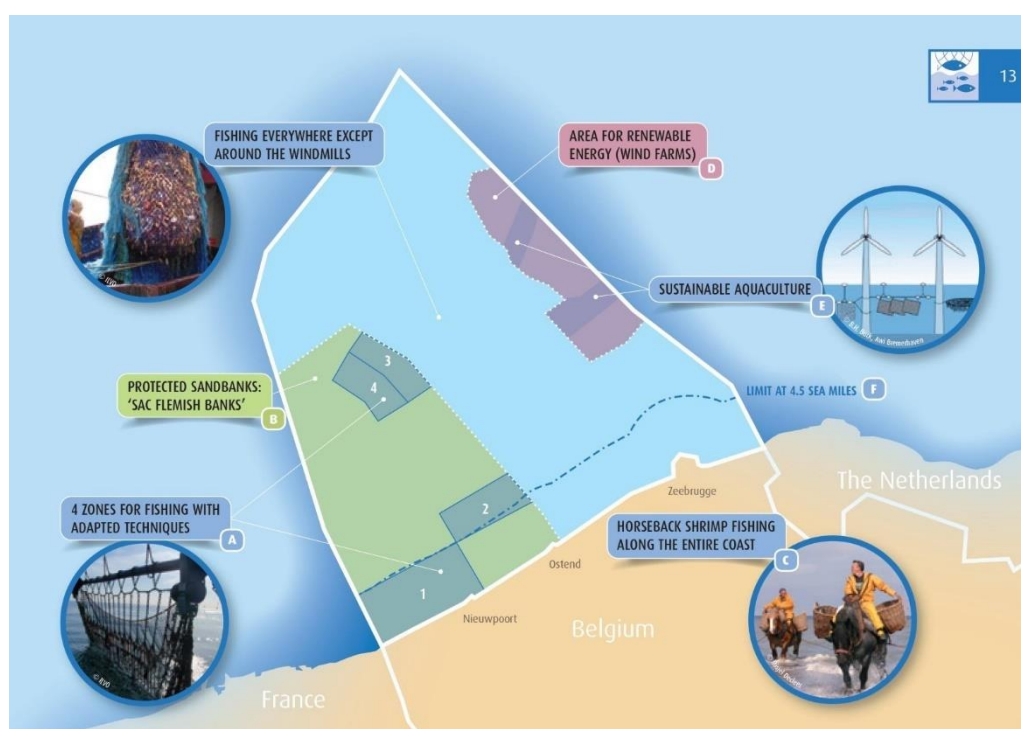


Figure 2. Restriction zones for sea-floor preservation within the Flemish Banks under MSP 2014–2020. The Flemish Banks MPA is shown in green (source: Van de Velde et al., 2015).

The measures for commercial fisheries inside these subzones were not converted into a European Union (EU) Delegated Act and were revoked by the European Parliament in 2018. The European Commission later withdrew the four subzones with fisheries measures (Figure 2) (Verlé et al., 2020). Consequently, the MSP 2020–2026 included three new areas with restrictions on activities that affect the seabed, such as trawl fishing (Figure 3) (Pecceu et al., 2021; Verhalle and Van de Velde, 2020). These were named 'Search Zone 1' (32 km²), 'Search Zone 2' (326 km²) and 'Search Zone 3' (354 km²). Search zones 2 and 3 are within the Flemish Banks MPA and will be considered further in the analyses of our work (Figure 3).

The process of formulating fishing restrictions in these new subzones is ongoing (Pecceu et al., 2021). Only a portion of the zones (± 325 km²) will have restrictions. In these areas no bottom-disturbing activities will be allowed, so use of mobile bottom-disturbing fishing gears will not be permitted (Table 1). These restrictions are in line with the requirements of the Belgium measures programme of the MSFD and the recent EU marine action plan (EC, 2023).



Figure 3. Search zones where measures for bottom-disturbing fishing can be imposed under MSP 2020-2026. The Flemish Banks MPA is shown in green (Source: Verhalle and Van de Velde, 2020).

Table 1. Overview of the different types of gear and whether they will be banned from the protected areas in the Flemish Banks (from Pecceu et al., 2021).

	Gear type code	Remark
Gear groups that should be banned in the closed area		
Beam trawl	TBB	
Shrimp trawl	TBS, TBC	
Pulse trawl	PUL	Under EU ban, forbidden
Otter trawl	OTB, OTM, OTT, PTB	
Seiners	SDN, SSC, PS	
Other gears	RB, HMD, MIS, PTM	
Gear groups that could be allowed in the closed area		
Passive fishing techniques	FPN, FPO, FYK, GN, GND, GNS, GTR, LHM, LHP	Circumstances in which these gears could be allowed need to be evaluated

	Gear type code	Remark
NB: TBB = beam trawls targeting demersal fish; TBS = beam trawls targeting shrimp; TBC = beam trawls targeting crustacea; PUL = pulse trawl targeting demersal fish; OTB = bottom otter trawls; OTM = midwater otter trawls; OTT = otter twin trawls; PTB = bottom pair trawl; SDN = Danish seines / anchored seine; SSC = Scottish seines / fly-shooting seine; PS = purse seine; DRB = boat dredge; HMD = mechanised dredges; MIS = miscellaneous gear; PTM = midwater pair trawls; FPN = stationary uncovered pound nets; FPO = pots and traps; FYK = fyke net; GN = gill nets (not specified); GND = drift netting; GNS = set gill nets; GTR = trammel nets; LHM = mechanised hand-lines and pole-lines; LHP = hand-operated hand-lines and pole-lines.		

2.4. Belgian institutional context

Belgian institutional complexity often creates problems of policy overlap, policy gaps and conflicts. Jurisdiction is divided over three levels of government: federal, regional and local (Cliquet et al., 2008). Jurisdiction over maritime affairs is divided between the federal state and the Flemish region, and within each level over several departments. Since there is no hierarchy between federal and regional governmental levels, each can adopt legislation and policy measures independently, within defined competencies (Pecceu et al., 2016). Fisheries belong to the competencies of the Flemish region, while the federal government is in charge of defining the overall rules and regulations for maritime activities in the BPNS. The latter ensures the coordination of all activities on the seabed, in the water column and on surface waters. However, Flanders has jurisdiction over fisheries and dredging activities (Cliquet et al., 2008; Pecceu et al., 2016).

2.5. History of the fishing industry and current status

For centuries, the Belgian coastal waters were an important area for fishing activities. The area offered fishing grounds for both demersal and pelagic fisheries applying both passive and active gears. The BPNS, including the Flemish Banks, provided a source of food, employment and income opportunities for local coastal communities (Lescrauwaet et al., 2010; Lescrauwaet et al., 2013b). An iconic example is the horseback shrimp fishing, a historical practice that used to exist along the North Sea coastline from France to the Netherlands. Currently it is limited to the community of Oostduinkerke (Acott et al., 2014) and is practised inside the Flemish Banks (Van de Velde et al., 2015).

Dutch and French fishers have a history of being active in the vicinity of the Belgian coast. A treaty in 1958 gave Dutch fishers unlimited access to fish all species in the 0–3 nautical mile (nm) zone. Their vessels may not exceed a GT of 70. Based on the Common Fisheries Policy (CFP), Dutch fishing vessels also have access to the 3–12 nm zone. An agreement in 1975 gave French fishers permission to fish herring and sprat in the Belgian territorial sea (Douvere and Maes, 2005). These agreements are still valid (Pecceu et al., 2014) and substantial Dutch activity persists in the BPNS and in the Flemish Banks (Pecceu et al., 2021).

Nieuwpoort, a harbour inside the Flemish Banks, has been a fishing port for about 850 years. In 1945, there were 99 vessels and 354 fishers. Since the 1950s, the importance of the fleet has steadily declined, as is the case for the whole Belgian fleet (Vandecasteele, 2014; Lescrauwaet et al., 2013a; Verlé et al., 2020). Over the past

decades, a number of factors (e.g. policy, market developments, business costs, changing fishing grounds) contributed to this overall decline, including various CFP programmes oriented to decommissioning. Investments resulted in vessels with higher average engine power and GT (Lescrauwaet et al., 2013a). However, the fishing vessels active in the BPNS and more specifically near the coast are still relatively small scale compared with the rest of the fleet. These vessels target mainly shrimp, sole and plaice and are harboured in Nieuwpoort and Ostend. They have long been operated by family businesses and make short trips. In 2017, they represented 18 % of the active Belgian fleet, but only accounted for 2 % of the total landed weight and 3 % of the value (Verlé et al., 2020).

Commercial fishers in Nieuwpoort operate almost exclusively in the BPNS and use towed gear, trammel nets, pots and fykes (Verlé et al., 2020). Shrimp usually represents a high proportion of the sold weight (Scherrens, 2022). The town promotes collaboration with a non-profit organisation that includes fish retailers and (retired) fishers and aims to keep a strong link with the fishing community (Acott et al., 2014). In addition, Dutch commercial sea-angling vessels regularly landed their catches in Nieuwpoort. It has been reported that a number of former Belgian recreational anglers have joined this Dutch commercial fleet and are part of Low Impact Fishery Southern North Sea (LIFSN) (van Winsen et al., 2016).

A study of fishing activity in the Flemish Banks between 2013 and 2019 showed that Belgian, Dutch and French vessels were active (Pecceu et al., 2021). Other Member States included in the study had negligible fishing activities. Fishing gears used were beam trawls, shrimp trawls, otter trawls, passive gear, seiners, and until August 2019, mainly pulse trawl (flatfish). Dutch vessels massively shifted from beam trawl to pulse trawl and this was visible in the data of the Flemish Banks in 2013 and 2014. Throughout Flemish Banks, the majority of activity came from Dutch pulse trawlers (Pecceu et al., 2021). For Belgian vessels, the area later designated as 'Search Zone 3' was important for shrimp trawls, beam trawls and passive gears. In the area later designated as 'Search Zone 2' Belgian vessels mainly used beam trawls and passive gears. French vessels using otter trawls and passive gears were also active in 'Search Zone 2'. It was estimated that the overall yearly potential loss for 'Search Zone 2' and 'Search Zone 3' was, respectively, 530 tonnes and EUR 2.42 million and 668 tonnes and EUR 2.45 million (Pecceu et al., 2021).

2.6. Recreational fishing

Nieuwpoort and Ostend also have an important marina for recreational vessels. Their activities occur mostly within the 3 nm zone. A study in 2016 identified approximately 100 recreational beam and otter trawlers. The majority of the Belgian recreational fishing fleet consists of anglers (about 87 %) (Verleye et al., 2022).

2.7. Fishery displacement

There are no fishery restrictions in force in Belgian waters, including in the Flemish Banks MPA. The process on implementing fishery management areas (restrictions) is ongoing. However, on several occasions, changes to the fishery sector have been mooted. Therefore, it is useful to investigate whether the possibility of measures being imposed in the future has already led to changes in fishing practices in the Flemish Banks MPA.

3. AIMS AND OBJECTIVES

The main objective of this report is to assess spatial reallocation of fishing activities in response to MPA implementation and during the process of establishing fishery measures in the Flemish Banks area over the period 2007–2022. The specific objective is to evaluate whether the management of fisheries in and around the Flemish Banks area has had an influence on fishing activity (effort, weight and value). The following questions will be investigated: (i) has fishing activity increased or decreased within the area? (ii) what changes have there been in fishing activity in specific areas (e.g. search zones) over the study period? (iii) has there been any change in the types of fishing activity? These questions are investigated in relation to the different periods of the MPA implementation process: (1) the designation of the MPA (2007–2012); (2) the first fishery measure process (2013–2018) and (3) the second fishery measure process (2019–2022).

4. METHODOLOGY

For this case study, possible changes in fisheries activity during the MPA designation process and fishery negotiations were investigated based on vessel monitoring system (VMS) data for the period 2007–2012 (before designation), the period 2013–2018 (first fishery measures process) and the period 2019–2022 (second fishery measure process). The first proposed fishery measures were revoked by the European Parliament in 2018.

A systematic review revealed only a few sources (see Annex 4 of main report for detailed methodology). The background review was mainly based on our own libraries, grey literature and expert knowledge. The research for this case study on the possible influence of the Flemish Banks MPA on fisheries is based on available data.

4.1. Data types and sources

Results from Pecceu et al. (2014) and Pecceu et al. (2021) were consulted and used in a first instance to conceptualise the overall approach to the case study. Additionally, results available for the period 2012–2021 based on the Belgian contribution to the data call (i.e. see Section 2 of the main report) were visualised. Gaps were identified in terms of the timeline coverage, the players involved and the measures for units of fishing effort. For example, effort can be expressed in terms of fishing pings, fishing hours or days, or in hours/days at sea.

This final version includes results of an ongoing study in the MPA (Verlé et al., 2023). Data between 2007 and 2022 were collected from six countries active in the BPNS. Data was available for Belgium, France and the Netherlands for the period 2007–2022 and for Germany, Denmark and the United Kingdom for 2009–2022.

Where possible, VMS and logbook data were processed in R using the *VMStools* package version 0.76 (Hintzen et al., 2012). However, because France uses a different formatting system, it was not possible to integrate data for France into the overall map of fishing effort. For each VMS ping, the activity of the vessel was defined based on its speed and the gear type used during the trip. In general, depending on the gear, fishing will occur within a certain speed interval. At lower speeds, the vessel is considered to be floating or in the harbour; at higher speeds, the vessel is considered to be steaming. Points close to a harbour are generally excluded.

4.2. Analysis

Data for the Flemish Banks and the BPNS were analysed separately in order to make a comparison over time. Particularly significant countries (the Netherlands, Belgium) and gears (beam trawl, shrimp trawl, seine, otter trawl and passive fishing) were also investigated separately. Gear groups were considered in line with the categories in Table 1. For the purpose of this study, we focused on the distribution of effort in term of fishing hours per period. We considered that the potential reallocation of fishing activities was best investigated via the location of the effort based on the estimated fishing pings.

Other gears (Table 1) and passive gears were not included separately, as the VMS methodology described above may not be accurate for defining fishing activity using such passive gears. It was also unclear how fishing activity was defined for the French data, and this country reported an important share of the overall passive gear activity. Furthermore, French 0–12 m vessels, mainly using passive gears, appeared from 2015 onwards, so were probably not using VMS systems before this. Nevertheless, the overall activity of passive gears remains limited in the BPNS.

5. RESULTS

5.1. General spatial and trend analyses

Since the designation of the Flemish Banks as a SAC in 2012, a shift towards more coastal fishing effort can be observed over the years. Figure 4 presents the overall effort between 2007 and 2022 and includes contributions of Belgian, Dutch, German, Danish and British vessels. On the north-east side of the BPNS, an area with no fishing appears over time (grey area, Figure 4); this coincides with the gradual developments within the wind-farm concession area. The first pylons were built in 2009–2010. As construction continued, a gradual disappearance of fishing activities within this concession area can be observed. The wind farm has been fully operational since 2020.

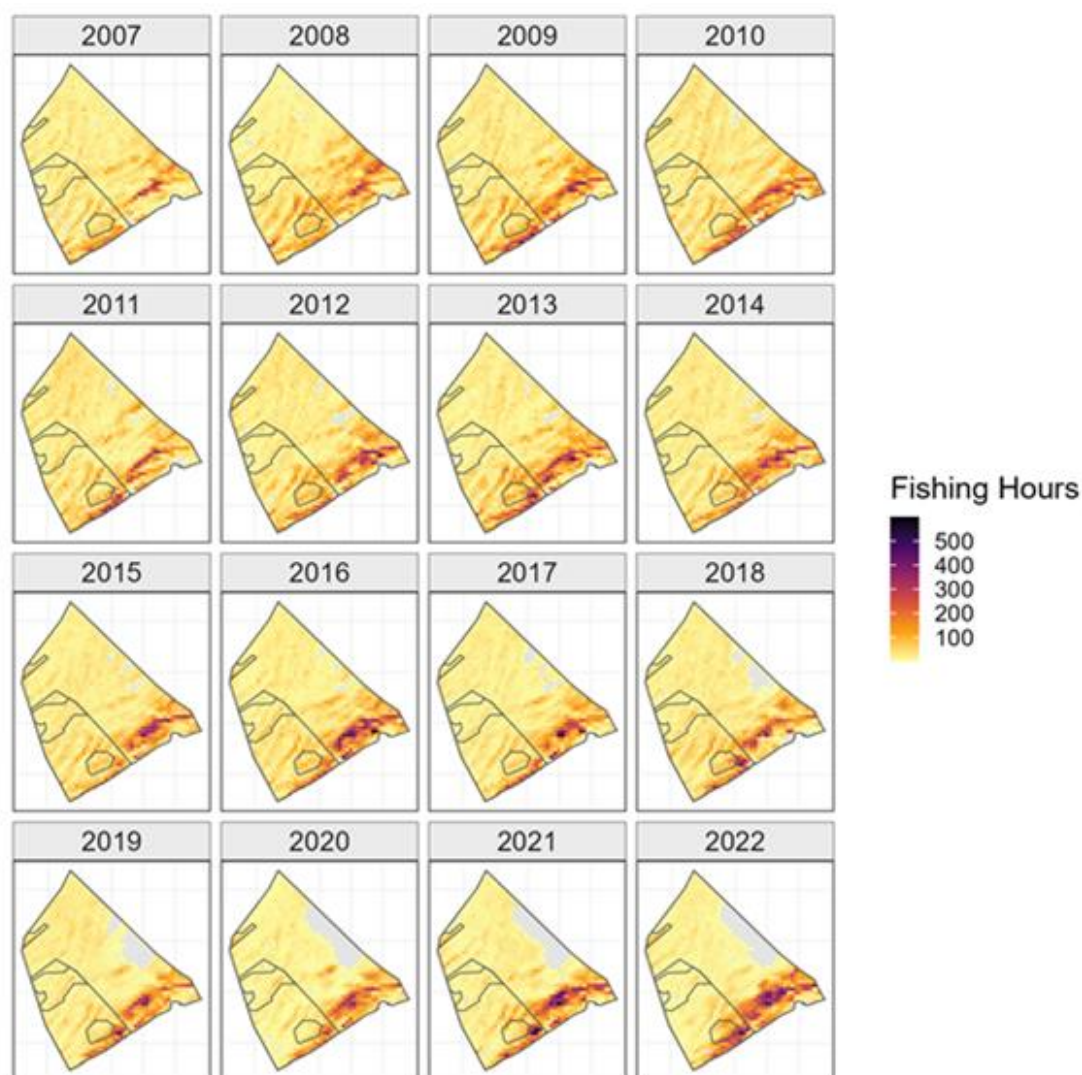


Figure 4. Total fishing effort in fishing hours in the BPNS between 2007 and 2022 (all gears combined). Data includes activities of the Belgian and Dutch fleets (2007–2022) and the German, Danish and British fleets (2009–2022).

NB: France could not be included, as its data could not be delivered in the required grid (because France uses a different grid). The information is presented on a 1.6 x 1.6km grid. The Flemish Banks polygon is also represented along with zones in which the imposition of fishing measures is proposed.

In Belgian waters, the Netherlands was the most important player, accounting for 66 % of fishing hours and ~80 % of landings by weight and value (2009–2022). Belgium was next, with close to 31 % of fishing hours and 17–18 % of landings. France was in third position, with about 1 % of the effort. Finally, the combined contribution of Denmark, Germany and the United Kingdom accounted for about 2 % of the fishing hours.

When considering total trends (Table 2, Figure 5), there seems to be an overall decline of the fishing effort and landed weight in the Flemish Banks as well as in the BPNS, at least for the period 2012 - 2020. The trend is more apparent for the landed weight than for the fishing effort and seems steeper for the BPNS than for the Flemish Banks. Since 2017, the landed weight is clearly lower for the same level of effort. On average, the Flemish Banks represent 33 % of the fishing effort in the BPNS.

Table 2. Fishing hours inside the Flemish Banks, in the BPNS and percent in the Flemish Banks between 2007 and 2022, all gears combined.

Year	Flemish Banks	BPNS	% Flemish Banks
2007	19 450	54 309	36
2008	22 837	64 894	35
2009	26 185	72 030	36
2010	22 992	70 100	33
2011	23 621	65 280	36
2012	22 268	69 860	32
2013	27 125	72 266	38
2014	21 032	68 440	31
2015	20 276	64 425	31
2016	22 478	73 880	30
2017	18 174	59 770	30
2018	24 444	62 417	39
2019	18 663	57 238	33
2020	14 477	50 847	28
2021	21 839	66 955	33
2022	23 060	71 093	32

NB: Includes data from Flemish Belgian, French and Dutch vessels (2007–2022) and German, Danish and British vessels (2009–2022).

No decline could be observed when considering the share of fishing effort in the Flemish Banks relative to the fishing effort in the BPNS (Figure 6, Table 2). A simple linear regression model was fitted through these percentages both for the entire period and the period after designation of the MPA (Table 3). Results showed that there is no significant negative relationship between years and relative fishing effort in the Flemish Banks (p -value > 0.1). Similarly, there was no significant decline in relative value of landings in the Flemish Banks (p -value > 0.5).

In terms of landed weight, there seems to be a slight decline since the designation, with a decrease of 0.8 % per year ($p = 0.013$). This decreasing trend is not significant when considering the entire period ($p = 0.141$). As there is some variation from year to year, we can see that a simple linear trend does not represent a convincing fit in most of our cases, as R^2 values are low (Table 3). For the trend of landed weight between 2012 and 2022 the highest R value was observed ($R^2 = 0.5$).



Figure 5. Fishing effort and landings in Flemish Banks and BPNS, 2007–2022. Top: total fishing effort in fishing hours; bottom: total landed weight in the BPNS and the Flemish Banks (all gears combined).

NB: Includes data from Belgian, French and Dutch vessels (2007–2022) and German, Danish and British vessels (2009–2022). The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA in 2012. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

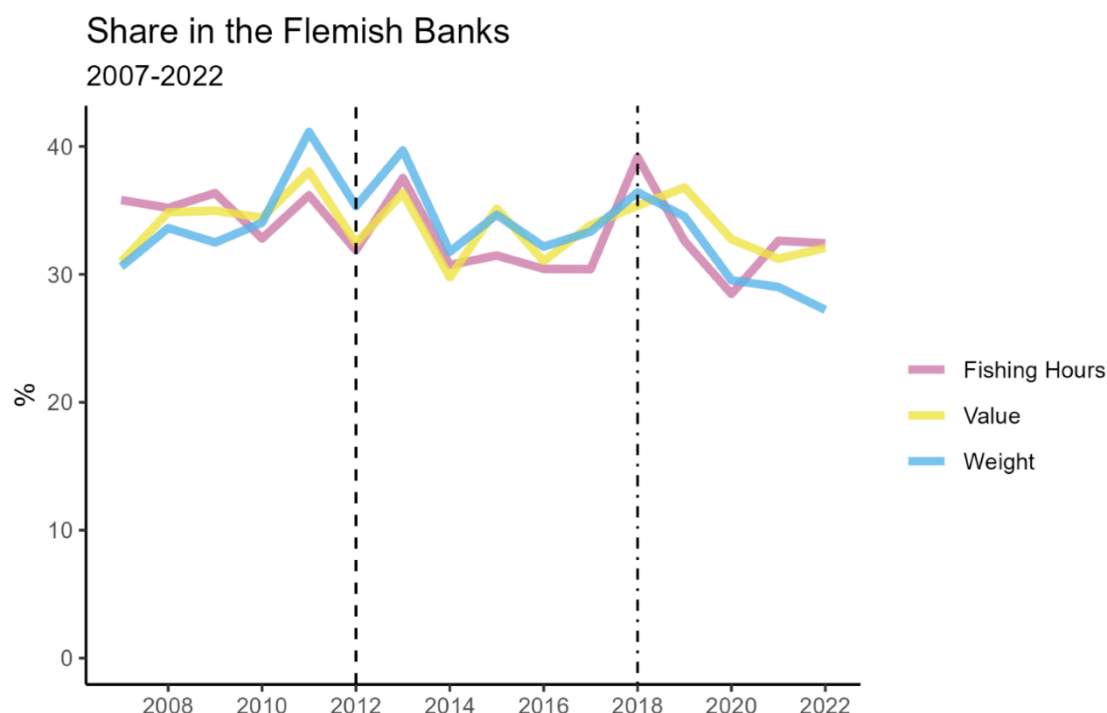


Figure 6. Percent of effort, weight and value of landings in the Flemish Banks relative to the Belgian part of the North Sea (all gears combined).

NB: Data from Belgian, French and Dutch vessels (2007–2022) and German, Danish and British vessels (2009–2022). The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

Table 3. Estimated coefficient (B1), probability of finding the given t statistic if the null hypothesis of no relationship were true ($\Pr(>|t|)$) and goodness of fit (R^2) for a simple linear regression on the share of fishing hours, weight and landed value (Y) in the Flemish Banks relative to the BPNS for the entire time period (2007–2022) and since its designation as an MPA (2012–2022)

Year	Y	B1	$\Pr(> t)$	R^2
2007–2022	%Effort	-0.2641	0.104	0.1776
2007–2022	% Weight	-0.2959	0.141	0.1484
2007–2022	% Value	-0.0831	0.538	0.0277
2012–2022	% Effort	-0.1148	0.723	0.0146
2012–2022	% Weight	-0.7811	0.0130	0.5142
2012–2022	% Value	-0.0534	0.823	0.0058

NB: Data from Belgian, French and Dutch vessels (2007–2022) and German, Danish and British vessels (2009–2022). R^2 is the coefficient of determination and measures how well the model predicts the outcome (goodness of fit of the model). When the R^2 is low, many points are far from the fitted line.

Figure 7 considers the fishing effort over three periods: before designation of the MPA (6 years), first fishery measure process (6 years) and the second fishery measure process (4 years). There is no significant difference in fishing effort between these periods. The contribution of 2020 (Covid year) affects the lower median of the period 2019–2022 compared to the other periods. When data for 2020 is removed, the medians equalise to the same level as the previous periods, both in Flemish Banks and the BPNS.

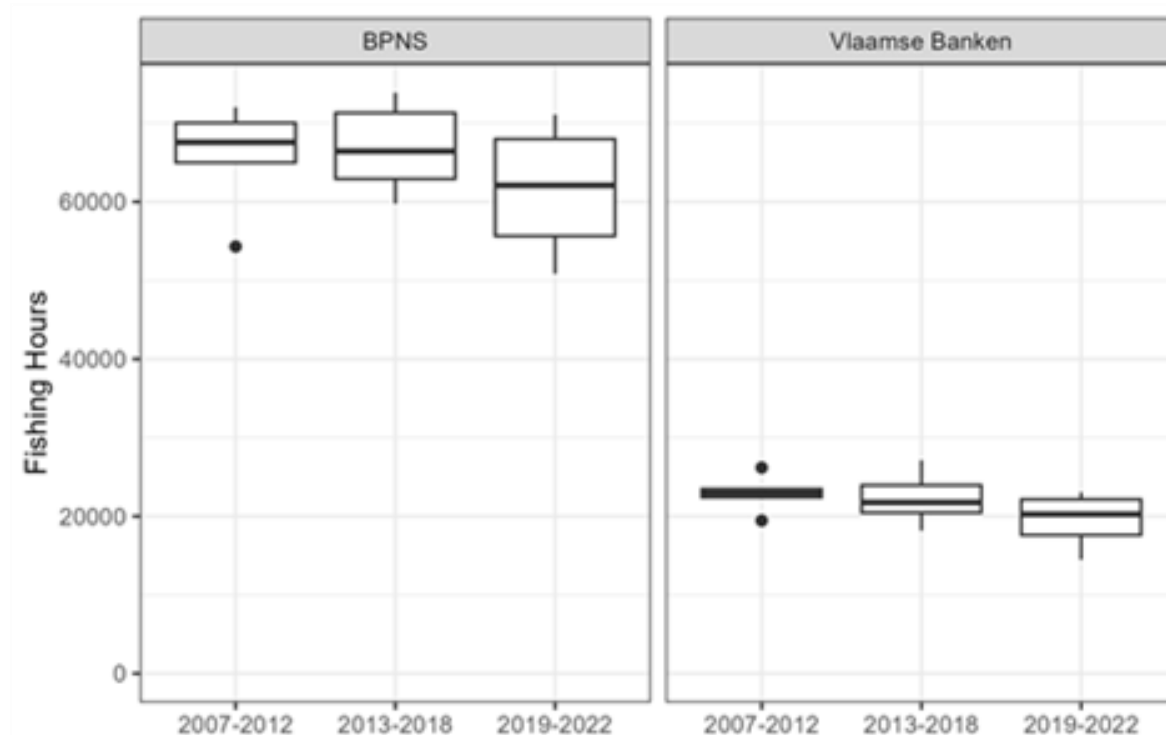


Figure 7. Boxplot (mean; 25/75 percentile) of fishing hours in the BPNS and in Flemish Banks for three time periods: before designation of the MPA (2007–2012), first fishery measure process (2013–2018) and second fishery measure process (2019–2022).

NB: Data from Belgian, French and Dutch vessels (2007–2022) and German, Danish and British vessels (2009–2022).

5.2. Activity per country

There was no change in the share of effort and landings in the Flemish Banks for Belgium and the Netherlands in the period 2007–2012 (Figure 8). A discrete decline may be observed for the Netherlands when considering only the period 2019–2022. The French data show an increasing and then decreasing trend over the time period. In absolute terms, French fishing effort (only 2 % of fishing effort in BPNS) was rather volatile both in the BPNS and in the Flemish Banks and it was not possible to see a convincing trend.

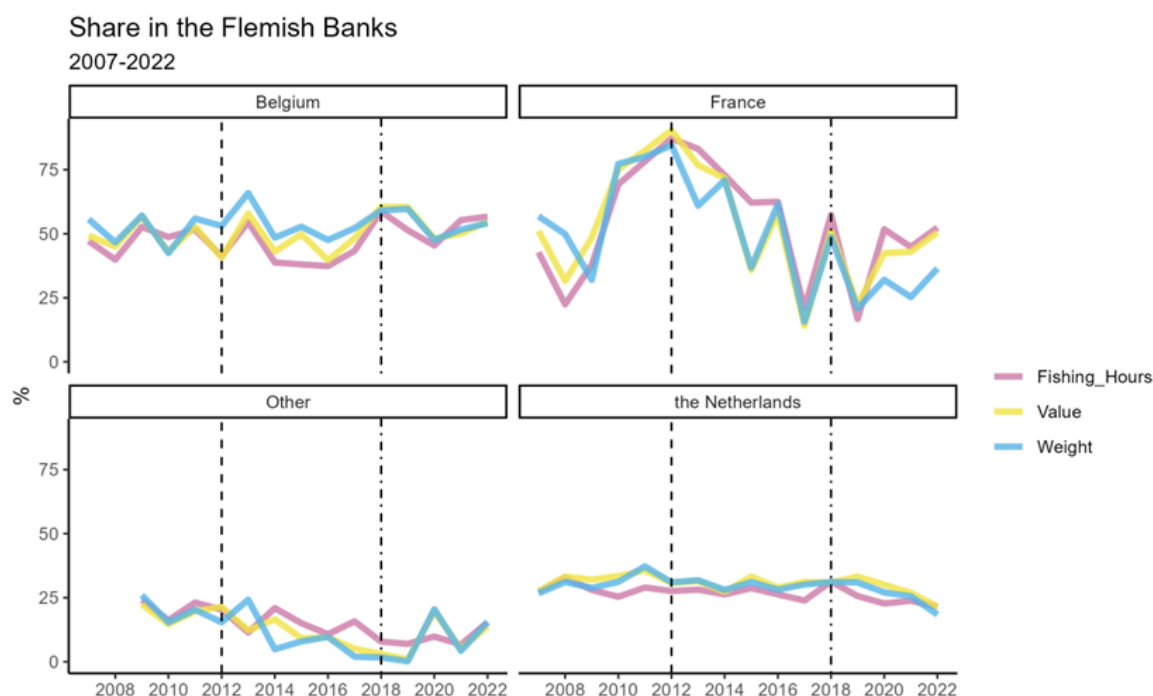


Figure 8. Percent of effort, weight and value of landings in the Flemish Banks relative to the BPNS per country between 2007 and 2022 (all gears combined).

NB: 'Other' includes Germany, Denmark and the UK. The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

5.3. Spatial and trend analyses of certain important gears

Figure 9 gives an overview of the total fishing effort per gear group in the Flemish Banks and the BPNS. Shrimp trawl, beam trawl and pulse trawl were the most important gears.

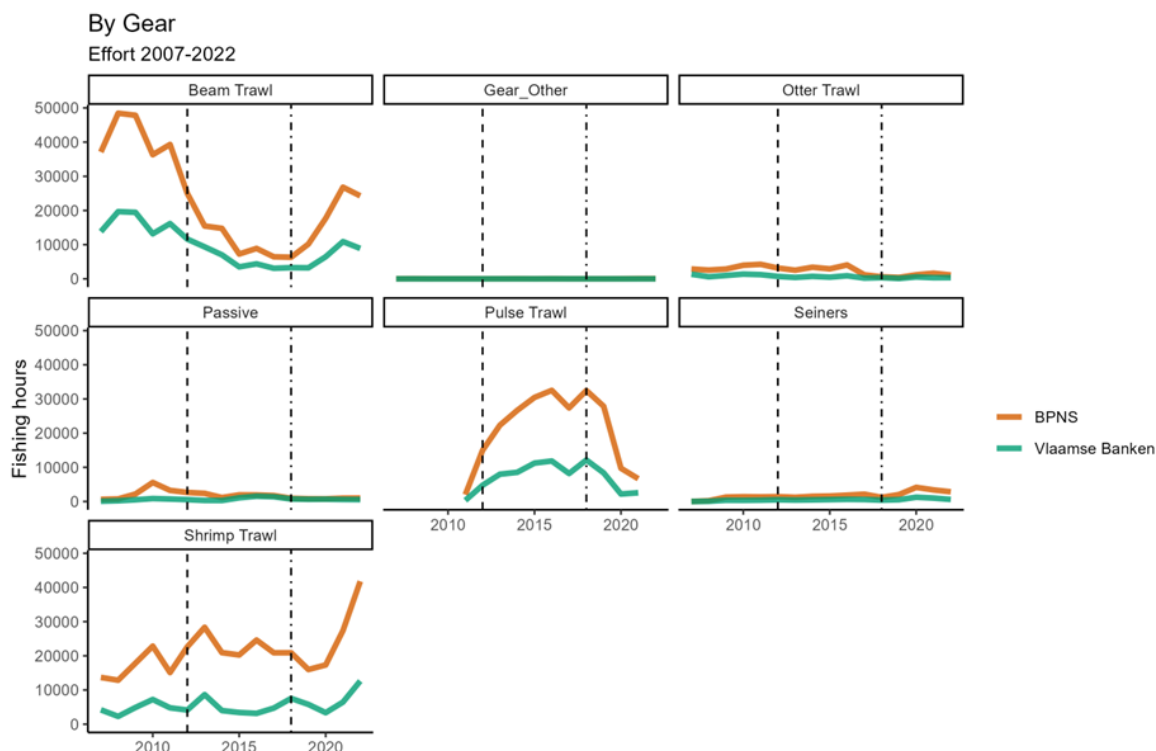


Figure 9. Total fishing effort in fishing hours per gear group, in the BPNS and the Flemish Banks.

NB: Data from Belgian, French and Dutch vessels (2007–2022) and German, Danish and British vessels (2009–2022). The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

5.3.1. Shrimp trawl

Trawling for brown shrimp (*Crangon crangon*) is an important activity for Belgian and to a lesser extent Dutch vessels. Fishing takes place along the coastline, including in the Flemish Banks (Figure 9). Within the category of trawling gears, shrimp trawls represent relatively small vessels when compared to trawling vessels active further offshore. They generally consist of vessels < 70 GT and ≤ 221 kW that are allowed to operate within the 3 nm zone. No trend for shrimp trawl was observed between activities inside and outside the Flemish Banks since the designation of the Flemish Banks as a SAC in 2012 (Figure 10).

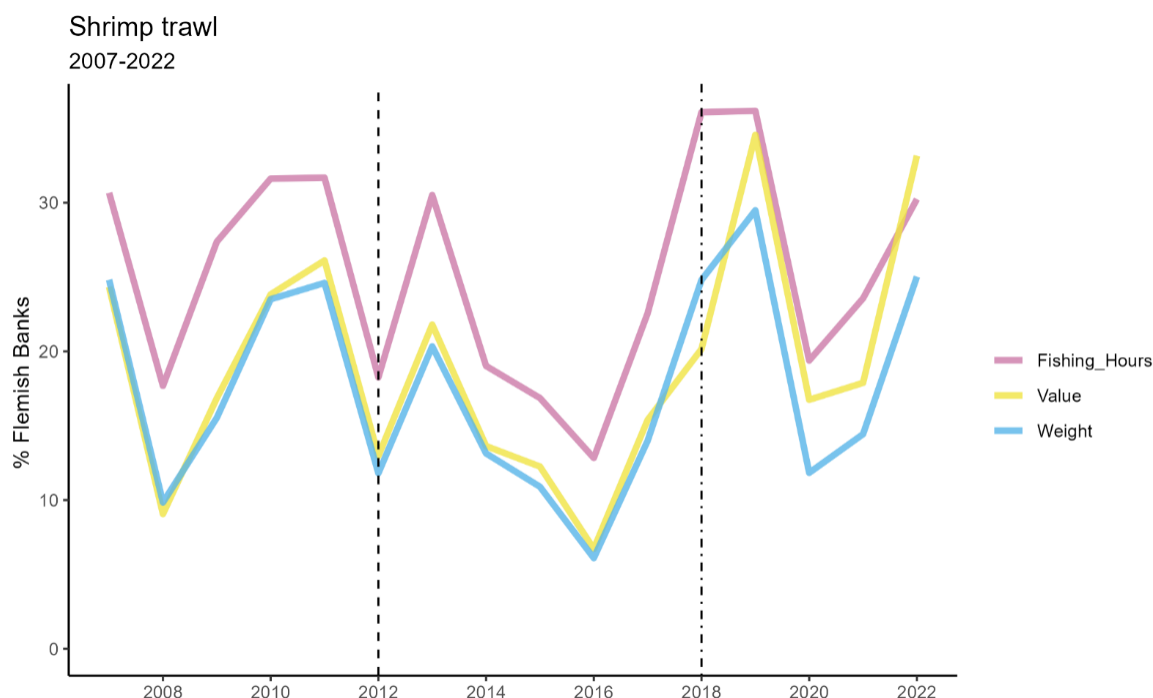


Figure 10. Percent of effort, weight and value of landings for shrimp trawl in the Flemish Banks relative to the BPNS between 2007 and 2022.

NB: The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

5.3.2. Beam trawl and pulse trawl

Beam trawling for flatfish is an important activity for Belgian and Dutch vessels. Smaller vessels are active closer to the coast and may alternate beam trawling and shrimp trawling depending on the season. Further offshore, larger vessels predominate. An important target species is sole (*Solea solea*), which could also be caught successfully with a pulse trawl. In the time series, the shift from beam trawling to pulse trawling made by many Dutch vessels is visible (Figure 9). This was not the case for the smaller coastal vessels that operate in the 3 nm zone. There was no pulse trawling for flatfish close to the coast (2011–2021). Pulse trawling stopped in 2021. Despite overall effort displacements for these gears over time, when combining them, no difference could be observed between relative effort of beam and pulse trawl in the Flemish Banks and the BPNS (Figure 11).

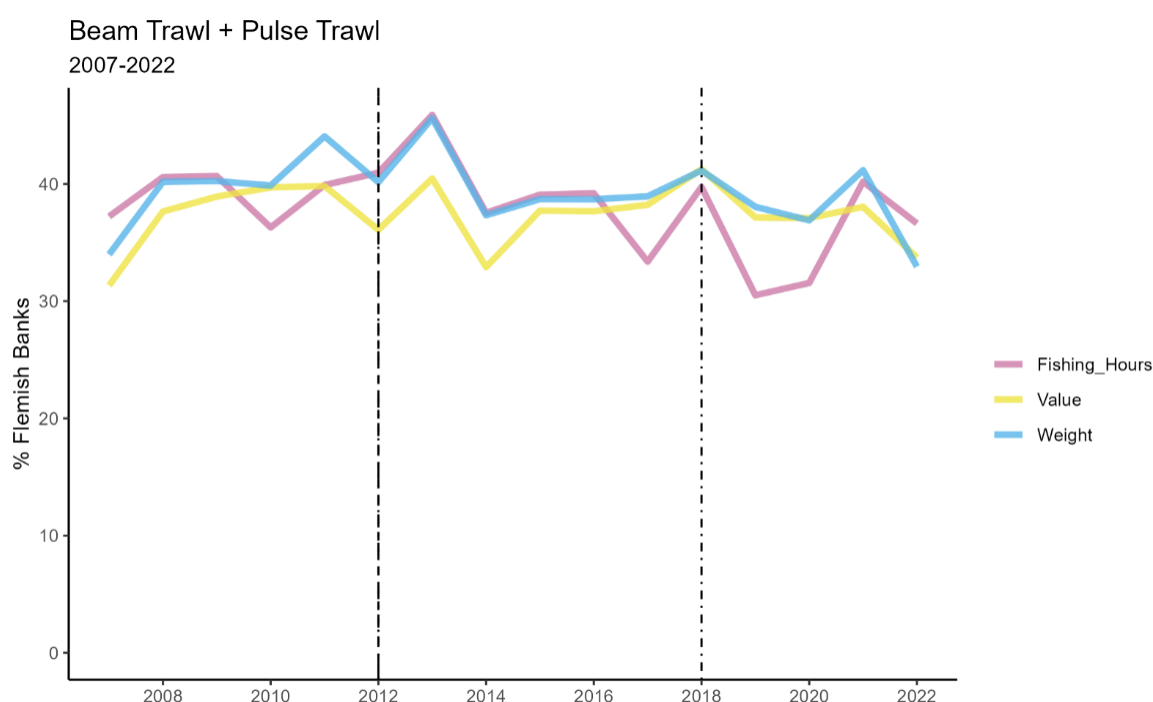


Figure 11. Percent of effort, weight and value of landings for beam and pulse trawl in the Flemish Banks relative to the BPNS between 2007 and 2022.

NB. The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2028, the year the first proposed fishing measures were revoked by the EU Parliament.

5.3.3. Otter Trawl

No trend for otter trawl could be observed in the relative effort in the Flemish Banks compared to the BPNS (Figure 12). The activity of otter trawl within and outside the Flemish Banks is variable but does not seem to be influenced by the designation or management actions planned.



Figure 12. Percent of effort, weight and value of landings for otter trawl in the Flemish Banks relative to the BPNS between 2007 and 2022.

NB: The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

5.3.4. Seine fishing

Predominantly Scottish seiners were introduced in the BPNS over time. However, their relative effort within the Flemish Banks compared to the BPNS seems relatively stable over time with no trend (Figure 13).



Figure 13. Percent of effort, weight and value of landings for seine fishing in the Flemish Banks relative to the BPNS between 2007 and 2022.

N.B: The vertical dashed line indicates 2012, the year of designation of the Flemish Banks as an MPA. The vertical dot-dashed line indicates 2018, the year the first proposed fishing measures were revoked by the EU Parliament.

6. DISCUSSION

An **overall declining trend in fishing activity and landings, especially in the period 2017–2020, could be observed in the BPNS**. There are a number of possible contributing factors:

In 2019, the EU decided to impose a definitive ban on pulse trawl fishing. This was implemented in the ensuing years. Pulse trawls were then (partly) replaced by beam trawls. The pulse trawl had a higher catch efficiency for target species sole compared to other gears (Rijnsdorp et al., 2020, van Overzee et al., 2023). Post-2017, the overall landings are clearly lower for about the same effort, possibly because fewer fish are present or catch efficiency changed (because of the switch back to 'normal' beam trawl, after the pulse ban) within the BPNS.

The Covid-19 pandemic influenced the trends in 2020. We see a dip for this year in the trend figures. Additionally, the median of fishing effort when considering 2019–2022 is influenced by events in 2020. During the pandemic, the demand for fresh seafood products was much lower which affected market prices initially. Consequently, different EU fishing fleets adopted alternative strategies to mitigate potential impacts (Carpenter

et al., 2023). For example, the Belgian fleet organised a vessel-rotation system in order to limit landings and maintain high prices (STECF, 2020). During this period, effort and landings were indeed lower. However, during the first lockdown, the impact on economic performance was not as great as one could have expected (Carpenter et al., 2023; STECF 2020).

In 2022 a decommissioning scheme was approved for vessels in the Dutch fleet targeting demersal fish in the greater North Sea (Hamon et al., 2023). Eligible vessels included beam trawls, otter trawls and flyshoot. Vessels targeting shrimp were not eligible. Of the eligible vessels, 50 % (71 out of 139) were registered to be scrapped on 1 April 2023, representing 13 % of the total Dutch fleet (Hamon et al., 2023). This is in effect a consequence of Brexit as the UK has claimed a larger share of the total allowable catch (TAC) than was agreed at the time of its membership. A permanent cessation of a part of the fleet mitigates the risk of shortage of fishing opportunities (van Oostenbrugge et al., 2022).

During the time period new/other players (for example, wind farms) received a designated area in which to conduct their activities in the BPNS and became competitors for space. Fishing grounds are declining as result of other uses of space in the BPNS and the neighbouring vicinity.

We also observed a **decline in the landings from the Flemish Banks overall** and a dip in fishing effort in 2020. However, in relative terms, the decline in the Flemish Banks is less apparent. **The Flemish Banks accounted for 33 % of the fishing effort in the BPNS** between 2007 and 2022, while the area also represents 30 % of the area of the BPNS (Pecceu et al., 2016). The Flemish Banks are not the only fishing area in the BPNS but do hold some important fishing grounds for the fishing industry. The share of fishing effort in the Flemish Banks remained relatively stable over the time period, therefore its relative importance did not diminish, even after the announcement of its designation as a SAC in 2012 or during the different fishing measures processes. On the other hand, no relative increases in fishing effort were observed, indicating that there are no sudden claims on the area either. Similarly, **neither the designation of the Flemish Banks, nor the proposed management actions have changed the combined fishing patterns of pulse and beam trawls, shrimp trawls, otter trawls and seining gear. Therefore, it seems that announcing possible measures and closures is not sufficient to lead to behavioural changes.**

We hypothesized that the Belgian fishing industry might be quicker to respond to potential changes in national legislation than other countries. These other countries might be less inclined to adapt their fishing grounds prematurely as they may be less involved in the discussion or less worried about finding alternatives. However, this does not seem to be the case: we saw no different trends for Belgian and Dutch data when considering the relative shares of the Flemish Banks compared to the BPNS. There was a decline for other countries, but it seems unlikely that this is linked to the Natura 2000 designation.

The French data showed a relative increase followed by a decline, but a lot of variation is seen from year to year. It was also difficult to make out a trend in absolute terms. It is likely that not all relevant vessels had VMS systems installed for the entire period considered. Implementing VMS systems in the EU happened in several phases. It first became mandatory for vessels ≥ 24 m (2000–2004), then for vessels ≥ 15 m (2005–2011) and finally for vessels ≥ 12 m (2012 onwards) (European Commission, 2003; European Commission 2009; Lee et al., 2010; European Commission, 2011). This could distort the image of French fishing activity in the Flemish Banks. Vessels < 15 m may

also be generally underrepresented in the analysis. Additionally, vessels < 12 m were not obliged to have VMS systems on board. In the data, French 0–12 m vessels appeared from 2015 onwards. They were mostly active in the Flemish Banks and used mostly passive gears.

The decreasing trend in the relative landed weight from the Flemish Banks since its designation as an MPA, is not convincing and caution is needed when drawing conclusions. Though significant at $p < 0.05$, the goodness of fit (R^2) is only 0.5 for weight, meaning that the trend line only partially explains the outcome. Possibly, with the declining contribution of the Dutch fleet targeting flatfish, relatively more of the overall weight was contributed by the shrimp fisheries. Shrimp is a high-value species with a lower weight than flatfish. So, for the same amount of effort, a shrimp trawl will land less weight. The shrimp trawls are active inside the Flemish Banks, but not exclusively (between 13 and 36 % of fishing effort). So, if the share to the landed weight from shrimp trawls becomes more important, this could entail a shift in the relative weight from the Flemish Banks compared to that from the BPNS.

The study indicated that the Flemish banks MPA Natura 2000 area is an important fishing area, so the implementation of the management areas (planned to be active by 2025) might have consequences for future fishery activities within the BPNS. For future work, it can be worthwhile to test possible fishery displacement effects in the BPNS based on fishery displacement models (e.g. Bastardie et al., 2014).

7. CONCLUSIONS

Since the designation of the Flemish Banks as a SAC in 2012, **a small decline, but no relative change in total fishing effort has been observed**. The analysis included data from six countries active in the BPNS between 2007 and 2022. The Netherlands and Belgium were the most important players. The relative importance of the Flemish Banks compared to the BPNS was investigated overall, as well as for certain important countries and gears.

Currently, there are no fishing restrictions in the Flemish Banks for commercial fisheries, despite previous efforts to implement management measures to preserve sea-floor integrity. The first proposed measures focused on defined subzones of the Flemish Banks but were revoked by the European Parliament in 2018. Consequently, new subzones were defined and the process of formulating fishing restrictions in these zones is ongoing.

The share of effort and landings in the Flemish Banks MPA compared to the BPNS was relatively stable overall and particularly for the Netherlands and Belgium, over the entire time period. Similar patterns could be observed when considering important gears. It appears that **fishers did not adjust their behaviour following the news of the designation of the MPA nor do they do so when fishing measures are pending**. The area holds important fishing grounds, therefore announcing possible measures and closures is not sufficient to lead to behavioural changes. Possible alternative fishing grounds are limited and declining because of other uses of space in the vicinity (overall trend in the BPNS is declining) – the BPNS is intensely and increasingly used for other activities such as a zone for renewable energy and area for sustainable aquaculture, and sand extraction areas. On the other hand, no increases in fishing effort were observed in the Flemish Banks relative to the BPNS, indicating that there are no claims on the area either.

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Case Study Report

The Nida-Perkone Marine Protected Area Latvia – Baltic Sea



Mapping of marine protected areas and their associated fishing activities

Ivars Putnis and Loreta Rozenfelde

***Institute of Food Safety, Animal Health and Environment
Latvia***

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LIST OF ABBREVIATIONS

Term	Description
BEF	Baltic Environmental Forum
BIOR	Institute of Food Safety, Animal Health and Environment
EEZ	Exclusive Economic Zone
EU	European Union
GES	Good Environmental Status
Ha	Hectare(s)
ICES	International Council for the Exploration of the Sea
LIAE	Latvian Institute of Aquatic Ecology
LIFE REEF	Project LIFE19 NAT/LV/000973 REEF 'Research of marine protected habitats in EEZ and determination of the necessary conservation status in Latvia'
MPA	Marine Protected Area
NCA	Nature Conservation Agency
NCP	Nature Conservation Plan
USSR	Union of Soviet Socialist Republics

1. EXECUTIVE SUMMARY

The Nida-Perkone Marine Protected Area (MPA) is a Natura 2000 site located in the south-western territorial sea of Latvia on the coastline of Rucava and Nīca parish. It covers an area of 367 km² and protects European Union (EU) essential habitats such as reefs. The long-term goal for the MPA is to ensure a favourable protection status for reef habitats and protected bird species, while maintaining their natural distribution and environmental functions and promoting the area's sustainable development by balancing nature conservation and socioeconomic interests.

Over the last decade, the biggest challenge for reef conservation has been non-indigenous species such as the round goby (*Neogobius melanostomus*). This fish species is a potential threat to habitat-forming species such as blue mussels. On the other hand, the round goby population density has potentially modified fishery patterns and socioeconomic conditions in coastal areas. In that context, the main aim of this case study is to investigate fishing activities in the Nida-Perkone MPA and analyse the possible changes in the fishery and its spatial dynamics after the round goby invasion.

To this end, Latvian fishery logbook data have been prepared to facilitate spatial and temporal fisheries analysis for the Nida-Perkone MPA. Coastal commercial fishery data covered the period 1995–2022, offshore fishery data covered the period 2004–2022 and coastal self-consumption fishery data covered the period 2014–2022.

Findings show that the ichthyofauna of the Nida-Perkone MPA has, in the short term, witnessed the ascendancy of the round goby as the most dominant fish species. Market opportunities and national fisheries policy promoted the rapid growth of a specialised goby fishery in the Nida-Perkone MPA. There is substantial pressure from the fishing sector to develop a round goby fishery based on the non-indigenous status of this species. On the one hand, round goby extraction could improve the ecosystem's health. However, there is a potential conflict between exploiting the novel resource and mitigating the bycatch risk for marine mammals and birds, which are also one of the focus organisms associated to conservation objectives of the Nida-Perkone MPA.

The scientific advice for national fishery management in Latvian coastal waters to reduce seabird and marine mammal bycatch and limit the impact of invasive species is still in progress and will be concluded by 2025. Although the EU Biodiversity Strategy for 2030 states that fisheries-management measures must be established in all MPAs, the intention to place 10 % of waters under a strict protection regime, could restrict the application of fishery-related measures necessary for controlling the round goby population in the MPA.

2. BACKGROUND

The Nida-Perkone Marine Protected Area (MPA) is a Natura 2000 site (code LV0900100) which was designated in 2010 (Regulations of the Cabinet of Ministers No. 17 of January 5, 2010). The MPA is on the Latvian coast of the Baltic Sea at the coastline of Rucava and Nīca parish (Figure 1 and Figure 2) and covers a surface area of 367 km² (36,703 hectares).

The MPA protects one habitat type and 20 species of the Habitats and Birds directives. A qualifying feature is the presence of reefs (Habitat type 1170). The MPA hosts numerous bird species: *Mergus merganser*, *Larus minutus*, *Gavia stellata*, *Gavia arctica*, *Cygnus olor*, *Anas platyrhynchos*, *Aythya marila*, *Aythya fuligula*, *Clangula hyemalis*, *Bucephala clangula*, *Mergus serrator*, *Haliaeetus albicilla*, *Larus ridibundus*, *Larus canus*, *Larus*

argentatus, *Larus marinus*, *Melanitta fusca* and *Melanitta nigra*. The MPA shelters internationally significant concentrations (more than 1 % of the flyway population) of goosander (*Mergus merganser*) and little gull (*Larus minutus*).

Spawning and feeding sites for coastal fish communities are present. The stony reef areas support diverse biological communities, representative of hard substratum with many associated species, including epibenthos. Red algae (*Furcellaria lumbricalis*), bivalve (*Mytilus trossulus*) and cirriped (*Balanus improvises*) are dominant species in the habitat, and brown algae (*Fucus vesiculosus*) are also abundant. Bivalves are an essential food source for some fish species and, in shallower areas, support the migrating waterbird populations by being a feeding ground (Baltic Environmental Forum, 2009).

Specific conservation objectives for the 2009-2018 period were adopted in 2011. An updated Nature Conservation Plan (NCP) is being developed nationally and will cover all Latvian MPAs. The Nature Conservation Agency (NCA) coordinates the work, and the new NCP will be adopted in 2025.

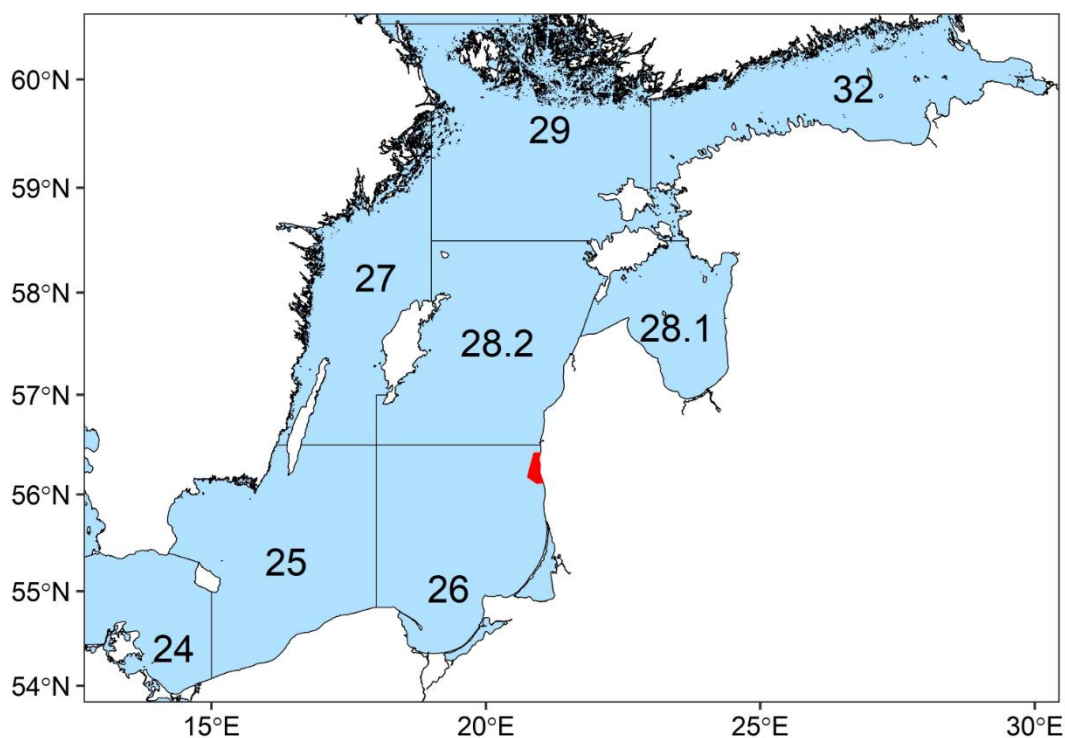


Figure 1. The location of the Nida-Perkone MPA in the Baltic Sea (in red).

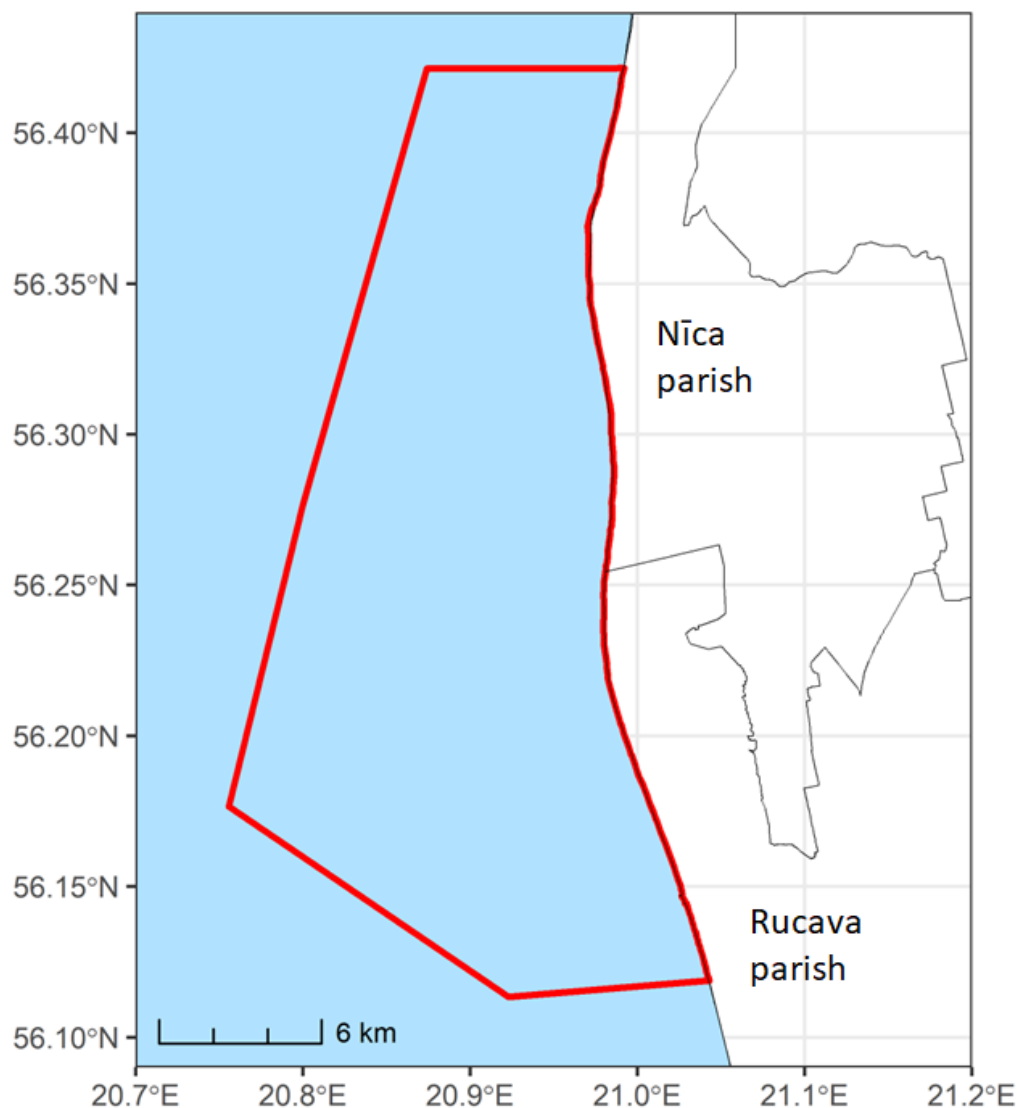


Figure 2. The borders of the Nida-Perkone MPA.

2.1. The MPA designation process at the national level

The NCP for the Nida-Perkone MPA was developed in 2008–2009. It was financed by the LIFE-Nature Programme of the European Union (EU) project 'Marine Protected Areas in the Eastern Baltic Sea' (LIFE05NAT/LV/000100).

The development of the plan was chaired by the public organisation Baltic Environment Forum (BEF) in cooperation with project partners: the Latvian Institute of Aquatic Ecology (LIAE), Latvian Ornithology Society, Latvian Fish Resources Agency and Maritime and Internal Waters Administration of the State Environmental Service. The NCP was designed simultaneously with the designation of the Nida-Perkone MPA. The proposal for establishing the Nida-Perkone MPA was submitted in December 2008 by the BEF and approved on 5 January 2010 by the Cabinet of Ministers of the Republic of Latvia.

A site-risk analysis has been carried out within the framework of the plan, taking into account the main impacts: coastal hydro-technical structures, bycatch of birds and mammals, direct disturbances caused by human activity, dumping of soil removed from the construction and deepening works in the sea, pollution, hazards from invasive species,

industrial fishing, oil product releases, mineral mining and the construction of potential offshore wind parks. The analysis concluded that there were no significant impacts on habitats and species of the Nida-Perkone MPA at the time of the study. However, at the same time, it was considered that such effects could appear in future as the use of marine space will likely increase for different purposes.

The long-term aim proposed for the site's preservation is to ensure a favourable protection status for protected bird species and reef habitats while maintaining their natural distribution and environmental functions and promoting the area's sustainable development by balancing nature conservation and socioeconomic interests.

To contribute to the preservation of marine biodiversity and to promote sustainable development of the MPA, the NCP set eight short-term objectives to be implemented within a ten-year cycle:

1. Ensure the conservation of habitats of EU importance – reef area of 22,268 ha – and the preservation of their functions at the current level (time of the development of the conservation plan (2009) as a baseline);
2. Ensure the proportion of bird populations of EU importance for conservation at the current level or increase for *Mergus merganser* and *Larus minutus*;
3. Ensure the efficient development and implementation of the monitoring and reporting system;
4. Ensure permanent administration and supervision of the MPA;
5. Promote sustainable tourism development in the MPA and ensure infrastructure aligns with MPA management objectives;
6. Preserve cultural and historical landmarks of the territory;
7. Promote public education on the natural and cultural value of the MPA;
8. Prepare/improve normative acts to help preserve the biological value of the MPA.

Within the Nida-Perkone MPA, it is not required or is practically impossible to perform habitats and species-oriented direct management measures; therefore, identified management activities were focused mainly on administration, control, research, monitoring, development of a favourable legal base and informing the public. Functional zoning was also developed, which provides three zones: a nature conservation zone, a neutral zone and a nature park zone (Figure 3). The justification for establishing each zone and identifying restricted actions was incorporated into the Cabinet Regulation for the Nida-Perkone MPA individual protection and management measures (No 652, adopted on 23/08/2011).

In the **nature conservation zone**, activities that cause mechanical damage to the specially protected habitats in the sea (reefs) are prohibited. Such activities include installing wind farms and extracting minerals, creating new dumping sites, and industrially removing algae and mussels. The **neutral zone** has been designed to ensure the sustainable economic activity of coastal settlements and the development of tourism infrastructure. In the **nature park zone**, environmental impact assessments are required before the creation of new dumping sites. The use of lead ammunition in bird hunting is prohibited throughout the MPA.

The NCP for the Nida-Perkone MPA has been developed in accordance with the Cabinet of Ministers No. 686 regulations, 'Regulations on the Content and Procedure for the Development of a Nature Management Plan for a Specially Protected Nature Area', adopted on 9 October 2007. The plan was launched by an informative meeting on 27 February 2008. During the plan's development, four sessions of the monitoring group were

organised: on 26 May and 30 October 2008, and on 13 March, 13 and 22 May 2009. A public consultation on the plan took place on 16 June 2009. Following that, the plan was sent to the municipal councils of Rucava and Nica for evaluation. The last meeting of the monitoring group was held on 21 August 2009, when it was declared that the development of the MPA NCP was complete (Baltic Environmental Forum, 2009).

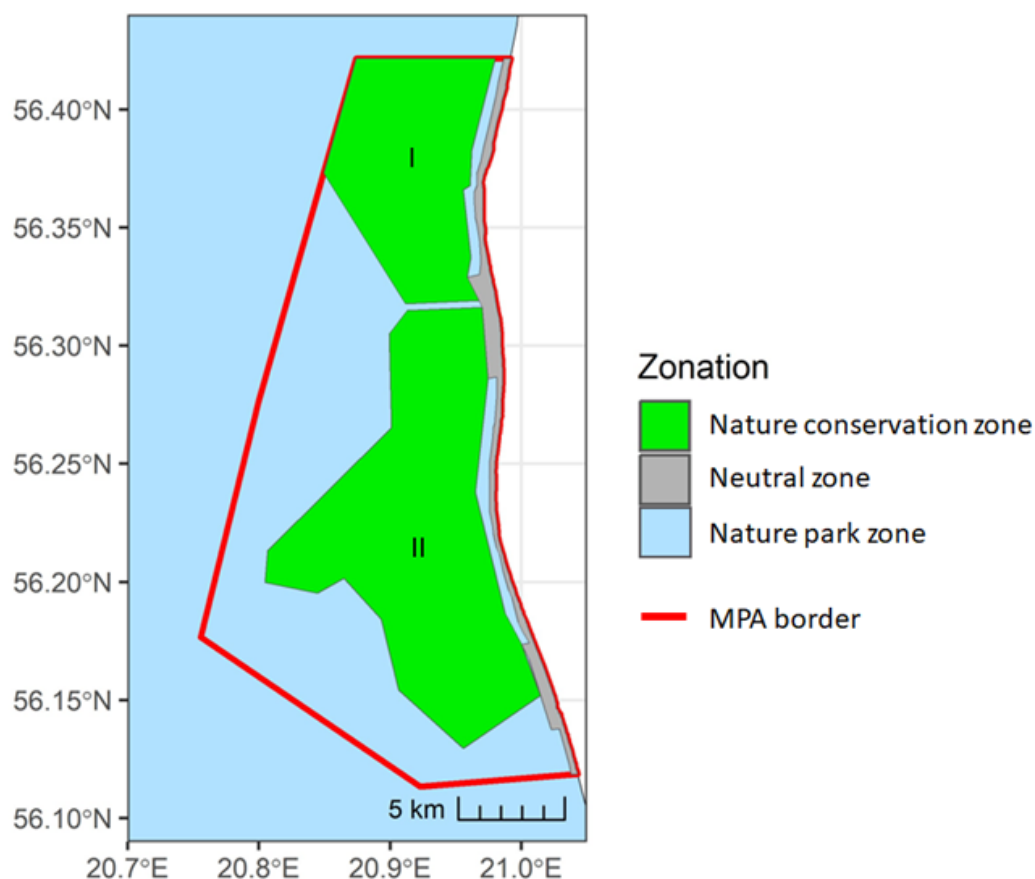


Figure 3. The Nida-Perkone MPA zonation.

2.2. Management measures

There are no MPA-specific fisheries restrictions in the Nida-Perkone MPA, and the fishery is regulated similarly to other marine areas in Latvia in accordance with Latvian legislation, e.g. Regulations on industrial fishing in territorial waters and waters of the economic zone (Regulations of the Cabinet of Ministers No. 296 of May 2, 2007) and Regulations on industrial fishing limits and the procedure for their use in coastal waters (Regulations of the Cabinet of Ministers No. 1375 of November 30, 2009). All current site-specific restrictions aim to prevent direct damage to the seafloor and protect bird species. However, an updated NCP for the conservation and management of marine habitats is being developed within the LIFE REEF project (LIFE REEF, 2020) and may include fisheries restrictions in the future, notably taking into consideration the EU Biodiversity Strategy for 2030, stating that fisheries-management measures must be established in all MPAs, and the EU marine action plan. The new NCP will be adopted in 2025. In the coming years, the LIFE REEF project also plans to develop scientific advice for national fishery management in Latvian coastal waters to reduce seabird and marine mammal bycatch and limit the impact of invasive species.

2.3. Fisheries

The Nida-Perkone MPA is an essential area for coastal small-scale fishery using passive gears. Historically, it was part of the coastal community lifestyle and provided many families' livelihoods. During the Soviet Union's occupation (1940–1990), the Latvian coastal area of the Baltic Sea also bordered the Union of Soviet Socialist Republics (USSR), so any offshore activity, including fishing, was strictly restricted or prohibited. Once Latvia's independence was restored (1990), the coastal fishery developed again. Historically, the main species in the Nida-Perkone coastal fishery were herring (*Clupea harengus*), flounder (*Platichthys* sp.), cod (*Gadus morhua*) and garfish (*Belone belone*).

The round goby (*Neogobius melanostomus*) is an invasive species of Ponto-Caspian origin. It was first observed in the Baltic Sea in the Gulf of Gdansk in 1990 (Skóra and Stolarski, 1993). The first recorded sighting of round goby in Latvian waters was in 2004 near Liepāja (the south-western corner of the Latvian coast, near the Nida-Perkone MPA) (Kruze et al., 2023). Because of the low biodiversity in the Baltic Sea ecosystem and the level of unoccupied ecological niches, the ensuing invasion of invasive brackish-water species (Ojaveer et al., 2015) was not a surprise. The population of round goby in Latvia grew rapidly: it became the dominant species in many coastal areas and posed intense predatory pressure on epibenthic molluscs (Kruze et al., 2023). Reefs and rocky habitats are suitable for non-native species, such as round goby, providing spawning areas and a rich food base (Karlson et al., 2007). The observational analysis conducted in the Nida-Perkone MPA in 2014 revealed a fivefold decrease in mussel biomass in coastal regions compared to the data recorded in 2006 (LIAE, 2017). Studies have also shown that the juvenile turbot (*Scophthalmus maximus*) population in the Nida-Perkone MPA declined following the invasion of round gobies, and have documented a negative correlation between dietary overlap and their abundance (Ustups et al., 2016). This evidence indicates that the presence and activity of round goby have had an impact on the Nida-Perkone MPA's ecological dynamics.

After the population increase, the round goby became an essential resource for the coastal fishery. Several management activities have been implemented in Latvia to effectively utilise the abundant round goby resource, including the design of specialised fishing gears and methods to minimise bycatch of non-target species.

The Latvian coastal fisheries management scheme involves annual data collection from commercial and scientific fisheries followed by information analysis and biological parameter estimates to assess stock status. All available information is used to develop scientific advice for the policymakers, enabling them to suggest necessary changes in fishing policy and defining allowable fishing gear limits in each coastal municipality. The Institute of Food Safety, Animal Health and Environment (BIOR) provided advice on the round goby fishing season and gill-net mesh size based on results from scientific coastal gill-net surveys. A specialised fishery using round goby gill nets in Latvia was initiated in 2015 (in Nida-Perkone in 2017). BIOR also tested the effectiveness of round goby trap-nets in cooperation with the local fishers. As a result, a new round goby trap-net has been used in Latvia since 2018. These specialised gears were similar to other coastal gears – the main difference was the smaller mesh size and a gill-net height limit (1.5 m). Seasonal and spatial fishing restrictions have also been introduced. Both gears led to an increase in round goby fishing selectivity and total landings. Population indices have decreased since the development of the specialised round goby fishery in recent years (Kruze et al., 2023). However, total landings for the coastal fishery in recent years have remained high at 550–933 tons per year (BIOR, 2022). Currently, round goby is the second-most fished species after herring in the entire Latvian coastal fishery. Historically, one of the highest round

goby commercial landings in the Baltic Sea has been observed on the south coast of Latvia (ICES, 2022). This territory also includes the Nida-Perkone MPA.

3. AIMS AND OBJECTIVES

The main aim of the case study is to investigate fishing activities in the Nida-Perkone MPA and analyse the possible changes in the fishery and its spatial dynamics after the round goby invasion. The case study's objectives are to analyse changes in coastal commercial and self-consumption fishery, offshore commercial fishery, and coastal fisheries management and to outline challenges and lessons learned from the perspective of invasive-species management.

4. METHODOLOGY

For this case study, Latvian fishery logbook data available at BIOR has been prepared to facilitate spatial and temporal fisheries analysis for the Nida-Perkone MPA. Coastal commercial fishery data covered the period 1995–2022, offshore fishery data covered the period 2004–2022 and coastal self-consumption fishery data covered the period 2014–2022.

Offshore fishery data contains detailed landing-site coordinates. All data sets enable the extraction of information on fishing time, landings per species, used fishing gear, and involved companies/fishers. Coastal commercial and self-consumption fishery data are reported on a municipality level, without detailed coordinates (Nida-Perkone MPA consist of two municipalities – Nīca and Rucava parish).

According to Latvian legislation (Regulations of the Cabinet of Ministers No. 296 of May 2, 2007), coastal waters are defined as areas up to 20 m depth (excluding offshore zones behind 20 m isobath). Thus, offshore fishery refers to areas with depths above 20 m and coastal fishery (including self-consumption fishery) refers to depths below 20 m (from the coastline). Self-consumption fishery refers to the practice of using fishing rights that are granted to an individual in accordance with established laws and regulations for the purpose of catching fish for personal consumption. This means that the fish caught cannot be sold in the market.

In recent years, the case study report authors were actively involved in national fisheries management processes and thus are sharing their experience regarding fisheries management at the Nida-Perkone coast.

Data analysis and visualisation were conducted in R (R Core Team, 2023) using the R-Studio environment (Posit Team, 2023). The main packages were *ggplot2* (Wickham, 2016), *sf* (Pebesma, 2018), *tidyverse* (Wickham et al., 2019), *rgdal* (Bivand et al., 2023), *viridis* (Garnier et al., 2023), *data.table* (Dowle and Srinivasan, 2023), *mapplots* (Gerritsen, 2018), *shapefiles* (Stabler, 2022) and *maptools* (Bivand and Lewin-Koh, 2022).

5. RESULTS

By annual landings, the coastal fishery was dominant in the investigated period (Figure 4). Offshore landings were low as only a minor part of the Nida-Perkone MPA is accessible for the offshore fishery (deeper than 20 m).

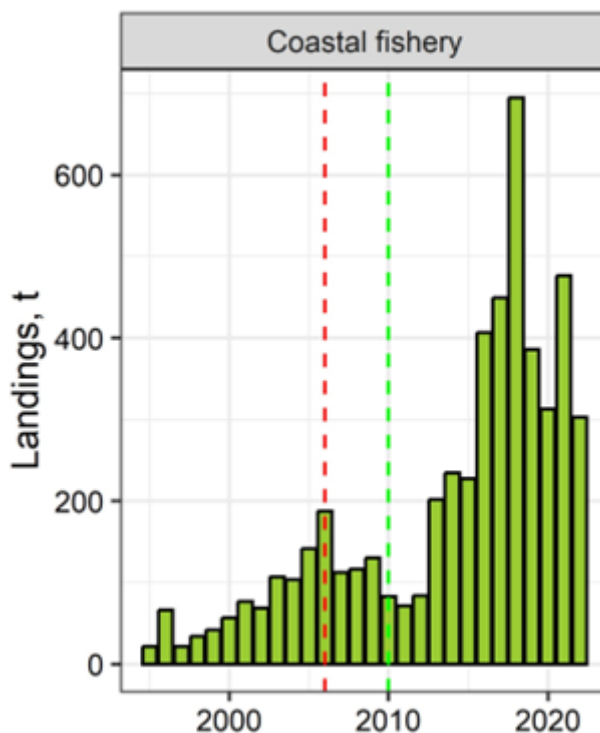


Figure 4. Annual total landings (t) of all species in the coastal fishery in the Nida-Perkone MPA. The red dotted line indicates the year when round goby was first recorded near the MPA (2006) and the green dotted line indicates the year when the MPA was established (2010).

5.1. Coastal fishery

Coastal-fishery landings were relatively stable until 2013. After 2013, landings increased gradually following the increase in the population of round goby, reaching a record-high value in 2018 (695 tonnes). The main fishing gears up to 2013 were gill nets, but afterwards, because of the round goby invasion, herring pound nets and round goby specialised fishing gears (round goby gill nets and trap-nets) were dominant (Figure 5). Herring pound nets target spring-spawning herring stock and are suitable for round goby fishery because of their small mesh size. The herring fishery overlaps with round goby seasonal activity in April–June.

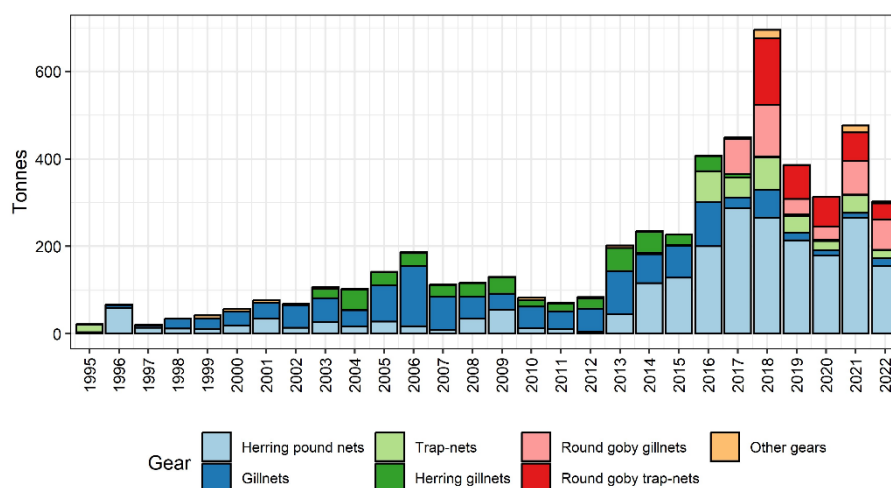


Figure 5. Annual landings (tonnes) by fishing gears in the coastal fishery in the Nida-Perkone MPA (1995–2022).

The number of coastal fishing companies has increased recently, likely due to the development of the round goby fishery (Figure 6). Most of them are fishing seasonally, and the annual number of fishing days at sea is below 60.

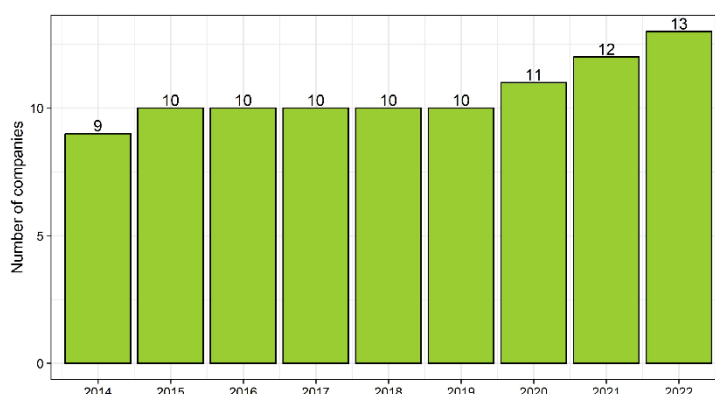


Figure 6. The number of coastal fishing companies operating in the Nida-Perkone MPA (2014–2022).

Total landings in the Nida-Perkone MPA coastal fishery depend on fishing regulations (number of allowed fishing gears, bycatch restrictions, etc.), fish stocks and fishing activity. In all year periods, the highest landings were observed in the northern part of the MPA, and no significant changes in spatial reallocation were seen (Figure 7).

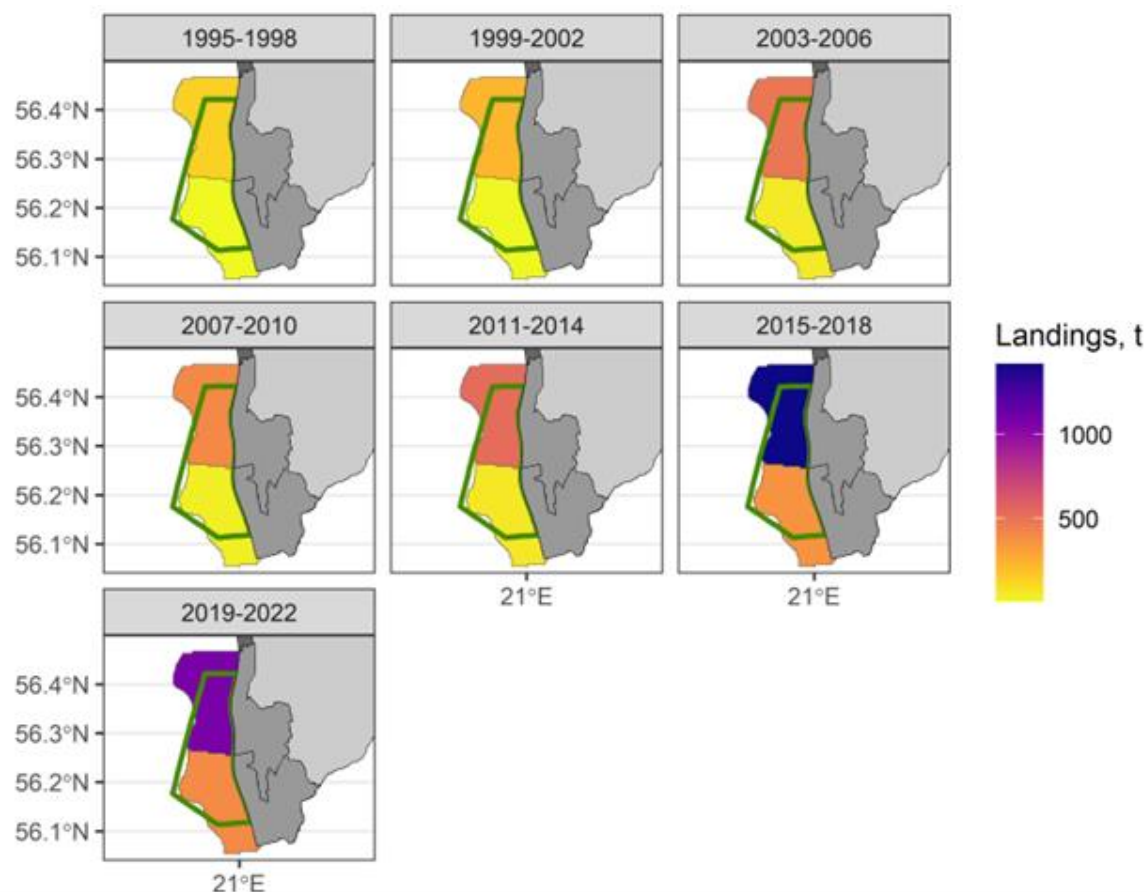


Figure 7. Spatial distribution of coastal landings (tonnes) in the Nida-Perkone MPA.

In recent years, catch rates have peaked in the second quarter from April to June, and fishes are caught mainly with herring pound nets, round goby gill nets and round goby trap-nets (Figure 8). Round goby, herring, cod, flounder and garfish are the main species in coastal landings (Figure 9). In recent years, round goby has become a dominant species, comprising more than 80 % of landings (Figure 10).

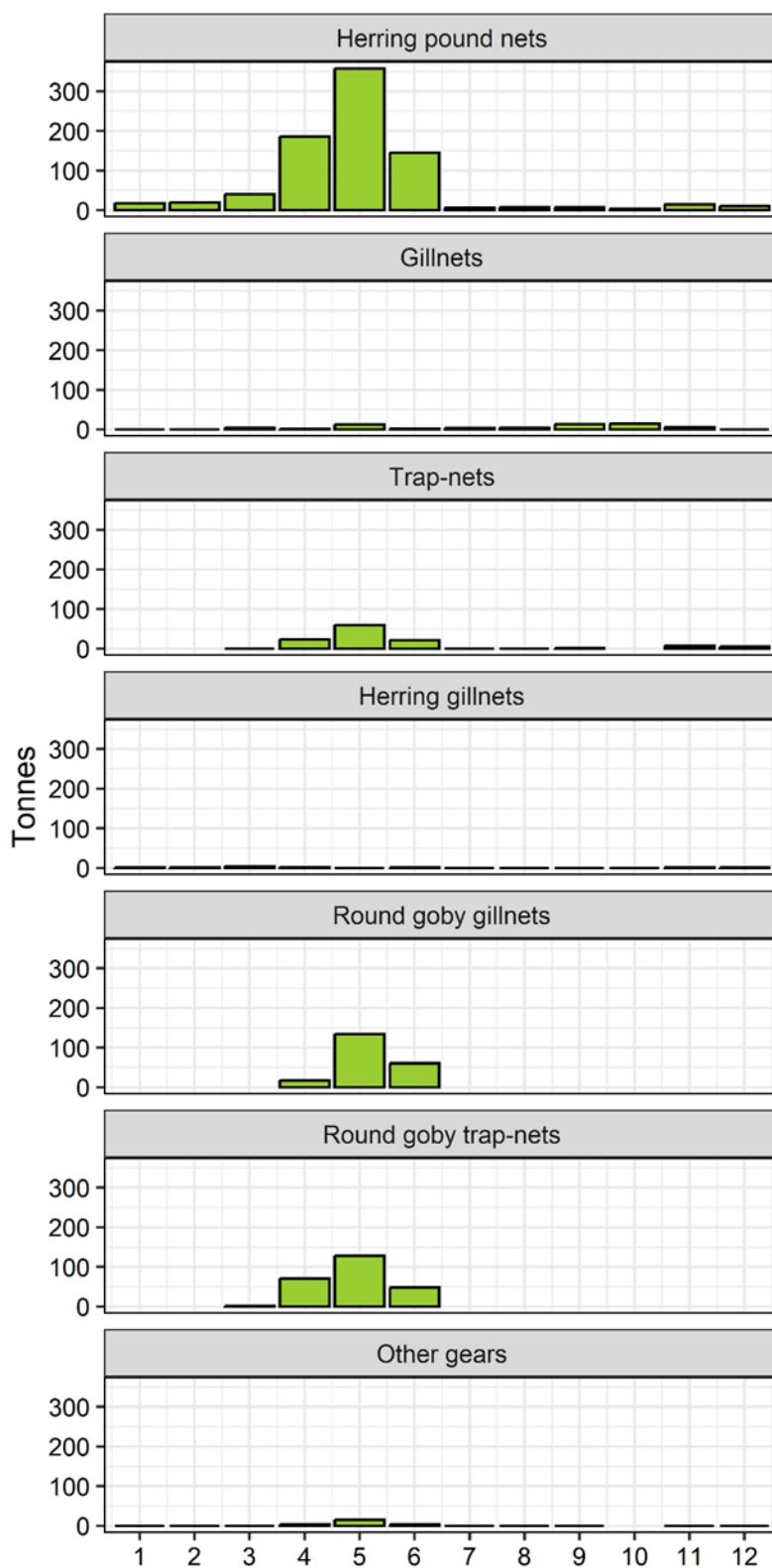


Figure 8. Monthly landings by coastal fishing gear in the Nida-Perkone MPA (2019-2022). Months on the x-axis: 1 = January, 12 = December.

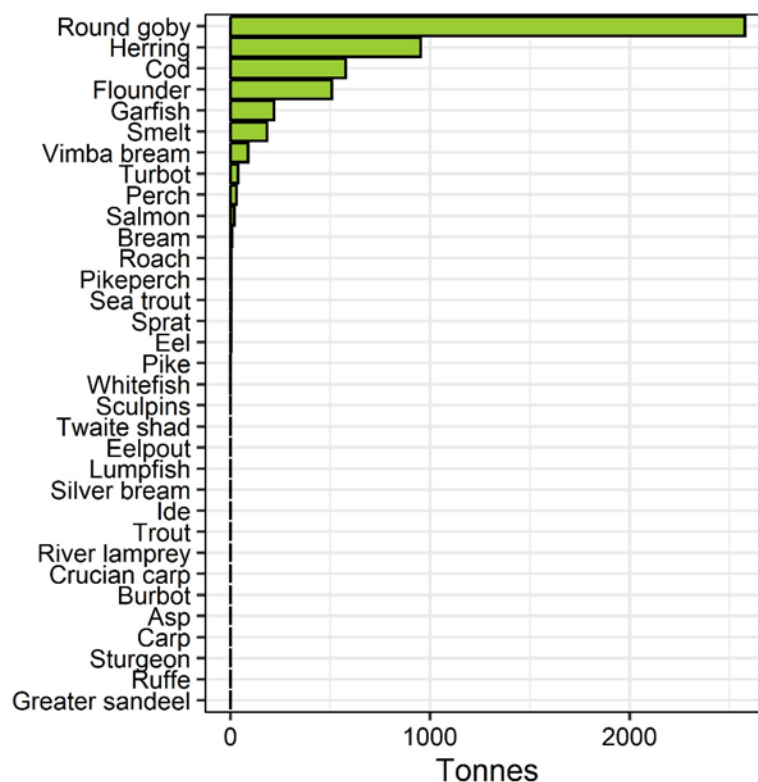


Figure 9. Landings (tonnes) by species in the Nida-Perkone MPA coastal fishery (1995–2022).

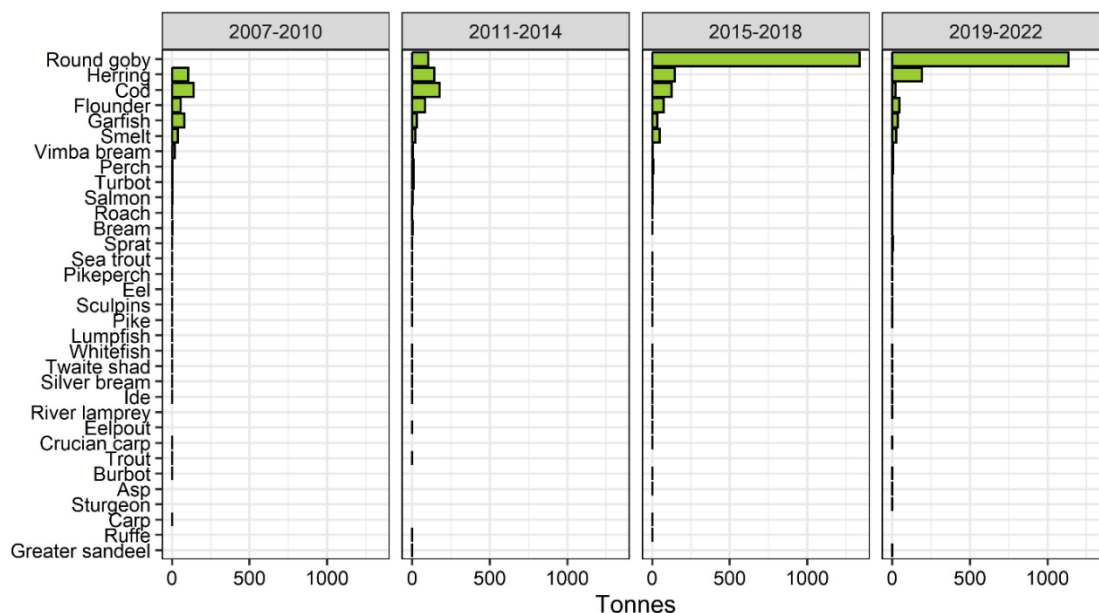


Figure 10. Landings (tonnes) by species by recent year periods in the Nida-Perkone MPA coastal fishery.

5.2. Self-consumption fishery

Self-consumption fishery is a part of recreational fishery where a limited number of residents can apply for fishing permits that allow the use of specific fishing gear. Landed fish are intended for self-consumption, and further selling is prohibited. In the Nida-Perkone MPA, this type of fishery is negligible, and the total landings comprise less than 1 % of the coastal commercial fishery landings (Figure 11).

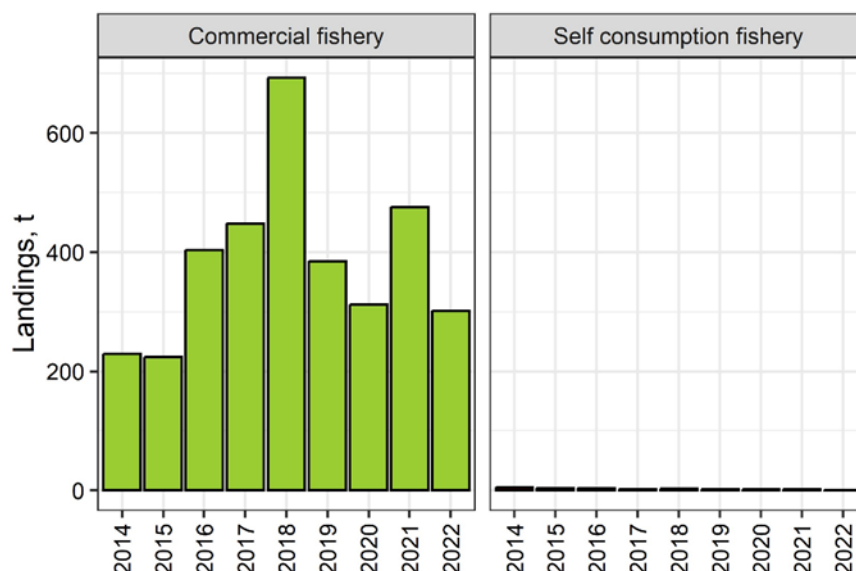


Figure 11. Annual landings (tonnes) in the commercial coastal fishery and self-consumption fishery in the Nida-Perkone MPA (2014–2022).

The main fishing gears are gill nets and herring gill nets, and in recent years, total landings have decreased (Figure 12). The highest landings were observed in the northern part of the MPA (Figure 13) from March to September (Figure 14).

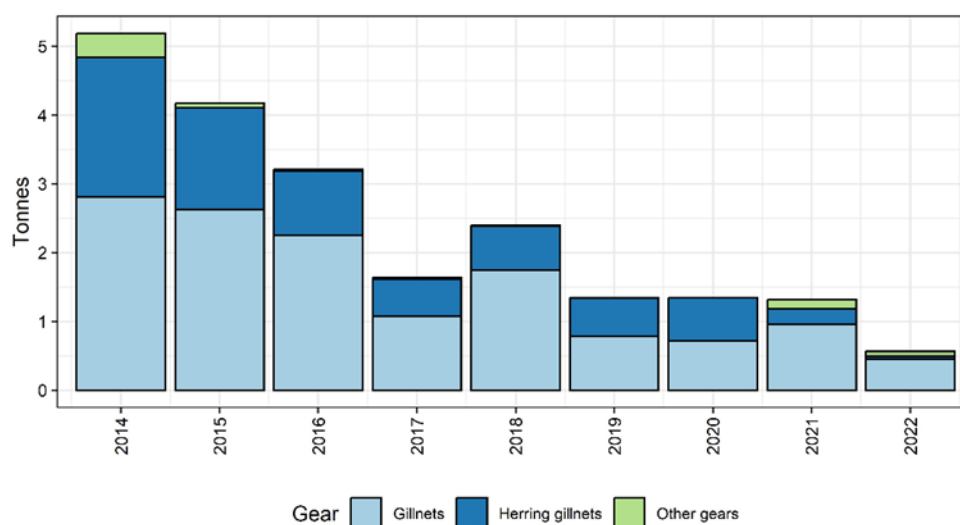


Figure 12. Annual landings (tonnes) by fishing gear in the self-consumption fishery in the Nida-Perkone MPA (2014–2022).

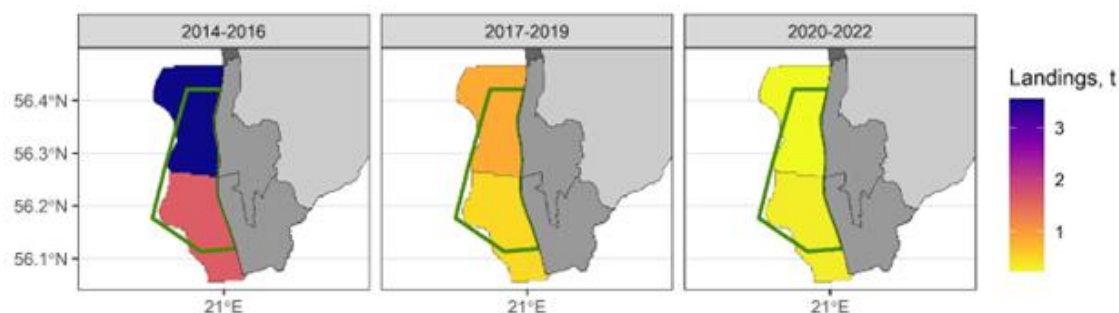


Figure 13. Spatial distribution of self-consumption landings (tonnes) in the Nida-Perkone MPA.

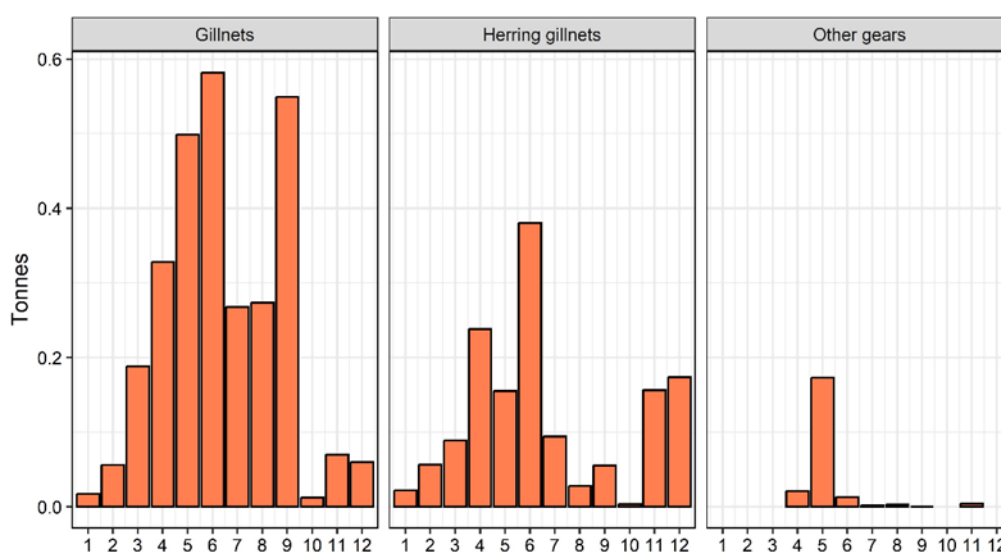


Figure 14. Monthly landings (tonnes) by self-consumption fishing gears in the Nida-Perkone MPA (2014–2022). Months on the x-axis: 1 = January, 12 = December.

There is a high variety of dominant species in the catches of the self-consumption fishery (Figure 15). However, in the coastal commercial fishery, only a few species are dominant. In recent years, flounder, herring and vimba bream (*Vimba vimba*) landings have decreased, and round goby has become one of the dominant species (Figure 16). However, the dominance of the round goby in the self-consumption fishery is lower than in the coastal commercial fishery (see Figure 10). These differences could be related to differences in deployed fishing gear and preferences for other target species in self-consumption fishery. In Latvia, people are often reluctant to catch this new species and prefer more traditional ones.

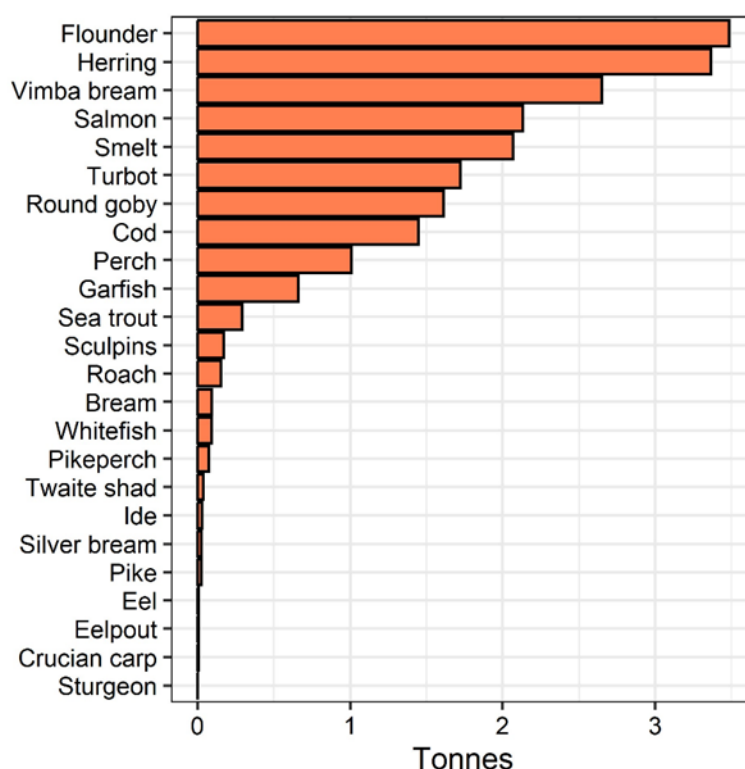


Figure 15. Landings (tonnes) by species in the Nida-Perkone MPA self-consumption fishery (2014–2022).

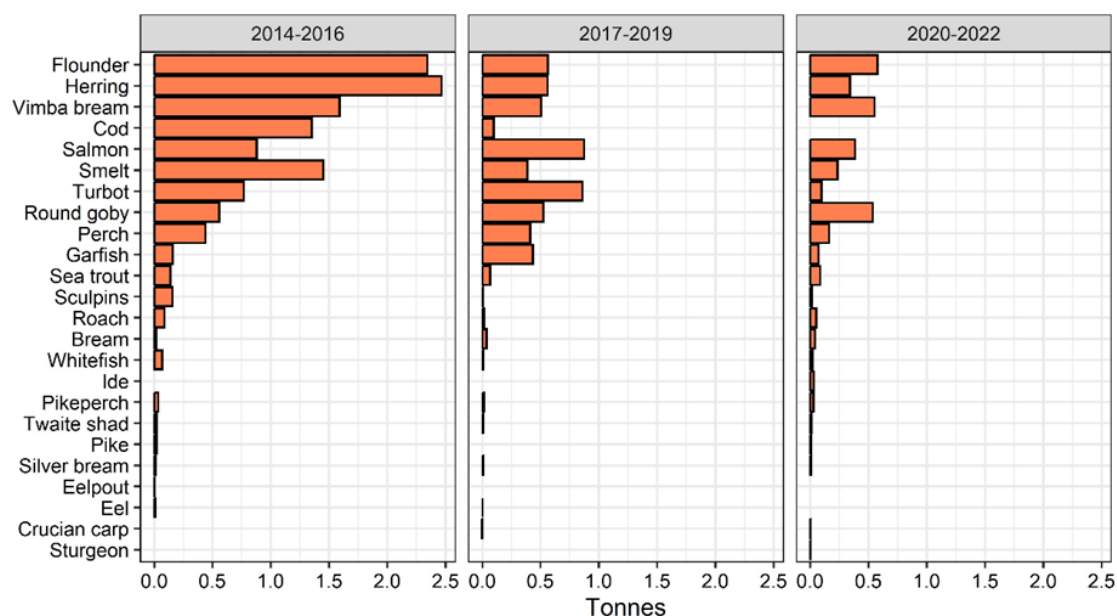


Figure 16. Landings (tonnes) by species by recent year periods in the Nida-Perkone MPA self-consumption fishery.

5.3. Offshore fishery

Offshore landings were generally low because only a minor part of the Nida-Perkone MPA is accessible for offshore fishery (deeper than 20 m). Only bottom gill nets were used, irregularly, in the analysed period. Landings fluctuated – and since 2019, no offshore landings have been recorded in the Nida-Perkone MPA (Figure 17). Offshore bottom gill-net fishery mainly occurs outside the Nida-Perkone MPA (Figure 18 and Figure 19).

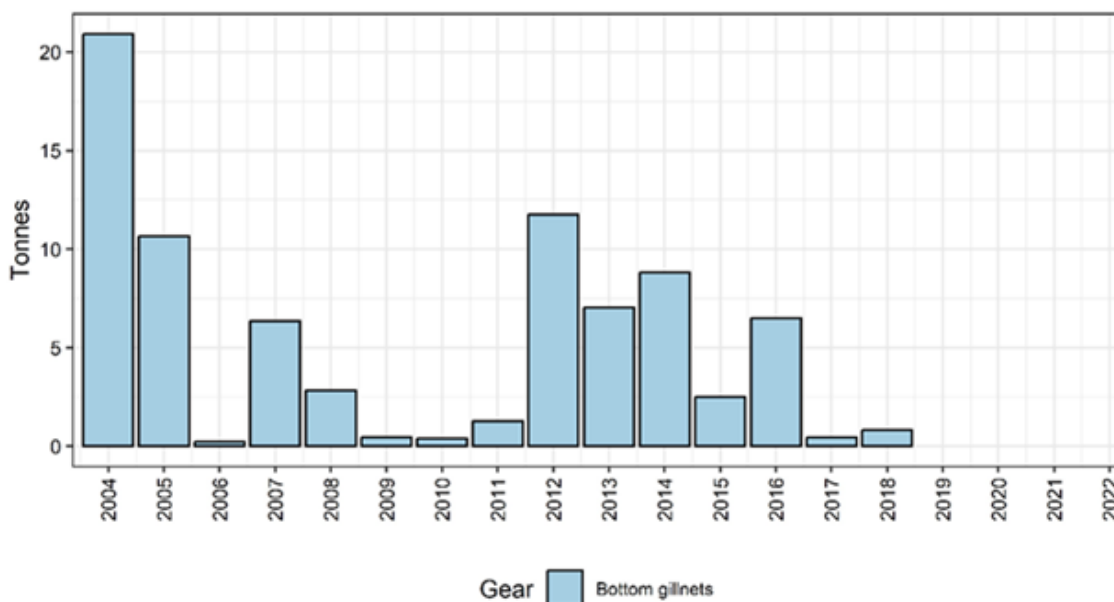


Figure 17. Annual landings (tonnes) of the offshore fishery in the Nida-Perkone MPA.

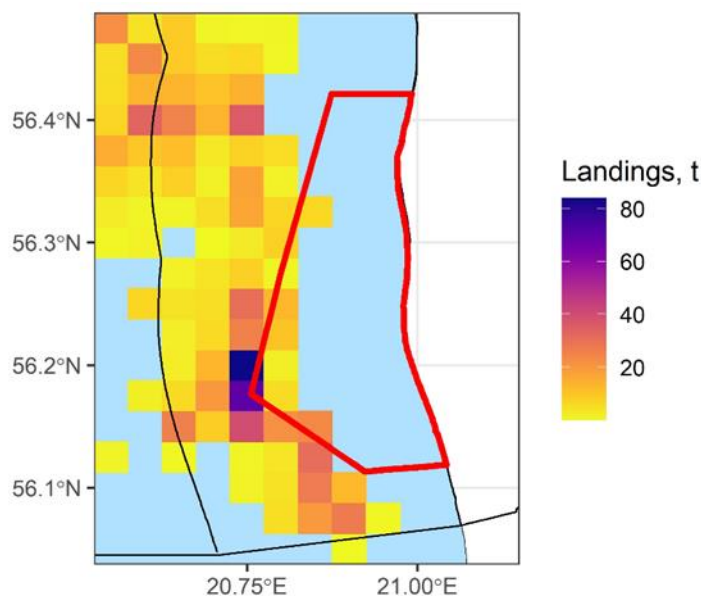


Figure 18. Spatial distribution of offshore bottom gill-net landings in the Nida-Perkone MPA and surrounding areas (2004–2018). The MPA border is in red. The border of the Latvian exclusive economic zone (EEZ) and the border of the Latvian territorial waters are in black.

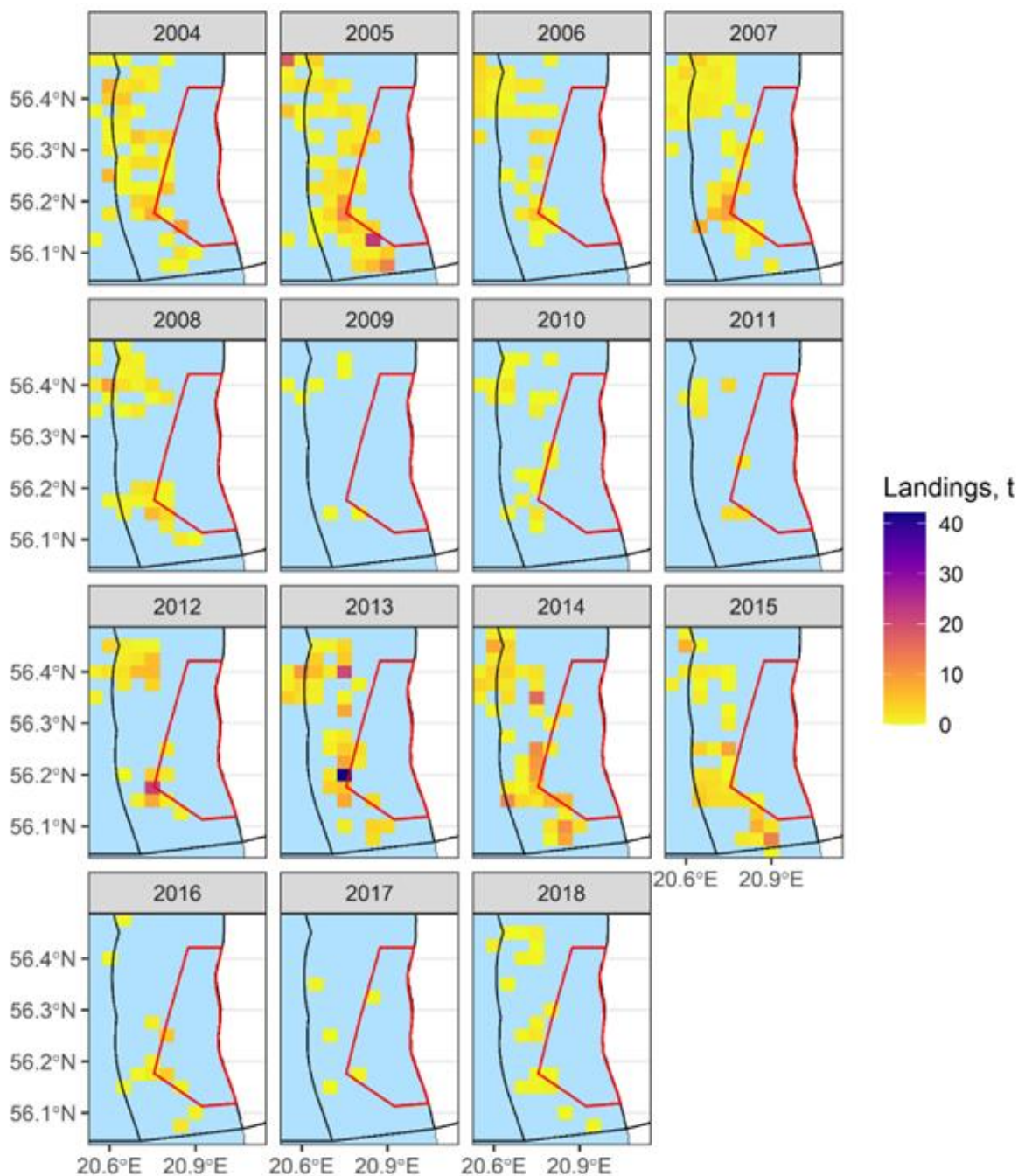


Figure 19. Spatial distribution of offshore bottom gill-net landings (tonnes) by year in the Nida-Perkone MPA and surrounding areas (2004-2018). The MPA border is in red. The border of the Latvian exclusive economic zone (EEZ) and the border of the Latvian territorial waters are in black.

Offshore fishing mainly occurs from October to January (Figure 20), and the main target species are cod and flounder (Figure 21). In recent years, offshore fishery has decreased and ultimately stopped in the MPA and its surrounding areas, mainly because of a decrease in the cod population and ensuing cod-fishing restrictions (Figure 22).

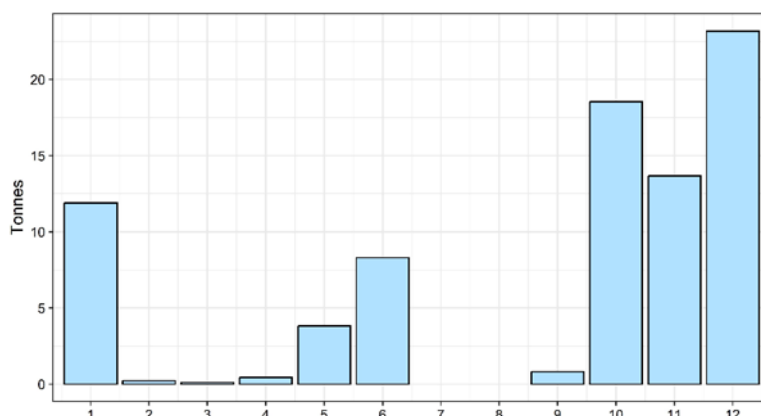


Figure 20. Monthly landings (tonnes) of offshore bottom gill nets in the Nida-Perkone MPA (2004–2018). Months on the x-axis: 1 = January, 12 = December.

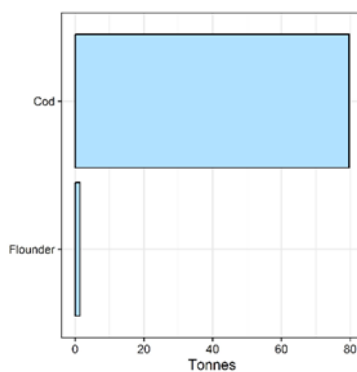


Figure 21. Landings (tonnes) by species in the Nida-Perkone MPA offshore bottom gill-net fishery (2004–2018).

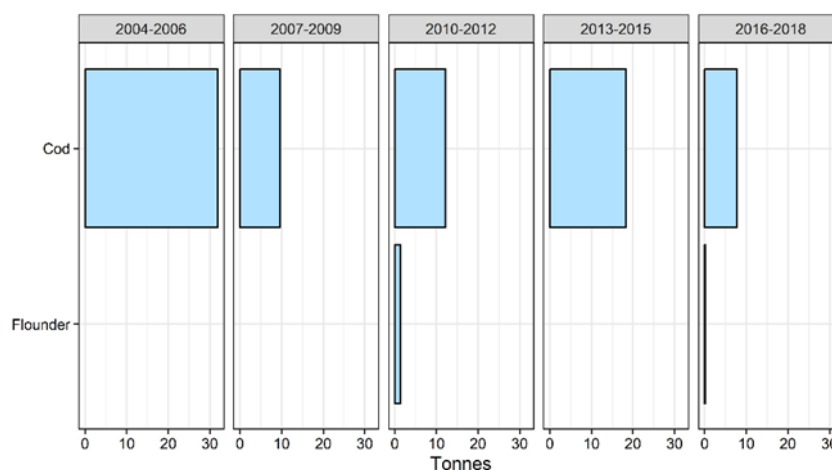


Figure 22. Landings (tonnes) by species and year periods in the Nida-Perkone MPA bottom gill-net fishery.

6. DISCUSSION

The Nida-Perkone MPA is an essential area for coastal fishery and for maintaining biological diversity (Baltic Environmental Forum, 2009). The main challenge is combining and saving both of these values and taking a precautionary approach. These protected areas are affected by activity-footprints, including the influence and consequences of pressures emanating from outside the management area, like shipping activities, bottom trawling and overfishing (Elliott et al., 2020). The Nida-Perkone MPA is exposed to relatively high shipping and fishing pressure, as it is located between two large ports in Liepaja and Klaipeda. Consequently, this anthropogenic-activity footprint introduces new vectors for introducing novel species within the MPA. It should be added that the primary vector of the introduction of the round goby to the Baltic Sea is most probably ship traffic (Corkum et al., 2004; Kornis et al., 2012; Kotta et al., 2016; Florin et al., 2018).

The highest landings of all species in the coastal commercial fishery were observed in the northern part of the Nida-Perkone MPA, and no significant changes in spatial reallocation were seen. Fishing regulations at the municipal level (Rucava and Nīca parish), such as the permitted number of fishing gear and fishing activity by local companies, play a crucial role in determining the total landings in the MPA coastal fishery. Historically, the municipality of Nīca had a higher number of authorised fishing gears (Regulations of the Cabinet of Ministers No. 1375 of November 30, 2009) with the potential for higher effort and landings. The main fishing gears up to 2013 were gill nets targeting mainly cod, but after the round goby invasion and the beginning of specialised round goby fishery, small mesh-size gears such as herring pound nets, round goby gill nets, and trap-nets were dominant. The coastal fishing effort after 2013 increased, resulting in significantly higher landings. A similar increase in fishing effort due to changes in fishing policy and the development of specialized round goby fishery was observed in other municipalities along the Latvian coast as well (Kruze et al., 2023).

Landings in coastal self-consumption fisheries were negligible compared with those in coastal commercial fisheries. The highest self-consumption fishery landings were also observed in the northern part of the MPA. Fishing activity in this segment is occasional and mainly performed by gillnets, targeting various species depending on the fishing season. Round goby is not a target species in the self-consumption fishery.

Offshore landings were generally low, consisting of bottom gill-net fisheries. Only a minor part of the MPA is accessible for offshore fishing (deeper than 20 m). In recent years, offshore fishing has decreased and ultimately stopped in the MPA and its surrounding areas, mainly due to a decrease in the cod population followed by additional fishing restrictions.

After the round goby invasion, our findings show that **the fishery, within the Nida-Perkone MPA, gradually switched towards round goby exploitation.** The reason for high catches was also a thriving export market to the Caspian Sea region, and this fish has turned from an unwanted fishing object into a precious catch. Only a minor quantity of round goby is sold on the local market. The fishery of round goby has led to a 'bioeconomic paradox', where a commercialised harvest for invasive species offers a compelling solution to control their abundances and reduce their impacts on ecosystems. However, reducing the population of invasive species can reduce their catch rates and make their harvest uneconomical (Harris et al., 2023). There is a lot of pressure from the Latvian fishers and fishing sector, who are interested in increasing fishing pressure for round goby, justified by the non-indigenous status of this species. On the other hand, because of the observed population decrease, management could take a precautionary approach. A thoughtless

increase in fishing pressure can increase the risk of bycatch of unwanted species or undersized individuals of desirable species.

It is rather difficult to compare the experience of other countries in the round goby fishery and its effect on the MPA territories. There is also no information on active round goby fishery management in other Baltic Sea countries (ICES, 2022). So far, there is no unified monitoring programme for assessment of the round goby population in the Baltic Sea, and estimates of abundance are scarce, although ecological impacts depend on the abundance of the species (Ojaveer et al., 2015). A targeted fishery for this species exists only in Latvia, Lithuania and Estonia (ICES, 2022). Based on modelling studies, **it is expected that protected areas will contain fewer round goby than unprotected areas** with similar habitat characteristics; this is because of stronger interspecific interactions with predators and competitors that prevent the establishment of the round goby. Therefore, MPAs can work as biodiversity conservationists by providing larger numbers of natural enemies that will reduce the round goby population (Holmes et al., 2019). However, this could be challenging for the Baltic Sea ecosystem, where natural predators like cod and turbot are scarce in many areas.

The overall impact of round goby is unequal. Round goby is an invasive species that potentially threatens habitat-forming species such as blue mussels. **In the Nida-Perkone MPA, round goby catches are one of the highest in Latvia** and have comprised 39 % of the total catch in Latvia since 2019. This might be because of the rocky habitats typical for the Nida-Perkone MPA coastal area. Such habitats provide spawning areas and a rich food base, including mussels for round goby. The observational analysis conducted in the Nida-Perkone MPA in 2014 revealed that mussel biomass within the coastal regions had decreased fivefold compared to that recorded in 2006 (LIAE, 2017). This could increase eutrophication risks. Studies also conducted in the Nida-Perkone MPA have demonstrated a decline in juvenile turbot (*Scophthalmus maximus*) abundance since the round goby invasion. These findings provide empirical evidence of a negative correlation between dietary overlap and the abundance of these species (Ustups et al., 2016). On the other hand, round goby extraction could improve the biological diversity, food resource availability for other fishes, ecological plasticity and socioeconomic aspects, which are also incorporated within MPA NCP.

The impact of the round goby on the MPA areas is expected to decrease. Recent studies show that population numbers of round goby in Latvian waters peaked in 2018, followed by a sharp decline. Intensive commercial fishing caused a considerable rapid decline of the species, but it is possible that the decline was amplified by a broader scale decline, as observed in many areas of the Baltic Sea (Kruze et al., 2023). Despite the decrease in numbers, the population most likely will not disappear from the local ichthyofauna and has the potential to become an important fishing object in other Baltic Sea areas. Assessing the fishery effect on the round goby population reduction is difficult because the population also depends on environmental changes and fishing activities in neighbouring countries.

Functional zoning, which foresees nature conservation and neutral zones, was developed for the Nida-Perkone MPA. A number of prohibited/regulated activities were included in the Nida-Perkone MPA NCP, which is currently being updated. As a result, these regulations should reduce anthropogenic pressure, ensure ecological functions, and balance nature conservation. Establishing these zones is unlikely to eradicate the round goby population in the Nida-Perkone MPA. New challenges could arise from the EU's intention to protect marine biodiversity, including placing 10 % of waters under a strict protection regime and fishery-related measures necessary for sustainable management of the round goby population.

7. CONCLUSIONS

Invasive species are one of the most significant issues in conservation biology today. Growing global trade and environmental change accentuate the spread and intensity of marine biological invasions, making their potential management a priority. To ensure a good environmental status (GES), introduced species must be at a level that does not affect the ecosystem negatively. Unfortunately, there is evidence that the round goby is having negative impacts on the Nida-Perkone MPA (Kruze et al., 2023, LIAE, 2017; Ustups et al., 2016).

For invasive fish species, catch quotas are not set, but the fishery could be regulated by defining fishing gears, fishing season and effort. An increase in the population of round goby, emerging market opportunities and national fisheries policy all contributed to the rapid growth of the specialised goby fishery in the Nida-Perkone MPA.

Findings of this case study show that the ichthyofauna of the Nida-Perkone MPA has, in the short term, witnessed the ascendancy of the round goby as the most dominant fish species. As a result, the Nida-Perkone MPA area is under high socioeconomic pressure from the fishing sector, which is interested in developing a round goby fishery, justified by the invasive nature of the species. On the one hand, round goby extraction could improve the ecosystem's health. However, there is a potential conflict between exploiting the novel resource and mitigating the bycatch risk for marine mammals and birds, which are also one of the focus organisms associated to conservation objectives of the Nida-Perkone MPA. Scientists providing advice try to take a precautionary approach and, in many cases, there are disagreements with fishers. A thoughtless increase in fishing pressure can result in an increased risk of bycatch of unwanted species or undersized individuals of desirable species.

The scientific advice for national fishery management in Latvian coastal waters to reduce seabird and marine mammal bycatch and limit the impact of invasive species is still in progress and will be concluded by 2025. The EU's intention to protect marine biodiversity, including placing 10 % of waters under a strict protection regime, could restrict the application of fishery-related measures necessary for controlling the round goby population in the MPA.

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Case Study Report

Adler Grund og Rønne Bank Denmark – Baltic Sea



Mapping of marine protected areas and their associated fishing activities

Elliot J. Brown

***Section for Ecosystem Based Marine Management
National Institute of Aquatic Resources***

The Technical University of Denmark

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LIST OF ABBREVIATIONS

Term	Description
AGRB	Adler Grund og Rønne Banke
AIC	Akaike Information Criterion
AIS	Automatic Identification System
BACI	Before-After, Control-Impact
BACI TS	Before-After, Control-Impact Time Series
DG ENV	Directorate-General for Environment
DG MARE	Directorate General for Maritime Affairs and Fisheries
EEZ	Exclusive Economic Zone
EMODnet	European Marine Observation and Data Network
eNGO	Environmental Non-Governmental Organisation
EU	European Union
EUNIS	European Nature Information System
Ha	Hectare(s)
ICES	International Council for the Exploration of the Sea
ICES WGSFD	ICES Working Group on Spatial Fisheries Data
LPUE	Landings Per Unit Effort
MBCG	Mobile Bottom-Contacting [Fishing] Gear
MMSI	Maritime Mobile Service Identity
MPA	Marine Protected Area
REML	REstricted Maximum Likelihood
SAC	Special Area of Conservation
SCI	Site of Community Importance
VMS	Vessel Monitoring System(s)

1. EXECUTIVE SUMMARY

The Adler Grund og Rønne Banke (AGRB) Marine Protected Area (MPA) is a large Natura 2000 area located south-west of the Danish Island of Bornholm in the Baltic Sea. The AGRB is designated as a Special Area of Conservation (SAC) under the European Union (EU) Habitats Directive based on the presence of rocky reefs, sand banks and harbour porpoise. Subsequent to the AGRB's SAC designation in 2016, EU regulations were proposed by a coalition of Baltic states, led by Denmark, to ban fishing with mobile bottom-contacting fishing gears (MBCG) on and around reefs in the MPA, creating a number of buffer zones. These regulations came into force in March 2017.

The aim of this case study report is to investigate whether the designation of the site as a Natura 2000 SAC or the subsequent fishing regulations had an effect on fisheries effort or landings in the MPA, and if so, whether these effects were dependent on the habitat being fished. To detect any change in fishing patterns in response to policy changes, we utilised a series of post-hoc Before-After, Impact-Control (BACI) experiments, and where the data allowed, included a time-series component (BACI-TS). These experiments were set up to test the effects of both the designation of the MPA as a SAC or the time in which fisheries restrictions came into force. Experiments were also carried out for the MPA as a whole, and for only those subsets of areas where fishing regulations were imposed.

Our results show a dominant retention of models based on the MBCG regulations coming into force, as opposed to the alternative models based on SAC designation date. Moreover, in the only two cases where the SAC designation date models were retained, no significant effect of SAC designation date was detected. This indicates that the only significant detectable changes occurred concurrently with the MBCG regulations. Therefore, while the SAC designation triggered changes to some human activities in the area, fisheries effort (and subsequent landings) appears to respond only to specific fisheries regulations.

In nearly all cases, the responses of landings biomass and landings value followed the response of effort. When considering the whole MPA, the significant decrease in fishing effort and landings appears driven by the changes in MBCG activities. In the spatial context of the MBCG regulated area, there was no significant changes in combined effort, nor in the effort of MBCG or gill net fisheries independently. This pattern is likely due to effort being in decline from the start of our time series for MBCG gears, within the area selected for their prohibition.

Without the effect of buffer zones to protect mapped reefs, it is unlikely that a difference in MBCG activities would have been detected, as these ordinarily did not operate on the reefs themselves, for practical reasons. However, there was no evidence that the habitat types that were fished changed in response to MPA designation or MBCG regulations. In a broader trend, across the MPA and both sets of controls, the gill net fisheries increased from zero to some occurrences after the MPA was designated and primarily MBCG regulations came into force. While not specific to the MPA, this is relevant because of the stated objective of protecting harbour porpoise in the MPA, which are known to entangle and drown in gill nets. This highlights the trade-offs in regulating subsets of fishing activities, while there are complex interactions of fishing across different ecosystem components.

While the SAC designation restricted other human activities in the MPA prior to the fisheries regulations, such as the extraction of sediments, it is primarily the result of MBCG regulations that we see a decrease in fisheries activities in the AGRB MPA.

2. BACKGROUND

2.1. General information

The Adler Grund og Rønne Banke Marine Protected Area (AGRB MPA), is located in the south-western Baltic Sea, south-west of the Danish Island of Bornholm. The MPA is outside Danish territorial waters but inside the Danish Exclusive Economic Zone (EEZ) and adjacent to the German EEZ. It covers an area of around 321 km² (32,124 hectares) over a contiguous, irregular rectangular area (Figure 1).

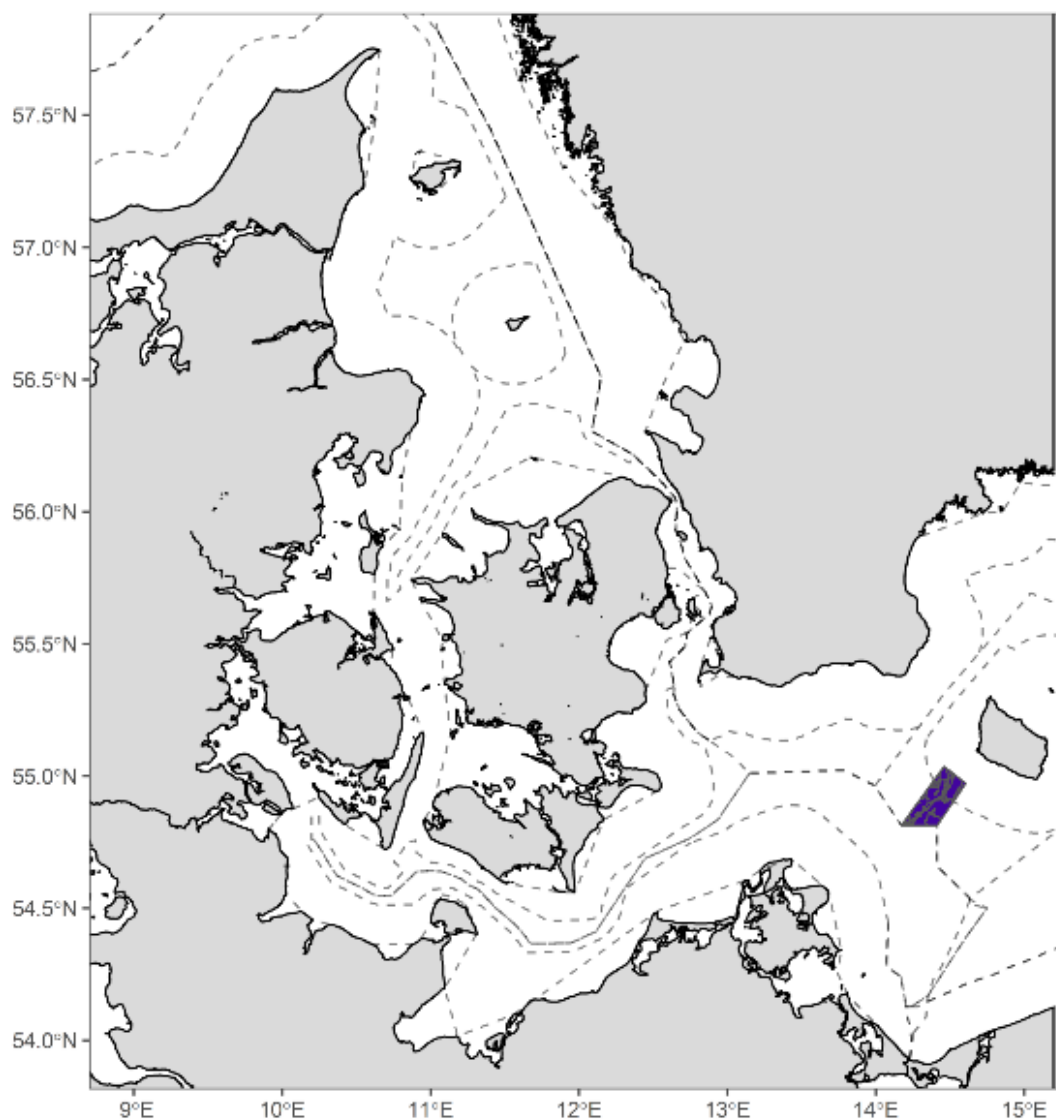


Figure 1. The position of the AGRP MPA in the western Baltic Sea, south-west of the Danish Island of Bornholm.

NB: the AGRB MPA is shown in purple. Dashed lines represent territorial waters, and EEZ borders.

2.2. Natura 2000 status

As part of the Natura 2000 European network of protected areas, the AGRB MPA was first proposed as a Site of Community Importance (SCI) in the end of 2009 and subsequently

designated as such in January 2011. The AGRB MPA was designated as a Special Area of Conservation (SAC) in April 2016.

As a part of the Natura 2000 network, the AGRB MPA was established to represent rare, threatened or characteristic habitats and species. In the case of the AGRB MPA, one species – the harbour porpoise – and two habitat types – rocky reefs and sandbanks – are named as the key components (Miljøstyrelsen, 2020).

While harbour porpoise is wide roaming, relatively high densities of the Baltic sub-population have been reported in the area of the AGRB MPA, especially in winter. Furthermore, the AGRB MPA is of a relatively large size, affording longer periods of residence to these more mobile species (Hansen and Høgslund, 2019).

The AGRB MPA contains a large system of rocky reefs that stretches in a south-west to north-east direction and continues into areas outside the MPA (Fredshavn, 2012) (Figure 2), including across into the German EEZ in the southwest. Surrounding this network of reefs, are various soft-bottom type habitats including large sandbanks at varying depths (Hansen and Høgslund, 2019).

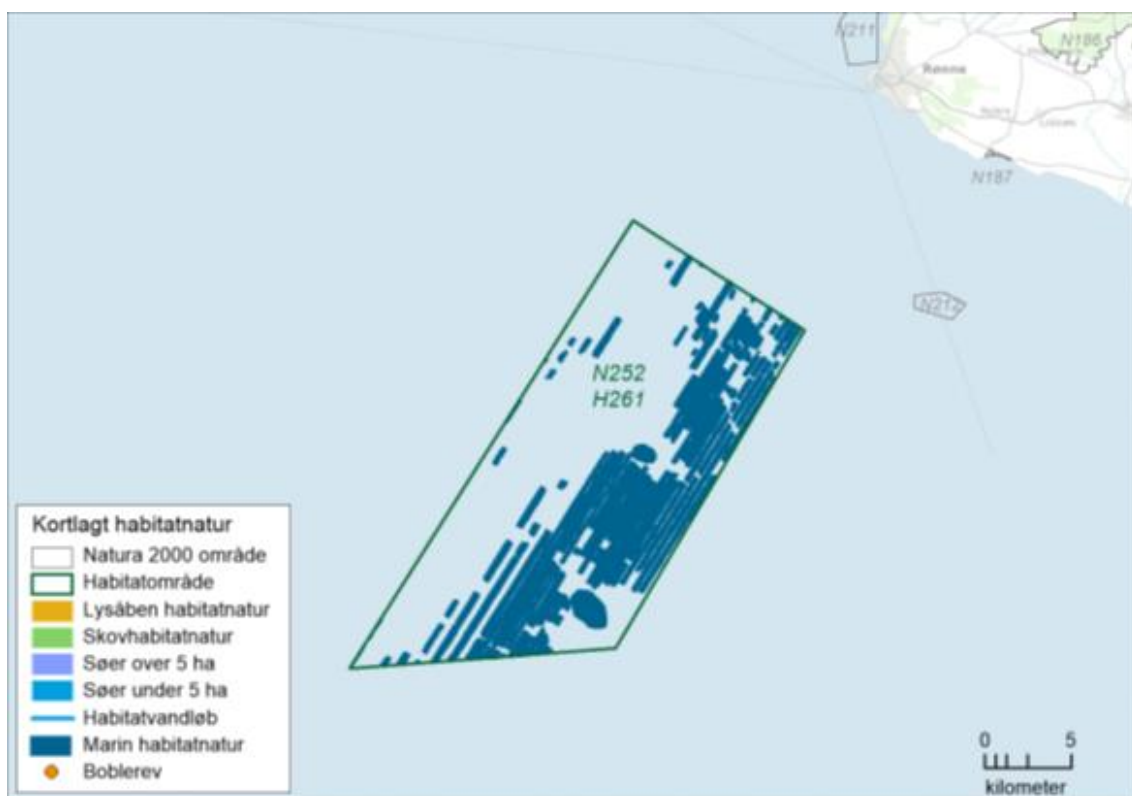


Figure 2. Map of the AGRB MPA showing mapped reef areas from a survey in 2012 that was used in the designation of the area as a SAC for rocky reefs.

NB: The green polygon delineates the MPA; mapped reef areas are shown in dark blue. Figure from the Danish Ministry for the environment (Miljøstyrelsen, 2020), based on Fredshavn (2012).

2.3. Fisheries regulations

In 2015, Denmark applied European legislation on a range of Natura 2000 SACs to protect reef habitats, including stone reefs, biogenic reefs and so-called 'bubble-reefs', which are created by mineral deposits from geological gas leaks. The new legislation prohibited the use of all gears on the bubble-reefs and prohibited Mobile Bottom-Contacting Gears

(MBCG) ⁽¹⁾ on the stone reefs and biogenic reefs. In 2017, the legislation was extended to include the AGRB MPA (European Commission, 2017), which also enlarged the prohibition with MBCG on or around (240 m buffer zone) the mapped reefs. Previously, there had been no additional fishing restrictions in force in the AGRB MPA, relative to the surrounding areas.

Prior to the joint proposal from Denmark, Sweden, Germany and Poland for fisheries regulations to be imposed in the common fisheries area, Denmark undertook a series of domestic and international stakeholder consultations (Danish AgriFish Agency, 2016). These began with the already established Natura 2000 dialogue forum (representatives from environmental non-governmental organisations (eNGOs), fisher organisations, research institutes and national authorities) before specific consultations were held with the most relevant industry organisation. Once consultations were completed, the draft proposal was reviewed by lawmakers and their representatives, and was scientifically peer reviewed, to ensure the proposal worked within domestic law and was based on the best available knowledge.

After the domestic draft proposal was finalised, Danish representatives held a series of bilateral meetings and/or corresponded electronically with other EU Member State representatives in Sweden, Germany, Poland, Finland, Estonia, Latvia and Lithuania while consulting with the European Commission, namely the Directorate-General for Maritime Affairs and Fisheries (DG MARE) and the Directorate-General for Environment (DG ENV), the primary fisheries advisory institute (International Council for the Exploration of the Sea, ICES), and the regional industry/stakeholder organisation (Baltic Sea Advisory Council). The result was a proposal for a Commission regulation brought forward by Denmark, explicitly backed by Germany, Sweden and Poland, and a priori accepted by Finland, Estonia, Latvia and Lithuania. Final consideration and consultation were facilitated via an in-person meeting in Brussels.

2.4. History of fisheries

As part of the consultation with fisher organisations and other Member States, an analysis of fisheries in the AGRB MPA area was undertaken using data submitted from Denmark, Sweden, Estonia, Germany and Poland. Utilising vessel monitoring system (VMS) and logbook data from all countries for vessels over 12 m, and automatic identification system (AIS) data for smaller vessels from Denmark and Sweden, data on types of fishery were collected and collated (Danish AgriFish Agency, 2016).

Denmark and Germany had carried out the majority of MBCG fishing activities inside the MPA area or nearby, prior to the SAC designation and prior to the MBCG regulations, with Sweden, Poland and Estonia each having small amounts of activity. The average annual value of MBCG landings varied from ~EUR 13 000 for Denmark to ~EUR 3,000 for Estonia (from the whole C-square-based area, for the period 2013-2017). Overall, this represents relatively low values but with peaks in value throughout the time series (e.g. ~EUR 47,500 from Denmark in 2012).

As part of a fundamental analysis of the AGRB MPA undertaken by the Danish environmental agency in 2020 (Miljøstyrelsen, 2020), a comparison of Danish fishing activity before and after the fisheries regulations came into force was presented. This analysis used the year 2018 as the start of the 'after' period and included 2017 in the before period. With only one year's data in the after period, this analysis illustrated that in 2018 only minor incursions of MBCG into the AGRB MPA were observed (Figure 3), and large sections that had been fished in the period 2013-2017 were no longer being fished

(Miljøstyrelsen, 2020). A systematic review of primary literature found no relevant literature on fisheries in the AGRB MPA (see Annex 4 of main report for detailed methodology).



Figure 3. Comparison of MBCG fishing intensity in the AGRB MPA in 2013–2017 (inclusive) and in 2018, indicative of the periods before and after regulations came into effect.

NB: Top panel= 2013–2017 (inclusive); bottom panel= 2018. Figures and underlying analyses carried out by National Institute of Aquatic Resources, the Technical University of Denmark and presented in the base analyses for the MPA by the Danish Ministry (Miljøstyrelsen, 2020).

3. AIMS AND OBJECTIVES

In this case study, we investigate whether the designation of the MPA as a SAC or the subsequent fisheries regulations within this region have affected fisheries effort and

landings within the MPA and if so, whether the impacts were specific to particular habitat types.

To address this, we targeted a series of specific objectives:

1. Attain high-resolution habitat maps for the AGRB MPA and surrounding areas and use these data to identify suitable contiguous control areas;
2. Create a custom national data call to attain high-resolution data on fisheries effort and landings (mass and value) relevant to the MPA and control sites;
3. Design and execute a series of post-hoc Before-After, Control-Impact (BACI) experiments to determine whether the designation as a SAC in 2016 or the subsequent fisheries regulations in 2017 affected fishing effort or landings;
4. Design and execute a series of post-hoc experiments to determine whether the designation as a SAC or subsequent fisheries regulations caused a change in habitats that were fished; and
5. Contextualise the results of the findings from aims 3 and 4 in the broader policies and management of marine space in the south-western Baltic Sea.

4. METHODOLOGY

4.1. Habitat maps

National high-resolution data on benthic habitats was collated and shared via the European Marine Observation and Data Network (EMODnet) platform in various formats. Therefore, for consistency within the project and for familiarity outside of the project, output from the 2021 EUSeaMap broad-scale predictive model, produced by EMODnet Seabed Habitats⁽²⁾ was utilised in this case study (Vasquez et al., 2021). The use of this model output also allows for comparisons with other sea-bed habitat classification systems and the European Nature Information System (EUNIS) 2012 classification system was utilised⁽³⁾. More detailed metadata and downloadable data products are available online⁽⁴⁾.

Initial maps of the habitats in the broader area were considered together with habitat maps of both the entire AGRB MPA and the MBCG regulated areas (Figure 4). Based on these maps, trade-offs were made to select control areas that were (as far as possible) contiguous, adjacent / close to the AGRB MPA/MBCG areas and covered habitat types as similar as possible to their corresponding AGRB MPA/MBCG sites.

4.2. Fisheries data

Four sources of data were aggregated to provide estimates of effort, landings mass and landings value:

- VMS data was obtained from the Danish Fisheries Agency. Vessels larger than 12 metres were required to have VMS as of 2012 in the Baltic Sea. The frequency of information from this data source is typically one ping per hour.
- AIS data was retrieved from a publicly available database⁽⁵⁾ for all Danish vessels carrying a transponder. The typical sending frequency is one ping per minute. Here the anonymous Maritime Mobile Service Identity (MMSI) number is coupled the vessel harbour number.

- Sales notes data containing information on the amounts landed and initial auction prices were obtained from the Danish Fisheries Agency.
- Vessel logbook data reporting on trip times, fishing times, gears used per trip and vessel specifications (e.g. length and engine power) were obtained from the Danish Fisheries Agency.

The following procedures were undertaken to align and merge the above data sources and to generate estimates of fishing effort and landings by vessel and gear at points along fishing tracks, in time.

- VMS and AIS data were aligned to increase the frequency of locations for each VMS trip. If there were data gaps, or if a vessel only had VMS, a non-linear interpolation based on heading between pings was made to estimate the track. Sailing / non-harbour points were then determined by overlapping the points with polygons of the Danish harbours and the sequence of points going in and out of the harbour polygon used to assign a trip ID.
- Fishing operations for the identified trips were identified primarily from speed filters, where the ranges of minimum and maximum speeds were determined by expert judgment in collaboration with the ICES Working Group on Spatial Fisheries Data (WGSFD).
- The sales notes and logbook data were combined by landing date, so the amounts and values for each species were coupled to a reported fishing trip. In some cases, the logbook also contained the reported start and end of each fishing operation, which increased the accuracy of the information on fishing activities.
- The fishing operation identified from the positions data and combined sales notes and logbook data were combined hierarchically.

If there is a reported fishing operation start and end, and these timestamps align with timestamps of position data, then it is possible to match the reported catch for the given fishing operation to the relevant points. If only trip start and end is reported and are within reasonable time from the positions data identified for the trip, the catch is allocated evenly between all the points of an identified fishing operation. If only the landing date is available, the landing dates and fishing operations are aligned in time, and all the catch from the closest landing date, after the fishing operation, is assigned to these points.

Following the above alignment and merging of the various sources of fisheries information, fishing activities and reported landings were further aggregated to each habitat polygon within the AGRB MPA and MBCG ban areas, as well as their corresponding control areas. Aggregation to these polygons were made by year, quarter, and métier level 5. Effort was aggregated as both fishing days and fishing kW days. Landings were aggregated per polygon by species and reported in both kilograms and sales value (DKK).

4.3. Fisheries changes in response to MPA

4.3.1. Analytical Procedure

In order to attribute any changes in fishing activities to the SAC designation or the implementation of gear regulations, a series of BACI studies were instigated. This approach accounts for changes in the system outside of the SAC designation (Impact) by selecting a similar site in the same context (control) and comparing the fisheries activities between

and within sites, both before and after an intervention or significant change has occurred at the impact site; namely SAC designation or MBCG regulations coming into force. The traditional BACI compares only per-site means, from before and after the intervention. However, when time-series data are available, with even time steps between observations, we can extend the BACI approach to account for both changes in means and trends (henceforth referred to as a BACI Time Series (BACI TS) analysis). This approach provides a more detailed account of the relative changes in the modelled system, while also testing for the key assumption that the control and impact sites have the same trends in the before period. For a review of the applications of BACI and BACI TS (see Wauchope et al., 2021).

The same procedure was applied to all analyses, where only the response variable (the description of fishing activity) or the date at which the intervention took place (SAC designation or MBCG regulations imposed) changed.

All responses (effort or landings) were log-transformed prior to model fitting after observing increasing residual errors in preliminary models. This ensured the assumption of normality of errors in the residuals of the models was adhered to or was at least more realistic.

4.3.2. Modelling approaches

A BACI TS model was fit where the response (y) changed depending on the question being investigated:

$$y = \alpha + \beta_1 \cdot BA + \beta_2 \cdot CI + \beta_3 \cdot T + \beta_4 \cdot BA:CI + \beta_5 \cdot BA:T + \beta_6 \cdot CI:T + \beta_7 \cdot BA:CI:T \quad \text{Eq. 1}$$

Where α = intercept, BA is a binary before/after variable that changed from before to after, depending on either the year of SAC designation (2016) or the year in which the MBCG regulations came into force (2017). CI is a binary control/impact variable, T is a time variable (year in our case), and β s are the coefficients of each term. All interactions are included in the model and are represented by the ':' between the variables.

A significant coefficient of the CI:T interaction (β_6) indicates that the control and impact sites were on different trajectories before the change point, therefore precluding further interpretation of the BACI analysis.

A significant three-way interaction coefficient (β_7) indicates that there was a relative difference in the slope at the impact site relative to the control site, accounting for how the control and impact sites were progressing before the breakpoint. This is one of the two major terms that we were looking for, and indicates an effect of the SAC designation or subsequent MBCG regulation.

The second major term of interest in our study was the interaction between BA and CI variables (β_4). A significant coefficient here indicates that there was a change between the control and impact sites immediately after the SAC designation or MBCG regulations came into force, whichever the context of the BA variable.

A significant coefficient of the BA:T interaction (β_5) indicates a trend change in the control site, and thus can be interpreted as a general change independent of the MPA. This term was of less interest to us and had no direct bearing on how the MPA affects fishing activities.

The individual variables were of less importance for our comparisons. Where there are significant higher-order interactions, then the coefficients of these individual terms are conditional on the interaction and therefore cannot be directly interpreted.

The time series available for the spatially explicit fisheries data required in these analyses are relatively short and are not split evenly on each side of the intervention. For example, the data are only available for the period 2012–2022, the MPA was designated a SAC in 2016 and the fisheries regulations restricting the use of MBCG in parts of the MPA came into force in 2017. This leaves four and seven years before and after SAC designation, respectively. Likewise, there are only five and six years before and after MBCG regulations came into force, respectively. To deal with the unbalanced number of years, we utilised the REstricted Maximum Likelihood (REML) estimation function to estimate model parameters, which is robust to this imbalance. Furthermore, to ensure that we were not overfitting our model with so few data, we sought to select the most parsimonious model between the full BACI TS model (Eq. 1) compared with the simpler BACI model:

$$y = a + \beta_1 \cdot BA + \beta_2 \cdot CI + \beta_4 \cdot BA:CI$$

Eq. 2

Where the variables are the same and the main change comes from the removal of all of the time series components of the model. In this simplified BACI model, the interpretation of the interaction between BA and CI (β_4) changes, such that a significant coefficient now represents a difference in the mean values (integrated across all within-group years) between the impact and control sites, after the intervention, relative to their difference before the intervention.

Prior to model selection, the individual model fits were interrogated by way of visual inspection of standardised residuals and tests were run to detect any significant deviations from the assumed response distribution, changes in residual variation or whether outliers were present. If both models had acceptable fits and no major trends in residuals, model selection was then done by selecting the model with the smallest Akaike Information Criterion (AIC) (Burnham and Anderson, 2007). We used the same procedure to retain one model out of the two different temporal contexts (SAC designation or MBCG regulation).

Once a final model was selected, it was interpreted according to the description of the coefficients above. Coefficients were deemed significant at the 0.05 level. No corrections to the accepted significance levels were required as no subsequent pair-wise tests were undertaken and all models were built to address independent hypotheses.

For each of the response variables: effort, landings mass, landings per unit effort and landings value, a broad to narrow series of analyses was undertaken. This included starting with data aggregated to include all gears and all species. Subsequently MBCG and gill nets were analysed independently, according to their importance in the context of the AGRB MPA.

4.4. Habitat use changes in response to the AGRB MPA

Using the same approach described for the analyses of fisheries changes generally, comparisons of BACI and BACI TS model parsimony were used to investigate changes in total fishing effort by habitat type. While habitat types were available in polygons with a high level of granularity (Figure 4), fisheries data were too scarce within these many categories. Therefore, an aggregation of habitat types into six categories was undertaken resulting in two physical descriptors, hard and soft bottom, together with three descriptors

of relative depth, namely euphotic (where the sea floor receives sunlight), shallow-shelf, and deep shelf. These re-categorisations were made by consulting the detailed habitat descriptions provided in EUNIS habitat types⁽⁶⁾.

Lastly, due to scarcity of fisheries data, only comparisons of effort from all gears combined were utilised in the analyses of habitat use by fishers.

4.5. Analytical tools

All data cleaning and analyses were undertaken in the R statistical programming language (R Core Team, 2021), accessed via R-studio (RStudio Team, 2021). Figures were plotted using the packages ggplot (Wickham, 2016), ggthemes (Arnold, 2021), viridis (Garnier et al., 2021), gridExtra (Auguie, 2017) and treemapify (Wilkins, 2021). Spatial data were manipulated and plotted as simple features using the package sf (Pebesma, 2018). Model fitting utilised glmmTMB (Brooks et al., 2017), residual checks were done with DHARMA (Hartig, 2018) and AIC based selection was facilitated by bbmle (Bolker and R Development Core Team, 2017).

5. RESULTS

5.1. Habitat maps

Detailed habitat maps of the whole AGRB MPA (SAC designated area) as well as the areas within the MPA with subsequent MBCG bans were constructed and complementary control areas selected (Figure 4).

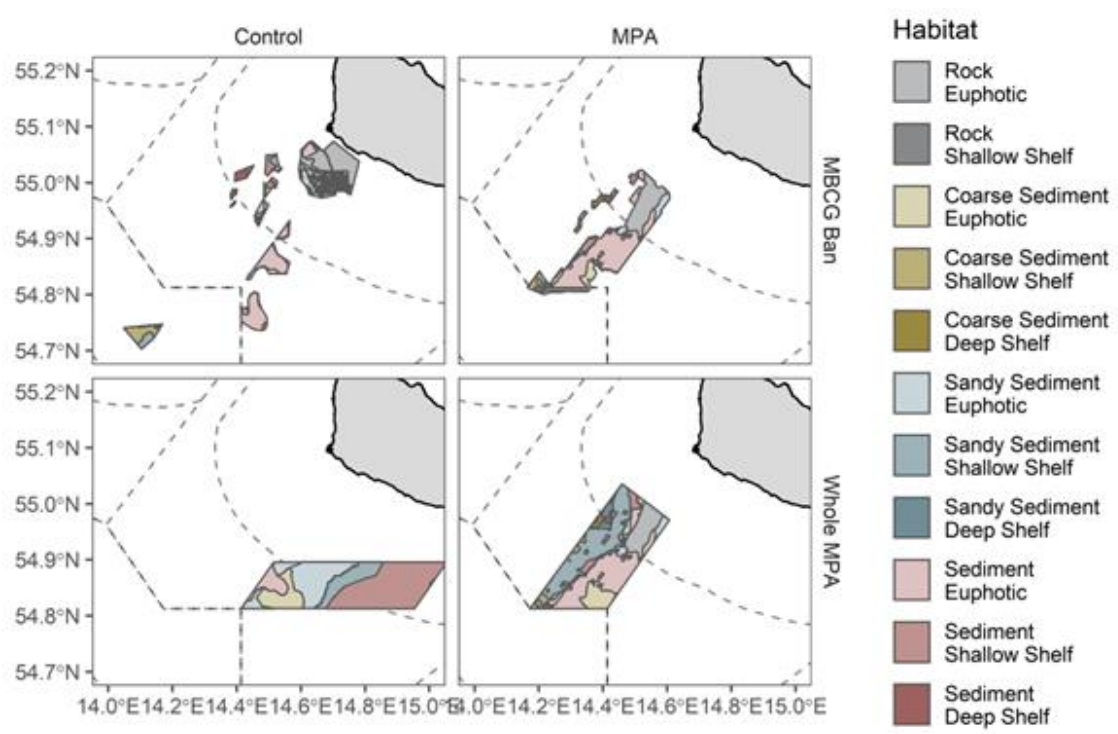


Figure 4. Site maps showing the MPA areas and the corresponding control areas selected for use in the subsequent BACI analyses for the whole AGRB SAC and the areas with bans on MBCG.

NB: MPA areas (right column) and the corresponding control areas selected for use in the subsequent BACI analyses (left column) for the whole AGRB SAC (bottom row) and the areas with bans on MBCG) (top row). Habitat descriptions are simplified descriptions of EUNIS 2012 habitat categories from the 2019 updated version (⁷).

For the whole AGRB MPA, one large contiguous area was selected as most representative. This area, as well as being adjacent to the MPA, contained the most similar make up of habitat types. For the MBCG area it was not possible to find a suitable contiguous area with similar habitat composition, thus a collection of areas surrounding the AGRB MPA with similar proportions of habitat type were selected. In both cases the trade-offs between these choices resulted in varying degrees of representativeness between habitats across the control and AGRB MPA areas (Table 1); however, the BACI methods employed in the analyses of fisheries activities are tolerant to differences, as they compare relative changes, rather than absolute values.

Table 1. Total areas (ha) of various habitat type aggregated by depth categories nested within substrate type (bold) across the paired control and MPA sites for the whole designated AGRB MPA and only those areas where MBCG bans are in place.

Habitat types	MBCG Ban		Whole MPA		Grand total
	Control	MPA	Control	MPA	
Coarse Sediment	3 358	2 736	3 582	4 010	13 686
Deep Shelf	108	256		336	699
Euphotic	1 973	1 741	3 582	2 813	10 109
Shallow Shelf	1 277	739		862	2 878
Rock	5 561	4 577	4	3 892	14 034
Euphotic	5 435	4 512	4	3 892	13 843
Shallow Shelf	126	65		0	191
Sandy Sediment	3 102	2 082	12 066	12 332	29 583
Deep Shelf	79	84		908	1 071
Euphotic	2 244	1 364	8 244	1 825	13 677
Shallow Shelf	780	634	3 823	9 599	14 835
Sediment	7 978	10 551	16 258	11 676	46 463
Deep Shelf	461	172		260	893
Euphotic	6 455	9 400	2 042	9 552	27 449
Shallow Shelf	1 062	979	14 216	1 864	18 121
Grand Total	19 999	19 946	31 911	31 910	103 767

5.2. Fisheries data

The aggregation of the four different sources of fisheries data led to the creation of four different data tables, containing data on:

1. fishing effort for the whole AGRB MPA and its control area;
2. fishing effort for the MBCG ban AGRB MPA and its control areas;
3. landings for the whole AGRB MPA and its control area and; and
4. landings for the MBCG ban AGRB MPA and its control areas.

The level of aggregation is described in the methods. These data were further processed to provide two more tables on landings per unit effort (LPUE).

5.3. Fisheries changes in response to MPA

5.3.1. Effort for the whole MPA

5.3.1.1. Effort of all gears

Before and after SAC designation

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 8.3$). However, none of the coefficients in the model were significant, indicating a valid BACI TS comparison with no significant changes after SAC designation, relative to changes in the control area.

Before and after EU regulations on MBCG

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 12.2$). There was a significant interaction of BA:CI:T, indicating a significant change in trend within the MPA, relative to the control area, after the implementation of MBCG regulations (Figure 5). There was no significant interaction between CI:T, indicating no significant difference in trends between the AGRB MPA and the control site prior to the imposition of MBCG regulations.

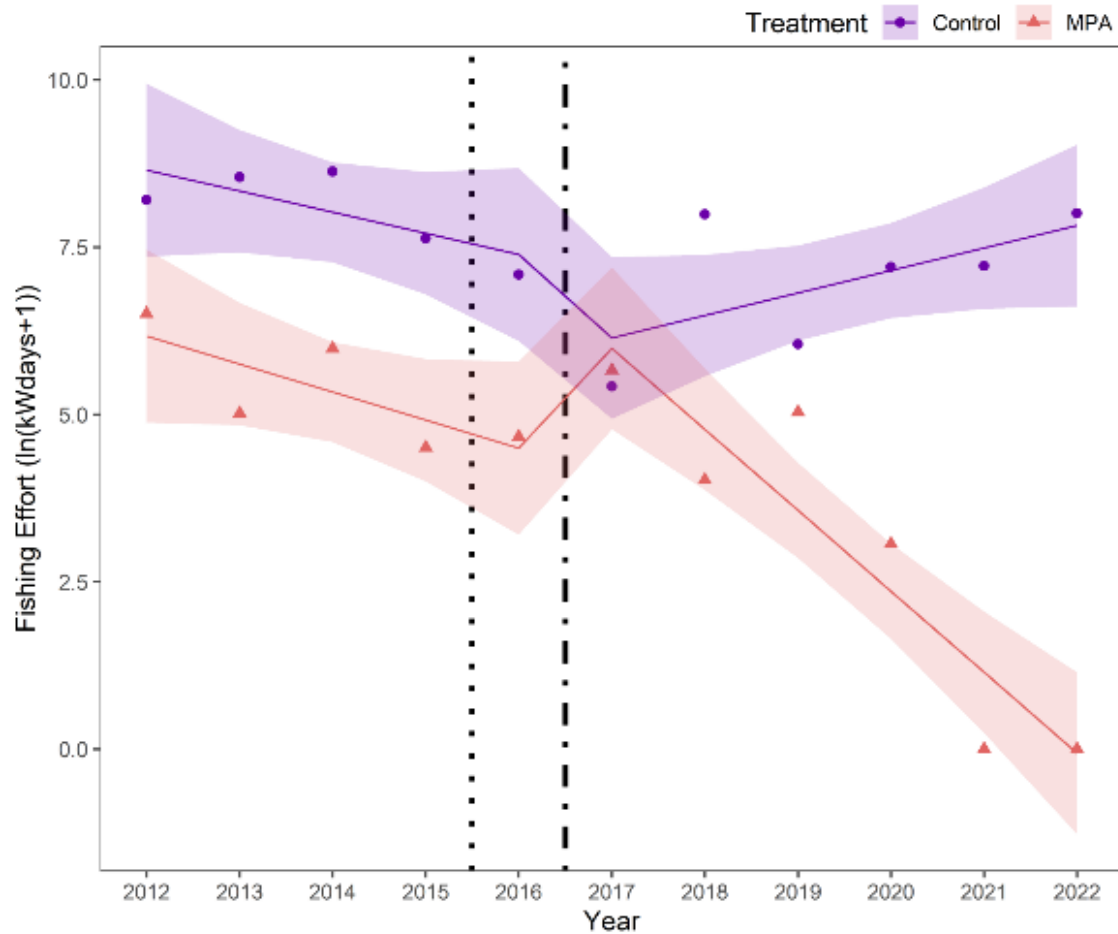


Figure 5. Total fishing effort (log transformed) over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations across both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is no difference in trend between sites before the MBCG regulations, but that there is a significant change in effort immediately after the regulations were imposed and that the trend in the MPA is significantly different from that of the control area after the MBCG regulations came into force.

5.3.1.2. Gear-specific effort

There was one dominant gear type per category of MBCG and non-MBCG, namely bottom trawls and gill nets respectively (Figure 6). The MBCG group consists entirely of bottom trawl gears, while the non-MBCG group also has contributions from pelagic trawls and some hook and line effort.

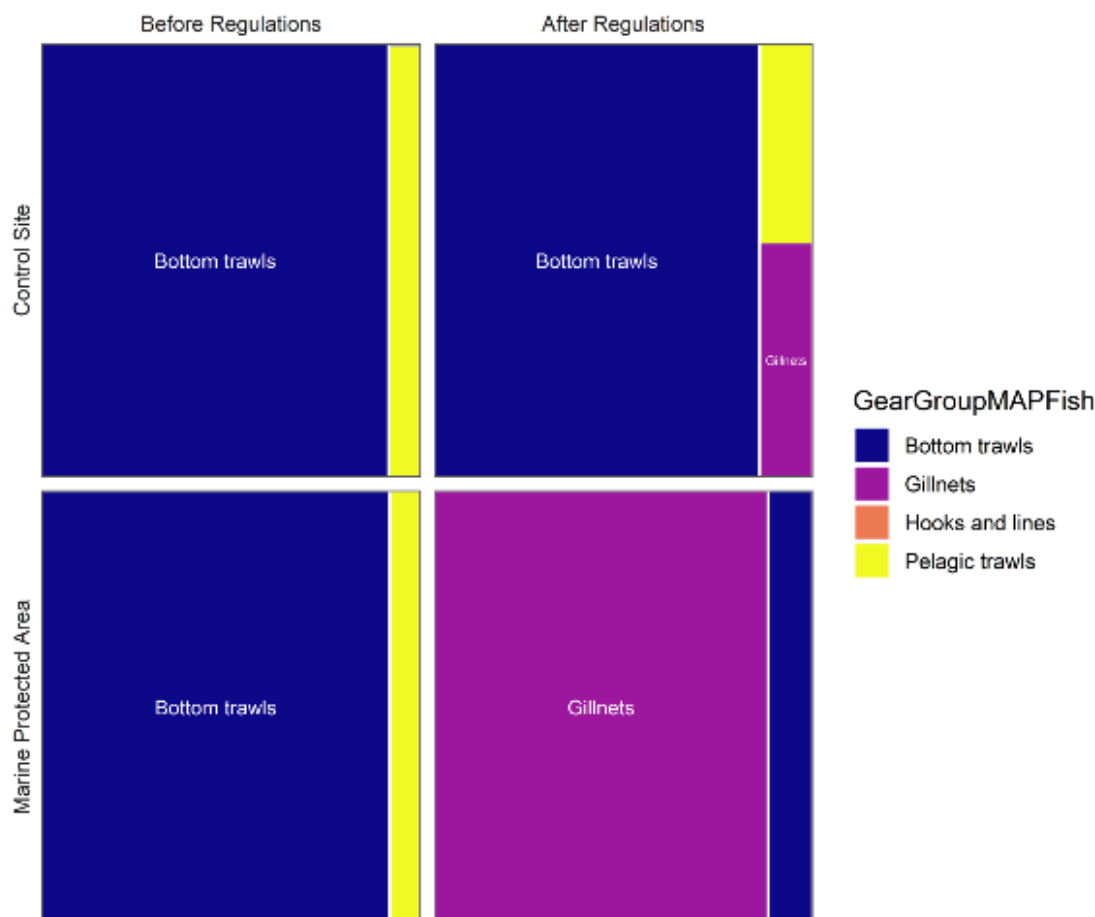


Figure 6. Treemap showing the relative contribution of different gear types to fishing effort, within each 'group' of the BACI analyses.

NB: The top-left panel represents the relative contributions of gears to the control site before regulations on MBCG came into force, while the bottom-right panel represents the relative contribution of gears to the full MPA area after the MBCG regulations were imposed. Colours illustrate the different gear groups that were active in the MPA and control areas over the study period.

5.3.1.3. Effort of mobile bottom-contacting gears

Before and after SAC designation

The simple BACI model was selected as the most parsimonious model ($\Delta AIC = 2.3$). There was a significant interaction of BA:CI, indicating a significant change in the mean annual effort of MBCG gears in the AGRB MPA compared to the control following designation. However, the model using pre- and post-fishing regulations better represented the data.

Before and after EU regulations on MBCG

The simple BACI model was selected as the most parsimonious model ($\Delta AIC = 4.9$). There was a significant interaction of BA:CI, indicating a significant change in the mean annual effort of MBCG gears in the MPA compared to the control following the implementation of MBCG regulations within the MPA (Figure 7).

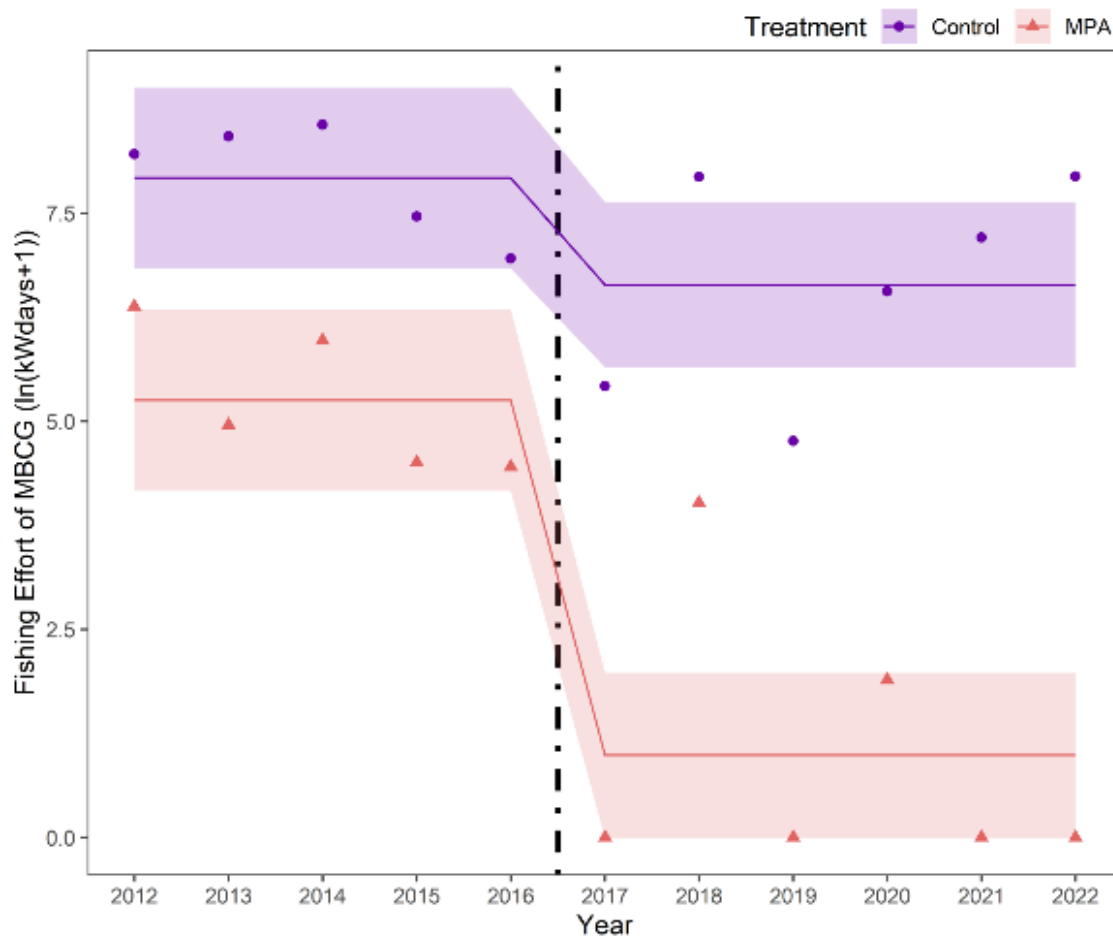


Figure 7. Fishing effort from MBCG (log transformed) over time, in the area that became the AGRB MPA (orange/triangles) and a matched control site (purple/circles).

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations across both MPA and control areas; ribbons indicate +/- 95% confidence intervals of the estimations. The full analytical model indicates that there is a significant difference in the difference of means between MBCG effort within the MPA, compared to that of the control area, from before to after the MPA was designated a SAC.

5.3.1.4. Gill net effort

Before and after SAC designation

Neither the full BACI TS nor the simplified BACI model produced satisfactory fits to the gill-net effort data. Therefore, we were unable to test whether any significant change occurred in response to SAC designation.

Before and after regulations on MBCG

The full BACI TS did not produce a satisfactory model fit and therefore could not be used to determine the more detailed effects of MBCG regulations on the use of gill nets in the AGRB MPA. The simplified BACI did provide an acceptable model fit; however, it only provided evidence of overall change in effort, with no differentiation between MPA and control sites (Figure 8).

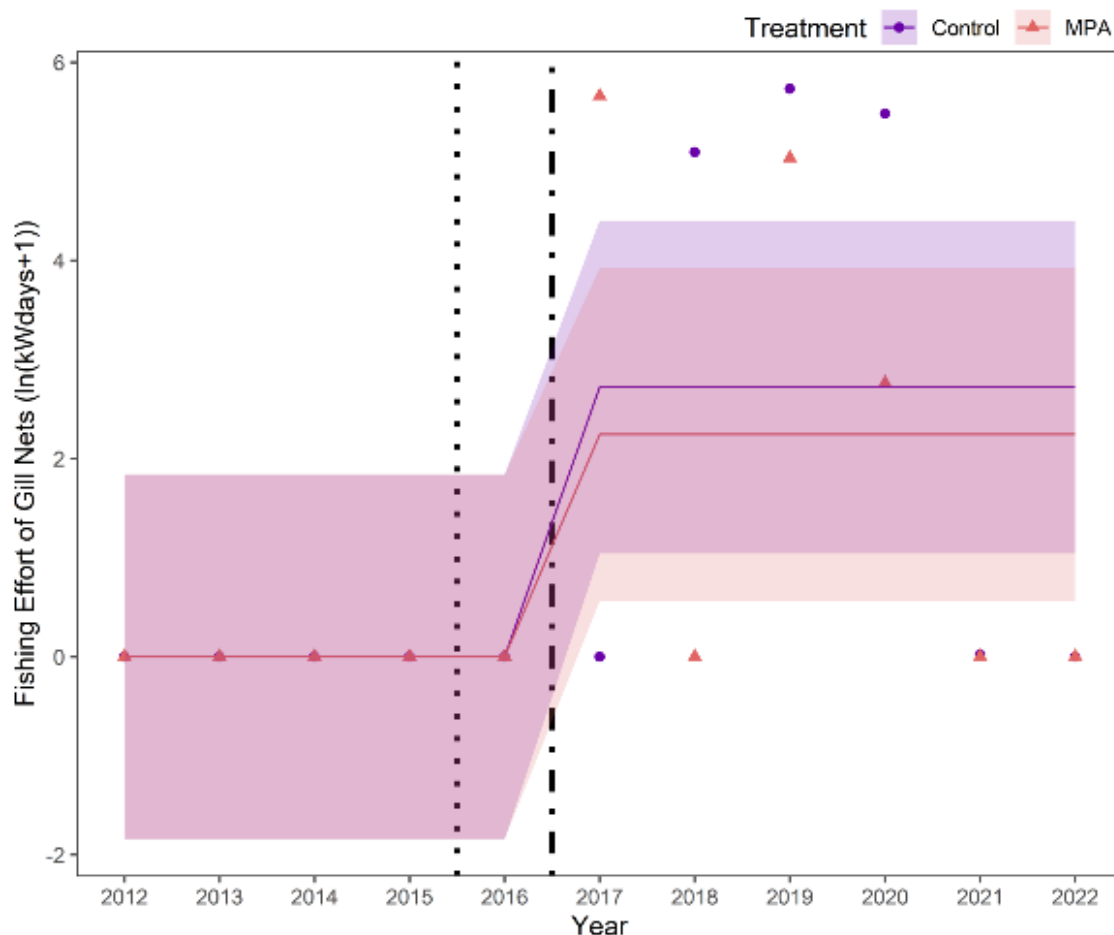


Figure 8. Fishing effort from gill nets (log transformed) over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations across both MPA and control areas; ribbons indicate +/- 95% confidence intervals of the estimations. The full analytical model indicates that there is no significant difference between the means across MPA and control sites. However, there is a significant difference in the pooled means before and after the imposition of the MBCG regulations, that is to say an overall increase in gill-net fisheries in both the MPA and control sites, with no difference between them.

5.3.2. Effort for MBCG regulated areas

5.3.2.1. Total effort combined

Before and after SAC designation

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 1$). However, none of the coefficients in the model were significant, indicating a valid BACI TS comparison with no significant changes in the MBCG regulated areas after SAC designation, relative to changes in the control area.

Before and after EU regulations on MBCG

As was found with the analysis considering SAC designation date, the full BACI TS model was selected as the most parsimonious model for the BACI centred on the MBCG regulation date ($\Delta AIC = 1$). This full model, also had no significant terms, indicating a valid BACI TS comparison with no significant changes in the MBCG regulated areas after EU regulation came into force, relative to changes in the control area (Figure 9).

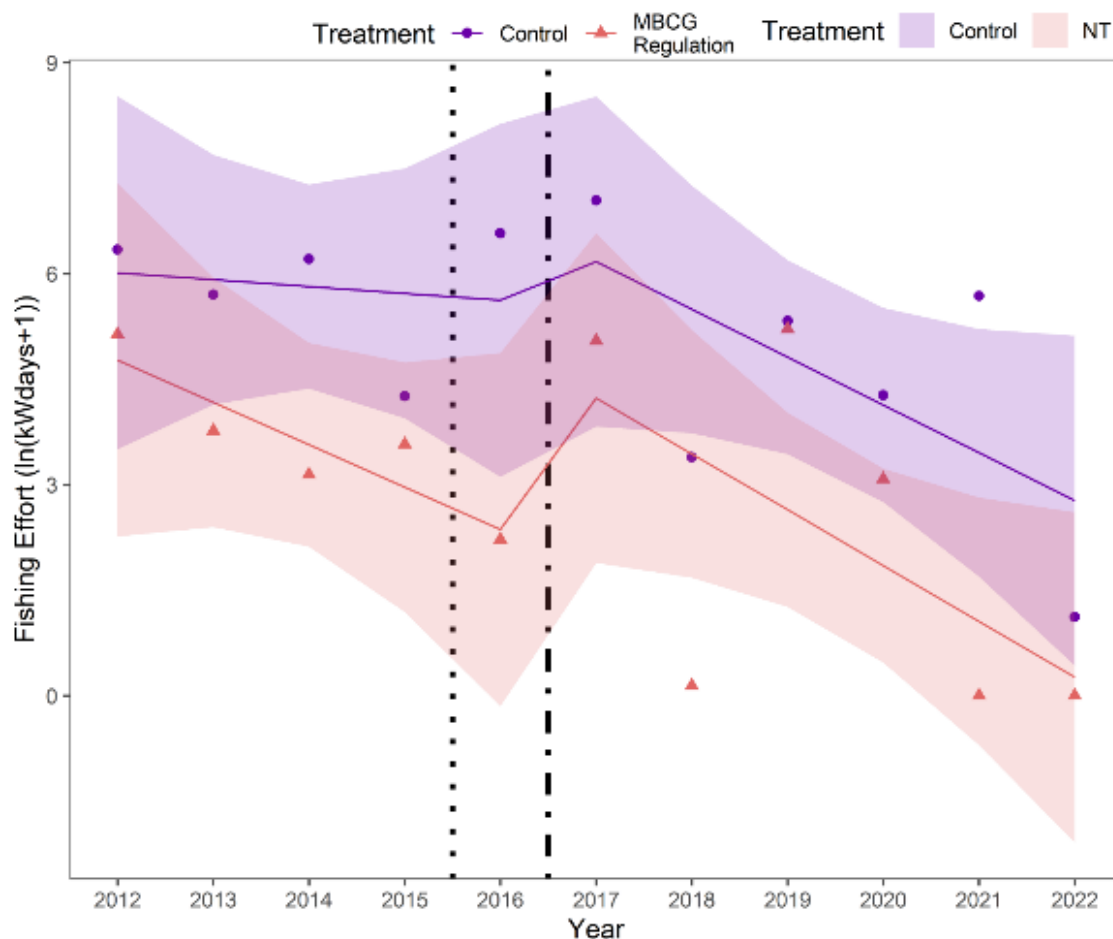


Figure 9. Fishing effort from all gears (log transformed) over time, in the specific area within the AGRB MPA that had MBCG restrictions imposed and matched control sites.

NB: Areas of the AGRB MPA that had MBCG restrictions imposed = orange/triangles; matched control sites = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the

reef areas within the MPA (2017). The trend lines are model estimations based on a break before and after the MBCG regulations across both the MBCG regulated part of the MPA (NT) and control areas; ribbons indicate $\pm 95\%$ confidence intervals of the estimations. The full analytical model indicates that there is no significant difference between the means across MPA and control sites, nor over time.

5.3.2.2. Gear-specific effort

The same dominance of gear types seen across the whole AGRB MPA (Figure 6) persisted in the areas in which the MBCG regulations were imposed (Figure 10). The gear MBCG group consists entirely of bottom trawl gears, while the non-MBCG group also has contributions from pelagic trawls and some hook and line effort.



Figure 10. Treemap showing the relative contribution of different gear types to fishing effort, within each 'group' of the BACI analyses.

NB: The top-left panel represents the relative contributions of gears to the control sites before regulations on MBCG came into force, while the bottom-right panel represents the relative contribution of gears to the area where MBCG regulations applied, within the MPA, after they were imposed. Colours illustrate the different gear groups that were active in the MPA and control areas over the study period.

5.3.2.3. MBCG effort

Before and after SAC designation

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 4.3$). However, none of the coefficients in the model were significant, indicating a valid BACI TS comparison with no significant changes in the MBCG regulated areas after SAC designation, relative to changes in the control area.

Before and after regulations on MBCG.

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 5.4$), in spite of some deviations in residuals that were most likely caused by the immediate drop to 'zero' values after the model breakpoint (when MBCG regulations were imposed). Based on the BACI TS model, there was a significant overall negative trend in effort shared by both control and MPA sites. Although the before and after periods appear different, this cannot be attributed to MBCG regulations. While there appears to be some difference between MPA and control sites, this was not significant and there was no discernible change in the size of this difference over time or between periods before and after MBCG regulation (Figure 11).

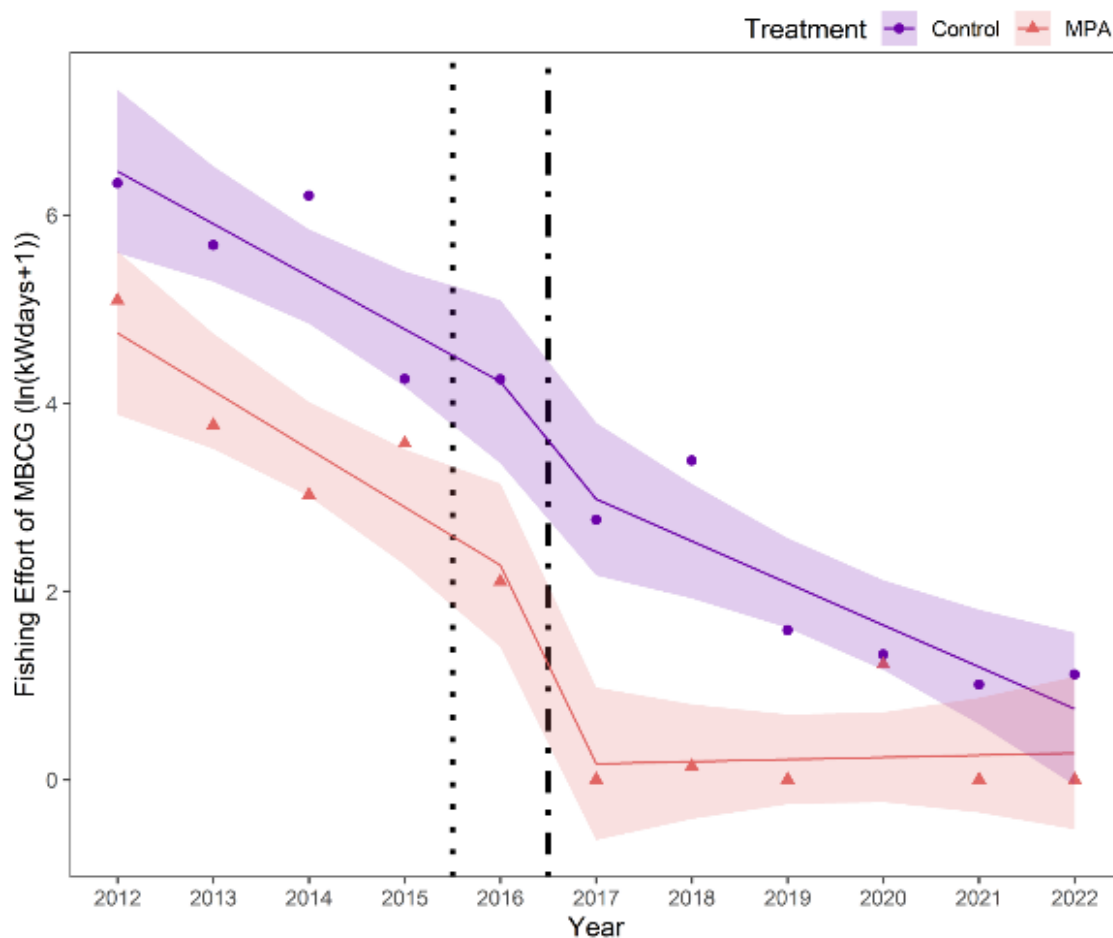


Figure 11. Fishing effort of mobile bottom contacting gears (log transformed) over time, in the specific area within the AGRB MPA that had MBCG restrictions imposed and matched control sites.

NB: Areas of the AGRB MPA that had MBCG restrictions imposed = orange/ triangles; matched control sites = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates

when the site was designated a SAC (2016), and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). The trend lines are model estimations based on a break before and after the MBCG regulations across both MPA and control areas; ribbons indicate +/- 95% confidence intervals of the estimations. The full analytical model indicates that both MBCG regulated sites and the control sites share a significant change over time and that there is no difference in this trend before and after regulations were imposed, such that the change cannot be directly attributed to the imposition of regulations.

5.3.2.4. Gill-net effort

Before and after SAC designation

The simple BACI model was selected as the most parsimonious model ($\Delta AIC = 1.3$). There was no significant interaction between the spatial (MBCG trawl ban and control) and temporal (before and after SAC designation) components of the model, and only a significant effect of the categorisation of before and after SAC designation (Figure 12). Therefore, we can infer that while gill-net fishing effort increased in the area after the MPA was designated a SAC, this change cannot be attributed to the SAC designation as it occurs simultaneously in the control areas.

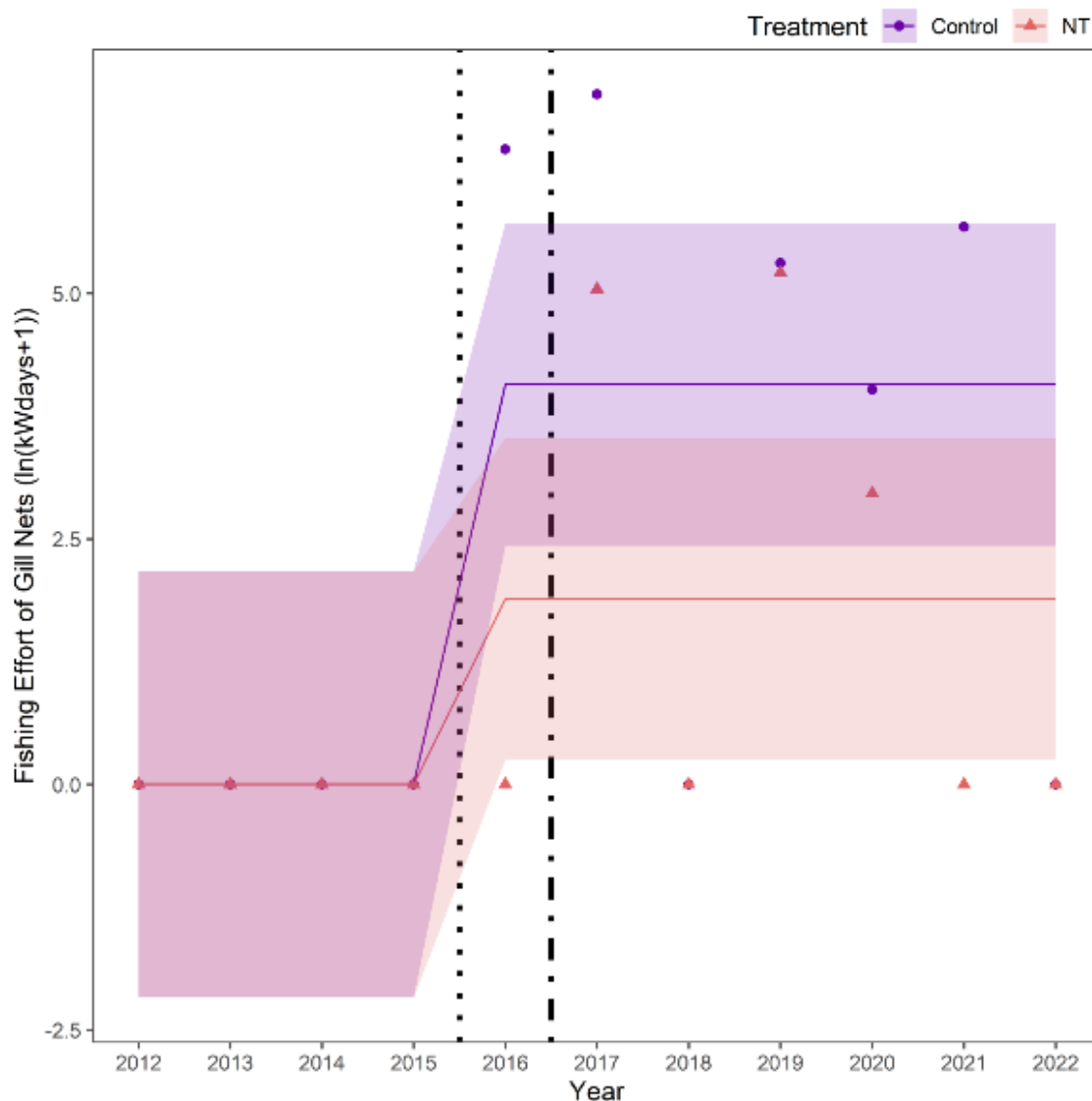


Figure 12. Fishing effort from gill nets (log transformed) over time, in the areas of the AGRB MPA (orange/triangles) that had MBCG restrictions imposed in them, and matched control sites (purple/circles).

NB: Areas of the AGRB MPA that had MBCG restrictions imposed = orange/ triangles; matched control sites = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the SAC designation for both the MBCG regulated part of the MPA (NT) and control areas; ribbons indicate $\pm 95\%$ confidence intervals of the estimations. The full analytical model indicates that there is no significant difference between the means across MPA and control sites. However, there is a significant difference in the pooled means before and after the area was designated a SAC.

Before and after regulations on MBCG.

The simplified BACI model was retained, as the full BACI TS model had large patterns in the residuals. These patterns were likely driven by the $\sim$ zero values for the gill-net effort in both MBCG regulated and control areas, before the MBCG regulations were imposed

(Figure 12). The retained model had no significant coefficients and is, therefore, unable to indicate any differences between the four contexts.

5.3.3. Species composition in landings

5.3.3.1. Species composition from the whole MPA

Landings from the MPA site were much lower than from the control site (Figure 13). These large landings in the control site can be attributed primarily to sand eels, and to a lesser extent Atlantic cod and some small catches of Atlantic herring and sprat. Sand eels were not as significant in landings from the MPA site, where they were landed in relatively small quantities before MBCG regulations were in force. Conversely, Atlantic cod played a larger role proportionally in landings from the MPA site, with minor contributions from Atlantic herring and sprat.

To view the relative differences within species across the four different contexts (before/after: AGRB MPA/control), landings values were log transformed. This allows us to view the relative changes across all species on the same scale but also makes comparisons between species more complex. While there were some increases of Atlantic salmon (< 1 kg, 1 064 %), turbot (122 kg, 288 %), and European plaice (1 219 kg, 90 %) within the MPA after the MBCG regulations were in force, landings of most species decreased (Figure 14).

5.3.3.2. Species composition from the MBCG regulated area

Within the MBCG regulated area and the respective control sites, Atlantic cod and herring dominate the landings (Figure 15), as opposed to the sand-eel-dominated landings of the whole AGRB MPA and its control (Figure 13). With an apparent decrease in landings of all species from MBCG fishing within the MBCG regulated areas, there appears to be an increase in landings from non-MBCG fishing. To visualise the differences between control and impact in an alternative way, the log transformed landings are shown in Figure 16.

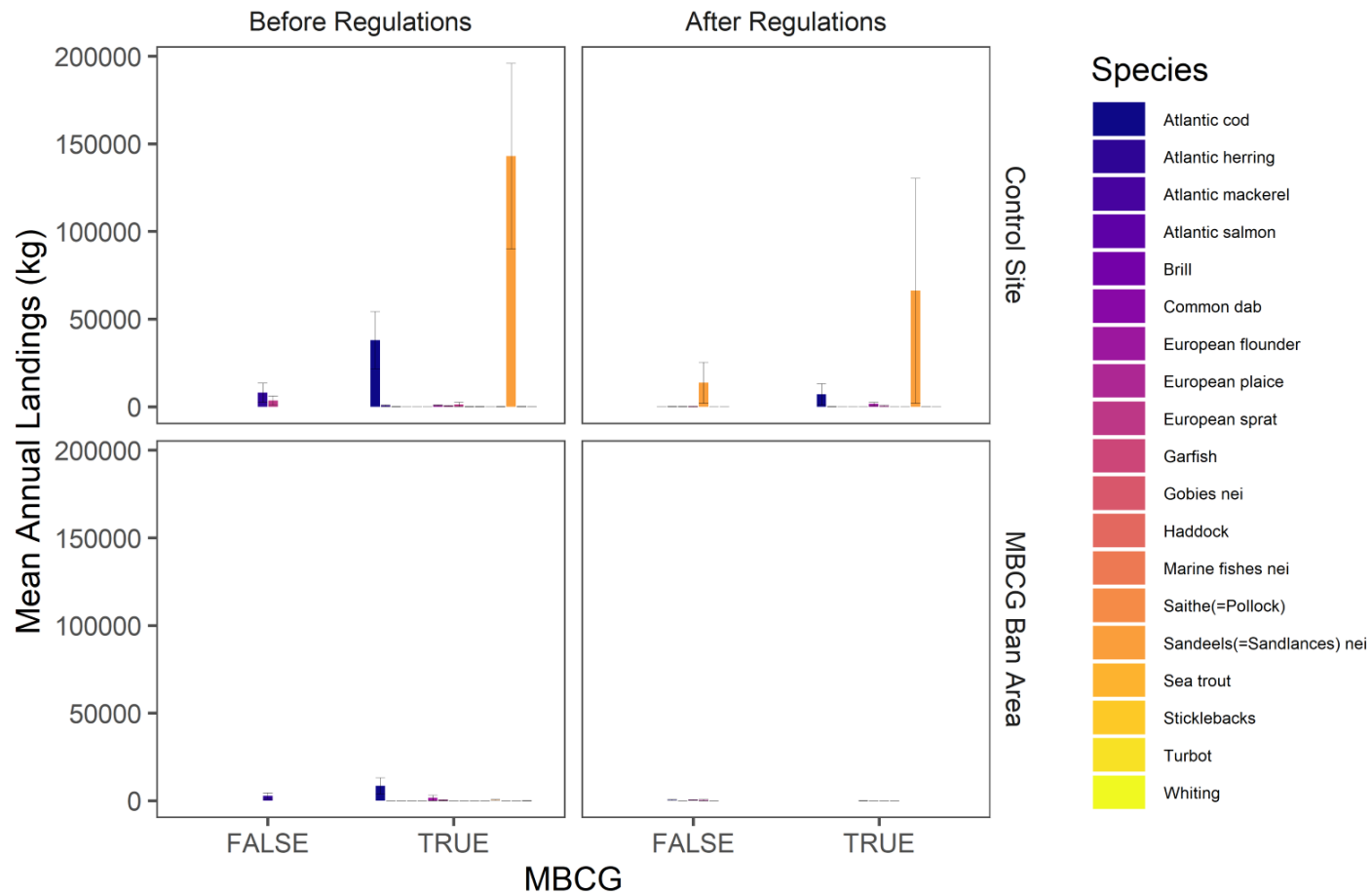


Figure 13. Mean annual landings (kg) of different species grouped by the gear type used to harvest them (MBCG or not = TRUE or FALSE, respectively) and plotted out in four different contexts.

NB: For example, in the top-left panel, landings are from the control site before MBCG regulations came into force within the MPA, while the bottom-right panel has landings from within the MPA after the MBCG regulations came into force.

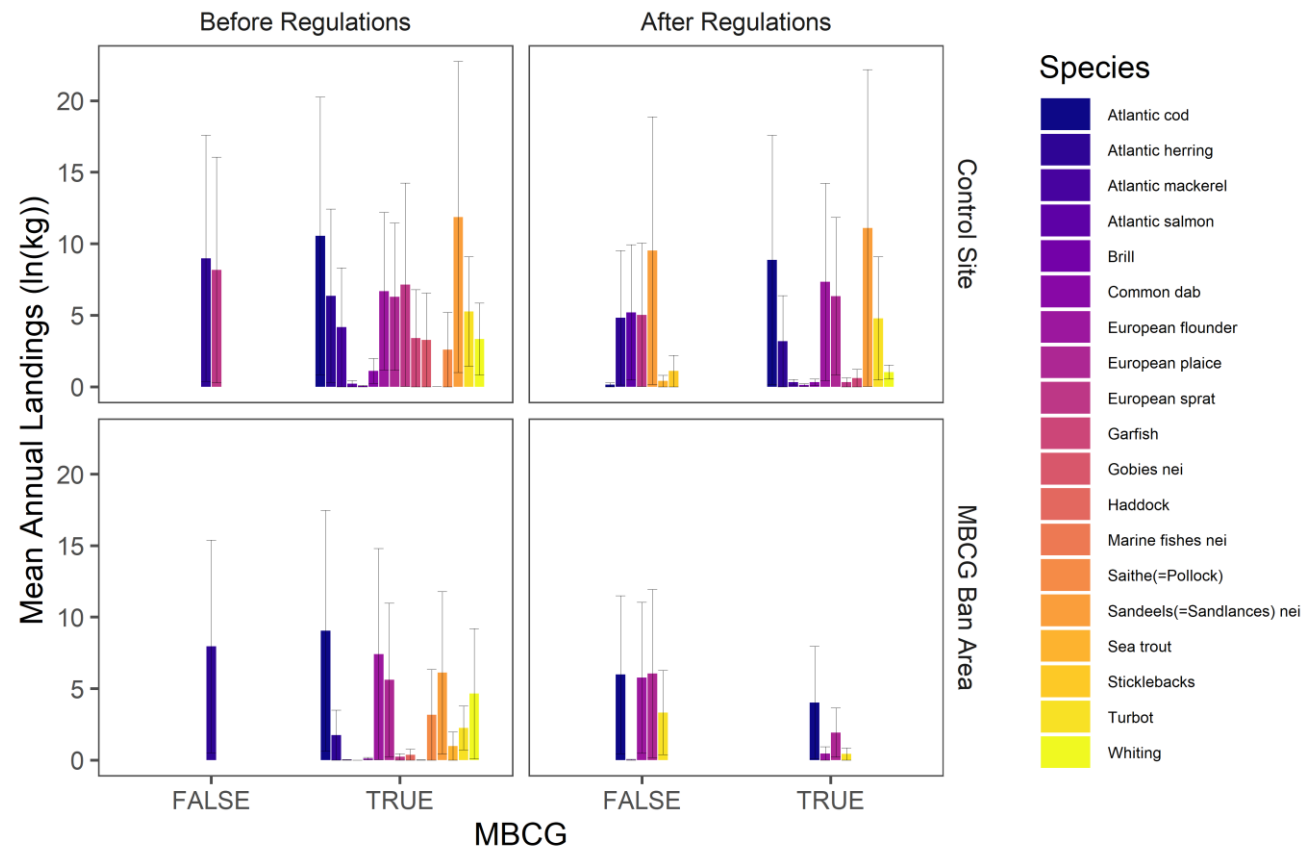


Figure 14. Natural log transformed landings, of different species (or species groups, coloured bars) from the different time and treatment groups.

NB: In the top-left panel, landings are from the control site before MBCG regulations were imposed within the MPA, while the bottom-right panel has landings from within the MPA after the MBCG regulations took force. TRUE indicates landings that come from MBCG, while FALSE come from other gears (predominantly gill nets). Landings are reported as mass, wet-weight in kilograms.

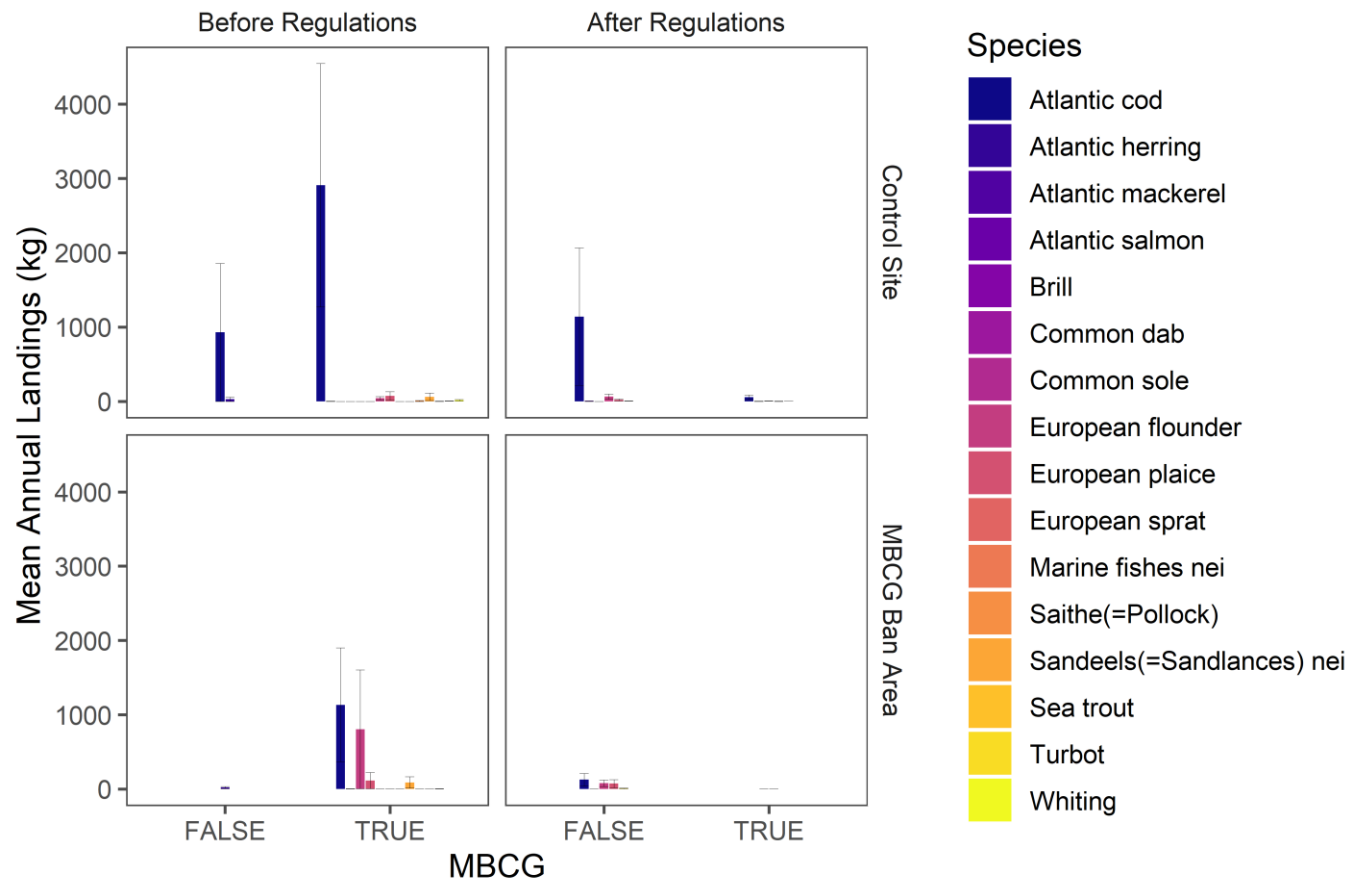


Figure 15. Mean annual landings (kg) of different species (colour) grouped by the gear type used to harvest them(MBCG or not = TRUE or FALSE, respectively) and plotted out in four different contexts.

NB: For example, the top-left panel represents landings from control sites prior to MBCG regulations coming into force, while the bottom-right panel represents the regulated area, after the MBCG regulations came into force.

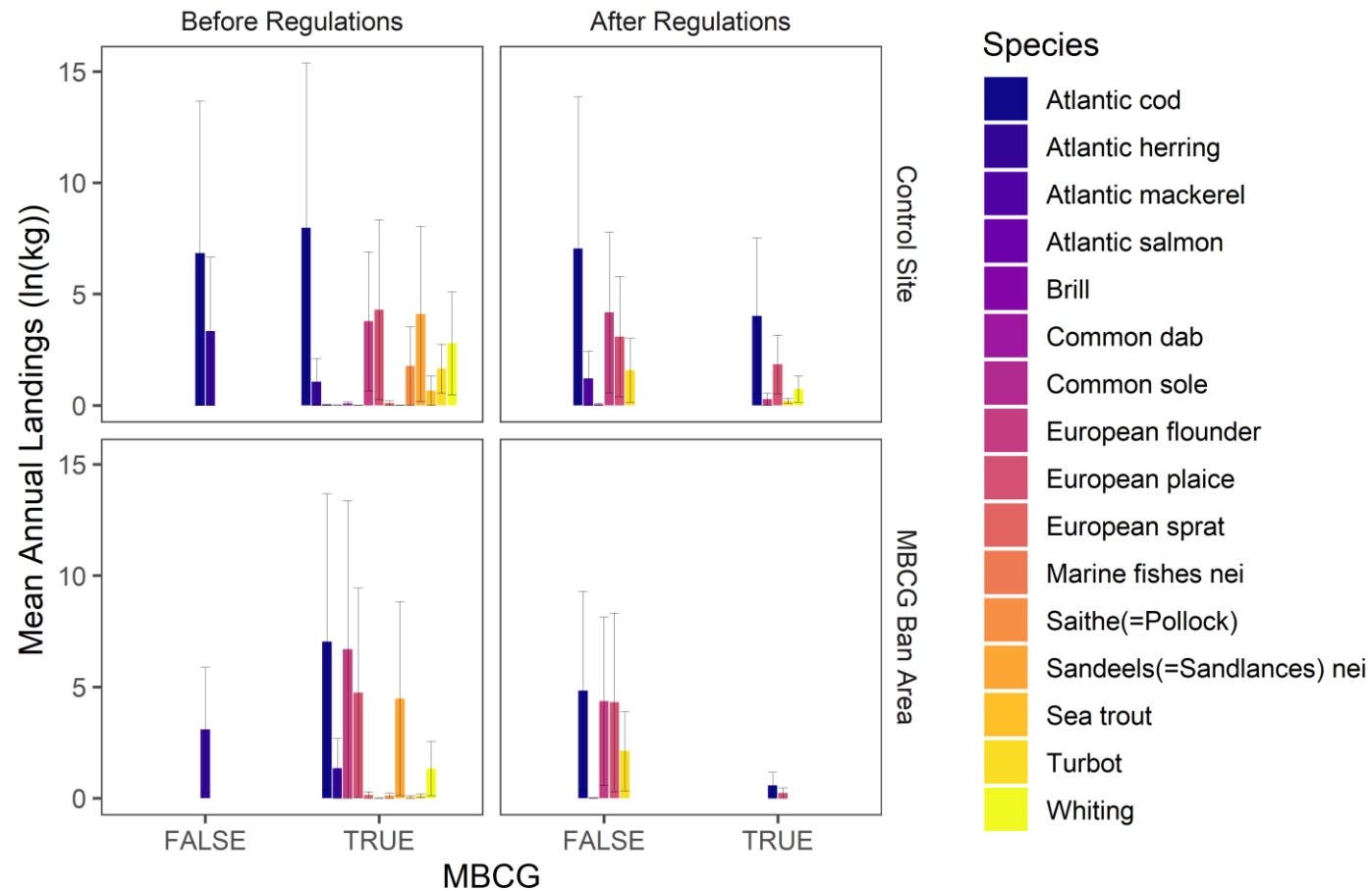


Figure 16. Mean annual landings (kg; log transformed) of different species (colour) grouped by the gear type used to harvest them, MBCG or not, and plotted out in four different contexts.

NB: For example, the top-left panel represents landings from control sites prior to MBCG regulations coming into force, while the bottom right panel represents the regulated area, after the MBCG regulations came into force

5.3.4. Total landings for the whole MPA

5.3.4.1. All gears and species

Before and after SAC designation

An analysis of total landings mass (for all species) found that the full BACI TS was the best model for comparing across the year in which the site was designated ($\Delta AIC = 12.5$). However, there were no significant terms in this model, indicating a valid BACI TS but no effect of designation on landings.

Before and after EU regulations on MBCG

The full BACI TS model was also the most parsimonious for the pre-post MBCG regulation scenario ($\Delta AIC = 12.4$). There was a significant three-way interaction of the before/after, control/impact and time step variables, indicating that there was a significant change in slope in the landings from the AGRB MPA after the MBCG regulations came into in force, relative to the changes in the control area. There was also a significant interaction between the before/after and control/impact variables, which we can interpret as there being an immediate change in the landings straight after the MBCG regulations were imposed (Figure 17).

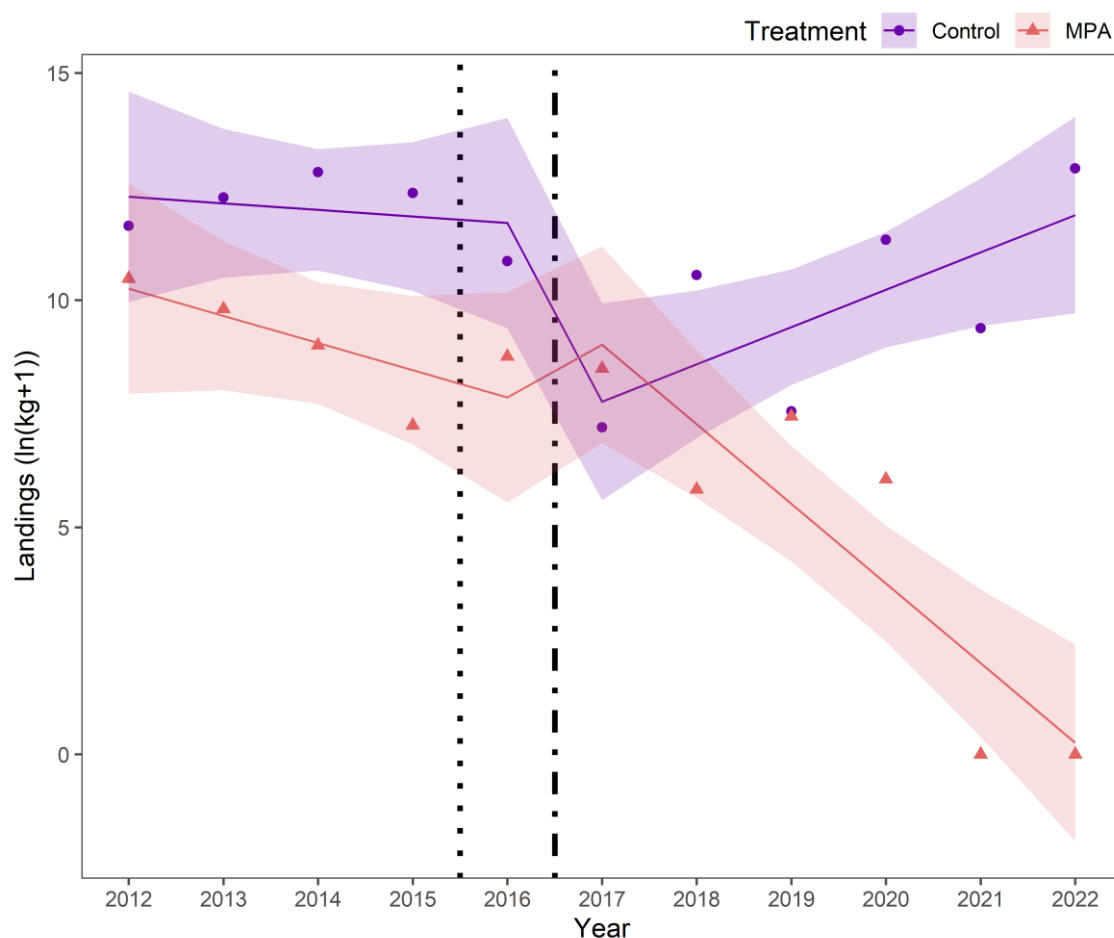


Figure 17. Annual landings mass of all species (log transformed) over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and

after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016), and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the SAC designation for both MPA and control areas; ribbons indicate $\pm 95\%$ confidence intervals of the estimations. The full analytical model indicates that there is a significant change in landings immediately after the regulations came into force and that the trend in the MPA is significantly different from that of the control area after the MBCG regulations came into force.

5.3.4.2. Gear-specific landings

Bottom trawl landings appear to decrease across both the control and MPA sites after MBGC regulations are imposed (Figure 18). Pelagic trawls in the control area appear unchanged from before to after, and contribute only minor relative landings in the MPA area. Landings from gill nets are only present after MBCG regulations are imposed, but are present in both control and MPA sites. However, gill nets contribute very little to landings relative to the other two gears.

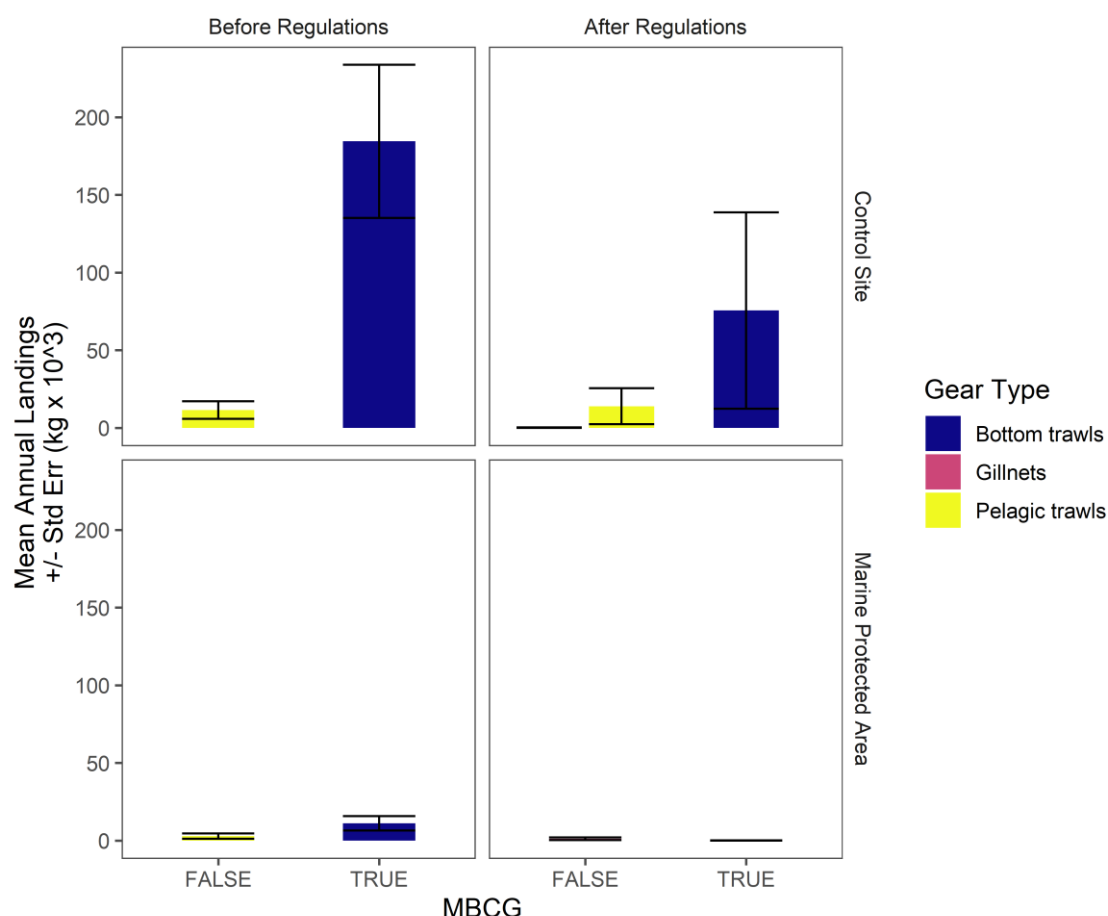


Figure 18. Mean annual landings caught by different gear types categorised as MBCG or not.

NB: These means are calculated for four different contexts, before and after MBCG regulations came into force (columns) and in the areas in which the MBCG regulations apply vs some matched, non-MPA, control areas (rows).

5.3.4.3. MBCG and all species

Before and after SAC designation

An analysis of total landings mass (for all species) found that the simplified BACI model was not an appropriate fit for the data, based on residual patterns, and therefore the full BACI TS model was retained. However, there were no significant terms in this model, indicating a valid BACI TS but no effect of SAC designation on landings.

Before and after EU regulations on MBCG

The simplified BACI model was retained based on model parsimony ($\Delta AIC = 0.8$). Based on the BACI model, there was a significant coefficient between the control/MPA and before/after regulation interaction term. This indicates that the difference between control and MPA sites was significant in the after period (Figure 19).

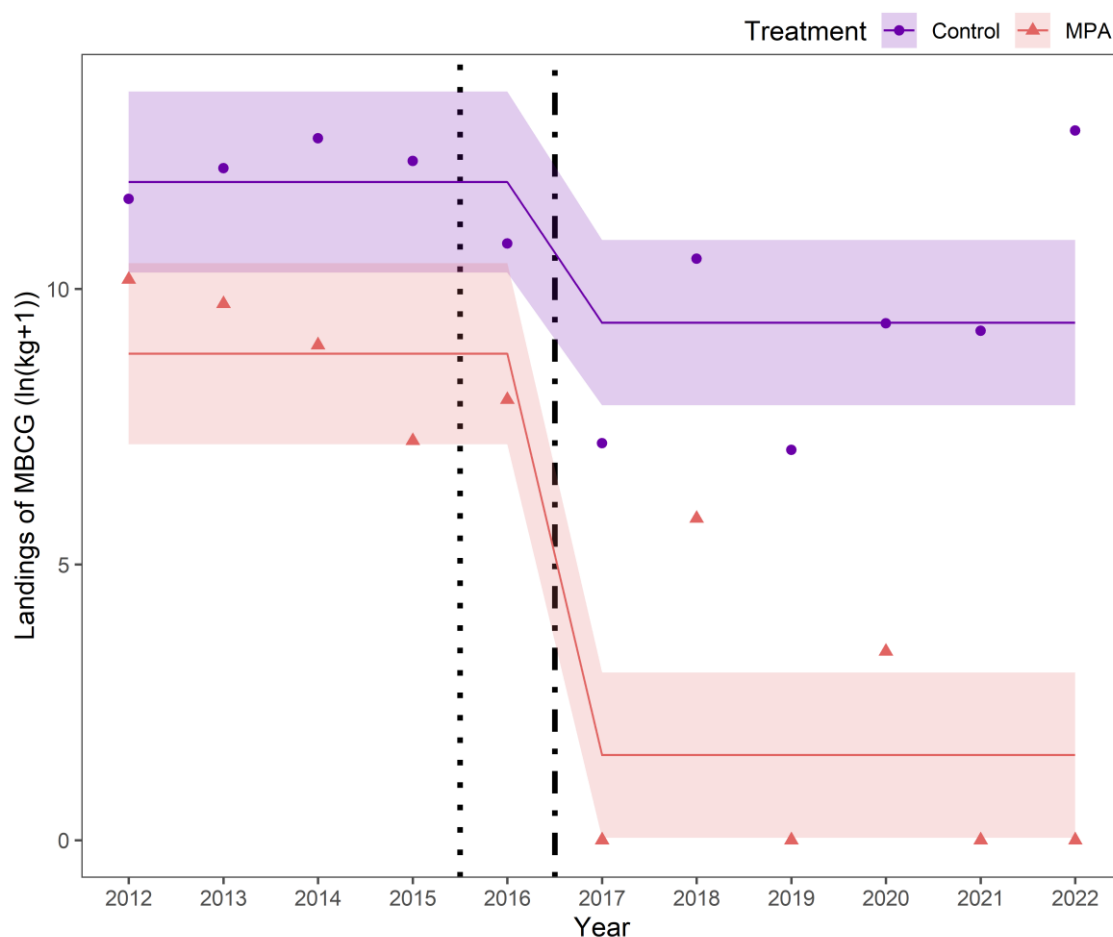


Figure 19. All species landings (kg, log transformed) from MBCG over time, in the specific area within the AGRB MPA that had MBCG restrictions imposed and matched control sites.

NB: Specific area within the AGRB MPA that had MBCG restrictions imposed = orange/ triangles; matched control sites purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCGs on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the SAC designation for both MPA and control areas;

ribbons indicate $\pm 95\%$ confidence intervals of the estimations. The full analytical model indicates that the difference between MPA and control sites (vertical distance between orange and purple lines) after the MBCG regulations were imposed is significantly different after the regulations came into force.

5.3.4.4. Gill nets and all species

Data from gill-net-only landings did not support the fitting of either the BACI TS or the BACI models, in either of the SAC designation date or MBCG regulation date contexts. Residual errors were unevenly distributed with strong patterns, indicating that models were unreliable and did not represent the data appropriately. However, it can be seen that landings from gill nets only started after MBCG regulations were imposed but eventually reduced to close to zero again, late in the time series (Figure 20).

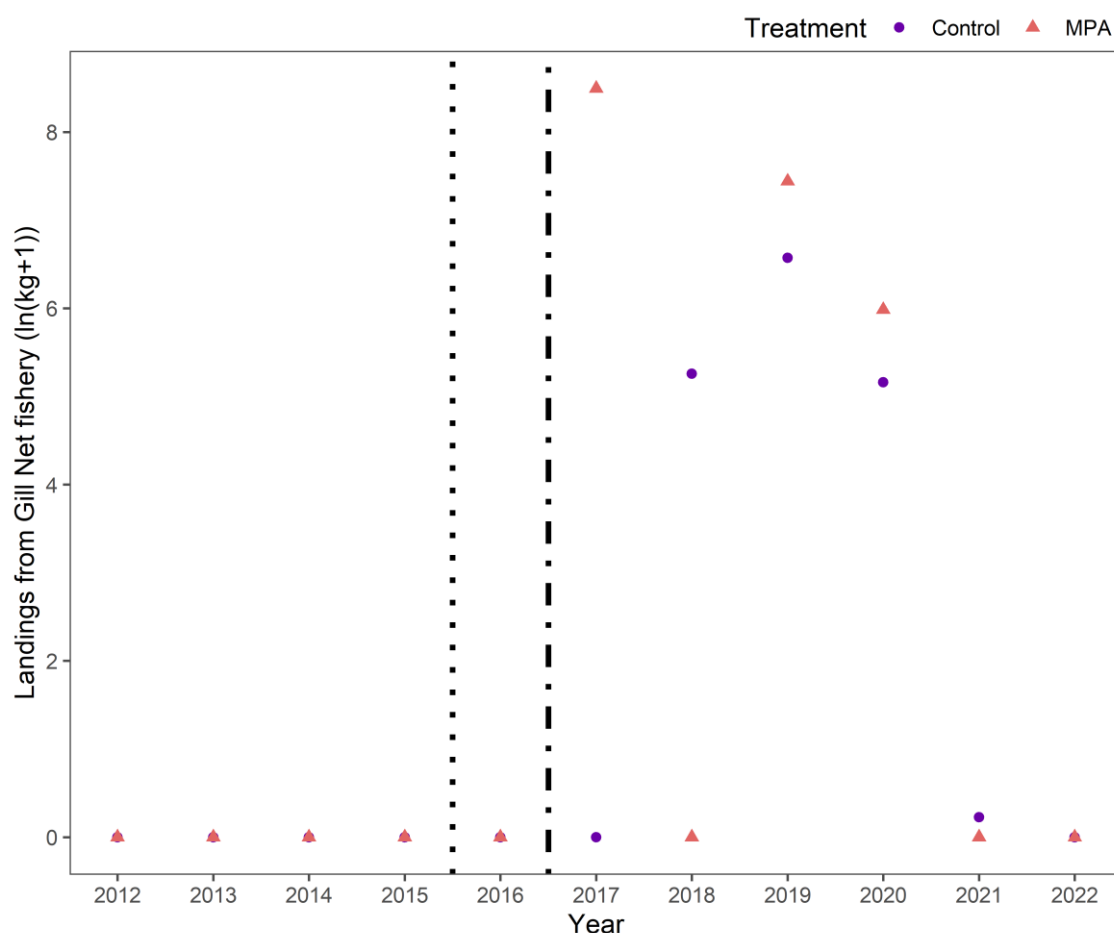


Figure 20. Landings mass of all species from gill nets (kg, log-transformed) over time.

NB: Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016), and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). No trendlines are displayed as no model could be reliably fitted to these data, and hence, no conclusions comparing before and after and the MPA vs control can be drawn, statistically.

5.3.5. Total landings for the MBCG regulated areas

5.3.5.1. All gears and species

Analyses of both contexts for Before/After, the designation year and the MBCG regulation year, the full BACI TS models were retained based on model parsimony ($\Delta AIC = 6$ and 5.3 , respectively). Neither model fit produced any significant interaction or first-order coefficients, indicating a valid BACI TS comparison, but no significant changes due to either intervention (Figure 21).

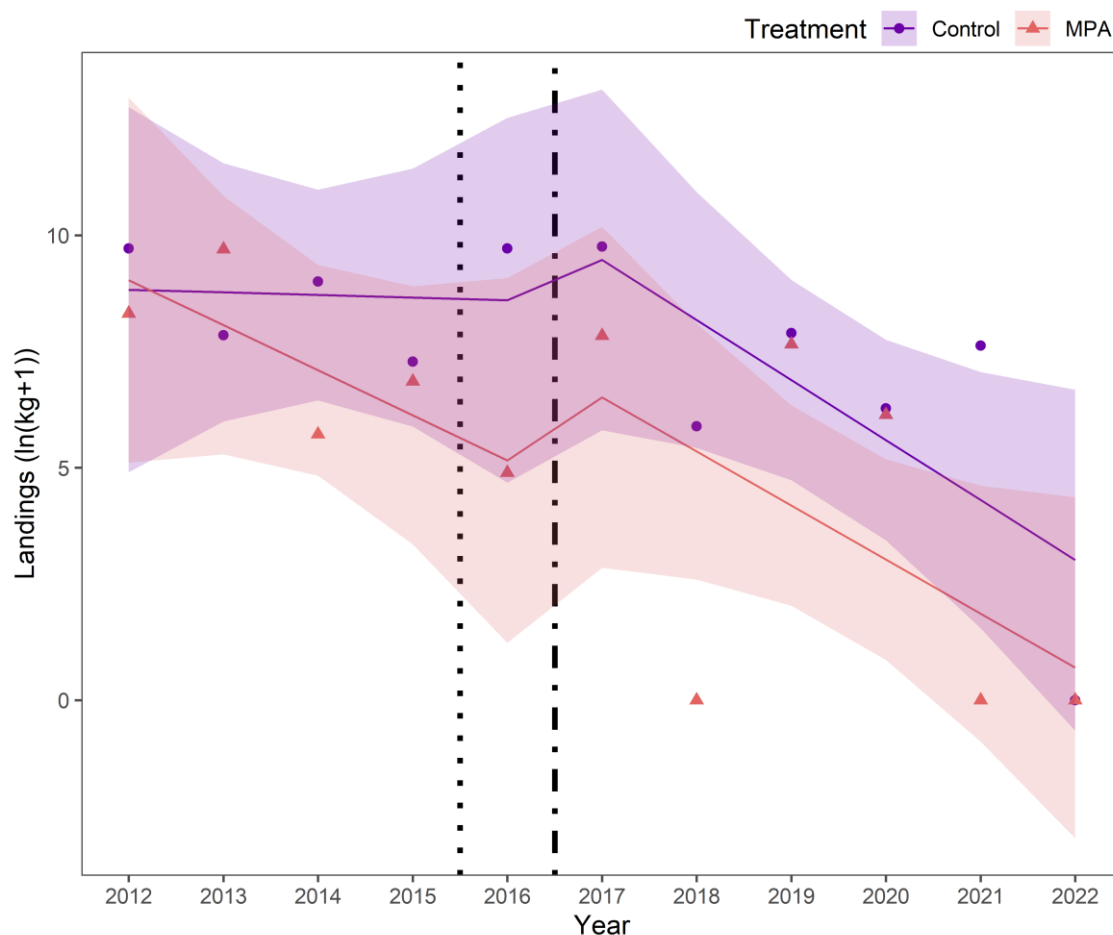


Figure 21. Total landings mass of all species, from all gears, (log transformed) over time, in the specific area within the AGRB MPA that had MBCG restrictions imposed and matched control sites.

NB: Area within the AGRB MPA that had MBCG restrictions imposed = orange/ triangles; matched control sites = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations for both MPA and control areas; ribbons indicate $\pm 95\%$ confidence intervals of the estimations. The full analytical model provides no evidence of a difference between the means across MPA and control sites, nor over time.

5.3.5.2. Gear-specific landings

Landings from bottom trawls appear to decrease across both the control and MBCG regulated areas after MBCG regulations come into force (Figure 22). While gill-net landings from the control sites appear relatively stable, they appear to increase within the MBCG regulated areas. As was demonstrated in the analyses of effort, the MBCG category is entirely made up of bottom trawl gears, while the landings from the non-MBCG category come predominantly from gill nets.

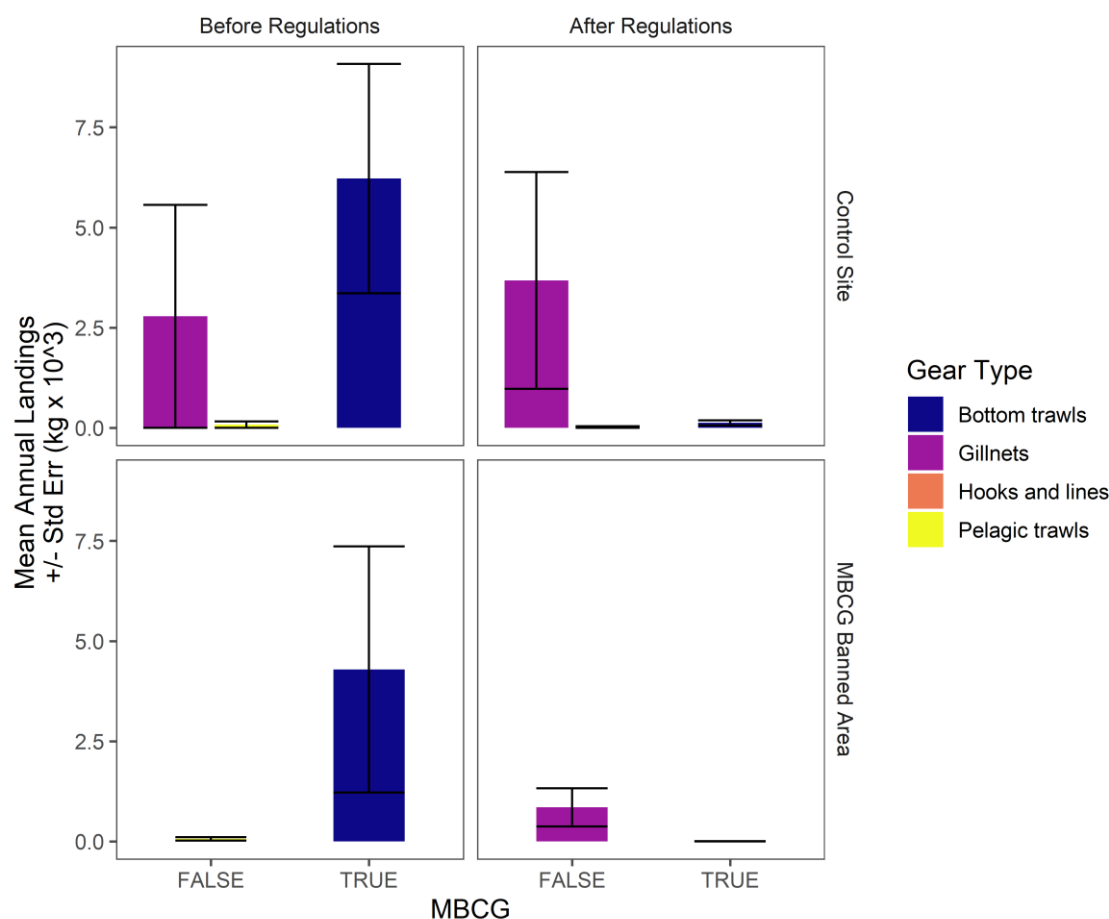


Figure 22. Mean annual landings mass +/- standard errors caught by different gear types categorised as MBCG or not.

NB: These means are calculated for four different contexts, before and after MBCG regulations came into force (columns) and in the areas in which the MBCG regulations apply vs some matched, non-MPA, control areas (rows).

5.3.5.3. MBCG and all species

Before and after SAC designation

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 12.2$). However, none of the coefficients in the model were significant. This indicates a valid BACI TS comparison with no significant changes in the MBCG regulated area after designation, relative to changes in the control area.

Before and after EU regulations on MBCG

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 11.6$). There was a significant interaction of BA:CI:T, indicating that the trend in landings in the MBCG regulated site was significantly different from that seen before the regulations, accounting for the change in the trend at the control site. Furthermore, a significant interaction between the before/after and MBCG/control terms indicates a change in the landings of the MBCG regulated site relative to the control site immediately after the regulations were put in place (Figure 23). No significant interaction between CI:T was found, indicating no significant difference in trends between the MPA and the control site prior to the imposition of the MBCG regulations.

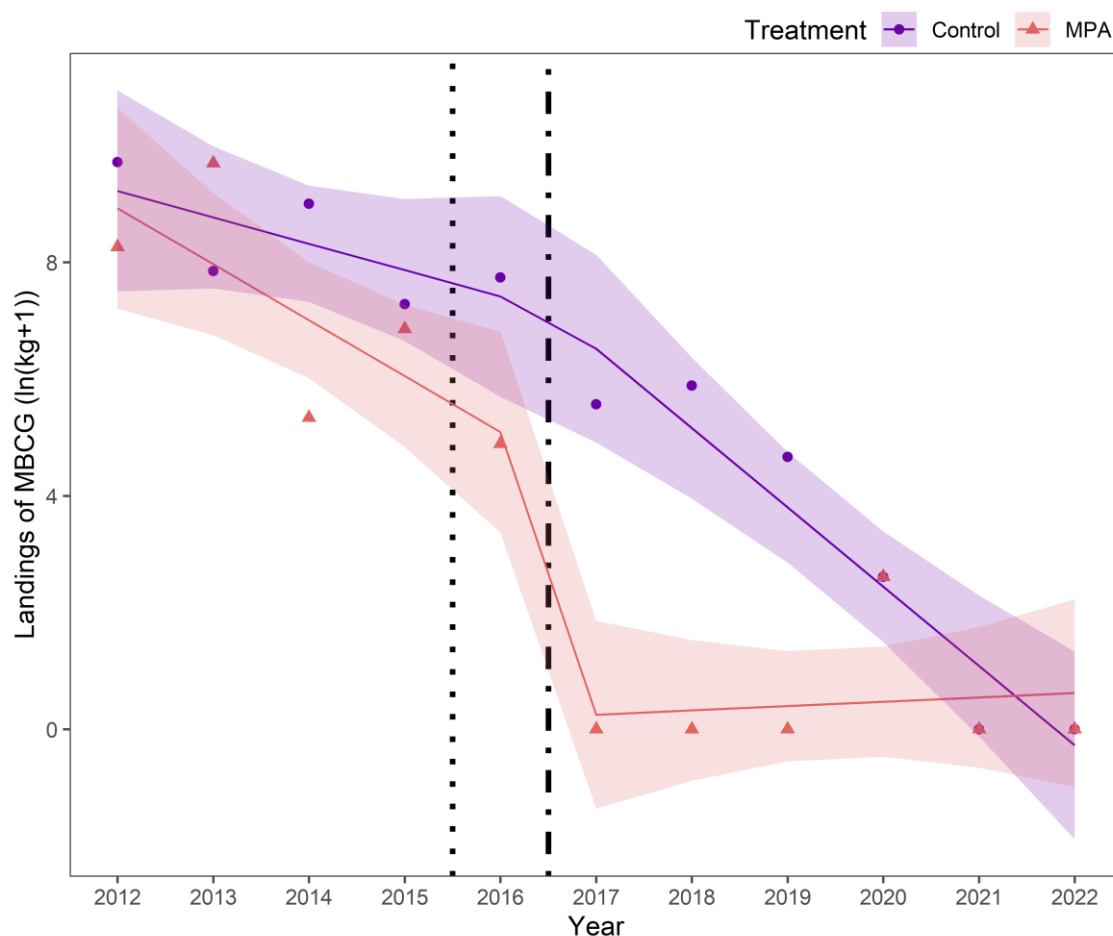


Figure 23. Annual landings mass of all species caught in MBCG (log transformed) over time, in the area of the AGRB MPA that had MBCG regulations imposed and matched control.

NB: Area of the AGRB MPA that had MBCG regulations imposed=orange/ triangles); matched control sites = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016), and the dot-dash line is when EU regulations were imposed restricting the use of MBCGs on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the imposition of MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is no difference in trend between sites before the MBCG regulations, but that there is a significant change in landings immediately after the regulations were imposed and that the trend in the MPA is significantly different from that of the control area after the MBCG regulations came into force.

5.3.5.4. Gill nets and all species

Data from gill-net-only landings did not support the fitting of either the BACI TS or the BACI models, in either of the SAC designation date or MBCG Regulation date contexts. Residual errors were skewed with strong patterns, indicating that models were unreliable and did not represent the data appropriately. This is likely due to the large numbers of zero annual landings and the fact that these are spread across both sides of the policy-change dates.

5.3.6. Landings per unit effort for the whole MPA

5.3.6.1. All species and gears combined

Before and after SAC designation

The full BACI TS provided the most parsimonious model fit ($\Delta AIC = 1.5$). However, none of the coefficients were significant, indicating that while the TS version of the BACI was appropriate, there was no significant difference in LPUE either between MPA and control, or before/after the site was designated.

Before and after EU Regulations on MBCG

The simplified BACI model was retained based on model parsimony ($\Delta AIC = 1.2$). Even in the simplified model, none of the model's coefficients, other than the intercept, were significant. Therefore, we cannot discern any differences between control and MPA sites, before or after MBCG regulations were imposed, interaction between these two groupings, or any change over time (Figure 24).

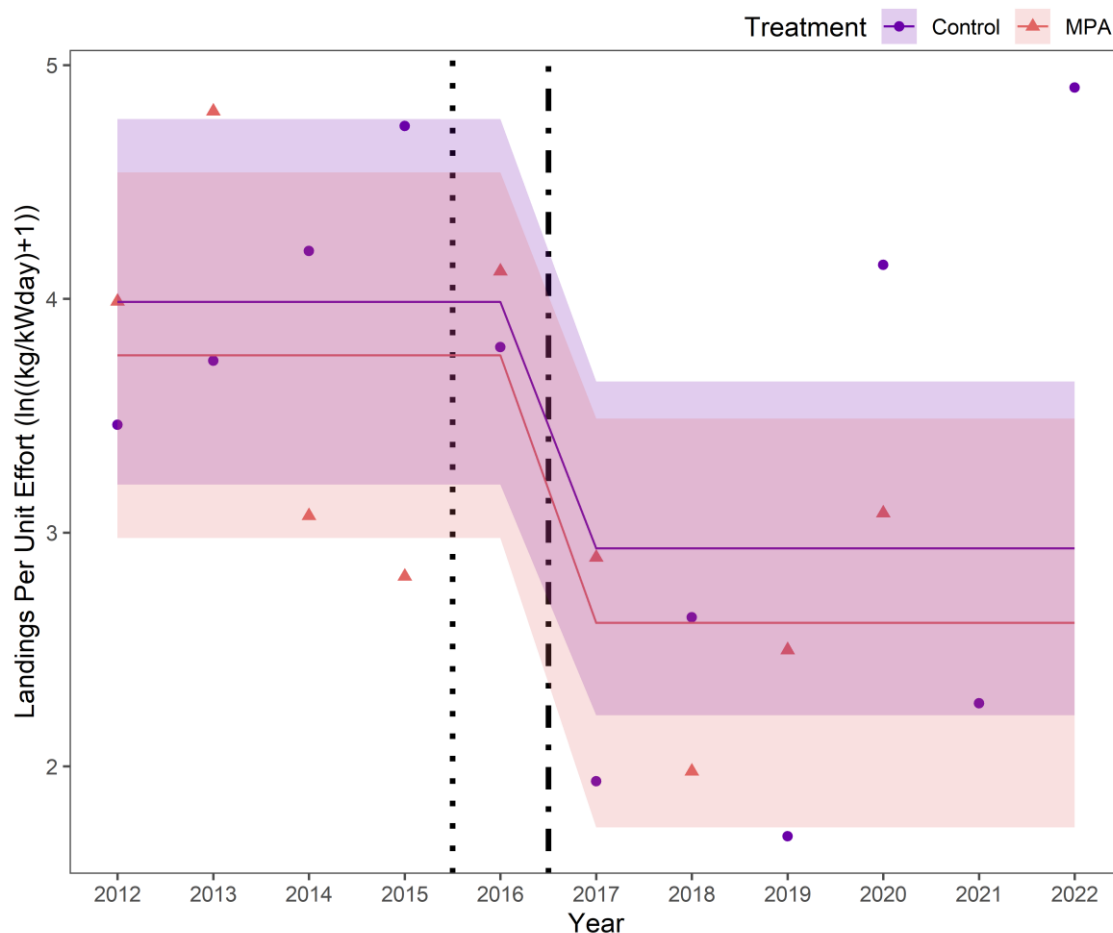


Figure 24. All species landings per unit effort (LPUE, kg/kWday, log transformed) from all gears pooled, over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/ triangles; matched control site = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line is when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). The trend line is simply the global mean of annual LPUE (log transformed), pooled for both the MPA and control areas combined as well as the before and after periods. The full analytical model indicates that there is no significant difference between the means across MPA and control sites, or over time.

5.3.6.2. MBCG and all species

Before and after SAC designation

The simple BACI model was retained based on parsimony ($\Delta AIC = 5$). There were no significant coefficients, indicating no detectable relative differences in average LPUE before and after designation between MBCG regulated and control sites (Figure 25).

Before and after EU regulations on MBCG

The simple BACI model was also retained when the comparative period was set to the implementation of MBCG regulations, in this case due to the poor fit of the full BACI TS model. This model was also a more parsimonious fit than the one that compared pre and post SAC designation ($\Delta AIC = 3.9$). However, this model also detected an overall significant change in LPUE shared by both the control and MPA (Figure 25). This indicates that changes in LPUE were independent of the implementation of regulations.

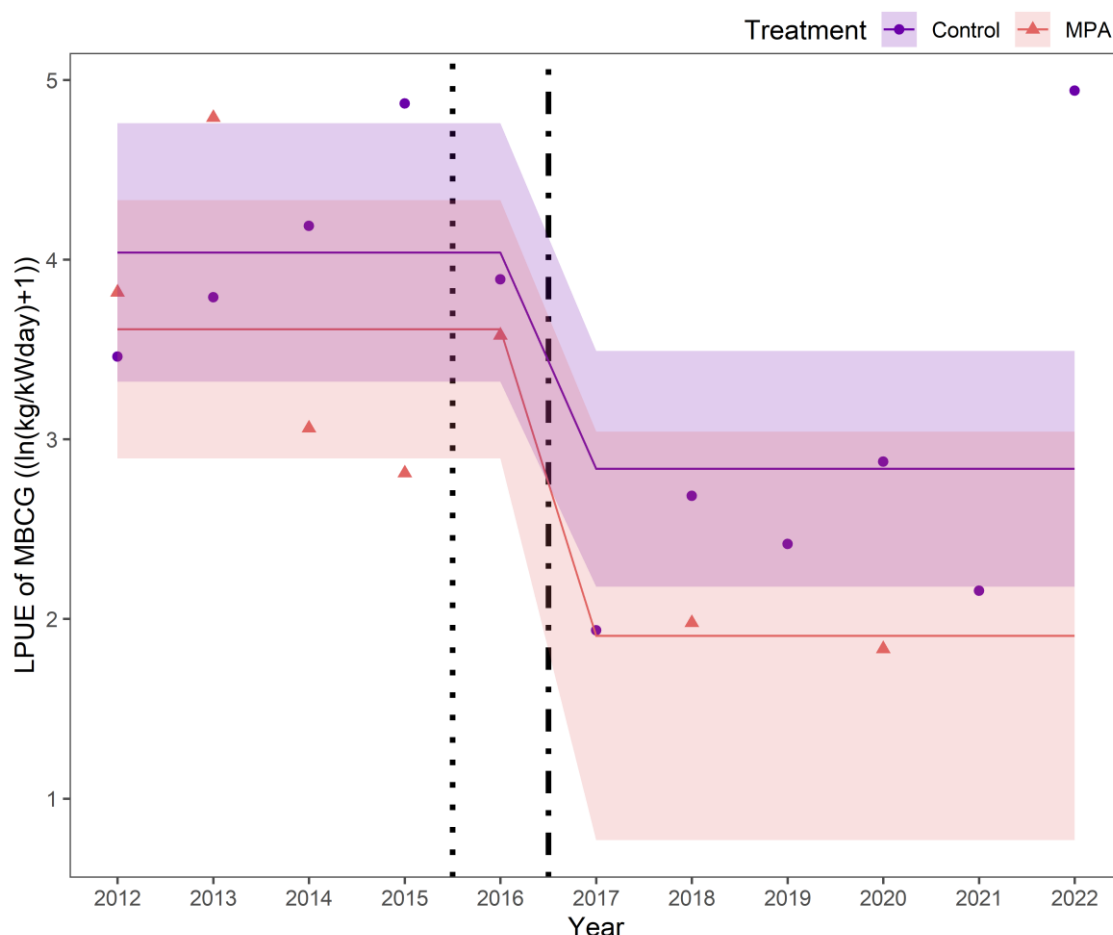


Figure 25. All species landings per unit effort (LPUE, kg/kWday, log transformed) from MBCG gears only, over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/ triangles; matched control site = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the imposition of MBCG regulations for both MPA and control areas; ribbons indicate $\pm 95\%$ confidence intervals of the estimations. The full analytical model indicates that there is no significant difference between the means across MPA and control sites, but an overall change in the LPUE for both areas after the implementation of the MBCG regulations.

5.3.6.3. Gill nets and all species

Because there was zero effort of gill nets prior to the MBCG regulations, no LPUE data can be calculated for the before periods, and hence no BACI analyses were run.

5.3.7. Landings per unit effort for the MBCG regulated areas

5.3.7.1. All species and gears combined

Before and after SAC designation

The simple BACI model was retained because the more complex model did not fit the data well (residual patterns). However, no coefficients were significant, hence we cannot describe any differences between the control or MBCG regulated areas, or before or after SAC designation, or between the two areas after SAC designation relative to before (Figure 26).

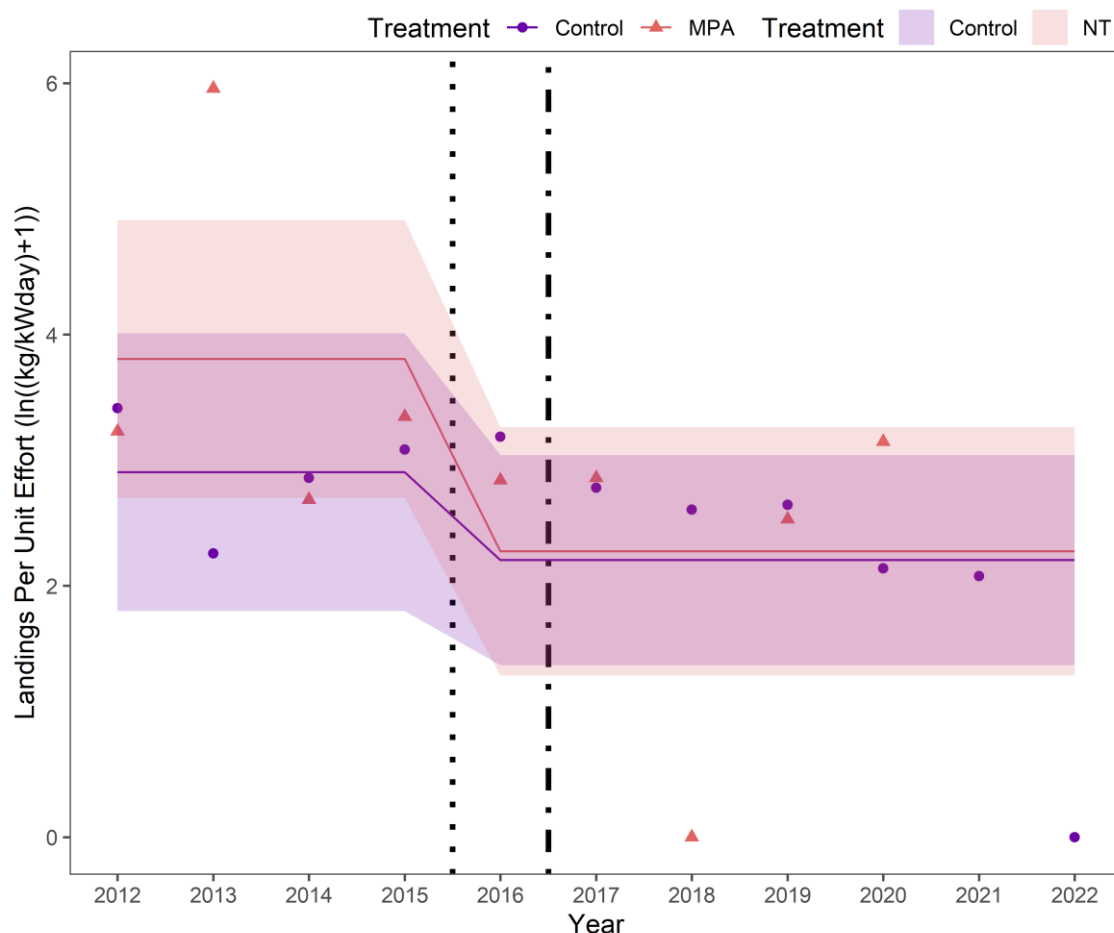


Figure 26. All species landings per unit effort (LPUE, kg/kWday, log transformed) from all gears, over time, in the area of the AGRB MPA that had MBCG regulations applied and matched control sites.

NB: Area of the AGRB MPA (orange/triangles) that had MBCG regulations applied = orange/triangles; matched control sites = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line

is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). The full analytical model indicates that there is no significant difference between the means across MPA and control sites, or over time.

Before and after EU regulations on MBCG

Neither BACI nor BACI TS models produced models with acceptable fits to the data. Therefore, there is no evidence of a change in LPUE for total fishing activities within the MBCG regulated area as a result of those regulations coming into force.

5.3.7.2. MBCG and all species

Before and after SAC designation

Both levels of complexity in the model produced equivalent fits according to model parsimony and both BACI and BACI TS models exhibited no significant coefficients, indicating no evidence of any changes in response to SAC designation nor time.

Before and after EU regulations on MBCG

Only the BACI model produced an acceptable fit to the available data and this model contains no significant coefficients, indicating no evidence of changes in response to MBCG regulations coming into force (Figure 27).

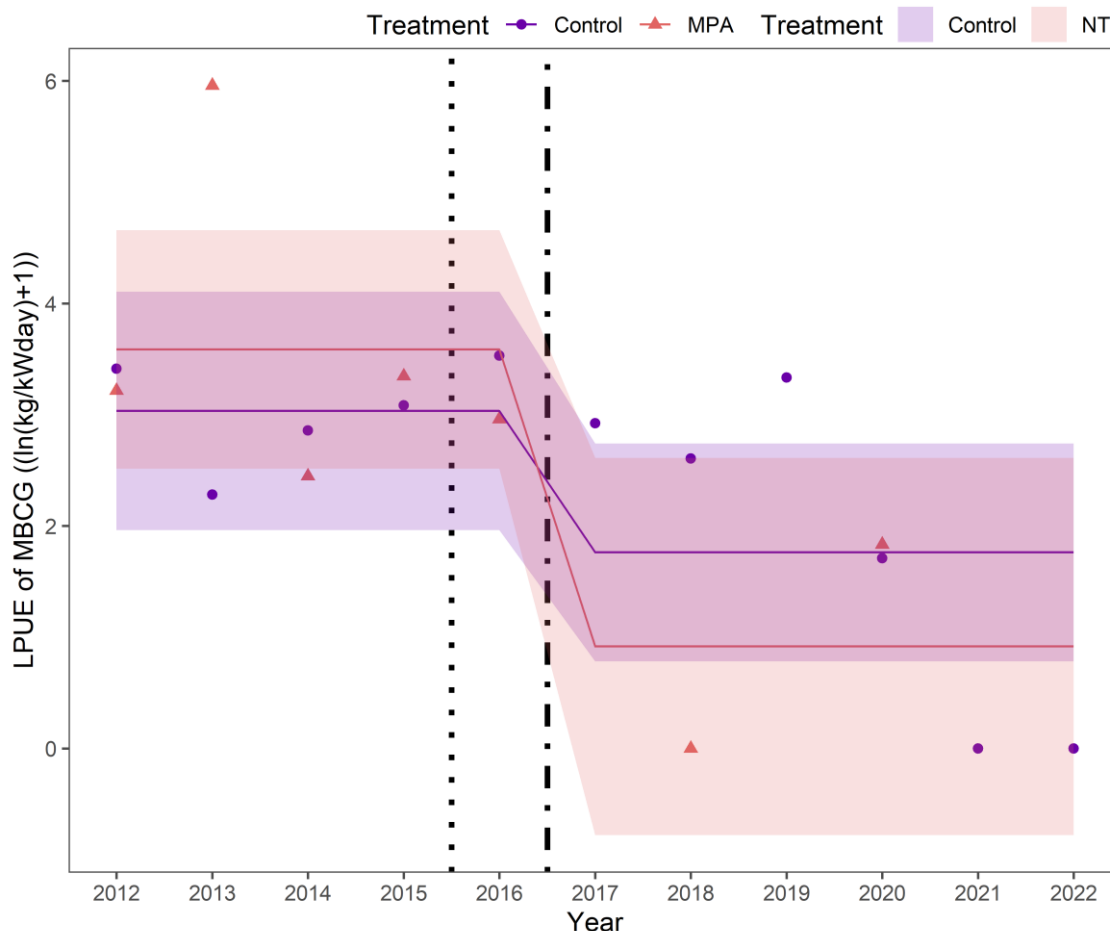


Figure 27. All species landings per unit effort (LPUE, kg/kWday, log transformed) from MBCG over time, in the area of the AGRB MPA that had MBCG regulations applied and matched control sites.

NB: Area of the AGRB MPA that had MBCG regulations applied = orange/triangles; matched control sites = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the imposition of MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is no significant difference in LPUE over time, between sites, or in response to the imposition of MBCG regulations.

5.3.7.3. Gill nets and all species

As there was no reported gill net effort prior to 2016, it is not possible to compare landings per unit effort before and after the SAC designation or the MBCG regulations.

5.3.8. Landings value for the whole MPA

5.3.8.1. All gears and species combined

Before and after SAC designation

The BACI TS model was retained as the simple BACI model did not fit the data, based on patterns in the model's residuals. This full BACI TS model produced no significant coefficients for any of the model terms, indicating no evidence of an impact of SAC designation on landings value from the MPA.

Before and after EU regulations on MBCG

In contrast, both the BACI and the BACI TS based on a before/after period around the MBCG regulations fit the data but the full model had a much more parsimonious fit ($\Delta AIC = 16$). This full BACI TS provides evidence of both an immediate relative shift in landings value (significant coefficient for the BA:CI term), and a relative change in trend between MPA and control sites after the MBCG regulations were imposed (significant coefficient for the BA:CI:Y term) (Figure 28).

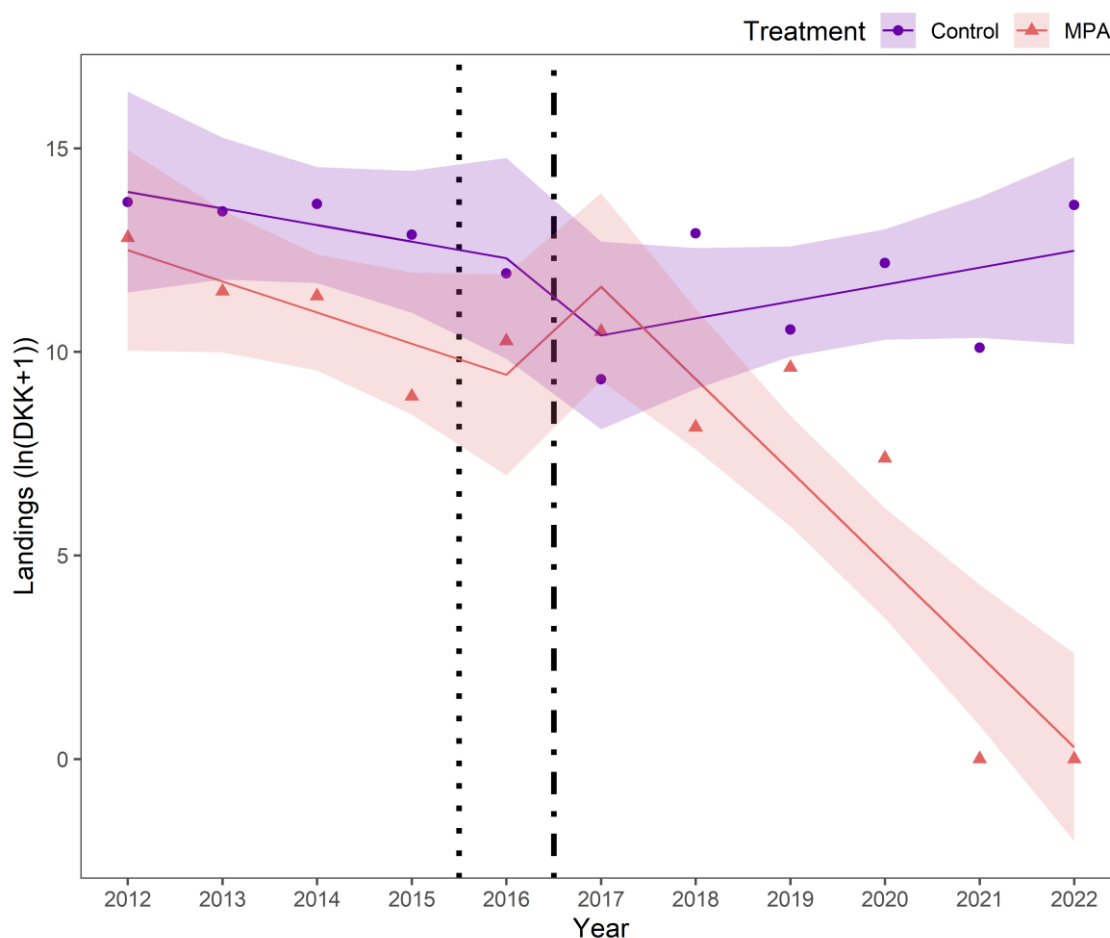


Figure 28. Annual landings value of all species, from all gears (DKK, log transformed) over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was

designated a SAC (2016), and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the SAC designation for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is a significant change in landings value immediately after the regulations were imposed and that the trend in the MPA is significantly different from that in the control area after the MBCG regulations came into force.

5.3.8.2. MBCG and all species combined

Before and after SAC designation

The full BACI TS was retained as the representative model as the simplified model did not fit the data based on observations of standardised residuals. However, none of the model coefficients were significant, indicating that the comparison with between MPA and control was valid but that there were no relative differences in landings value between these two areas or significant changes over time relative to the date of SAC designation.

Before and after EU regulations on MBCG

Because of the combination of observations of zero landings value and relatively high landings value in the MPA after MBCG regulations were imposed, there are some residual deviations. However, the simple BACI model was the most parsimonious and appears to fit the data acceptably well. This model has a significant interaction effect of the MPA vs control from before to after the MBCG regulations were imposed. This indicates a change in the mean difference between the two sites, in response to the imposition of MBCG regulations (Figure 29).

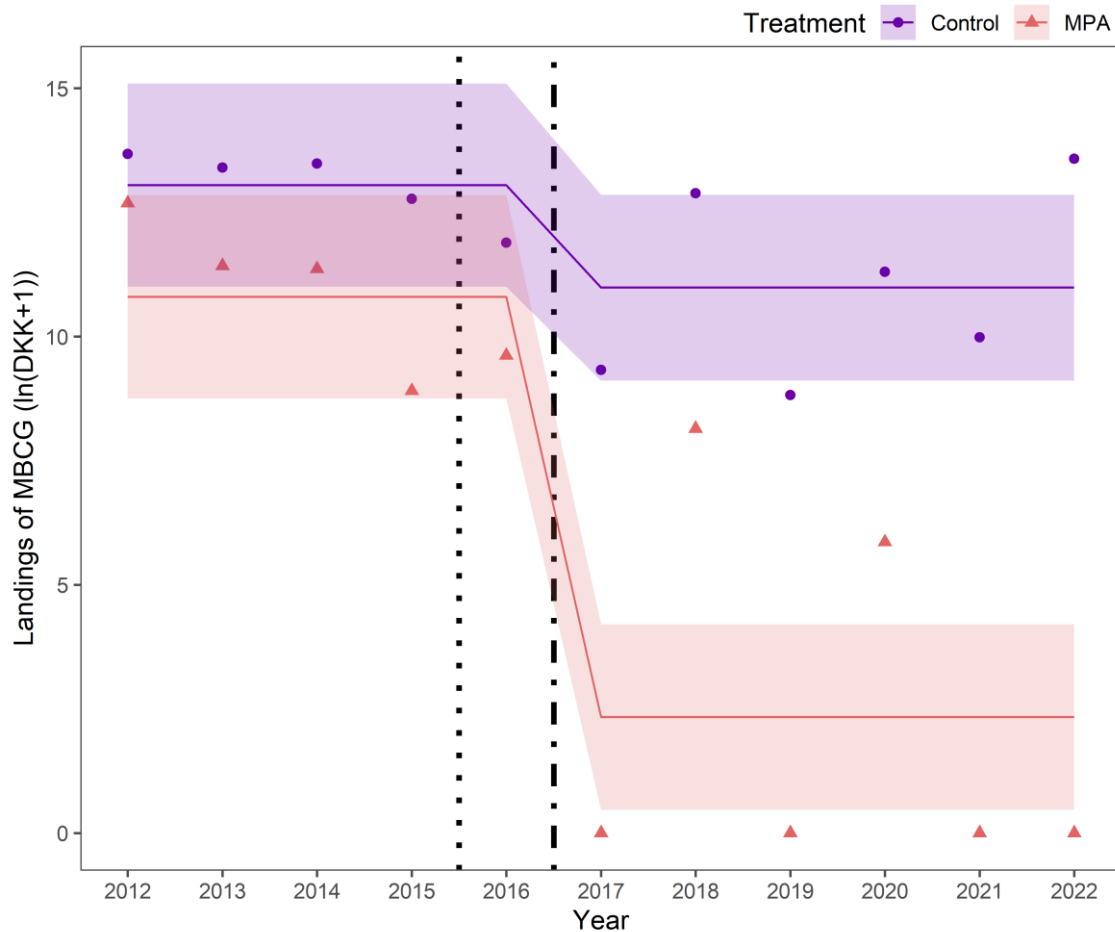


Figure 29. All species landings value (DKK, log transformed) from MBCG over time, in the area of the AGRB MPA and a matched control site.

NB: Area of the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the imposition of MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is a significant change in the difference between the overall means between MPA and control sites before and after the MBCG regulations came into force.

5.3.8.3. Gill nets and all species combined

Model fits for pre/post SAC designation and the fit for the full BACI TS with pre/post MBCG regulations were very poor, with large patterns in the models' residuals. The only retained model was the simple BACI with the before after period based on the MBCG regulations. This model shows no significant difference in gill-net landings value in the MPA vs the control; however, the pooled mean landings values of both sites does show a significant increase after the regulations were imposed (Figure 30).

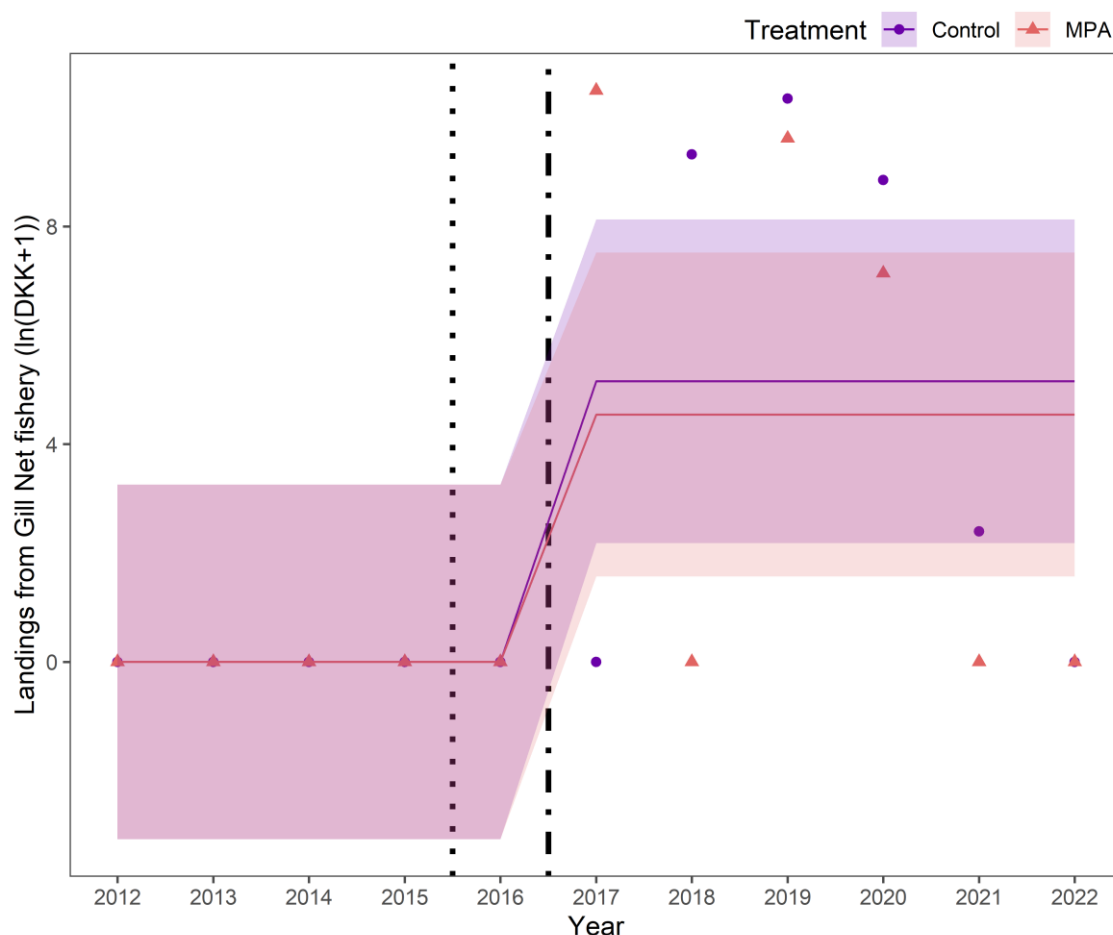


Figure 30. Landings value from gill nets (DKK, log transformed) over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations across both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is no significant difference between the means across MPA and control sites. However, there is a significant difference in the pooled means before and after the imposition of the MBCG regulations.

5.3.9. Landings value for the MBCG regulated areas

5.3.9.1. All gears and species combined

Before and after SAC designation

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 8$). However, none of the coefficients in the model were significant, indicating a valid BACI TS comparison with no significant changes in the MBCG regulated areas after SAC designation, relative to changes in the control area.

Before and after EU regulations on MBCG

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 7.8$). However, none of the coefficients in the model were significant, indicating a valid BACI TS comparison with no significant changes in the MBCG regulated areas after the regulations were put into place, relative to changes in the control area (Figure 31). The model based on the date of MBCG regulations coming into force showed slightly more uniform residuals than the model based on SAC designation date.

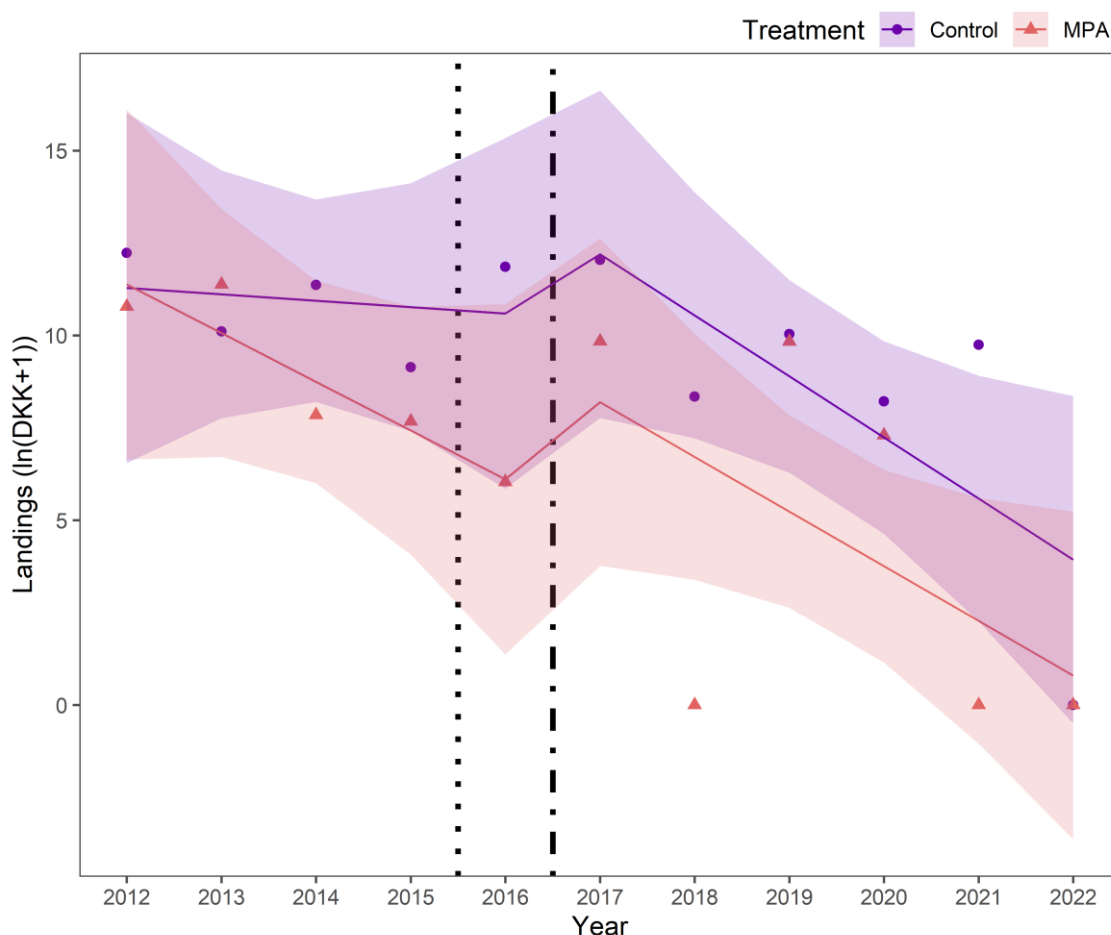


Figure 31. Landings value of all species, from all gears (DKK, log transformed) over time, in the specific area within the AGRB MPA that had MBCG restrictions imposed and matched control sites.

NB: Specific area within the AGRB MPA that had MBCG restrictions imposed = orange/triangles; matched control sites = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the

dotted line indicates when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model provides no evidence of a difference between the means across MPA and control sites, or over time.

5.3.9.2. MBCG and species combined

Before and after SAC designation

The full BACI TS model was selected as the most parsimonious model ($\Delta AIC = 13$). However, none of the coefficients in the model were significant, indicating a valid BACI TS comparison with no significant changes in the landings value from MBCG in MBCG regulated areas after SAC designation, relative to changes in the control area.

Before and after EU regulations on MBCG

The full BACI TS model was also the most parsimonious when considering the periods before and after MBCG regulations ($\Delta AIC = 13.1$). This full BACI TS provides evidence of both an immediate relative shift in landings value (significant coefficient for the BA:CI term), and a relative change in trend between MPA and control sites after the MBCG regulations were imposed (significant coefficient for the BA:CI:Y term) (Figure 32).

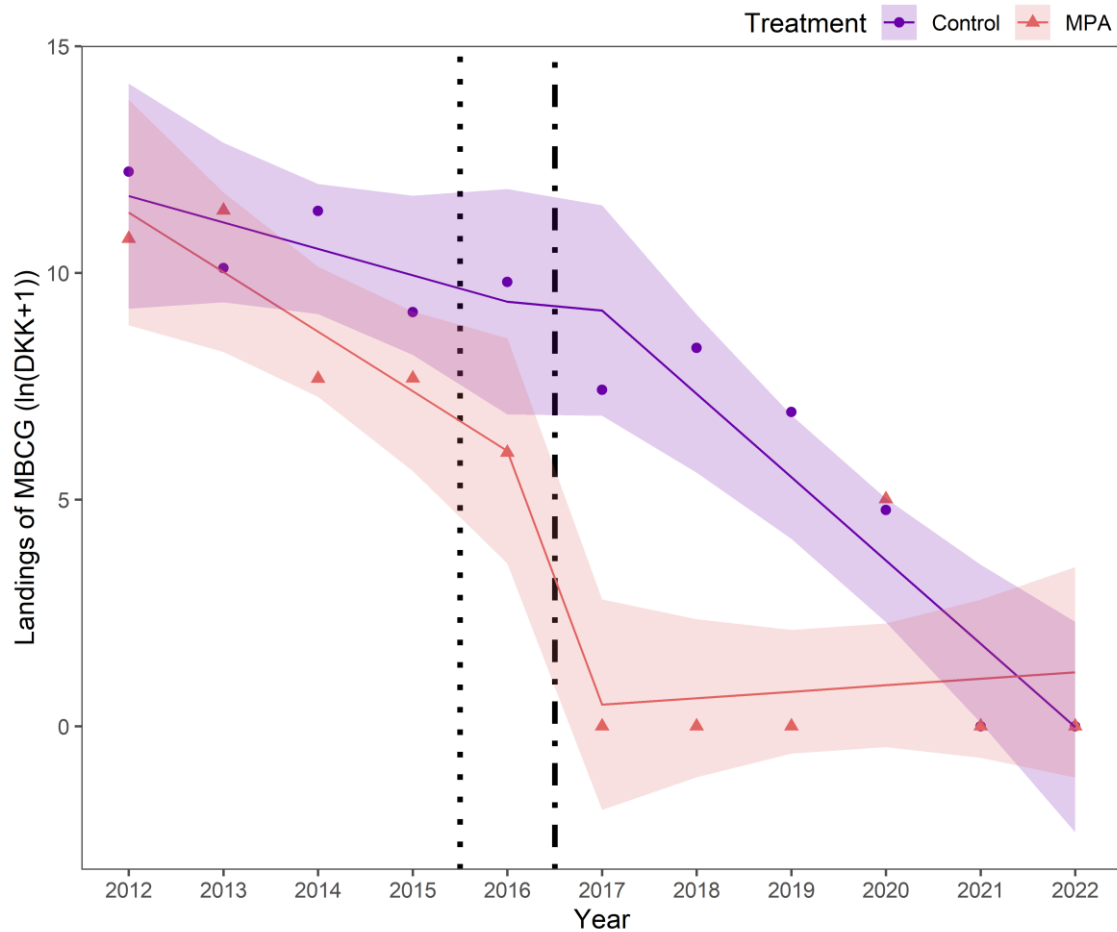


Figure 32. Annual landings value of all species caught in MCBG (DKK, log transformed) over time, in the area that became MCBG regulated within the AGRB MPA and matched control sites.

NB: Area that became MCBG regulated within the AGRB MPA = orange/ triangles; matched control sites = purple/ circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016), and the dot-dash line is when EU regulations were imposed restricting the use of MCBG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the SAC designation for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates that there is a significant change in landings value immediately after the regulations were imposed and that the trend in the MPA is significantly different from that of the control area after the MCBG regulations came into force.

5.3.9.3. Gill nets and all species combined

Before and after SAC designation

The full BACI TS was the most parsimonious of the two models ($\Delta AIC = 3.3$) but showed no evidence of a difference between the areas with MBCG regulations and the controls (Figure 33). This was also the best-fitting model compared to those fit for the period before and after the imposition of the MBCG regulations.

Before and after EU regulations on MBCG

The full BACI TS was the most parsimonious of the two models ($\Delta AIC = 6.4$) but showed no evidence of a difference between the areas with MBCG regulations and the controls.

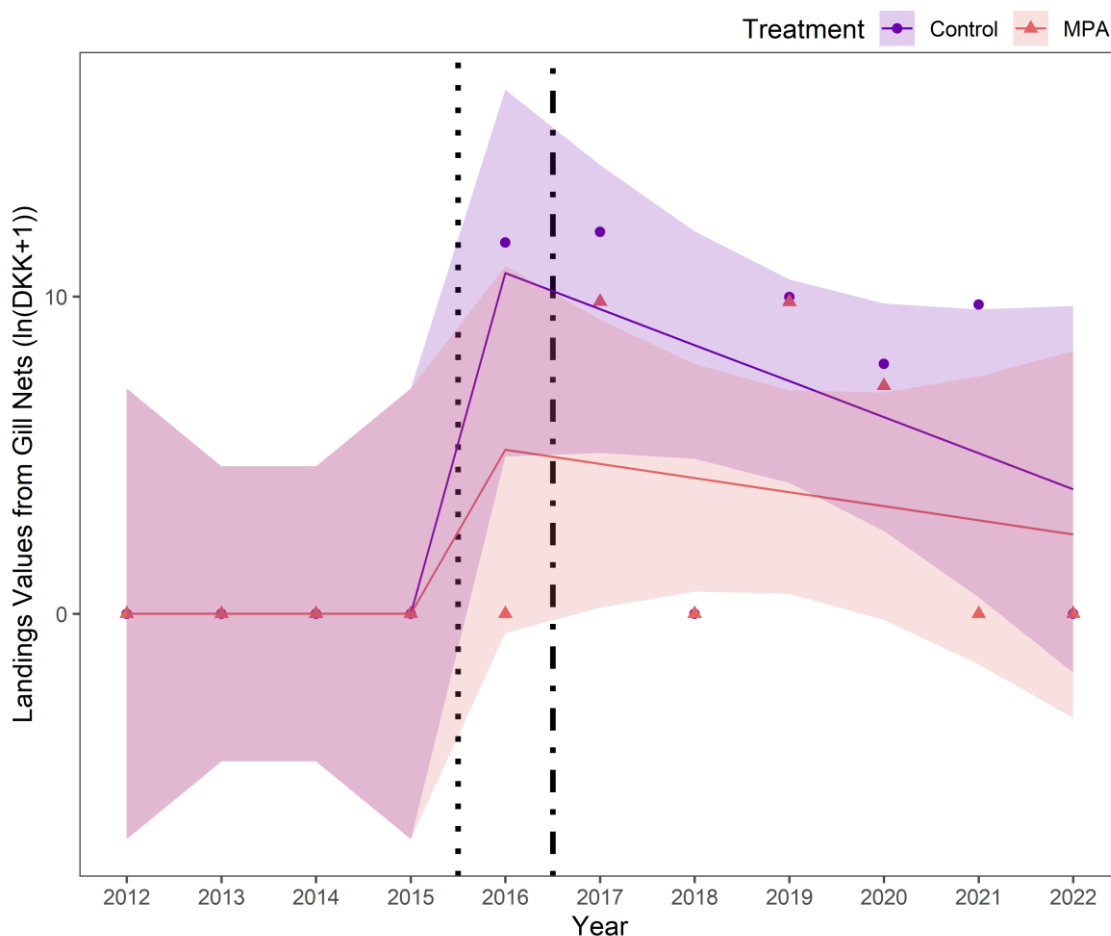


Figure 33. Landings value of all species, from Gill Nets (DKK, log transformed) over time, in the specific area within the AGRB MPA that had MBCG restrictions imposed and matched control sites.

NB: Specific area within the AGRB MPA that had MBCG restrictions imposed = orange/triangles; matched control sites = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016) and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full

analytical model provides no evidence of a difference between the means across MPA and control sites, or over time.

5.3.10. Summary of fisheries changes results

Of all cases where suitable models were retained (20; Table 2), models basing the before/after variable on SAC designation date only provided the most appropriate models in two cases (Table 3). The remaining 18 cases retained the model utilising the MBCG regulation date as the most appropriate. The two models with the SAC designation date as the temporal context provided no evidence of a difference between control and MPA sites (MBCG spatial context).

In nearly all cases, three of the four responses / metrics for fisheries change (namely: effort, landings mass and landings value) behaved the same in the retained models.

In the whole MPA context and considering all gears at once, all three responses showed significant differences between MPA and controls both immediately after the implementation of MBCG regulations and in the trends over subsequent years. In the analyses of the MBCG gears in isolation, only the simplified BACI models were retained, but all three responses still showed the same significant relative changes in annual means between MPA and control sites after the implementation of MBCG regulations. The analyses of gill-net-only fishing activity showed the same patterns in effort and landings value across both sites, with no difference attributable to changes in regulations, but there was no evidence of a change in total mass of fish extracted with these gears between the MPA and control site.

When limiting analyses to only those fishing activities taking place in the area with MBCG regulations, there was no evidence of any changes in fishing activity when all gears were grouped together. When analysing only MBCG responses, there was no significant difference in the effort with the MPA relative to control sites; however, there was a significant decrease in both landing metrics in the MPA relative to the control sites after the regulations were imposed. Contrary to the MBCG effort, gill-net effort increased across both sites from around the time of MBCG regulations, but this increase was comparable between the MPA and the control sites, meaning it could not be attributed to the MPA.

Irrespective of which gear groups were analysed or which spatial contexts were considered, there was no evidence of a change in LPUE in response to either implementation step in the MPA (SAC or MBGC regulations).

Table 2. Summary of retained models by spatial context, for various gear groups and four key responses/metrics of change in the fishery.

Spatial context	Gears	Response	Temporal context	Model selection	Δ AIC	Mean difference / immediate difference	Relative difference in trend
Full MPA	All gears	Effort (kW.days)	MBCG regulations	BACI TS	12.2	Significant	Significant
		Landings (DKK)	MBCG regulations	BACI TS	16	Significant	Significant
		Landings (kg)	MBCG regulations	BACI TS	12.4	Significant	Significant
		LPUE (kg/kWday)	MBCG regulations	BACI	1.2	Not significant	NA
	Gill nets only	Effort (kW.days)	MBCG regulations	BACI	Based on fit	Not significant	NA
		Landings (DKK)	MBCG regulations	BACI	Based on fit	Not significant	NA
		Landings (kg)	MBCG regulations	Poor fits	NA	NA	NA
		LPUE (kg/kWday)	MBCG regulations	No effort before	NA	NA	NA
	MBCG gears only	Effort (kW.days)	MBCG regulations	BACI	4.9	Significant	NA
		Landings (DKK)	MBCG regulations	BACI	11	Significant	NA
		Landings (kg)	MBCG regulations	BACI	0.8	Significant	NA
		LPUE (kg/kWday)	MBCG regulations	BACI	Based on fit	Not significant	NA
MBCG regulations	All gears	Effort (kW.days)	MBCG regulations	BACI TS	1	Not significant	Not significant
		Landings (DKK)	MBCG regulations	BACI TS	7.8	Not significant	Not significant
		Landings (kg)	MBCG regulations	BACI TS	5.3	Not significant	Not significant
		LPUE (kg/kWday)	MBCG regulations	BACI	Based on fit	Not significant	NA
	Gill nets only	Effort (kW.days)	SAC designation	BACI	1.3	Not significant	NA
		Landings (DKK)	SAC designation	BACI TS	3.3	Not significant	Not significant
		Landings (kg)	MBCG regulations	Poor fits	NA	NA	NA
		LPUE (kg/kWday)	MBCG regulations	No effort before	NA	NA	NA
	MBCG gears only	Effort (kW.days)	MBCG regulations	BACI TS	5.4	Not significant	Not significant
		Landings (DKK)	MBCG regulations	BACI TS	13.1	Significant	Significant
		Landings (kg)	MBCG regulations	BACI TS	11.6	Significant	Significant
		LPUE (kg/kWday)	MBCG regulations	BACI	Based on fit	Not significant	NA

NB: See Methods (Modelling approaches) for the context of the two right-most columns. 'NA' values across the three right-hand columns are due to a model not being fit (missing data) or a model not being fit (see column ' Δ AIC'). 'NA' in only the right-most column is where the best model was a BACI, without the time-series explicit analyses, and thus, no relative difference in trends were tested.

Table 3. Numbers of retained models by spatial context (bold left column) and temporal context (plain font left column) across the four key responses / metrics of change in the fishery.

Spatial and Temporal Contexts	Responses				
	Effort (kW.days)	Landings (DKK)	Landings (kg)	LPUE (kg/kWday)	Grand Total
Full MPA					
MBCG regulations	3	3	2	2	10
MBCG regulations					
MBCG regulations	2	2	2	2	8
SAC designation	1	1			2
Grand Total	6	6	4	4	20

5.4. Habitat use changes in response to MPA

5.4.1. Effort across habitats

To investigate the effect of SAC designation or MBCG regulations on the interaction of fisheries and various habitat types, we first investigated which habitats received which levels of effort (Figure 34). Some habitats were never fished (i.e. hard-bottom habitats in the deep and shallow shelf), while others were only fished in the MPA area and not the control (hard euphotic and soft deep shelf). Only two habitats (soft euphotic and soft shallow shelf) experienced fishing effort in both control and MPA areas. Therefore, the BACI analyses were only possible for these two habitat types.

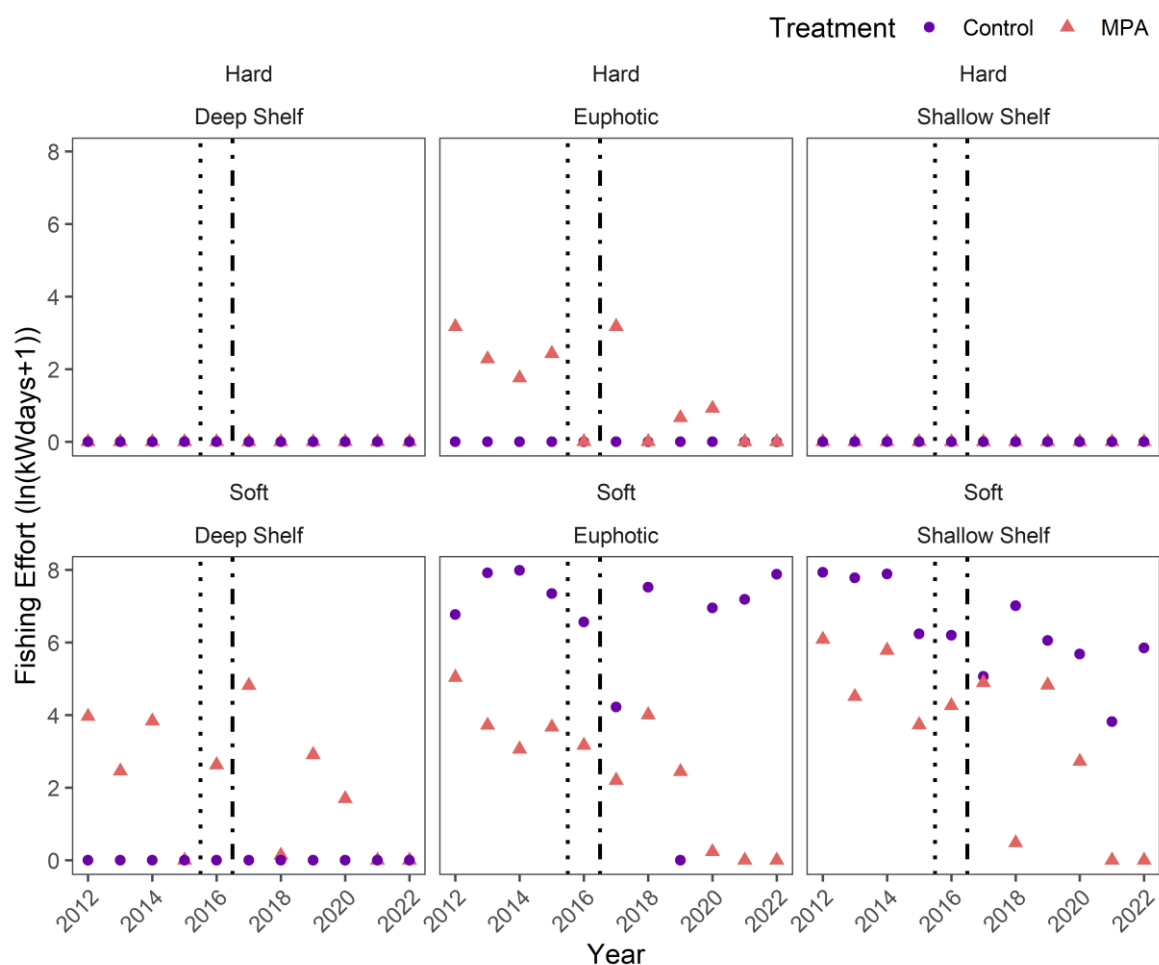


Figure 34. Total fishing effort (log transformed) over time, in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a Special SAC (2016), and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Individual panels show the effort by each of six habitat categories, with hard-bottom (top row) and soft-bottom (bottom row) habitats divided by relative depths (columns).

5.4.2. Effort on soft euphotic habitats

The simplified BACI model was the only model to appropriately fit the data for both scenarios with the SAC designation date or the MBCG regulation date as before/after designators. In both cases, no significant coefficients were found for the interaction between the MPA/control and before/after variables, indicating no evidence that either the SAC designation or MBCG regulations caused a change in effort on soft euphotic habitats (Figure 35). From these two BACI models, the one based on the MBCG regulation date provided the best fit and AIC ($\Delta AIC = 0.9$) and this model indicated a significant overall difference in the means of effort between MPA and control site, but no change in this relative difference before and after the implementation of the MBCG regulations.

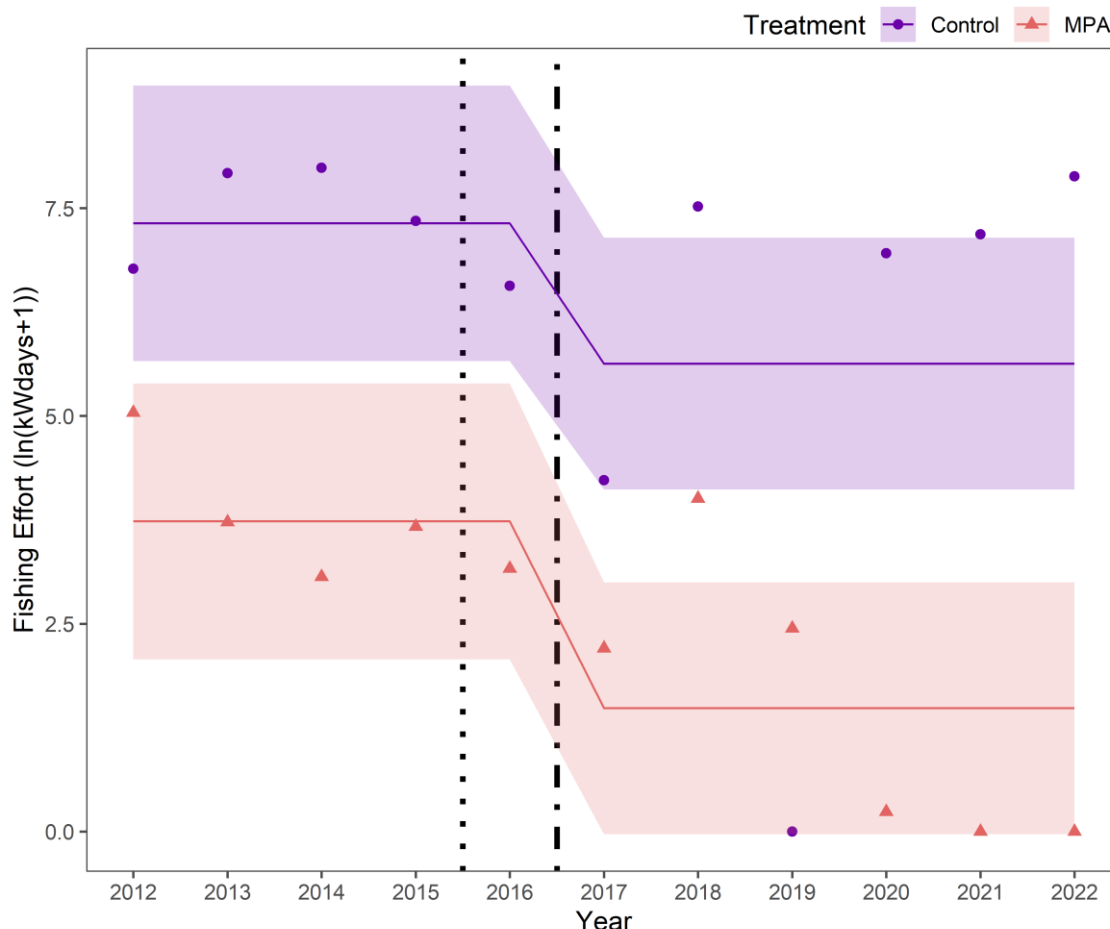


Figure 35. Total fishing effort (log transformed) over time, for soft eutrophic habitats in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016), and the dot-dash line is when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the imposition of MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates only a difference between MPA and control site, but no significant change in response to the implementation of MBCG regulations and no a significant effect of the MPA relative to control after the MBCG regulations were put in place.

5.4.3. Effort on soft shallow shelf habitats

Before and after SAC designation

The full BACI TS model was retained as it had a slightly more parsimonious fit than the simplified model ($\Delta AIC = 1.4$). However, no coefficients were deemed significant, indicating a valid BACI TS model but with no evidence of effects from SAC designation, or time since designation (Figure 36).

Before and after EU Regulations on MBCG

The full BACI TS and the simplified BACI model achieved the same level of parsimony with equivalent residual patterns ($\Delta AI = 0$). The simpler model reported a significant difference in overall means between MPA and control sites, but did not provide any evidence that this difference was affected by the implementation of the MBCG regulations. When considering the variability across the time series, the BACI TS did not detect this difference between overall means. The model fit based on the SAC designation date as the before/after period produced a better fit than both of these models; therefore, that one was retained.

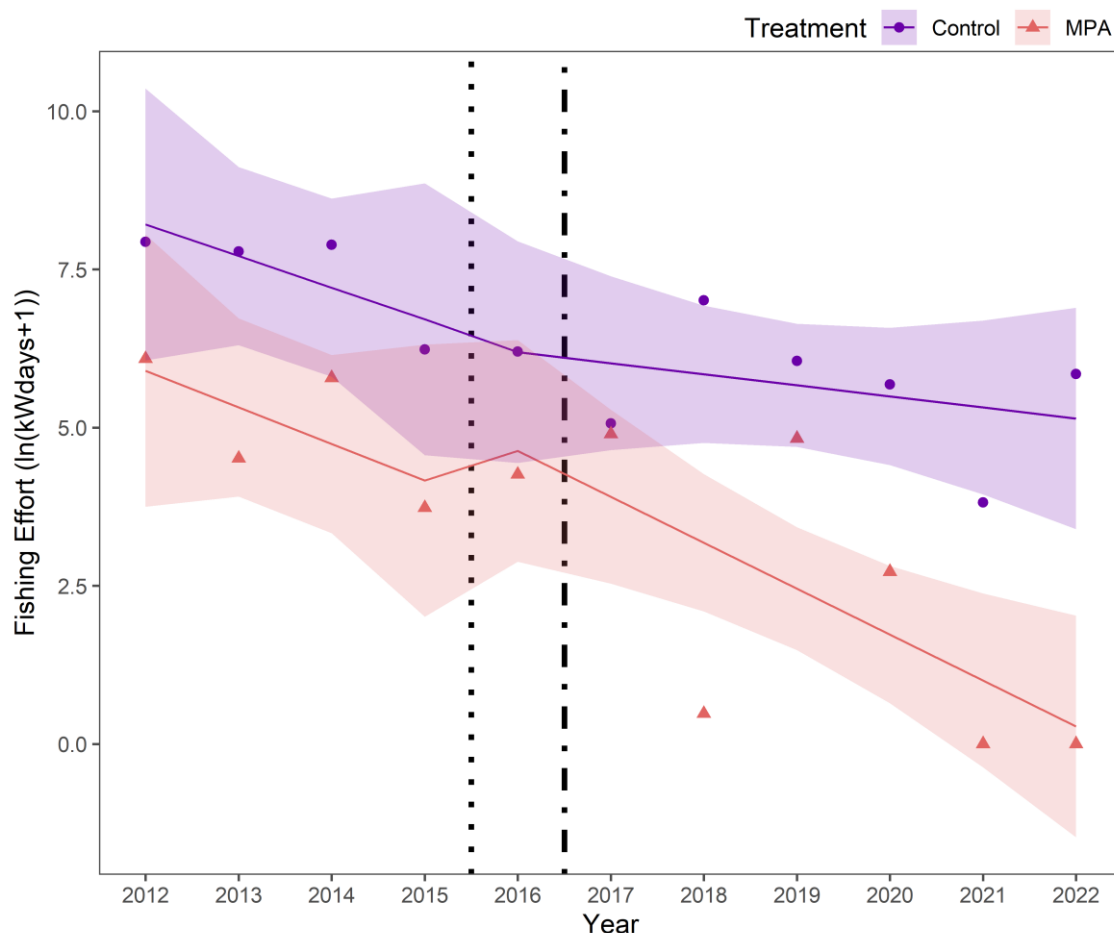


Figure 36. Total fishing effort (log transformed) over time, for soft shallow shelf habitats in the area that became the AGRB MPA and a matched control site.

NB: Area that became the AGRB MPA = orange/triangles; matched control site = purple/circles. Vertical lines represent the breaks between before and after periods in the analyses: the dotted line indicates when the site was designated a SAC (2016), and the dot-dash line indicates when EU regulations were imposed restricting the use of MBCG on the reef areas within the MPA (2017). Trend lines are model estimations based on a break before and after the imposition of MBCG regulations for both MPA and control areas; ribbons indicate +/- 95 % confidence intervals of the estimations. The full analytical model indicates no significant difference between MPA and control site, no significant change in response SAC designation and no significant effect over time.

6. DISCUSSION

6.1. Interpretation of results

Our results show a dominant retention of models based on the MBCG regulations coming into force, as opposed to the alternative models based on SAC designation date. Moreover, in the only two cases where the SAC designation date models were retained, no significant effect of SAC designation date was detected. This indicates that **most detectable changes occurred concurrently with the MBCG regulations**, which directly affected segments of the fishery. It may be that either SAC designation was not sufficient to effect a change in fisher behaviour around the AGRB MPA, or that any change in behaviour occurred with a lag after the designation date, such that it was better represented by using the later date of the MBCG regulations. This second hypothesis appears less likely when we look at those cases in the results where the data supported fitting the full BACI TS model. Here we see that changes in fisheries activity occur immediately after the MBCG regulations are put in place, rather than a gradual change that may be expected by a lagged voluntary response.

In nearly all cases the **responses of landings mass and landings value followed the response of effort**. This could be expected, as landings should be correlated to effort. However, if the change in effort had an impact on the local population size (i.e. increased abundance), then one would anticipate cases with no discernible change in landings mass, or higher-quality large fish maintaining landings value. This was not seen in our analyses. Furthermore, none of our analyses of landings per unit effort showed evidence of any change in response to the site designation, or to the implementation of MBCG regulations. One caveat to this interpretation, is that there was no gill-net activity in the MPA (or in most years in the controls) in our 'before' periods. This means that no LPUE value could be calculated, and therefore, no analyses could be done on the effect of the MPA or MBCG regulations on the LPUE of gill-net fisheries.

When **considering the whole MPA, we can see that the significant decreases in fishing effort and landings appear to have been driven by changes in MBCG activities. Contrastingly, the effort of gill netting increases immediately after the regulations on MBCGs comes into force**, but this seems to occur across the broader region, and not just in the MPA itself. Nevertheless, this increase in gill-net effort reduces the observed effects in the models considering responses from all gears combined (for example, see the significant initial relative rise in effort within the MPA for all gears). This comparison both justifies and supports the analysis of fishing activities that are disaggregated by gear or style of fishing.

In the spatial context of the MBCG regulated area, we see no significant changes in combined effort, or in the effort of MBCG or gill-net fisheries independently. This might appear counter-intuitive, as one could expect that the prohibition of the use of MBCG gears would lead to a significant decrease in their recorded effort. However, it seems that the effort from fishing with MBCG gears, within the area selected for their prohibition, was already in decline from the start of our time series. The observed rapid decline in MBCG effort immediately after the regulations came into force was therefore not statistically distinguishable from the overall downward trend of MBCG effort in both the regulated area and the controls. Immediately after the SAC designation and the MBCG regulations came into force, gill-net-fisheries effort rose from zero to their highest levels for the control area and MBCG regulated area, respectively. However, this rise was coupled with sporadic years

of zero effort and there was therefore **no significant change in gill-net activities in response to the MPA designation or MBCG regulations.**

There was **no evidence that the habitat types that were fished changed in response to MPA designation or MBCG regulations.** There were general decreasing trends in effort across the two habitats where we had enough data to analyse them. However, these were the same trends for both the control and MPA sites. Furthermore, these overall decreasing trends appear to be driven by the aforementioned decrease in MBCG effort that is itself most likely linked to a decrease in fishing opportunities for the highly valued Atlantic cod.

6.2. Fisheries and conservation objectives

While the AGRB MPA was designated for the protection of reefs, sand banks and harbour porpoise, the only fisheries regulations imposed are for the protection of reefs. These protections are only applied to the reefs and some buffer zones around them, within the MPA. These restrictions appear to have had no significant impact on fisheries activities, as the gears that were prohibited were already in decline in the regulated area, and continued to decline in a similar manner in the control areas. The efficacy of these regulations may then be questioned; however, their imposition did have **a significant effect on MBGC gears across the whole MPA.** The reasons for this are unknown but may be due to fishers with MBCGs not finding the rest of the MPA a suitable place to fish, or because they are being overly cautious in order to avoid penalties for fishing within the MBCG exclusion areas within the MPA. **This *de facto* protection, afforded to the rest of the MPA, goes some way to protect the sand-bank habitats from physical disturbance, but the actual effect of this reduction in MBCG activity has not been monitored or documented.**

In a broader trend, **across the MPA and both sets of controls, the gill-net fisheries increased from zero to some occurrences, after the MPA was designated and MBCG regulations came into force, albeit, without statistical significance.** While the use of gill nets is documented to have a lower impact on benthic communities and structure than MBCGs, use of gill nets increases the risk of bycatch of marine mammals, including harbour porpoise. Therefore, any shift from MBCG to gill nets that is brought about by the MBCG regulations trades off the protection of benthic habitats (reefs and sand banks) against the protection of harbour porpoise bycatch.

The overall combined effect of SAC designation and MBCG regulations is a decrease in fisheries activities in the MPA. The protection of the reef and sand-bank habitats could be thought to have an effect on the production or abundance of exploited fish species, as a subsequent or indirect effect of trying to attain other conservation goals. The export of this extra production from the MPA to surrounding fisheries is termed 'spillover' and is an outcome that can demonstrate the value of MPAs to fishers and fisher communities, over and above the desired conservation goals. In this case, the continued fishing activities within the MPA show no evidence of an increase in LPUE, which is a proxy for abundance. Without an increase in abundance within the MPA, spillover of biomass to the surrounding areas is less likely. Therefore, we can deduce that fisheries spillover from this MPA is unlikely at this stage in its development. However, the longer an MPA is left with reduced pressures, the more mature the community becomes and the more likely spillover can occur and to greater (although marginally) magnitudes (Di Lorenzo et al., 2020). Future studies of the AGRB MPA should investigate both relative abundance of fish within and outside the MPA as well as attempt to trace movements of individuals from the MPA to detect whether or when the MPA produces this spillover.

6.3. Broader marine management

The SAC designation restricted other human activities in the MPA prior to the fisheries regulations, such as the extraction of sediments. These other human activities are becoming more spatially restricted as Denmark implements its marine spatial plan, with areas dedicated to conservation, mineral extraction, sediment extraction, navigation and renewable energy installations, among others (Havstrategiloven, Miljø- og Fødevaremin., j.nr. 2019-8697, 2019). An aspect of this spatial allocation of activities is the exclusion of others. One activity that does not have explicitly allocated areas is fishing. While we are able to show that fishing effort decreased in the MPA relative to some select control sites in response to MBCG regulations, we cannot say how this changed effort across the broader region. With policies excluding fishers from more areas for different purposes, it could be assumed that fishing effort will become more and more concentrated on few fishing grounds. This broader national, or even regional, level question of effort concentration and subsequent environmental impacts should be addressed in future work.

In the south-western Baltic Sea, many fisheries have been in decline, including the highly socially and economically valued Atlantic cod (ICES, 2023). With fewer fishing opportunities and an ultimate collapse of the two stocks that support the fisheries in the region around the AGRB MPA there are greater economic forces acting on fisher behaviour than simple area-based restrictions. Equally, the volatility of the size and distribution of the small pelagic fisheries drives variation in the effort and landings from any given area of the south-western Baltic Sea. While the AGRB MPA seeks to conserve the harbour porpoise population, any developments in population size should be presented in the context of decreased fishing opportunities for cod and small pelagics, and increased fishing opportunities for flatfish.

6.4. Limitations

The approach used in this case study comes from a long pedigree of ecological methodology. However, the methodology can only provide results as good as the input data. In this case, we have fairly detailed fishing records; however, these are limited to a relatively short time series, extending only a few years either side of the two interventions considered (SAC designation and MBCG regulations coming into force). Furthermore, the level of fishing activity in the MPA has always been quite low, especially for certain gear types (for example, gill nets). This leads to many years with zero or very low effort, making some analyses impossible and leaving very low contrasts in data for detecting significant differences.

While our data on fishing effort is derived from relatively high-resolution position data, the exact location of fishing activities in smaller vessels is determined by simple speed filters. While this approach is widely used when determining fishing position from AIS data, more complex modelling methods (for example, random forest models of various input variables (ICES, 2022)) that are currently in development achieve greater accuracy and should be employed to provide fisheries data to analyses such as the current report presents, in future studies.

Furthermore, the allocation of landings data to fishing activities is most often averaged by day or trip, meaning that the total landings from a day are shared across the various fishing activities undertaken that day. This reduces the contrast in our data between MPA and control areas and so makes finding differences more difficult. Thus, while this is a

limitation, it means that our inferences are more conservative than if we were better able to allocate landings.

These analyses consider only the Danish fleet. German fishers and to a lesser extent Swedish and Polish fishers have utilised this area historically, so having data from these countries' vessels would better represent the full picture. However, Danish vessels were historically the most dominant and operated at the largest scale, and therefore, should be representative of the overall changes occurring in the MPA.

7. CONCLUSIONS

While the designation of the Adler Grund og Rønne Banke MPA triggered changes to some human activities in the area, fisheries effort (and subsequent landings) appears to respond only to specific fisheries regulations.

Regulations on MBCG had a significant effect on the effort taking place inside the MPA. Seemingly unintuitively, this effect was only significant across the whole MPA, and not within the sub-set of the MPA where the regulations applied. There was a general increase in gill-net-fisheries effort that corresponded with the imposition of MBCG regulations, but this effort was not significantly different in the MPA compared to control sites. The overall reduction in effort within the MPA may not necessarily lead to increased effort outside the MPA, because fishing opportunities for key species are, in general, decreasing in the region. Any effort that is displaced from the MPA will also be restricted by the marine spatial plan's exclusion of fisheries from areas allocated to other uses, such as wind energy installations.

The AGRB MPA illustrates how trade-offs must be made when deciding between no-take MPAs and targeted restrictions, while also attempting to address/prioritise all conservation objectives assigned to an MPA. In this case, the MBCG regulations aimed to reduce impacts on reefs, indirectly reduced impacts on sand banks, but potentially indirectly increased gill-net activities in the broader region, increasing the risk of bycatch of harbour porpoise.

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Case Study Report

The Bratten Offshore Natura 2000 Area Sweden – Skagerrak



Example of the diverse deep reef habitats in Bratten with the basket star *Gorgonocephalus caputmedusae* sitting on the fan coral *Paramuricea placomus* and surrounded by the sponges *Mycale lingua* (upper right) and *Phakellia ventilabrium* (lower left). Photo by Tomas Lundälv

Mapping of marine protected areas and their associated fishing activities

Mattias Sköld and Patrik Jonsson

***Swedish University of Agricultural Sciences
Department of Aquatic Resources***

Sweden

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LIST OF ABBREVIATIONS

Term	Description
AIS	Automatic Identification System
ANOVA	Analysis of Variance
BACI	Before-After, Control-Impact
CFP	Common Fisheries Policy
EEZ	Exclusive Economic Zone
EU	European Union
ICES	International Council for the Exploration of the Sea
MPA	Marine Protected Area
NGO	Non-Governmental Organisation
OSPAR	The Convention for the Protection of the Marine Environment of the North-East Atlantic
ROV	Remotely Operated Vehicle
SAC	Special Area of Conservation
SwAM	Swedish Agency for Marine and Water Management
VMS	Vessel Monitoring System

1. EXECUTIVE SUMMARY

In this case study we have evaluated the potential displacement of the fishery within and around the Bratten marine protected area (MPA). This is an area with species and habitats of high conservation value with spectacular bathymetry characterised by steep rock walls, canyons and pockmarks on the slope (100 – 500 m) towards the Norwegian Deep in the Skagerrak. The exposed rock walls host deep water Coral gardens, sponge communities, large predatory fish, and dense sea pen fields in the surrounding soft bottoms.

The area was designated as a Natura 2000 site in 2011 for reef and pockmarks according to the Habitats Directive and is part of OSPAR's network of marine protected areas. A major challenge for the management and conservation of the area is that Bratten is situated within one of the most important fishing grounds in the Skagerrak for Northern shrimp *Pandalus borealis* and demersal fish, located outside territorial waters in the Swedish exclusive economic zone, and intensively fished by bottom trawlers from Sweden and Denmark. In addition, the integration between the EU nature conservation policy and the Common Fisheries Policy (CFP) was poorly developed during the early phases of the management process. This made the application of an ecosystem approach to fisheries management in practice slow and difficult. The aim of the measures finally established, was to ensure adequate protection from harmful fishing activities interacting with designated habitats and species in the Bratten MPA. The measures include the establishment of no-take zones where all fisheries are prohibited, and for control purposes compulsory use of automatic identification system (AIS) for all vessels fishing in the area.

This case study compared fishing activity before and after closure of no-take zones within the Bratten MPA. The potential displacement of the Swedish trawl fishery was analysed using vessel monitoring systems (VMS) data coupled with logbook information. Findings show that reallocation of the *Pandalus* fishery occurred within the MPA following enforcement of the no-take zones. This displacement of the trawl fishery complies with the intention of the measures to protect the highly valued reef and pockmark habitats from physical disturbance along with nearby seapen fields. Trade-offs were made to allow for trawlers to utilize passages through the area. The trawlers accordingly ceased to fish in the no-take zones and intensified their efforts in the passages between zones and to the north-east within the MPA. There was no significant reduction in fishing effort within the MPA, and we found no indications of displacement to areas outside the Bratten MPA. Rather, the variability in effort within the MPA correlated with the effort and fishing opportunities linked to the overall variation in availability of northern shrimp between years within the Skagerrak.

The fishery regulations in the Bratten MPA were negotiated with fishers' organisations from Sweden and Denmark, and authorities considered the arguments from the fishers that it was important to keep passages through the area open. This may explain that effort could be withheld within the MPA, and that compliance with the regulations has been high. In addition, the regulations have been strongly enforced by detailed vessel monitoring covering essentially all vessels operating in the MPA. Conservation targets need long-term monitoring to fully evaluate the effects of fisheries regulations and conservation status. A monitoring program to study the effects of the fishery regulations is in place.

2. BACKGROUND

Marine protected areas (MPAs) with fishery measures may lead to redistribution of the regulated fishing activities to surrounding areas because of loss of fishing grounds (Murawski et al., 2005). Decreased effort within MPAs may then be balanced by increased effort outside the protected areas. Unless considered in the planning process, intensification of fishing in the areas surrounding MPAs could have negative effects on sensitive habitats and / or non-target species. For the fishing industry, the catch per unit effort might be decreased and fleet behaviour changed leading to higher costs (Hilborn et al., 2021). The MPAs that may provide the highest conservation benefits are likely those in which high levels of protection are imposed. For example, the inclusion of areas of full protection (i.e., no-take zones), prohibiting extractive activities (Edgar et al., 2014). To better understand the fishers' responses to MPA establishment, it is thus essential not only to explore the spatial distribution of fishing activities and stakeholders' perceptions before establishing MPAs, but also to investigate the effects of reallocation (displacement) after enforcement.

2.1. The Bratten MPA and the process of establishment

The Bratten MPA is a large area (1,209 km²) hosting species and habitats of high conservation value¹ with spectacular bathymetry characterised by steep rock walls, canyons and pockmarks on the slope (100–500 m) towards the Norwegian Deep in the Skagerrak (Figure 1). The exposed rock walls host deep-water coral gardens, sponge communities, large predatory fish, and dense seapen fields in the surrounding soft bottoms. The area was designated as a Natura 2000 site (Special Area of Conservation, SAC) for reef structures in 2011 and is part of the Convention for the Protection of the Marine Environment of the North-East Atlantic's (OSPAR's) network of MPAs.

A major challenge for the management and conservation of the area was that Bratten is situated within one of the most important fishing grounds in the Skagerrak for Northern shrimp *Pandalus borealis* and demersal fish, located outside territorial waters in the Swedish exclusive economic zone (EEZ), and intensively fished mainly by bottom trawlers from Sweden and Denmark. In addition, the integration between the EU Habitats Directive, the management of the Natura 2000 network and the Common Fisheries Policy (CFP) was poorly developed during the early phases of the process.

The fisheries conservation measures were based on a proposal from the County Administrative Board of Västra Götaland, undertaken within a European Union (EU) Interreg project: Hav möter Land (Sea meets Land). The Interreg project ran between 2010 and 2013 with the involvement of Sweden (lead partner), Norway and Denmark. In Sweden, the County Administrative Board is responsible for the management of the Natura 2000 sites, and the Swedish Agency for Marine and Water Management (SwAM) is the authority implementing the proposed fisheries measures. Through an assignment, SwAM was given the mandate from the government to conduct the international negotiations on the basis of articles 11 and 18 in the CFP with concerned Member States with the aim of formulating a joint recommendation for the Bratten area (SwAM, 2016). The aim of the

¹ Habitats directive: reef 1170. OSPAR list of threatened and declining species and habitats: Seapen and burrowing megafauna communities, Coral gardens, Deep sea sponge aggregations

measures finally established was to ensure adequate protection from harmful fishing activities interacting with designated habitats and species in Bratten.

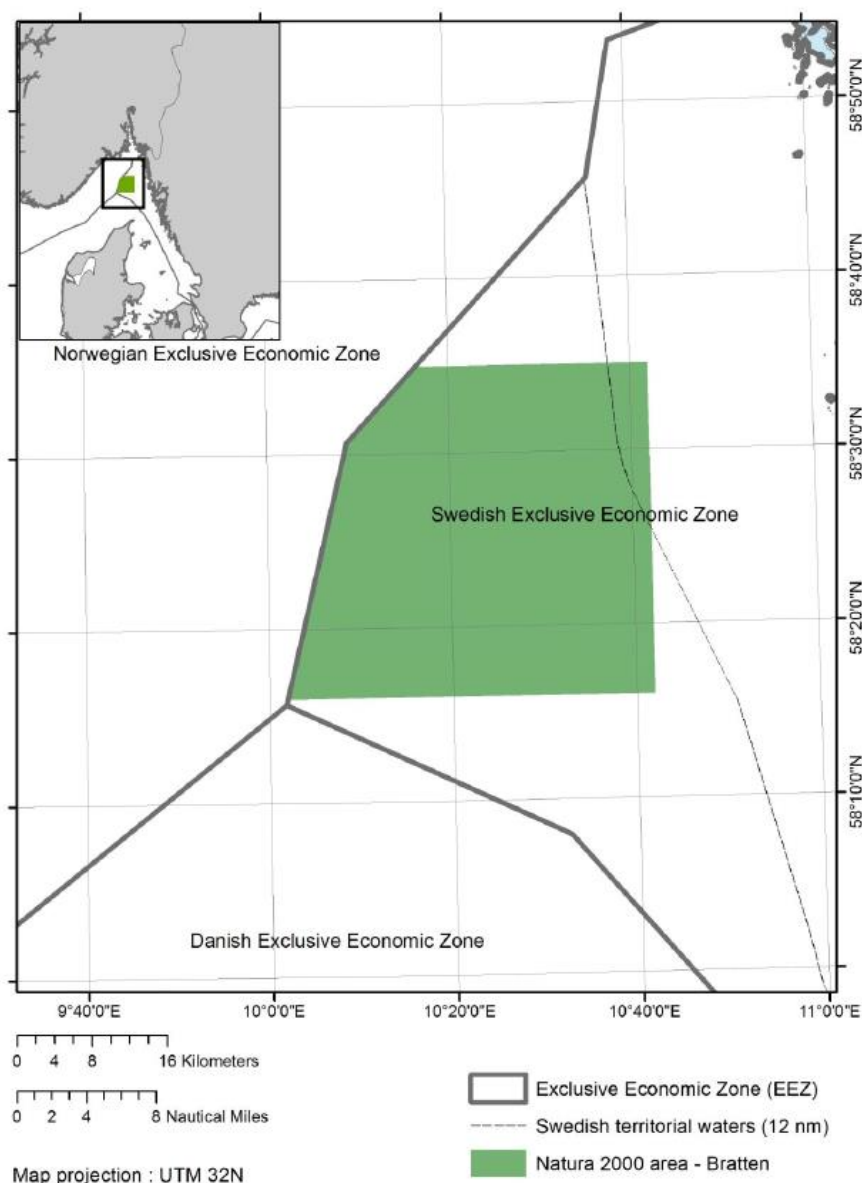


Figure 1. Location of the Bratten MPA in the Skagerrak (SwAM, 2016).

2.2. Commercial fisheries and regulations

The proposal for fisheries regulations was developed in extensive dialogues with representatives of the fishing industry, sport fishermen, various authorities and research institutions from Sweden, Norway and Denmark. Environmental non-governmental organisations (NGOs) were invited to comment on the proposal. The proposed measures were contested by the fishing industry claiming that high economical values were at stake. On the contrary the environmental NGOs argued that larger closures within Bratten were necessary to ensure adequate protection of the biological features. The Swedish and Danish fishery target Northern shrimp *P. borealis*, Norway lobster *Nephrops norvegicus* and demersal fish using bottom otter trawls and are very active in the area. Fishers' activity

was mapped and considered in the design of the closures to allow for passages of trawlers with the gears deployed through parts of the area. The measures include the establishment of no-take zones covering 27 % of the MPA, where all fisheries are prohibited, and for control purposes compulsory use of an automatic identification system (AIS) for all vessels fishing in the area. The regulations for commercial fisheries were regulated by the European Commission and enforced in 2017 (EU, 2017) (Figure 2).

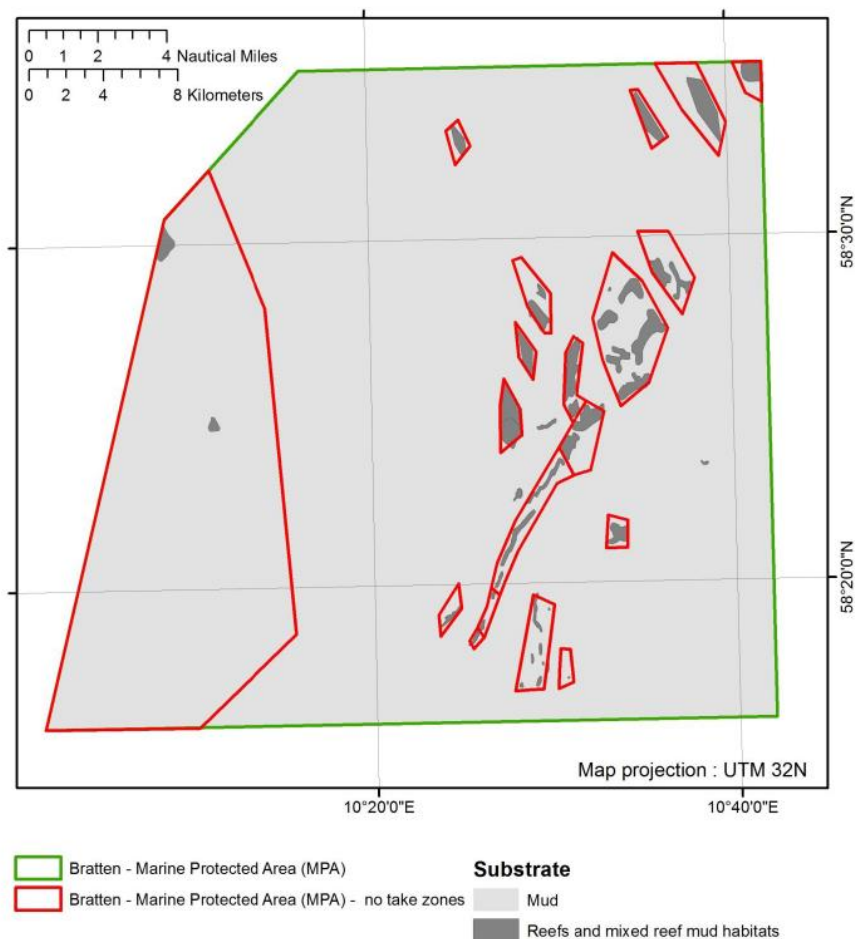


Figure 2. Map of major habitats and closures in the Bratten MPA (SwAM, 2016).

2.3. Recreational fisheries

The practice of recreational fishing is limited in the area, and threats to designated species and habitats are in general considered to be relatively low; however, remotely operated vehicle (ROV) surveys have repeatedly shown torn fishing lines around corals, and in many cases fatal effects on colonies. It has, however, been debated whether these lines are from the recreational fishery or from commercial fisheries, e.g. demersal long lines, but instances where recreational fishermen have caught specimens of corals have been reported. Recreational fisheries focus on large specimens and sensitive species groups such as sharks and rays. In the case of directed fishery for large individuals of the relatively stationary tusk *Brosme brosme*, effects on local populations cannot be excluded. Even if 'catch and release' is applied when fishing for skates, rays and sharks, handling will lead to an increased risk of injury that may increase mortality, and all species with a swim bladder risk severe injury and mortality due to barotrauma. Fisheries conservation

measures for recreational fisheries were enforced in 2017 through the Swedish national legislation by closing several of the zones also to recreational fisheries (i.e. no-take zones; Figure 2).

When introducing regulations for commercial and recreational fisheries in the Bratten MPA, efforts were made to minimise the cost to ongoing fisheries while securing the conservation targets within the area. The resulting consequences for fisheries will likely result in avoidance of the no-take zones but may have wider consequences for the behaviour of the fishery since the spatial allocation of trawl hauls covering long distances may be changed, as well as the effort deployed. This may affect the economy of the fishery because of changes in catches and effort, including use of fuel.

3. AIMS AND OBJECTIVES

The main aim of this report is to assess spatial reallocation of commercial fishing activities in response to fishing regulations in the Bratten MPA.

The specific objectives of this case study were to analyse the potential displacement of the bottom otter trawl fisheries from the no-take zones within the Bratten MPA; and analyse the potential displacement of the bottom otter trawl fisheries from the Bratten MPA to the surrounding sea area.

4. METHODOLOGY

The study compared fishing activity before (2013–2016) and after (2017–2022) closure of the no-take zones within the Bratten MPA. The displacement of the Swedish trawl fishery was analysed from hourly satellite vessel monitoring system (VMS) data coupled with logbook information using the VMStools R package (Hintzen et al., 2012) and applying speed filters to detect fishing activity (Gerritsen and Lordan, 2011).

Fisheries affected by the closures were bottom otter trawl fisheries grouped into four métiers: northern shrimp (*Pandalus borealis*), demersal fish (Demersal_120), mixed crustacean and demersal fish (MCD) and Norway lobster otter trawlers using sorting grid (Nephrops_grid). The vessels covered over the study period were all ≥ 12 m length, which comprises essentially all the active vessels in this offshore area. VMS and logbook data was obtained from the Swedish Agency for Marine and Water Management.

After quantitatively having analysed the different métiers, we concluded that the overall dominating fishery was for *Pandalus*. Accordingly, the displacement of the otter trawl fisheries is illustrated as the difference in VMS pings within 500 x 500 m grid within the area by the dominating *Pandalus* fishery and the yearly average for the periods before and after the closure on the same grid.

A Before-After, Control-Impact comparison (BACI) for the Bratten as a whole was performed for the dominating *Pandalus* fishery by comparing the fishing hours spent within the Bratten area with the hours spent outside in the Skagerrak, before and after the closure of the no-take zones in Bratten in 2017 using a two-factor analysis of variance (ANOVA) with square-root transformed data on fishing hours per year.

We also analysed a potential correlation in hours spent inside the Bratten MPA with hours spent in the fishery outside in the Skagerrak.

Due to lack of data from other nations having access to the area, the assessment was only done for the Swedish commercial fishery. No analysis was done for the recreational fishery because of the absence of data.

5. RESULTS

A table was constructed for each fishery by analysing hours spent per year within the no-take zones, the Bratten MPA and the Skagerrak (outside and in total) to evaluate the temporal evolution of the fisheries and identify potential difference before and after closure of the no-take zones (Table 1).

Overall, the *Pandalus* fishery with landings of mainly Northern shrimp but also saithe *Pollachius virens*, witch flounder *Glyptocephalus cynoglossus* and cod *Gadus morhua* was dominant in both the Bratten (93–99 % of the total hours) and the closed zones. The Demersal 120 fish fishery use larger meshes (120 mm) in the bottom trawl and lands a mixture of fish species mainly witch flounder, saithe and cod. The mixed crustacean and demersal fish (MCD) use slightly smaller meshes (90 mm) lands essentially the same as the Demersal 120 fishery but have a larger share of Norway lobster *Nephrops norvegicus*. Selective bottom trawling for Norway lobster with sorting grid is of minor importance in the Bratten MPA (Table 1).

As expected, the *Pandalus* trawl fishery decreased sharply in the no-take zones after implementation of the regulations in 2017. The remaining hours detected in the zones averaged around 28 per year, likely due to steaming at slow speed or illegal fishing activities.

The displacement of the *Pandalus* trawlers occurred within the Bratten MPA to the north-east and was concentrated in three available passages between the no-take zones (Figure 3 and Figure 4). No overall displacement effect of the *Pandalus* fishery to areas outside the Bratten MPA in the Skagerrak could be detected analysed as a BACI comparison (two-factor ANOVA $F_{1,1} = 1.52$, $p = 0.24$). On average, per year, 8 313 hours were spent within the Bratten MPA and 34 671 outside in the Skagerrak. The hours spent trawling for *Pandalus* within Bratten in the period 2013–2022 varied and correlated with hours spent outside in the Skagerrak ($R^2 = 0.52$, $p = 0.02$; Figure 5). The yearly catches of *Pandalus* estimated by ICES for the same period correlated with the total hours spent trawling for *Pandalus* in the Skagerrak, albeit only at a significance level of $\alpha = 0.10$. ($R^2 = 0.35$, $p = 0.07$).

Table 1. Swedish fishing hours spent per year within the Bratten MPA, the no-take zones within Bratten (Bratten No Take), outside the Bratten MPA in the Skagerrak and in total in the Skagerrak.

		Before closure of the no-take zones				After closure of the no-take zones					
Metier	Area	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Pandalus	Bratten MPA	9,177	9,552	6,454	9,182	8,139	9,435	6,454	8,170	8,538	6,016
	Bratten No Take	626	566	305	354	64	47	11	12	19	13
	Outside MPA	38,401	39,932	33,353	40,713	34,517	31,914	31,152	34,457	33,246	29,026
	Skagerrak total	48,204	50,050	40,111	50,429	42,720	41,396	37,617	42,640	41,802	35,055
Demersal 120	Bratten MPA	35	429	494	299	435	307	74	51	125	111
	Bratten No Take	1	6	15	9	1	1	-	-	1	-
	Outside MPA	2,017	4,130	4,445	3,674	4,561	6,827	5,377	4,299	5,488	5,427
	Skagerrak total	2,053	4,566	4,954	3,983	4,997	7,135	5,451	4,350	5,614	5,538
MCD	Bratten MPA	496	192	8	12	75	114	93	22	161	49
	Bratten No Take	37	5	-	-	5	2	-	1	2	2
	Outside MPA	10,720	7,679	8,141	8,748	12,992	10,864	15,523	16,211	17,371	14,455
	Skagerrak total	11,252	7,876	8,149	8,760	13,072	10,980	15,616	16,234	17,535	14,506
Nephrops_grid	Bratten MPA	-	-	-	1	4	-	11	4	14	-
	Bratten No Take	-	-	-	-	-	-	-	-	-	-
	Outside MPA	16,502	18,804	14,256	17,098	10,216	10,005	15,128	15,861	19,159	17,500
	Skagerrak total	16,502	18,804	14,256	17,099	10,020	10,005	15,139	15,865	19,173	17,500

NB. Fisheries using bottom otter trawls are: Pandalus = Northern shrimp (*Pandalus borealis*), Demersal_120 = demersal fish, MCD = mixed crustacean and fish, and Nephrops_grid = Norway lobster (*Nephrops norvegicus*) using sorting grid.

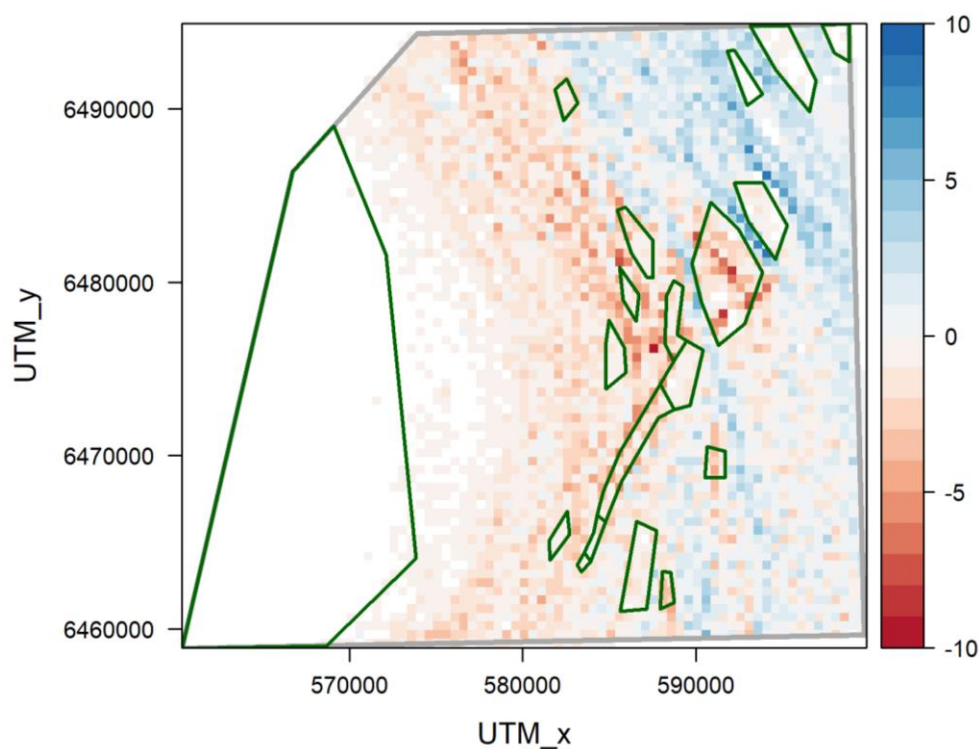


Figure 3. Displacement as difference in VMS pings (right scalebar with blue colour indicate increase in effort, red indicate decrease) of the Swedish *Pandalus* bottom otter trawl fishery within the Bratten MPA comparing effort in 500 X 500 m grid cells before (2013–2016) with after (2017–2022) enforcement of the closed no-take zones within the Bratten MPA. The Bratten MPA is delineated in grey; closed no-take zones are delineated in green.

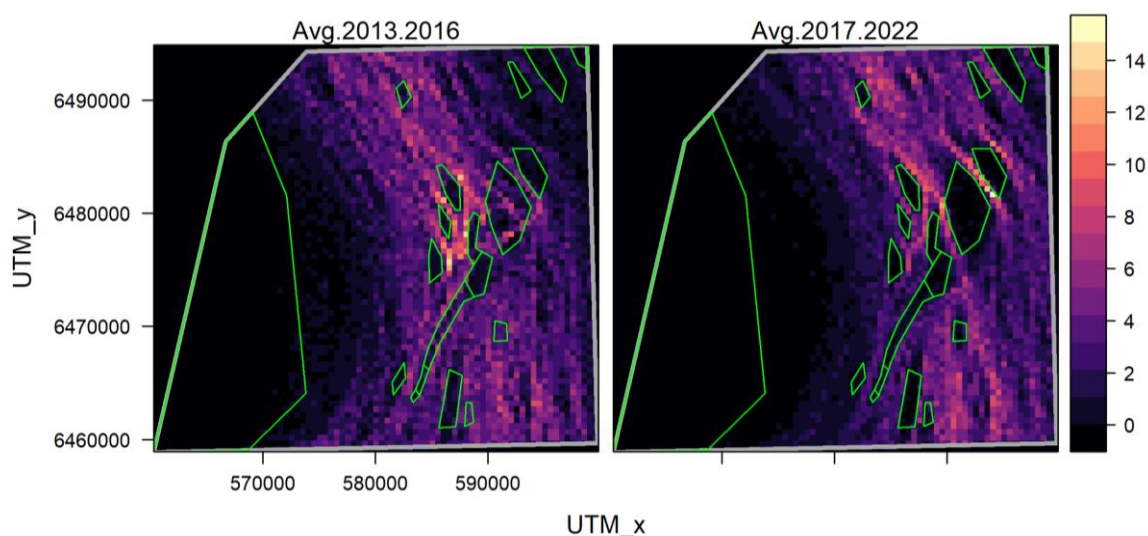


Figure 4. Averaged yearly VMS pings in the Swedish *Pandalus* bottom otter trawl fishery within the Bratten MPA in 500 X 500 m grid cells before (2013–2016 left) and after (2017–2022 right) enforcement of the closed no-take zones (green borders).

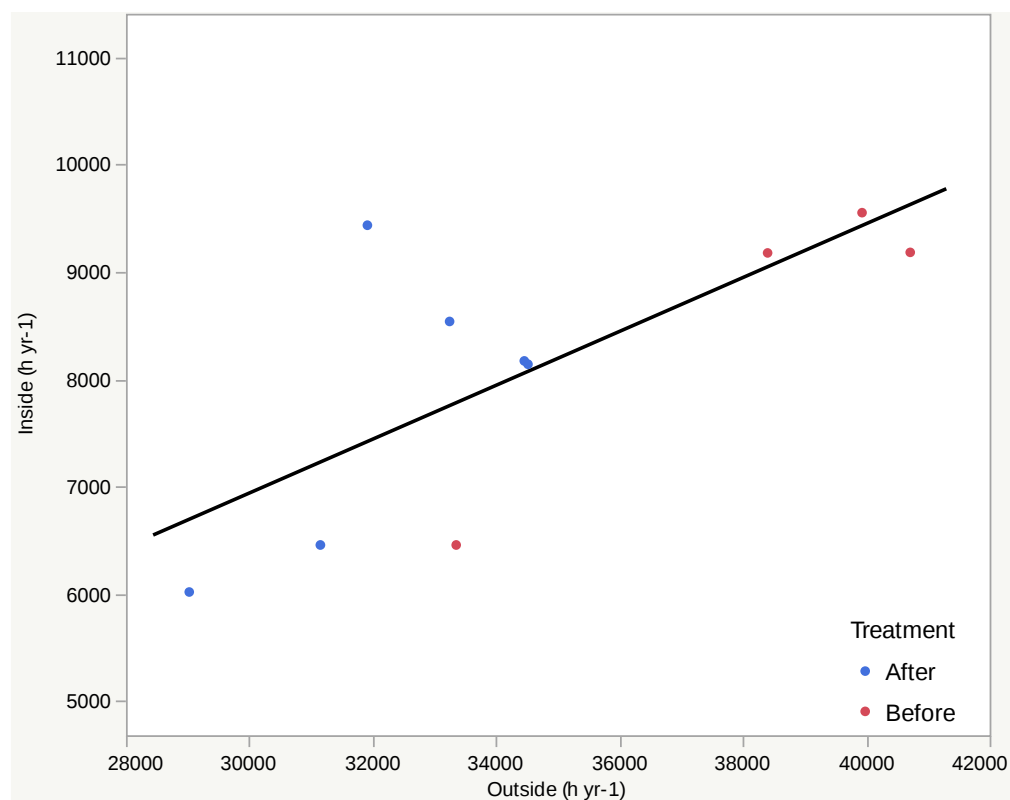


Figure 5. Correlation between hours spent trawling for *Pandalus* per year outside the Bratten MPA in the Skagerrak versus inside the MPA. Red = 2013–2016, before the closure of the zones; blue = 2017–2022, after the no-take zones were enforced.

6. DISCUSSION

The Bratten MPA, a Natura 2000 SAC area, is designed mainly to protect reef habitats and the deeper canyons and pockmarks in the area, and the areas surrounding these habitats, which include soft seafloor with sea pens and burrowing megafauna (SwAM, 2016). Our analysis indicated that displacement of the dominating *Pandalus* fishery following the enforcement in 2017 occurred within the Bratten MPA to the north-east and was concentrated in three available passages between the closed no-take zones. As communicated by the fishers during the negotiation process for fishing regulations in Bratten, passages mainly in the north to south direction were of vital importance to the ongoing fishery in the area as the trawlers follow the slope, and that hauls continue for long distances through the area. The most utilised passages identified in the Bratten area were kept open to minimise the cost to fisheries and the conflict between conservation targets and the trawl fishery. This work shows that trawler effort has indeed increased in these areas, as expected. The large no-take zone in the west had very little effort the years prior to closure, and no significant shift in effort from that area was expected.

The concentration of effort to the passages indicates a high level of compliance with the regulations; this can be attributed to the trade-offs made in the process of establishing the no-take zones and acceptance by the fishermen, as well as strict enforcement and control of the fishery. All vessels are equipped and continuously monitored by the

authorities with VMS. AIS is mandatory for all fishing vessels entering the area, and VMS for all vessels $\geq 12\text{m}$ (EU, 2017).

For the Bratten MPA as a whole, we analysed the potential displacement to the outside Bratten within the Skagerrak for the dominating *Pandalus* fishery. Our results showed no differences in trends over time inside versus outside the MPA. The effort inside the MPA differed between years but correlated to the effort spent in the Skagerrak as a whole. We interpret this to be effects of overall fishing opportunities and catch efficiency linked to the overall variation in availability of *Pandalus* between years, and not an effect linked to the displacement of the trawlers within the Bratten MPA.

The Bratten MPA is situated in the Swedish EEZ outside territorial waters, where other nations have access. Bratten is mainly an important fishing ground for Danish fisheries. It is an important catch area for halibut *Hippoglossus hippoglossus*, with 61 % of landings from the Skagerrak area originating from the MPA in 2014. However, in terms of landing volume, northern shrimp also dominates the Danish landings from the area with amounts about the same as the Swedish landings of northern shrimp per year (SwAM, 2016). Before the enforcement of the closures, the Danish fishery had a similar spatial distribution to that of the Swedish fishery but slightly more concentrated to the north-east (SwAM 2016). However, since we didn't manage to access these data, this study is limited to analysis of displacement of the Swedish fishery.

Increased local effort may lead to local depletion of fish resources (Bartolino et al., 2012), and it would be interesting to investigate whether the finer-scale displacement has led to changes in catches in the trawl hauls within the Bratten MPA, in particular in those hauls where the fishery has concentrated. However, this was not feasible with the available data since landings are reported by trip, which could cover several days, and not individual hauls. In addition, even individual hauls are long and continuous over long distances in the *Pandalus* fishery (4–11 hours at approximately 2–2.5 knots) and may cross the borders of the Bratten MPA as well as the areas where the fishery has concentrated. In summary, given the spatial resolution of available data, we were not able to make a constructive displacement analysis on potential impact on fine-scale catches. In addition, in contrast to the fishery-independent dataset of VMS observations used in this study, logbook and sale slips depend on reports from the fishery, which might be influenced by, for example, discards of catches or misreporting that may influence the outcome of the results. Changes in reporting by the fishery are further influenced by the enforcement of the EU landing obligation (EU, 2019) during the study period.

In summary, **reallocation of the *Pandalus* fishery has occurred within the Bratten MPA following enforcement of the no-take zones**. With regard to the intention of the measures to protect the highly valued reef and pockmark habitats from physical disturbance along with nearby sea pen fields, and the trade-offs made to allow for trawlers to utilize passages through the area, we conclude that the **objective to remove threats from direct impact of bottom trawling have been met within the Bratten MPA**. However, indirect effects from, for example, resuspension of sediments (Linders et al., 2017) or food web interactions (van de Wolfshaar et al., 2020), may still occur.

Conservation targets like coral garden habitats and sea pen fields need long-term monitoring to fully evaluate the effects of fisheries regulations and conservation status. A monitoring program to study the effects of the fishery regulations started in 2019 will be reviewed every 3 years. The focus of these studies includes the recovery of sea pen

fields, infauna and hard bottom organisms, in particular coral gardens and deep-sea sponge communities.

7. CONCLUSIONS

Reallocation of the *Pandalus* fishery occurred within the Bratten MPA following enforcement of the no-take zones. This displacement of the trawl fishery complies with the intention of the measures to protect the highly valued reef and pockmark habitats from physical disturbance along with nearby sea pen fields. Trade-offs were made to allow for trawlers to utilize passages through the area. The trawlers accordingly ceased to fish in the no-take zones and intensified their efforts in the passages between zones and to the north-east within the MPA. There was **no significant reduction in fishing effort within the MPA**, and we found **no indications of displacement to areas outside the Bratten MPA**. Rather, the variability in effort within the MPA correlated with the effort and fishing opportunities linked to the overall variation in availability of northern shrimp between years within the Skagerrak.

The fishery regulations in the Bratten MPA were negotiated with fishers' organisations from Sweden and Denmark, and authorities considered the arguments from the fishers that it was important to keep passages through the area open. This may explain that effort could be withheld within the MPA, and that compliance with the regulations has been high. In addition, the regulations have been strongly enforced by detailed vessel monitoring covering essentially all vessels operating in the MPA.

Conservation targets need long-term monitoring to fully evaluate the effects of fisheries regulations and conservation status. A monitoring program to study the effects of the fishery regulations started in 2019.

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Case Study Report

Dundalk Bay Ireland – Celtic Seas



Mapping of marine protected areas and their associated fishing activities

Aylis Emerit, Oliver Tully, David Reid

Marine Institute, Rinvile, Oranmore, Co. Galway

Ireland

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LIST OF ABBREVIATIONS

Term	Description
BACI	Before-After Control-Impact
BIM	Bord Iascaigh Mhara
CFP	Common Fisheries Policy
CI	Control Impact
DAFM	Department of Agriculture, Food and the Marine
DHLGH	Department of Housing, Local Government and Heritage
EU	European Union
FND	Fishery Natura Declaration
FNP	Fishery Natura Plan
IUCN	International Union for Conservation of Nature
I-WeBS	Irish Wetland Bird Survey
MPA	Marine Protected Area
NIS	North Irish Sea
NPWS	National Parks and Wildlife Service
SAC	Special Area of Conservation
SCI	Site of Community Importance
SFPA	Sea-Fisheries Protection Authority
SPA	Special Protection Area
TAC	Total Allowable Catch
VMS/iVMS	Vessel Monitoring System(s)/Inshore Vessel Monitoring System(s)

1. EXECUTIVE SUMMARY

Dundalk Bay, Co. Louth, in Ireland is a shallow east-facing bay, opening into the Irish Sea. The bay was first designated as a Special Protection Area (SPA) under the European Union (EU) Birds Directive in 1994, as the site is a regular feeding ground for important communities of wintering waterbirds that are of national and international importance and of significant conservation value. The bay was later designated as a Site of Community Importance (SCI) in 2002, and later as a Special Area of Conservation (SAC) under the EU Habitats Directive in 2019 for its extensive saltmarshes and intertidal sand/mudflats that hold rich fauna of bivalves, molluscs, crustaceans and marine worms.

Commercial hydraulic dredging for cockles and razor clam occurs annually in the SAC and SPA respectively; this is managed through fishery management plans tailored to those specific fisheries. This report describes how fisheries have responded to the designation of the site as a marine protected area (MPA) and how the management of fisheries has been adapted in response to the designation. The report therefore synthesises all documentation on the MPA, the fisheries occurring within it, mitigation measures to manage the impact of fishing on the site and monitoring programmes that provide evidence of the efficacy of mitigation.

This report finds that the favourable conservation status of ecological features in the site can be maintained with the co-existence of managed fisheries. Management of fishing activity has been achieved through explicit and detailed fishery management plans that define how the fishery operates in the site. These plans have been implemented successfully for cockle fisheries over the past 15 years without any displacement of cockle fishing from the SAC. Fishing for razor clams has not been displaced but annual monitoring of the fishery with an Inshore Vessel Monitoring System (iVMS), razor clam and benthic bivalve surveys and periodic aerial digital surveys for seabirds that feed on bivalves are undertaken. A closed area of approximately 3 km² was introduced in the razor clam fishing area in 2023 to monitor changes in benthic communities following removal of fishing pressure. This closure has not led to any significant displacement of fishing from the SAC.

Limiting entry to fisheries in Natura 2000 sites increases the potential for fishing stakeholders to make better and long-term decisions and this can be achieved through a co-operative approach between conservation and fishery authorities and fishers. This is likely to be more successful than top-down management as prior communication by conservation authorities of their intention to designate sites is very important to fishers. The primary objective of fishery plans in a Natura 2000 site is protection of the features for which the site is designated. Fishing for economic gain is a secondary objective and can only be achieved if the first objective is met. This is why periodic reviews of the status of the site and the fishery plan is important to, if necessary, reset the conditions under which fisheries operate in Natura 2000 sites, and to identify at an early-stage risk of deterioration of designated features. Monitoring data is thus essential to the proper implementation of fishery plans in Natura 2000 sites.

2. BACKGROUND

Dundalk Bay, situated in County Louth, is a shallow and east-facing bay opening into the Irish Sea to the east of Ireland (ICES Celtic Seas ecoregion); it has a centre point of longitude -6.3384 and latitude 53.9586. Formed under the Natura 2000 European network of protected areas aiming to protect habitats and species of importance at the European scale (Council Directive 92/43/EEC; Directive 2009/147/EC), Dundalk Bay was designated as a Site of Community Importance (SCI) in 2002 under the Habitats Directive (92/43/EEC) and later as a Special Area of Conservation (SAC) in 2019; this latter designation aimed mainly at protecting estuarine and intertidal mud and sand flats (Table 1). The bay had also been designated as a Special Protection Area (SPA) under the Birds Directive (2009/147/EC) in 1994. At a national level in Ireland, the designation of Dundalk Bay (and all such protected areas) is the responsibility of the Heritage Division of the Department of Housing, Local Government and Heritage (DHLGH), and the National Parks and Wildlife Service (NPWS) within that Department (DHLGH, 2020).

Table 1. Habitats of qualifying interests*, listed under Annex I of the EU Habitats Directive for Dundalk Bay SAC (site code 000455).

Habitat Code	Site Name
1130	Estuaries
1140	Mudflats and sandflats not covered by seawater at low tide
1220	Perennial vegetation of stony banks
1310	<i>Salicornia</i> and other annuals colonising mud and sand
1330	Atlantic salt meadows (<i>Glauco-Puccinellietalia maritima</i>)
1410	Mediterranean salt meadows (<i>Juncetalia maritimi</i>)

* Priority habitats under the Habitats Directive.

The Habitats Directive was transposed into Irish law in the European Communities (Birds and Natural Habitats) Regulations of 2011, as amended in 2021. However, Habitats and Birds (Habitats and Birds Directives) regulations for sea fisheries are laid out in the following 2009 regulations: Natural Habitats and Birds; Sea-fisheries 2009 (S.I. 346 of 2009 as amended by S.I. 397 of 2010, S.I. 237 of 2012, S.I. 457 of 2021, S.I. 293/2021). These regulations relate more specifically to the impact of sea-fisheries on the marine environment. To ensure compliance with the requirements of the Habitats and Birds Directives, which require activities to be compatible with the conservation objectives for which the sites are designated, a fishery management plan (Fishery Natura Plan) for Dundalk Bay was first introduced in 2009. This management plan, which, with amendments, has been running since introduction, proposed a series of management measures regulating fishing activity within the bay (DAFM, 2022). The Common Fisheries

Policy (Regulation 1380/2013) ⁽²⁾ also aims to grow the fisheries and aquaculture sectors in the European Union (EU) by ensuring that activities are sustainable.

Within Dundalk Bay, the area designated as a SAC consists of approximately 5,196 hectares (51.96 km²), of which 92.8 % is a marine area (EC, 2020a); the SPA, which extends seaward beyond the border of the SAC has a total area of 13,238 hectares, of which 97.5 % is a marine area (EC, 2020b). The Irish NPWS, the authority responsible for the conservation of these sites, oversees, monitors and reports on the status of the habitats and species for which the site is designated. Parameters such as species populations, demographic structure and distribution, and the distribution and range of benthic community structure and function are reported under Article 12 and Article 17 of the Habitats and Birds Directives. The favourable conservation conditions for a particular habitat or species at that site and the integrity of the site is achieved when these parameters are stable or increasing (NPWS, 2011). The conservation objectives of the SAC and SPA were adopted in 2011, nine years after Dundalk Bay was designated as an SCI.

Within Dundalk Bay, the qualifying habitats – estuaries, mud and sandflats, and saltmarshes – support a diverse array of marine benthic communities (Figure 1). Estuaries, mud and sandflats are dominated by bivalve molluscs, marine worms and crustaceans. Of the 11 intertidal biotopes identified in a 2008–2010 benthic habitat monitoring survey, polychaetes and the bivalve *Macomangulus tenuis* (formerly known as *Angulus tenuis*, MolluscaBase eds. 2023) were found to be most dominant within the Dundalk Bay littoral fine sand community (Clarke and Tully, 2011).

In addition, the fine sand or muddy sand shores hold highly abundant populations of cockle (*Cerastoderma edule*) with accompanying communities of polychaetes including *Eteone longa*, *Scoloplos armiger*, *Pygospio elegans*, *Capitella capitata*, shrimps such as *Crangon crangon*, amphipods such as *Bathyporeia* sp., as well as molluscs such as *Macoma balthica* (Clarke and Tully, 2011). These benthic communities represent the main food resource for waterfowl (including waders and gulls) feeding in the intertidal area of Dundalk Bay (DAHG, 2014).

⁽²⁾ Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC [2013] OJ L354/22.

Mapping of marine protected areas and their associated fishing activities: MAPAFISH Annex 5 - Case study reports

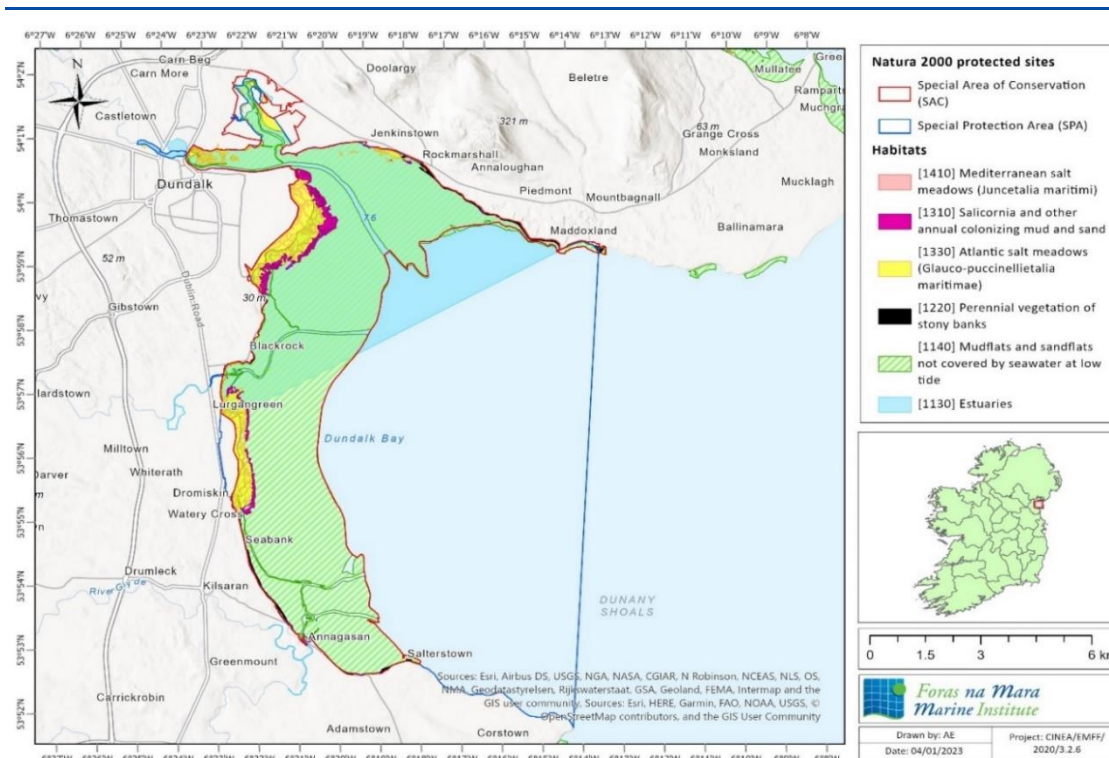


Figure 1. Habitats of qualifying conservation interests listed under the Habitats Directive for Dundalk Bay SAC and SPA.

Whilst the SPA supports the largest concentration of wintering waterfowl on the east coast of Ireland, with more than 20,000 waterbirds regularly using the site (EC, 2020b), the bay also supports 23 species (Table 2) of national importance (i.e. it regularly supports 1 % or more of the all-Ireland estimate of a species) and a further four species of international importance (i.e. it regularly supports 1 % or more of the flyway population of one species or subspecies of waterbird), including the bar-tailed godwit (*Limosa lapponica*), the black-tailed godwit (*Limosa limosa*), the knot (*Calidris canutus*) and the light-bellied brent goose (*Branta bernicla hrota*) (Fitzgerald et al., 2021).

Table 2. Species of qualifying interests, listed under the Birds Directive for Dundalk Bay SPA (site code 004026).

Species Code	Species Name	Legal Protection Status
A005	Great Crested Grebe (<i>Podiceps cristatus</i>)	Article I of the EU Birds Directive
A043	Greylag Goose (<i>Anser anser</i>)	Annex II and III of the EU Birds Directive
A046	Light-bellied Brent Goose (<i>Branta bernicla hrota</i>)	Annex II of the EU Birds Directive
A048	Shelduck (<i>Tadorna tadorna</i>)	Article I of the EU Birds Directive
A052	Teal (<i>Anas crecca</i>)	Annex II and III of the EU Birds Directive
A053	Mallard (<i>Anas platyrhynchos</i>)	Annex II and III of the EU Birds Directive

Species Code	Species Name	Legal Protection Status
A054	Pintail (<i>Anas acuta</i>)	Annex II and III of the EU Birds Directive
A065	Common Scoter (<i>Melanitta nigra</i>)	Annex II and III of the EU Birds Directive
A069	Red-breasted Merganser (<i>Mergus serrator</i>)	Annex II of the EU Birds Directive
A130	Oystercatcher (<i>Haematopus ostralegus</i>)	Annex II of the EU Birds Directive
A137	Ringed Plover (<i>Charadrius hiaticula</i>)	Article I of the EU Birds Directive
A140	Golden Plover (<i>Pluvialis apricaria</i>)	Annex I, II and III of the EU Birds Directive
A141	Grey Plover (<i>Pluvialis squatarola</i>)	Annex II of the EU Birds Directive
A142	Lapwing (<i>Vanellus vanellus</i>)	Annex II of the EU Birds Directive
A143	Knot (<i>Calidris canutus</i>)	Annex II of the EU Birds Directive
A149	Dunlin (<i>Calidris alpina</i>)	Article I of the EU Birds Directive
A156	Black-tailed Godwit (<i>Limosa limosa</i>)	Annex II of the EU Birds Directive
A157	Bar-tailed Godwit (<i>Limosa lapponica</i>)	Annex I and II of the EU Birds Directive
A160	Curlew (<i>Numenius arquata</i>)	Annex II of the EU Birds Directive
A162	Redshank (<i>Tringa totanus</i>)	Annex II of the EU Birds Directive
A179	Black-headed Gull (<i>Chroicocephalus ridibundus</i>)	Annex II of the EU Birds Directive
A182	Common Gull (<i>Larus canus</i>)	Annex II of the EU Birds Directive
A184	Herring Gull (<i>Larus argentatus</i>)	Annex II of the EU Birds Directive
A999	Wetlands & Waterbirds	-

Large numbers of waders, mainly oystercatchers (*Haematopus ostralegus*), bar-tailed godwits (*Limosa lapponica*) and redshanks (*Tringa totanus*), feed regularly in the sand and mudflats in Dundalk Bay, while many grazing birds, notably the light-bellied brent geese (*Branta bernicla hrota*) and wigeon (*Anas penelope*), feed on the saltmarshes and other vegetation present in the bay (Crowe, 2005). Furthermore, the bay represents the main roosting site for greylag geese (*Anser anser*) and whooper swans (*Cygnus cygnus*), with the saltmarshes used as high-tide roosts by all waterbird species (Crowe, 2005) listed in Table 2. A large number of seabirds (gull species) also regularly present within the site.

Because the bay is a site of high conservation value, inadvertent disturbance activities are generally small-scale and localised (Crowe, 2005). Recreational activities include leisure fishing, a range of at-sea activities (unpowered and powered water crafts), horse-riding, walking, clay-pigeon shooting, bait digging and mollusc gathering as well as wildfowling (Crowe, 2005; Marine Institute, 2021a). Although most activities are usually of low impact in the bay, the main pressures to the area include discharge into the estuary and pollution to surface waters (EC, 2020a). A sewage treatment works and discharge outlet in Dundalk

Harbour (Figure 2) has been operating since 1999; a 2020 environmental report has found the treatment plant to be non-compliant with the emission limit values set in the Wastewater Discharge Licence for Total Phosphorus and Total Nitrogen parameters (UISCE, 2020). Water pollution (eutrophication) in the estuary is thus an important pressure currently affecting the bay.

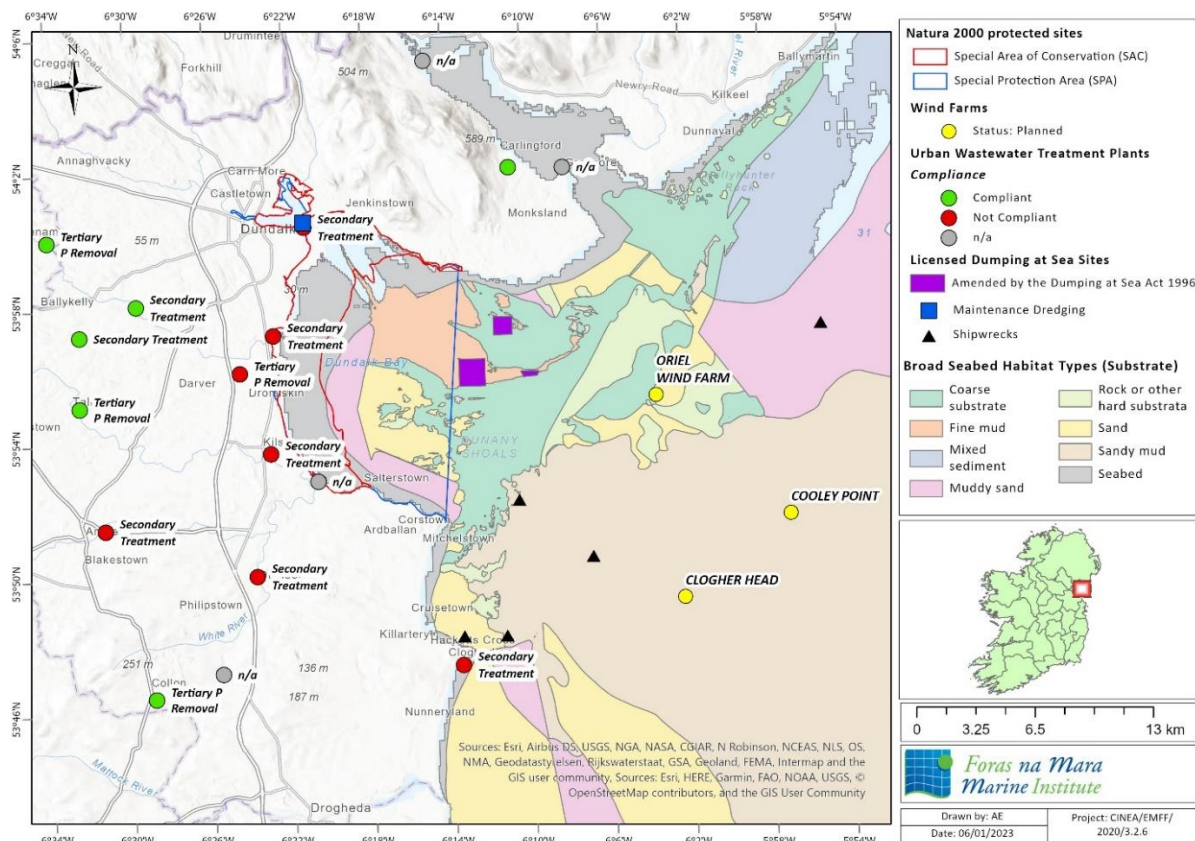


Figure 2. Active industrial and commercial activities within Dundalk Bay

Although a number of development proposals have been issued over the years, these mainly consist of small-scale industrial port developments. However, an offshore wind farm (Oriol Windfarm Ltd), east of the SPA, has made significant progress in 2019 in carrying out survey works of the proposed development site to investigate the suitability of the area for the construction of the wind farm. These works included a set of geotechnical site investigations, with engineering assessments and the completion of an Environmental Impact Assessment coupled with a Natura Impact Statement. The project is currently applying for a Foreshore Licence that will enable the completion of further marine survey works, which will enable the planning application to advance to development permission (DHLGH, 2022). The wind farm would be established significantly east of the SPA but with possible access of power lines through this site (Figure 2).

Vessel access to Dundalk Port had become limited over time because of ongoing sediment accretion, thus maintenance dredging has become necessary to restore adequate depth in the channel (Anthony D Bates Partnership LLP, 2021) with aggregate extraction also present along the edge of the north shore (Crowe, 2005). All above activities are currently active within the boundaries of Dundalk Bay SAC and SPA.

2.1. Fishing activities

Commercial hydraulic dredging was first introduced in Dundalk Bay in the late 1990s, which represented, and still represents, the main fishing activity of the area. The fishery was established to land **cockles (*Cerastoderma edule*)** occurring within the intertidal habitat of the bay in both the SAC and in the SPA, which overlaps with two marine community types listed in the specific conservation objectives for the site: fine sand and muddy fine sand community complexes. These communities are home to a wide diversity of bivalves and other invertebrates that are important sources of prey for many waterbirds and seabirds protected under the SPA.

The depletion of such prey populations could have a negative impact on bird populations or lead to displacement of them to other coastal sites. Moreover, dredging for cockles disturbs sediments to 5 cm depth, and is indiscriminate and unselective in the capture of non-target organisms. Dredging and sediment disturbance could lead to loss of fine sediment materials at the fishing site and change in the capacity of the physical habitat to support these same biological communities (Marine Institute, 2021a). Cockle stocks and other prey species for birds thus need to be maintained in favourable condition to ensure the ecological integrity of the site (Marine Institute, 2021a).

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Cerastoderma edule

In order to comply with the requirements of the EU Habitats and Birds Directives (and as transposed to Irish legislation in the European Birds and Habitats Regulations), a fishery plan for cockle was described and assessed in 2009 by the Dundalk Cockle Local Advisory Committee (DCLAC, 2009) with the aim of ensuring that the fishery is sustainable and protects the conservation of habitats and species within the site (DAFM, 2021). The management measures put into place mainly involved fishing gear and spatio-temporal restrictions with explicit catch quotas. Subsequently, five-year Fishery Natura Plans (FNPs) were developed, subject to appropriate assessment and implemented in 2011–2015, 2016–2020 and 2021–2025.

Commercial hydraulic dredging for **razor clams (*Ensis siliqua*)** also occurs in the subtidal habitats of Dundalk Bay, although unlike the cockle fishery that occurs in the SAC, the razor clam fishery occurs solely in the SPA and seaward of the cockle fishery. The fishery developed in the 1980s and has gone through varying cycles of increasing and decreasing production (Marine Institute, 2021e). Surficial bivalves, associated with razor clams, are important food sources for diving sea-ducks. This fishery is included as a possible in

combination effects pressure in appropriate assessment of the cockle fishery (Marine Institute, 2021a). Some crab and lobster trap fisheries also occur at the outer edge of the SPA, although at a very small scale. With regulations of various degrees currently enforced in all the above fisheries, all are monitored and controlled by the Sea-Fisheries Protection Agency (SFPA), responsible for the regulation of the sea-fisheries and the seafood production sectors, operating under the aegis of the Department of Agriculture, Food and the Marine (DAFM).

@MarineInstitute



Ensis siliqua, Dundalk Bay Ireland

2.2. Key elements of the Marine Protected Area

Under the International Union for Conservation of Nature (IUCN) Marine Protected Area (MPA) categories detailed in the IUCN Best Practice Protected Area guidelines series, Dundalk Bay SAC and Dundalk Bay SPA form the type IV sub-division category defined as a 'habitat/species management area' with a uniform multiple-use level of protection (Day et al., 2019). Its permanence of protection is year-round. Compliance monitoring is undertaken by the SFPA and consists of monitoring of records from vessel monitoring systems (VMS) (in collaboration with the Marine Institute), monitoring daily landings by vessels, ensuring compliance with food safety legislation, which includes tracking the geographic origin of catches, and monitoring of biotoxins in shellfish and the mandatory reporting of sales under the EU Buyers and Sellers Regulation (S.I. No. 260/2007 – Sea-Fisheries (First Marketing of Fish) Regulations, 2007) ⁽³⁾. Although a fishery management plan for cockle fishing is in place for Dundalk Bay and there are various management measures in place for the razor clam fishery in the wider area of the north-west Irish Sea, there is currently no management of recreational fishing; however, the level of such activity is thought to be very low according to the fishers operating in the bay.

With MPAs aiming at protecting ecosystems to achieve long-term conservation of habitats and species, there is widespread concern that many forms of bottom-dredging on benthic

⁽³⁾ Sea Fisheries (First Marketing of Fish) Regulations (2007) (S.I. No. 260 of 2017).

communities could cause irreversible damage to intertidal and subtidal sediments (Piersma et al., 2001). Depending on the scale of the fishery and the local hydrological conditions of the area fished, not only might target species be removed, but non-target fauna can be affected as well, which becomes of increasing concern for ecosystem functioning (Hiddink, 2003). Although hydraulic suction dredges operate by fluidising the sand using water jets and lifting the sediment with the cockles passing through a sieve (Hiddink, 2003), the mesh size, if chosen wisely, can allow juvenile cockles to be released in the water, enabling cockle populations to recover for the next harvesting year. The need for management plans to include gear specifications is thus important to reduce the likelihood of overfishing and over-exploiting the stock.

2.3. History of the cockle fishery

The cockle fishery in Dundalk Bay does not have a long history. When fishers started harvesting cockles (mostly hand gathering and raking) and landing cockles in different markets in the 1990s, the fishery was essentially open access: in other words, there was no limit on the number of vessels operating in the bay. In the early 2000s, small dredge fisheries emerged, which prompted landings to be recorded to report on the contribution made to the national economy (Fahy et al., 2005). In essence, during that time, and according to fishers, stocks were overfished.

In 2008, the DAFM closed the fishery because Ireland was also held accountable at the time of failing to comply with its obligations under Article 6 of the Birds and Habitats Directive (Council Directive 92/43/EEC; Directive 2009/147/EC). Those obligations entailed undertaking appropriate assessments of the implications of a project/plan in view of the conservation objectives of a designated site (Article 6 of the Habitats Directive 92/43/EEC), as well as building conservation measures to ensure the survival and maintenance of habitats and species. Undertaking of research to inform protection and management is also mandated by the Directive (Article 4 and Article 10 of the Birds Directive 2009/147/EC). From 2008, the necessary procedures of Article 6 were completed, additional fishery management measures were introduced and a research and monitoring programme was established. This programme of measures evolved between 2008 and 2023.

2.3.1. Cockle fishery management plan development

From 2008, when the appropriate assessments were carried out, the DAFM, the Marine Institute, Bord Iascaigh Mhara (BIM) and the industry developed a cockle fishery management plan to regulate the fishery. This referenced the baseline ecological data on benthic communities and wading birds for the site (collected by NPWS and the Marine Institute) and also the Before-After, Control-Impact (BACI) and Control-Impact (CI) studies on the effects of the fishery on habitats and species to ensure that the fishery did not compromise the conservation objectives for ecological features protected in the site. Annual cockle surveys and surveys of associated bivalves, marine benthic community analysis and overwintering bird surveys and dietary studies were developed. Key metrics were derived to evaluate effects of fishing on the site including the relationship between cockle biomass and the number of birds that the site supported, distribution maps for the dominant bivalve species, and compliance and performance monitoring of the fishery.

From 2011, five-year fishery plans (Fishery Natura Plans) were developed and each were subject to new appropriate assessments. The third FNP (2021–2025) is in its third year.

3. AIMS AND OBJECTIVES

The main aim of this report is to describe the protected features in Dundalk Bay MPA, the fisheries that operate in the Bay and the significance of those fisheries for the long-term management and conservation of the MPA. Specifically, we assess how the profile and management of fisheries have been adapted in response to the designation of the site as an MPA and describes the process by which this was achieved.

4. METHODOLOGY

The Marine Institute was centrally involved in advising on the fishery management procedures and measures as well as assessing impacts of fishing on the site with the fishery management authority from 2008 to present. All documentation for Dundalk Bay is therefore held by the Marine Institute or by the fishery management authority (DAFM).

4.1. Annual cockle biomass surveys in the SAC

Annual total allowable catch (TAC) for cockle fishing vessels in Dundalk Bay is estimated from annual mid-summer cockle stock surveys undertaken by the Marine Institute (sampling from 2007 to 2022). The resulting TAC recommendations and other fishery measures are given legal effect through Fishery Natura Declarations (FNDs) issued by the DAFM annually (DAFM, 2022).

Estimates of the cockle population size are made via a stratified randomised quadrat survey (Figure 3) along the intertidal mud and sand flats of the bay, all other bivalve species collected (mainly *M. tenuis*, *M. balthica* and *Donax* sp.) are identified to species level (where applicable) and counted. A geostatistical model is used to interpolate abundance between survey stations and to provide an overall estimate of abundance of different size classes of cockle.

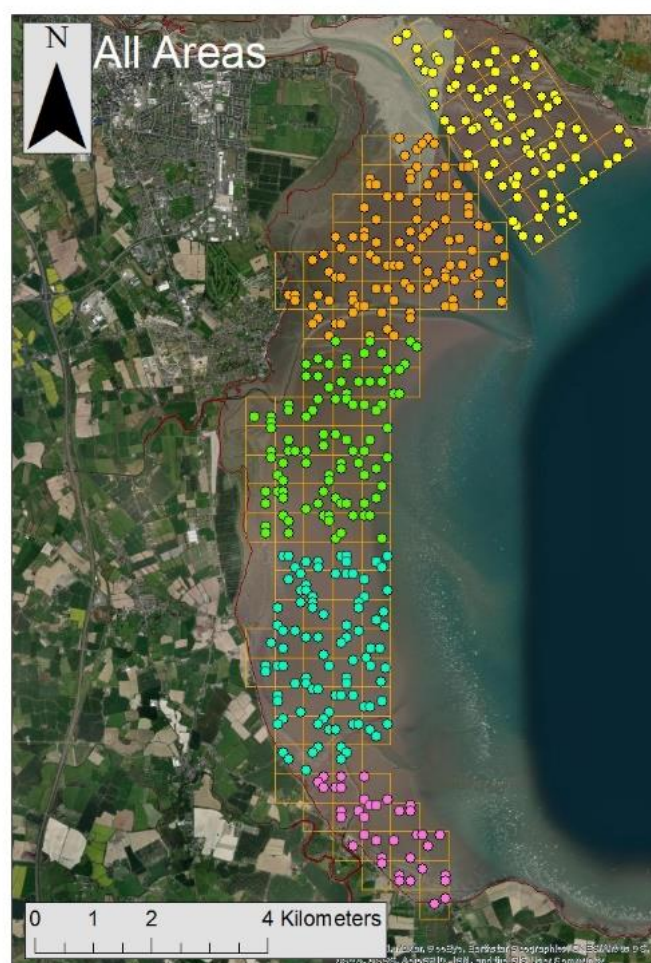


Figure 3. Cockle survey stations in Dundalk Bay on a 400m grid. Colours represent different survey sections.

4.2. Annual biomass surveys for razor clams in the SPA

The North Irish Sea (NIS) razor clam (*Ensis siliqua*) fishing area has been surveyed annually by the Marine Institute since 2017, using commercial vessels and standard hydraulic dredges. The survey includes approximately 160 stations within Dundalk Bay SPA (Figure 4). Although the abundance and size distribution estimates of razor clams are the primary focus of the survey, the total number of all other bycaught bivalves are also recorded. Surveys are stratified using VMS effort data, which is presumed to reflect different abundance of razor clams as vessels target such areas. As fishing methods utilise hydraulically pressurised water to fluidise sediments in front of the dredge, fishing depth is usually limited, thus the fishery occurs generally at depths between 4 and 14 m (Marine Institute, 2021c). A geostatistical model is used to interpolate abundance between survey stations and to provide an overall estimate of abundance of different size classes of razor clam.

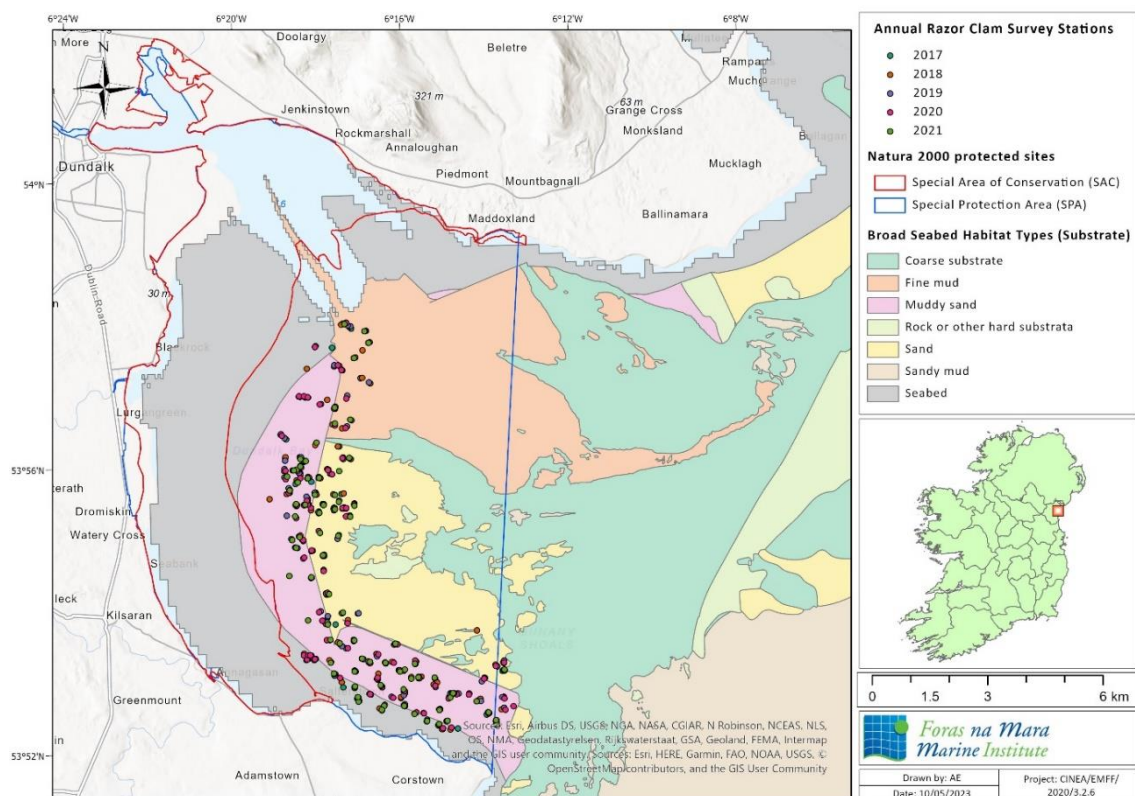


Figure 4. Location of North Irish Sea razor clam (*Ensis siliqua*) survey stations between 2017 and 2021 in Dundalk Bay.

4.3. Verifying fishing activity

From 2016, vessels in the cockle fishery that also fished for razor clams in the SPA reported their spatial location using Inshore Vessel Monitoring Systems (iVMS). In 2023, all vessels fishing for cockles reported iVMS data. Such information is mandatory for the razor clam fishery under the Razor Clam (Conservation of Stocks) Regulations (S.I. No. 206 of 2015) ⁽⁴⁾. The iVMS data is used as a fishery control measure. Data is viewable on mobile devices in real time by fishery control officers.

4.4. Before-After, Control Impact studies and Control-Impact studies

To inform the Appropriate Assessment of the impact of hydraulic dredging for cockles on the Conservation Objectives of Dundalk Bay, a series of studies and surveys were initiated between 2008 and 2010.

The first Control-Impact (CI) survey in 2008 aimed to assess the impacts of hydraulic dredging on sediments and benthic fauna in Dundalk Bay. A dredge and rake fishery for cockles had occurred in a restricted area of the bay in the autumn of 2007, so sampling was undertaken in August 2008 from areas that were open to fishing and areas that were

⁽⁴⁾ Razor Clam (Conservation of Stocks) Regulations 2015. S.I. No. 206 of 2015. [irishstatutebook.ie](https://www.irishstatutebook.ie/eli/2015/si/206/enacted/a1) (accessed 11 January 2024).

closed to fishing in 2007 to assess for recovery after the previous fishery (Clarke and Tully, 2011).

A BACI study was also undertaken between 2009 and 2010 to study the effects of cockle suction and non-suction dredging on the sediments and benthos of Dundalk Bay. Surveys were done immediately before, immediately after and four months after the closure in 2009 of a fishery that extracted 108 tonnes of cockles from a standing stock of 2 158 tonnes, to monitor the impacts of dredging activity and determine recovery rates if effects did indeed occur (Clarke and Tully, 2011). A second impact study was initiated in 2012, this time looking at mortality of bivalves in visible dredge tracks compared to areas outside of dredge tracks, the day following fishing occurred (Marine Institute, 2021b).

4.5. Waterbird Distribution Study and the Irish Wetland Bird Survey

To inform the Appropriate Assessment of the impact of hydraulic dredging for cockles on the Conservation Objectives of Dundalk Bay, a waterbird count and a foraging behaviour study was initiated in 2009 and 2010.

The foraging behaviour examined the relationship between cockle fishing and bird distribution by carrying out a series of waterbird counts on six dates in February and March 2010 in areas that had been fished in the autumn of 2009 and in comparable unfished areas (Marine Institute, 2011). Although the waterbird count and foraging behaviour study acted as a baseline to assess bird abundance and density in the bay, I-WeBS, coordinated by BirdWatch Ireland, is the main programme used to inform on the conservation status of birds in Ireland, and especially to provide annual count data, which is the key metric used in the FNPs. Since 1994, the programme has relied on hundreds of volunteers and NPWS rangers each year to monitor wintering waterbird populations at their wetlands across Ireland (BirdWatch Ireland, 2023). It is the primary tool for assessing the relationship between post-fishery cockle biomass and oystercatcher numbers. This data is presented in the FNP reviews.

5. RESULTS

5.1. Fishery Natura Plan (2021–2025)

The FNP for cockles (*Cerastoderma edule*) in Dundalk Bay SAC and SPA has been implemented since 2011 with a restriction on the number of licences issued per management plan. The plan then sets out measures to be undertaken by the vessel owners holding permits to fish cockles in the bay. The plans include daily catch limits, minimum catch rate closure conditions, overall season TAC, minimum landing sizes and closed seasons (Table 3).

Table 3. Fishery restrictions in respect with the Fishery Natura Plan 2021-2025 for cockle (*Cerastoderma edule*) in Dundalk Bay, Co. Louth, Ireland.

Restriction	Description
Cockle fishing permit only	Fisheries Natura Permit issued under Regulation 10 of the European Union (S.I. No. 290 of 2013).
Limited number of vessels allowed in the fishery	Limit of 33 permits as in the fishing plan for 2016–2020. The plan of 2021–2025 has reduced it to 28 permits.
Daily catch allowances	Up to 1 tonne per vessel per day to ensure equitable distribution of catch.
Spatial and temporal conditions are applied	Vessels cannot fish on more than one tide in any 24 hours, and on any tide less than 4.2 m
Gear specifications and restrictions	Cockle suction dredge with a dredge width greater than 0.75 m or a cockle non-suction dredge with a dredge width greater than 1 m is not permitted.
Minimum legal landing size	Cockles with a maximum width less than 17 mm are not permitted on board and should be returned to the sea. However, the operational size due to market considerations is 22 mm.
Harvest rates are set in relation to annual biomass estimates	In 2023, the survey biomass estimate of 2 603 tonnes provides for a TAC of 867 tonnes.
Minimum biomass	The fishery will not open when biomass is less than 1 000 tonnes as estimated from the mid-year annual survey.
Fishery closure due to in-season depletion of catch	If rates decline to 250 kg per vessel, the fishery will close, irrespective of other harvest rules. A previous option to increase exploitation rates from 33 % to 50 % when biomass exceeds 3 000 tonnes was removed in order to sustain harvest over the years when recruitment is low.
Hand-gathering of cockles	No commercial hand-gathering of cockles is allowed within the bay.
Seasonal closures to minimise disturbance to birds and habitats	Fishery closes on 1 November, provided this is preceded by a period of 14 weeks during which the fishery is open to allow sufficient fishing opportunity to take available quota. If this is not the case, the closing date will be later as required.

5.2. Annual cockle biomass surveys in the SAC

Annual surveys of cockle biomass and distribution were undertaken in the period 2007–2023 (Table 4, Figure 5). The fishery TAC is estimated from the survey biomass using pre-agreed harvest control rules set out in the FNP. The harvest control rules have been modified in each successive FNP.

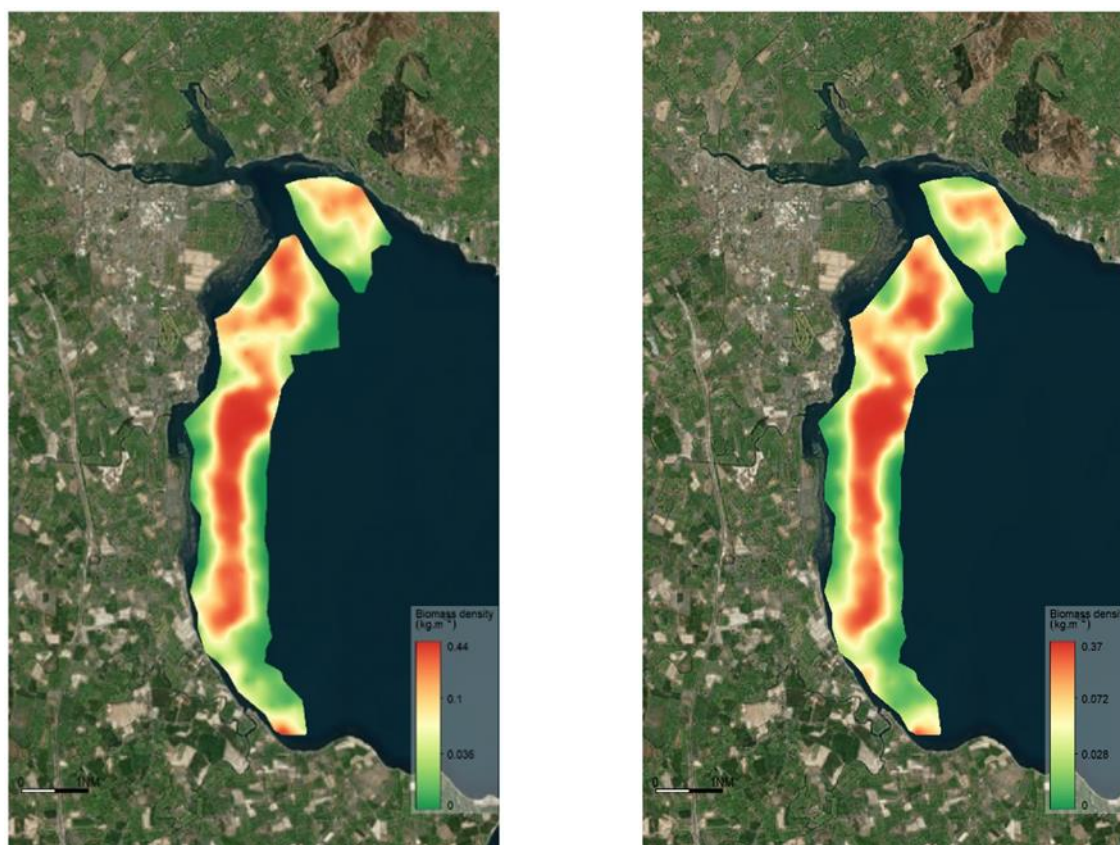


Figure 5. Distribution and density (kgs.m-2) of all cockles (left) and commercial cockles (> 22 mm shell width) (right) in Dundalk Bay in May 2023.

Table 4. Cockle (*Cerastoderma edule*) biomass trends with Total Allowable Catch and landings for the respective years in Dundalk Bay between 2007 and 2023.

Year	Survey month	Biomass (tonnes)		TAC (tonnes)	Landings (tonnes)	
		Mean	95 % CL		Vessels	Hand gatherers
2007	March	2 277	172	950	668	0
2008	August	3 588	1 905	0	0	0.28
2009	June	2 158	720	719	108	0
2010	May	814	314	0	0	0.25
2011	May	1 531	94	510	325	9.4
2012	May	1 234	87	400	394	0
2013	June	1 260	99	416	343	0
2014	June	972	188	324	0	0
2015	June	1 034	100	345	0	0

Year	Survey month	Biomass (tonnes)		TAC	Landings (tonnes)	
		Mean	95 % CL	(tonnes)	Vessels	Hand gatherers
2016	July	1 878	87	626	626	0
2017	June	2 316	95	772	772	0
2018	June	1 785	175	542	542	0
2019	July	3 790	110	600	594	0
2020	May–June	3 420	870	1 128	1 128	0
2021	May–June	1 927	537	642	638	0
2022	May	1 826	360	608	0	0
2023	May	2 603	578	867	-	-

From 2007 to 2023 cockle stocks have varied from a high of 3,588 tonnes in 2008 to a low of 814 tonnes in 2010. In the most recent five years, biomass has ranged from 1,826–3,790 tonnes (Figure 6). Cockle landings have not exceeded the scientific TAC. In some years, although a TAC was allowed for, no fishery occurred. This was through voluntary agreement of fishers who hedged forgone income in one year in lieu of a better income in the following year as they forecasted evolution of the biomass.

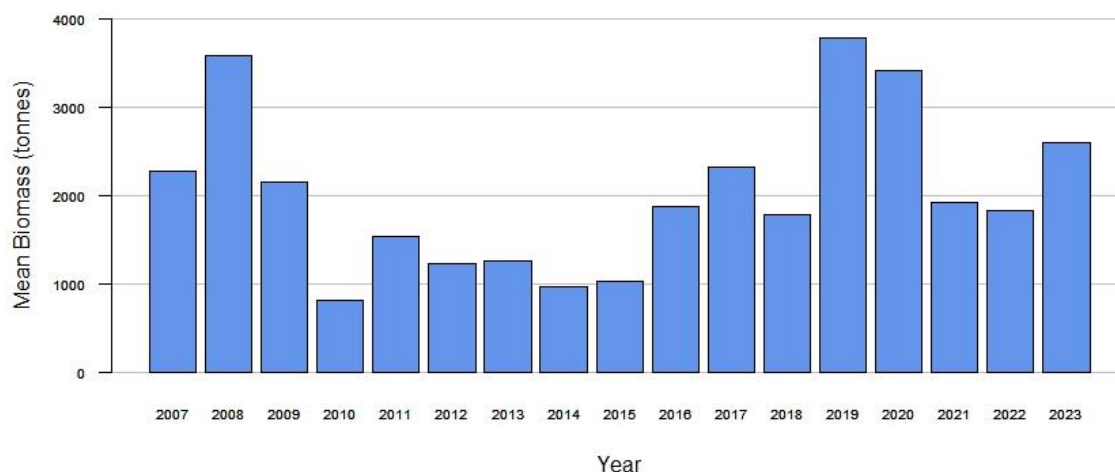


Figure 6. Cockle (*Cerastoderma edule*) biomass trends in Dundalk Bay between 2007 and 2023.

Populations of *M. tenuis* and *M. balthica* also occur in high densities in Dundalk Bay. *M. balthica* occurs in the upper shore and is not exposed to the cockle fishery, which occurs in the mid and lower shore. Mortality of *M. tenuis* occurs as a result of the abrasion pressure caused by the fishery but the species has high resilience to fishing pressure (Marine Institute, 2021d).

Annual surveys of both species have shown that densities were stable between 2013 and 2020, (Figure 7).

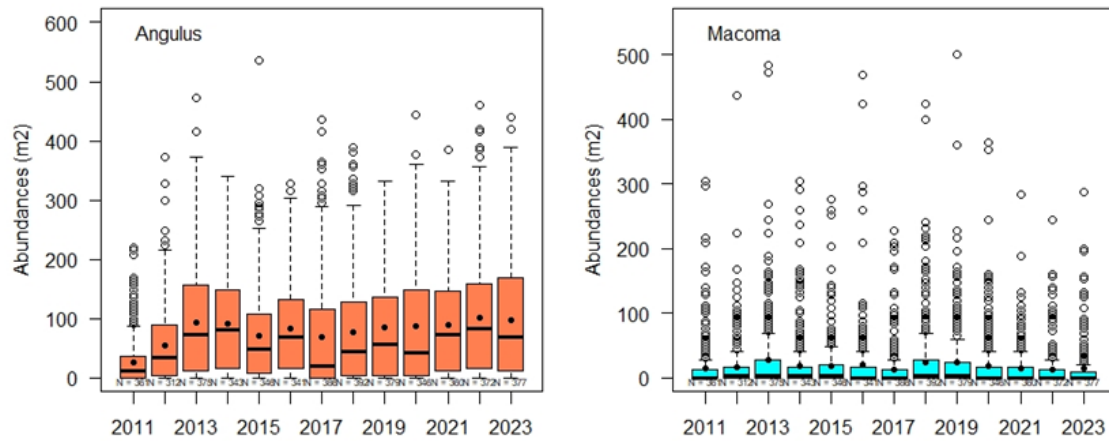


Figure 7. Average densities of *Macomangulus tenuis* and *Macoma balthica* in intertidal sediments in Dundalk Bay in 2011–2020.

5.3. Annual biomass surveys for razor clams in the SPA

Surveys undertaken for razor clams (*Ensis siliqua*) annually since 2017 have shown that the population within Dundalk Bay fluctuated in total biomass, with a 41.5 % decrease from 1,505 tonnes in 2017 to 880 tonnes in 2018, (Table 5). However, biomass increased to 1,972 tonnes in 2020. Razor clam distribution in 2023 was overall denser in the southern part of the bay (Figure 8).



Figure 8. Distribution of *Ensis siliqua* in the north Irish Sea, in June 2023. Dundalk Bay is in the northernmost (top) section.

5.3.1. Landings

Landings for the NIS razor clam fishery are reported in shellfish stocks and fisheries reviews published by BIM and the Marine Institute.

Landings of razor clam across Dundalk Bay increased between 2013 and 2015 (from 140 tonnes to 300 tonnes), coinciding with an increase in fishing vessels from 21 in 2012 to 49 in 2015 in the North Irish Sea (Marine Institute and BIM, 2019). The Dundalk Bay and Gormanstown (South of Dundalk) production areas accounted for 619 of 716 tonnes of landings in 2018 and 91 % of the landings (488 of 533 tonnes) in 2019 (Marine Institute and BIM, 2019).

5.3.2. Exploitation rates

Harvest rates for Dundalk Bay ranged from 15 % in 2017 to 31 % the following year. These rates are higher than those advised for the NIS fishery (advisory 15 % harvest rate based on voluntary TAC agreements; Marine Institute and BIM, 2018). Furthermore, with the results of the annual razor clam surveys, the abundance of deep-burrowing bivalve infauna caught by bycatch were reported to have declined by more than 5 % between 2017 and 2020 (Marine Institute, 2021c). It is suspected that the fishery causes mortality of deep-burrowing species, although the removal of razor clams by fishing may allow other species to recruit. However, in other areas of the North Irish Sea where the razor clam fishery occurs, bycaught species remain at high levels. The effects are therefore uncertain. In response to this a new closed area was put in place in 2022 so that changes in abundance of these species in the absence of fishing could be monitored.

Table 5. Biomass of razor clams (*Ensis siliqua*) reported from the North Irish Sea razor clam surveys carried out in Dundalk Bay between 2017 and 2022.

Year	Variable	Biomass (tonnes)	Simulated 95 % CLs	
			Lower	Upper
2017	Biomass all size classes	1 504.9	1 484.6	1 778.0
	Biomass > 130 mm	1 357.3	1 351.5	1 622.3
	Biomass > 150 mm	1 036.9	1 053.7	1 308.7
2018	Biomass all size classes	880.3	911.5	1 163.2
	Biomass > 130 mm	838.8	948.9	1 222.7
	Biomass > 150 mm	679.2	714.3	926.3
2019	Biomass all size classes	1 213.7	1 209.6	1 530.6
	Biomass > 130 mm	1 174.0	1 174.4	1 488.1
	Biomass > 150 mm	1 008.2	1 030.6	1 314.3
2020	Biomass all size classes	1 972.1	1 963.3	2 356.9
	Biomass > 130 mm	1 773.0	1 775.4	2 175.3
	Biomass > 150 mm	1 642.1	1 659.6	2 055.7
2021	Biomass all size classes	1 800.0	1 682.9	2 041.8
	Biomass > 130 mm	1 748.4	1 625.8	1 991.0
	Biomass > 150 mm	1 514.3	1 413.9	1 732.4
2022	Biomass all size classes	1 913.8	1 892.3	2 332.7
	Biomass > 130 mm	1 894.1	1 869.5	2 308.0
	Biomass > 150 mm	1 744.3	1 694.3	2 162.9

All bivalves (other than razor clam) showed low abundances in razor clam surveys (both per m² and per station surveyed) between 2017 and 2021 in the bay compared to other areas (in the NIS). There has been a constant decrease since 2017 apart from a slight recovery in 2020 and 2021 (Marine Institute, 2021e). Species richness (total number of species recorded) has also declined since 2017, although it increased outside the bay during the same timeframe. The bivalve *Arctica islandica* has not been recorded by the surveys since 2018, despite the species showing slow growth and long life span (Marine Institute, 2021e).

5.3.3. Verifying fishing activity

iVMS effort peaked in 2018 and 2019 in Dundalk Bay (Figure 9), with various hotspots of activity. The VMS effort data has two distinct 'clusters' of activity: one within the SAC representing the cockle fishery and one within the SPA representing the razor clam fishery. There is no spatial overlap between these two fisheries. In 2021 there was no cockle fishery (Figure 9) as biomass was low in that year and fishers voluntarily agreed not to fish the TAC.

Mapping of marine protected areas and their associated fishing activities: MAPAFISH Annex 5 - Case study reports

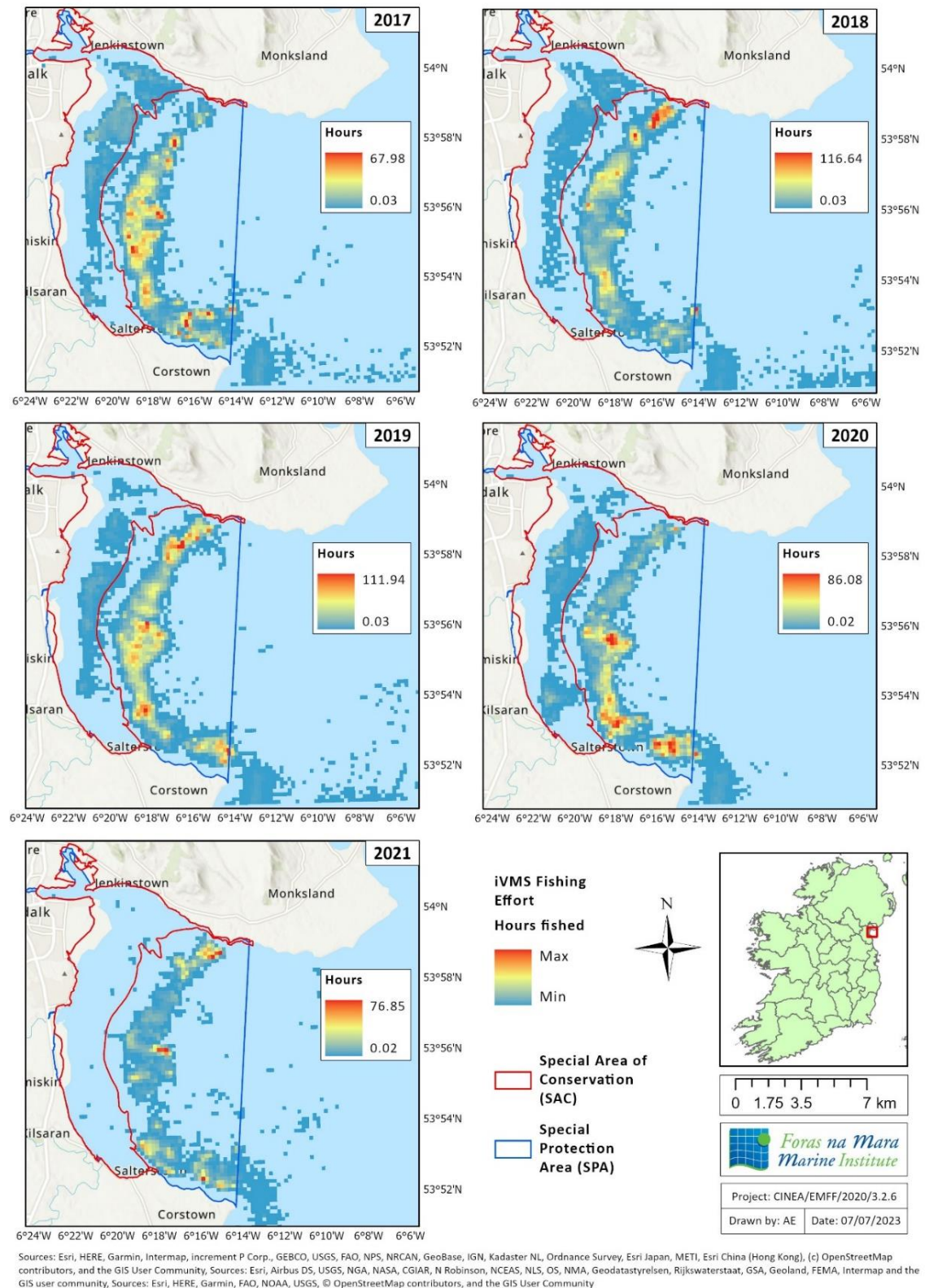


Figure 9. Inshore Vessel Monitoring System (iVMS) effort in hours fished between 2017 and 2021 in Dundalk Bay.

5.3.4. Before-After, Control-Impact studies and Control Impact studies

No significant effect was found in the 2008 survey. However, both the BACI study of 2009–2010 and the CI study of 2012 found that fishing causes mortality of cockles (*Cerastoderma edule*). In fact, the BACI study failed to detect significant effects on benthic sediments and benthic faunal communities with the exception of the dominant species of bivalves in the system: *Cerastoderma edule*, *Macomangulus tenuis* and *Macoma balthica*. Significant spatial and temporal variability in abundance of taxonomic groups and species was observed; however, this variability was unrelated to fishing effects (Clarke and Tully, 2011). The mortality rate of cockles in the system can be explained by their highly brittle thin shell and their occurrence in the top few centimetres of sediments. This makes them highly vulnerable to capture by cockle fishing gear in surface sediments (Marine Institute, 2021b).

5.3.5. Waterbird distribution study and I-WeBS

The results of the waterbird distribution study showed no consistent pattern of distribution for most species between control and impact sections. The mean proportion of oystercatchers (*Haematopus ostralegus*), dunlin (*Calidris alpina*), bar-tailed godwit (*Limosa limosa*) and common gull (*Larus canus*) in the impact zone was close to 50 %, meaning that if cockle fishing in the autumn of 2009 caused habitat differences between control and impact zones, no detectable effects on the distribution of these species could be seen in February and March 2010 in Dundalk Bay (Marine Institute, 2011). The knot (*Calidris canutus*) was the only species to have significantly higher numbers occurring in control zones of one sector of the bay, however it was not possible at the time to specify whether this was due to cockle fishing or other causes such as habitat differences or effects of prey depletion earlier in the winter (Marine Institute, 2011).

The waterbird foraging behaviour study also did not show any significant changes, with foraging behaviours in most species being generally very similar between control and impact zones. The only difference that may have been significant was found in oystercatchers (*Haematopus ostralegus*) in lower shore habitat outside the transects of the survey, compared to birds within the transects (Marine Institute, 2011). This indicated the importance of the survey design for these studies and the need for control and impact areas within the same lateral zones in the shoreline.

I-WeBS bird counts recorded over the winter season showed a long-term average (1994 to 2019) count to be stable in a range of between 30 000 and 61,000 birds (Figure 10). The highest number of all bird species in Dundalk Bay was 61,255 in the winter of 2003/04. The trend of year-on-year decline between 2011 and 2015 was reversed between 2015 and 2019.

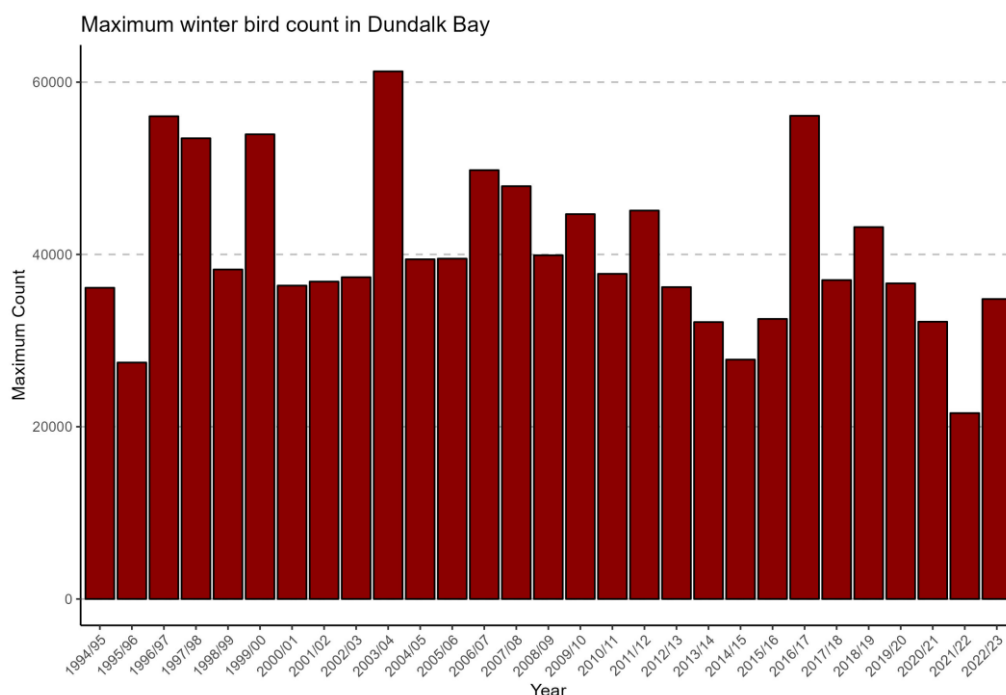


Figure 10. Bird trend (all species) from September to February during the winter seasons from 1994/95 to 2022/23 in Dundalk Bay.

As oystercatchers predominantly feed on cockles and on other prey present in the grasslands bordering Dundalk Bay, monitoring their count and distribution is important to assess the extent of the impact of fishing. I-WeBS counts reported highest numbers of oystercatchers in 2006/07 (Figure 11). Numbers decreased from 2007 to 2014 and fluctuated without trend from 2015–2022.

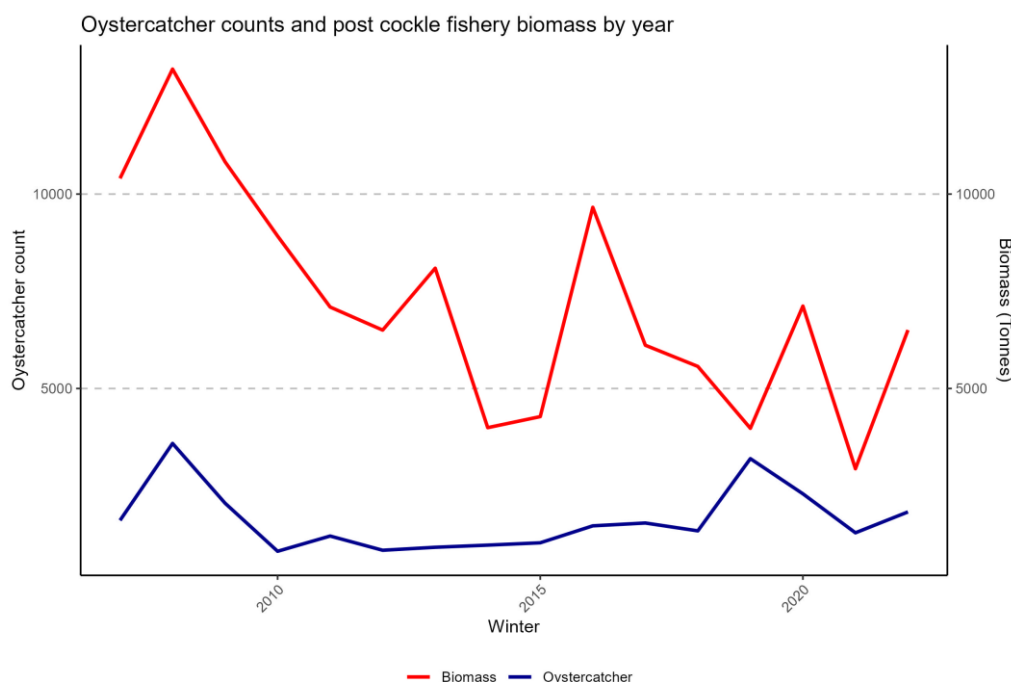


Figure 11. Oystercatcher (*Haematopus ostralegus*) trend (peak count number) from September to February between 2004/05 and 2022/23 and post fishery cockle biomass in Dundalk Bay.

6. DISCUSSION

6.1. Implementation of FNP

With the implementation of the FNP for cockles in Dundalk Bay since 2011, and the required appropriate assessments being produced prior to adopting these plans, Dundalk Bay has more than a decade of fishery and environmental data available to monitor any possible changes in structure, function and distribution of the marine communities and waterbirds of the bay. Harvest control rules for the fishery are revised regularly as is the supporting policy on the access of new vessels to the fishery (Marine Institute, 2021d). In the current FNP (2021–2025), there is no fishery when cockle biomass is below 1 000 tonnes whilst harvest control rules for biomass between 1,000 and 1,500 tonnes never reduce biomass to less than 1,000 tonnes. This limit reference point is to protect wading birds that feed on cockles. At higher biomass, the harvest rate is 33 %, (Cockle Fishery, 2021). Although the legal landing size of cockles is currently 17 mm shell width, the current effective minimum landing size is of 22 mm shell width set to optimise the price in the market place and separate the cockles in Dundalk from those landed in UK fisheries. In-season depletion of catch is also closely monitored within the plans, leading to closures of the fishery if catches decline to 250 kg per vessel per day (Cockle fishery, 2021). In-season fishing depletion was detected during the first half of the second FNP, however depletion rates were always lower than the 33 % harvest control rule, which was verified through the annual stock biomass estimate surveys (Marine Institute, 2021d). The survey estimates were even found to underestimate the fishable biomass at the start of the fishing season as cockles would show significant increases in size between the surveys and the opening of the fishery (Marine Institute, 2021d).

6.2. Monitoring of dredging activity impacts

Hydraulic suction and non-suction dredging for cockles and razor clams is a common commercial fishing activity in intertidal and sub-tidal marine habitats. These activities have potential impacts on macrobenthic communities and non-target species (Wijnhoven et al., 2011). Where such fishing occurs in Natura 2000 sites, assessments of their significance for features that are protected by the Habitats (92/43/EEC) and Birds Directives (2009/147/EC) have to be assessed. In such areas the ecological integrity (distribution structure, range and function) of habitats has to be maintained at favourable conservation status (Clarke and Tully, 2011 and 2014). The significance and likelihood of ecological effects depend on the scale, intensity and frequency of dredging activity (Hiddink *et al.*, 2006). Furthermore, marine benthic communities and protected species have variable sensitivities to dredge fishing. Sensitivity in this case has two components: resistance to the pressure or the degree to which a unit of pressure changes the feature; and resilience or recoverability of the feature. In Dundalk Bay, extensive monitoring of benthic communities, specific species of those communities and wading-bird populations have been monitored annually since 1990 (wading birds) and 2007 (habitats). Fisheries management measures and plans have been adopted and also adapted in response to any signals of effects in monitoring data. In 2022 an area was closed to mobile bottom fishing gears to evaluate whether changes in benthic communities would result from removal of fishing and to find a true baseline reference point for these benthic communities in the site. Monitoring is ongoing.

6.3. BACI, CI Studies and implications

The BACI studies undertaken by Clarke and Tully (2011, 2014) failed to detect any significant effect of cockle fishing on benthic sediments or on the overall community structure of the site but found short-term effects on cockle, as would be expected because it is the target species for the fishery, and on a thin shelled bivalve (*M. tenuis*) which is not a target species for the fishery. However, these species have low sensitivity to fishing pressure – their resilience is high due because of their species life history. Possible effects of fishing are difficult to detect in these cases and long-term robust monitoring programmes provide evidence of the degree to which the communities are stable in time and space.

6.3.1. Annual cockle biomass surveys in the SAC

Survey data for 15 years (cockle) and seven years (razor clam) provide estimates of the biomass of cockles and razor clams which are used to advice the TAC. The cockle surveys also provide evidence that mortality effects on *M. tenuis* are short-lived (as suggested in the 2011 BACI study; Clarke and Tully, 2011) and insignificant in relation to inter-annual variation in abundance. *M. tenuis* is seen to be highly variable across years, and short-term fishery-induced changes in abundance are not carried over (cumulative) between seasons (Marine Institute, 2021b). Disturbance on *M. tenuis* and *C. edule* with respect to the long-term stability of marine intertidal communities in Dundalk Bay SAC is thus found to not be significant (Marine Institute, 2021b).

Increases in biomass of cockles depend on spat settlement, spat survival and growth rates. The surveys also provide data on the size and age of cockles and enable short-term forecasting of biomass. Spat settlement and overwinter survival are also affected by severe weather conditions, such as seen in 2010–2011.

6.3.2. Annual biomass surveys for razor clams in the SPA

Biomass of razor clams in the NIS increased from 2017–2018 and have been stable at around 9 000 tonnes between 2019 and 2023. A TAC was introduced in 2022. However, no specific TAC was developed for Dundalk Bay. Although indicators such as catch per unit effort declined from 2015, they have been stable at about 200 kgs.hr⁻¹ in recent years. Dundalk Bay Razor Clam Assessments in 2019 (Marine Institute, 2019) revealed the possible need to limit the area of Dundalk Bay that is fished in conjunction with an applied TAC based on percentage biomass from annual surveys. An alternative approach was adopted in 2022 with the introduction of a closed area in which there is no fishing.

Within the razor clam fishery, as with cockle fishing, bycatch is also of concern, as the deep burrowing bivalve infauna of the SPA saw a 50 % decline in abundance between 2017 and 2020. In the Irish Sea, a large number of those deep burrowing bivalves occupy the same habitat as razor clams, and the fishing of the latter causes some mortality depending on the species (Marine Institute, 2021c). Furthermore, community analysis of the data in Dundalk revealed that the bay has a different community characterised by lower diversity and abundance of deep burrowing infaunal bivalves compared to areas south of the Bay (Marine Institute, 2021c) which suggests a need for more fishery management controls. Impacts on infauna and sediments are thus expected from such fishing, and disturbance and long-term effects will depend on the footprint of the fishery and capacity of fauna to

recover (Marine Institute and BIM, 2019). Breen et al. (2022) found no evidence of effects of the razor clam fishery on diving sea-ducks that feed on surficial benthic bivalves.

As the razor clam fishery operates within the SPA of Dundalk Bay, it is also mandatory to assess it to ensure compliance the NIS fishery regulations, and to determine its direct and potential in-combination effects on the cockle fishery in the SAC. Dredging for cockles disturbs sediments down to 5 cm depth, while dredging for razor clams disturbs sediments down to 25 cm. There is no spatial overlap between the two fisheries and the marine communities and bird species that may be affected are different. In combination effects are therefore unlikely. Both fisheries, as seen previously, are unselective in the capture of non-target organisms and may result in the depletion of sea-floor fauna within habitats protected under Natura 2000. Such changes to structure and function of habitats may have a knock-on effect on some species of bird that utilise these habitats as sources of food (Marine Institute, 2021b). A wide-range of bivalve feeders are of Special Conservation Interest within the conservation objectives of SPAs, including oystercatchers, knots, shell-ducks and common scoters.

6.3.3. Waterbird trends

The I-WeBS scheme provides a key data set for evaluation of effects of fisheries on wading birds in the Bay. Two other contracts were commissioned to complement this data, one in 2009-2010 by NPWS, and one between 2012 and 2015, contracted by the Marine Institute (Marine Institute, 2021d). Although these studies were successful overall at providing a good insight on bird populations of Dundalk Bay, several factors affected the quality of the data produced, and considerable practical difficulties made the design and execution of the studies particularly challenging (for example, difficulty in identifying control and impact stations as the fishing parameter was quite broad in the Bay; some areas of the bay were difficult for bird counters to access). The limitations of this study show the importance of gathering data on a systematic basis, especially when it comes to providing a baseline on species distribution that may vary over time, whether it is due to human activity or environmental pressures.

Low-tide data in Dundalk Bay suggests that the high-tide counts have underestimated the number of birds using the Bay by about 30 % (Marine Institute, 2021d). However, I-WeBS is the longest-running programme in Ireland to have provided continuous trend assessments on waterbirds at more than 250 sites across the country (BirdWatch Ireland, 2023), and this data will enable potential impacts such as fishing to be evaluated as a possible cause of decline in bird populations. The Marine Institute also commissioned a digital aerial wading-bird survey in the Bay in 2019 to evaluate whether this method could be used instead of ground-based counting. The analysis of these data is as yet incomplete.

6.4. Status of fishing activity in Dundalk Bay in relation to the MPA

Recent risk and appropriate assessments for razor clams and cockles have shown these fisheries could have physical disturbing effects on some bivalve feeders in the Bay (Marine Institute, 2021b). It is thus important that fishery management plans for both cockle and razor clam fisheries ensure the sustainability of those activities without compromising the conservation status of species and habitats designated under the SAC and the SPA.

The cockle and razor clam fisheries have not been displaced from Dundalk Bay following designation of the site as a SAC and SPA. The cockle fishery in the Bay has operated as a limited-entry fishery since 2008 and all permit holders have fished the Bay

each year in which there is a TAC available as determined by annual surveys and harvest control rules. So called fishery natura permits are issued to the 28 vessel owners annually by the licensing authority (DAFM). Landings by these vessels fishing for cockles in the Bay, collated by the Sea Fishery Protection Authority (SFPA) and iVMS data sourced and collated by the Marine Institute provide the evidence of ongoing fishing activity and uptake of the TAC. The TAC is the limiting factor in controlling the total fishing effort. These data are reported annually by the Marine Institute ⁽⁵⁾. Cockle fishing, as with other bivalve fisheries, can only occur in areas that are microbiologically classified for the production of cockles. Dundalk Bay is the only classified production area for cockles on the east coast ⁽⁶⁾; therefore, if cockle fishing occurs on the east coast, it has to occur in Dundalk Bay. Also, the fishery is tied to intertidal mud flat habitats where cockles are distributed.

The razor clam fishery occurs in shallow sub-tidal waters of the North Irish Sea including Dundalk Bay ⁽⁷⁾. The fishery has not been displaced legally or operationally from Dundalk Bay. This is evidenced by high-resolution iVMS data, which has been available from 2016. The iVMS data shows expansion of the fishery from the south of the Bay in 2016 to central and northern areas in more recent years. Landings of razor clams from the Dundalk Bay classified production area for razor clams, collated by the SFPA, is evidence of ongoing activity. In 2022, a small area of the Bay was closed to fishing but this accounts for a minor proportion of the distribution of razor clams in the Bay (DAFM, 2023). This is the only spatial restriction in place. No other technical or operational restrictions have been established. Therefore, is no reason to expect any displacement. The annual research surveys of razor clams provide distribution, abundance and grade structure of razor clams in the Bay and south along the east coast of Ireland. The activity of the fleet follows the distribution and is evidenced in the iVMS data.

The designation of the site as a SAC and SPA has driven improved management of fisheries at the site and in collaboration with fishers. Initial discussions with fishers regarding the possible implications of the designation for their livelihoods were difficult, mainly because of uncertainty about how fisheries could co-exist with the designated site and with the conservation requirements. There was significant scientific uncertainty about the possible impacts of continued fishing but also uncertainty in the process and procedures that would be involved in implementing newly transposed national legislation. Guidance on implementation of the Habitats Directive at EU level was generic and many previous case studies of discussions between Member States and the European Commission had been resolved only through case law in the European Court of Justice. Therefore, protocols, procedures and thresholds of impact had to be developed. Over the past 10 years, the second and third five-year fishery management plans operating at the site have worked well. Fishers are key active partners in the annual cockle survey and in the annual razor clam survey. The introduction of a limited-entry permit system has increased the sense of ownership and stewardship in the fishing community. This is evidenced by voluntary collective decisions not to harvest cockles in a given year when market conditions are poor (2022) and to forego potentially good harvests when cockle biomass is very high (the measures in the third fishery plan are more conservative and driven by industry). The annual scientific monitoring has shown that cockle stocks and the abundance and distribution of characterising bivalve species, although highly recruitment dependent, are

⁽⁵⁾ Shellfish Fisheries Reviews, 2009–2022, <https://www.marine.ie/> (accessed 11 January 2024).

⁽⁶⁾ www.sfpa.ie (accessed 11 January 2024).

⁽⁷⁾ Shellfish Fisheries Reviews, 2009–2022, <https://www.marine.ie/> (accessed 11 January 2024).

relatively stable. Waterbirds are declining but this is most likely because of factors outside the site and such declines are also evident at regional, national and EU level (Burke et al., 2018). Further engagement of other stakeholders, including all recreational users, and increasing the awareness of the importance of the site in the local community are important next steps in protecting the site.

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Case Study Report

The Madeira Archipelago Portugal – Macaronesia



Mapping of marine protected areas and their associated fishing activities

Nuno Castro

MRAG Europe

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LIST OF ABBREVIATIONS

Term	Description
AIS	Automatic Identification System
API	Application Programming Interface
CECAF	Fishery Committee for the Eastern Central Atlantic
DGPA	Direção-Geral das Pescas e Aquicultura (General Directorate for Fisheries and Aquaculture)
DOP	Departamento de Oceanografia e Pescas (Department of Oceanography and Fisheries, University of Azores)
DREM	Direção Regional de Estatística da Madeira (Regional Statistical Office of Madeira)
DROTE	Direção Regional do Ordenamento do Território
EEZ	Exclusive Economic Zone
EMODnet	European Marine Observation and Data Network
EU	European Union
EUNIS	European Nature Information System
FAO	Food and Agriculture Organization
GDP	Gross Domestic Product
GFW	Global Fishing Watch
Ha	Hectare(s)
ICCAT	International Commission for the Conservation of Atlantic Tunas
ICES	International Council for the Exploration of the Sea
IFCN	Instituto das Florestas e Conservação da Natureza – IP-RAM
INE	Instituto Nacional de Estatística (Statistics Portugal)
IUCN	International Union for Conservation of Nature
kW	Kilowatt
MPA	Marine Protected Area
nm	Nautical Mile
SAC	Special Area of Conservation
SPA	Special Protection Area
SRA	Secretaria Regional Ambiente, Recursos Naturais e Alterações Climáticas (Regional Secretariat for the Environment, Natural Resources, and Climate Change)

1. EXECUTIVE SUMMARY

This case study aimed to explore fisheries' spatial reallocation in response to the implementation of marine protected areas (MPAs) in the Madeira archipelago. This analysis was based on a systematic literature review and an exploration of publicly available automatic identification system (AIS) data. Additionally, a synopsis of the habitats found throughout the MPAs was described, and MPA policies to regulate fishing activities were reviewed to identify and describe possible complementary management measures.

Two MPAs were included in the analysis because of the limited data timeframe (2012–2022): Ponta do Pargo MPA and Selvagens MPA. In the Ponta do Pargo MPA, where fishing is allowed, the changes in fishing patterns could be attributed either to natural variations in activities or to the implementation of the MPA, making it difficult to draw conclusions. The Selvagens MPA had its protection extended to 12 nautical miles (nm) in May 2022. The analysis conducted in this MPA suggests that its expansion did not result in a redistribution of fishing activities (at least in 2022). Interestingly, fishing near the Selvagens MPA ceased months before the 12 nm extension.

Within several MPAs in Madeira, restrictions have likely been enacted over areas of little historical fishing activity, with such designation then unlikely to have a substantial impact on regional fisheries activity and the long-term sustainability of such activities. In addition, as fishing effort in this region is predominantly centred on deep-water habitats, designating coastal habitats as MPAs (where the majority of MPAs in Madeira are situated) is unlikely to have substantial impacts on the sustainability of the local fishing industry but will minimise any negative interactions with such operations.

Unlike in other Macaronesia islands, a top-down MPA implementation appears to be effective in Madeira and is the most common method used to designate and implement MPAs. Such implementation has been wholly undertaken without prior studies or stakeholder engagement. However, within Madeira, this implementation strategy appears to be effective as fishing occurs offshore and in deeper waters. It is suitable when MPAs are implemented in areas of little historical fishing activity or in areas where there is likely little presence of local and regional stakeholders.

This report underlines the potential next steps for Madeira MPAs, such as making fisheries data more accessible, enhancing monitoring, and updating MPA management plans. Despite this, Madeira has already designated a range of MPAs, with a focus on the use of no-take areas, including the largest no-take MPA within Europe, with the Selvagens MPA covering 2,677 km² with no-take restrictions.

Even without MPA protection, within Madeira, a range of gear restrictions have been implemented for fishing activities greater than 200 metres in depth (to protect deep-water coral reefs). Such restrictions have been enacted to preserve deep-water habitats and may be expected to reduce any fishing impacts on such habitats. In this respect, Madeira has taken substantial positive steps towards ensuring that the main fishing habitats supporting its deep-water fishery are protected, potentially ensuring the sustainability of such fishing activities.

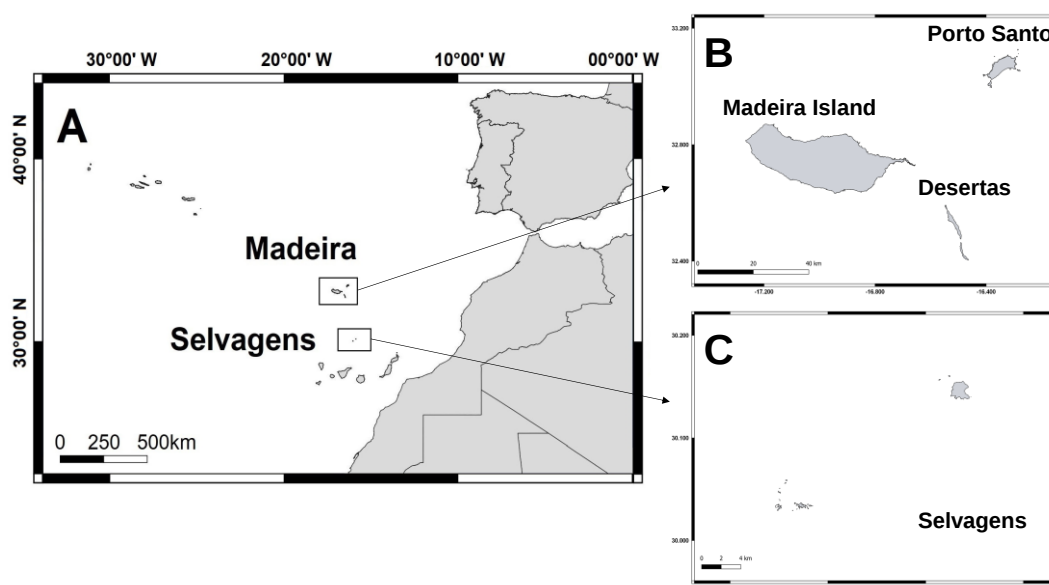
Overall, a lack of fisheries data, little to no regular MPA monitoring, and outdated management plans potentially reduce the effectiveness of MPAs. However, with the inherent diversity of MPA zoning and the specific tailoring of MPA strategies to specific deep-water habitats amid enhanced preservation of traditional and small-scale fisheries,

the region appears to be on track to meet the European Union (EU) Biodiversity Strategy 2030 goals.

2. BACKGROUND

Madeira, officially the Autonomous Region of Madeira, is an autonomous region of Portugal. This oceanic archipelago is located in the North Atlantic at 32°24' N, between 16°16' and 17°16' W (Figure 1). The archipelago comprises Madeira and Porto Santo, the two main populated islands, the Desertas Islands and (as a separate archipelago) the Selvagens Islands. Both the Desertas and Selvagens Islands are small, uninhabited island systems⁸. The Madeira archipelago (comprising Madeira, Porto Santo and the Desertas Islands) is 630 km west of Morocco and 900 km from Lisbon, Portugal's capital. In comparison, the Selvagens Islands are situated 280 km south of Madeira and 165 km north of the Canary Islands (Spain). Madeira is part of the Macaronesia biogeographical region along with the Azores, the Canary Islands and Cabo Verde.

According to recent census data available from the Regional Statistical Office of Madeira (DREM), the island of Madeira (termed Madeira thereafter) has 245,595 inhabitants, while Porto Santo has 5,149 inhabitants (DREM, 2022a). Overall, the archipelago is characterised by a relatively high population density (317.2 inhabitants/km² in 2019), almost three times higher than the Portuguese national average. According to Statistics Portugal (Instituto Nacional de Estatística, INE), the population is concentrated on the two main islands, Madeira and Porto Santo (797 and 43 km² respectively), with the highest density occurring in the Funchal municipality (1,367 inhabitants/km²) and the lowest in Port Moniz (28.3 inhabitants/km²) (INE, 2020). The other islands (Desertas and Selvagens) are uninhabited nature reserves.



⁸ The Desertas Islands comprise the three uninhabited islands: Deserta Grande Island, Bugio Island and Ilhéu Chão; the Selvagens Islands comprises two main islands: Selvagem Grande Island and Selvagem Pequena Island.

Figure 1. Map of the Madeira archipelago showing (A) the location of the study area in the Northeast Atlantic, including (B) Madeira Island, Porto Santo, and Desertas, and (C) the Selvagens Islands

The archipelago contributes substantially to the Portuguese gross domestic product (GDP). The archipelago's total GDP for 2019 was EUR 5,069 million, accounting for 2.4, % of the total Portuguese GDP, while per capita GDP was EUR 24,266, 4 % lower than in mainland Portugal (EUR 25,299; 2019 data). The regional economy relies heavily on the tertiary sector (e.g. provision of services to businesses and final consumers), in which growth has been predominantly driven by tourism. In this respect, tourism is the primary source of revenue for the regional economy, comprising 77 % of jobs in the Madeiran tertiary sector. The primary sector, mainly agriculture, accounts for only 9 % of employment, while the secondary sector (e.g. manufacturing, food processing and metal fabrication) makes up 14 % of employment (MRAG, 2022).

Madeira has been an autonomous region since 1976 under the terms of the Portuguese Constitution. Despite this, the Portuguese constitution specifies both a regional and national connection, obliging their administrations to maintain democratic principles and promote regional interests while still reinforcing national unity. In this respect, the regional autonomy does not affect the integrity of the State's sovereignty over Madeira, which is exercised under the framework of the Portuguese Constitution. Despite this, Madeira legislates on matters of specific interest and exercises executive authority in areas such as promoting the economy, development and quality of life, environment, heritage, and regional administration organisation. Madeira is endowed with political and administrative statutes and self-governing bodies such as the Legislative Assembly of Madeira and the Regional Government (Lanceiro, 2010).

The Portuguese Exclusive Economic Zone (EEZ) comprises three subareas: the mainland (287,521 km²), the Azores (930,687 km²) and Madeira (442,248 km²) (Figure 2). In this respect, Madeiran fisheries are managed under the European Union (EU) Common Fisheries Policy. Most of the Madeira EEZ is located within the Food and Agriculture Organization (FAO) Statistical Area 34, except for a small northernmost part within Area 27. The majority of the waters of Madeira lie under the remit of the International Commission for the Conservation of Atlantic Tunas (ICCAT) and the Fishery Committee for the Eastern Central Atlantic (CECAF) (MRAG, 2022).

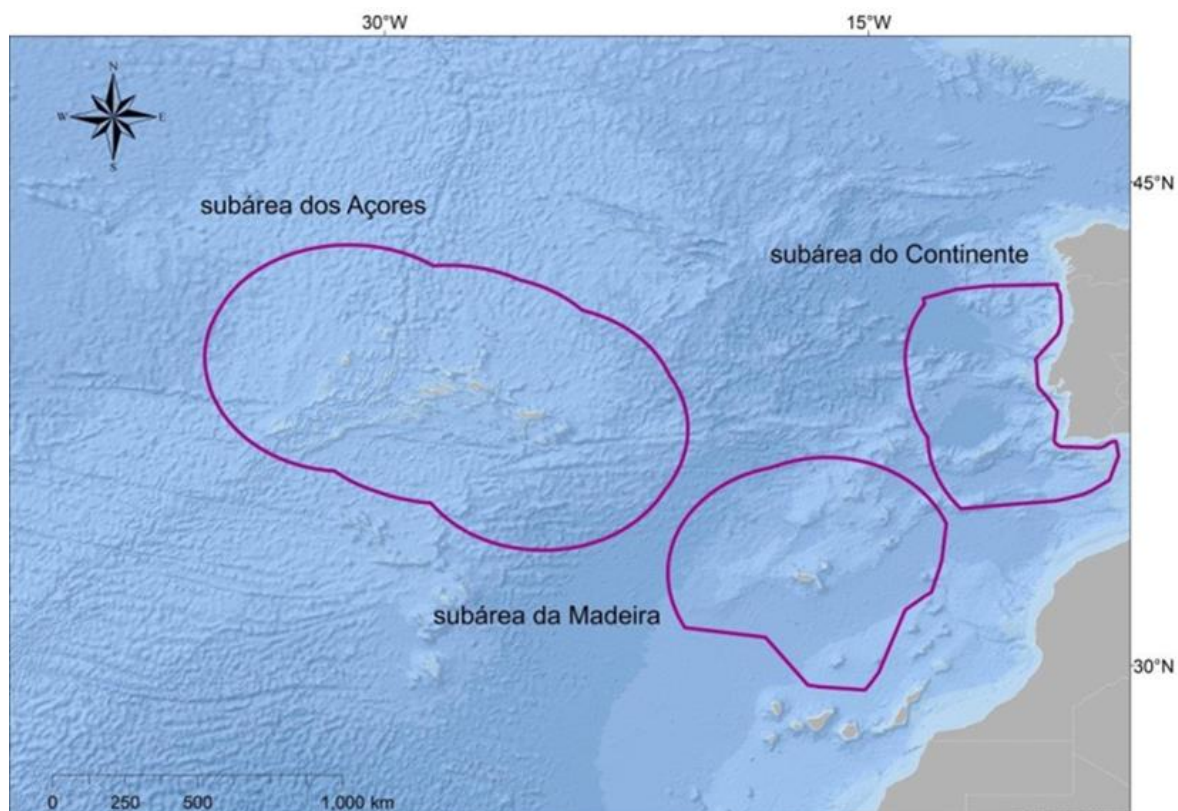


Figure 2. Map of the Portuguese EEZ with its three subareas: mainland, Azores and Madeira (Source: www.dgrm.mm.gov.pt).

Both the Madeira archipelago and Selvagens Islands are of volcanic origin. The Selvagens Islands are the oldest, estimated to be 25.7 million years old. Madeira and the Desertas Islands are the youngest at 7.0 and 5.5 million years, respectively, while Porto Santo falls in between with an age of 18.8 million years (Florencio et al., 2021). According to Spalding et al. (2007), the Madeiran archipelago lies within the temperate northern Atlantic ecoregion, is surrounded by oligotrophic waters (Canning-Clode et al., 2008), with a coastline and shallow waters dominated by rocky reefs (platforms and boulders) and sandy benthic habitats (Monteiro et al., 2021). The average seawater temperature is approximately 20.4°C, with seasonal variations of between 16 and 26°C (Schäfer et al., 2019). The archipelago is influenced by several oceanographic mechanisms, including the Azores anticyclone and the Gulf and Canary currents (Freitas et al., 2019).

The marine waters surrounding the Madeira archipelago form habitat and feeding opportunities for more than 20 whale species, several migratory marine birds, and the monk seal (*Monachus monachus*). In terms of native fishes, there are several regionally important species, including the dusky grouper (*Epinephelus marginatus*), the Mediterranean parrotfish (*Sparisoma cretense*), the striped barracuda (*Sphyraena viridensis*), the round stingray (*Taeniura grabata*) and a variety of temperate and warm temperate fishes.

2.1. Fishing history and current statistics

Fishing in Madeira dates back to the 15th century, with Portuguese settlers colonising the archipelago, primarily using artisanal hand-harvesting methods (Sousa et al., 2019a). The high availability of coastal resources and their easy accessibility encouraged the exploitation of marine shellfish along the coastal stretches of the islands. Fishing activity in Madeira intensified with the growth of human settlements on the islands and technological advancements that brought previously inaccessible coastal areas within reach (Silva and Menezes, 1921; Sousa et al., 2019b). According to Noronha (1925), the (now termed 'traditional') deep-water fishery began in the 1800s when local fishers targeted 'oil fishes' (e.g. deep-water squalid sharks, between 600 and 1,000 m depth) primarily for their fatty livers, oil of which was used for home lighting. Deep-water fishing techniques remained almost unchanged for more than a century until the introduction of monofilament drifting longlines in 1982, which replaced hemp, resulting in an increased number of hooks and increases in the overall durability and workability of the fishing gear (Martins and Ferreira, 1995). These improvements in fishing gears, in addition to better-equipped vessels, helped expand the fishing industry within Madeira, resulting in the use of new fishing grounds (e.g. seamounts) and significant improvements in overall fishing yields (Martins and Ferreira, 1995).

The 1980s were an important time in the development of the fishing industry in Madeira. More cost-effective air transportation allowed the export of fresh products to mainland Portugal and foreign countries (Morato, 2012). In addition, financial support from several political mechanisms (e.g. the European Commission, Regulation 639/2004) reshaped the national Madeiran fleet. This was predominantly through the acquisition of modern vessels equipped with enhanced fishing technology (e.g. the replacement of wooden vessels with modern fibreglass or iron vessels and the use of sonar equipment to target fish schools more easily). Such fleet development increased vessel autonomy and allowed fishing to progress into offshore and deeper areas (e.g. Machete et al., 2010; Pinho et al., 2001; Sousa et al., 2019e).

According to Vallerani et al. (2017), in 2004, the outermost regions, including Madeira, were assigned funds for fleet modernisation to cover the construction of new vessels or to increase the capacity of existing vessels (Regulation (EC) 639/2004). Madeira was one of the regions that appears to have not taken advantage of the opportunity. For example, at present, the small-scale fishing fleet of Madeira (i.e. vessels < 12 metres in length) has an average age of 42 years (i.e. launched 1981), while the average age of vessels > 12 metres and > 24 metres is 21 years (i.e. launched 2002), and those longer than 24 metres in length have an average age of 19 years (i.e. launched 2004). Ageing vessels are less secure and more inefficient than newer vessels and offer less-attractive working conditions for new fishers. In addition, as the waters of Madeira are oligotrophic, inshore resources are relatively low in abundance, resulting in limited opportunities for coastal fishing activities. Such low levels of inshore stock status make offshore resources virtually the only resource that can support the further development of fisheries.

The structure of the fishing fleet has changed over the last 15 years, and this has been associated with reduced numbers of vessels and higher total power. For example, according to the Madeira Regional Service for Statistics (DREM, 2022b), the number of registered fishing vessels has decreased over the last 15 years, peaking in 2007 (155), with the lowest number of vessels registered in 2019 (91). Despite this, total power (kilowatt; kW) has increased over the same period, from an average of 70.3 kW per vessel in 2006 to 111.2 kW per vessel in 2018 (Figure 3). Lastly, according to INE (2021), data from 2021 show that half of the Madeiran fishing fleet was less than 10 metres in length, 30 % were

between 10 and 15 metres in length, and 11 % were between 15 and 24 metres in length. In addition, the largest vessels (> 24 m) comprised the smallest percentage of the entire fleet, with vessels between 24 and 40 metres in length comprising just 3 % of the fleet and vessels 40 metres or more in length comprising 6 % of the fleet (Figure 4).

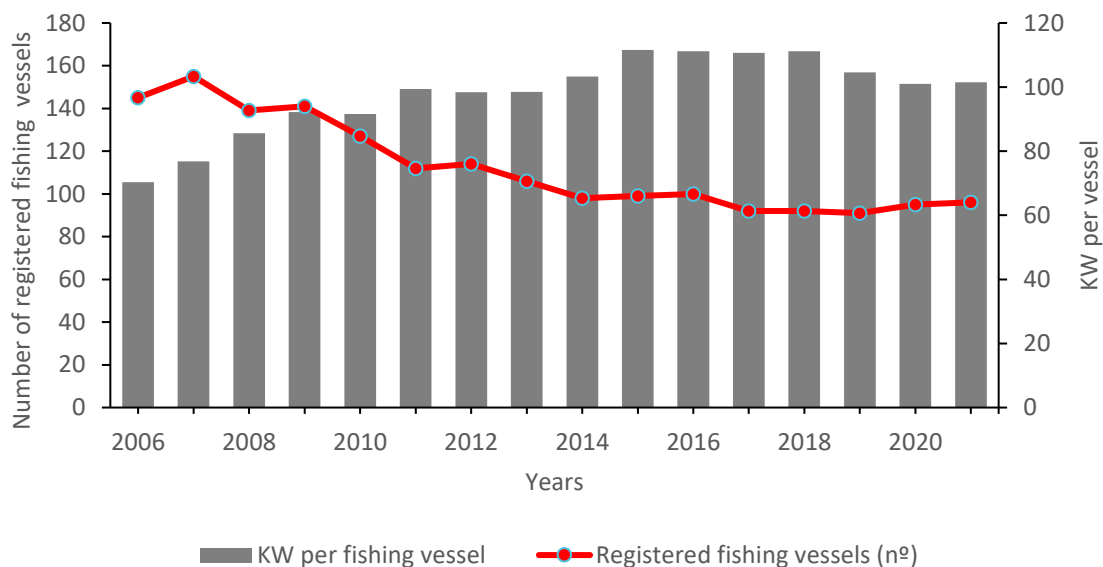


Figure 3. Number of registered fishing vessels in the Madeira archipelago and vessels' total power (kW) over the last 15 years (2006–2021) (Source: DREM, 2022b).



Figure 4. (a) Artisanal fishing vessel (< 12 m in length); (b) longliner (23 m in length); (c) longliner (< 12m in length); and (d) pole and line vessel (> 25m in length) (Source: (a) @Got2globe; (b) @Marco Paulo Jarimba/Marinetrtraffic.com; (c) @Rui Marote /Fuchalnoticias.net; (d) @Rodrigo Freitas/Marinetrtraffic.com).

In the last 15 years, the average annual landings in Madeira totalled approximately 6,000 tonnes, which is approximately EUR 16 million. Landings per fishing vessel have fluctuated across this time, with the lowest recorded in 2010 at 37 tonnes per vessel and high points in 2017, 2018, and 2019 at 87, 81 and 88 tonnes, respectively (Figure 5).

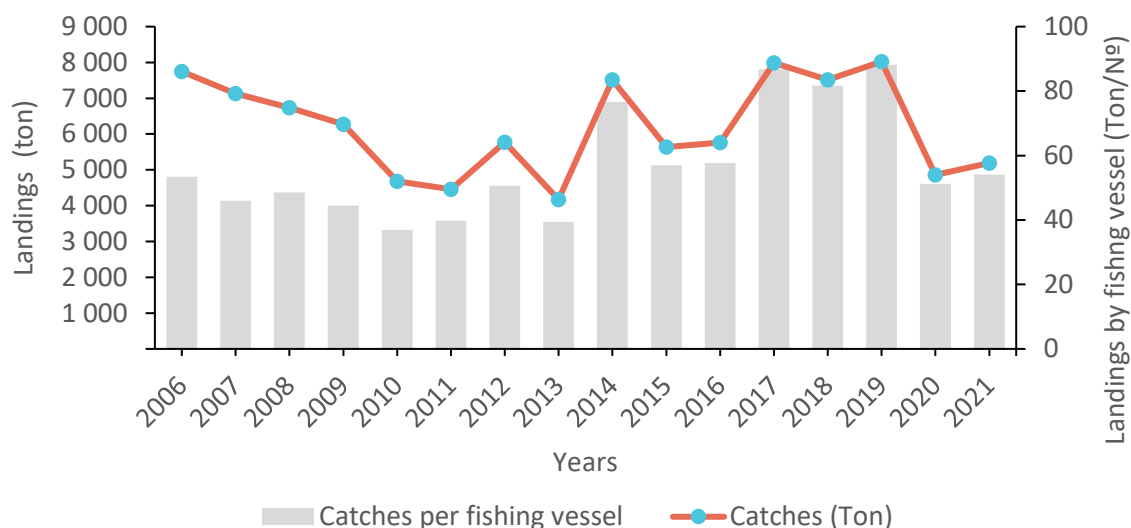


Figure 5. Total landings (metric ton) and landings by fishing vessel (metric ton/Nº) in the Madeira archipelago over the last 15 years (2006–2021) (Source: DREM, 2022b).

2.2. Commercial fishing activities

The Madeira archipelago's geographic and oceanographic characteristics, predominantly the narrowness of the continental shelf, reduced continental slope, high abyssal depth (average depth of around 4,000 m) and oligotrophic waters, restrict the range of economically feasible fishing activities. For this reason, the range of exploitable species in the inshore areas is reduced, requiring fisheries to focus their effort on productive offshore locations, such as seamounts and areas of deep upwelling (DGPA/MADRP, 2007; SRA, 2014). In this respect, the fishing sector in Madeira is divided into three categories that focus predominantly on deep-water or pelagic fishing: (i) deep-water fishing (drifting longlines for scabbardfishes); (ii) tuna fishing (pole and line fishing); and (iii) fishing that focuses on small pelagic fisheries (purse-seine) (DROTA, 2020). In addition, there is a range of other much more small-scale fishing activities, including traps and artisanal hand harvesting for limpet resources (Sousa et al., 2019a; 2019b; 2020a).

Commercial fishing in the Madeira archipelago (Porto Santo, Madeira, Desertas and Selvagens islands) predominantly comprises several pelagic and deep-water species (Table 1). The main catches (data from 2021) comprise tuna (54.1 %, including bigeye tuna [*Thunnus obesus*], albacore [*Thunnus alalunga*] and skipjack tuna [*Katsuwonus pelamis*]), black scabbardfish (36.1 %, *Aphanopus* spp.), blue jack mackerel (3.3 %, *Trachurus picturatus*) and Atlantic chub mackerel (2.1 %, *Scomber colias*) (DREM, 2022b) (Table 1). Importantly, the fishery for black scabbardfish is one of the oldest deep-water fisheries in the Atlantic and one of the region's most economically important ones (Morato, 2012).

Table 1. Description of the fisheries in the Madeira archipelago. For the Domestic commercial fisheries, the International Council for the Exploration of the Sea (ICES) codes for each métier are displayed in brackets. Source: Adapted from MRAG (2022).

Domestic commercial fisheries						
Pole and line (LHP_LPF)	Drifting longliners (LLD_DWS)	Artisanal handline (LHM_LPF)	Artisanal handline (LHM_FIF)	Artisanal bottom longline (LLS_FIF)	Artisanal hand harvesting (MISC_MOL)	Purse seiners (PS_SPF)
Bigeye tuna, albacore, skipjack tuna	Black and intermediate scabbard fish	Bigeye tuna, albacore, skipjack tuna	Bogue (<i>Boops boops</i>), Red porgy (<i>Pagrus pagrus</i>), forkbeard (<i>Phycis phycis</i>), Atlantic wreckfish (<i>Polyprion americanus</i>), blacktail comber (<i>Serranus atricauda</i>), porgies (<i>Pagellus sp.</i>), barracuda (<i>Sphyræna sp.</i>), sea bream (<i>Diplodus sp.</i>), barred hogfish (<i>Bodianus scrofa</i>), comb grouper (<i>Mycteroperca fusca</i>)	Bogue, Red porgy, forkbeard, Atlantic wreckfish, blacktail comber, porgies, barracuda, sea bream, barred hogfish, comb grouper	Limpets (<i>Patella aspera</i> , <i>Patella candei</i>), topshell (<i>Phorcus sauciatu</i> s)	blue jack mackerel, Atlantic chub mackerel, European pilchard (<i>Sardina pilchardus</i>), bogue, curled picarel (<i>Centracanthus cirrus</i>), Madeiran sardinella (<i>Sardinella maderensis</i>), longspine snipefish (<i>Macroramphosus scolopax</i>), boarfish (<i>Capros aper</i>).
Vessels 12–18m, 24–40m	Vessels < 10 m, 12–18m	Vessels < 10 m	Vessels < 10 m	Vessels < 10 m	Vessels < 10 m	Vessels 18–24 m
Domestic recreational fisheries						
Big game fishing		Atlantic blue marlin (<i>Makaira nigricans</i>), white marlin (<i>Kajikia albida</i>), wahoo (<i>Acanthocybium solandri</i>), mahi Kaikai (<i>Coryphaena hippurus</i>), pompano dolphinfish (<i>Coryphaena equiselis</i>), big eye tuna, albacore, skipjack tuna. 43 teleost species, 2 elasmobranchs and 6 invertebrates. Such species include white sea bream (<i>Diplodus sargus</i>), Mediterranean parrotfish (<i>Sparisoma cretense</i>), salema porgy (<i>Sarpa salpa</i>), red porgy, bogue, mullet (<i>Chelon spp.</i>), pompano (<i>Trachinotus ovatus</i>), and several other porgies and sea bream (F. Sparidae).				
Shore fishing						
Spearfishing						
International fisheries (Drifting longliners)						
Vessel type 1		Black scabbardfish, intermediate scabbardfish				

Annual total landings within Madeira primarily fluctuate because of variations in catch associated with the tuna fisheries. Landings were lowest in 2011 (1,368 tonnes), with highs in 2017, 2018, and 2019 at 5,153, 4,683 and 5,131 tonnes, respectively (Figure 6). Variation in annual total catch is also associated with catches of black scabbardfish. The highest catches for this species were recorded in 2006, 2007 and 2008 (2,717, 2,922 and 3,019 tonnes, respectively), with the lowest catches in 2012 (1,716 tonnes) (Figure 6). Atlantic chub mackerel, blue jack mackerel, and other teleost fishes also add to yearly fluctuations in the catch, averaging approximately 200, 381 and 309 tonnes per year (Figure 6). In economic terms, tuna and black scabbardfish comprise EUR 7 million and EUR 6.6 million per year on average to the regional economy (DREM, 2022b), while blue jack mackerel and Atlantic chub mackerel contribute EUR 390,000 and EUR 220,000 per year (DREM, 2022b).

The harvesting and consumption of intertidal invertebrates (e.g. the limpet *Patella* spp and the topshell snail *Phorcus sauciatis*) have high socioeconomic importance within Madeira (Sousa et al., 2019a). For limpets, relatively large landings are apparent: between 2006 and 2017, landings were approximately 110 tonnes annually, with an average annual value of EUR 500,000 (Sousa et al., 2019b). There are no official landing numbers for topshells, as their harvesting is unregulated and does not need to be sold in official auctions (Sousa et al., 2019c).

A number of gear restrictions within the Madeiran islands limit the range of fishing activities that can be undertaken. Bottom trawls (or the use of towed gear that interacts with the benthos), gill nets, entangling nets or trammel nets are prohibited or limited to depths greater than 200 metres (Regulation (EC) No 1811/2004; Council Regulation (EC) No 1568/2005). These permanent restrictions have been implemented to protect deep-water coral reefs, which are included in the list of endangered habitats in the framework of the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention).

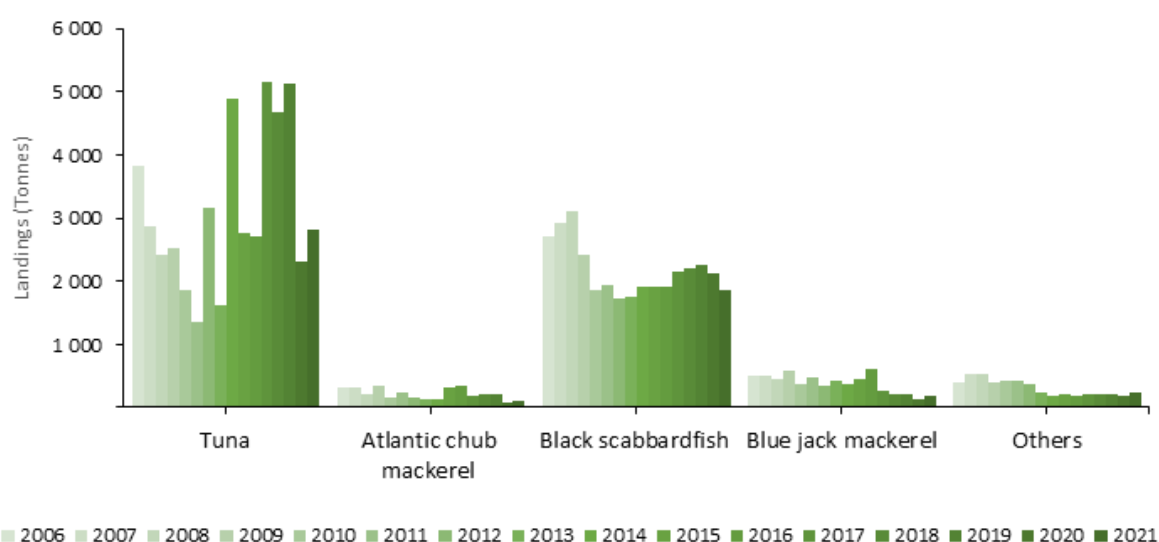


Figure 6. Total landings (tonnes) of the primary organisms fished in the Madeira archipelago between 2006 and 2021 (Source: DREM, 2022b).

2.3. Recreational fishing

The recreational fishing sector in the Madeira archipelago comprises several activities, including boat fishing, shore angling, shellfish harvesting and spearfishing (Martínez-Escauriaza et al.; 2020a; 2020b; 2021a; 2021b). Until 2016, data regarding this fishing sector was almost non-existent (Martínez-Escauriaza et al., 2020a). In 2016, legislation for recreational fishing in Madeira was established (Regional Legislative Decree N° 484/2016, November 14, 2016), creating a series of rules, including requiring a licence issued by the Regional Fisheries Directorate. Such licencing has shown that in 2017, 4,825 shore-angling licences had been provided, while 1,778 spearfishing licenses were provided (Martínez-Escauriaza et al.; 2020a; 2020b; 2021a; 2021b). Moreover, in the same year (2017), it was estimated that in total, the recreational fishing sector (e.g. recreational boat fishery, shore angling and spearfishing) landed approximately 1,500 tonnes of catch (Martínez-Escauriaza et al., 2021a).

The game-fishing fleet has grown in the past decade and is most likely associated with increased tourism. A pilot study under the European Union Multiannual Union Programme for data collection (2017–2019 and extension to 2020–2021) registered 31 vessels, a 106 % increase from 2003. Atlantic blue marlin is the most frequently captured within this fishery, followed by the Atlantic white marlin and other large pelagic fishes. Generally, this fishing fleet enacts a catch-and-release technique, which is believed to limit the impact of this fishing type on these resources. The annual economic gain from big-game fishing in Madeira is estimated at approximately EUR 2 million (MRAG, 2022).

2.4. Marine protected areas in the Madeira archipelago

The Selvagens Islands in Portugal were the first location within Madeira to be designated as a nature reserve (named the Selvagens Islands Nature Reserve), which was implemented in 1971 (Table 2). Over the next 50 years, several other protected areas were established in the Madeira archipelago, with the latest being implemented in 2018. In 2023, the Madeira archipelago has eight MPAs (Table 2 and Figure 7). One MPA is completely no-take (Selvagens MPA: in 2022, this MPA became the first in Madeira to ban all extractive activities, including fishing), three include zoning for no-take areas but may also comprise partially protected areas (Garajau Partial Natural reserve; Desertas Islands Nature Reserve; and Porto Santo Marine Protected Area network), three are wholly comprised of partially protected areas (Rocha do Navio Nature Reserve; Cabo Girão Protected Area; Ponta do Pargo Protected Area), while one protected area has no fisheries restrictions (Ponta de São Lourenço Protected Area) (Freitas et al., 2004).

The MPAs in Madeira vary substantially in size, ranging from less than 5 km², such as Garajau and Cabo Girão (both excluded from the case study because of their small size), to the Selvagens MPA, the largest MPA in the North Atlantic, with full protection covering 2,677 km² (Alves et al., 2022). However, Madeira also holds a marine 'Site of Community Importance' called Cetaceans of Madeira (PTMMD0001) (not included in this case study because of the paucity of data on the impact of this protection). This site was created in late 2017 to protect bottlenose dolphins (*Tursiops truncatus*), other species of cetaceans, sea turtles, and monk seals (Regional Resolution n° 699/2016). The Cetaceans of Madeira MPA covers a total area of 681 980 ha, which corresponds to a polygon covering all coastal marine waters around Madeira, the Desertas Islands and Porto Santo (encompassing one nautical mile away from each coastline). The development of a special programme plan (i.e. management) for this MPA is still ongoing (Regional legislative dispatch n°221/2020).

MPAs within Madeira comprise an array of International Union for Conservation of Nature (IUCN) categories. These range from strict nature reserves (Category Ia: Selvagens MPA, Desertas Islands) to those categorised as areas for sustainable use of natural resources (Category VI: Ponta do Pargo Protected Area). Some MPAs are designated as Special Areas of Conservation (SACs) and/or Special Protection Areas (SPAs) and may overlap with one another and, in some cases, protect terrestrial and marine elements. In addition, some of the MPAs are zoned for multiple uses. For example, the Desertas Islands and Porto Santo MPAs network both have no-take areas as well as complementary protected areas with less stringent regulations.

The majority of MPAs in Madeira have been established through a top-down approach led by the central government⁹. These MPAs have been implemented predominantly to protect marine species with a high risk of extinction (e.g. monk seal), protect the marine environment (e.g. Rhodoliths), conserve nature and/or promote artisanal fisheries¹⁰.

Table 2. Marine protected areas of the Madeira archipelago with the extent of the marine area (km²), fisheries restrictions, implementation year and IUCN category
Source: data retrieved from <https://ifcn.madeira.gov.pt/> (accessed 12 January 2024).

Implementation year	Area	Marine area (km ²)	Type of protection	MPA designation type
1971 (extended in May 2022)	Selvagens MPA	2,677	No-take zone	IUCN 1A – includes a SAC (PTSEL0001) and an overlapping SPA (PTZPE0062).
1982	Ponta de São Lourenço Protected Area	26.09	No fisheries restrictions	IUCN V – includes a SAC (PTMAD0003) and an overlapping SPA (PTZPE0064).
1986	Garajau Partial Natural reserve	3.11	No-take zone*	IUCN IB
1990	Desertas Islands Nature Reserve	119.77	Includes a no-take zone* and a partially protected area (bottom trawling, trammel nets, and purse-seine* are not allowed).	IUCN IA – includes a SAC (PTDES0001) and an overlapping SPA (PTZPE0063).

⁹ The implementation of Garajau is the exception to this criteria, which was proposed by a group of scuba divers (Castro et al., 1985).

¹⁰ <https://ifcn.madeira.gov.pt/> (accessed 12 January 2024).

Implementation year	Area	Marine area (km ²)	Type of protection	MPA designation type
1997 (SAC from 2009)	Rocha do Navio Nature Reserve	17.52	Partially protected area (bottom trawling, trammel nets, and purse-seine* are not allowed)	IUCN IB – includes a SAC (PTMAD0004)
2008	Porto Santo MPA network	26.67	Includes a no-take zone and a partially protected area (commercial fishing is not allowed*).	IUCN IB – includes a SAC (PTMAD0003)
2017	Cabo Girão Protected Area*	2.55	Partially protected area (professional, recreational, fishing and harvesting are permitted under specific regulations)	IUCN VI
2018	Ponta do Pargo Protected Area	15.40	Partially protected area (professional, recreational, fishing and harvesting are permitted under specific regulations)	IUCN VI

*Except for capturing bait-fish for tuna fishing.



Figure 7. The location of MPAs in the Madeira archipelago is highlighted in red

In 2018, the Ponta do Pargo Protected Area (Regional Legislative Decree – 19/2018/M) was created, which includes a land and marine area located in the south-west area of Madeira Island. This park (hereafter referred to as Ponta do Pargo MPA), categorised as IUCN category VI, has been implemented to protect, enhance, and sustainably utilise resources in the region, complying with the provisions of the National Strategy for the Sea and the requirements of the Marine Strategy Framework Directive¹¹. In this respect, with the respective licences, commercial fishing, recreational fishing, and artisanal harvesting are allowed in this MPA (Regional Legislative Decree – 19/2018/M).

In November 2019, the Regional Government of Madeira announced the creation of the largest total MPA in Europe through the approval of a new legal regime focused on the Selvagens MPA. This significantly expanded the protection of the waters of this archipelago. This decision was supported by scientific and legal studies and by data collected in expeditions carried out in recent years, with some declaring the area as a pristine location (Friedlander et al., 2017). The declaration of this marine area of total protection covers a territory of 2,677 km², 12 nautical miles around the Selvagens Islands. In this extension, all species became fully protected from extractive activities.

3. AIMS AND OBJECTIVES

The main aim of this case study was to assess the spatial reallocation of fishing activities in response to MPA implementation in the Madeira archipelago. This analysis relied on two substantial data sources: a broad literature review and an analysis of publicly available automatic identification system (AIS) data. In support of this work, this report also provides a synopsis of the habitats found throughout the MPAs. Lastly, Madeira's policies to regulate fishing activities in the vicinity of selected MPAs are reviewed to identify and describe possible complementary management measures.

4. METHODS

4.1. Systematic literature review of fishing activities

A systematic literature review was conducted to verify fishing activities in and around Madeiran MPAs and document the dynamics of fisheries activity and potential strategies adopted following MPA implementation (see Annex 4 in the main report for detailed methodology). The case study; and 14 papers had no reports of fishing activities – effort, catches, landings or profit.

4.2. Fisheries reallocation after MPA implementation

A targeted data call was conducted to the Madeiran regional authorities to request high-resolution vessel monitoring system data for larger commercial vessels, AIS and log-book data (where available), and any additional metrics of small-scale fishing activities that may not be publicly available. This data call was sent to the Secretaria Regional de Mar e Pescas, the Direção Regional do Mar and the Direção Regional das Pescas. However, despite several

¹¹ <https://ifcn.madeira.gov.pt/> (accessed 12 January 2024).

emails to request such data, only the Direção Regional das Pescas replied, stating that the Direção Regional do Mar had the only access to such data. Unfortunately, follow-up emails to the Direção Regional do Mar did not result in any replies.

As no fisheries data were available, this study used Global Fishing Watch (GFW) data via the GFW application programming interface (API) using Rstudio (RStudio Team, 2020) in order to gain insight into how the establishment of MPAs in Madeira has affected the distribution of fishing activities spatially and temporally. Setting up the mapping data required the use of a shapefile outlining the EEZ and islands' borders derived from a shapefile by Flanders Marine Institute (2019). The shapefile of the Madeiran MPAs was sourced from Instituto das Florestas e Conservação da Natureza IP-RAM (IFCN).

Using the GFW API in Rstudio, fishing data was obtained across the Madeiran EEZ for 2012, 2018 and 2022. This data was received at the highest resolution possible and summarised into $0.1^{\circ} \times 0.1^{\circ}$ grid square cells (the resolution of $0.1^{\circ} \times 0.1^{\circ}$ is equivalent to approximately $11 \times 11 \text{ km}^2$ at the equator). These data were then plotted as maps for each of the three years obtained across the entire EEZ, with all gear types aggregated to show total annual effort (fishing hours) within the region while also displaying the MPAs that had been established for that particular period in time (i.e. MPAs established in 2022 were not shown on the 2012 and 2018 maps). These map plots were then repeated as faceted plots, disaggregated for all gear types present within the data to visualise the distribution of effort for different gears in a year.

Secondly, a more focused mapping analysis was conducted for two specific MPAs: Ponta do Pargo MPA and Selvagens MPA. As Ponta do Pargo was established in 2018, data were retrieved from GFW for the period 2017–2019. These annual datasets were again summarised into $0.1^{\circ} \times 0.1^{\circ}$ grid square cells, providing annual maps of total fishing effort distribution within and surrounding the Ponta do Pargo MPA. Further, maps of 2018–2019 were repeated but disaggregated by gear type, displaying the spatial distribution of effort for each gear within and surrounding the Ponta do Pargo MPA.

The Selvagens MPA extended its protection in May 2022. For this MPA, we conducted two different analyses. The first assessment was based on 2022 only. This 2022 dataset was plotted by month to determine whether fishing was taking place within the boundary of the MPA before and after its extension. Faceted plots were produced by disaggregating all gear types within the data, visualising the distribution of fishing effort (total hours) for different gears in a year. The second analysis used all the available GFW data (2012–2022), where the map plots were generated per year, with the MPA boundaries before and after the extension. The faceted plots were created with disaggregated gear types present within the data and by the country flag. In both analyses, the data were summarised into $0.1^{\circ} \times 0.1^{\circ}$ grid square cells.

4.3. Description of habitats and bathymetry within the MPAs

To gather information pertaining to habitats and bathymetry in MPAs and surrounding areas searches were conducted on the European Nature Information System (EUNIS) and the European Marine Observation and Data Network (EMODnet) to identify habitats, substrate types and bathymetry. A literature review was performed to complement the former searches, using SciVerse Scopus (<https://www.scopus.com>) and Google Scholar databases (<https://scholar.google.com/>). The search terms used to locate and acquire relevant literature were related to the topic in hand (e.g. "Madeira Island" or "Madeira archipelago", "habitat", "depth" and related synonyms). A grey literature search was also performed.

A targeted data call was made to relevant regional authorities to complement the existing information sources gathered. To access certain data, including information gathered from scientific and oceanographic cruises, it must be requested from the regional government institutions, as they are responsible for its management and dissemination. This call was made to two regional government institutions, IFCN and DROTE (Direção Regional do Ordenamento do Território), involved in MPAs in Madeira (Horta e Costa, 2017). IFCN is the competent authority in managing Madeira archipelago MPAs, and DROTE is responsible for Marine spatial planning and MPA implementation.

5. RESULTS

5.1. Systematic literature review of fishing activities

Of the four documents identified for use in the literature review, these provided details of interactions between fisheries and MPAs in the Madeira archipelago. However, none of these documents examined the effects on fishing before and after MPA implementation. Three of these four studies focused on artisanal hand harvesting. These studies showed that topshell populations inside MPAs had larger shell lengths and sustained a higher catch per unit of effort (Sousa et al., 2019d). In comparison, the effects of protection on limpet populations resulted in a differential increase in size at first maturity, shell size, and capture per unit effort according to the degree of protection (Sousa et al., 2020b). Old and enforced MPAs showed the best-preserved limpet populations in the study area, with larger adult specimens and a more balanced population.

This review also found a single policy report that analysed the level of protection of Portuguese MPAs (SCIAENA, 2021). In this report, the authors studied contemporary fishing effort occurring inside MPAs based on information from GFW, using AIS satellite signals to track down large-scale fishing vessels (> 15 m)¹². Between 2015 and 2018, the authors found that MPAs in the Madeira archipelago had the lowest overall fishing effort compared to MPAs within the Azores and the Portuguese mainland. It was argued that the predominance of longlines in very deep waters in the region results in low fishing activity inside or near MPAs. Lastly, Marques (2009) surveyed fishers' opinions on MPAs within Madeira. This work shows that the majority of fishers had a positive view of MPAs, including their potential support for biodiversity. Most fishers stated that their fishing activities were conducted in distant waters, so coastal MPAs did not affect them or their actions.

5.2. Fisheries reallocation after MPA implementation

5.2.1. Madeira archipelago

The GFW analysis showed a rise in fishing effort across the Madeira archipelago from 2012 to 2022 (Figure 8). Most fishing activities in 2012 were centred on several seamounts located between Madeira and mainland Portugal rather than in the vicinity of the Madeira archipelago and its protected areas, which aligns with the recent analysis of such data

¹² Such data potentially underestimate overall fishing effort because AIS data is predominantly from vessels longer than 15 m, which make up only 20 % of the fleet fishing Madeiran waters (SCIAENA, 2021).

(Wienberg et al., 2013). In 2018, fishing activities increased, with the expansion of fishing grounds closer to the Madeira archipelago. Most fishing activities were centred on Madeira Island, Porto Santo and inside the Selvagens MPA. In 2022, fishing intensity increased again, though with relatively little change in the overall fishing structure. Importantly, there was an absence of fishing surrounding the Selvagens MPA (especially in 2022, following the extension of the MPA) and a rise in fishing activities within the western border of the Madeiran EEZ¹³.

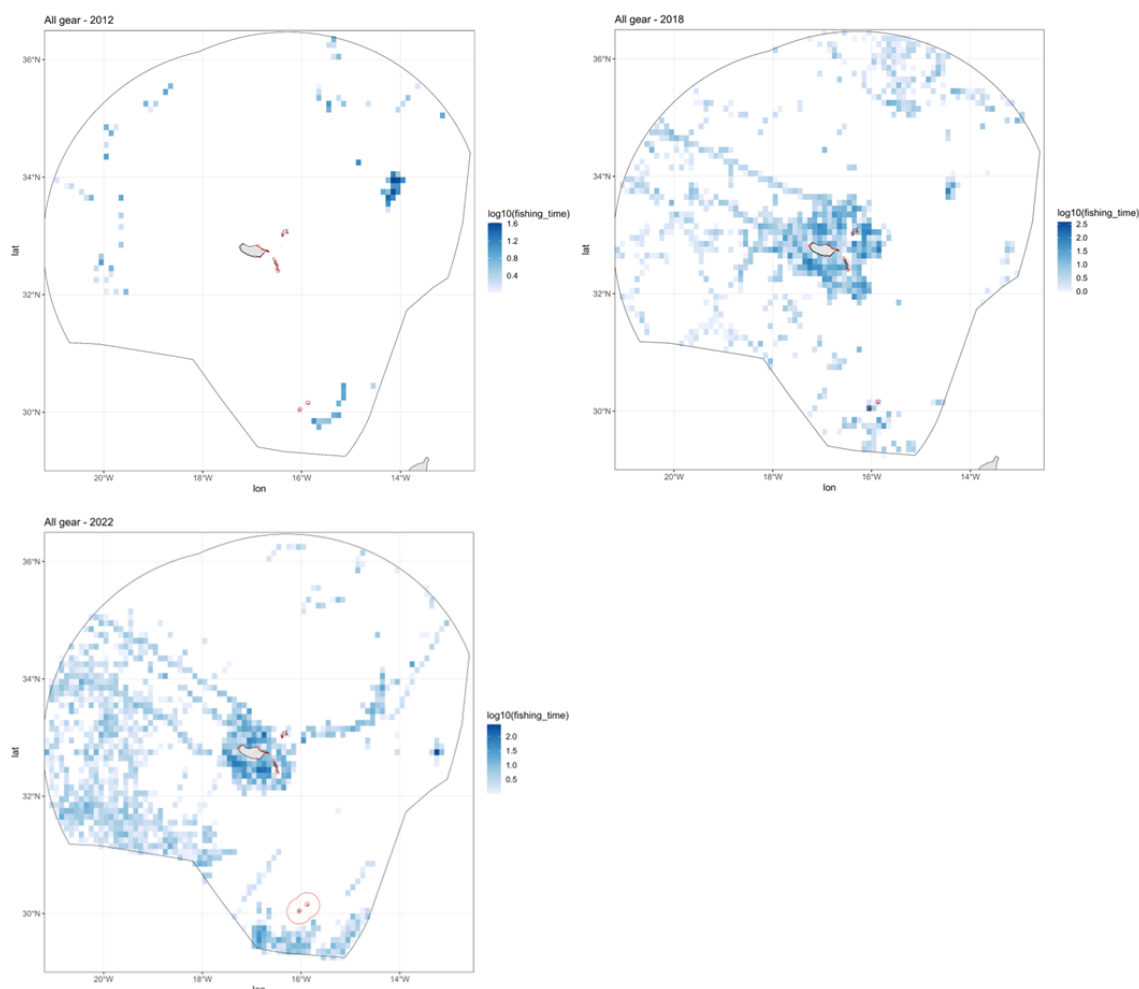


Figure 8. Total fishing effort in the Madeira archipelago in 2012, 2018 and 2022. Plots were obtained from GFW; red lines indicate the locations of the MPAs.

The disaggregated métiers made it possible to verify that most fishing was conducted with drifting longlines and pole and line, regardless of the year (Figure 9). In addition, the activities of the fishing fleet in recent years (2018 and 2022) have been characterised by increases in the intensity of fishing effort across the métiers (Figure 10, Figure 11). Such activities are particularly close to the Madeira MPAs, with various fishing techniques applied, such as drifting longlines, pole and line, and purse-seine, among others.

¹³ Please note, the levels of fishing activity found within this analysis may also be attributed to the evolution of the Global Fishing Watch dataset (e.g. Park et al., 2023) and the mandatory use of AIS in all EU-flagged fishing vessels over 15 metres (which became effective in May 2014).

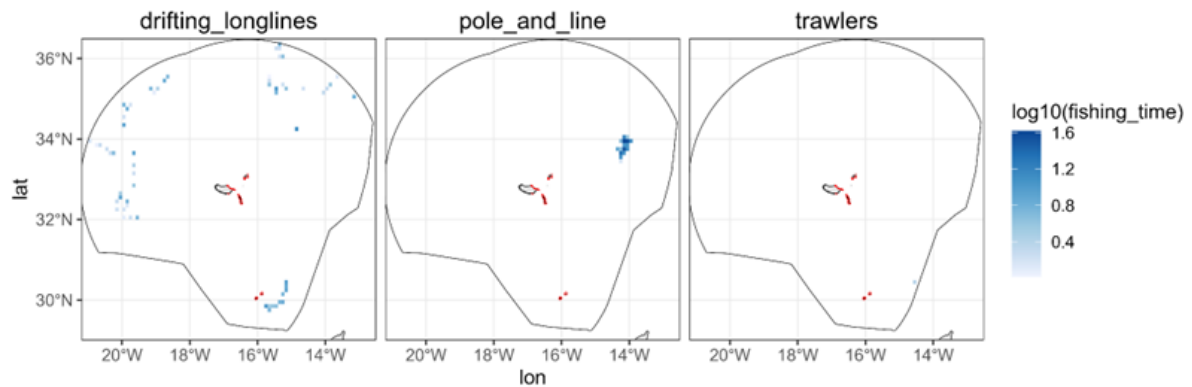


Figure 9. Total fishing effort in the Madeira archipelago for 2012 by fishing gear. Plots were obtained from GFW; red lines indicate the locations of the MPAs.

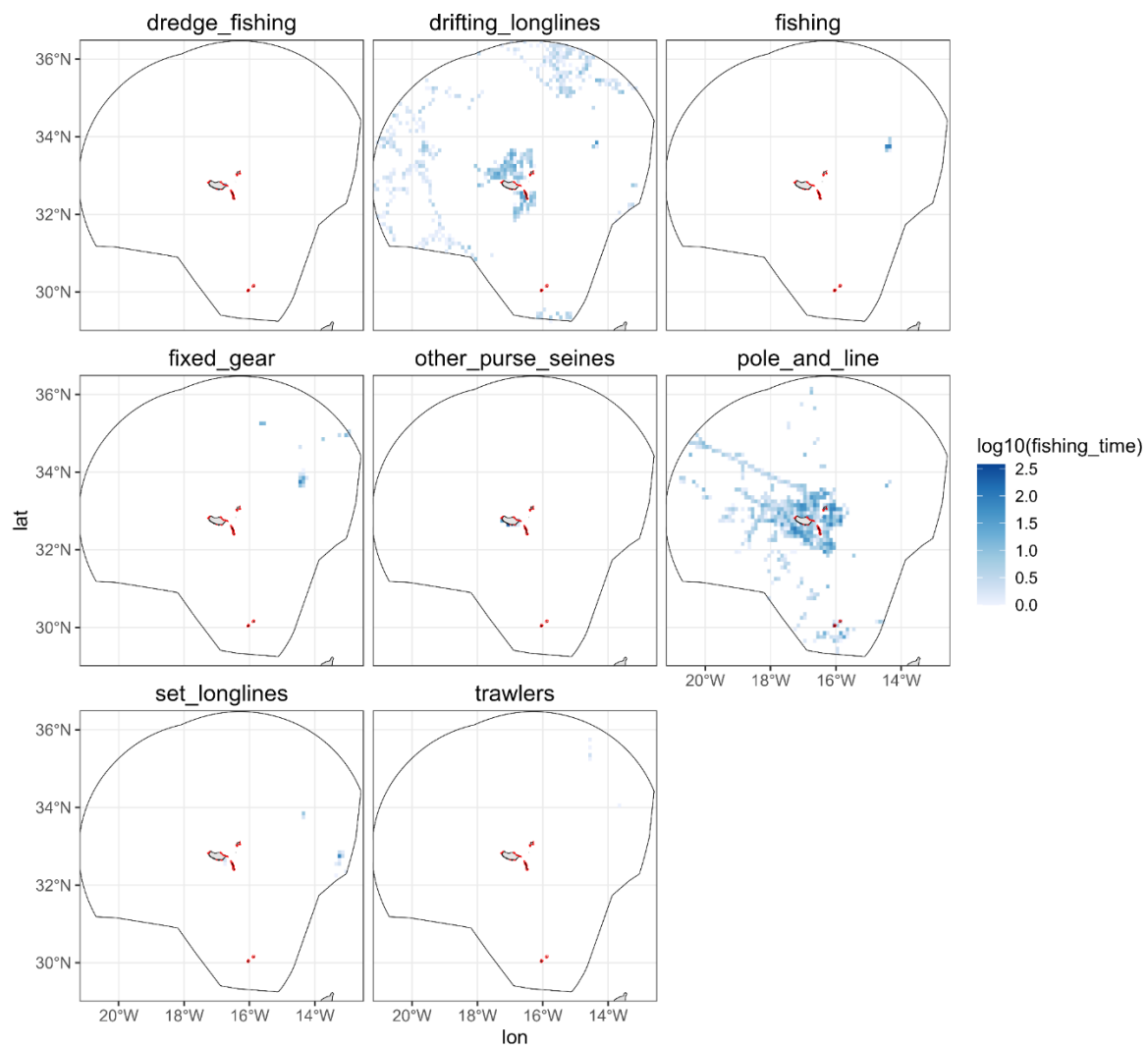


Figure 10. Total fishing effort in the Madeira archipelago for 2018 by fishing gear. Plots were obtained from GFW; red lines indicate the locations of the MPAs. The term 'fishing' is used when GFW cannot define with confidence which fishing gear is being used.

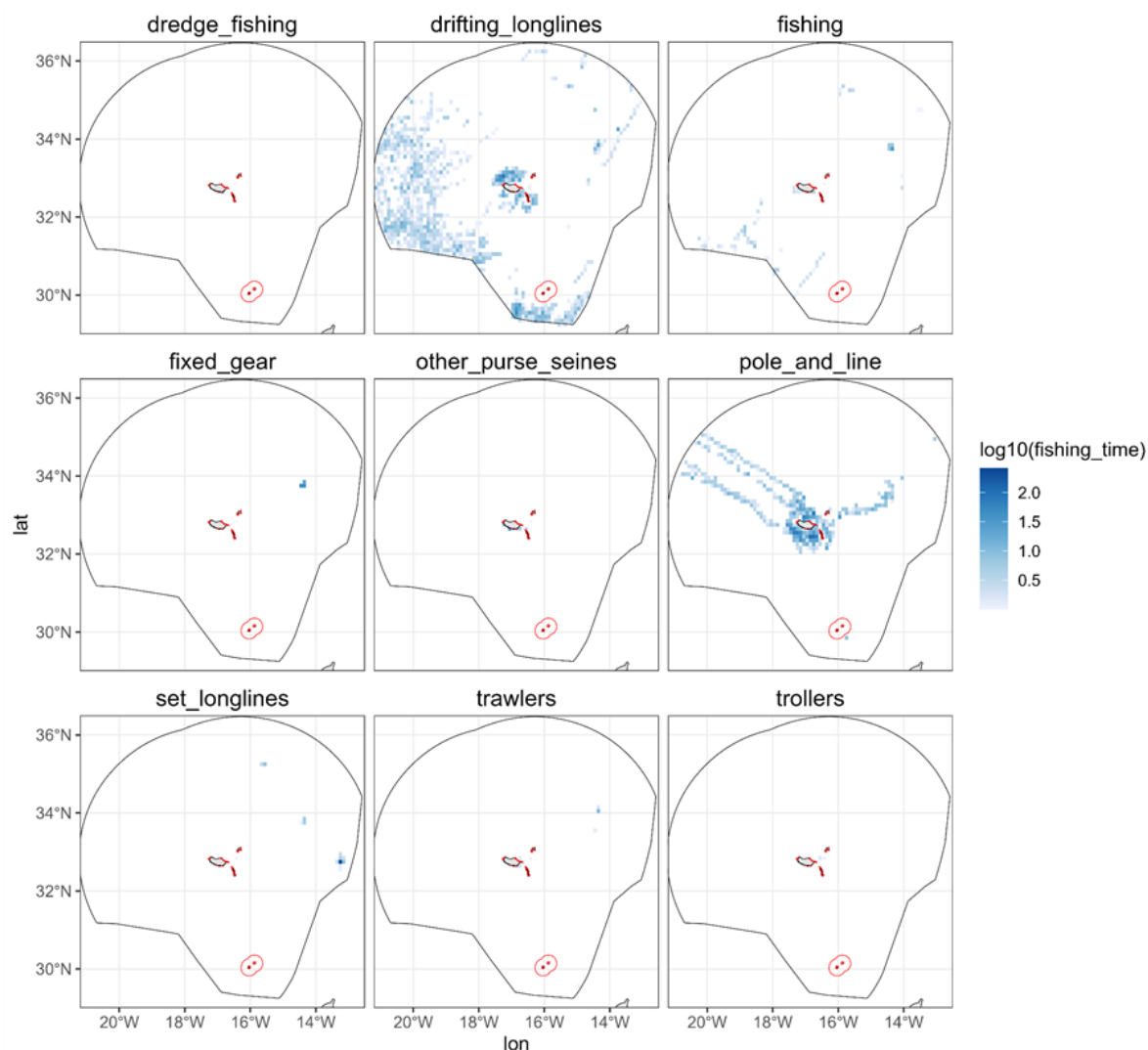


Figure 11. Total fishing effort in the Madeira archipelago for 2022 by fishing gear. Plots were obtained from GFW; red lines indicate the locations of the MPAs. The term 'fishing' is used when GFW cannot define with confidence which fishing gear is being used.

5.2.2. Ponta do Pargo MPA

Before the Ponta do Pargo MPA was established in 2018, there was moderate fishing activity in this area. For example, in 2017, most of the fishing occurring in that MPA and its boundaries were undertaken by pole and line, drifting longlines and purse-seine (Figure 12). Following the implementation of the MPA, fishing activities remained variable in the following years (Figure 13). The dominant fishing gears detected in and surrounding the MPA were pole and line in 2018 and 2019 (Figure 13).

Within this MPA, commercial fishing, recreational fishing, and artisanal harvesting are allowed with the respective licences (Regional Legislative Decree – 19/2018/M). Therefore, despite the implementation of the MPA, there is no reason to suggest that fishing activities should be reduced.

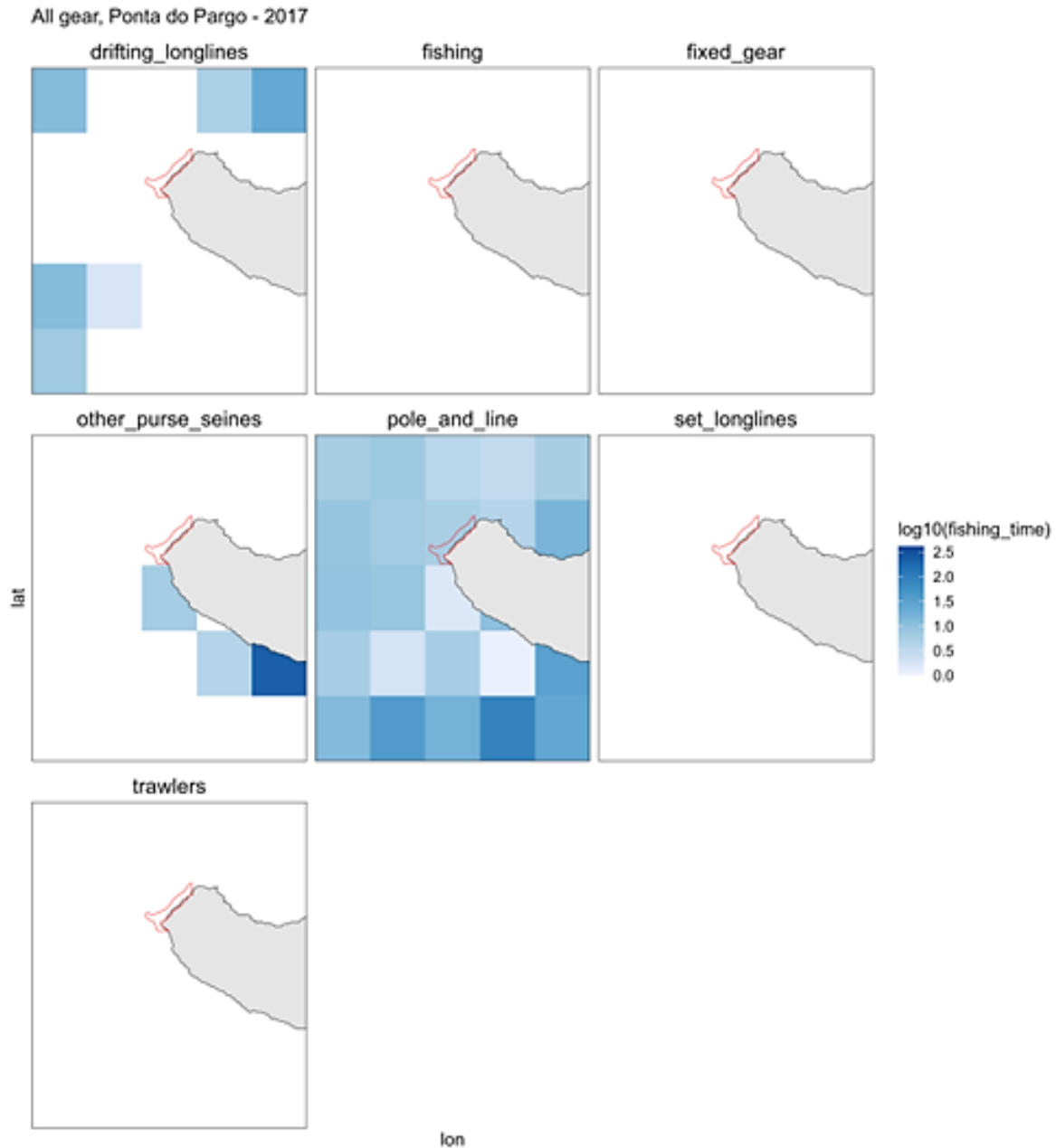


Figure 12. Total fishing effort in the Madeira archipelago in 2017, by fishing gear, before the Ponta do Pargo MPA was established. Red line indicates Ponta do Pargo MPA. The term 'fishing' is used when GFW cannot confidently define which fishing gear is being used. Similar to 'fixed gear' is when GFW cannot accurately define which fixed gear is being used (e.g. pots and traps, longlines and gill nets).

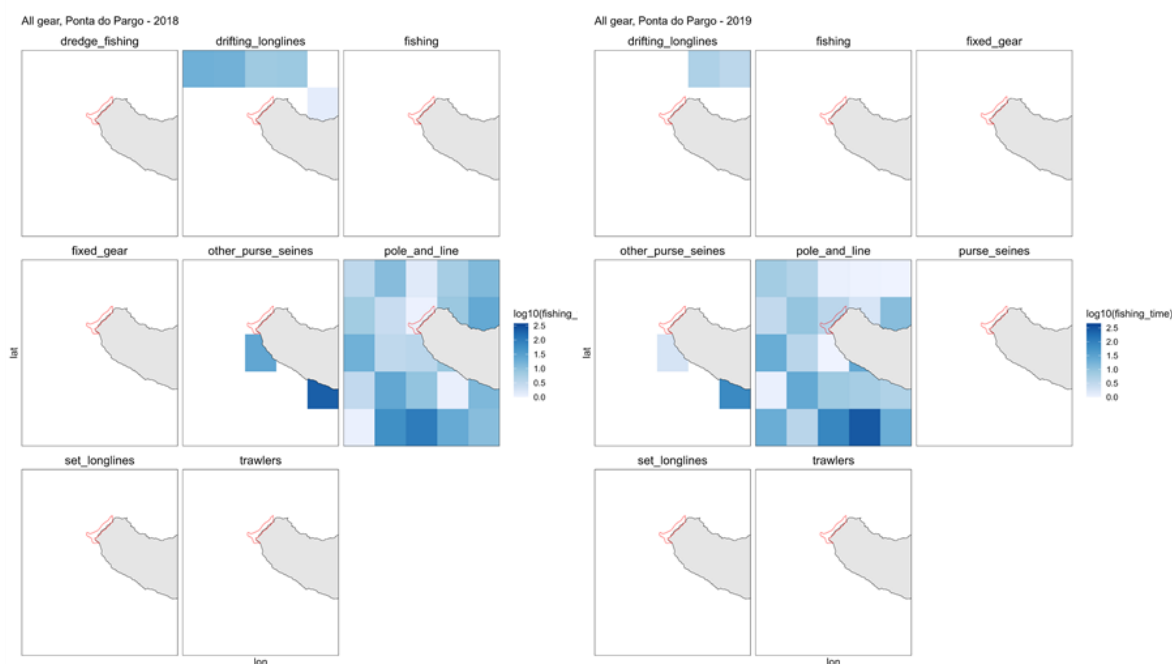


Figure 13. Total fishing effort by fishing gear from 2018 to 2019 after implementing the Ponta do Pargo MPA. Red line indicates Ponta do Pargo MPA. The term 'fishing' is used when GFW cannot confidently define which fishing gear is being used. Similar to 'fixed gear', is when GFW cannot accurately define which fixed gear is being used (e.g. pots and traps, longlines and gill nets).

5.2.3. Selvagens MPA

In 2022, the monthly AIS activity surrounding the Selvagens MPA showed no evidence of fishing activities within or around its boundaries, both before and after its establishment in May 2022 (Figure 14). Where fishing occurs in the Selvagens MPA region, Portuguese and Spanish vessels were the most frequently recorded (Figure 15, Figure 16 and Figure 17). There was only a record of Belize (BLZ) vessels in 2012 (Figure 15). Over the years, the Portuguese fishing vessels were the only ones fishing inside/near the MPA borders. As for the Spanish fleet, they fished around the 12 nm mark (Figure 15, Figure 16, and Figure 17). The fishing intensity increased for both fleets until 2022 when no Portuguese vessels were detected (Figure 17).

Pole and line was the most utilised fishing gear inside/near the Selvagens MPA over the studied period (Figure 18, Figure 19, and Figure 20). Other fishing gears, such as drifting longlines and purse seine, were recorded out or near the 12 nm mark (Figure 18, Figure 19 and Figure 20). In 2022, all the observed fishing gear was used outside the MPA border (Figure 20).

This analysis suggests that the expansion of the MPA did not result in a redistribution of fishing activities (at least in 2022). In comparison, the temporal analysis verified that pole and line fishing occurred over the years inside/near the border of Selvagens MPA, and in 2022, that fishing stopped. Such cessation of fishing activities within the border of the Selvagens MPA was likely associated with substantial public interest, as the extension of the MPA was a highly broadcasted event with several

international and national entities¹⁴. Government agencies regularly visited the Selvagens MPA in 2022, leading fishers to pursue other fishing grounds to avoid the authorities.

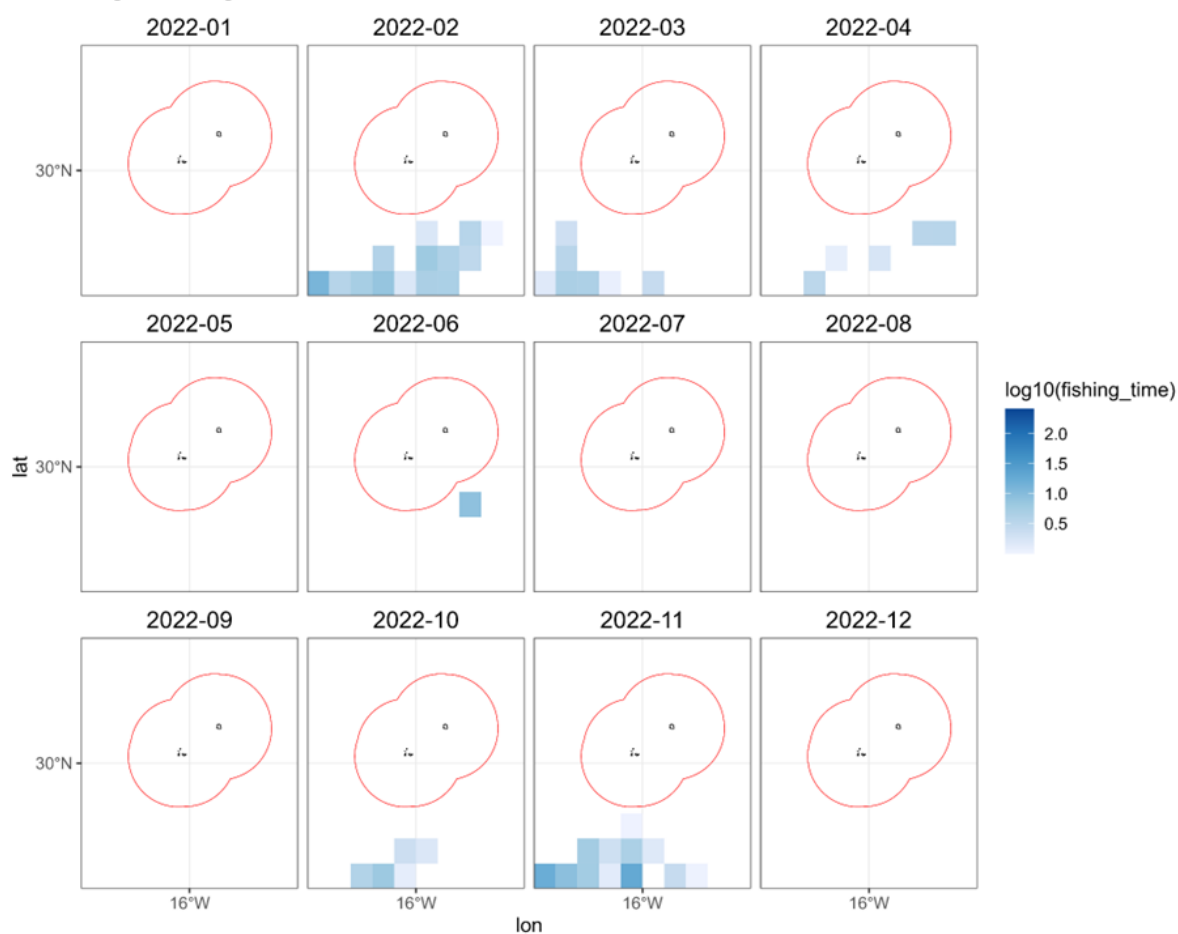


Figure 14. Total fishing effort by month for 2022. Based on data from GFW, the red border indicates the Selvagens MPA extension in May 2022.

¹⁴ <https://ifcn.madeira.gov.pt/> (accessed 12 January 2024).

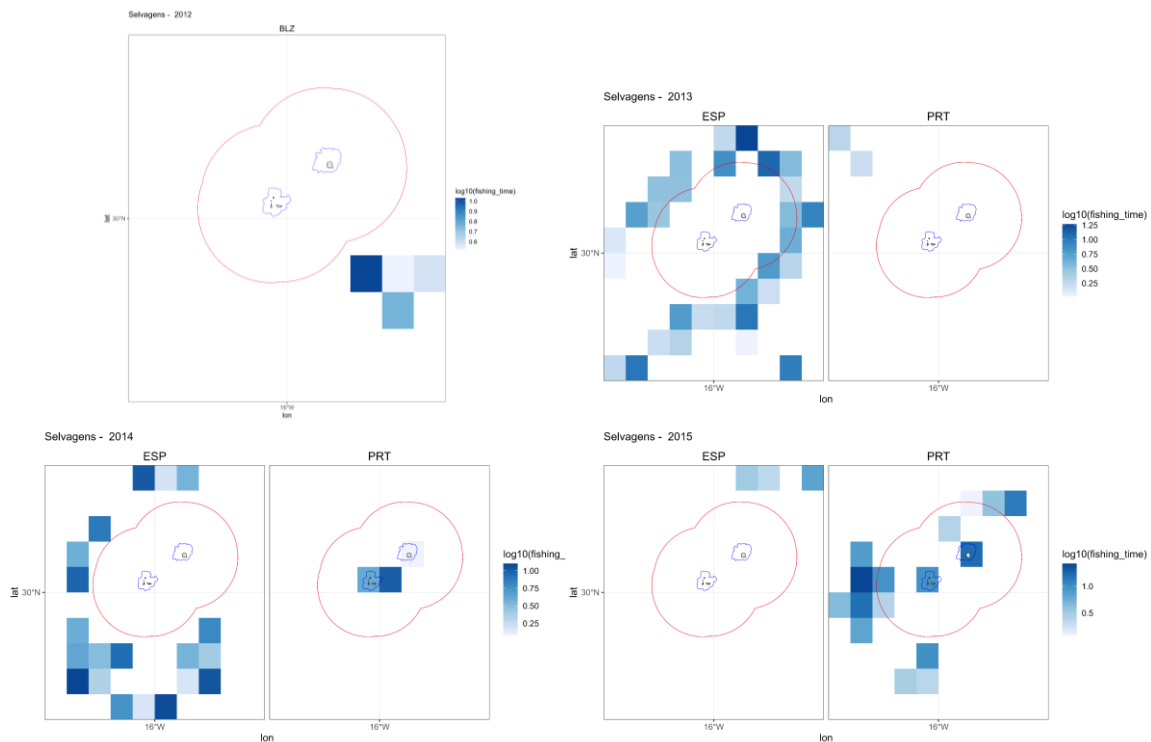


Figure 15. Total fishing effort by country for 2012, 2013, 2014 and 2015. Based on data from GFW. The blue border indicates the Selvagens MPA and the red border indicates the Selvagens MPA extension in May 2022.

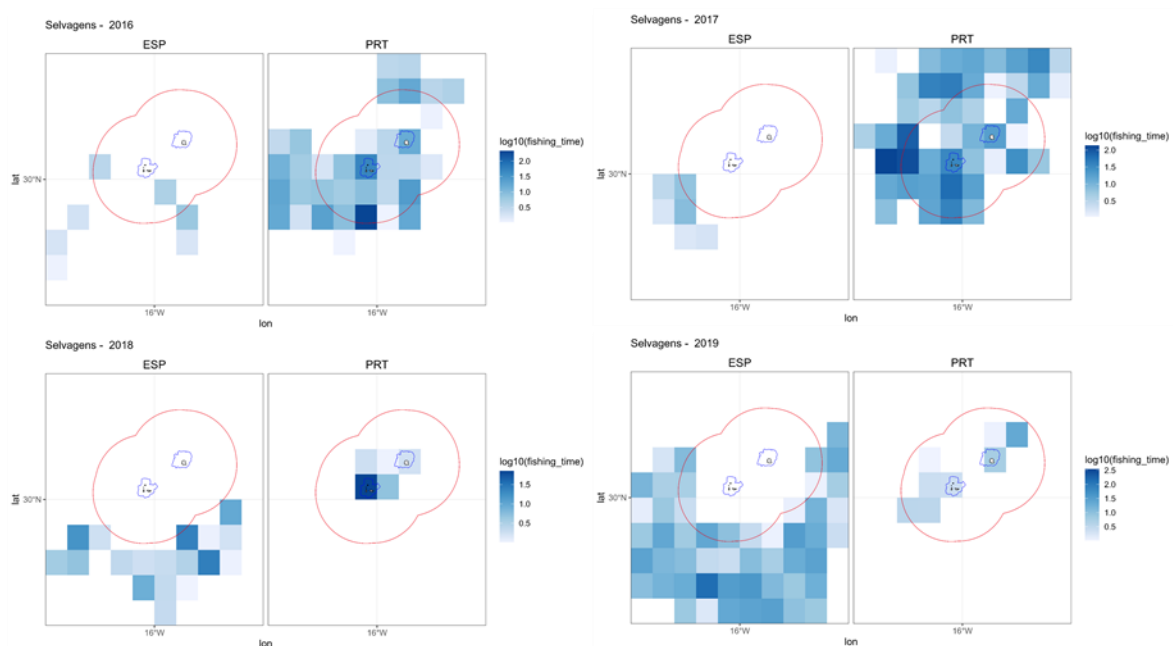


Figure 16. Total fishing effort by country for 2016, 2017, 2018 and 2019. Based on data from GFW. The blue border indicates the Selvagens MPA, and the red border indicates the extension in May 2022.

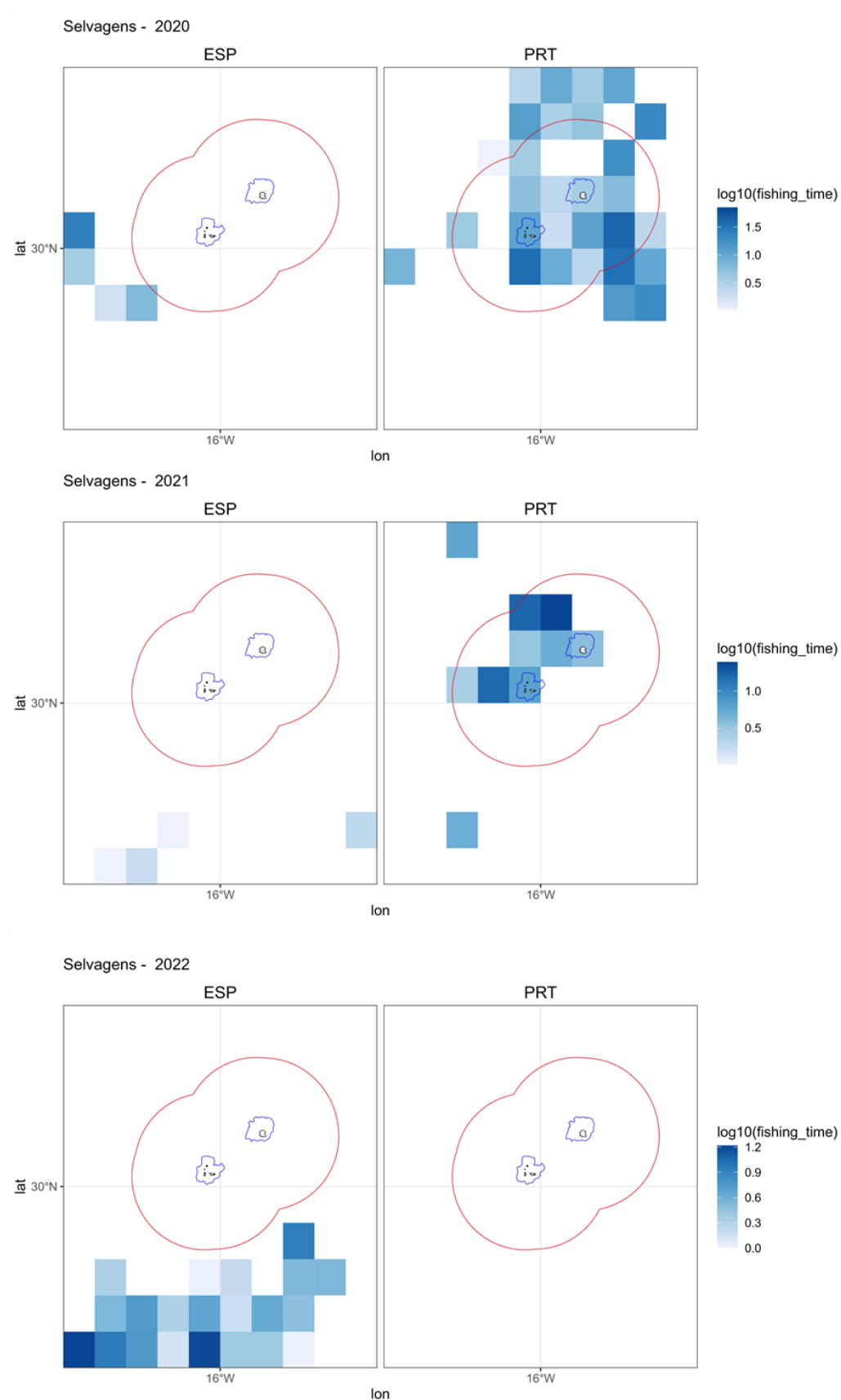


Figure 17. Total fishing effort, by country, for 2020, 2021 and 2022. Based on data from GFW. The blue border indicates the Selvagens MPA, and the red border indicates the extension in May 2022.

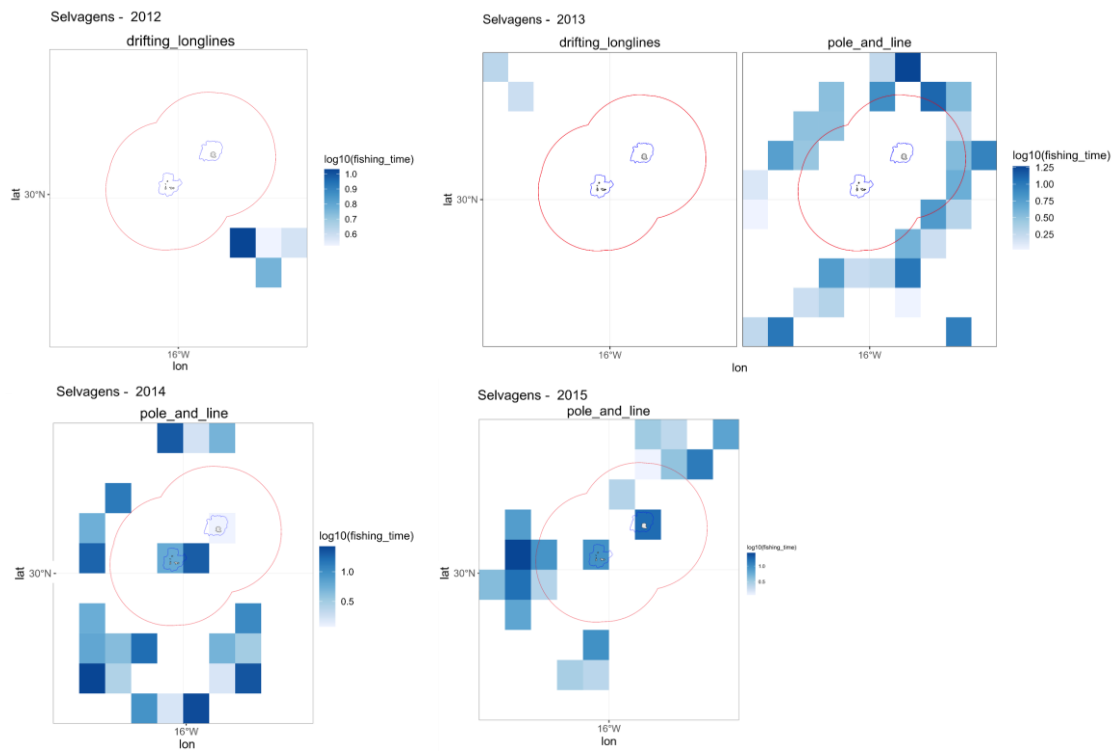


Figure 18. Total fishing effort, by fishing gear, for 2012, 2013, 2014 and 2015. Based on data from GFW. The blue border indicates the Selvagens MPA. The red border indicates the extension in May 2022.

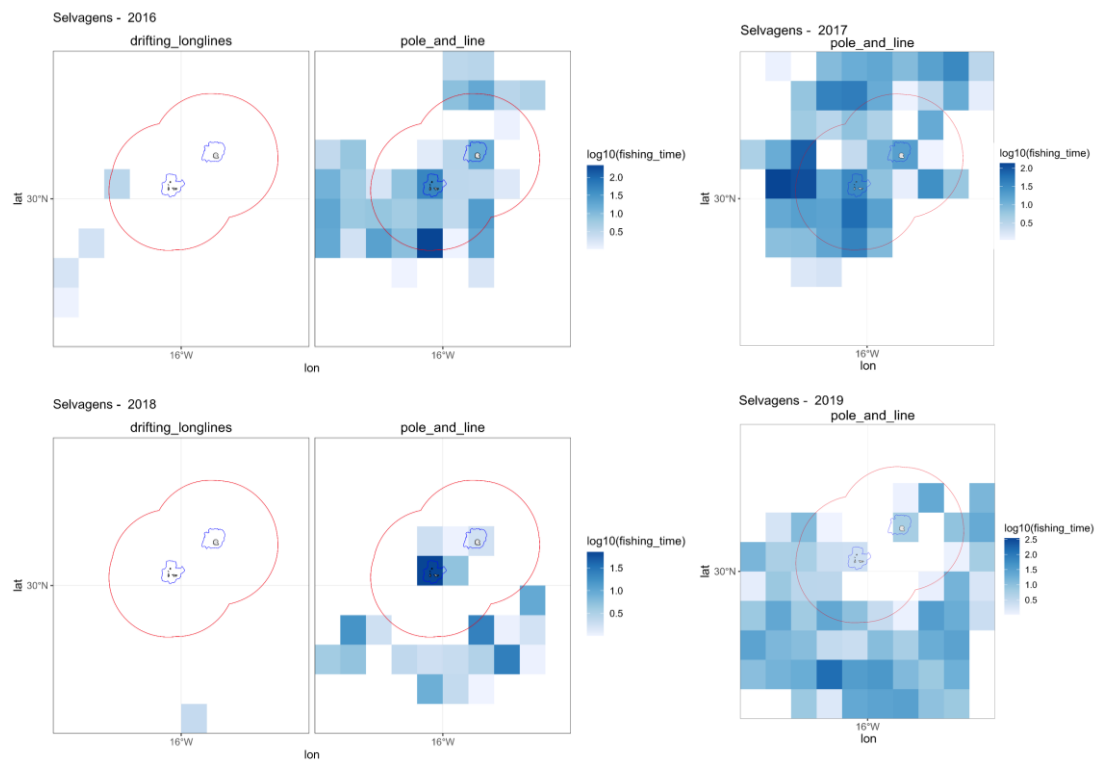


Figure 19. Total fishing effort, by fishing gear, for 2016, 2017, 2018 and 2019. Based on data from GFW. The blue border indicates the Selvagens MPA, and the red border indicates the extension in May 2022.

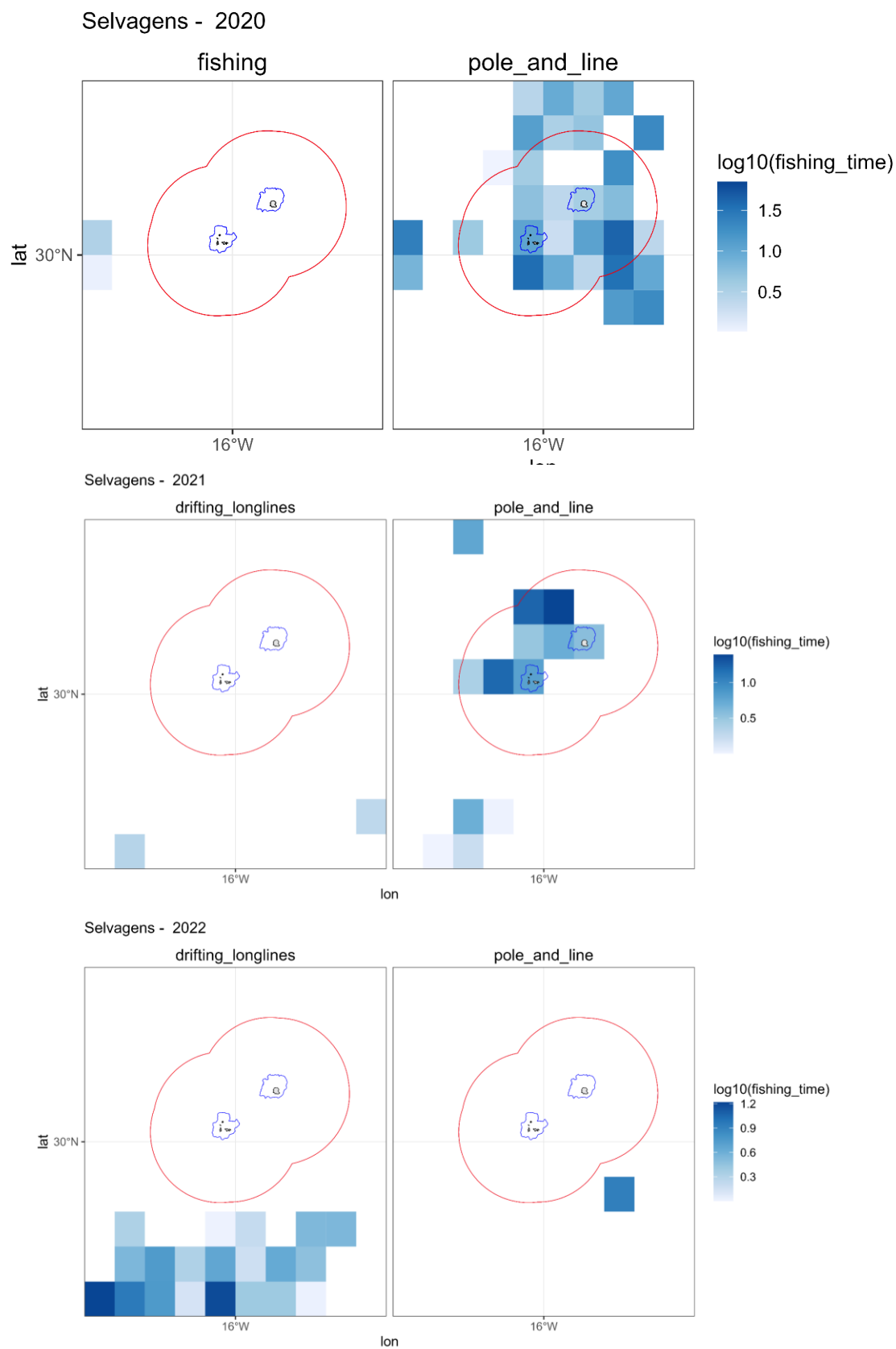


Figure 20. Total fishing effort, by fishing gear, for 2020, 2021 and 2022. Based on data from GFW. The blue border indicates the Selvagens MPA, and the red border indicates the extension in May 2022.

5.3. Description of habitats and bathymetry within the MPAs

5.3.1. Habitats

In understanding how important the habitats that are encapsulated within Madeiran MPAs, below we mapped the different habitats and provide a description of these found between different MPAs. Habitats cover both intertidal and subtidal habitats, but also a range of species of conservation importance within the EU (Table 3).

Table 3. Range of habitat types identified in Madeiran MPAs (Habitats Directive Annex I codes).

MPA	Description of habitat type
Desertas Natural reserve	• Large shallow inlets and bays (1160) • Vegetated sea cliffs with endemic flora of the Macaronesian coasts (1250) • Thermo-Mediterranean and pre-desert scrub (5330) • Sub-merged or partially submerged sea caves (8330)
Ponta de São Lourenço protected area	• Large shallow inlets and bays (1160); • Vegetated sea cliffs with endemic flora of the Macaronesian coasts (1250) • Thermo-Mediterranean and pre-desert scrub (5330) • Submerged or partially submerged sea caves (8330) • Endemic forests with <i>Juniperus</i> spp. (9560)
Ponta do Pargo MPA	• Sandbanks which are slightly covered by sea water all the time (1110) • Large shallow inlets and bays (1160) • Sub-merged or partially submerged sea caves (8330)
Porto Santo MPA network	• Large shallow inlets and bays (1160) • Vegetated sea cliffs with endemic flora of the Macaronesian coasts (1250) • Thermo-Mediterranean and pre-desert scrub (5330) • Submerged or partially submerged sea caves (8330) • Endemic forests with <i>Juniperus</i> spp. (9560)
Rocha do Navio Natural reserve	• Vegetated sea cliffs with endemic flora of the Macaronesian coasts (1250) • Thermo-Mediterranean and pre-desert scrub (5330) • Submerged or partially submerged sea caves (8330)
Selvagens MPA	• Sandbanks which are slightly covered by sea water all the time (1110) • Mudflats and sandflats not covered by seawater at low tide (1140) • Large shallow inlets and bays (1160) • Vegetated sea cliffs with endemic flora of the Macaronesian coasts (1250) • Thermo-Mediterranean and pre-desert scrub (5330)

Two articles mentioned *Cymodocea nodosa*, the only seagrass species reported in the Madeiran archipelago (Ribeiro et al., 2022; Schäfer et al., 2021). This species was first

reported in the 1970s, and until 2017, its observation was sparse (Ribeiro and Neves, 2020). It was thought to be extinct in this region (Schäfer et al., 2021). Schäfer et al. (2021) reported the discovery of *C. nodosa*, suggesting that a seagrass patch inside the Ponta de São Lourenço protected area was growing for the first time in decades, possibly expanding into a meadow (Figure 21). Ribeiro et al. (2022), also studying *C. nodosa*, provided a map of the sediment size of a part of the Ponta de São Lourenço protected area. It is important here to note that *C. nodosa* is included in several international protection lists, such as the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention), the EU Habitats Directive and the Bern Convention, which call for monitoring and protection of seagrass meadows (OSPAR, 2010). Additionally, *C. nodosa* is also encompassed in several national and regional regulations in the Mediterranean Sea (Orfanidis et al., 2010; Sghaier et al., 2017) and in the Canary Islands (Barberá et al., 2005; Herrera et al., 2014).

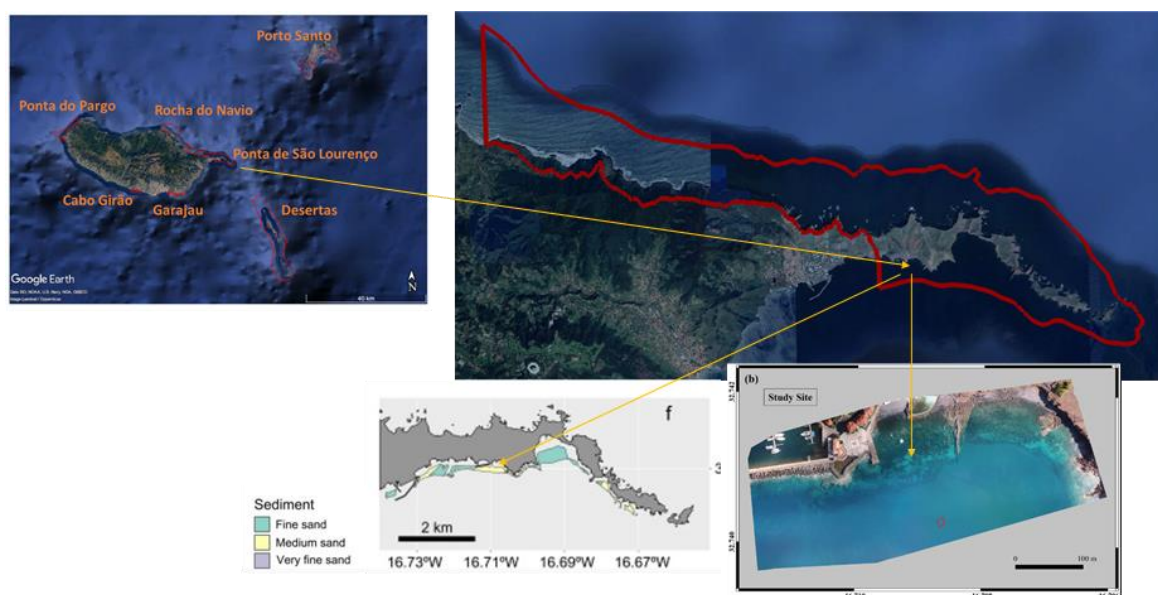


Figure 21. Indication of mapped *Cymodocea nodosa* inside Ponta de São Lourenço protected area and sediment characteristics in the exact location. Source: *Cymodocea nodosa*, Schäfer et al. (2021); sediment characteristics, Ribeiro et al., 2022.

Another study highlighted the presence of rhodoliths (Neves et al., 2021), some species of which are listed in Annex V of the Habitats Directive (Council Directive 92/43/EEC). The Directive requires Member States to ensure that the exploitation and harvesting of these species in the wild are consistent with maintaining their favourable conservation status. Neves et al. (2021) mapped rhodoliths beds in the Madeira archipelago, mentioning that these are more common and vast than previously assumed. This last article provided maps of the substrate type of the main rhodoliths beds in the Madeira and Porto Santo islands. In Porto Santo Island, Neves et al. (2021) mapped a location called 'Baixa das bicudas', which is located on the border of the no-take area inside the Porto Santo MPA network (Figure 22).

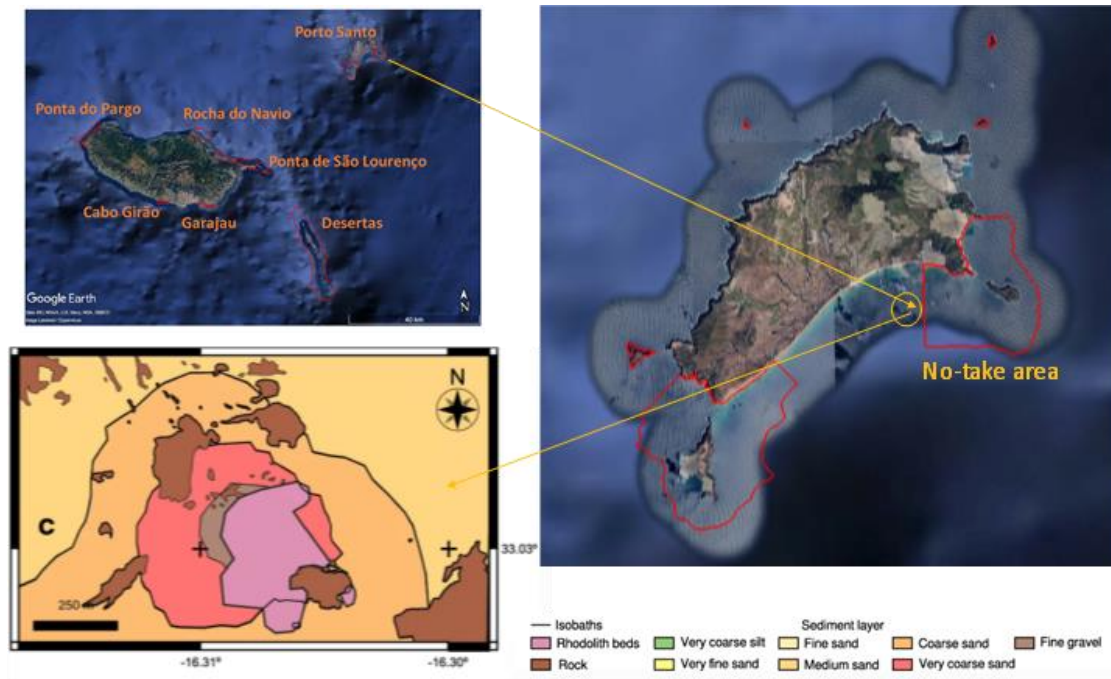


Figure 22. Indication of mapped rhodoliths beds and the sediment layer near the No-take area of the Porto Santo Marine Protected Areas network (source: Neves et al., 2021).

Finally, the last retained document gives insights into the green macroalga *Caulerpa ashmeadii*, discovered in Porto Santo for the first time (Ribeiro et al., 2022). This species was recorded on the Porto Santo MPA network, inside and outside the no-take area (Figure 23).

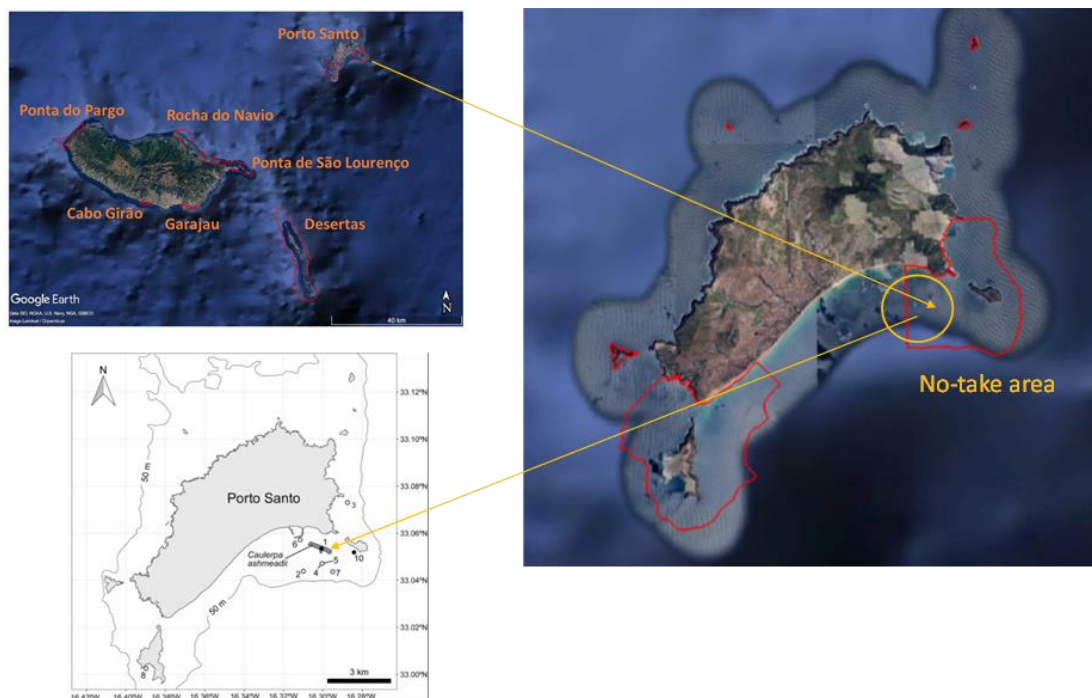


Figure 23. Location of the green macroalga *Caulerpa ashmeadii*, discovered in Porto Santo for the first time inside and outside the no-take area of the Porto Santo Marine protected areas network (Source: Ribeiro et al., 2022).

One phenomenon that is occurring in Madeira coastal habitats (up to 20–30 m depth) is a phase shift from macroalgae habitats to barren grounds as a result of intense sea-urchin grazing (Gizzi et al., 2020). The population of sea urchins has risen, possibly due to anthropogenic pressure (i.e. overfishing and consequent reduction in urchin predators and global warming). Since August 2022, the regional government has banned the fishing of two species, *Epinephelus marginatus* and *Bodianus scrofa*, by any fishing activity (commercial or recreational) (Resolution of the Regional Government Council nº604/2022). The latter species, *Bodianus scrofa*, is a natural urchin predator (Gizzi et al., 2021). Despite some recent studies in the last few years, Madeira still has gaps in data requirements regarding benthic habitats important for fisheries management, especially in small-scale fisheries (MRAG, 2022).

5.3.2. Bathymetry

Most MPAs have depths ranging from 50 to 100 metres, with the Selvagens MPA having the greatest depths (Figure 24). Coastal marine environments in Madeira Island typically consist of rocky shores characterised by small basalt-boulder beaches and basalt blocks, whilst subtidal substrates mainly comprise basalt that ends in a sand bottom around 20 m deep (Ribeiro, 2008; Monteiro et al., 2021).

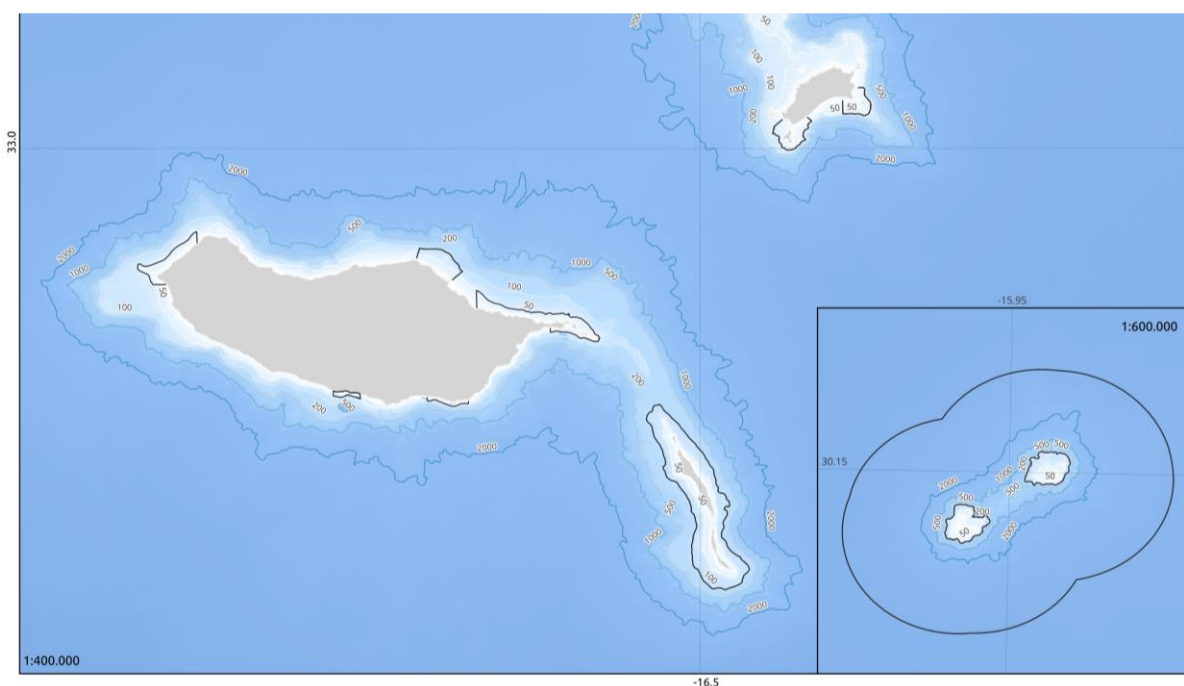


Figure 24. Map of the different MPAs in the Madeira archipelago, with the mean depth layer for the Madeira archipelago. MPAs delineated in black; mean depths sourced from the European Marine Observation and Data Network (EMODnet).

5.4. Regulations for MPAs in the Madeira archipelago

The regulations for MPAs in the Madeira archipelago are unclear: implementation, rules and classification are not fully defined, and the sharing of responsibilities between the region and the Portuguese State is not well established. As Horta e Costa (2017) noted, this might be particularly significant for MPAs beyond the territorial sea. Most of the MPAs

in Madeira are implemented by the government without prior studies or stakeholder consultation, with some exceptions (e.g. Ribeiro and Neves, 2020). However, stakeholder engagement is essential, especially in an insular context (Gutierrez et al., 2023).

None of the documents analysed mention a strategy for monitoring and enforcement of the MPAs. Horta e Costa (2017) noted this as a technical and operational gap (possibly legal) in MPAs across Portugal, which must also be addressed.

Monitoring is essential for verifying the effects of MPAs and making informed decisions in their management. However, monitoring in Portuguese MPAs is currently lacking in terms of legal and operational terms (Horta e Costa, 2017). To the best of our knowledge, in Madeira, there are only two monitoring programmes in place, one for Cabo Girão (e.g. GIRO, 2019) and another for Porto Santo to monitor an artificial ship reef placed inside an MPA (Neves et al., 2018). The effects of Madeiran MPAs are primarily dependent on the research conducted inside the MPAs. One example is a study undertaken by Friedlander et al. (2017), which verified that the Selvagens MPA was substantially different from other extractive MPAs with benthic communities with diverse algal assemblages and high fish biomass, including an abundance of large commercially-important species. This paper was the trigger that allowed the extension of that MPA and its classification as a no-take zone. There are cases of protected species (e.g. *Cymodocea nodosa* and Rhodoliths) that are not mentioned in the management plans or on the IFCN website¹⁵ (Ribeiro et al., 2022; Schäfer et al., 2021). Despite these gaps, MPAs in Madeira appear to be a positive example to follow and not just 'paper parks' in the Portuguese context: they are strongly protected by usage restrictions and have dedicated managers and surveillance (Horta e Costa, 2017).

6. DISCUSSION

MPAs play a key role in conserving and managing marine biodiversity and fisheries (e.g. Jones, 2002; Halpern et al., 2010; Vandeperre et al., 2011; Grorud-Colvert et al., 2021). This case study examined the importance of MPAs in potentially structuring fishing activities within the Madeira archipelago by focusing on two specific areas: Ponta do Pargo MPA and Selvagens MPA. The study explored their establishment, management strategies, and the complex relationship with fisheries, particularly in the context of the Madeira fishing fleet.

This case study identified three major pillars in Madeira MPAs' significance. One is related to the archipelago being a biodiversity hotspot, hosting unparalleled marine biodiversity, encompassing a diverse array of species (e.g. monk seal, cetaceans, sea turtles, endemic fish, limpets, seagrass, and rhodoliths) (Pires et al., 2008; Neves et al., 2021; Schäfer et al., 2021; Ribeiro et al., 2022). The second is related to protecting various marine habitats, from rocky reefs to deeper ecosystems. Finally, economic and social interests in MPA implementation and management seem to contribute to sustainable resource utilisation, benefiting both the environment and local communities. Over the years, the region has created a series of MPAs to ensure the sustainability of marine habitats and their relevant species while preserving traditional and small-scale fisheries (e.g. Ribeiro and Neves, 2020; Sousa et al., 2019b 2020b; Alves et al., 2022).

¹⁵ <https://ifcn.madeira.gov.pt/> (accessed 12 January 2024).

A range of different levels of restriction are associated with Madeiran MPAs, though with a focus on the use of no-take areas. Grorud-Colvert et al. (2021) developed an MPA policy framework that referred to the importance of protection against human activities in assuring the enhancement of marine biodiversity and promoting sustainable fisheries. In fact, Madeira has designated the largest no-take MPA within Europe, with the Selvagens MPA covering 2 677 km² with no-take restrictions. Although rigorous, such restrictions have likely been enacted over areas of little historical fishing activity, with such designation unlikely to have a substantial impact on regional fisheries activity and the long-term sustainability of such activities. However, the creation of this MPA provided Madeira (and Portugal) with support in assuring 30 % overall protection (and within this, 10 % full protection) as defined by the EU Biodiversity Strategy. Given the importance of fishing vessels operating near the Selvagens MPA before its extension, a zoning strategy (encapsulating both no-take zones and areas of restricted fishing activities) could have been implemented to support historical fishing activities. However, for this MPA, **full protection was perhaps ideal because of the area's isolation (uninhabited archipelago and long distance to Madeira Island) and the subsequent difficulty in monitoring and enforcement**. Isolation is an important conservation feature, and even before the area was fully protected and with active fisheries, this MPA was considered a pristine location (Friedlander et al., 2017).

As **fishing effort is centred predominantly in deep-water habitats within Madeira, designating coastal habitats as MPAs (most MPAs in Madeira are coastal habitats) is unlikely to have a substantial impact on the sustainability of the local fishing industry**. However, it will minimise any negative interactions with such operations. Although a range of species may utilise coastal habitats in their juvenile phase (for example, elasmobranchs and some teleost species), there is the potential for low levels of ecological connectivity between such habitats and those in which commercial species are exploited. MPA implementation that is more focused on protecting areas of high fishing effort (i.e. offshore deep-water habitats within Madeira) is likely to have a much more positive impact on fished species. The majority of the studies regarding interactions between fisheries and MPAs in the Madeira archipelago are related to limpet harvesting (e.g. Sousa et al., 2019d; 2020b). However, some works have explored the protection effect of the region MPAs (Ribeiro, 2008; Almada, 2017), including recent scientific campaigns in several MPAs in the Madeira archipelago (MARE-Madeira, unpublished data).

In support of preserving deep-water habitats within Madeira, outside MPAs, substantial fishing restrictions have been enacted on habitats > 200 m deep throughout the Madeira EEZ. Such restrictions may be expected to have the same positive effect on deep-water habitats as MPAs that have been enacted on such habitats, i.e. the reduction of fishing impacts that may reduce the quality or quantity of such habitats. In this respect, Madeira has made substantial positive steps towards protecting the main fishing habitats, supporting their deep-water fishery, and potentially ensuring the sustainability of such fishing activities. Recent evidence verified that Madeira's deep-water habitat is a hotspot of biological diversity (deep water kelps and coral gardens) that likely contributes to the provision of dispersal corridors of foundation species at the central Northeast Atlantic (Braga-Rodrigues et al., 2022).

Using different levels of zoning within MPAs (e.g. no-take and areas of varied restrictions) will be essential to conserve biodiversity and accommodate important local and regional fishing and other extraction activities (e.g. harvesting). In this respect, only three MPAs in Madeira (Garajau Partial Natural Reserve, Desertas Islands Nature Reserve and Porto Santo Marine Protected Area network) have been designed to accommodate such a balanced approach. Importantly, **preliminary records have identified a protection effect in the 'no take' zones in both the Desertas Island Nature Reserve and Porto**

Santo Marine Protected Area network from recent scientific campaigns (MARE-Madeira, unpublished data).

Despite a long history of substantial actions supporting fisheries management, within Madeira, there has been no simple way to obtain regional landing data. This is likely associated with the lack of coordination in developing tools and platforms to facilitate communication among scientists and managers within the study area. Moreover, there is no data sharing among institutions, despite (for example) the Regional Directorate of Fisheries collecting fisheries landings since the 1930s (e.g. Hermida and Delgado, 2016). Such lack of data availability is likely associated with many fisheries studies conducted in Madeira being undertaken by regional departments (e.g. Regional Directorate of Fisheries and Regional Directorate of the Sea). This situation contrasts with other Macaronesia locations (e.g. the Azores), where universities and research institutes carry out fisheries studies. Therefore, data is much more readily available (e.g. documents produced within the Department of Oceanography and Fisheries (DOP) of the Azores University).

In comparison to other Macaronesian islands, within Madeira, a top-down MPA implementation appears to be effective, and the most common method undertaken to designate and implement MPAs (albeit bottom-up initiatives have also been successfully implemented in the region (e.g. Garajau Partial Natural Reserve). Moreover, the implementation within such a top-down procedure was undertaken without comprehensive data being available prior to the establishment of the MPAs. Such **top-down management may be suitable in such cases where MPAs are implemented in areas of little historical fishing activity or in areas where there is likely little presence of local and regional stakeholders.**

The lack of fisheries data conditioned the present case study. The only way to verify whether MPA implementation affected the distribution of fisheries was by using GFW data. Since the MPAs in Madeira vary significantly in area (e.g. 2.55 km² to 2,677 km²; see Table 2 for more details), the results of the GFW analysis should be interpreted with caution because of the size of the grid cells compared to the size of the MPAs. However, for the present analysis, the resolution used was sufficient to capture the overall response of the fishing vessels to MPA implementation.

Because of temporal constraints on the MPA implementation in Madeira, only two MPAs were available: Ponta do Pargo (implemented in 2018) and Selvagens MPA (extended in May 2022). Ponta do Pargo MPA allows commercial fishing, recreational fishing and artisanal harvesting. Therefore, **despite implementing the MPA, there is no evidence to suggest that fishing activities have been reduced or modified.** Pole and line was the most used fishing gear in the area. This type of fishery is dependent on tuna migration behaviour, which is influenced by environmental variables (e.g. Gouveia and Mejuto, 2003). Pole and line was the most used in the Selvagens MPA, where the expansion of the MPA did not result in a redistribution of fishing activities (at least in 2022). In this MPA, fishing activities stopped within the border of the Selvagens MPA. The public interest associated with the MPA expansion, the tuna migratory routes (as mentioned before), or illegal fishing by blocking the AIS system (e.g. Kurekin et al., 2019) could explain the lack of vessels operating in the area.

The main concern while conducting this case study was the lack of data. For this study, several unsuccessful attempts were made to retrieve data. The present study also

detected a lack of research on fisheries outside the regional government entities. Tools/platforms to facilitate communication between scientists and managers would be required. The region might need to invest in joint efforts to plan, manage, and enforce MPAs with stakeholder involvement. Researchers, fishers, and the regional government should adopt strategies that allow adjustments based on new data, changing conditions, and stakeholder feedback.

7. CONCLUSIONS

This case study, which was conducted in the Madeira archipelago, provides insightful findings regarding the impact of MPAs on fisheries' dynamics. The study provides broader implications for the entire archipelago through a comprehensive approach involving a literature review, GFW (AIS) data analysis, and examination of MPA policies.

Analysis of the overall archipelago reveals that **MPAs are located in areas with historically low fishing activity, suggesting limited direct impact on regional fisheries**. This finding underscores the **importance of considering existing fishing patterns when designing MPA strategies**. At the metier level, outcomes vary between the two focal MPAs, Ponta do Pargo and Selvagens. In Ponta do Pargo MPA, where fishing is permitted, the study observes ambiguous changes in fishing patterns (pole and line, drifting longlines and purse-seine), making it challenging to attribute shifts in fishing dynamics (e.g. effort and gear usage) solely to MPA implementation. Conversely, the extension of Selvagens MPA did not significantly alter fishing activities, indicating potential stability or resilience in fishing practices within this area, where pole and line was the most relevant fishing gear.

The habitat description provided in this case study offers valuable insights into the ecological characteristics of the MPAs. It highlights the diversity of habitats present within these areas, including sensitive coastal (e.g. seagrass) and deep-water habitats (e.g. deepwater kelp and coral gardens).

Overall, while both MPAs aim to conserve marine biodiversity, their effectiveness and impact on fisheries differ. Ponta do Pargo MPA's allowance of fishing complicates interpretation, while Selvagens MPA's extension appears to have limited immediate effects on fishing activities. Within the region, there is a lack of fisheries data, little to no regular ecological monitoring of MPAs, and outdated management plans for most MPAs. Despite this, with its diverse MPAs, tailored MPA strategies, and preservation of traditional and small-scale fisheries, the region appears to be on track to fulfil the EU Biodiversity Strategy 2030 targets.

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Case Study Report

The Professor Luiz Saldanha Marine Park Portugal – The Iberian Cost



Mapping of marine protected areas and their associated fishing activities

Henrique Folhas, Raquel Pereira and Gonalo Carvalho

Sciaena

Portugal

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LIST OF ABBREVIATIONS

Term	Description
AIS	Automatic Identification System
ANOVA	Analysis of variance
AP	Average price
BA	Buffer Area
DGRM	Directorate-General of Natural Resources, Safety and Maritime Services
EEZ	Exclusive Economic Zone
EU	European Union
FPA	Fully Protected Area
MPA	Marine Protected Area
PMLS	Professor Luiz Saldanha Marine Park
PPA	Partially Protected Area
PNA	Arrábida Natural Park
VMS	Vessel Monitoring System

1. EXECUTIVE SUMMARY

The Professor Luiz Saldanha Marine Park (*Parque Marinho Professor Luiz Saldanha* (PMLS)) was created in 1998 and the entire area is part of the Arrábida Natural Park, which is included in the Natura 2000 network of protected areas. This marine protected area (MPA) was the first marine park to be created in continental Portugal, with the adoption of its management plan in 2005. The creation of the park has been publicly criticized during its creation and implementation, especially by commercial fishers, due to the long history of fishing in the PMLS area by vessels from Sesimbra and Setúbal.

Due to the small size of commercial vessels operating in and around the PMLS, existing automatic identification system and vessel monitoring system datasets could not be used to monitor and evaluate the reallocation of fishing activities before and after implementation of the MPA. Instead, the main aim of this report was to evaluate indirect measures, such as changes in landings and market prices, resulting from the implementation of the marine park.

The results showed that implementation of the PMLS did not have a negative impact on landings for several main commercial species, including white sea bream and octopus. Specifically, total landings from multi-gear fisheries from three main ports in Lisboa, Sesimbra and Sines, declined prior to implementation of PMLS and subsequently became more stable after its implementation. In addition, analysis of the gillnet fisheries in the same location also showed the total landings to increase after implementation of the PMLS in 2005. These observed trends were supported by an increase in total landings and fish price between 2000 and 2015, from the port of Sesimbra.

Through either effort displacement or adaptation to fisheries regulations, this study showed that local small-scale fisheries have remained viable, with increased landings and fish prices for many commercial species after the park was implemented. It was noted, however, that to fully quantify the impacts of PMLS, including the reallocation of fishing effort, additional monitoring and assessment of the environment, fisheries and resources is required.

2. BACKGROUND

Marine protected areas (MPAs) have been widely studied and several reviews have confirmed their global potential to restore marine resources and ecosystems (Halpern, 2003; Hamilton et al., 2010). However, in many cases there is a general lack of data prior to the implementation of MPAs, which is particularly evident in temperate regions. This lack of data limits the ability to assess the effectiveness of MPAs and interpreting positive responses. For example, observed differences in habitat quality between the reserve and other fished areas may predate implementation of the MPA.

Compounding this, where fisheries are a concern, commercial landings data for coastal artisanal fisheries worldwide are scarce and potentially biased (Batista et al., 2015). Further to this, critics argue that evidence of the economic benefits of MPAs is weak, particularly regarding fisheries. This continued opposition to MPAs for fisheries slows progress towards conservation targets and undermines the economic and ecological sustainability of the oceans (Costello, 2024).

To date, there are 74 legally designated MPAs throughout the entire Portuguese territory (Mainland Portugal $n = 8$; Azores $n = 60$; Madeira $n = 6$) (Cunha, 2021)¹⁶. Most of these are in territorial waters (up to 12 nautical miles), including the Professor Luiz Saldanha Marine Park, which is the focus for this study.

2.1. The Professor Luiz Saldanha Marine Park

The Professor Luiz Saldanha Marine Park (*Parque Marinho Professor Luiz Saldanha*, PMLS) was created in 1998 and the entire area is part of the Arrábida Natural Park (PNA), which is included in the Natura 2000 network of protected areas. The MPA covers approximately 5,300 hectares (53 km²), encompassing waters up to 100 metres in depth and spanning the coastline from Portinho da Arrábida (southwest of the city of Setúbal) to Cape Espichel and Praia da Foz (Figure 1).

This area is a biogeographic and oceanographic transition zone between warm and cold temperate waters and is also near the northern limit of the main northeast Atlantic upwelling events (Wooster et al. 1976), making this area an important hotspot of diversity (Henriques et al., 1999; Gonçalves et al., 2003).

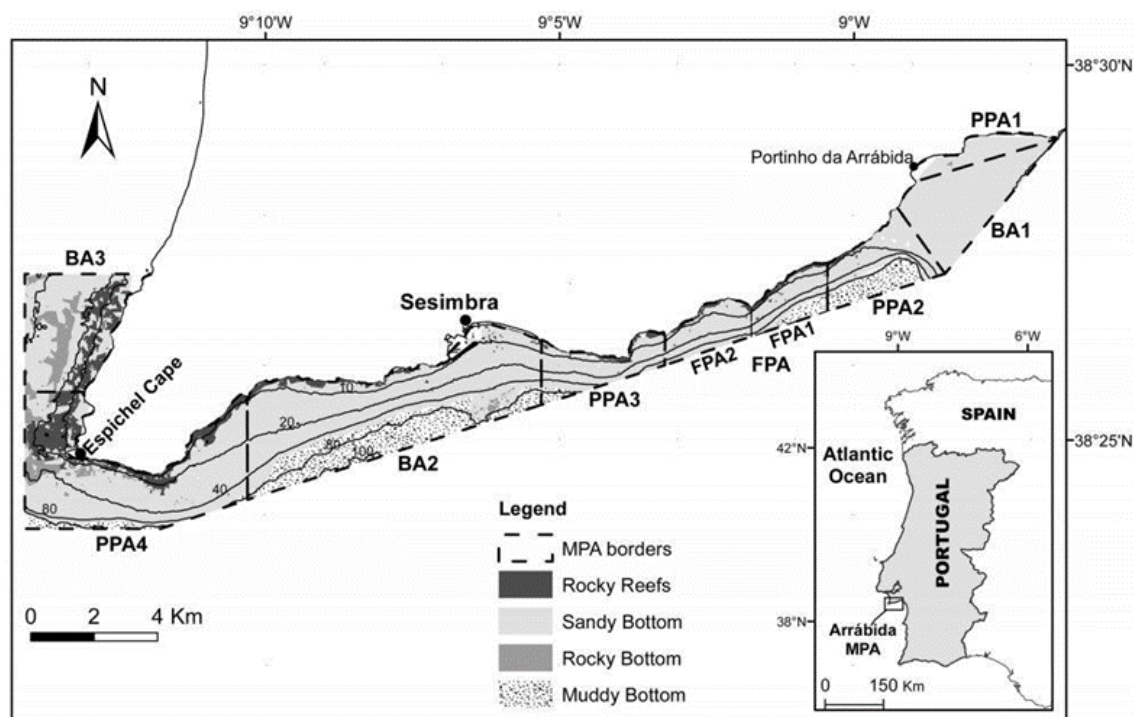


Figure 1. The PMLS location and zoning. FPA: fully protected area; PPA 1 to 4: partially protected areas; BA 1 to 3: buffer areas (Adapted from source - Horta e Costa et al., 2013b)

¹⁶ Portuguese MPAs were listed until 2020 from previous studies by Horta e Costa et al., (2019) and reviewed with up-to-date information collected online from national databases of MPAs, such as the Portuguese Maritime Spatial Planning Plan (PSOEM for Mainland, Madeira, and Continental Shelf subdivision, <https://www.psoem.pt/>; PSOEMA for Azores region, <https://oema.dram.azores.gov.pt/>)

Within the PMLS, the intertidal zone is steep, and various subtidal habitats are present, including rocky reefs, kelp forests, seagrass beds and sand banks. The rocky reefs create a complex variety of macro- and microhabitats which supports a high diversity of algae, invertebrates and fish, including important commercial species, such as the white sea bream (*Diplodus sargus*) and common octopus (*Octopus vulgaris*). Below 60 metres, corals and sponges dominate, forming complex habitats attracting a variety of communities. Overall, the PMLS is home to over 1,500 marine animal and plant species (Henriques et al., 1999; Horta e Costa et al., 2013b), making its natural richness unique at a European level.

2.2. PMLS management

The PMLS was the first marine park to be created in continental Portugal (Regulatory Decree No. 23/98 [1998]) along the coast of the Setúbal Peninsula. This legislation came 33 years after the first initiatives for the creation of an "underwater national park" in the Sesimbra area and 22 years after the creation of the PNA in the terrestrial part of the Arrábida mountains. Despite the designation of the marine park in 1998, the management plan was only approved in 2005 (Portuguese legislation, Council of Ministers Resolution 141/2005) (Figure 2).

The management objectives of the PMLS are to increase the marine biodiversity of the area (Horta e Costa et al., 2013b), promote the recovery of local seagrass (i.e. phanerogam) communities (Cunha et al., 2014), stimulate scientific research applied to the conservation, information, awareness raising and environmental education as well as promote ecotourism and traditional regional economic activities such as fishing with lines and hooks.

The PMLS is divided into eight zones with varying regulations, from a fully protected area where no activities are allowed, to partially protected areas which allow sustainable artisanal fishing, but prohibit large commercial fishing operations (see Figure 1). The plan divides the eight zones into three levels of protection: a fully protected area (4 km²), four partially protected areas (21 km²) and three buffer areas (28 km²). The fully protected area is a no-take, no-go area (with the exception of research, monitoring and education purposes). The partially protected areas allow local commercial fishing with traps and lines, but only beyond 200 m from shore, and do not allow extractive recreational activities (e.g. angling, spearfishing) to occur. In the buffer areas, fishing vessels less than 7 meters in length and recreational fishing are allowed.

The introduction of the management plan was delayed and contested at the public presentation stage. User conflicts with the management entity and among themselves were heated and remained so after implementation (Carneiro, 2011; Vasconcelos et al., 2012). In this respect, there has been a varied history in the development and implementation of the PMLS management plan, encompassing lengthy and contested discussions. These have been predominantly within the public presentation stage, but with conflicts between stakeholders and MPA management maintained after implementation (Carneiro, 2011; Vasconcelos et al., 2012). Opposition to the management plan has remained particularly strong among commercial fishers, due to the long history of fishing in the PMLS area by vessels from Sesimbra and Setúbal (Horta e Costa et al., 2013a,b). Further to this, the lack of perceived ecological effectiveness, good governance, and adequate management of social impacts within the PMLS seems to be contributing to a low level of support (Horta e Costa et al., 2013a,b).

The park's management plan was implemented with a transitional period for fisheries in which the different areas were gradually implemented during the first four years, with all

the buffer areas, partially protected area 1 and half of the fully protected areas (with partially protected area regulations) established in mid-2006, and all the partially protected areas and the second half of the fully protected areas (with partially protected area regulations) in mid-2007. The first half of the fully protected area started with regulations in mid-2008 and full implementation of the management plan was achieved in mid-2009 with the second half of the fully protected area (Horta e Costa et al., 2013b).

As in most Portuguese MPAs, the governance structure in the PMLS is top-down, as protection measures are defined by decision-makers, with a public consultation process that confers minor participation, decision power and knowledge recognition to local stakeholders (Vasconcelos et al., 2012). Such issues are caused by a lack of staff, budget, as well as poor enforcement and monitoring (Álvarez-Fernández et al., 2017), which undermines adequate management, including of social impacts (Gill et al., 2019). Despite this, recent initiatives are helping to minimize the opposition to the implementation of the park. For example, the MARGov project has focused on the construction of a Model of Collaborative Governance with local stakeholders. Within this, stakeholders have identified critical factors for the setting up of a table of sustainability to support collaborative management with the involvement of all stakeholders.

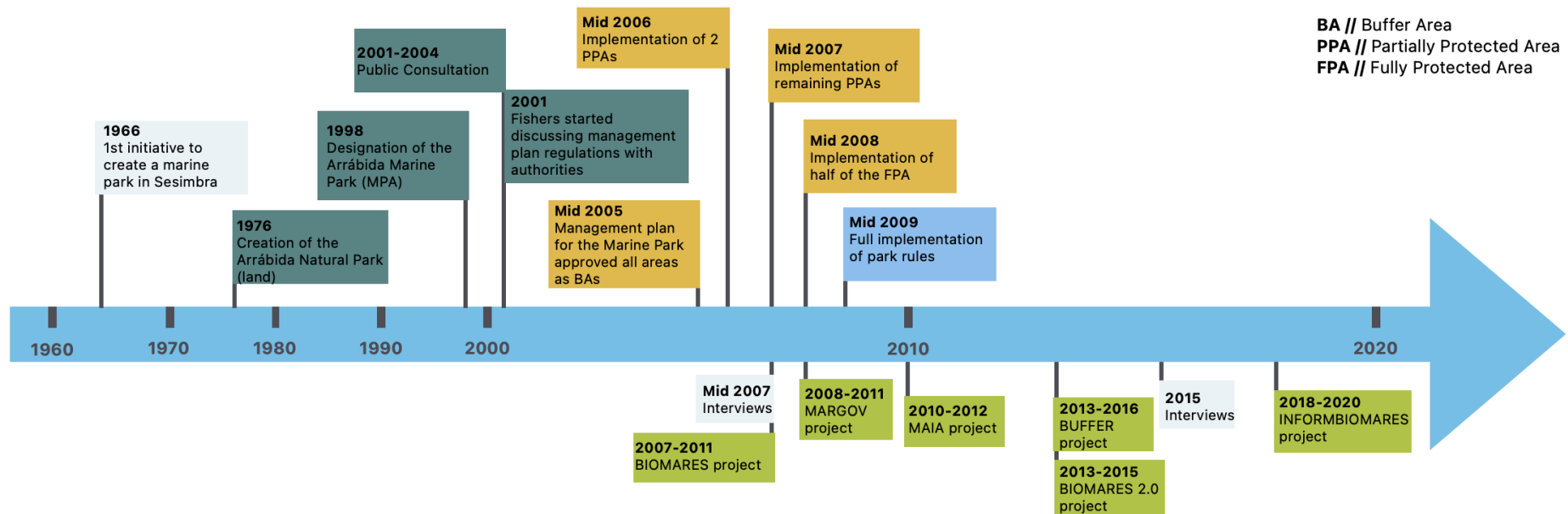


Figure 2. Chronological description of events related to the PMLS, including the creation of the Natural Park, the MPA designation, the implementation of the management plan and respective transitional period, and main projects occurring in the PMLS. Figure based on image from Batista (2007.)

2.3. Fisheries regulations

With respect to fisheries, the PMLS management plan defines limits and protection measures to various activities, namely, to protect the local small-scale fisheries (vessels smaller than 7 m in length) which have a high socio-economic importance in the area. However, the regulatory decree of 1998 prohibits certain fishing activities, including commercial diving for bivalves or other marine organisms, spearfishing, trawling, dredging, purse-seining and discarding fish.

Commercial fishing licences for the park are allocated only to fishers from Sesimbra, a town within the geographical limits of the MPA, and are renewed annually only if active. Permitted fishing gear is dependent on the area of the marine park. For example, gillnets, lines and traps are permitted in buffer areas but usage must be more than 400 m from the coast. Pots and handline jigging are permitted in partially protected areas, but usage must be more than 200 m from the coast. The black scabbard fish (*Aphanopus carbo*), despite being an important species for the vessels registered in the Sesimbra harbour, is caught by longline gear at depths greater than 200 m, and therefore the fishery takes place outside the marine park site (Bordalo-Machado and Figueiredo, 2009).

The most important fishing gears within the park include: traps used mainly to target octopus (Octopodidae); trammel nets which target species such as soles (*Solea senegalensis* and *Solea solea*) and cuttlefish (*Sepia officinalis*); longlines which target mostly Sparidae; and jigs which are used to catch cephalopod species (octopus, cuttlefish and squid (*Loligo vulgaris*)). Most of the vessels operating with longlines and jigs are less than 4 m in total length and are operated by a single fisher, while vessels that use traps and nets are typically 5-7 m in length and are usually operated by two fishers (Batista, 2007).

Fishers active in the PMLS changed the initial planned regulations by managing to have some gears (traps and jigs) in the partially protected areas and gill nets in the buffer areas permitted. By adding fishing gears to the partial protected areas, the size of the no-take area was reduced from about 25 km² to 4 km², indicating fishers' influence during the consultation process.

3. AIMS AND OBJECTIVES

The original objective of this study was to assess whether there has been spatial reallocation of fishing activities in response to the PMLS implementation and the impact of this potential reallocation on fisheries landings (i.e. socio-economic impacts). However, due to the limited data available, the main objective of this study is to evaluate the effects on local fishing activity resulting from the implementation of the PMLS, including possible spatial reallocation and changes in landings of two main commercial species (*Diplodus sargus* and *Octopus vulgaris*).

4. METHODOLOGY

4.1. Data types and sources

Despite the extensive literature available on the PMLS, there is a lack of quantitative survey data available to determine the local and regional socio-economic impact of the implementation of this MPA. Importantly, unlike other more recent MPAs within Portugal, there was no assessment of fisheries prior to implementation of the PMLS (e.g. interviews, surveys). Furthermore, investigation of temporal fishing activities around PMLS, especially spatial patterns, is hampered by the size of fishing vessels employed. Since the majority of vessels operating in and round the area are small-scale (i.e. less than 7 m in length) they are not required to have Automatic Identification System (AIS) or a Vessel Monitoring System (VMS) onboard. Thus, the original objective to assess the spatial relocation of fishing activities had to be reconsidered.

To determine the potential repercussions of the PMLS on local fisheries, public fisheries landing data from inside and adjacent to the MPA was analysed. Specifically, this included three data sets obtained from the Portuguese Directorate-General of Natural Resources, Safety and Maritime Services (DGRM).

4.1.1. Total annual landings from main ports around PMLS

Annual total landings (tonnes) were analysed from the three main ports around the PMLS (Lisboa, Sesimbra and Sines) between 1995 and 2014 for (i) multi-gear and (ii) gillnets. The main objective with using this larger dataset across a wider area was to determine whether total landings have changed in surrounding areas since the implementation of the PMLS in 2005.

4.1.2. Total annual landings and fish prices from Sesimbra

Trends in total annual landings from local fish auctions were analysed for the port of Sesimbra from two separate datasets. The first dataset includes a time series of total landings for the period 2000 – 2015, whereas the second includes more recent data for two important commercial species (white sea bream and common octopus) caught within the MPA during the period 2017 - 2019. In addition to the above total landings, the annual average price (EUR/Kg) was analysed for the same time periods and species.

4.1.3. Total landings and fish prices across three time periods for five species from Sesimbra

Monthly total landings (tonnes) and average price (EUR/Kg) data, obtained for the fishing port of Sesimbra, were analysed for five key species for three distinct time periods: 2000-2004; 2005; 2006-2015 (i.e. before/after implementation). These main commercial species caught using hooks, jigs and nets in the PMLS and surrounding area (Stratoudakis et al., 2015):

- Octopodidae
- Cuttlefish (*Sepia officinalis*)

- Sea bream (*Diplodus* spp.)
- Gilthead sea bream (*Sparus aurata*) and
- Sea bass (*Dicentrarchus* spp.)

Regarding Octopodidae, due to the lack of a continuous time series for each species included in this family, it was necessary to group together all the species of this family caught in or around the PMLS so that they could be compared with the other species analysed in this study.

While this dataset cannot evaluate the spatial reallocation of the fleet, it could help identify the potential impact of the marine protection on fisheries species.

4.2. Data analysis

Trends in annual datasets were analysed using linear regression models. Statistical analysis of data between different time periods for five commercial species was undertaken using a Kruskal-Wallis one-way analysis of variance (ANOVA), with Bonferroni's Method for Pairwise Multiple Comparisons test used if comparisons were statistically significant at 0.05.

5. RESULTS

5.1. Total annual landings from main ports around PMLS

Total landings data from multi-gear in the three main ports closest to PMLS (Lisboa, Sesimbra, Sines) for the period before and after implementation (1995 – 2014) are shown in Figure 3. The results show a declining trend in landings between 1995 to 2002, prior to implementation of the marine park. The decline then appeared to slightly increase before becoming more stable up to 2014. In comparison, landings data from gillnets show relatively stable catches throughout the period 1995 to 2014, with a slight decline (before implementation of the PMLS) between 1999 and 2003, but increasing gradually after 2005 (Figure 4).

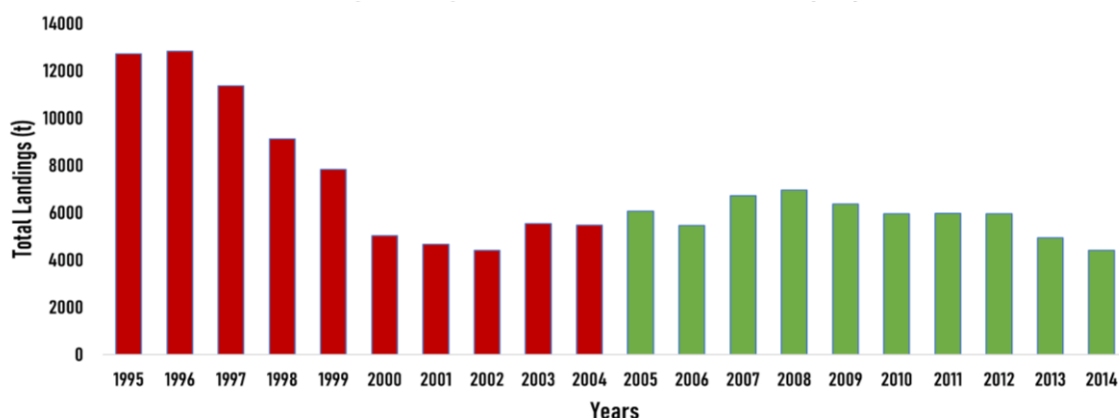


Figure 3. Total landings (tonnes) per year from multi-gear in the three ports closest to the PMLS (Lisboa, Sesimbra, Sines). Red bars = before implementation of PMLS; green bars = after implementation of the PMLS. (Data source: DGRM).

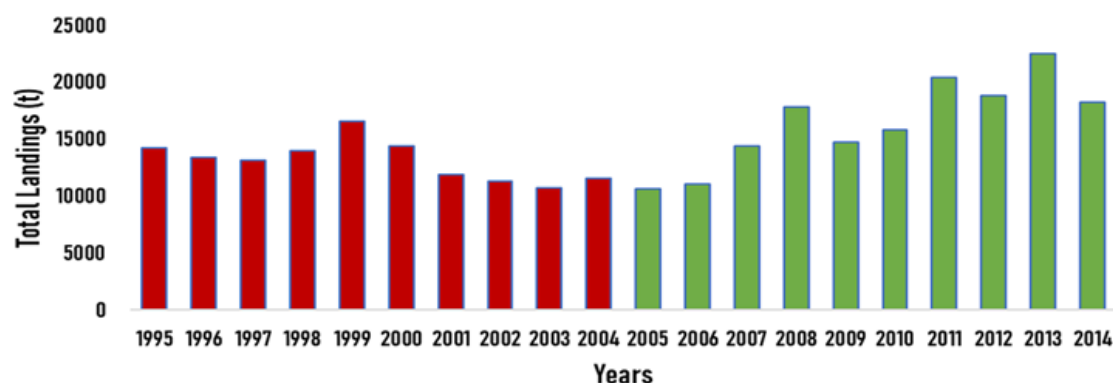


Figure 4. Total landings (tonnes) per year from gillnets in the three ports closest to the PMLS (Lisboa, Sesimbra, Sines). Red bars = before implementation of PMLS; green bars = after implementation of the PMLS. (Data source: DGRM).

5.2. Total annual landings and fish prices from Sesimbra

Total landings data from local fish auctions within Sesimbra have shown a steady increase in volume since 2000, especially after 2005 (Figure 5). More recent data from 2017-2019 further support this trend, with annual total landings around 30,000 tonnes in 2019 (Figure 6). In addition to landings data, the average fish price has shown a similar increasing trend between 2000 and 2015 (Figure 7).

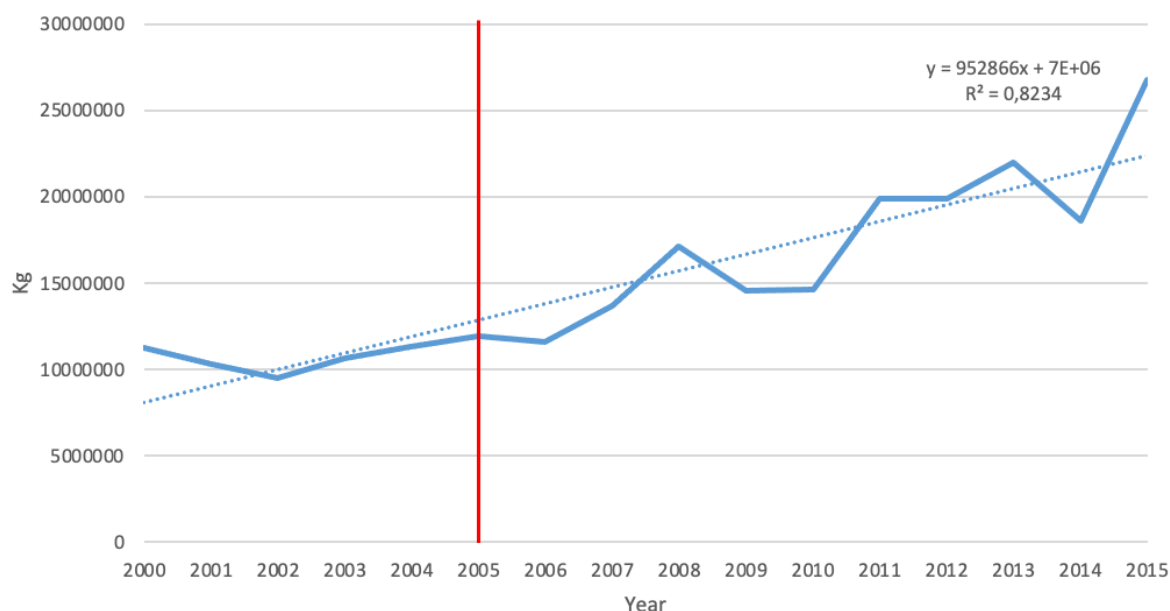


Figure 5. Total landed in Sesimbra fishing port from 2000 to 2015. The vertical red line represents the implementation of PMLS. (Data source: DGRM).

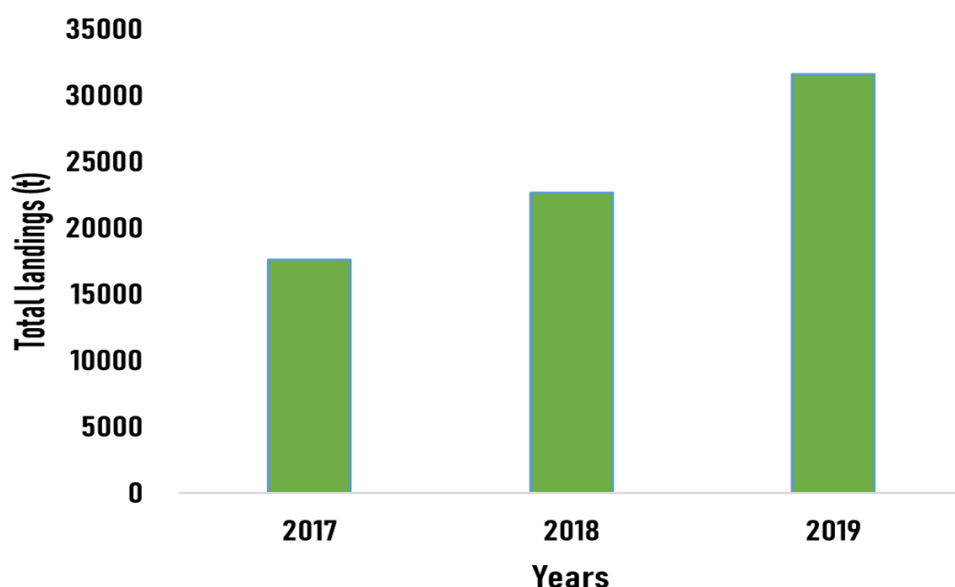


Figure 6. Total landings in the port of Sesimbra in 2017, 2018 and 2019. (Data source: DGRM).

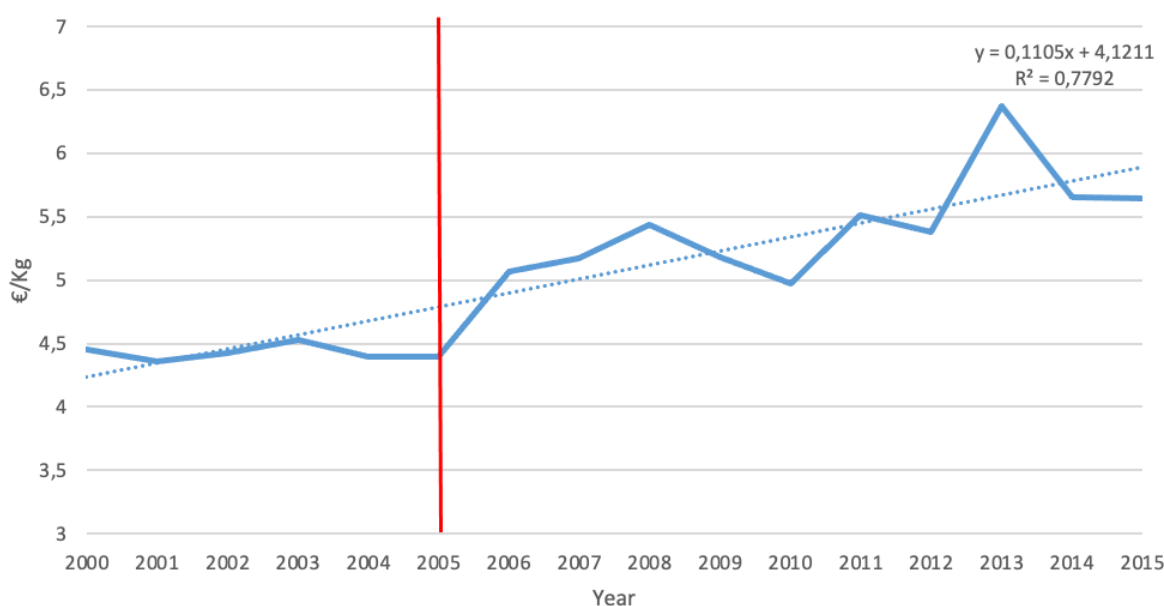


Figure 7. Average price of landings in Sesimbra fishing Port from 2000 to 2015. The vertical red line represents the implementation of PMLS. (Data source: DGRM).

Further detailed analysis of the average price of two main commercial species (white sea bream and common octopus) landed at Sesimbra fishing port between 2017 and 2019 is shown in the table below (Table 1).

The results show the average price of both species has increased between 2017 and 2018, while white sea bream continued to increase in 2019. Even though octopus did not show a similar trend in 2019, the minimum price at fish auction was higher than in 2017 (Table 1). Such differences in price over this period may be associated with the loss of fishing grounds caused by the MPA designation, the spatial competition between trap fisheries and

a decrease in the use of nets. The latter are important in catching octopus within traditional fishing grounds, which are now only permitted in the buffer areas.

Table 1. Average price (EUR/Kg), including minimum, maximum, average and standard deviation (SD). (Data source: DGRM).

Species	Year	Min	Max	Average (SD)
White sea bream (<i>Diplodus sargus</i>)	2017	1.24	3.92	2.48 (+/- 0.88)
	2018	0.69	3.52	2.61 (+/- 0.80)
	2019	0.98	4.30	3.15 (+/- 0.96)
Common octopus (<i>Octopus vulgaris</i>)	2017	6.02	9.71	7.99 (+/- 1.02)
	2018	6.9	10.57	8.98 (+/- 1.43)
	2019	6.51	8.40	7.47 (+/- 0.45)

5.3. Total landings and fish prices across three time periods for five species from Sesimbra

Five main commercial species / species groups were chosen for further detailed analysis of landings: Octopodidae, cuttlefish, sea bream, gilthead sea bream and sea bass. Cephalopods (Octopodidae and cuttlefish) are part of the traditional recipes of the southernmost region of Lisbon, while Sparidae and sea bass are important species for Portuguese small-scale fisheries. These species are caught by the small artisanal fleet, which are an important fleet in the area of the marine park, with species caught using hooks, jigs and nets (Stratoudakis et al., 2015).

5.3.1. Octopodidae

These are important catches for the Portuguese small-scale fleet, since they comprise one of the most landed catches and have higher value at fish auction for the Portuguese fishing sector (Pilar-Fonseca et al., 2014). They are caught mainly with pots and traps, one of the few fishing gears permitted in the marine park. Although landings of Octopodidae have not changed significantly over time (Table 2), average price has increased significantly, with the most substantial difference between implementation of the PMLS and after this time period (Table 3). Octopodidae significantly differed between contexts on the Kruskal Wallis test only for AP ($H(2) = 6,947$, $p < 0.05$).

5.3.2. Cuttlefish (*Sepia officinalis*)

Mean total landings and the average price of cuttlefish have differed significantly between years on the Kruskal Wallis test (total landings: ($H(2) = 7,029$ $p < 0.05$); average price ($H(2) = 14,144$ $p < 0.001$)). Total landings decreased significantly between pre-implementation and implementation and after implementation of the PMLS (Table 2). In comparison, the average price of cuttlefish has increased consistently over time (Table 3).

5.3.3. Sea bream (*Diplodus* spp.)

There were significant differences in total landings and average price for sea bream across years on the Kruskal Wallis test (total landings: $H(2) = 53,333$ $p < 0.001$; average price $H(2) = 6,875$ $p < 0.05$). Changes in landings were due to substantial reductions, with landings slightly decreasing at implementation, but showing a four-fold reduction after implementation (Table 2). Such reductions in landings have resulted in a moderate increase in the average price of this fish at market (Table 3).

5.3.4. Gilthead sea bream (*Sparus aurata*)

There were significant differences in total landings and average price for gilthead sea bream across years on the Kruskal Wallis test (total landings: $H(2) = 11,269$ $p < 0.005$; average price $H(2) = 6,870$ $p < 0.05$). Such differences in total landings were associated with a slight drop after PMLS implementation, but a near doubling of landings following implementation (Table 2). With regards to average price, gilthead sea bream has shown a moderate increase over time (Table 3).

5.3.5. Sea bass (*Dicentrarchus* spp.)

There were significant differences in total landings and average price for sea bass across years on the Kruskal Wallis test (total landings: $H(2) = 10,149$ $p < 0.01$; average price: AP $H(2) = 8,704$ $p < 0.05$). Such changes in landings were associated with a moderate increase at PMLS implementation, but a doubling of landings following implementation (Table 2). Average price has shown fluctuations, with a decrease at implementation, and a relatively moderate increase after implementation (Table 3).

Table 2. Average (SD) total landings (kg) for five main commercial species across three time periods associated with the implementation of the PMLS (before, 2000–2004; implementation, 2005; after, 2006–2015). (Data source: DGRM). The letters a and b refer to comparison of landings between respective time periods that are significantly different at 0.05 level using the Kruskal Wallis test (absence of letter signifies not significant).

Species	2000–2004	2005	2006–2015
1. Octopodidae	31,271.9 (31,097.7)	22,994.8 (26,140.2)	29,605.4 (32,824.8)
2. Cuttlefish (<i>Sepia officinalis</i>)	6,758.4 ^{ab} (5,250.6)	4,373.3 ^a (3,797.8)	4,860.7 ^b (4,816.2)
3. Sea bream (<i>Diplodus</i> spp.)	292.8 ^{ab} (227.6)	202.7 ^a (275.8)	57.9 ^b (98.4)
4. Gilthead sea bream (<i>Sparus aurata</i>)	855.6 ^a (1,109.1)	819.8 ^b (1,034.6)	1,772.5 ^{ab} (2,966.7)
5. Sea bass (<i>Dicentrarchus</i> spp.)	369.4 ^a (552.5)	443.9 ^b (803.4)	797.5 ^{ab} (1,331.6)

Table 3. Average (SD) price (EUR/kg) for five main commercial species across three time periods associated with the implementation of the PMLS (before, 2000–2004; implementation, 2005; after, 2006–2015). (Data source: DGRM). The letters a and b refer to comparison of landings between respective time periods that are significantly different at 0.05 level using the Kruskal Wallis test (absence of letter signifies not significant).

Species	2000–2004	2005	2006–2015
1. Octopodidae	3.15 (2.0)	2.79 ^a (1.7)	3.22 ^a (2.2)
2. Cuttlefish (<i>Sepia officinalis</i>)	5.88 ^{ab} (1.8)	6.93 ^a (2.2)	7.3 ^b (2.4)
3. Sea bream (<i>Diplodus</i> spp.)	8.58 ^{ab} (1.4)	9.34 ^a (1.9)	9.31 ^b (1.8)
4. Gilthead sea bream (<i>Sparus aurata</i>)	10.1 ^a (2.5)	11.18 (2.2)	11.4 ^a (1.8)
5. Sea bass (<i>Dicentrarchus</i> spp.)	10.03 ^a (3.0)	11.55 ^a (3.7)	10.92 (3.8)

6. DISCUSSION

Despite initial low support by fishers, our results help demonstrate implementation of the PMLS has not led to overall negative impacts on local fishing activities, including total landings or fish price. Moreover, implementation of management and fisheries regulations has had immediate benefits, such as the exclusion of spearfishing, which can significantly impact high trophic level species such as large sparids, seabass and octopus (Horta e Costa et al., 2013a).

Our results show that total landings from multi-gear fisheries from three main ports in Lisboa, Sesimbra and Sines, declined prior to implementation of PMLS and subsequently became more stable after its implementation. In addition, analysis of total landings from gillnet fisheries in the same location also showed these to increase after implementation of the PMLS in 2005. Further to this, these trends were supported by an increase in total landings and fish price between 2000 and 2015, from the port of Sesimbra. Data from the same port for two of the most important commercial species (*Diplodus sargus* and *Octopus vulgaris*) showed local fishers also benefit from increasing fish prices, especially between 2017 and 2019.

Detailed analysis of monthly total landings and fish price was given for five main commercial species landed in Sesimbra. The results showed that landings of sea bass and gilthead sea bream had significantly increased since the implementation of PMLS, whereas total landings of cuttlefish and Octopodidae had remained relatively stable. However, a significant decline in total landings of sea bream was reported between the time when PMLS was implemented in 2005 and subsequent years up to and including 2015. Without an index of stock abundance, such as catch-per-unit-effort, it is unclear for example, whether the observed decline in total landings of sea bream in Sesimbra is due to reduced fishing effort and/or changes in targeting behaviour of fishers, or whether the population itself has declined.

Lédée et al., (2012) showed that individual fishers showed distinct strategies, with some operating in a broader area whereas others kept preferred territories, some of them being adjacent to a no-take area. Further to this, an in-depth study by Horta e Costa et al., (2013a) on the spatial relocation of the fishers operating in the limits of the marine park, reported that they had shown adaptations to multiple protection measures in an MPA, and therefore changes in total landings may be due to changes in targeting behaviour. While these observations are consistent with previous work on catches from the MPA (Horta e Costa et al., 2013b) it highlights the importance of monitoring and evaluation of species within and around the marine park. The positive effect of MPA protection on population abundance of commercially exploited species may take several years to show (i.e. reserve effect) (Claudet et al., 2008).

Since implementation of the PMLS in 2005, views and opinions of some stakeholders towards the marine park have recently changed. For example, the fishing community of Sesimbra, which was the most impacted by the implementation of the PMLS, also had the biggest opposition to its implementation. However, this community is now a strong advocate for the PMLS (Vasconcelos et al., 2012). Such a change in stance regarding the PMLS is likely to be associated with the increase in total revenue associated with landings of both white seabream and octopus. In addition, vessels fishing inside the MPA did not suffer a decline in revenues as many expected due to the implementation of fisheries restrictions inside the park (Batista et al., 2011; Lester and Halpern, 2008). While these findings are encouraging, it is not clear if the current levels of resource exploitation are sustainable and if other factors may be influencing the landing trends. The lack of a monitoring program for the park doesn't allow the observation and understanding of other variables, such as the fishing effort.

7. CONCLUSIONS

Due to the small size of commercial vessels (less than 7 m total length) operating in and around the PMLS, existing AIS and VMS datasets could not be used to monitor the temporal-spatial distribution of fleets, and therefore the reallocation of fishing activities before and after implementation of the MPA. Instead, the main aim of the case study was to evaluate indirect measures, such as changes in local fishing activity resulting from the implementation of the PMLS in 2005. Specifically, this includes landings and market prices of five commercially important species groups. This analysis was also used to look for evidence of spatial relocation of fishing activities.

From the data analysed and coherence between the results, it can be concluded that the **implementation of the PMLS didn't cause any negative impacts on total landings and average prices of main commercially important species** captured in and around the marine park. Further to this, the fishing activities that were most directly impacted by the creation of the park did not collapse, as was feared by the local community. Either by effort displacement or by adapting to the restrictions created, there are clear indicators that the port of Sesimbra and the small-scale fleet continued to prosper and even improved their landings and revenues after the creation of the park.

The PMLS can be seen as a good example of MPA implementation within the EU, despite the existing limitations in terms of enforcement and management. This study highlighted the potential benefits for both fishers and fisheries following protection of local resources. It is noted, however, that such indirect measures used to evaluate changes in fishing activities, such as relocation of fishing effort, do not provide a full understanding of the impacts of management intervention. Instead, there is a need within the PMLS to conduct

ecological and fisheries monitoring and assessment, to quantify the impact on biodiversity but also on human activities that occur within this park.

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Case Study Report

La Palma Island and La Graciosa Island Canary Islands. Spain – Macaronesia



Mapping of marine protected areas and their associated fishing activities

José Carlos Mendoza, Jesús Manuel Falcón, Bertín García, Noemi Dionis, Alba Jurado-Ruzafa, Antonio Punzón & Pablo Martín-Sosa

***Spanish Institute of Oceanography
Spanish National Research Council***

Spain

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LIST OF ABBREVIATIONS

Term	Description
AIS	Automated Identification System
CANZEC	CANarias – Zonas de Especial Conservación
EMODnet	European Marine Observation and Data Network
EUNIS	European Nature Information System
GFW	Global Fishing Watch
GIS	Geographic Information Systems
Ha	Hectare(s)
IEO-CSIC	Instituto Español de Oceanografía – Consejo Superior de Investigaciones Científicas
MAPA	Ministry of Agriculture, Fisheries and Food
MPA	Marine Protected Area
NOAA	National Oceanic and Atmospheric Administration
RESMARCAN – IEO-CSIC	REServas MARinas CANarias
SAC	Special Area of Conservation
TEPESCO	Análisis de la interacción Actividad Pesquera y el Ecosistema: caracterización, impacto, vulnerabilidad y áreas marinas

1. EXECUTIVE SUMMARY

The main aim of this work was to examine whether there have been changes in the spatial strategy of fishing activities in response to the implementation of two marine protected areas (MPAs) in the Canary Islands (Spain), namely the La Graciosa Island MPA and the La Palma Island MPA. Based on available fishing effort quantitative and semi-quantitative data, spatial fishing strategies and potential changes in habitat use were explored inside and outside each MPA during the period 2008–2023.

Although this work was unable to utilise data from before designation of both MPAs, our analysis has shown that the fishing restrictions within each MPA results in the abandonment or disuse of small-scale fishing gears (e.g. traps and trammel nets) inside the MPA restricted zone. In this respect, implementation of both MPAs has led to long-term displacement of restricted fishing gears to the external boundary of each MPA, including fishing being undertaken further offshore.

Although there has likely been displacement of fishing activities away from each MPA following implementation, this has resulted in the overlap of fishing grounds and strategies between the artisanal and industrial fleet in the external MPA boundaries. This scenario leaves the small-scale fishery at a disadvantage in terms of catches that could be managed with longline specific zonation rules (buffer areas) in the surroundings of La Graciosa Island MPA. Importantly, such displacement (especially where coastal habitats are limited, e.g. La Graciosa Island MPA) may mean that artisanal fishing boats may navigate longer distances to reach suitable fishing grounds. Such changes to fishing behaviour are not likely to have been accounted for in the implementation of the MPAs and has likely led to increased fuel costs and lower returns on catches for this fleet. Further work will be needed to understand the full repercussions of such displacement on the economic viability of different fishing métiers.

Regarding the potential changes in habitat use, the EMODnet habitats data is not sufficient in terms of specificity and resolution. There is need to incorporate the still ongoing cartography studies in the IEO-CSIC of deeper habitats where longline professional fishing activity and small-scale trapping activity have potential impacts on the seabed.

The impact of an MPA was found to be related to fishing gears restrictions. The likelihood of change in fishing activities would likely be low following implementation of an MPA covering a very small area, as was the case for the La Palma Island MPA, where fishermen who decided to continue using restricted fishing gears (traps and trammel nets) continued to fish near the MPA boundaries. Otherwise, a large area such as the La Graciosa Island MPA meant a high likelihood of change in fishing activities. Only a few larger small-scale fishing boats displaced their traditional fishing activity (traps) to waters outside the MPA boundaries.

Despite relatively low total coverage, the designation and implementation of both MPAs have affected fishing activities within both La Graciosa and La Palma. For example, the no-take zone of La Graciosa Island MPA represents only 1.7 % of the total protected area (i.e. of the MPA) in the Chinijo archipelago, while within the La Palma Island MPA only one relatively small no-take area was designated, despite the original plans indicating that two were planned. These relatively small no-take areas show that protection within the Canary Islands is a viable management measure for fishing activities. Future updates of both MPAs could entail increases in the coverage of no-take areas, which will only further increase the utility of both MPAs in supporting regional fishing sustainability.

2. BACKGROUND

The Canary Islands are a volcanic archipelago of Spain (Macaronesia), located to the northwest of the African mainland around 100 km west of the Saharan coast. The Canarian archipelago covers 7 492 km² and is situated between 27°39'N–29°24'N and 13°25'W–18°10'W (Figure 1). The archipelago comprises eight major islands (La Graciosa, Lanzarote, Fuerteventura, Gran Canaria, Tenerife, La Gomera, La Palma and El Hierro) and several islets. In general, a large portion of coastline is characterised by a sublittoral narrow island platform, which limits the primary productivity. Accordingly, the waters around the Canary Islands are defined as oligotrophic systems (Aristegui et al., 2001; Bode et al., 2001). The islands' (eastern subtropical Atlantic (Barton et al. 1998)) marine assemblages consist of a combination of tropical, subtropical and temperate species, with highly diversified vulnerable marine ecosystems harbouring low populations of coastal species with complex interactions (Riera et al., 2014).

Coastal fisheries resources have been overexploited in the Canary Islands (Riera and Delgado, 2019; González et al., 2020). Prior to the implementation of the Canary Islands MPAs, overfishing of top predatory fishes had resulted in ecological disequilibrium with consequences for the structure and function of the entire benthic communities (Clemente et al., 2009; 2010). For example, the proliferation of the grazing activity of sea urchins caused by overfishing of natural fish predators had resulted in extensive barren grounds (Hernández et al., 2008).

In 2024, the Canarian archipelago has a total of 27 marine protected areas (MPAs). Three of these MPAs have been designated under national legislation (Bacallado et al., 1989; Revenga, 2015): La Graciosa Island and islets to the north of Lanzarote (hereafter, **La Graciosa Island MPA**) in Lanzarote (1995), La Restinga MPA in El Hierro (1996) and La Palma Island MPA in La Palma (hereafter, **La Palma Island MPA**) (2010) (Figure 1). These three MPAs are designed as 'marine reserves with fishing interest' for the conservation of coastal fisheries resources. However, vulnerable species and sensitive habitats were not considered when the MPAs were established. Despite high political involvement in decision-making regarding the zoning of the La Palma Island MPA has been critical, for example, in reverting the barren grounds into a healthier and more equilibrated ecosystem (Sangil et al., 2012). If approved by the Spanish government, three proposed MPAs under national legislation (in La Gomera, Tenerife and Gran Canaria) will improve the Canarian MPA network (De La Cruz-Modino and Pascual-Fernandez, 2010). However, these proposals are currently at a standstill period.

2.1. La Graciosa Island and islets to the north of Lanzarote

The island of La Graciosa was uninhabited until the late 19th century, when it was settled by inhabitants of Lanzarote. However, it was used for the development of harvesting activities such as fishing and shellfish gleaning. From a very small population in 1875, when a fish-salting factory was established, the population on La Graciosa increased substantially, reaching approximately 80 houses in 1943. During this period, the population was immersed in a socio-economic crisis, with families surviving mainly by marketing fish and shellfish to Lanzarote residents. Between 1940 and 1980, important social stratification processes brought change to the island, including the arrival of the first tourists. Political families enriched through deep-sea fishing (i.e. targeting tuna) invested in La Graciosa facilities to develop the tourism sector (Socorro, 2000; Socorro and Socorro, 2004). Other poorer families, unable to undertake fishing, emigrated to Lanzarote. However, from 1980 onwards, intensive professional fishing using traps and longlines, as well as high levels of

recreational fishing associated with increasing tourism development led to deterioration of the local marine habitats surrounding La Graciosa. Traditionally, La Graciosa has been exploited by foreign vessels targeting, among other species, the European hake (*Merluccius merluccius*), though such commercial fishing decreased substantially during the 1990s, and no large commercial vessels now operate in this area.

In 1986, prompted by local ecologists and the regional government, the Chinijo archipelago (comprising La Graciosa and the northern islets) within the Famara cliff in Lanzarote was declared a Natural Park by the Canarian Government (BOC-1986-058-001). This designation was enacted to reduce the likelihood of development of large-scale tourism within La Graciosa. In practice, however, the protection only encompasses seabird-monitoring surveys and has resulted in constant conflict between the local population and the Canarian Government regarding the management of the Natural Park. While the Natural Park designation has halted attempts to build large hotel complexes, small-scale private initiatives have been able to develop with impunity despite the law prohibiting, for example, the transfer of property to outsiders. The declaration of the Natural Park neither encompassed conserving fisheries resources, as these are under the control of the Spanish Ministry of Agriculture, Fisheries and Food (*Ministerio de Agricultura, Pesca y Alimentación* [MAPA]). For this reason, added to the permanent conflicts over the management of the Natural Park, La Graciosa was declared an MPA.

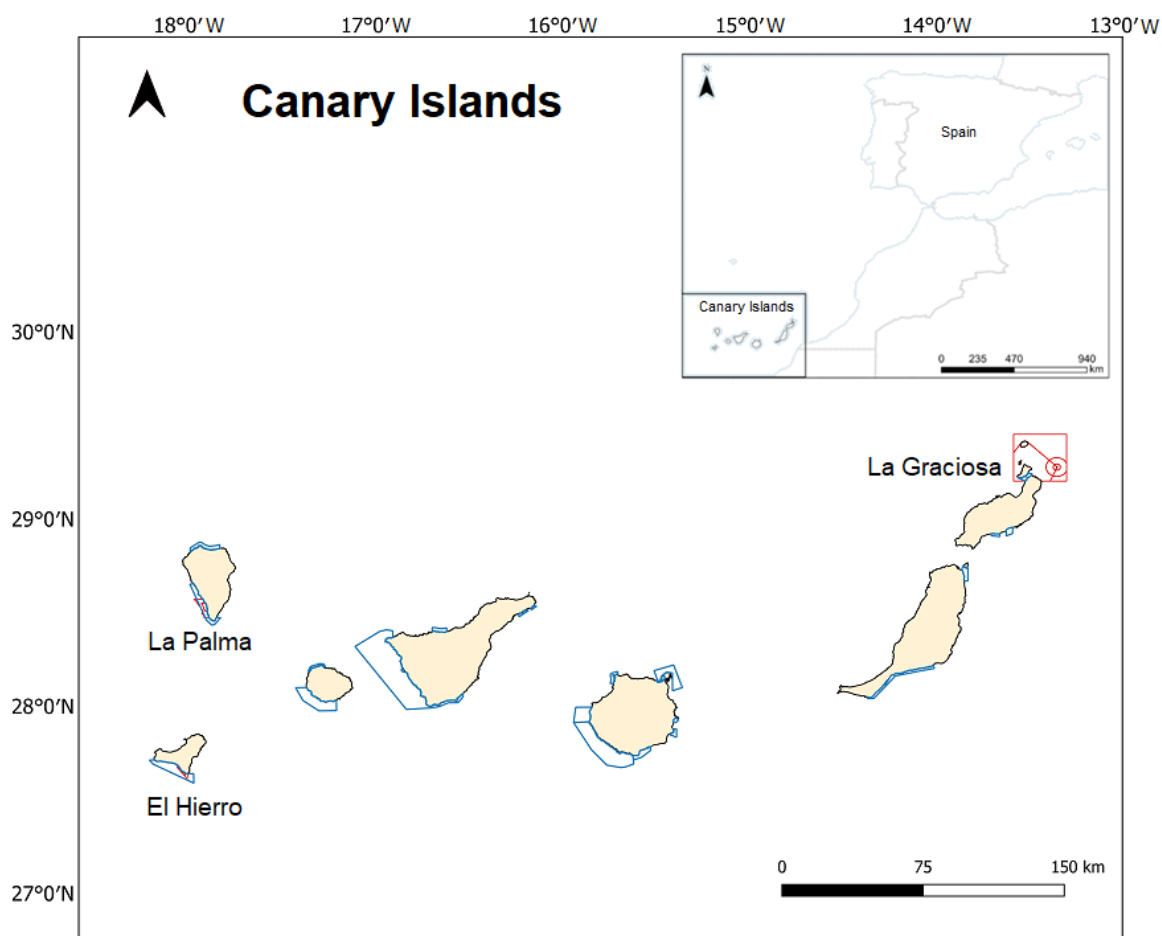


Figure 1. Location of the Canarian archipelago and its MPAs: SACs (Natura 2000) in blue and Marine reserves with fishing interest (El Hierro, La Palma and La Graciosa) in red.

In 1995, the La Graciosa Island MPA was designated (BOE-A-1995-13433) to protect the sea around the Chinijo archipelago to satisfy demands of the local fishing sector (Figure 2). The main concern of local fishermen was to ensure protection from foreign fishers, especially for spearfishing and trammel net fishing activities. Despite this, local fishers led by the president of the cofradia of La Graciosa (i.e. fishing association) did not support the MPA designation because of the potential loss of traditional fishing gears (e.g. traps, longlines, trammel nets). However, local fishers had little input into the design process for the La Graciosa Island MPA (Chuenpagdee et al., 2013; De la Cruz-Modino and Pascual-Fernández, 2013). The lack of formal discussions and clear information, and the lack of empowerment to negotiate regulations, made the fishing association hesitant to support the MPA designation.

Such lack of support from local fishermen has resulted in changes to the initial designation plans. In the initial studies examining MPAs within the La Graciosa Island MPA (e.g. Bacallado et al., 1989) the development of two no-take areas was suggested around the Chinijo archipelago (around El Roque del Este and El Roque del Oeste). However, the MAPA decrees only support a single no-take area (Roque del Este) and there was no scientific evidence to show such a small area would generate conservation benefits throughout the region.

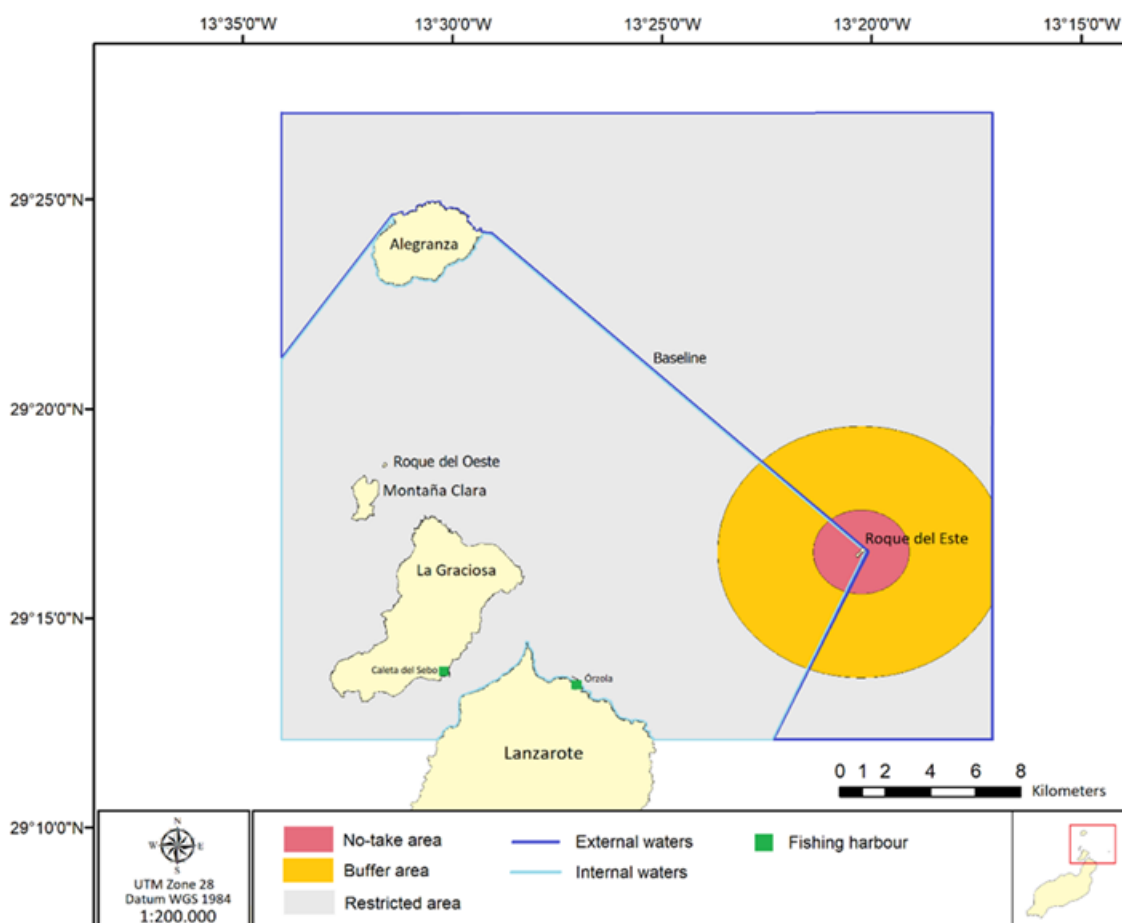


Figure 2. Location of La Graciosa and the northern islets (Chinijo archipelago) of Lanzarote La Graciosa MPA zonation: No-take area in red, Buffer area in orange and fisheries restricted area in grey.

2.1.1. La Graciosa Island MPA

The La Graciosa Island MPA covers an area of 70 439 hectares (ha) around La Graciosa and the northern islets of Lanzarote and is located between the meridians 13°34'W and 13°17'W and the parallels 29°27'N and 29°12'N. The La Graciosa Island MPA comprises external waters beyond baselines (delimitation of Spanish jurisdictional waters), which are under the remit of MAPA, as well as internal waters inside baselines, which are under the remit of the Comunidad Autónoma de las Islas Canarias (Canary Islands Regional Government). The limits of the protected area include the bathymetric range from 0 to 1,000 m deep, although the vast majority of the sea bottom within the La Graciosa Island MPA is up to 200 m deep because of the rocky insular shelf to the north of Lanzarote.

The La Graciosa Island MPA has three zones: a no-take area, a buffer area and a fisheries restricted area (Figure 2). The **no-take area** covers approximately 1,200 ha and encompasses the 'Roque del Este', located on the eastern side of the MPA. In this zone, only authorised scientific activities are allowed. The no-take area represents only 1.7 % of the MPA, which is far less than initially proposed (Bacallado et al., 1989).

The **buffer area**, covering a one-mile radius from the no-take area boundary, only allows tuna fishing to be undertaken by pole and line. The following fishing activities are prohibited: recreational fishing in waters beyond baselines, except trolling for migratory pelagic species at no less than two miles from the no-take area, and this only with prior authorisation from the Functional Area of Agriculture and Fisheries of Las Palmas (*Area Funcional de Agricultura y Pesca de Las Palmas*); recreational fishing at less than 500 m from Montaña Clara and Roque del Oeste and at less than two miles from Roque del Este (no-take area); professional fishing by fishermen who do not usually fish in the area and who are not in the relevant census for vessels authorised to fish in the marine reserve; shellfish gathering (removal of limpets, sea snails (*Phorcus* spp.) and any other invertebrates of gastronomic interest).

In the **restricted area** (which covers the majority of the La Graciosa Island MPA), a range of activities are allowed. Exploitation by authorised local fishermen (La Graciosa and Lanzarote) using traditional fishing gears and recreational fishermen only with trolling (external and internal waters) and hook and line (internal waters) are allowed. For example, in 2022, 50 artisanal fishing boats from four fishing harbours were authorised to fish in the restricted zone of the MPA according to the official fishing fleet census in La Graciosa (BOE-A-202221656). In addition, professional fishing was permitted, including fishing with bait (angling and line) and traditional fishing for salema (*Sarpa salpa*) and migratory pelagic species. Recreational fishing uses hook and line and is undertaken from the shore on the coast of La Graciosa and the coast of Lanzarote between Punta del Palo and to the far west of Playa de la Cantería. Recreational fishing with hook and line onboard a vessel (not land bound), except 500 m from Montaña Clara and Roque del Oeste coastline, is allowed. Recreational fishing is not allowed in internal waters without prior authorisation from the Viceconsejería de Pesca del Gobierno de Canarias. Trolling of migratory pelagic species in water beyond baselines (external waters) is allowed.

The La Graciosa Island MPA is managed by technical administrative staff and local government representatives, but the team suffers from a lack of capacity and support given the large size of the MPA and the complexity of fishing activities allowed. Consequent limitations in surveillance of the La Graciosa Island MPA have resulted in a certain degree of poaching (Socorro and Socorro, 2004). However, banning of less-selective fishing gears within the La Graciosa Island MPA (e.g. traps, longlines, trammel nets) has been an important regulation to enhance the sustainability of marine resources within the MPA.

Surveillance of the MPA is undertaken by fisheries rangers in collaboration with marine authorities. The fisheries rangers for the La Graciosa Island MPA patrol on a boat targeting the fishing activities inside the MPA. There are only two surveillance boats (one from MAPA and another from the Gobierno de Canarias). When the fisheries rangers detect infringements, the Spanish Guardia Civil is the competent authority to impose penalties.

2.2. La Palma Island

A preliminary coastal study in 1998, promoted by the collaborative agreement between the Government of the Canary Islands (Gobierno de Canarias: Consejería de Agricultura, Pesca y Alimentación) and the University of La Laguna, examined the viability of designating an MPA in La Palma Island (Barquín-Diez et al., 1999). This study explored the south coast of the Fuencaliente municipality, following on from previous work examining potential areas for protection proposed in Bacallado et al. (1989), utilising designation criteria stipulated in a National Oceanic and Atmospheric Administration (NOAA) MPA technical report (Plan Development Team, 1990) and criteria of the Nature Conservancy Council (1979). Accordingly, areas near the capital of La Palma (eastern region) or in the northern region were discarded; the capital, Santa Cruz de La Palma, is home to the island's only commercial port, while adverse marine conditions are found in the north region. Importantly, Barquín-Diez et al. (1999) not only assessed the viability of the areas in which to implement an MPA, but also how such an MPA should be zoned: two no-take areas on each side (W–E) of the island (four in total) with restricted areas, with buffer zones around each to support marine connectivity.

Pressure from local fishermen and the recreational fishing sector resulted in significant changes to the initial and scientifically-based zonation plan (Martín-García et al., 2015). Such changes included the reduction of the total surface area and the number of no-take areas by half compared to the proposal of the University of La Laguna (Barquín-Diez et al., 1999). The definitive plan proposed the protection of the south-western side of the island with an MPA. However, this design had no ecological or scientific basis.

The outcome of the stakeholder engagement and final decision to focus only on protecting the south-western side of La Palma, resulted in the La Palma Island Marine Reserve being established in 2001 by MAPA, under Ministerial Order of 18 July 2001 (BOE-185-2001) (Figure 3). This was implemented at the behest of the Government of the Canary Islands (Gobierno de Canarias) and the acceptance of the artisanal fisheries sector. The La Palma Island MPA was not requested by the fishing sector, although its initial lack of acceptance for this protected area has reduced.

Subsequent studies after the La Palma Island MPA implementation suggested that areas adjacent to the La Palma Island MPA had not benefited from ecological or fisheries spillover (Denny and Babcock, 2004; Ashworth and Ormond, 2005). Such impacts were deemed due to intense fishing pressure around the MPA boundaries (Sangil et al., 2013a, 2013b). Therefore, more marine protected zones such as the initial proposal in Barquín-Diez et al. (1999) and legal regulations are needed to complement marine environmental protection around the island (Pintado et al., 2006; Ayala, 2008).

2.2.1. La Palma Island MPA

The La Palma Island MPA covers an area of 3,455 ha, from 0 to 1,000 metres depth, encompassing 15 km of coastline. This MPA is divided into two different zones: a no-take zone and a restricted zone. The **no-take zone** is located in the centre, from the coastline to the 500 m isobath and from 28° 32.88' N (El Remo area) to 28° 30.19' N (Punta del Hombre area) north latitude (Figure 3). The **restricted zone** is located from 28° 34.10' N (El Charco Verde area) to 28° 28.24' N (Las Celdas area).

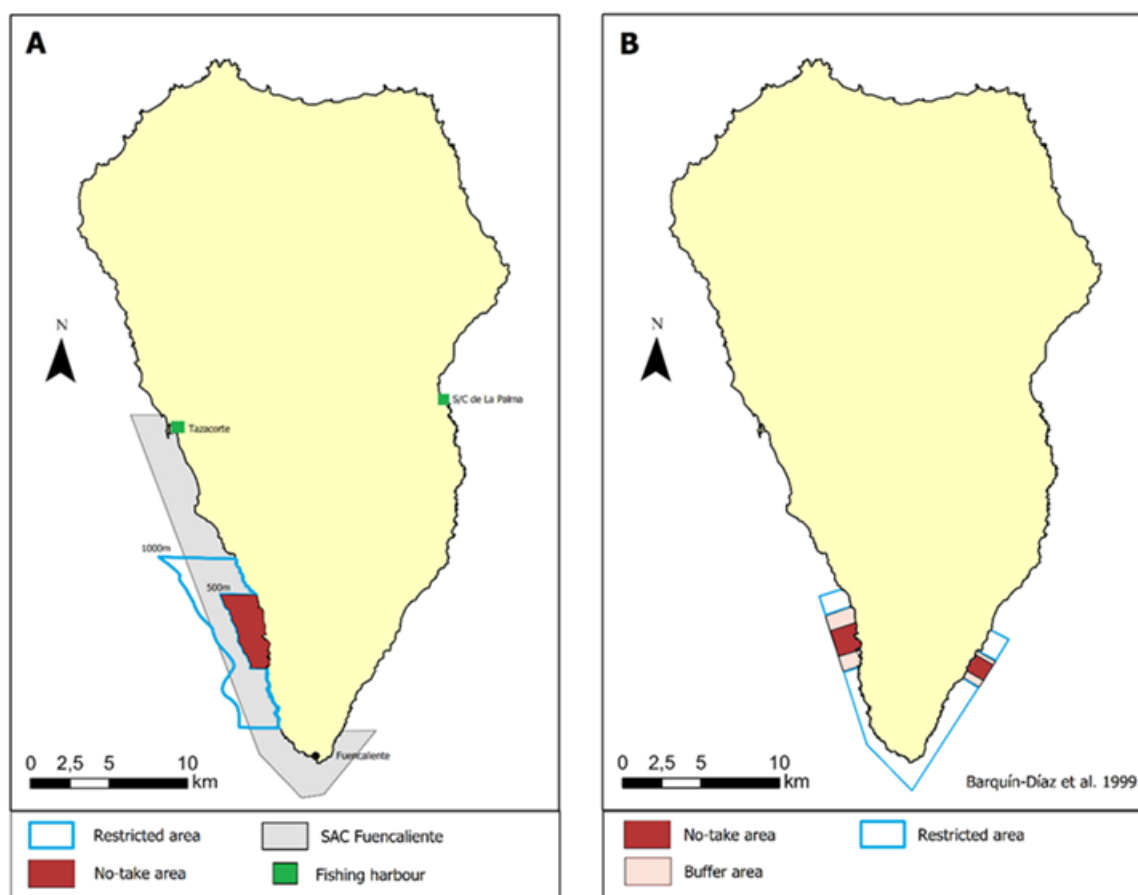


Figure 3. A) Contemporary location of the La Palma Island MPA; and B) initial proposed MPA boundaries (Barquín-Díaz et al., 1999).

A range of fishing activities are allowed within the La Palma Island MPA, and these differ depending on the level of fishing restrictions. In the no-take zone, any fishing activity, harvesting and scuba diving are prohibited, unless for authorised scientific purposes. For artisanal vessels utilising the restricted areas (surrounding the no-take zone), these must be registered in a census under the conditions pursuant to BOE-A-2001-6008. The requirements to be registered are: having a base harbour in Tazacorte or Santa Cruz de La Palma; registered in the use of small-scale fishing gears; and proof that the vessel has been fishing in the area in the two years prior to the establishment of the La Palma Island MPA with permitted fishing gears. For example, in 2020, 45 artisanal fishing boats were authorised to fish in the restricted zone of the La Palma Island MPA according to the official fishing fleet census in La Palma (BOE-A-2020-13180). Where fishing is permitted, activities include pole and line, surface trolling and tuna and live bait for tuna fishing. In addition, within the restricted area, any types of recreational fishing are prohibited, except fishing from the shore outside the no-take reserve and under the conditions pursuant to Order APA/1936/2002 of 18 July 2002 (BOE-181-2002). The recreational fishers are allowed to

fish only from the coast in the restricted area with a daily fishing quota of 5 kg (BOE-A-1986-27546).

Surveillance of the MPA is undertaken by fisheries rangers in collaboration with marine authorities. The La Palma Island MPA fisheries rangers service patrols on a boat targeting the fishing activities and law enforcement inside the La Palma Island MPA. However, when an infringement is reported by the fisheries rangers, the Spanish Guardia Civil is the competent authority to impose penalties.

3. AIMS AND OBJECTIVES

The main aim of this report was to determine whether there have been changes in the spatial strategy of fishing activities in response to the implementation of the La Graciosa Island MPA and La Palma Island MPA. Based on the available data on fishing activities within the Canary Islands, this case study had the following specific objectives:

1. to develop physical and benthic habitat maps (European Nature Information System (EUNIS) habitat type / European Marine Observation and Data Network (EMODnet)) for both the La Graciosa Island MPA and La Palma Island MPA and surrounding areas;
2. to map temporal changes in fishing effort (i.e. fishing hours) and fishing gear use within the La Graciosa Island MPA and La Palma Island MPA in the period 2008- 2023 to assess whether there are differences in fishing activity between inside and outside each MPA across time;
3. to examine changes in habitat use by fishing fleet to MPA implementation based on previously built maps.

4. METHODOLOGY

4.1. Data types and sources

Spatial fishing strategies in the La Graciosa Island MPA and the La Palma Island MPA were explored, based on different data sources, after respective MPA designation. Importantly, there was no available information on fishing activities undertaken throughout the regions covered by the two MPAs before designation. Therefore, this analysis does not provide a synopsis of the direct effects of both MPAs on the potential reallocation of fishing activities. However, the data can be used to gain an understanding of how fishing activities have been structured around the MPAs, and whether this has changed over the time series of the data available.

4.1.1. Fishing effort of artisanal fleet – quantitative and semi-quantitative data

To determine the main fishing area distribution of the artisanal fleet, fishing effort data for each of the MPAs as soon after designation (La Graciosa Island MPA: 1995, La Palma Island

MPA: 2001) was examined – this encompassed two distinct time points: quantitative data in 2008 and semi-quantitative data in 2023 ⁽¹⁷⁾.

4.1.1.1. RESMARCAN (2008)

The 2008 artisanal data were collated from onboard observations associated with a collaboration agreement on MPA monitoring between MAPA and the Spanish Institute of Oceanography – Spanish National Research Council (IEO-CSIC) in the Canary Islands (RESMARCAN) (Martín-Sosa et al., 2010; Martín-Sosa et al., 2011). Such quantitative data encompassed fishing effort data (expressed as 'fishing days'), which represents fishing activity of small-scale fisheries in 2008. The collection of such data was mainly from within the boundary of each MPA. All fishing effort points (.csv) were converted into a shapefile (.shp).

4.1.1.2. TEPESCO (2023)

Artisanal fishing effort data from 2023 were also examined. The data were sourced from in situ interviews of artisanal fishermen from both MPAs. Such data encompassed a qualitative description of the fishing effort areas, expressed as number of fishing boats (quantitative) fishing on each square from a 1 x 1 km grid and was also used to contrast the 2008 fishing activity. The data were collated from the monitoring and evaluation of fishing activities in the Special Areas of Conservation (SACs) of the Canary Islands by IEO-CSIC (TEPESCO) (IEO-CSIC; Ministry of Science, Innovation and Universities – unpublished data).

4.1.2. Fishing effort of commercial fleet – automated information system analysis

To further represent the fishing activities throughout the two MPAs, professional fishing fleet activity in the surrounding areas was assessed using automated information system (AIS) data. Available AIS data from the Global Fishing **Watch (GFW) were used to represent three time-steps (2012, 2016 and 2020) after MPA implementation within the La Graciosa Island MPA and the La Palma Island MPA**. AIS data (fishing hours) were processed in RStudio (RStudio Team, 2020) to obtain a skewed file (.csv) for the Canary Island region, representing fishing activity of larger fishing vessels that operate with other fishing gears (longlines, traps and tuna fisheries) in the surrounding areas. The skewed AIS data were also converted into a shapefile (.shp).

To obtain a global view of the fishing activity in both MPAs, processed spatial fishing effort data from GFW, RESMARCAN (2008) and TEPESCO (2023) were plotted by time-step year using different symbol (e.g. heat map and graduated symbols). These map plots were represented disaggregated by gear type, visualising the different fishing strategies.

⁽¹⁷⁾ The experimental design in this case study was only suitable for an After (A) analysis.

4.1.3. EMODnet data

To develop a benthic habitat layer, EMODnet broad-scale sea-bed habitats data for Europe, were incorporated in the basemap (background layer based on GIS). This includes physical habitat maps (EUSeaMap) and observations gathered from surveys across Europe. The EUSeaMap project has brought together a European consortium of specialists in benthic ecology and seabed habitat mapping. The broad-scale predictive mapping methods are repeatable and ensure that the predictive maps can continue to be improved in the future.

The seabed substrate that is used as input to EUSeaMap is the result of combining multiple datasets at multiple resolutions into a single dataset that contains the highest resolution available at each location. The EUSeaMap predictive model includes the sublittoral zone with roughly 100 meters resolution. Due to the lack of detailed substrate data and the resolution of the model, it is difficult to predict sublittoral habitats at this scale. EUSeaMap requires a combination of seabed types to be classified according to a modified version of the Folk (1954) classification system with 7 classes (Rock, Coarse substrate, Mixed sediment, Sand, Muddy sand, Sandy mud, Mud) and biogenic substrate (Vasquez et al., 2023).

The EUSeaMap (2023) was obtained as an ArcGIS geodatabase file ⁽¹⁸⁾ and incorporated in a base map within the fishing effort layers (AIS/RESMARCAN/TEPESCO) to analyse the habitat use overlapping with fishing activity.

5. RESULTS

5.1. Quantitative and semi-quantitative results for fishing in the La Graciosa Island MPA and the La Palma Island MPA

The quantitative data (RESMARCAN) illustrated that after MPA designation, the small-scale fishing fleet of the La Graciosa Island MPA decided to stop using restricted fishing gears (traps, trammel nets and longlines) in MPA waters. Only a few large fishing boats, displaced out of the La Graciosa Island MPA boundaries, still fished with traps. Such data also show the displacement of the fishing boats operating with traps. The longline fishery (restricted in MPA waters) was replaced by artisanal pole and line (electric reel).

In La Palma Island MPA the RESMARCAN data illustrated that the fishing gear restrictions resulted in the displacement of the restricted fishing gears outside of the MPA. However, the small size of the MPA did not account for increased fishing effort or fuel costs. These fishing strategies are currently (2023) ongoing based on fisher interviews (TEPESCO) in the La Graciosa Island (undertaken by large fishing boats) and La Palma Island MPAs and the surrounding areas. The fishing scenario in the La Graciosa Island MPA is also highlighted in the 2023 spatial distribution, where *Merluccius merluccius* traditional fishing grounds overlap with commercial vessels' fishing areas.

⁽¹⁸⁾ www.emodnet.ec.europa.eu/

5.2. La Graciosa Island MPA – analysis of available spatial data

5.2.1. Small-scale fishing activity

The RESMARCAN data summarises the spatial distribution of small-scale (artisanal) pole and line and traps (Figure 4), within the region, while the TEPESCO (2023) data also show the currently spatial distribution activities for the same fishing gears based on semi-quantitative data Figure 5). Such fishing is predominantly pole and line, although trap activities are also being undertaken. The use of traps on the edges and adjacent to the La Graciosa Island MPA are shown, with such fishing being undertaken by the small-scale fleet on non-specified substrata habitats (250–350 m depth).

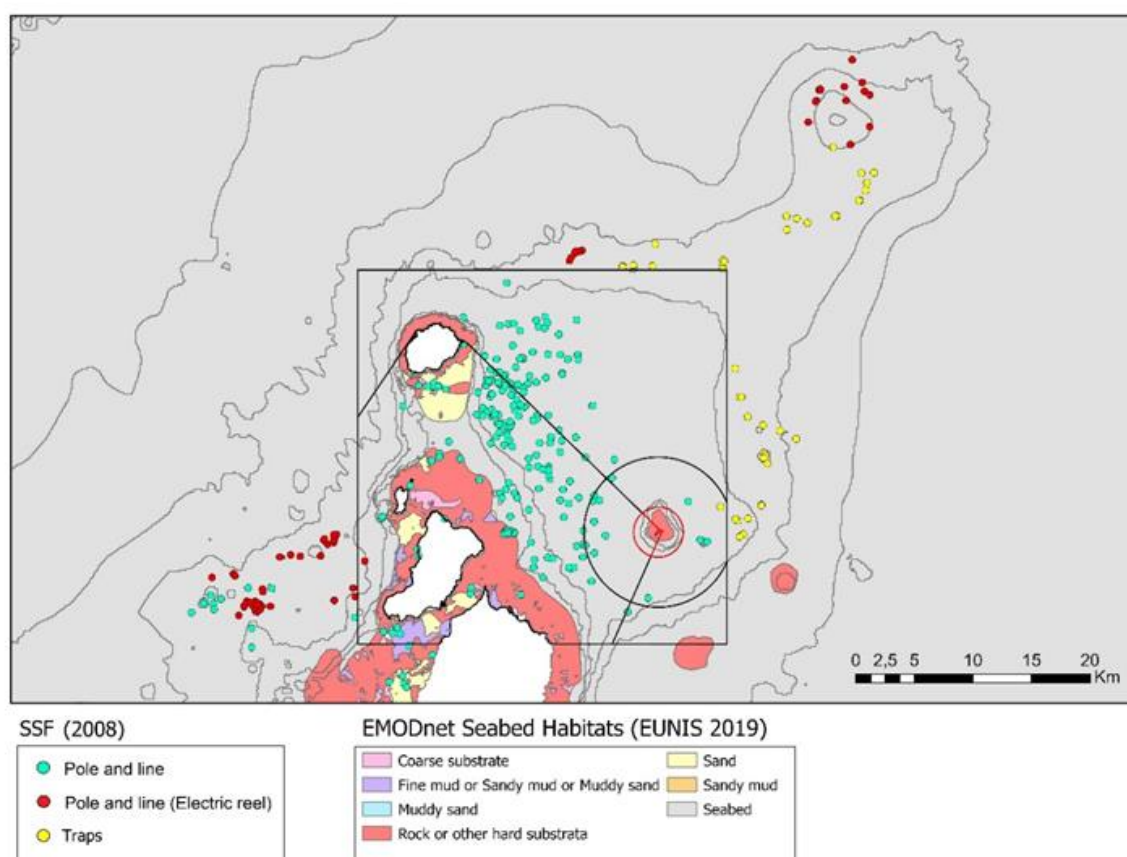


Figure 4. La Graciosa Island, 2008 small-scale fishery activities (RESMARCAN)

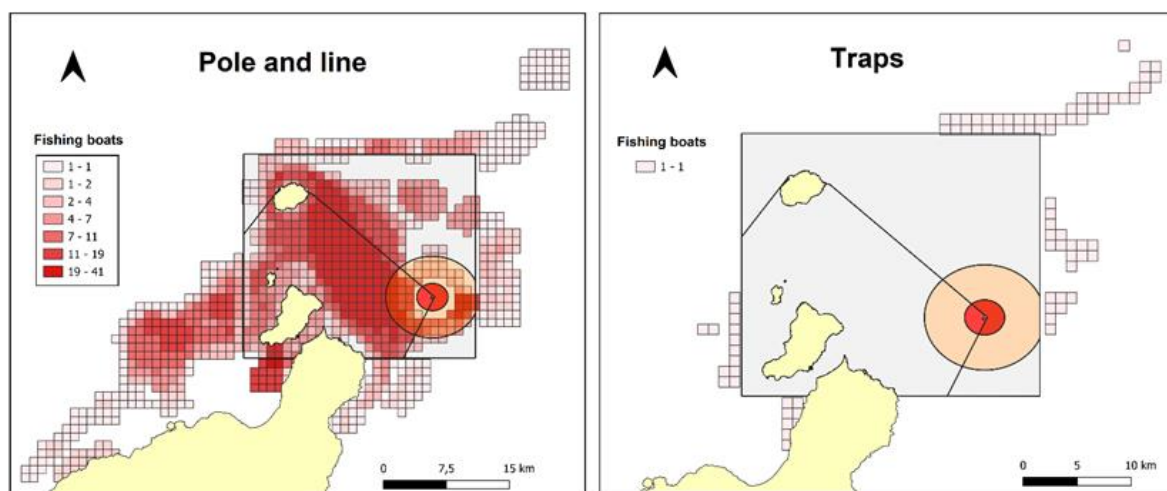


Figure 5. La Graciosa Island, 2023 small-scale fishery activities (TEPESCO)

5.2.2. Commercial fishing activity

The 2012 (AIS) data show pole and line fishing activity (for tuna) far away and outside the designated area, at the eastern region of the Chinijo archipelago (Figure 6). When comparing data for 2012 and 2016, it can be seen that the commercial fleet displaced the pole and line activities adjacent to the MPA (Figure 7).

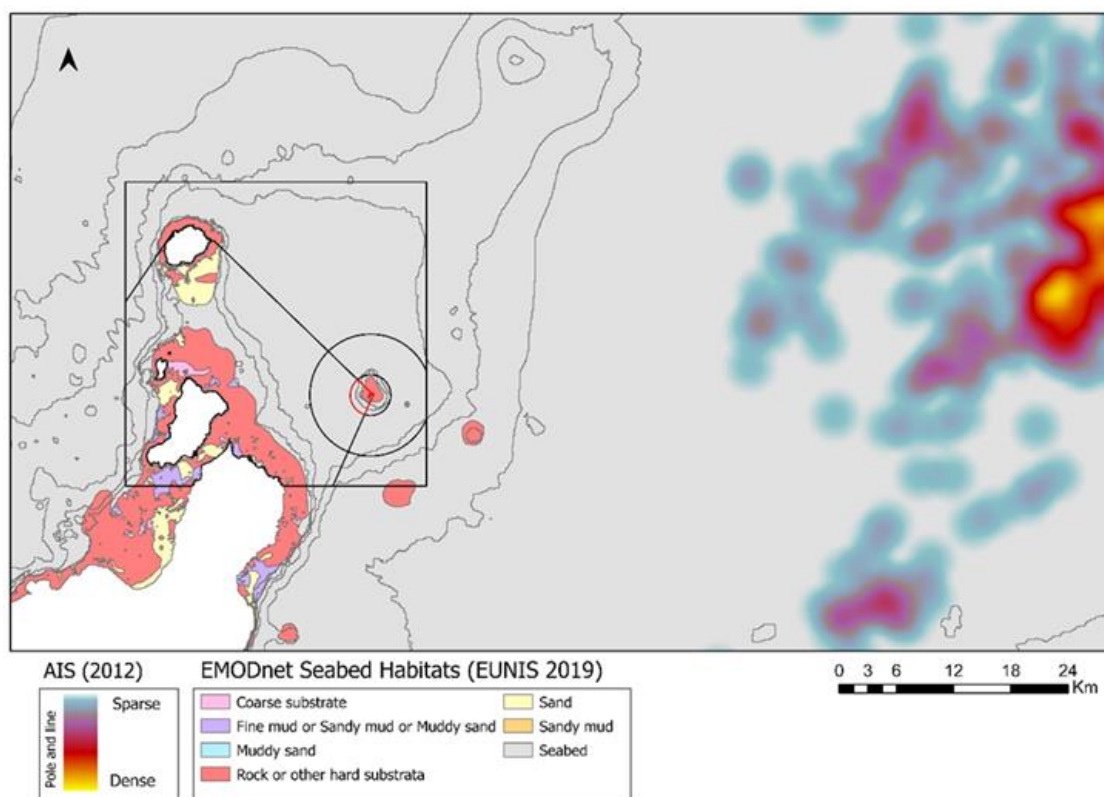


Figure 6. La Graciosa Island, 2012 (AIS) commercial fishing activity

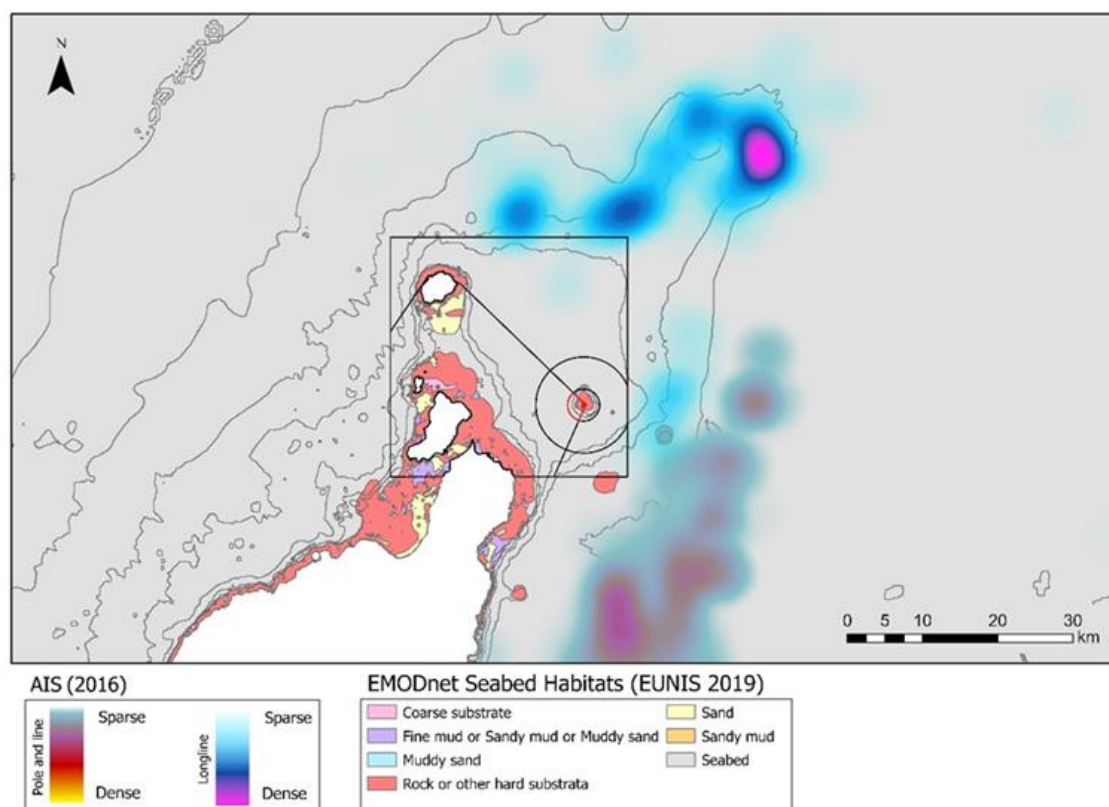


Figure 7. La Graciosa Island, 2016 (AIS) commercial fishing activity

Importantly, there is a substantial amount of pole and line activity to the south-east of the MPA (although this does not overlap with the designated area). In addition, there is longline activity to the north of the designated area, overlapping with artisanal electric reel and trap fishing grounds. However, substantial longline fishing to the north-east of the designated area does not overlap with this area.

In 2020, all fishing activity (both longline and pole and line) is outside the designated area (Figure 8). However, longline fishing densities exploited other fishing grounds (east and southwest) overlapping currently (2023) artisanal fishing grounds of electric reel. Where fishing activities are being undertaken (i.e. longline), although these are adjacent to the designated area, there is virtually no overlap between the designated area and such fishing activities. Regarding the pole and line activity, the 2020 data shows dispersion around the designated area.

Overall, from 2012 to 2020 fishing effort (AIS) patterns of the industrial fishery increased mainly on the external boundaries of the MPA. The main fishing gears used were likely associated with tuna fisheries, with longliners changing the fishing effort area between 2016 and 2020 when they started to target the European hake (*Merluccius merluccius*).

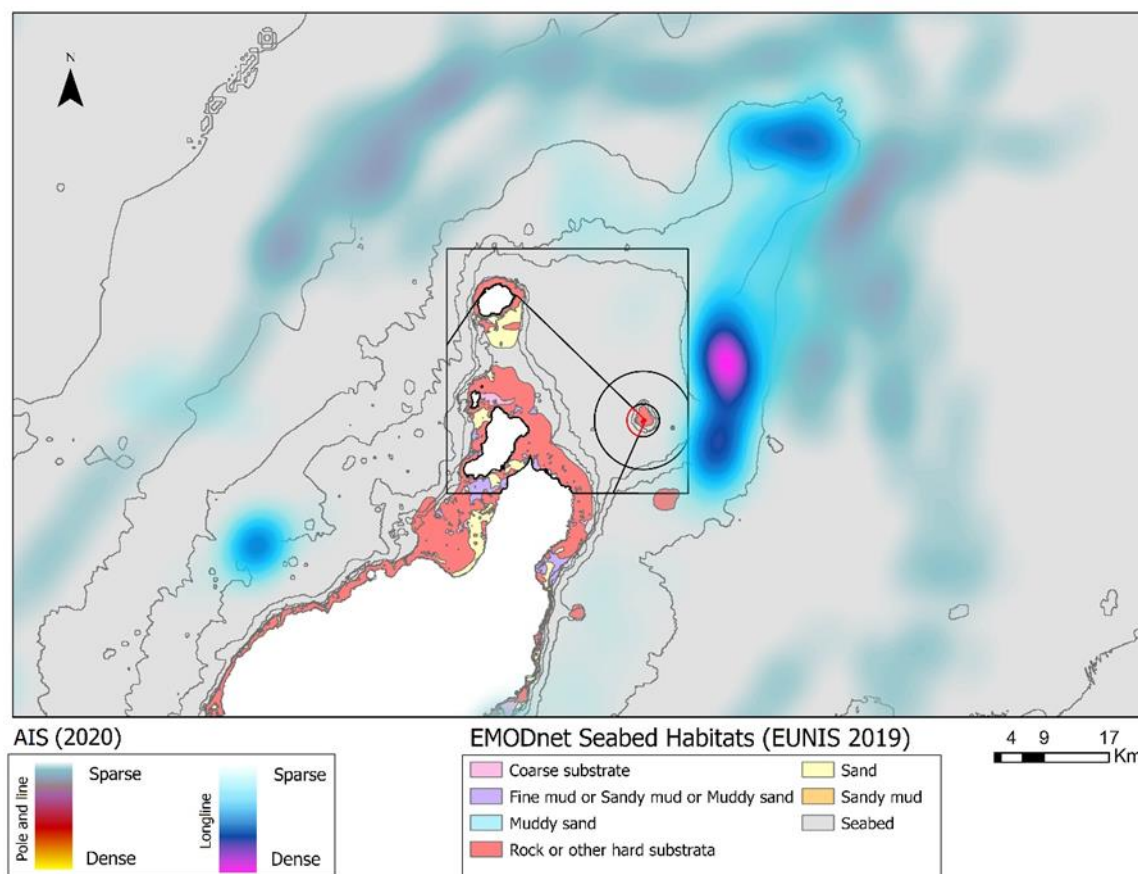


Figure 8. La Graciosa Island, 2020 (AIS) commercial fishing activity

5.3. La Palma Island MPA – analysis of available spatial data

5.3.1. Small-scale fishing activity

In 2008 (pole and line, small-scale artisanal trapping, trammel nets, purse seine and longlines), there was a substantial array of fishing activity in the La Palma Island MPA and both north and south of the MPA (Figure 9). Within the MPA, also shown in 2023 (Figure 10), the majority of fishing activity is based on pole and line fishing within the restricted zone.

Between 2008 and 2023, there were several types of small-scale artisanal fishing activity adjacent to and outside the La Palma Island MPA. These were predominantly trapping, but also sporadic use of small-scale purse seine (live bait), longline and trammel nets. Importantly, the majority of such activity, although closely adjacent to the restricted area of the MPA, was not substantially undertaken within this zone, and there is no evidence to suggest that any of these activities were undertaken within the no-take zone.

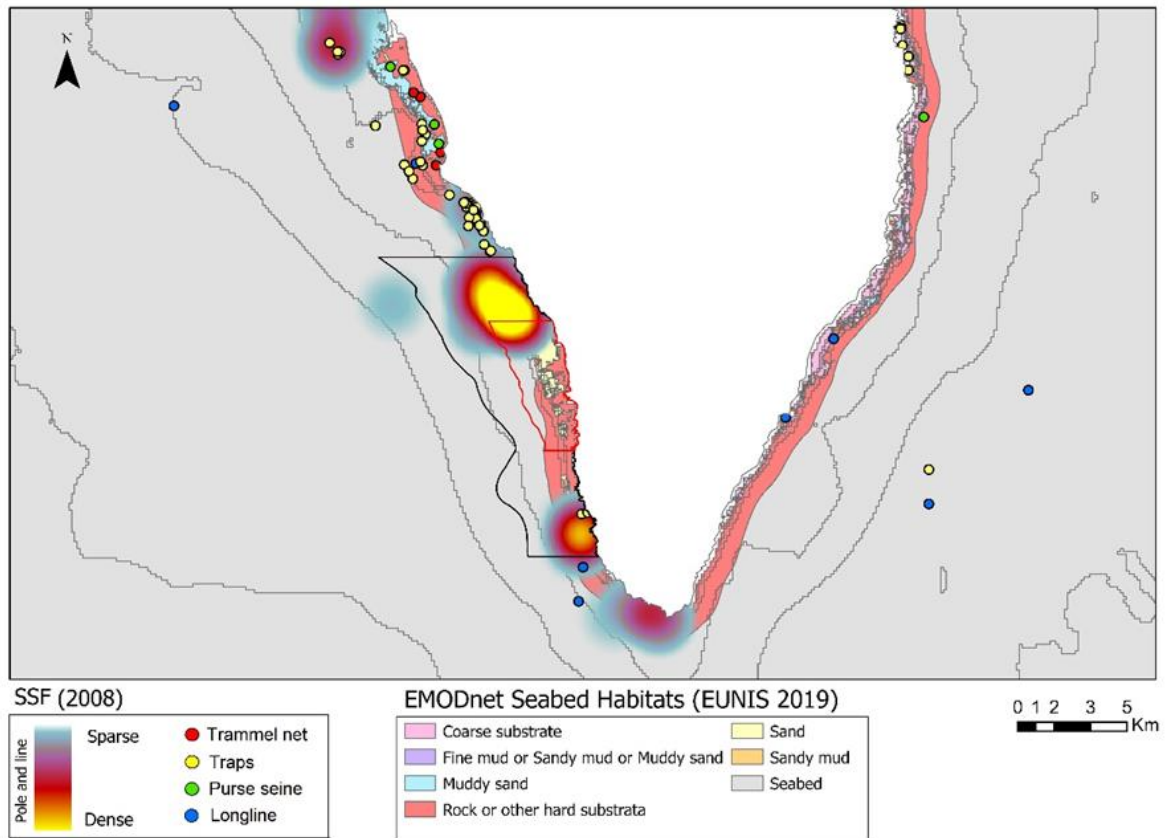


Figure 9. La Palma Island, 2008 small-scale fishing activity (RESMARCAN)

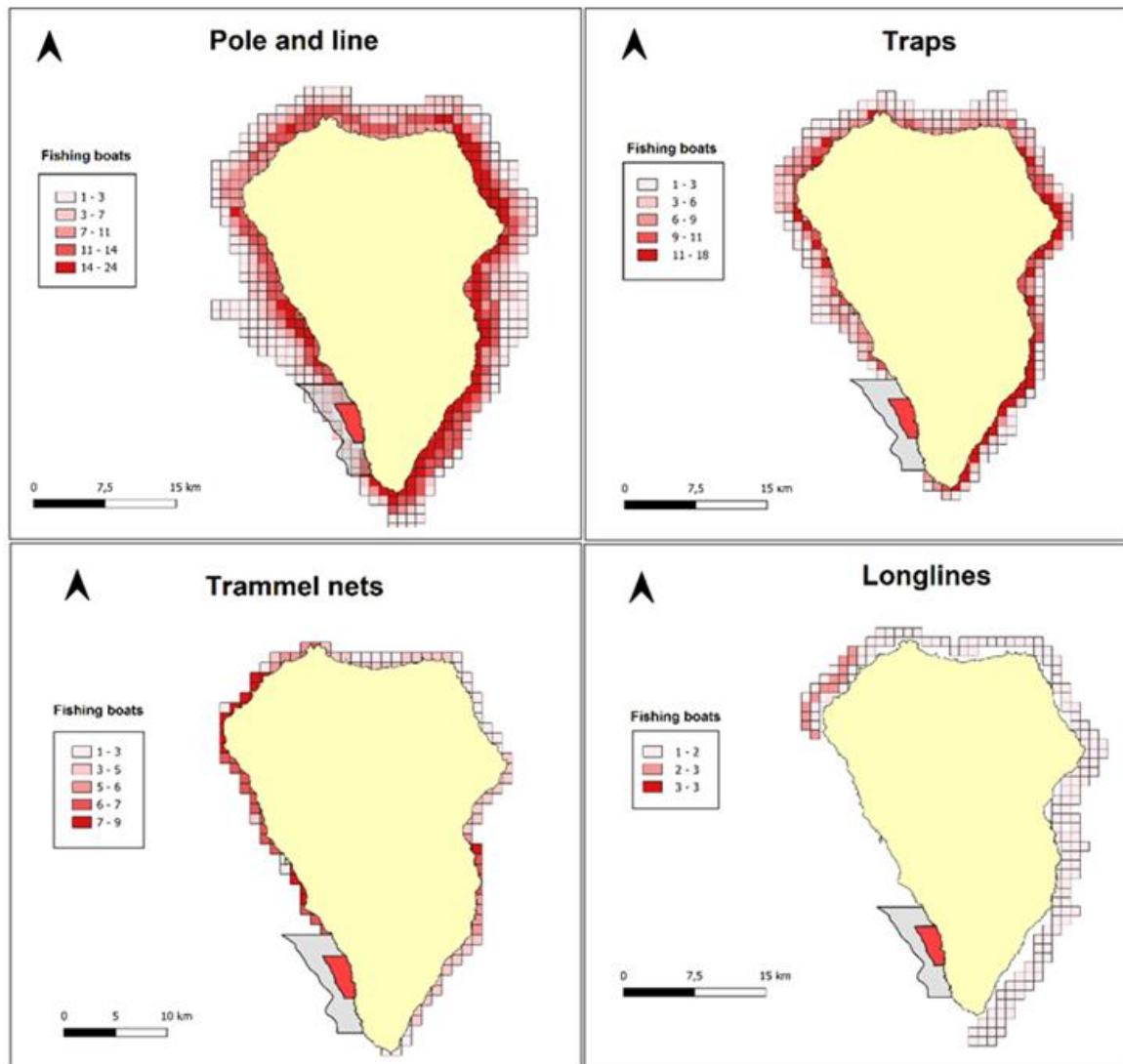


Figure 10. La Palma Island, 2023 small-scale fishing activity (TEPESCO)

5.3.2. Commercial fishing activity

In 2016, all commercial fishing activities (identified by AIS) were substantially undertaken outside the La Palma Island MPA (Figure 11). This encompasses pole and line activity (tuna fisheries). Importantly, this shows the movement of fishing activity, not only away from the no-take area (i.e. with the restricted zone), but to completely outside the designated area.

Commercial fishing activity in 2020 increased to the north and south of the designated area, though with little evidence (as in 2016) to suggest such activity within the designated area (Figure 12).

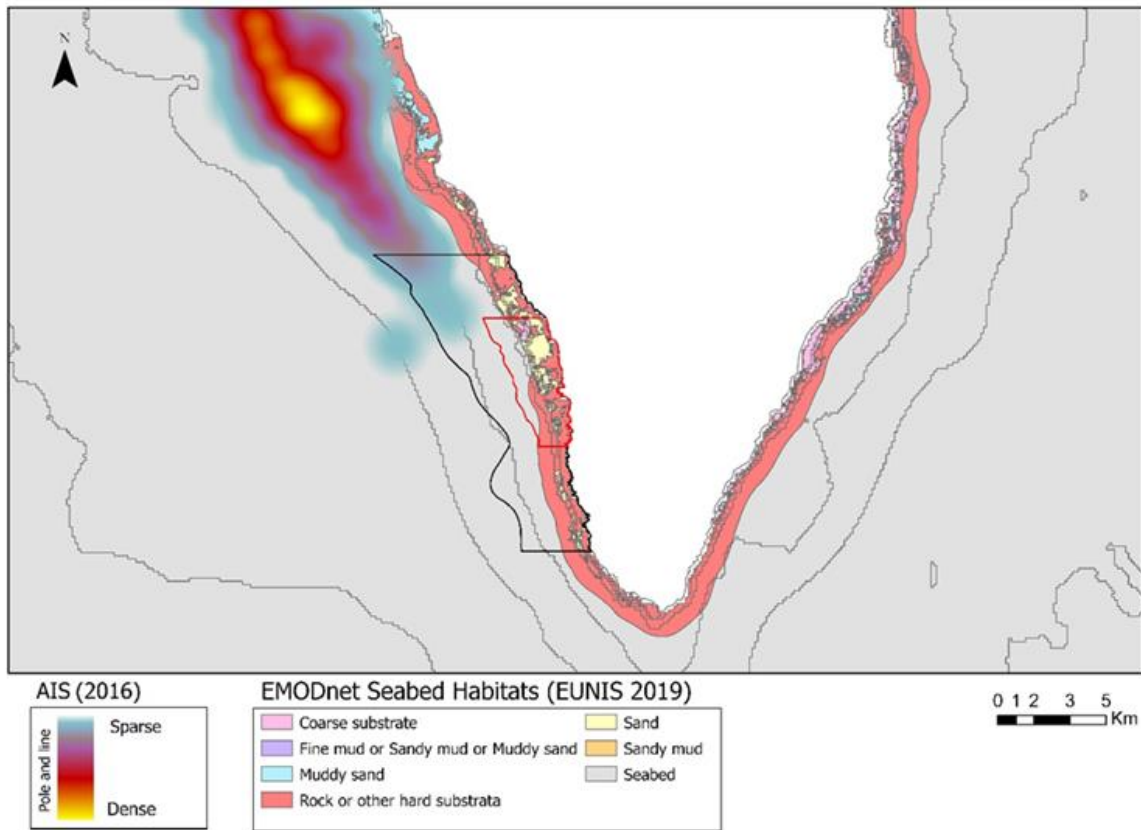


Figure 11. La Palma Island, 2016 (AIS) commercial fishing activity

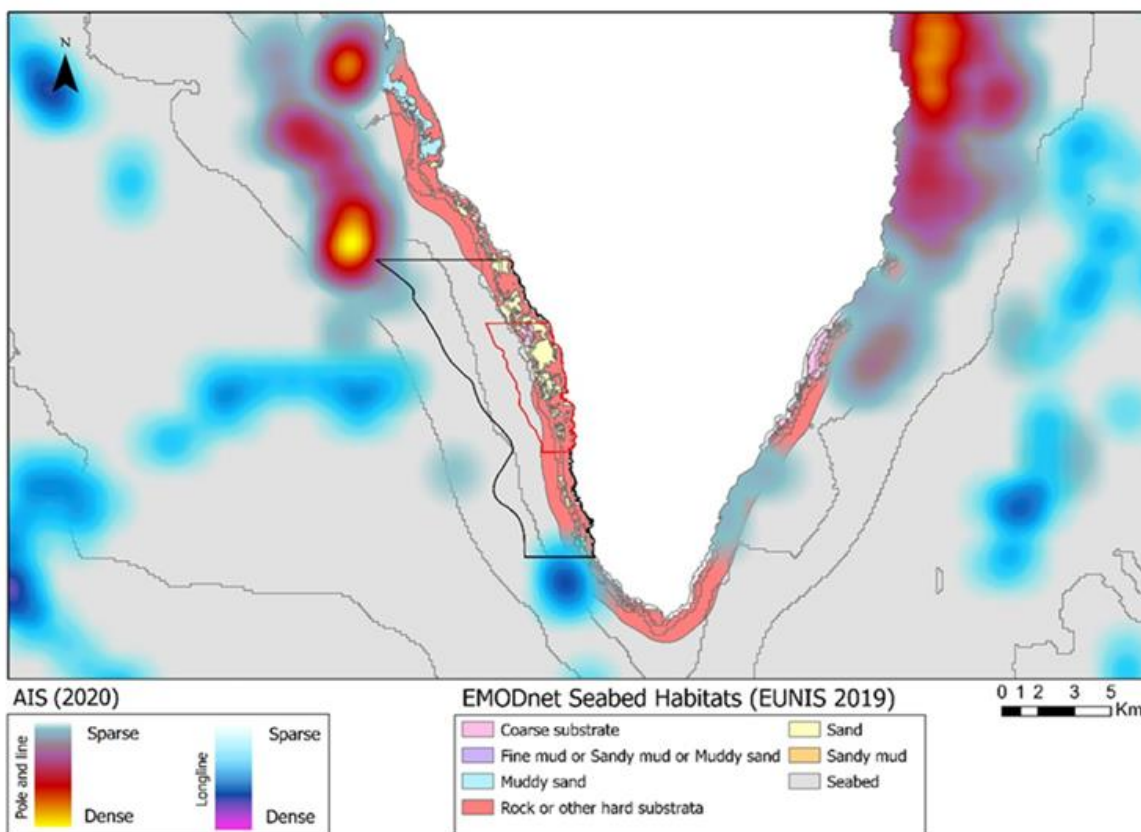


Figure 12. La Palma Island, 2020 (AIS) commercial fishing activity

5.4. Analysis of EMODnet broad-scale sea-bed habitats data

Within the La Graciosa Island MPA the highest fishing effort density corresponds with fishing activities being undertaken on “seabed” habitats (80–150 m depth) where the type of substrata for this fishing grounds is not available in EMODnet (Figure 4). Moreover, outside the La Graciosa Island MPA there are other exploited fishing grounds (rocky outcrops and muddy sand habitats) by the small-scale fleet (250–350m depth) and commercial longliners where the “seabed” habitat does not provide accurate or relevant information (Figure 4).

Within and adjacent to the La Palma Island MPA, the main small-scale fishing activity is in the coastal rocky-sand habitats, but also in deeper (likely sandy) habitats away from the shore (Figure 9). The lack of a better habitat resolution data does not make an analysis of habitat change possible in the 2008–2023 period.

6. DISCUSSION

The establishment of MPAs in the Canary Islands and the resulting spatial fishing restrictions triggered **important changes in the use of traditional fishing gears, fishing grounds and target species**. The main changes were associated with the abandonment or disuse of restricted fishing gears (e.g. traps, purse seine and trammel nets) inside the MPA. Although no pre-designation fishing-effort data were available within this analysis (i.e. before 1995 for the La Graciosa Island MPA; before 2001 for the La Palma Island MPA) the results of **this analysis show the displacement of the majority of fishing gears after 2008 to the external MPA boundaries and offshore fishing grounds**. Such spatial distribution data shows that changes to fishing regulations have had substantial impacts on the fishing strategies undertaken within the La Graciosa Island and La Palma Island MPAs. The artisanal fleet adaptations to fishing regulations since the establishment of the MPAs have not been quantified in economic or social terms. Such fishing adaptations described in RESMARCAN projects (Martin-Sosa et al., 2010, Martin-Sosa et al., 2011) need a mid- and long-term of temporal data to evaluate properly the fishing changes. The available fishing effort data are limited to a snapshot of two distinct time points. However, **the results evidence the adaptive capacity of fishing activity to MPA establishment**.

Between the two MPAs, differences in the total size of the MPA showed little effect on how fishing activity changed. Specifically, despite the small size of the La Palma Island MPA, the cessation of activities within the MPA was relatively similar to that found in the larger La Graciosa Island MPA.

Within both regions, between 2012 and 2020 both **longline and pole and line commercial fishing activity moved closer to the MPA boundaries**, increasing the fishing effort with little intrusion in 2020. In comparison, between 2008 and 2023, small-scale fishing activity also showed a dispersal of fishing spatial distribution.

Currently, looking at spatial fishing strategies outside the La Graciosa Island MPA, very few artisanal fishing boats operate with traps, while in the La Palma Island MPA activity with trammel nets and traps remains. The longline fisheries of the La Graciosa Island MPA artisanal fleet have been replaced by pole and line (electric reel) where only industrial longliners are still operating with this fishing gear. **The consequence of this adaptation to the MPA regulation is the overlap of fishing grounds and strategies between the artisanal and industrial fleet at the external MPA boundaries**. This scenario leaves the artisanal fishery at a disadvantage in terms of catches that could be managed

with longline-specific zonation rules (buffer areas) in the surroundings of the La Graciosa Island MPA.

Furthermore, the current fishing scenario in the La Graciosa Island MPA is focused on the exploitation of few fishing resources. The diversification of fishing activities and exploited resources could be helpful to minimise the MPA's influence on the behaviour of artisanal fishers (Mendoza et al., 2022). This multi-species approach may include the return of traditional fishing gears with low impact on habitats (Cochrane, 2005) (e.g. crustacean traps). The recovery of some overexploited and offshore traditional fishing grounds (e.g. *Merluccius merluccius*) would also contribute to fishing fleet diversification. However, navigating longer distances to reach offshore grounds was not accounted for in MPA implementation and has increased fuel costs. Therefore, MPA influence on fisher behaviour must be considered when evaluating the effects of marine spatial conservation measures.

The designation of the no-take area within the La Graciosa Island MPA displaced very little fishing effort considering the low representation (1.7 %) in relation to the total protected area. In this regard, it would be advisable to review the initial MPA proposal zoning (Bacallado et al., 1989) and increase the no-take percentage with an additional fully protected area (Claudet et al., 2008). It could also be suitable to allow traditional tuna fishing (hook and line) in the no-take area, as is the case in other Canary Islands MPAs (El Hierro).

Overall, studies characterising the dynamics of the artisanal fishery in the Canarian MPAs are scarce (Castro et al., 2019; Gonzalez et al., 2020). Effort data before MPA implementation is practically non-existent in the Canary Islands. Therefore, there is a need to obtain a great amount of fisheries dynamics data both inside and outside (surroundings) the protected areas. However, this requires a well-organized programme of on-board sampling as well as details of the season, gear type and habitat features. The importance of characterising the spatio-temporal patterns is essential to inform the design of future fisheries spatial management measures, including for MPAs. Currently, the IEO-CSIC in the Canary Islands are developing the small-scale fishery spatial distribution for the entire archipelago. The semi-quantitative spatial distribution included in this report (TEPESCO) will be validated with a GPS tracking programme during the next years.

Finally, EMODnet seabed habitats data is not sufficient in terms of specificity and resolution when examining the potential impact of the analysed fishery spatial distribution. This is the principal scope of EUSeaMaps, due to the lack of available knowledge on the seabed types. However, it is expected that more seabed substrate data will become available in the future (Vasquez, et al., 2017). The main fishing activity in the Canary Islands occurs within a short and abrupt volcanic basin followed by a seabed slope habitat reaching great depths. This means that a large percentage of the fishing activity overlaps at the EMODnet seabed habitat (circalittoral), which does not have a detailed substrata type. The lack of information about deeper habitats and the particular cartography of the Canary Islands limit the potential impact analysis for the different fishing gears (e.g. traps, longlines and trammel nets). The characterisation and regulation of those seabed rocky-sand habitats where artisanal and industrial fleets overlap when targeting some fishing resources (e.g. *Merluccius merluccius*) could reduce the resource competition in traditional fishing grounds.

Finally, there is need to incorporate the still ongoing cartography studies in the IEO-CSIC of deeper habitats where longline professional fishing activity and small-scale trapping activity have potential impacts on the seabed. This could affect the fishing grounds and habitats where some small-scale fishing boats operate. Specifically, the latest MAPA studies are focused on mapping the biogenic communities and characterisation of benthic

communities belonging to circalittoral reef habitats of the marine SACs in the Canary Islands (CANZEC).

7. CONCLUSIONS

This report describes the influence of MPA restrictions on the spatial fishing effort reallocation during the 2008-2023 period. The main findings show that the fishing restrictions within each MPA results in the abandonment or disuse of small-scale fishing gears (e.g. traps and trammel nets) inside the MPA restricted zone. In this respect, implementation of both MPAs has led to long-term displacement of restricted fishing gears to the external boundary of each MPA, including fishing being undertaken further offshore. This has resulted in different strategies around the MPA boundaries where the artisanal and industrial fleets overlap targeting some fishing resources. The artisanal fishing effort spatial distribution also evidenced the fishing gears restrictions (passive gears) inside the MPA. This scenario highlighted the need to incorporate additional fishing diversification measures to minimize the MPA influence on the traditional artisanal fisheries behaviour. In this sense, this study highlights the need to develop a permanent programme of onboard observers to monitor the Canarian small-scale fishery activity throughout archipelago.

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