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B-USEFUL

User-oriented Solutions for Improved Monitoring and Management of Biodiversity and Ecosystem services in vulnerable European Seas

Deliverable 4.3 Match/mismatch between European marine protected areas and 'hotspots' of sensitivity or risk from climate change and fishing

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Executive summary

Understanding the vulnerabilities of marine life to human pressures is key for Horizon project B-USEFUL to develop “user-oriented tools and solutions to conserve and protect marine biodiversity” in support of the EU Green Deal, Biodiversity Strategy 2030, and national biodiversity policies. The primary aim of work package 4 (WP4) is to *identify habitats and key species at risk of extinction in sensitive ecosystems* by developing a hierarchical risk-based framework.

In two previous deliverable reports, WP4 assessed the sensitivities and risks posed to European marine biodiversity by two key pressures: climate change and fishing. The first of these reports (D4.1) looked at *individual species'* sensitivities to these two pressures, while the second (D4.2) assessed the *community-level and habitat-level* sensitivities and risks to climate change and fishing. Building on these findings, the present, third deliverable report aims at assessing the match, or potential mismatch, of sensitivities and risk with biodiversity protection. More specifically, we pursue the following research questions:

(1) Where are the most important ‘biodiversity hotspots’ – based on species richness, sensitivity and/or risk from climate change and fishing – within Europe's different regional seas?

(2) To what extent are the most important ‘hotspots’ protected through Marine Protected Areas (MPAs) and/or fishery-restricted areas, taking into account the varying levels of protection offered?

Firstly, we describe the approach to categorising MPAs according to the level of protection afforded, harmonised across European regional seas. Based on *the MPA Guide* but extended beyond EU waters to include non-EU waters, we distinguish between fully, highly, lightly and minimally protected MPAs, and levels of protection below that.

Next, we identify biodiversity and sensitivity hotspots across European regional seas and assess their match or mismatch with MPAs and/or fishery-restricted areas (FRAs). The key messages emerging, by basin and species group assessed, are that:

- Many biodiversity and sensitivity hotspots in the Western Mediterranean Sea fall within MPAs or FRAs, but the level of protection is generally low. In the Eastern Mediterranean Sea, most hotspots fall outside MPAs. Improving protection requires strengthening MPAs, expanding coverage especially in the Eastern basin, and integrating FRAs to enhance conservation effectiveness.
- Within the North-East Atlantic, groundfish communities in the Northern North Sea emerged as highly sensitive to warming, but with areas at high risk largely outside MPAs. The Bay of Biscay and Iberian Coast emerged as large climate-risk hotspots; but strict protection is almost absent. Therefore, improving protection requires strengthening existing MPAs, creating new ones, expanding boundaries, and prioritising ecological effectiveness over simple area-based targets.

- In Icelandic waters, a limited extent of groundfish biodiversity hotspots identified emerged as protected, mainly through seasonal FRAs. The most urgent gaps concern Arctic species hotspots in northern and western waters. Protection could be strengthened by upgrading seasonal closures, expanding protected areas, or applying species-specific measures, while ensuring climate-sensitive Arctic species are adequately safeguarded.
- In the North Sea, only 19% of vulnerable epibenthic communities and 8.5% of highly sensitive habitats lie within MPAs, mostly with inadequate protection. Major mismatches stem from fishing and dredging being allowed. Improving protection may require stepwise strengthening of restrictions, targeted safeguards for sensitive habitats, and adaptive measures as vulnerability hotspots shift over time.
- Most benthic habitats in continental Portugal's Exclusive Economic Zone (EEZ) lack effective protection, with restricted MPAs covering only 0.02%. Sensitive habitats – maërl, seamounts, rocky reefs, macroalgae forests, and mud volcanoes – require stricter protection and better mapping. A resilient European MPA network must ensure habitat representativity, connectivity, and protection of climate refugia.
- In the Eastern Mediterranean, Lessepsian non-indigenous species (NIS, of Indo-Pacific origin) increasingly threaten MPAs and vulnerable nursery habitats. While some areas remain unaffected, rapid spread demands continuous monitoring and adaptive management. Effective response requires targeted NIS control, stakeholder-supported harvesting incentives, and a dedicated European-wide NIS management framework.

Across all study areas, our methodology has yielded valuable new information that can support managing and implementing MPAs across European basins. We could clearly see impacts of pressures from fisheries and climate change, either determining that systems were impacted, or that systems were recovering or had already adapted. Together with information on biodiversity and species composition, we have proven that a vulnerability/risk framework adds value beyond current management approaches to MPAs. The quality of managing MPAs could be improved by applying this methodology, thereby improving the levels of populations, species and biodiversity.

For the first time, this study has done a comprehensive mapping of biodiversity hotspots, vulnerabilities and risks across European seas, and assessed to what extent these hotspots fall within the existing MPA network. The assessment shows that many hotspots are located outside MPAs or where within the network, are often lacking adequate levels of protection, prompting a need to reassess MPA design and management. Future work should expand the approach to more taxa, include shifts in the locations of hotspots over time, and integrate stakeholder interests for truly holistic MPA planning and management.

The role of this deliverable

This deliverable (D4.3) is the third of three reports in WP4 of EU project "*User-oriented Solutions for Improved Monitoring and Management of Biodiversity and Ecosystem services in vulnerable European Seas*" (B-USEFUL) that together comprise "*Risk and vulnerability*". Understanding the sensitivities of Europe's marine species to key pressures (both environmental and human-induced) will support the overarching aim of B-USEFUL to develop tools and solutions to manage marine biodiversity. This work supports international policies including the EU Green Deal and Biodiversity Strategy 2030. It also supports national policies: within the UK, it is relevant for the Marine Environment Plan and UK Biodiversity Strategy; and for Icelandic waters, the work supports the Nordic Biodiversity Framework.

Throughout B-USEFUL WP4, the emphasis is on two highly dominating pressures: (1) climate change and (2) fishing pressure (and associated physical seabed disturbance). Anthropogenically accelerated climate change in combination with (over-)exploitation of marine life are seen as key drivers of biodiversity loss, both globally and in Europe's regional seas. These drivers operate through multiple ecological mechanisms including trophic restructuring, habitat degradation, shifts in species distributions and loss of functional diversity (IPCC 2022; IPBES 2019). Hence, understanding the mechanisms by which climate change and fishing pressure impact marine ecosystems is crucial for biodiversity conservation and sustainable resource management.

In WP4, functional approaches based on species' biological traits are used, firstly, to assess the *sensitivities and vulnerabilities* of marine species to the impacts of climate change and fishing pressure (Deliverable D4.1, Engelhard et al. 2024). An important step has been the development of two new trait-based sensitivity indicators: (i) *sensitivity to climate change* (SCC) and (ii) *sensitivity to fishing pressure* (S_{FP}) presented in D4.1 (Engelhard et al. 2024). The approach is described in a paper published since then (Polo et al. 2025) and allows a 'scoring' of Europe's marine species according to their sensitivities to these two pressures.

The second step has been to scale up from species-level, to assess *community-level* sensitivities and risks to European marine biodiversity (Deliverable D4.2, Rozemeijer et al. 2025). Community-level sensitivity is calculated based on the weighted average sensitivities across all species that comprise a local marine community; this allows establishing where local 'community-level sensitivity hotspots' exist, as well as assessing changes in community-level sensitivity over time. Sensitivities, in combination with spatio-temporal variations in the *exposure to pressures* (either climate change or fishing pressure, or both) allow the estimation of community-level *risks* (to either pressure, or both). Thus, in addition to 'sensitivity hotspots', also 'risk hotspots' have been calculated for European marine life as part of D4.2 (Rozemeijer et al. 2025). This has been done for Europe's major marine regions (Mediterranean, North Sea, North East Atlantic, Icelandic and Greenlandic waters).

As part of D4.2, we have moreover assessed the sensitivity of *marine benthic habitats* to climate change and fishing pressure (three climatic stressors, i.e. temperature rise, ocean acidification and hypoxia; and five main fishing gear stressors). Finally, D4.2 has assessed the risks posed by a major threat to the Mediterranean Sea: that of non-indigenous species (NIS), with focus on the infamous 'Lessepsian' species (those that have entered the Mediterranean via the Suez Canal).

The present, third deliverable of WP4, then, builds on D4.1 and D4.2. It has the following, overarching aims:

(1) To identify important biodiversity 'hotspots' in European regional seas (based on either local species richness, or high community-sensitivity, or high community-level risk), and then
(2) Assess to what extent hotspots either are or are not appropriately protected – either through marine protected areas (MPAs, which is preferred) or through fishery-restricted areas (FRAs, noting that the level of protection is generally lower).
(3) The study will also discuss where there are clear 'mismatches' between biodiversity, sensitivity or risk hotspots and protection or conservation measures, and provide policy recommendations.

The report aims to achieve these objectives for marine fish and invertebrate communities in a broad swathe of European regional seas, ranging from Icelandic waters and the North Sea, Celtic Seas, Bay of Biscay through to the Mediterranean Sea. In combination, it is hoped that these approaches are of relevance to the management and conservation of European marine biodiversity risks to climate change and anthropogenic pressures.

Contents

1 General introduction

1.1 Background

Europe's marine biodiversity faces multiple pressures arising from climate change, fisheries, habitat loss and pollution (Burrows et al. 2011; Poloczanska et al. 2013). Halting biodiversity loss requires robust scientific advice to guide management plans and assess conservation measures, especially the design and placement of marine protected areas (MPAs). Decision-support tools are required to help safeguard 'biodiversity hotspots' and vulnerable ecosystems, while safeguarding key ecosystem services, such as food provisioning and climate regulation.

The Horizon project B-USEFUL develops user-oriented decision-support tools to help conserve marine biodiversity, building on existing European data and governance frameworks. In doing so, it contributes to the policy goals of the European Green Deal and EU Biodiversity Strategy 2030. It also contributes to aligned national policies, including the UK's National Biodiversity Strategy and Action Plan (NBSAP), and for Icelandic and Greenlandic waters, the Nordic Biodiversity Framework.

WP4 identifies habitats and species at risk in sensitive ecosystems by developing a hierarchical risk-based framework, adapted from the climate-risk approach developed by the Intergovernmental Panel on Climate Change (IPCC 2014). The three deliverables of WP4 are to:

(1) Identify species- or habitat-level sensitivities or risk in different European regional seas (D4.1; Engelhard et al. 2024);
(2) Assess trends and patterns in community-level sensitivities and risk (D4.2; Rozemeijer et al. 2025); and
(3) Evaluate spatial overlap or mismatch between biodiversity hotspots – areas of locally high species richness, community sensitivity or risk – and existing marine protected (MPA) or otherwise spatially managed areas.

This report (D4.3) addresses the third objective: the spatial match or mismatch between biodiversity hotspots and current marine protection, and its implications for biodiversity management.

1.2 Aim of this deliverable

MPAs are central to European and international strategies to conserve marine biodiversity and mitigate ongoing loss. The European MPA network aims to safeguard sensitive habitats and species, while supporting delivery of key ecosystem services (European Commission 2023; European Environment Agency 2024; Defra 2024). MPAs are designed to offer spatial protection, targeting threatened or declining features; ensuring habitat and species representativity; and promoting connectivity and resilience to deliver conservation benefits (Chaniotis et al. 2018). Nevertheless, it is often unclear how well current MPAs overlap with areas of highest biodiversity, ecological sensitivity, or risk for marine organisms, including bottom-living fish and invertebrates. It may also be unclear whether other spatial measures (e.g. fishery-restricted areas) align with these 'biodiversity hotspots'. Clarifying this alignment is important for assessing network performance and identifying where expanded or strengthened protection could deliver substantial biodiversity gains.

Throughout WP4 the emphasis is on two dominant pressures on marine ecosystems: (1) anthropogenic climate change; and (2) fishing pressure (and associated seabed disturbance). Together, these pressures are widely recognised as primary drivers of biodiversity loss globally across the world's oceans and in Europe's regional seas (Rijnsdorp et al. 2010, Poloczanska et al. 2013, Kroodsma et al. 2018). Biodiversity loss, in turn, strongly affects ocean ecosystem functions and services (Worm et al. 2006), including supporting and regulating services and the provisioning of sustainable seafood (Jennings et al. 2016). Many studies demonstrate recent and ongoing impacts of climate change and fishing pressure in European waters, including changes in species composition (Hiddink et al. 2006, 2008, McHugh et al. 2010, Receveur et al. 2024) and shifts in abundance and distributions of many fish species (Perry et al. 2005, Engelhard et al. 2011, 2014, Azzurro et al. 2019, Baudron et al. 2020), including elasmobranchs (skates and sharks: Sguotti et al. 2016, Fortibuoni et al. 2017, Chatzimentor et al. 2022). These drivers have also affected other marine life (e.g. benthic invertebrates: Greenstreet et al. 2007, Hiddink et al. 2015, Beauchard et al. 2023, cephalopods: van der Kooij et al. 2016, Oesterwind et al. 2022, seabirds: Davies et al. 2013). In many cases, the relative contributions of climate change and fishing remain poorly known (Rijnsdorp et al. 2009, Gissi et al. 2021). Hence, better understanding the risks imposed by these drivers is important for biodiversity conservation and sustainable resource management (Gissi et al. 2021, IPCC 2022).

In B-USEFUL WP4 biological trait-based approaches are used to assess (i) the *sensitivities* of marine species, habitats, and communities to climate change and fishing pressure; this was the central theme of the first deliverable report (D4.1: Engelhard et al. 2024). For the second deliverable report, information about the sensitivities of marine life was then combined with (ii) the levels of *exposure to the two pressures* posed by climate change and fishing across European regional seas. This combination of *sensitivity* and levels of *exposure to pressure* allowed us to assess (iii) the *risks* (or *vulnerabilities*) that marine communities are subject to – as well as the spatial and temporal variations in the levels of sensitivity, pressure or risk.

In this third deliverable report, we build further on D4.1 and D4.2 and ask:

(3) Where are the most important 'biodiversity hotspots' within different regional seas, and for a given study group – based on metrics including species richness, community-level sensitivity, and community-level risk to climate change and fishing pressure?

(4) To what extent are the most important 'hotspots' protected – through MPAs and/or fishery-restricted areas, that may vary in the level of protection offered (from 'fully protected' to offering very little protection to the particular animal group examined)?

1.3 Overview of the report's structure

In Chapter 2, we describe the approach followed throughout this report to categorising MPAs according to the level of protection afforded, harmonised across European regional seas. Building on key studies by Aminian-Biquet et al. (2024a, 2024b) but now extending this beyond EU waters to include non-EU waters, we distinguish between fully, highly, lightly and minimally protected MPAs, and levels of protection below that.

In Chapter 3 we identify biodiversity and sensitivity hotspots in the Mediterranean Sea and assess their match or mismatch with MPAs and/or fishery-restricted areas.

Chapter 4 applies the same methodology to the North-East Atlantic (Greater North Sea, Celtic Seas, and Bay of Biscay & Portuguese Coast ecoregions), looking at sensitivity and risk hotspots to climate change and fishing pressure and their overlap with MPAs.

In Chapter 5, we cover Icelandic waters and look at hotspots of species richness, sensitivity and risk, as well as of sensitive Arctic species, and to what extent these overlap with protected and/or fishery-restricted areas.

Chapter 6 covers epibenthic invertebrate communities in the North Sea, and shows that hotspots of sensitivity or vulnerability may not be static but can move over time – with implications for their overlap with MPAs (which in many cases were found to provide relatively limited protection to vulnerable epibenthic communities).

Chapter 7 focuses on sensitive benthic habitats in Portuguese continental waters, and demonstrates how climate change risk can be integrated into benthic habitat conservation planning.

We then show, in Chapter 8, how non-indigenous species (NIS) in the Eastern Mediterranean – in particular those that entered through the Suez Canal ('Lessepsian' species') – are increasingly entering MPAs, as well as vulnerable nursery habitats for native foodfish species, calling for management action to be taken.

The report closes with Chapter 9, providing an overview of the key messages emerging from the preceding chapters, and a consideration of initial management implications and wider perspectives.

2 Categorising MPAs, harmonised across European seas

2.1 Introduction

MPAs are a central instrument in global and European strategies to halt marine biodiversity loss and mitigate cumulative human pressures on ocean ecosystems (IPBES 2019). While coverage targets for MPAs have expanded rapidly, assessing whether protected areas are located where biodiversity values are highest, and whether they offer meaningful protection, remains challenging (Zhao & Costello 2026). This deliverable (D4.3) specifically examines the spatial match and/or mismatch between biodiversity hotspots and existing marine protection. This implied that it was essential to rely on clear and operational definitions of MPA protection. In the European context, this requirement is particularly pressing because MPAs are designated under a mosaic of legal frameworks and governance systems. These span multiple European regional seas and encompass both EU and non-EU countries, notably the United Kingdom (Aminian-Biquet et al. 2024b). Ensuring consistency in how protection levels are defined and interpreted across these jurisdictions is a prerequisite for robust large-scale spatial analyses and for meaningful comparisons of conservation performance (Grorud-Colvert et al. 2021).

To address this challenge, we build on recent advances in the classification of MPAs that explicitly link regulatory provisions to expected biodiversity outcomes. More specifically, we draw on the framework developed in *The MPA Guide* (Grorud-Colvert et al. 2021), which distinguishes MPAs according to both their stage of establishment and their level of protection, based on the type, intensity, and impact of activities permitted within their boundaries. This approach complements existing categorisations, such as IUCN protected area categories, by focusing on what MPAs do in practice rather than solely on stated management objectives. By classifying MPAs along a gradient from minimally protected to fully protected, the framework provides a transparent and transferable basis for assessing how different regulatory regimes are likely to contribute to biodiversity conservation, even in data-limited contexts (Grorud-Colvert et al. 2021; Aminian-Biquet et al. 2024b).

Methodologically, our aim was therefore twofold. First, we harmonised existing classifications of MPA protection levels derived from an impact-based framework that is applicable across diverse governance and legal settings, rather than generating a new classification. Second, we integrated these previously compiled datasets consistently across European regional seas, combining information for MPAs designated by EU member states under EU and regional conventions with equivalent data for non-EU countries (notably the UK). By doing so, we established a coherent and spatially comprehensive foundation for subsequent analyses of how current marine protection aligns with patterns of marine biodiversity and ecological risk, and for evaluating whether existing MPA networks are positioned to contribute effectively to emerging conservation targets across Europe.

2.2 Methods

This assessment builds on harmonised datasets describing the regulation of human activities within MPAs across European seas. Core information on MPA regulations and associated protection levels was derived from the dataset developed by Aminian-Biquet et al. (2024a), which applies the protection-level framework of The MPA Guide at large spatial scale. The resulting dataset comprises 4,858 MPAs across EU waters, providing spatially explicit information on permitted activities and their potential ecological impacts. Additional data for UK waters (391 MPAs) were obtained from an MSc thesis completed recently (Sim 2025) that applied the same MPA Guide methodology to UK MPAs, ensuring methodological consistency across European regional seas.

To account for uncertainty in how regulated activities may be implemented and enforced, two alternative scenarios of potential impact were originally computed by Aminian-Biquet et al. (2024a). In the present study, we retained their "Scenario 1", which represents the lowest plausible range of impacts for each regulated activity, and is considered a conservative approach for estimating protection levels because it assumes the least impactful interpretation of allowed uses in the absence of detailed information. Scenario 1 typically yields higher protection level classifications (more areas as fully, highly, or lightly protected) compared with Scenario 2, which assumes the highest plausible impacts and often classifies more MPAs as minimally protected or incompatible with conservation objectives. Aminian-Biquet et al. (2024b) showed that, when protection levels from Scenario 1 are compared with expert-based assessments, Scenario 1 tends to describe higher protection levels than Scenario 2, indicating that it represents a cautious estimate of potential biodiversity protection in data-limited contexts.

Protection levels were defined following the MPA Guide categories:

- ***Fully protected***, where no extractive or destructive activities are permitted;
- ***Highly protected***, where only limited, low-impact extractive activities may occur;
- ***Lightly protected***, where moderate extractive and other potentially impactful activities are allowed;
- ***Minimally protected***, where extensive extraction is permitted but the area still meets the minimum criteria of a marine protected area;
- ***Incompatible** with conservation objectives*, where one or more highly impactful activities are allowed that undermine biodiversity conservation; and
- ***Unclassified***, where insufficient or unclear information on permitted activities prevented assignment to a protection level (Grorud-Colvert et al. 2021) (*Figure 2-1*).

Protection levels were assessed based on the potential impacts of seven main categories of human activities, as defined in the MPA Guide framework:

i. Mining and oil and gas extraction
ii. Dredging and dumping
iii. Anchoring
iv. Coastal and offshore infrastructure
v. Aquaculture

vi. Fishing activities, including subsistence, commercial, and recreational fishing and the use of different gear types
vii. Non-extractive activities, such as tourism, recreational use, and cultural or traditional practices

For each activity, impact was evaluated based on its *type, intensity, spatial extent, duration,* and *frequency* relative to biodiversity conservation objectives. Activities were then classified along an impact gradient ranging from negligible to incompatible with conservation.

Several dataset extensions were implemented to improve spatial completeness. A large offshore MPA covering Greenlandic waters was added using a shapefile obtained from the World Database on Protected Areas (UNEP-WCMC and IUCN 2025). In addition, MPAs located in Icelandic waters, and one nationally designated MPA for Albania were added based on the most recent spatial information available from the same source.

Where spatial units overlapped and multiple protection levels applied to the same area, the highest level of protection was retained. This followed the hierarchy: unclassified < incompatible < minimally protected < lightly protected < highly protected < fully protected. Areas classified as unclassified or incompatible with conservation objectives were subsequently grouped, and for analytical purposes, treated as 'unprotected'.

For the Mediterranean Sea, *Fisheries Restricted Areas (FRAs)* were also included (data available at https://www.fao.org/gfcm/data/maps/fras/fr/). These areas represent other area-based management tools for which biodiversity conservation is not the primary objective and therefore they do not qualify as MPAs. However, they may confer incidental conservation benefits and, where they effectively contribute to biodiversity conservation, could qualify as *Other Effective area-based Conservation Measures (OECMs)*. These areas were analysed separately from MPAs, but included in spatial overlays to assess their potential contribution to reducing biodiversity risk.

MPAs in Norwegian waters were not included in this analysis. While comparable national efforts to categorise protection levels using approaches consistent with the MPA Guide are nearing completion for Norway, these data were not yet publicly available and could not be incorporated at the time of this assessment.

Finally, it is important to note that MPA datasets and associated protection-level classifications are dynamic. New protected areas are regularly designated; existing boundaries and regulations may be amended; and protection levels may be strengthened or weakened over time. Despite these limitations, the dataset used here represents the most comprehensive and up-to-date information currently available for assessing spatial patterns of marine protection across European regional seas.

The majority of the mapped MPA surface is classed as 'unclassified' (54.1%) or 'incompatible with conservation objectives' (23.9%), together accounting for 78.0% of the total area, while 'minimally' and 'lightly protected' MPAs represent 15.6% and 4.7% (20.3% combined). Areas under high or full protection remain limited (1.2% 'fully protected' and 0.5% 'highly protected') (*Figure 2-1*).

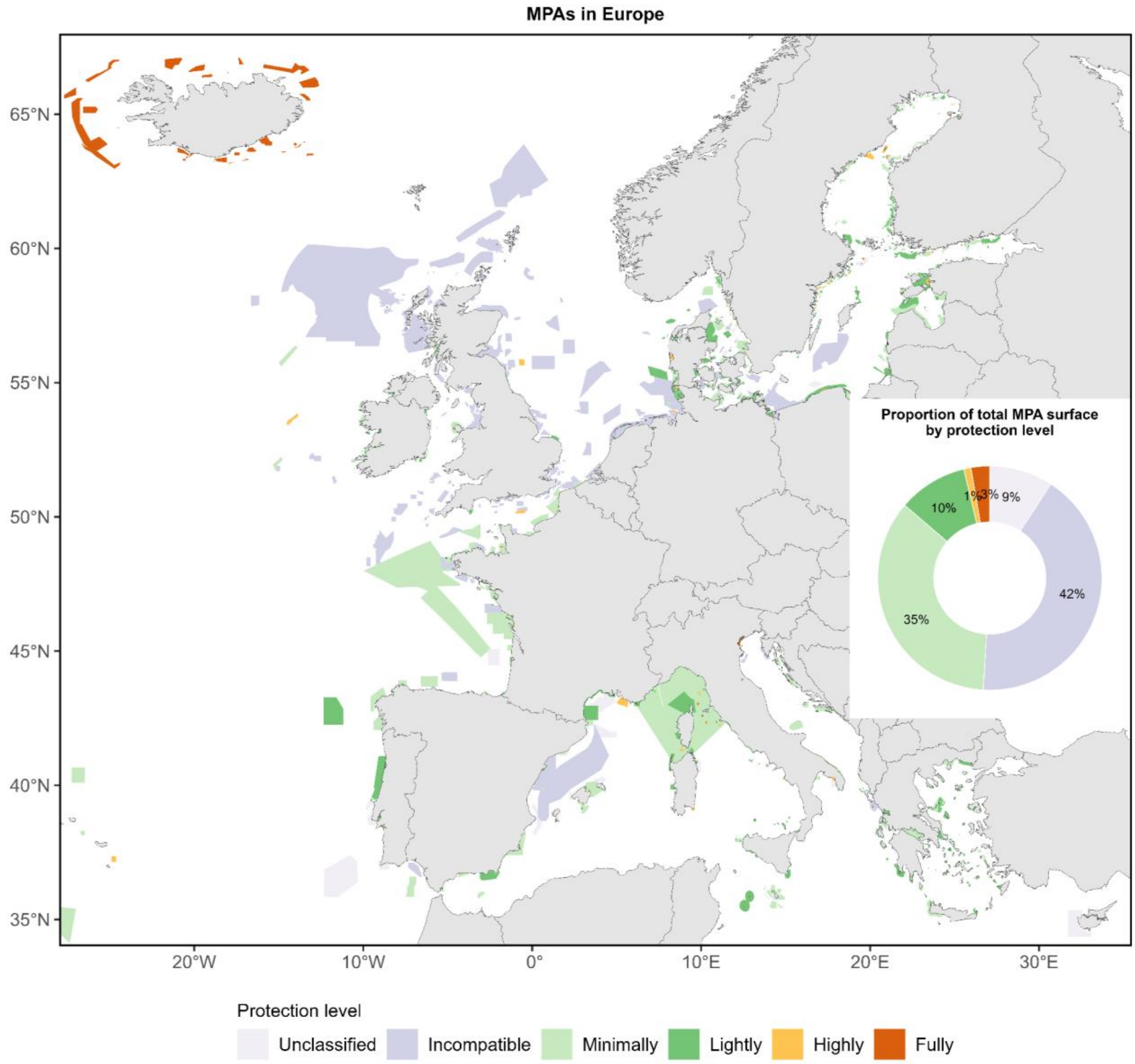

Figure 2-1 Spatial distribution of marine protected areas (MPAs) in Europe by protection level, with proportional surface summary. Fisheries Restricted Areas (FRAs) are not represented here. For clarity, Greenland is also not represented on the map. The left panel maps MPA polygons across the Baltic Sea, Mediterranean Sea, and NE Atlantic (including UK waters), classified into six protection categories: unclassified, incompatible, minimally protected, lightly protected, highly protected, and fully protected. The circular chart shows the proportion of total mapped MPA surface represented by each category, using the same colour scheme as the map legend.

2.3 Discussion

By harmonising protection-level classifications across EU and non-EU waters using a consistent, impact-based framework, this work provides a robust foundation for evaluating how current spatial management aligns with biodiversity sensitivity and risk patterns identified in B-USEFUL deliverable D4.3. The approach allows differences in legal designations and governance systems to be translated into a common metric of effective protection, facilitating large-scale spatial comparisons across European regional seas.

While data gaps remain, notably for Norwegian waters, and protection levels may not fully capture on-the-ground enforcement or compliance, the classification represents the best available approximation of effective protection based on current regulatory frameworks. As such, these results should be interpreted as baseline assessment, suitable for identifying broad spatial patterns, informing scenario analyses, and supporting strategic discussions on where strengthened or expanded protection could most effectively reduce biodiversity risk under ongoing climate change and fishing pressure.

2.4 Data availability

The spatial datasets used and processed in this study, including the classified MPA shapefiles and associated protection-level attributes, are available via the following link: https://github.com/MoullecF/EU_MPA_B_USEFUL. These data build on publicly available sources and harmonised classifications and are intended to support transparency, reproducibility, and future comparative analyses.

3 Mediterranean Sea

3.1 Introduction

MPAs represent a traditional tool for protecting marine biodiversity, aiming to safeguard species and habitats, and to conserve ecosystems and marine biodiversity while regulating human activities (Fraschetti et al. 2022). The Mediterranean Basin is recognised as a biodiversity hotspot (Myers et al. 2000), supporting high levels of marine life including roughly 17,000 species, with a notable proportion of endemism found nowhere else on Earth (Coll et al. 2010).

MPAs encompass a wide spectrum of objectives, governance arrangement and levels of protection. These range from multiple-use areas that may permit low-impact activities, such as artisanal fishing and ecotourism, to fully protected or no-take zones, where extractive activities are prohibited (Boubekri et al. 2026, Andradi-Brown et al. 2023), providing tools to protect nurseries, benthic communities and biodiversity hotspots, and reinforcing the role of area-based measures within wider management portfolios (Bastardie et al. 2025). This variability in governance or protection levels implies that at the basin scale, the designation of MPAs has not translated into homogeneous reductions of human pressures, as a large share of nominally protected waters remains weakly regulated or only marginally different from surrounding areas (Claudet et al. 2020, Boubekri et al. 2026). This heterogeneity reflects the fact that key policy instruments provide broad definitions and conservation commitments, and that there is no international regulatory standard for protection categories and allowed activities. Although frameworks such as the Convention on Biological Diversity and the Barcelona Convention promote marine conservation, they do not strictly define protection categories or allowed activities. In combination, this has resulted in heterogeneous protection levels across the Mediterranean Basin (Claudet et al. 2020).

The wide diversity of protection instruments adopted in the Mediterranean context reflects the basin's complex socio-ecological context (Fraschetti et al. 2018). According to the MAPAMED database (Hogg et al. 2025), which compiles official designations of marine protected sites in the Mediterranean Sea under the Barcelona Convention, no less than 1,126 MPAs within the Mediterranean basin have been designated up to 2019. However, only a small portion is actively managed (44%), while many others lack full management or a high level of protection. The status of management effectiveness varies considerably across the region, with ongoing efforts to improve conservation outcomes, regulation of fishing activities, and implementation of management plans to meet international targets such as the EU Biodiversity Strategy 2030, aimed at legally protecting 30% of our seas, of which 10% should be strictly protected by 2030 (EU 2020).

In practice, the MPA definition and designation are grounded on jointly evaluating ecological value and exposure to human uses. The former includes the presence of high biodiversity and vulnerable habitats, such as seagrass meadows, submarine canyons, cold-water coral reefs, and essential fish habitats; the latter include human pressures acting on these, such as fishing activities, maritime traffic, anchoring, pollution, and coastal development. All of this contributes to cumulative impacts that increase ecological risk and motivate spatial protection and zoning decisions (Murray et al. 2025). As a result, Mediterranean MPAs often aim not only to conserve biodiversity, but also to mitigate specific anthropogenic pressures and manage trade-offs between conservation and sustainable use (Abdulla et al. 2009).

Within this broader framework, Other Effective Area-Based Conservation Measures (OECMs) have gained increasing relevance. OECMs are spatially defined areas that deliver effective in-situ conservation of biodiversity (CBD 2018). Although OECMs are not formally classified as MPAs, they are recognised as a complementary type of area-based conservation measure counting toward global and regional commitment to protect at least 30% of marine areas by 2030 (the "30×30" biodiversity target; Estradivari et al. 2022). In the Mediterranean basin, Fisheries Restricted Areas (FRAs) established under the General Fisheries Commission for the Mediterranean (GFCM) can complement MPAs by addressing sector-specific pressures, particularly bottom-contact fishing, over large offshore and deep-sea areas that are often underrepresented in traditional MPA networks (Abdulla et al. 2009, Gabrié et al. 2012).

Despite these conservation tools, the Mediterranean Sea faces multiple and combined pressures, including rapid warming, intensive fishing and habitat disturbances; these can affect species' survival, growth, reproduction, and overall ecosystem functioning (Crozier & Hutchings 2014, Collie et al. 2017, Hidalgo et al. 2022). To strategically align conservation challenges to protection areas, researchers increasingly apply the concept of ecological risk, which considers both intrinsic sensitivity of species and communities, determined by life-history and ecological traits (Polo et al. 2025), and their exposure to environmental and/or anthropogenic stressors (Rozemeijer et al. 2025). This approach could be very helpful in identifying areas where conservation interventions are most urgently needed.

The Mediterranean Sea presents heterogeneous hotspots of ecological risk, as sensitivity and exposure vary across the basin (Chapter 3 in Rozemeijer et al. 2025). Climate change risk (R_{CC}) and fishing pressure risk (R_{FP}) are particularly high where sensitive communities overlap with where these threats are most present (Hidalgo et al. 2022, Chatzimentor et al. 2023, FAO 2023). However, it is unclear to what extent the Mediterranean MPA network effectively covers basin-scale hotspots of biodiversity and ecological risk, also considering that most designated MPA waters are only weakly protected and thus unlikely to substantially reduce key pressures (Coll et al. 2012, Claudet et al. 2020). Consequently, analysing and mapping these sensitivity and risk hotspots, as well as biodiversity distribution and hotspots across the basin, is very informative to identify where biodiversity is most at risk, and conservation tools (e.g., MPAs, FRAs) can optimise the effectiveness of spatial protection measures.

Therefore, this study aims to:

(1) Quantify the spatial overlap between basin-scale hotspots for demersal species (including fish, cephalopods, crustaceans) and the current protection network, considering both MPAs alone and the configuration including FRAs.
(2) Identify and discuss systematic mismatches, where hotspot footprints for demersal species remain largely outside effectively protected areas.
(3) Prioritise candidate areas of high conservation value through spatial prioritisation (Zonation).

3.2 Methods

Data on the presence and abundance of demersal species (fish, cephalopods, and crustaceans) in the Mediterranean Sea were obtained from the Mediterranean International Bottom Trawl Survey (MEDITS) conducted between 1999 and 2021. Subsequent trait analysis was carried out according to Rozemeijer et al. (2025). The spatial congruence between ecologically relevant hotspots and marine protected areas (MPAs) was quantified through a spatial overlap analysis carried out within a harmonised, spatial framework across the Mediterranean study domain (which excluded the southernmost parts of the basin). The Mediterranean Sea wide-scale biodiversity analyses relied on the species' spatio–temporal distributions estimated by means of HMSC (Hierarchical Modelling of Species Communities), developed as part of Deliverable 3.1 (Lindegren et al. 2025). Biodiversity hotspots were represented as polygon layers depicting spatial persistence of selected biodiversity indices (e.g., species richness, Shannon diversity) estimated in the context of Deliverable 3.2 (Hidalgo et al. 2026). Additionally, in the frame of Deliverable 4.2 (Rozemeijer et al. 2025), using a trait-based approach, two different community-level sensitivity layers were estimated, describing sensitivity to fishing pressure (S_{FP}) and to climate change (S_{CC}) (Polo et al. 2025). These sensitivity layers were used, in combination with sea surface temperature (SST) and fishing pressure (FP) exposure information, in order to respectively estimate community-level risk layers for fishing pressure (R_{FP}) and for climate change (R_{CC}).

The MPAs' level of protection was represented through polygon layers classified according to the MPA Guide framework (Grorud-Colvert et al. 2021, Aminian-Biquet et al. 2024a, Aminian-Biquet et al. 2024b), as described in Chapter 2. Specifically, MPA zones were aggregated into four protection levels (fully, highly, lightly, and minimally protected), and, where spatial overlaps occurred among zones, a priority rule was applied to assign a single protection class to each overlapping area (fully > highly > lightly > minimally). Two further categories were used and interpreted as not effectively protected, unclassified, and incompatible, accounting respectively for data-deficient areas and for areas where the permitted activities include impacts classified as incompatible with the conservation of nature under the MPA Guide framework. For the purposes of this analysis, GFCM FRAs were treated as OECMs, because they are spatially defined measures primarily targeting fisheries pressures and can contribute to *in-situ* biodiversity conservation outcomes. FRAs were integrated into the protection categories used for MPAs, following the same classification as for MPAs (details in Table A-1). In both protection configurations, all layers were analysed in terms of intersected surface area reprojected to a metric coordinate system, allowing direct comparison across hotspot types and protection configurations. The overlap analysis was aimed at assessing the spatial overlap between hotspots and the potential regulatory level of protection areas, rather than the realised effectiveness of protection.

The spatial overlap analysis was conducted across the Mediterranean Sea using a regular spatial grid covering the whole study domain, from 0 to 1000 m depth, and areas for which the biological information is available, owing to the MEDITS scientific trawl surveys. The grid consisted of square polygons of 0.1° of resolution (~10x10 km). The analysis was implemented under two alternative configurations: (1) an MPA-only scenario and (2) an extended scenario accounting also for the GFCM Fishery Restricted Areas (FRAs) across the Mediterranean Seas, and other relevant spatial management measures. Before the overlay operation, all polygon layers were validated and geometrically cleaned to remove topology artefacts introduced by clipping, dissolving, and overlaps, ensuring robust spatial intersections. Firstly, the intersection between hotspots polygons and grid cells was computed to quantify the hotspot area per cell; then, for each protection level, the intersection between the hotspot portion of each cell and the corresponding disjoint MPA layer was calculated. Ultimately, the areas were aggregated per cell and expressed in km^2. All analyses were performed in R, using integrated GIS workflows for spatial data handling and analysis.

A spatial conservation prioritisation analysis was conducted to rank Mediterranean areas according to their relative conservation value, and to highlight areas where current protection measures (MPAs, and MPAs + FRAs) do not align with biodiversity, sensitivity, and risk hotspots. The analysis was conducted using Zonation5 software, a decision-support framework for spatial conservation prioritisation that ranks all planning units (here, grid cells) from highest to lowest conservation value, given a set of biodiversity features, optionally accounting for connectivity assumptions, and (where relevant) costs or pressures (Moilanen et al. 2024). Zonation generates a continuous priority ranking and enables performance to be assessed for any chosen retained fraction of the seascape (Moilanen et al. 2022, Moilanen et al. 2009). In the present application, Zonation was used to identify priority areas and, more specifically, to assess spatial mismatches between priority areas and existing conservation measures under two protection configurations considered. These protection configurations implied aiming for protection of top 30% of top 10% of conservation features.

The spatial conservation prioritisation analysis was run separately for the West and East Mediterranean, since the underlying species distribution models were developed independently for the two sub-basins. Biodiversity information was represented by the average gridded distributions of 151 species in common between the two models, considering the time series spanning from 2012 to 2021 complemented by community-level biodiversity indices (species richness, Shannon diversity, and Pielou's evenness) derived above. To broaden the ecological interpretation beyond biodiversity patterns alone, community-level sensitivity layers for fishing pressure and climate-change drivers were also included, together with derived risk layers. In addition, elasmobranch-focused layers were used to further emphasise the conservation of vulnerable taxa (IUCN 2012). Fishing effort was incorporated directly into prioritisation as a negative component (cost) to penalise areas characterised by higher fishing effort (Moilanen et al. 2022, Moilanen et al. 2009). Finally, a single-feature connectivity setting was applied to elicit spatially aggregated solutions, thereby reducing granularity in the spatial prioritisation results (Lehtomäki et al. 2013).

Five scenario configurations were analysed to contrast the mismatches between hotspots areas and MPA protection levels. **Scenario 1** combined biodiversity features with sensitivity layers; **scenario 2** combined biodiversity features with risk layers. **Scenario 3** integrated biodiversity, sensitivity, and risk simultaneously, further including fishing effort as a negative feature, allowing for moving from a purely ecological prioritisation towards a more management-oriented prioritisation in which the expected cost of reducing pressure is represented explicitly (Moilanen et al. 2009, Moilanen et al. 2022). Scenarios 4 and 5 retained the same feature set as Scenario 3 but introduced hierarchical masks to reflect conservation planning supporting the prioritisation according to the existing protection networks. Scenario 4 imposed the hierarchy of existing MPAs as a protection spatial constraint, while scenario 5 applied an analogous hierarchy but based on the combined network of MPAs and FRA. Scenarios were compared in R using performance indicators, as the mean proportion of features retained along the prioritisation ranking based on the overlap among the top 30% and the top 10% priority cells in each scenario. In addition, the mismatch between areas with higher biodiversity value and the actual protection network was assessed by identifying priority areas selected outside the existing MPA network.

3.3 Results

Within the analysis domain, the total protected area differed substantially between sub-basins (*Table 3-1*). In the East Mediterranean, protected MPA levels covered 32,939 km² (7.52%) out of 437,868 km² of the study domain, whereas in the West Mediterranean, they covered 66,942 km² (30.89%) out of 216,722 km². Fully and highly protected levels were negligible in both sub-basins (≤0.12% in the East and ≤1.29% in the West), while most protected area was assigned to minimally and lightly protected categories.

Table 3-1. Surface area (km²) and relative coverage (%) of the protection network within the analysis domain (0–1000 m) for the East and West Mediterranean sub-basins. Areas are reported by protection level (Fully, Highly, Lightly, Minimally), together with the unprotected areas (in which the categories Unclassified and Unprotected are also included).

	EAST		**WEST**	
Surface (km²)	**Surface (km²)**	**Percentage (%)**	**Surface (km²)**	**Percentage (%)**
Total area	**437868**	**100.00**	**216722**	**100.00**
Fully	155	0.04	104	0.05
Highly	542	0.12	2787	1.29
Lightly	12557	2.87	14565	6.72
Minimally	19686	4.50	49487	22.83
Total protected	**32939**	**7.52**	**66942**	**30.89**
Unclassified	2306	0.53	11853	5.47
Incompatible	3104	0.71	13853	6.39
Unprotected	399518	91.24	124076	57.25
Total unprotected	**404929**	**92.48**	**149781**	**69.11**

Across hotspot types, there was a marked spatial misalignment between basin-scale hotspots and areas that can be interpreted as effectively protected. Under the actual MPA network, the distribution of hotspot surface across the protection levels (*Table 3-2*, left side, *Figure 3-1* to *Figure 3-5*) underlined that all hotspot types are mostly in unprotected areas. This was particularly pronounced for both risk hotspot types, with R_{CC} hotspots mostly outside protection (94.8% unprotected), likewise R_{FP} (80.1%) and S_{FP} (83.5%) hotspots. Also for species richness and S_{CC}, with somewhat higher coverages than both risk layers R_{CC} and R_{FP}, most hotspot areas still fell outside protection (73.9% and 69.1% unprotected, respectively). By contrast, the hotspot surface within areas of highest protection level was only marginal: fully protected MPAs showed negligible overlap (or approximately 0% due to numerical rounding) with any hotspot layer. Highly protected MPAs contributed only marginally, with 1.7% overlap with hotspots for species richness, 1.6% for S_{CC}, and 0.1% for R_{CC}.

Table 3-2. The percentages surface area of different hotspot types, overlapping with different MPA protection levels, for the MPA-only scenario (left side) and MPA + FRA scenario (right side). Hotspot types are species richness, climate change risk (R_{CC}), fishing pressure risk (R_{FP}), climate change sensitivity (S_{CC}), and fishing pressure sensitivity (S_{FP}). Values in bold highlight the differences observed in the combined MPA + FRA scenario compared with the MPA-only scenario.

	MPAs					**MPAs + FRAs**				
Protection level	**Species Richness**	**S_{CC}**	**S_{FP}**	**R_{CC}**	**R_{FP}**	**Species Richness**	**S_{CC}**	**S_{FP}**	**R_{CC}**	**R_{FP}**
Fully	0.0	0.0	0.0	0	0	0	0	0	0	0
Highly	1.7	1.6	0.0	0.1	0	**2.0**	**1.7**	0.0	0.1	0
Lightly	9.5	7.7	2.4	1.2	7.0	9.5	7.7	2.4	**1.4**	**9.8**
Minimally	7.3	15.1	11.4	1.4	6.2	**7.4**	**17.3**	**11.7**	1.4	6.2
Unclassified	2.1	2.9	0.5	0.3	4.6	2.1	**2.5**	0.5	0.3	4.6
Incompatible	5.5	3.8	2.1	2.1	2.2	5.5	3.8	2.1	**2.2**	**2.7**
Unprotected	73.9	69.1	83.5	94.8	80.1	**73.5**	**67**	**83.2**	**94.5**	**76.7**

Hotspot coverage increased mainly through the lower protection categories. Under the MPA network, the combined contribution of lightly + minimally protected MPAs accounted for 9.5% and 7.3% of species richness hotspots, respectively, and 7.7% and 15.1% of S_{CC} hotspots, but remained substantially lower for risk hotspots, especially R_{CC} (only 1.2% and 1.4%, for lightly and minimally protected respectively). The other categories of protection (unclassified + incompatible) reached a total of 7.6% for species richness and 6.7% for S_{CC}, and were proportionally higher for some risk and sensitivity layers (for example, R_{CC}: 2.4%, R_{FP}: 6.8%) (*Table 3-2*).

The inclusion of FRAs produced only modest redistribution in hotspot coverage (i.e. MPA + FRA scenario; see right side). The proportion of unprotected hotspot surface decreased slightly for all layers compared to the MPA network alone but remained dominant across all hotspot types (> 67%). The largest absolute reduction was observed for R_{FP} (a decline of 3.4% in the unprotected fraction), whereas changes were minimal for R_{CC} (−0.3%) and S_{FP} (−0.3%). Importantly, this extended configuration did not alter the qualitative pattern. Full protection level remained absent, while highly protected areas contributed marginally, but slightly more than MPAs alone, with 2% for species richness, 1.7% for $S_{CC,}$ and 0.1% for $R_{CC.}$ Lightly and minimally protected MPAs showed moderate representation also in this case. Consequently, even when FRAs were considered, R_{CC} hotspots continued to show the weakest alignment with protection, and R_{CC} exhibited the weakest overlap with effective protection in both MPA and MPA + FRA scenarios.

The radar plots reported in *Figure A-1* in the appendix provide a graphical representation of the proportional distribution of hotspot surfaces across protection levels under both protection network configurations. All hotspot types were predominantly located within the unprotected category, which accounts for incompatible, unclassified, and unprotected categories. The fully protected category was almost not represented at all, and the highly protected category likewise contributed marginally, while lightly and minimally protected MPAs accounted for a limited but appreciable share.

A complementary perspective is provided by assessing the proportions of areas covered by different MPA protection levels that overlap with the surfaces for different hotspot types (*Table 3-3*). Concerning the MPA scenario (*Table 3-3*, left side), only a small fraction of fully protected MPA areas overlapped with hotspots (4.1% with species richness hotspots, 1% for S_{CC}, and 0.2 of R_{CC}). Hotspot-intersecting MPAs were largely composed of highly and lightly protected areas, particularly for species richness and S_{CC} (e.g., species richness: 29.9% of highly and 20.1% of lightly protected MPAs; S_{CC}: 27.2% of highly and 16.2% of lightly protected MPAs). Nevertheless, a large share of hotspot-intersecting areas fell into incompatible and unclassified strata, with the incompatible strata at a higher proportion than the unprotected strata. This showed that a non-negligible component of the intersection occurs in zones that cannot be interpreted as delivering strong biodiversity protection, either because protection level cannot be confidently assigned or because activities are classified as incompatible with conservation objectives. When FRAs were added (*Table 3-3*, right panel), the internal composition of hotspot-intersecting areas shifted only slightly. For highly protected MPAs, the proportion overlapping with species richness hotspots and with S_{CC} hotspots decreased relative to MPA-only (overlap with species richness hotpots changing from 29.9% to 16.3%; overlap with S_{CC} changing from 27.2% to 13.4%), while for lightly protected and incompatible this proportion remained broadly comparable.

Table 3-3. The percentages of MPAs of different protection levels that overlap with different hotspot types, for the MPA-only scenario (left side) and MPA + FRA scenario (right side). Hotspot types are species richness, climate change risk (R_{CC}), fishing pressure risk (R_{FP}), climate change sensitivity (S_{CC}), and fishing pressure sensitivity (S_{FP}).

	MPAs					MPAs + FRAs				
Protection level	**Species Richness**	**S_{CC}**	**S_{FP}**	**R_{CC}**	**R_{FP}**	**Species Richness**	**S_{CC}**	**S_{FP}**	**R_{CC}**	**R_{FP}**
Fully	4.1	1.0	0.0	0.2	0.0	4.1	1.0	0.0	0.2	0.0
Highly	29.9	27.2	0.2	1.6	0.1	16.3	13.4	0.1	0.8	0.0
Lightly	20.1	16.2	5.4	1.6	9.4	18.1	14.6	4.8	1.7	11.8
Minimally	6.1	12.5	9.9	0.8	3.2	5.9	13.8	9.7	0.7	3.1
Unclassified	8.7	11.6	2.1	0.9	11.8	8.8	10.4	2.1	0.9	12.0
Incompatible	18.8	12.7	7.6	4.7	4.6	18.2	12.3	7.4	4.7	5.6
Unprotected	8.1	7.6	9.6	6.7	5.6	8.3	7.5	9.7	6.8	5.4

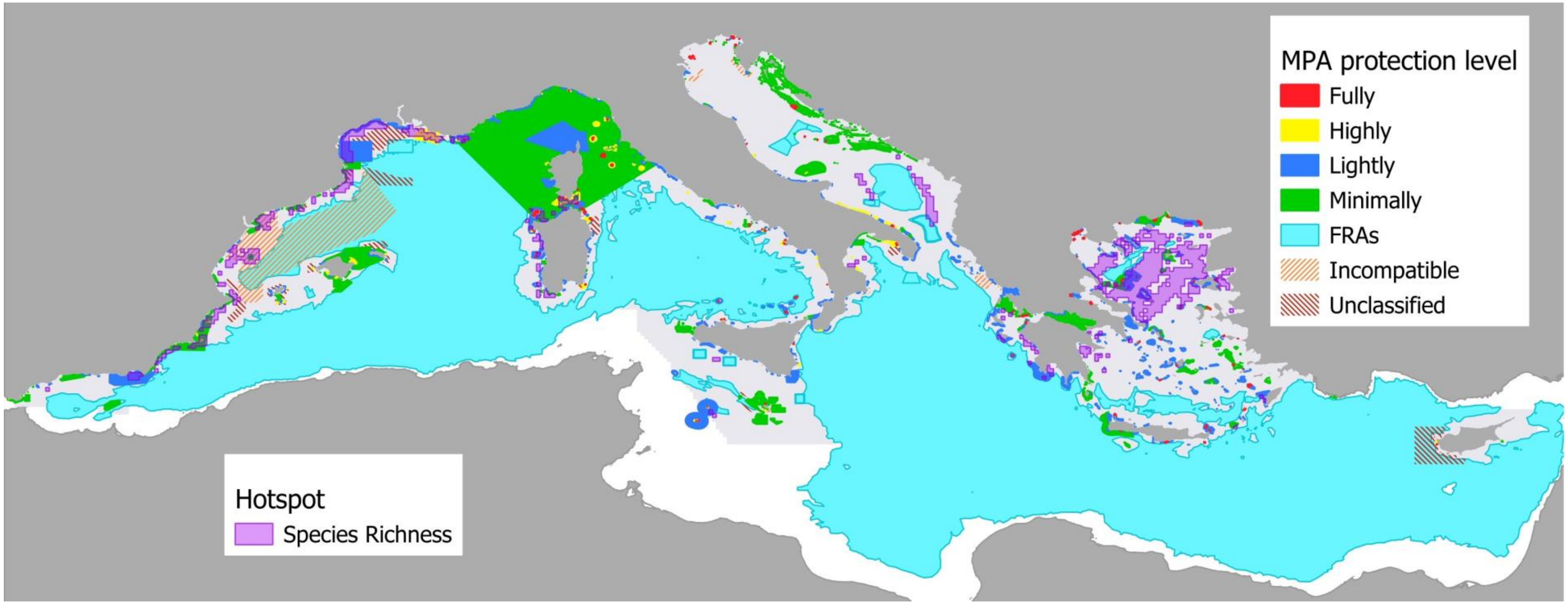

Figure 3-1. Map showing the overlap of hotspots of species richness with the MPA network and FRAs in the Mediterranean Sea.

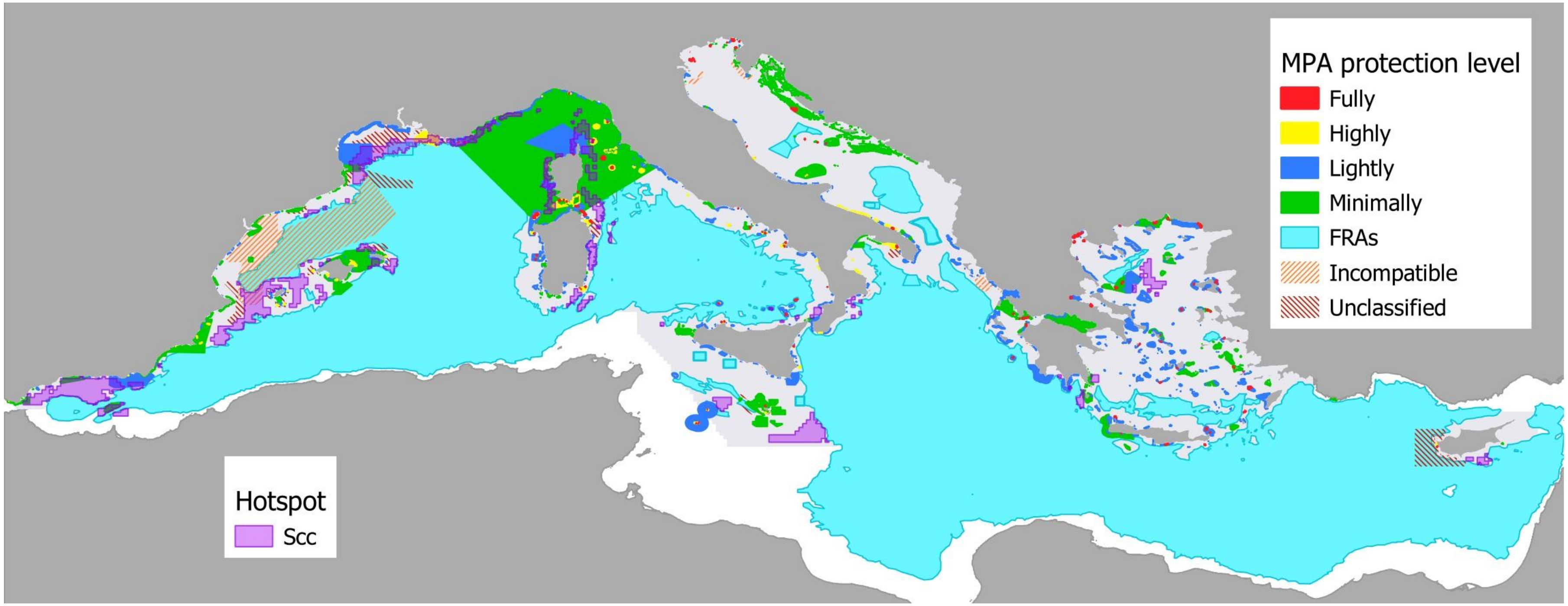

Figure 3-2. Map showing the overlap of hotspots of sensitivity to climate change (S_{CC}) with the MPA network and FRAs in the Mediterranean Sea.

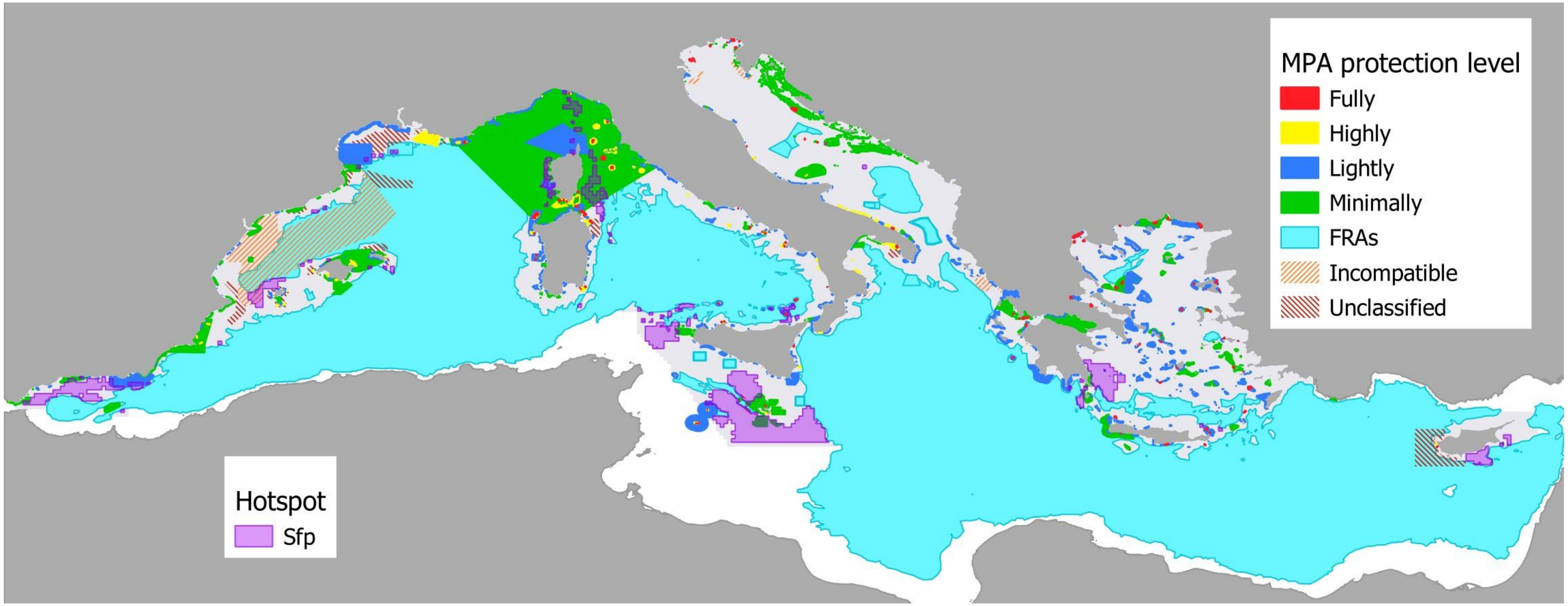

Figure 3-3. Map showing the overlap of hotspots of sensitivity to fishing pressure (S_{FP}) with the MPA network and FRAs in the Mediterranean Sea.

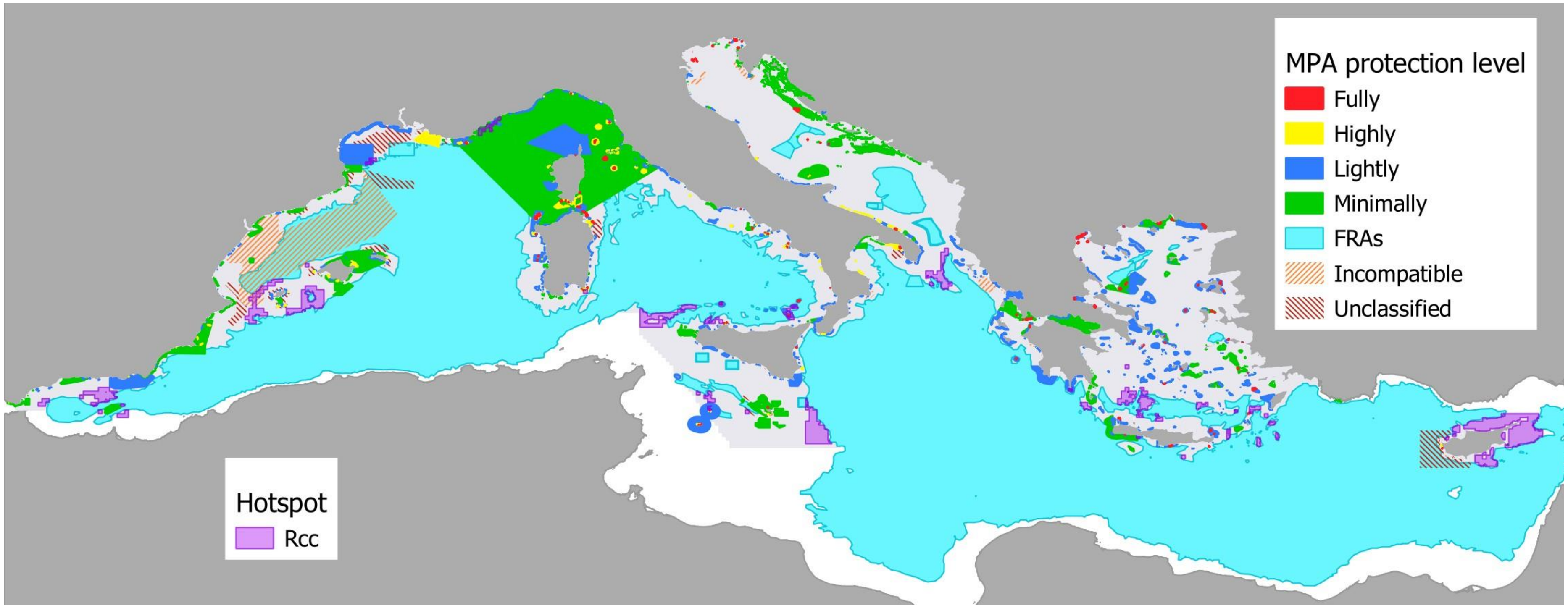

Figure 3-4. Map showing the overlap of hotspots of risk to climate change (R_{CC}) with the MPA network and FRAs in the Mediterranean Sea.

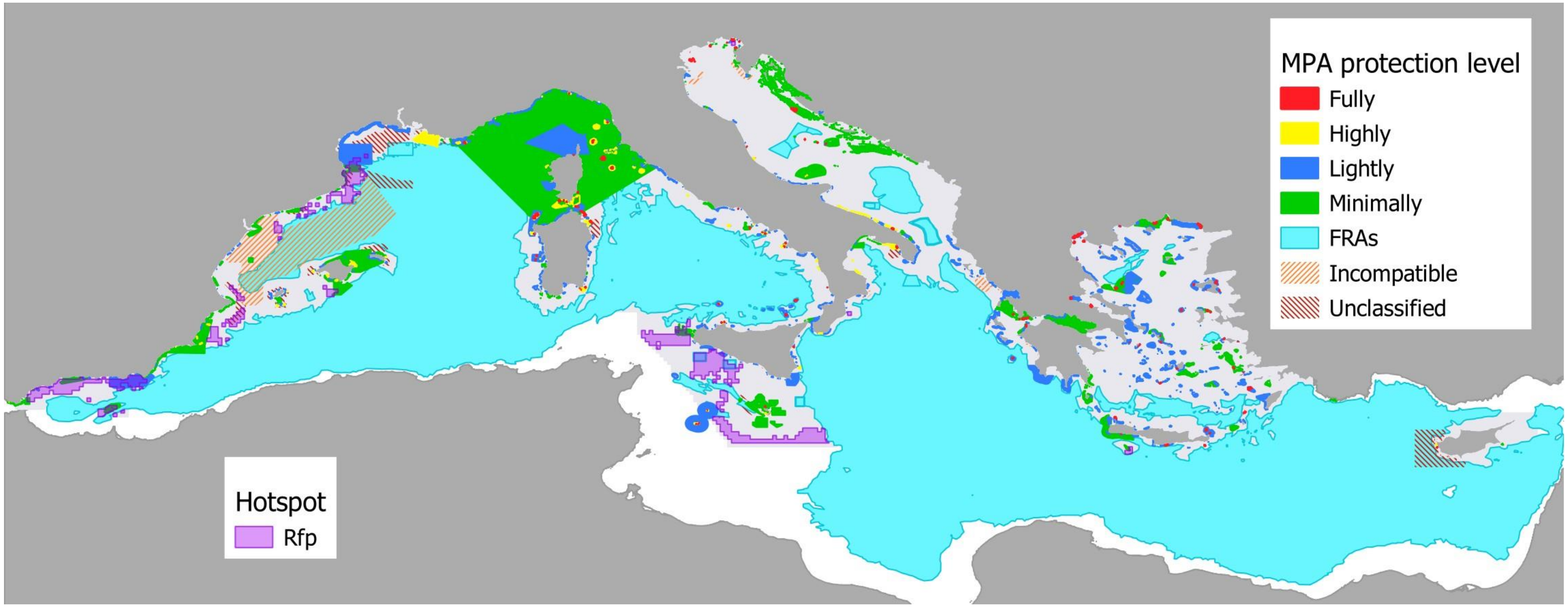

Figure 3-5. Map showing the overlap of hotspots of risk to fishing pressure (R_{FP}) with the MPA network and FRAs in the Mediterranean Sea.

The Zonation prioritisation generated consistent priority rankings for both the East and West Mediterranean basins. This allowed a direct comparison of scenario behaviour using summary statistics from the Zonation runs, performance index summaries at two retained-fraction thresholds, and spatial agreement among top-priority selections.

A clear and consistent pattern emerged from the area under the curve (AUC) reported for each scenario (*Table 3.-4*), which summarises the mean feature retained along the full priority gradient. In both basins, the highest AUC values were reported under Scenario 2 (Biodiversity + Risk) (East: 0.66; West: 0.62), closely followed by Scenario 1 (Biodiversity + Sensitivity) (East: 0.64; West: 0.62). The inclusion of the fishing effort as a negative weighted feature resulted in slightly lower AUC values (Scenario 3; East: 0.62; West: 0.59). The lowest AUC values were consistently associated with the scenarios that included hierarchical constraints linked to existing protection networks (Scenario 4 and 5, respectively for MPA and MPA/FRA configurations) (East: 0.59 and 0.59; West: 0.54 and 0.54). Taken together, these diagnostics indicate that scenarios 1 and 2 retained higher average feature retention over the full prioritisation ranking (reflected on higher AUC), whereas scenarios 4 and 5 produced more constrained solutions with lower retention performance (reflected in lower AUC).

The Zonation scenarios are being compared by checking how well they keep ("retain") conservation features inside the areas ranked as highest priority. The evaluation is done for the top 30% and top 10% of the priority map, which are treated as two alternative protection targets. For each target, two summaries are shown: the mean and the minimum retention value across all features. Higher values mean that the chosen top-priority areas capture more of the conservation features, so the scenario performs better. In the East Mediterranean, mean values for 30% thresholds (*Table 3.-4*) ranged from 0.44 (Scenario 5) to 0.52 (Scenario 2), with scenarios 1 and 2 reporting better performances than scenarios 3, 4, and 5. At 10% retained (*Table 3.-4*), mostly the same performance patterns were depicted, with Scenario 2 showing the highest mean features retention (0.240). In the Western Mediterranean, the separation between scenario groups was more pronounced. Considering the 30% threshold, mean retention was highest for scenarios 1 and 2 (0.464 and 0.465, respectively) and markedly lower under scenarios 4 and 5 (0.315 for both scenarios), and the same schema reported also for the top 10% retained, Scenario 2 again yielded the highest mean retention (0.224), whereas Scenarios 4 and 5 were substantially lower (0.149 for both scenarios). The similar values reported for scenarios 4 and 5 depended on the lower spatial extent of the FRAs in the western side of the basin in comparison to the eastern one. The map in *Figure 3-6* reports the spatial agreement among the five Zonation scenarios, highlighting the areas corresponding to the complete agreement among all five scenarios (100%), and those where the agreement was reached by at least 3 of 5 scenarios (60%).

Table 3.-4. Comparison of Zonation prioritisation scenarios in the Western (WEST) and Eastern (EAST) Mediterranean. For each scenario, feature retention is evaluated within the top 30% and top 10% of the priority ranking (interpreted as alternative protection targets). Values under mean and min represent, respectively, the average and the minimum proportion of conservation features retained within the selected fraction of the landscape (dimensionless, bounded between 0 and 1). Higher mean/min values indicate stronger representation of conservation features within the selected priority fraction. AUC is reported as a complementary overall performance metric across the full ranking (higher AUC indicates better overall retention). In scenarios including cost, this corresponds to fishing effort (fishing days) used as a negative cost layer.

	WEST					EAST				
Scenario	**AUC**	**Top 30%**		**Top 10%**		**AUC**	**Top 30%**		**Top 10%**	
		mean	**min**	**mean**	**min**		**mean**	**min**	**mean**	**min**
1. Biodiversity + Sensitivity	0.62	0.46	0.16	0.21	0.04	0.64	0.50	0.17	0.22	0.06
2. Biodiversity + Risk	0.62	0.46	0.20	0.22	0.06	0.66	0.52	0.18	0.24	0.07
3. Biodiversity + Sensitivity + Risk + Cost	0.59	0.44	0.21	0.22	0.07	0.62	0.47	0.14	0.23	0.06
4. Biodiversity + Sensitivity + Risk + Cost + MPA	0.54	0.31	0.00	0.15	0.00	0.59	0.45	0.16	0.13	0.01
5. Biodiversity + Sensitivity + Risk + Cost + MPA/FRA	0.54	0.32	0.00	0.15	0.00	0.59	0.44	0.18	0.15	0.01

The agreement-based mismatch analysis, evaluated against the current MPA network (protected levels only, see *Table 3-1*) and supported by the spatial patterns shown in the agreement maps (top 30% hotspots and agreement classes) is displayed in *Figure 3-7*. This revealed different patterns between the Mediterranean Sea's sub-basins in terms of the relative retention of biodiversity, sensitivity, and risk features. A qualitative inspection of the agreement maps indicates a substantial spatial overlap with some existing FRAs in the Central-Eastern Mediterranean Sea, especially in the Adriatic Sea, including Jabuka/Pomo Pit, Otranto Channel, and Bari Canyon FRAs. In contrast, in the Western sub-basin, such overlap was not observed. This is due to the limited FRA coverage within the western study domain. Notably, the large deep-sea trawling restriction FRA beyond 1000 m depth is outside the study area domain and therefore not represented in the analysis (*Figure 3-7*).

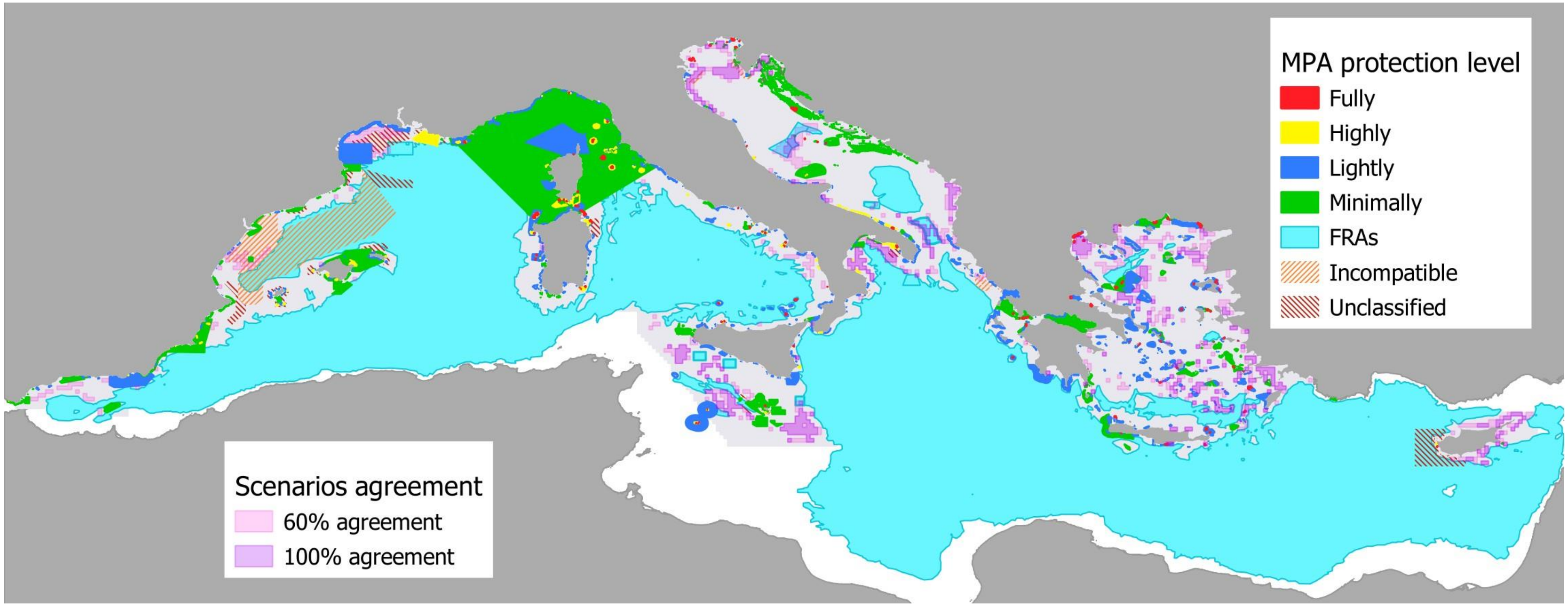

Figure 3-6. Spatial agreement among the five Zonation scenarios (see Table 3.-4 for an overview of these). Cells in 100% agreement were selected by all five scenarios, whereas cells in 60% agreement were selected by at least 3 of 5 scenarios. The MPA network is shown by protection level for comparison with the multi-scenario priority areas.

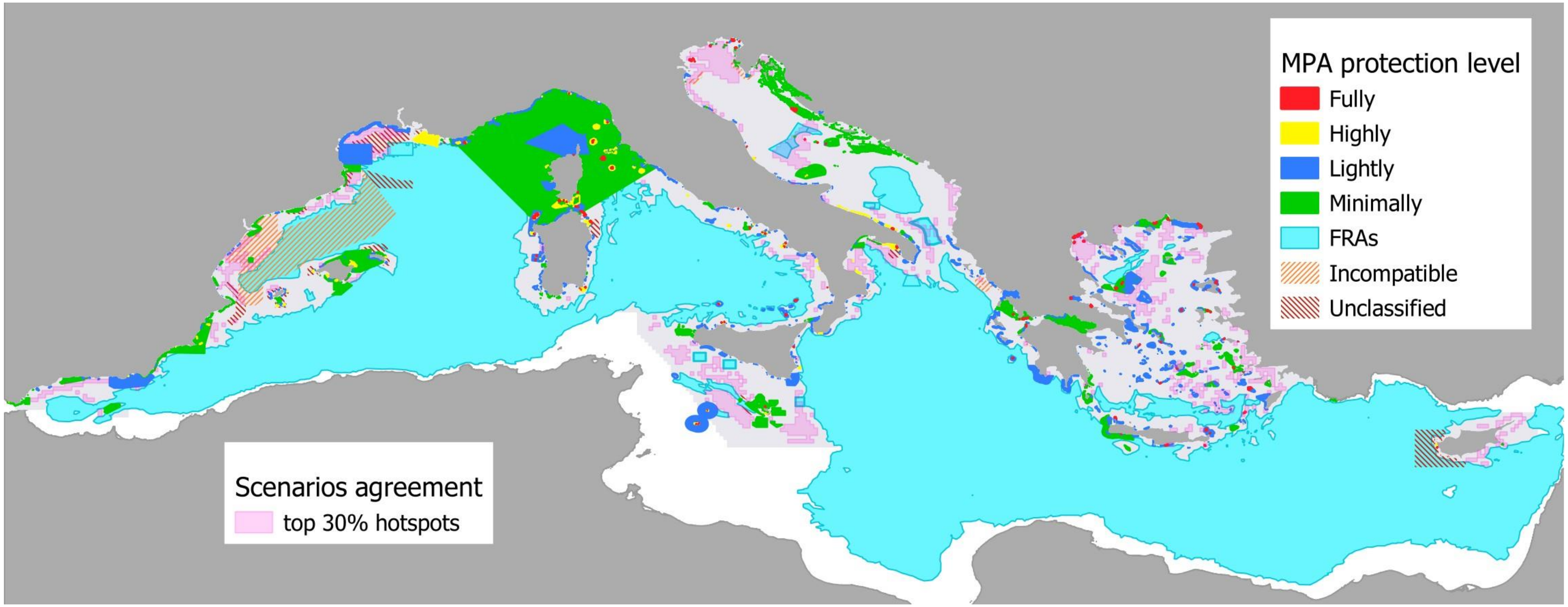

Figure 3-7. Spatial mismatch between the current MPA network and the priority areas identified by the Zonation solutions based on scenarios 1 and 2 (as defined in Table 3.-4). The top 30% agreement hotspots (pink) represent cells simultaneously selected by both scenarios, highlighting priority areas that are not covered by the existing protected network.

In both sub-basins, the relative representation of biodiversity, sensitivity, and risk features differed consistently between the current protected MPA network and the top 30% agreement area derived from solutions of Scenarios 1 and 2. In the Eastern Mediterranean, the agreement area showed a higher average feature content than the protected MPA levels across all feature groups, with mean values going on average from 0.119 in the protected MPA network to 0.199 in the top 30% agreement area outside the actual closure footprint. Conversely, in the Western Mediterranean, protected MPA levels retained a larger average fraction of feature content (0.385) than the additional top 30% agreement area selected outside MPAs (0.131).

3.4 Discussion

The Western Mediterranean Sea is characterised by a higher proportion of MPAs (~30.89% area coverage) than the Eastern basin (7.52%). Although MPAs are valuable tools for safeguarding biodiversity conservation and fisheries sustainability (Edgar et al. 2014, García-Charton et al. 2008, Lynham & Villaseñor-Derbez 2024, Boubekri et al. 2026), the results presented here showed that fully protected areas are negligible in terms of hotspot coverage, while lightly and minimally protected MPAs represented a major portion. Only a very low percentage of hotspots are covered by high protection levels (< 2%). Across hotspot types, the largest share of hotspot surface lies outside the protection network.

The generally low protection level accounted by the actual MPA network indicates that the present protection footprint is weakly aligned with areas of highest ecological value or highest exposure-weighted vulnerability. Indeed, although the actual protection level is better able to intercept the areas where the higher species richness and abundance of highly climate-sensitive species, the consistently weak representation of climate change-related and fishing effort risk hotspots further raises concerns regarding ecosystem resilience, population persistence, and the adaptive capacity under anthropogenic and environmental stresses in future decades (Andrello et al. 2017, Magris et al. 2018, Roberts et al. 2017). The integration of Fisheries Restricted Areas (FRAs) slightly improved hotspot coverage (particularly in the eastern side of the basin, where the FRAs coverage is higher), but the magnitude of change remained modest.

In the Mediterranean Sea, where fully protected areas represent only 0.04% of the basin (MedPAN and UNEP/MAP-SPA/RAC 2023), these findings indicate that achieving meaningful conservation and fisheries sustainability outcomes will require (1) strategically aligning higher protection levels with biodiversity and climate-risk hotspots within the already existing MPA network, and also (2) expanding spatial coverage. Upgrading protection in hotspot areas can, in the short-term, impose adjustment costs to fisheries through reduced access and effort reallocation (Sève et al. 2023, Bastardie et al. 2025). However, Mediterranean evidence shows that well-enforced no-take cores combined with regulated buffer zones and co-management can generate measurable long-term benefits via spillover and improved catch rates, with documented gains in catches and, in some cases, fishers' revenues (Goñi et al. 2010, Di Franco et al. 2016).

Sever spatial mismatches emerge in the Mediterranean Sea, particularly in the southernmost part and in the Western basin, where the dominance of unprotected surfaces within R_{CC} (approximately 95%) and R_{FP} hotspots (approximately 80%) represents the most severe gaps. These gaps raise concerns about the current capacity of the protection network to effectively mitigate impacts. These areas are, indeed, widely recognised as experiencing rapid warming and intense exploitation (Lejeusne et al. 2010, Juza et al. 2022, FAO 2023). These mismatches are particularly relevant given that climate exposure cannot be reduced locally; management action must instead focus on reducing co-occurring human pressures and safeguarding the most vulnerable components of the system (Hidalgo et al. 2022, Pastor et al. 2024).

On the other side, the limited overlap between R_{FP} hotspots and highly regulated areas is coherent with the persistence of long-time exploited fishing grounds. It indicates that the current spatial configuration of stringent measures is not concentrated where reductions in fishing mortality would be most directly targeted toward higher-risk communities (FAO 2023, FAO 2025). It is important to consider both drivers together since it has been shown that the effects of warming waters can negatively modulate the potential benefits of closure areas, raising the need for coupling climate adaptation strategies (Bastardie et al. 2025).

Prioritisation analysis conducted with Zonation approach further corroborates the potential mismatch between the actual Mediterranean MPA network and high-value areas but using a complementary approach in comparison to overlap analysis, which provides a diagnostic of how much of the highest feature values fall within each protection class to quantify coverage gaps. In contrast, Zonation jointly optimises multiple conservation features and ranks areas by their combined contribution to overall representation, likely including locations that are not hotspots for any single layer but represent efficient multi-feature compromises (Kukkala & Moilanen 2013, Moilanen et al. 2022). Scenarios based on biodiversity and sensitivity (Scenario 1) and biodiversity and risk (Scenario 2) achieved higher AUC values and higher feature retention at both the top 10% and 30% fraction than scenarios incorporating hierarchical masks based on the existing MPA network or the combined MPA+FRA network (Scenarios 4 and 5) (*Table 3.-4*). This pattern is consistent with the expected effect of the use of hierarchical masks in Zonation, which can force predefined areas (e.g., existing protected areas) into the top of the ranking and thereby constrain the solution space, typically reducing the achievable feature retention outside those constrained areas relative to an unconstrained prioritisation (Kukkala & Moilanen 2013, Moilanen et al. 2022).

Importantly, the differences between MPA-constrained and unconstrained scenarios should be interpreted against the marked differences in protection level between the two sub-basins. The Western domain is characterised by a much larger protected area than the East (>30% overall), but this is dominated by areas minimally protected and including a non-negligible highly protected component. In contrast, the Eastern domain includes a smaller protected footprint (<8% overall), with minimally and highly protected areas representing a far smaller portion of total area. Because the optimisation in Scenarios 1–3 did not account for protection levels *per se*, the systematically slightly higher AUC values observed in the

Eastern basin should be interpreted as a higher "prioritisation efficiency" in terms of feature retention within the same retained fractions. Conversely, in the Western basin, the wider surface covered by protection areas means that hierarchical masks allocate much of the top-ranked fraction into the existing network, which is dominated by minimally protected zones; this reduces the scope for the prioritisation to select high-value areas outside the network and helps explain the drop in performance under constrained scenarios.

The mismatch analysis provides a complementary perspective on the adequacy of the coverage capability of the actual protection network. By using the existing MPAs (or the combined MPA + FRA configuration) as the baseline, and comparing them with the priority areas in agreement emerging from unconstrained Zonation scenarios (1 and 2), it becomes possible to assess whether high multi-feature conservation value is effectively included within the current network or remains concentrated outside it. In this perspective, agreement between biodiversity-based scenarios identifies spatial compromises where multiple biodiversity, sensitivity and risk components co-occur at relatively high levels, even if none of them reaches its local maximum. Therefore, the agreement-based priorities can be interpreted as candidate areas where expanding or realigning protection could increase the retention of biodiversity, sensitivity and risk features beyond what is currently achieved by the existing network, particularly when the current network is dominated by low-stringency designations or was not originally designed around basin-scale, multi-objective priorities.

In the East Mediterranean, the currently protected network is relatively small and dominated by low-protection categories, so a substantial portion of biodiversity, sensitivity, and risk features remains weakly represented. Accordingly, the agreement area emerging from the unconstrained Zonation scenarios captures markedly higher multi-feature content than the existing protected footprint, indicating that expanding protection outside current MPAs could provide substantial gains in features representation. This is particularly relevant, considering the spatial overlap of some Eastern Mediterranean priority areas outside MPAs with existing FRAs, especially in the Adriatic Sea (e.g. Jabuka/Pomo Pit, Otranto channel, and Bari canyon FRAs). This is relevant in terms of feasibility because fisheries restrictions already in place can decrease the marginal costs and barriers to implementing additional measures that address other drivers or improve overall conservation performance (Abdulla et al. 2009, Bastardie et al. 2025). At the same time, potential short-term socio-economic effects of tightening spatial measures (e.g., effort displacement) remain a realistic constraint that should be anticipated in implementation (Maina et al. 2021). On the other side, in the Western Mediterranean, the protected footprint is larger and already retains higher average feature content, so the gain from additional features not yet within the MPA network would be smaller. However, much of this retained content falls within lightly or minimally protected zones (Giakoumi et al. 2017, Sciberras et al. 2013, Zupan et al. 2018).

3.5 Conclusions

This study indicates that the current Mediterranean protection footprint is unevenly distributed between the West and East sub-basins and is insufficiently aligned with biodiversity and risk priorities at the basin scale. Strict protection is underrepresented in the Mediterranean Sea, and most hotspot surfaces remain outside the network. The most relevant gaps concern climate- and fishing-risk hotspots for demersal species which are predominantly unprotected, suggesting limited capacity to reduce cumulative impacts unless higher-stringency measures are better targeted. In this context, the Eastern Mediterranean is expected to benefit most from strategic expansion of the MPA network, particularly by leveraging priority areas that already have fisheries restrictions in place so that pathways for broader conservation action are likely more feasible. Conversely, in the Western Mediterranean, the MPA network already retains higher feature content but the level of protection is generally low. Hence the largest gains are expected to come from strengthening protection levels and improving effectiveness within the existing network, complemented by selective gap-filling where external priorities persist.

4 Northeast Atlantic: Greater North Sea, Celtic Seas, Bay of Biscay and Iberian Coast

4.1 Introduction

Marine ecosystems across the Northeast Atlantic are increasingly strained by multiple anthropogenic pressures, with commercial fishing and climate change among the most pervasive drivers of change (Halpern et al. 2008, Holt et al. 2012, Punzón et al. 2016). Long-term shifts in sea temperature and associated conditions are altering species distributions and community composition (Thompson et al. 2023, Carroll et al. 2024), while fishing continues to shape population structure, size composition and habitat condition across the region and globally (Pinnegar et al. 2002, Shephard et al. 2012, Johnson et al. 2015). Together, these pressures are reshaping communities and affecting the capacity of marine systems to sustain biodiversity and support ecosystem services such as food provision (Poloczanska et al. 2013, Link & Watson 2019, du Pontavice et al. 2020). As these trends intensify, demand for robust, integrated and forward-looking management approaches will increase.

MPAs are widely accepted as important tools for supporting ecosystem resilience and safeguarding biodiversity (Halpern et al. 2009, Fuchs et al. 2026). This is reflected in global and regional commitments to expand and strengthen protection at sea. Under the Kunming–Montreal Global Biodiversity Framework (GBF) adopted under the Convention on Biological Diversity, Target 3 calls for effective conservation and management of at least 30% of coastal and marine areas by 2030, focusing on areas important for biodiversity and ecosystem functions. The EU Biodiversity Strategy for 2030 similarly commits to legally protecting 30% of EU seas, with one third (10% of EU seas) under strict protection. OSPAR has also committed to an ecologically coherent MPA network in the North-East Atlantic, aligned with “30 by 30”.

A critical part of assessing progress towards these biodiversity targets—and informing future placement of MPAs—is spatial analysis to determine how well communities and ecosystems are represented within protected areas. Targets emphasise ecologically representative, well-connected conservation, implying protection should meaningfully cover biodiversity and ecosystem functions rather than simply increasing area totals. Yet evidence remains limited on how well the existing European MPA network captures fish communities most sensitive to, or at highest risk from, climate change and fishing.

To address this gap, this section identifies community-level hotspots of sensitivity and risk to climate change (CC) and fishing pressure (FP) for bottom-dwelling fish (see also the previous deliverable reports: Engelhard et al. 2024, Rozemeijer et al. 2025). It then assesses to what extent these overlap with the spatial distribution of MPAs across the Greater North Sea, Celtic Seas, and Bay of Biscay & Iberian Coast. We aim to:

(1) Identify ‘hotspots’ based on community-level sensitivity and risk;
(2) Assess overlap between ‘hotspots’ (of bottom oriented fish) and MPAs and fishery-restricted areas;
(3) Identify and discuss clear ‘mismatches’ between ‘hotspots’ and protected areas.

By quantifying the proportion of hotspot areas within current protection layers, and identifying under protected and well covered hotspots, the analysis provides a screening level assessment of alignment between existing protection and areas where fish communities may be most vulnerable under ongoing climate change and continued fishing pressure'

4.2 Methods

This section builds on the community-level sensitivity for bottom-oriented fish and risk layers developed in the preceding report (Rozemeijer et al. 2025). We briefly summarise the previously used data and methods, and outline additional methods specific to this report.

4.2.1 Fishing pressure data

Fishing pressure data for the Greater North Sea, Celtic Seas and Bay of Biscay & Iberian Coast were taken from the International Council for the Exploration of the Sea (ICES) "OSPAR request" spatial fishing-intensity product (ICES 2018 and updates). The product provides annual swept area ratio (SAR) at C-square (0.05° × 0.05°) resolution based on VMS-derived vessel positions. We used surface SAR only for gears affecting the seafloor (bottom trawls, beam trawls, dredges and demersal seiners). SAR values were aggregated to ICES rectangles (0.5° × 1°) to match the ecological data. Internationally complete coverage was available for 2009–2020; incomplete cells along the Portuguese and NW Spanish coasts were removed to avoid misrepresenting fishing intensity.

4.2.2 Climate data

Environmental variables were sourced from NEMO-MEDUSA reanalysis (Yool et al. 2013), providing annual sea surface and bottom temperatures on a 0.25° grid across the Northeast Atlantic (20°W–15°E, 35°N–70°N). Values were aggregated to ICES rectangles in WGS84 to align with the resolution and projection used for community sensitivity and risk analyses.

4.2.3 Species-level sensitivities

Species abundance data were taken from the FISHGLOB database (Maureaud et al. 2024) and cropped to the study extent, resulting in 285 species. Sensitivity to fishing pressure (S_{FP}) and climate change (S_{CC}) was calculated from standardised life-history and ecological traits as described in Engelhard et al. (2024) and Polo et al. (2025). Trait values (categorical scored 0–1; continuous rescaled 0–1) were summed and rescaled to produce species-level S_{FP} and S_{CC} scores. For this report, S_{CC} incorporated thermal affinities only (SST or SBT). Species-level scores formed the basis for community-level indices used here. Full trait definitions and scoring protocols are in D4.1 (Engelhard et al. 2024) and D4.2 (Rozemeijer et al. 2025).

4.2.4 Community-level sensitivities

Community-level sensitivity scores were calculated following Polo et al. (2024). Species-level sensitivity scores were aggregated across hauls within each ICES rectangle in a given year and weighted by ^{10}log-transformed abundance of each species. Species abundances were logged to reduce the influence of overly abundant taxa and give more weight to rarer species.

4.2.5 Fishing pressure and climate change risk

Weighted community-level sensitivity scores were combined with each exposure (of climate change and fishing pressure) for every ICES grid cell and year. Fishing effort data (surface SAR) and SST were scaled between 0 and 1 prior to calculating risk. Because fishing effort data were skewed, SAR values were ^{10}log-transformed prior to scaling. Fishing pressure risk was then calculated as the mean of the weighted community-level sensitivity score (S_{FP}) and surface SAR for each ICES grid cell per year. Similarly, climate change risk was calculated as the mean of the weighted community-level sensitivity score (S_{CC}) and SST for each ICES grid cell per year.

4.2.6 Hotspots of sensitivity and risk

For hotspot detection, we focused on the time period from 2015 to 2020. Annual sensitivity and risk values were aggregated to one value per grid cell by calculating the mean across available years. To identify statistically significant spatial clustering of high and low values of community-level sensitivity and risk (fishing pressure and climate change separately), hotspots were calculated using the Getis-Ord Gi* statistic (Gi*), a local indicator of spatial association that evaluates each spatial unit in relation to its neighbours (Getis & Ord 1992). The Gi* approach determines whether the local sum of a variable in a cell and its neighbouring cells is significantly higher (hotspot) or lower (coldspot) than expected under spatial randomness. Neighbourhood structure was defined using polygon contiguity, where each grid cell was connected to adjacent cells sharing a boundary or vertex (Queen contiguity). Following the conventional Gi* formulation, the focal cell was included in its own neighbourhood when calculating local statistics.

For each grid cell the Getis–Ord Gi* statistic compares the sum of risk values in the cell's local neighbourhood (cell plus neighbours) to the overall study-area distribution, producing a standardised z-score and p-value that indicate clustering strength and significance. Positive significant z-scores indicate hotspots (clusters of high values), while negative significant z-scores indicate coldspots (clusters of low values); the magnitude reflects clustering intensity. We identified robust high-risk clusters by classifying ≥95% hotspots as cells with Gi* z-score > 0 and $p \leq 0.05$ (two-sided), and did not further categorise lower-confidence hotspots or coldspots for mapping. Hotspot maps were masked by survey locations with a 10km buffer to avoid predictions in unsampled areas (within the ICES rectangle-sized grid cells). Masked hotspots were used for plotting and MPA overlap analyses (*Figure B-1* in Appendix B).

4.2.7 Hotspot overlap with MPAs

Using the MPA dataset collated and described in Chapter 2 we quantified protection coverage of high-sensitivity and high-risk clusters by intersecting hotspot polygons with the MPA layer. Each hotspot cell was classified as overlapping at least one MPA, or not overlapping any MPA. We calculated (i) the total number of ≥95% hotspot cells, (ii) the number and proportion overlapping MPAs, and (iii) the number and proportion not overlapping MPAs. Overlapping hotspots were further summarised by MPA protection category by counting hotspot cells intersecting MPAs in each category. Where hotspots intersected multiple MPA designations, counts were non-exclusive (i.e. a hotspot can contribute to more than one category).

4.3 Results

4.3.1 Hotspots of community-level sensitivity

Community-level sensitivity hotspots, reflecting areas dominated by species with high sensitivity scores, varied for climate and fishing pressure. All hotspots of highly climate sensitive communities occurred in the northern North Sea and the Skagerrak Strait (orange in *Figure 4-1*, right panel). Clusters of communities with the highest sensitivities to fishing were found in the central Irish Sea, Celtic Sea and northwest areas of the Bay of Biscay (*Figure 4-1***Error! Reference source not found.**, left panel).

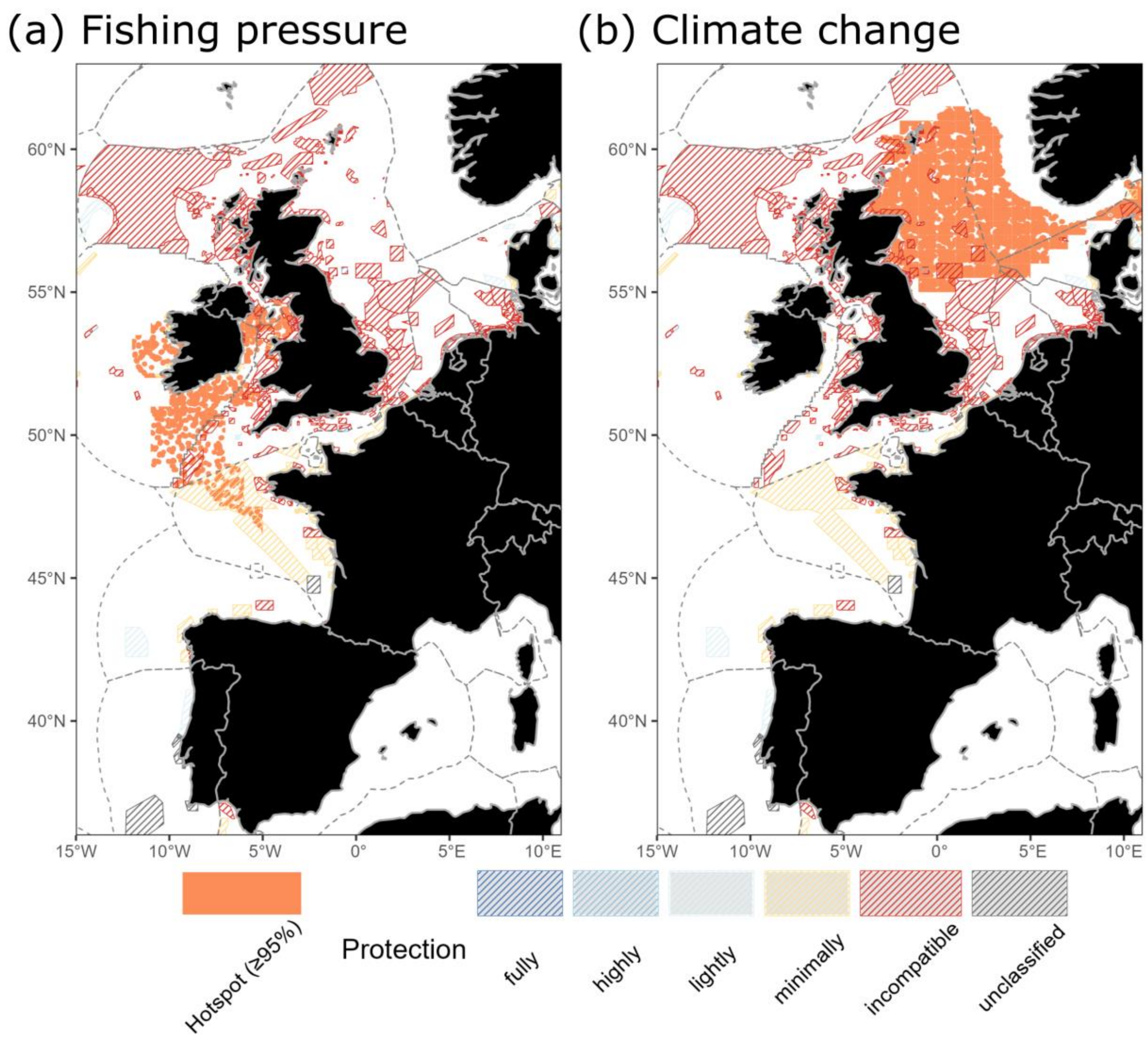

Figure 4-1 Hotspots of community-level ***sensitivity*** *to fishing (a) and climate change (b) and their overlap with marine protected areas. Hotspots of sensitivity are shown in orange. Hashed areas indicate MPAs and colours represent the level of protection (from fully protected to unclassified). EEZs marked with dashed grey lines. Cells are classified as hotspots at 95% confidence represent areas where high sensitivity/risk values are spatially clustered more strongly than expected by chance.*

4.3.2 Hotspots of community-level risk

Hotspots of climate change risk, based on the mean 2015–2020 risk scores and Getis–Ord Gi* ≥95% threshold, formed one main cluster in the more southerly areas of our study region, covering all surveyed grid cells in the Bay of Biscay and Iberian Coast (*Figure 4-2*, right panel). Fishing risk hotspots were more spatially fragmented than climate risk hotspots, with similar hotspots in the Celtic Sea and northern Bay of Biscay, but additional hotspots in the eastern English Channel, northern North Sea near Orkney and the Shetland Islands, and the Skagerrak Strait (*Figure 4-2*, left panel). Note that fishing pressure risk could not be computed south of 45°N due to incomplete fishing effort data.

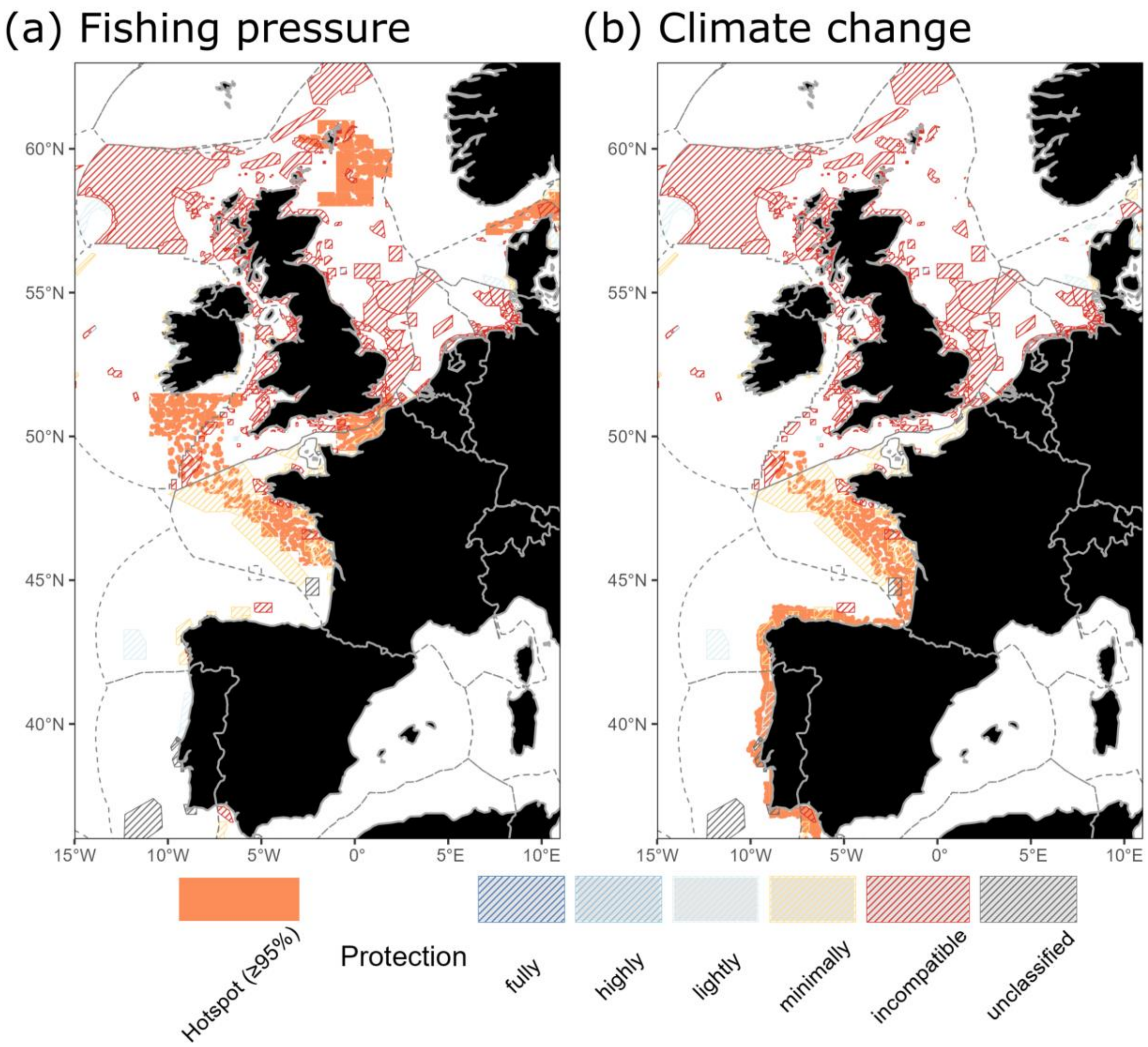

*Figure 4-2 Hotspots of community-level **risk** to fishing (a) and climate change (b) and their overlap with marine protected areas. Hotspots of risk are shown in orange. Hashed areas indicate MPAs and colours represent the level of protection (from fully protected to unclassified). EEZs marked with dashed grey lines. Note that fishing pressure risk could not be computed south of 45°N due to incomplete fishing effort data*

4.3.3 Overlap between hotspots and MPAs

Most climate-sensitive groundfish hotspots fall outside MPAs in the Northeast Atlantic (90%, or 247,475 km^2; *Figure 4-3a*). Of the total 274,877 km^2 of climate-sensitive hotspots, only a very small fraction falls under fully or highly protected areas (fully: 2.77 km^2, 0.001%; highly: 492 km^2, 0.18%; *Figure 4-3b*). Hotspots of fishing-sensitive communities were slightly better protected by MPAs (22.2% within MPAs, 77.8% outside), but again there was minimal coverage by highly, fully or highly protected MPAs. Of the areas that were protected, 64% are only minimally protected and 35% overlaps with MPAs that are incompatible (*Figure 4-3b*).

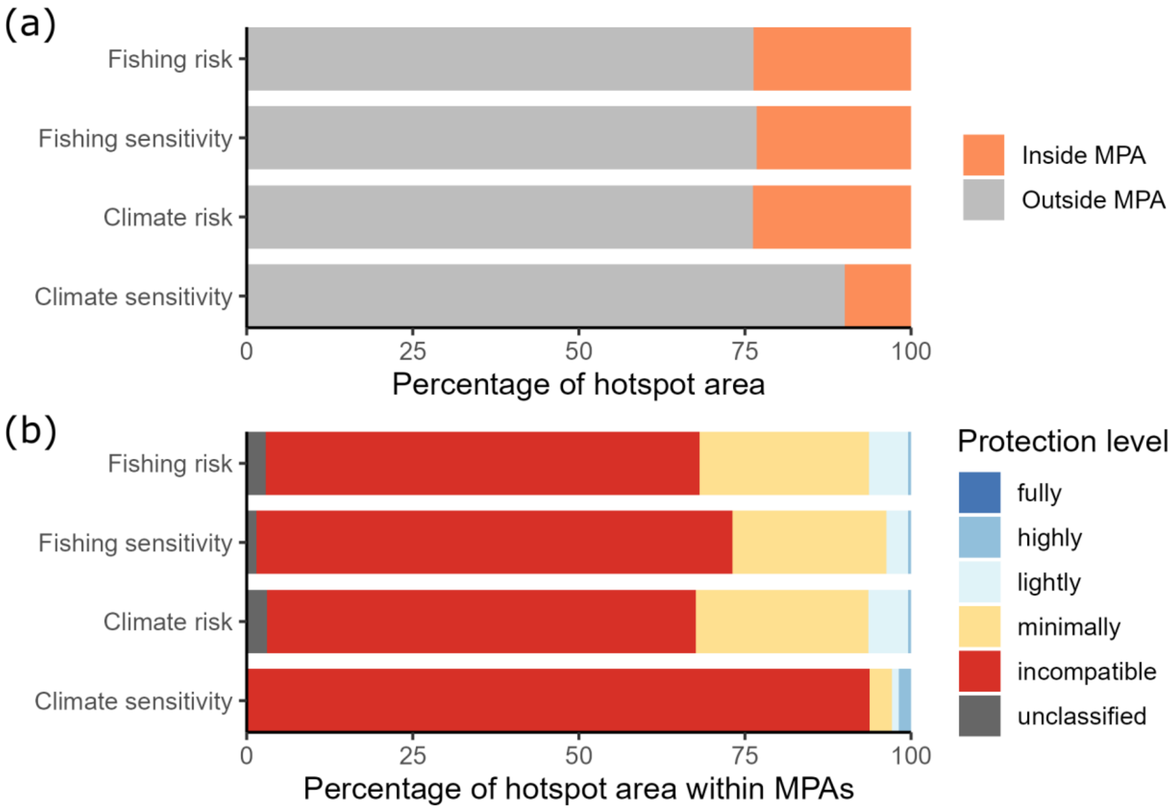

Figure 4-3 Proportion overlap of climate and fishing sensitivity and risk hotspots with marine protected areas. (a) percentage of total hotspot area occurring inside versus outside MPAs. Bars show the proportion of total hotspot area (km^2) within each metric that overlaps any designated MPA compared to areas outside MPAs. (b) protection level of hotspot area occurring within MPAs only. Bars show the proportional distribution of hotspot area across protection categories (fully, highly, lightly, minimally protected, incompatible and unclassified), recalculated to sum to 100% within the subset of hotspot area located inside MPAs.

Overlap patterns for risk hotspots were broadly consistent with those for sensitivity hotspots, but showed slightly greater representation within MPAs (*Figure 4-1*, *Figure 4-2*; *Figure 4-3*). More hotspots of climate risk are protected than climate sensitivity, but still only 23% and the rest occurring outside MPAs (*Figure 4-3a*). Fully and highly protected areas accounted for only 0.002% (5km^2) and 0.5% (950km^2), respectively, of the at-risk hotspots that are protected. Of these, 25% of area is only minimally protected and 65% overlaps with incompatible MPAs (*Figure 4-3b*). Fishing pressure risk hotspots showed a similar amount of match and mismatch with MPAs, with 76% of the area outside MPAs and limited coverage in MPAs with stricter protection (*Figure 4-3*). Of the hotspots that are protected, 65% (133,908 km^2) overlap with incompatible MPAs and 26% (52,372km^2) are minimally protected.

4.4 Discussion and conclusions

Building directly on the previously derived community-level sensitivity and risk indices in Deliverable 4.2, this study identifies statistically significant hotspots of bottom fish communities particularly sensitive and at risk from anthropogenic pressures (fisheries and climate change) and provides new evidence for how well these areas are represented within the existing MPA network across the Greater North Sea, Celtic Seas and Bay of Biscay & Iberian Coast. We found that hotspot locations differed across the study region: climate-sensitive communities clustered in the north (*Figure 4-1*), whereas climate-risk hotspots occurred in the south (*Figure 4-2*); fishing sensitivity and risk hotspots were concentrated in the Irish Sea–Celtic Sea–northern Bay of Biscay and with additional risk clusters in the English Channel and in the northern North Sea (*Figure 4-1*, *Figure 4-2*). Across hotspot types, overlap with MPAs was limited and, where present, largely occurred in minimally protected or incompatible categories, with negligible coverage by fully or highly protected areas. Collectively, these findings imply that progress toward "30 by 30" based on area alone is unlikely to safeguard the fish communities most sensitive to, or at highest risk from, climate change and fishing. This result supports those increasingly being found across the global oceans, that there is a mismatch between the existing degree of protection and biodiversity (Lindegren et al. 2018; Klein et el. 2015). This underscores the need to improve both the representativity and the effectiveness of protection measures.

Community-level sensitivity hotspots were structured geographically, with climate sensitivity concentrated in the northern North Sea and the Skagerrak Strait, while fishing sensitivity was highest in the central Irish Sea, Celtic Sea and north-western Bay of Biscay. These areas represent locations where the composition of fish communities inherently contains species that are more vulnerable to temperature change or fishing disturbance (Engelhard et al. 2015; Couce et al. 2020), irrespective of present exposure levels. In contrast, risk hotspots highlight where climate change or fishing are likely to have the greatest realised ecological impact. Climate change-risk was characterised by a large, cohesive cluster spanning the Bay of Biscay and Iberian Coast, indicating that the magnitude and spatial extent of warming exposure in this region is sufficiently strong to elevate risk across a broad area. Fishing-risk hotspots were more spatially fragmented, occurring in the Celtic Sea, northern Bay of Biscay, eastern English Channel, waters around Orkney and Shetland, and again the Skagerrak Strait, consistent with the heterogenous spatial footprint of fishing effort across the Northeast Atlantic (see previous deliverable report Rozemeijer et al. 2025). These patterns indicate that hotspots of community sensitivity do not always co-locate with community risk: highly sensitive communities may experience low realised risk if exposure is low, while communities of moderate sensitivity may be at high risk when exposed to intense fishing or warming.

From a management perspective, this distinction between hotspots of high-sensitivity and high-risk communities is critical (IPCC 2014). The northern North Sea-Skagerrak sensitivity hotspot represents a region where community vulnerability to warming is intrinsically high and thus could act as an "early-warning" system for rapid climate-driven ecological change, even if warming exposure is currently less intense than further south (Brown et al. 2022; von Schuckmann et al. 2024). Protecting these sensitive areas in the north may help maintain resilience and slow functional loss (White et al. 2025). Conversely, the wide spatial spread of climate risk (higher exposure) in the Bay of Biscay and Iberian Coast suggests that this region may require system-scale climate-adaptive management, rather than isolated site-based interventions (Cannizzo et al. 2025). New MPAs in these locations could improve the capacity of populations to withstand and recover from climatic extremes (Roberts et al. 2017).

Overlaying hotspot areas with the existing MPA network shows that the current distribution of MPAs provides limited coverage for the most vulnerable fish communities. This finding supports broader global evidence of a persistent mismatch between biodiversity patterns and MPA coverage (Klein et al. 2015; Lindegren et al. 2018; Claudet et al. 2020; Conners et al. 2022; Lin et al. 2026). Climate sensitive hotspots are by far the least protected, with approximately 90% of their total area lying entirely outside MPAs. Where overlap exists, the coverage is overwhelmingly characterised by minimal or incompatible protection categories, and the amount of fully or highly protected area is negligible. These patterns suggest that present protection is unlikely to translate into strong ecological outcomes for vulnerable fish communities (Aminian-Biquet et al. 2024b). Regions of high sensitivity, where fish communities are composed of species least tolerant to high temperatures, have not historically been a focus of MPA designation and strengthening protection in these areas represents a substantial gap in the representativity of the current network.

Hotspots of sensitive and at-risk communities to fishing have a slightly greater overlap with MPAs, with around a quarter of their area falling within existing designations. However, the majority of this overlap is again in categories that permit extractive or disruptive activities, limiting the capacity of these MPAs to reduce pressures most relevant to fishing-sensitive communities (Perry et al. 2022). This raises an important point that spatial overlap alone does not equate to reduced pressure or improved ecosystem condition (Mazaris et al. 2019; Claudet et al. 2020). If MPAs allow high-impact fishing gears, particularly bottom-contacting gears which forms a large number of the MPAs in the Northeast Atlantic region (Perry et al.

2022), they are unlikely to provide measurable benefits to communities composed of species with low resilience to disturbance (Aminian-Biquet et al. 2024b). Thus, the quality of protection, not merely its spatial footprint, is central to interpreting MPA effectiveness (Pieraccini et al. 2016). Evidence from global assessments shows that MPA outcomes are highly variable and depend strongly on whether sites are effectively managed and resourced (Lester et al. 2009, Di Franco et al. 2016, Mast et al. 2025). In particular, Gill et al. (2017) demonstrate that inadequate staff and budget are widespread across MPAs globally, with staff capacity reported as inadequate or below optimum in the vast majority of sites and budgets frequently insufficient for basic management needs. In this context, the dominance of low-integrity protection categories intersecting hotspots in our analysis suggests a double limitation: regulations may not constrain key pressures in hotspot areas, and even where rules exist, inadequate capacity could further reduce compliance, monitoring, and adaptive management, diminishing the likelihood that MPAs meaningfully safeguard sensitive or at-risk communities (Álvarez-Fernández et al. 2017).

Our results highlight several regions where management attention is particularly urgent. The climate-sensitive hotspot in the northern North Sea and Skagerrak stands out as one of the poorest-protected ecologically important areas in the Northeast Atlantic. This region could benefit from enhanced protection because communities there may undergo rapid structural change under continued warming, and maintaining local ecosystem integrity could help buffer climatic impacts (Benedetti-Cecchi et al. 2024; Fuchs et al. 2026). Similarly, the extensive climate-risk hotspot of the Bay of Biscay and Iberian Coast (see also Chust et al. 2022) represents a large, contiguous region facing high climate exposure, yet strict protection is nearly absent. Strengthening existing MPAs or identifying new areas for higher levels of protection may be required to build resilience across this climate-vulnerable system (Barnett & Baskett 2015). For fishing-risk hotspots, the fragmented spatial pattern suggests that targeted, pressure-specific management measures, such as gear restrictions, seasonal closures or spatial effort limits, could be effective (Pike et al. 2024), particularly in regions where full protection is socially or operationally challenging (Allison et al. 1998).

Moderate-priority areas include the coincidence of fishing-sensitivity and fishing-risk hotspots in parts of the Celtic Sea and northern Bay of Biscay, where existing MPAs provide a foundation that could be strengthened through revised zoning or improved regulation. These likely mark areas where trait-vulnerable communities are currently exposed to high disturbance and thus where management intervention could yield relatively rapid ecological benefits (Horta e Costa et al. 2025). In such areas, upgrading minimally protected zones to higher-integrity protection categories, aligning permitted activities with conservation objectives, or adjusting boundaries to incorporate adjacent hotspot cells could substantially improve ecological outcomes without the need for entirely new designations (Grorud-Colvert et al. 2021; Pike et al. 2024; Horta e Costa et al. 2025). Lower-priority hotspots may include smaller or less exposed clusters, or regions where existing management frameworks already provide partial mitigation of dominant pressures. However, these areas should remain under observation, as emerging pressures or changing climate conditions may increase their importance over time.

These findings support several broader policy recommendations. First, improving the quality of protection within existing MPAs intersecting hotspot regions may yield the most rapid, immediate gains. Many hotspots are already partially covered by low-integrity MPAs and strengthening management within these areas, by reducing high-impact fishing gears, limiting extractive activities or re-designating zones to higher protection levels, would improve alignment between protection and ecological vulnerability (Claudet et al. 2020; Perry et al. 2022; Aminian-Biquet et al. 2024b). Second, where hotspots fall just outside existing MPA boundaries, expanding these boundaries could increase representativity with relatively low administrative cost while enhancing ecological coherence (Brander et al. 2020). Third, the results identify a clear need for new area-based protections in regions of persistent, large mismatch, particularly in the northern North Sea–Skagerrak and the Bay of Biscay–Iberian Coast, where the existing network does not meaningfully cover areas of greatest sensitivity or risk. In cases where fully protected MPAs are not feasible, alternative mechanisms like temporary closures, fishery management plans or other effective area-based conservation measures could be used to reduce pressures within hotspot cells (Cashion et al. 2020).

Finally, due to the spatial resolution of the analyses and the varying quality of information on MPA regulations, these findings should be interpreted as a screening-level assessment rather than a definitive evaluation of site-level protection effectiveness. Targeted monitoring programmes in under-represented or partially protected hotspots would help identify whether current management measures are sufficient to safeguard sensitive communities, and whether additional actions are required (White et al. 2025). Monitoring should be designed to detect changes in community composition, trait structure and abundance, enabling evaluation of both climate-driven trends and the impacts of fishing pressure. Such evidence will be essential to support the implementation of more effective, ecologically coherent and climate-resilient protection measures across the Northeast Atlantic.

5 Icelandic waters

5.1 Introduction

Iceland is an island nation located in the North Atlantic, and as such receives the influence from two distinct water masses: the Atlantic and the Arctic/Polar waters (Stefánsson 1962; Malmberg and Valdimarsson 2003; Jónsson and Valdimarsson 2005). Warm, saline waters are transported on the Icelandic coastal shelf by the Irminger current, which splits from the Gulf Stream and flows westward after encountering the Iceland-Faroe Ridge. The current then continues flowing clockwise around the island, losing strength as it reaches its northern coasts (*Figure 5-1*) (Valdimarsson and Malmberg 1999). Thus, hydrographic conditions on the northern portion of the coastal shelf are defined by a mixture of Polar (East Greenlandic Current), Arctic (East Icelandic Current) and Atlantic waters (*Figure 5-1*), resulting in lower temperature and salinity than in the south west (Stefánsson 1962; Astthorsson et al. 2007).

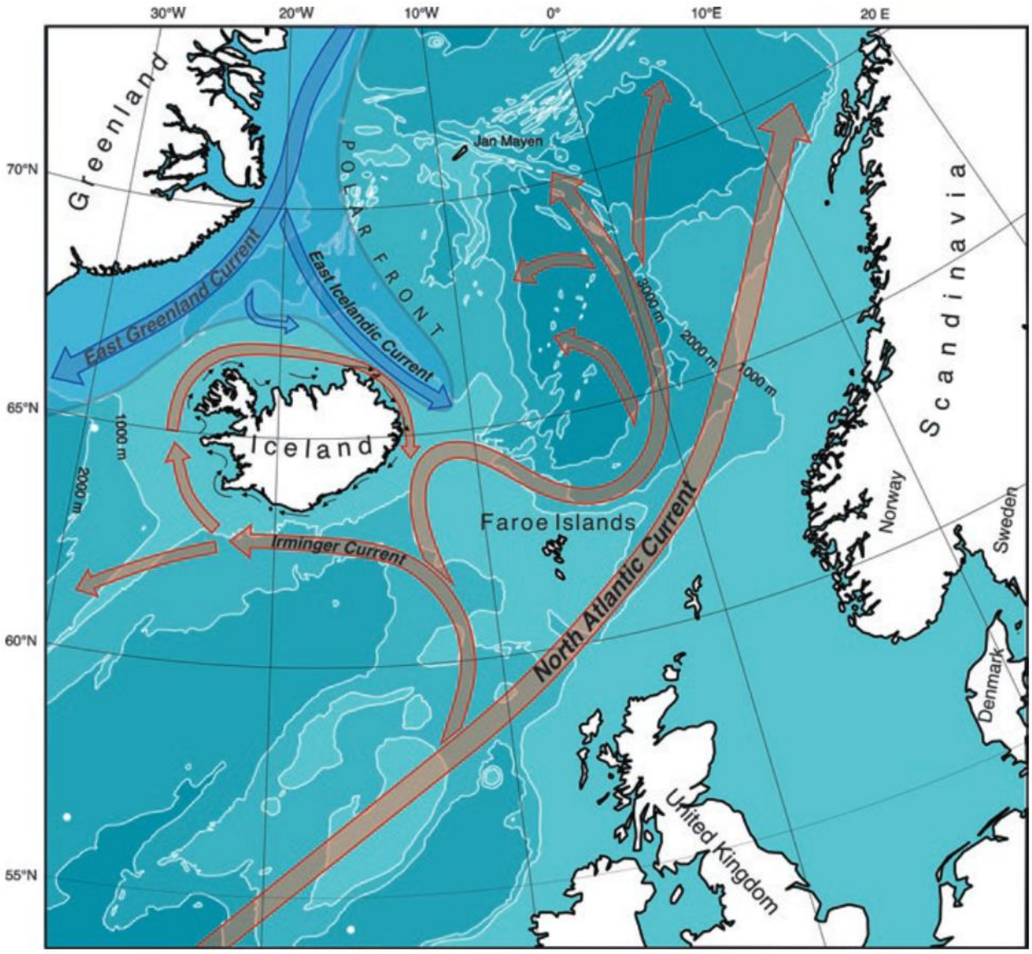

Figure 5-1. The direction of the Atlantic (red) and Arctic/Polar (light and dark blue) water masses around Iceland. From Símonarson et al. (2021).

This heterogeneity in water qualities has been linked to the presence of two separate species assemblages: one located predominantly to the south and west of the island, characterised by a greater fraction of Atlantic species, and one in the north and east, with a larger fraction of Arctic species (Astthorsson et al. 2007; Stefánsdóttir et al. 2010; Mecklenburg et al. 2011; Símonarson et al. 2021). Notably, transport of the Irminger current along northern Iceland is variable, with years of stronger inflow being associated with warmer local seawater temperatures (Malmberg and Valdimarsson 2003; Jónsson and Valdimarsson 2005). Changes in the composition of marine faunal assemblages in response to this variability have been reported in the past (Vilhjálmsson 1997; Stefánsdóttir et al. 2010; Valdimarsson et al. 2012; Sólmundsson et al. 2025), with shifts towards more southern species expected during periods of warming (Björnsson and Pálsson 2004; Astthorsson et al. 2007; Sólmundsson et al. 2025). In the area, northward expansions in co-occurrence with increasing sea temperatures have already been observed for several temperate and sub-arctic marine species (Stefánsdóttir et al. 2010; Valdimarsson et al. 2012; Campana et al. 2020; Sólmundsson et al. 2025).

Resilience of marine ecosystems to changes induced by climate variability can be reduced by exceptional mortality caused by fishing (Ling et al. 2009; Roberts et al. 2017; Cheung et al. 2024), as well as by habitat degradation from activities such as bottom trawling (Palanques et al. 2014; Handley et al. 2014; Martín et al. 2014). Iceland is one of the major fishing nations in the world (FAO 2018), hence the Icelandic seas are subject to relatively high levels of fishing. Nevertheless, recent decades saw a decrease in effort by several fishing methods (MFRI 2024), and commercial fish stocks around Iceland are generally considered well managed (Kleisner et al. 2022; FAO 2025). Part of this management strategy is the implementation of fishery-restricted areas, which include seasonal and permanent closures to fishing methods such as gillnets, bottom trawling and bottom-contact longlines (Ólafsdóttir et al. 2024). These closures have been implemented with the aim of protecting either vulnerable marine habitats, most prominently cold-water coral reefs, or commercially important fish species during their spawning period (Ólafsdóttir et al. 2024). Therefore, none of these closures were created with the aim of protecting biodiversity *per se*, or increasing the climate resilience of demersal communities. Aside from fishery-restricted areas, marine protected areas are also present. However, the level of protection this network offers to demersal fish communities is limited, as these areas are either small, or do not regulate the use of bottom-contact fishing g ear (e.g., the protected area in Breiðafjörður, which aims at conserving the bay's islands and coasts) (Alþingi 1995; Ólafsdóttir et al. 2024; The Icelandic Environmental Agency 2026).

The aim of this work is to assess the level of protection that the network of fishery-restricted areas around Iceland provides to groundfish species that are expected to, or have already been affected by climate change, fishing, and other human activities. To do so, we quantified the amount of overlap between these areas and hotspots of species belonging to relevant groups, i.e. species sensitive to climate change; those sensitive to fishing pressure; Arctic species; and species categorised as threatened or near threatened by the International Union for Conservation of Nature (IUCN Red List) (IUCN 2026). Furthermore, we included commercially important species, to assess the level of protection that fishery-restricted areas offer to species targeted by the local industry. Therefore, the specific research questions are:

(1) Where are the hotspots of groundfish species most threatened by climate change and anthropogenic pressures, around Iceland?

(2) To what extent do these hotspots overlap with the current fishery-restricted area network?
(3) Which areas should be prioritised by future conservation measures aimed at conserving demersal fish biodiversity?

5.2 Methods

5.2.1 Preparation of feature raster layers

We used the joint species distribution model described in Lindegren et al. (2025) to produce predictions of biomass (kg/km^2) for 79 demersal and bathydemersal fish species at a 0.004°x0.004° of longitude and latitude scale. After producing these predictions, we obtained the average value for 2015–2024 and transformed it using the natural logarithm of 1+X. Twenty raster layers outlining the distribution of commercially important and at-risk species were included in the analysis without further elaboration. These species were selected using information available in the Fisheries Overview by the Marine and Freshwater Research Institute of Iceland (MFRI 2024), with categorisations on their Red List status taken from the International Union for Conservation of Nature obtained through the R package 'rredlist' (Gearty & Chamberlain 2025).

To produce the distribution layers of species sensitive to climate change, sensitive to fishing pressure, and with Arctic biogeographic affiliation, we first split species in groups depending on their depth affinity, from information collected and outlined in Rozemeijer et al. (2025). We obtained 5 groups of similar size: species with a depth affinity <127m, between 127m and 216m, between 217m and 431m, between 432m and 701m, and >701m. This was done to avoid possible biases in the prioritisation; for example, had this split not been introduced, a particularly abundant shallow-water species might have led the corresponding group layer to have high values in coastal areas, leading to shallow-water areas receiving a greater weight in the prioritisation at the detriment of deep-water areas.

To select Arctic species, we used previously developed biogeographic classifications (Rozemeijer et al. 2025; Sólmundsson et al. 2025). Likewise, to select species with high sensitivity to climate change or fishing pressure we relied on species-specific scores in trait-based indices developed previously – the sensitivity to climate change (S_{CC}) and fishing pressure (S_{FP}) (Rozemeijer et al. 2025; Polo et al. 2025). If S_{CC} or S_{FP} of a given species was ≥0.5, the species was considered sensitive to the corresponding pressure. Through this procedure we obtained 15 species-group layers.

To obtain the contours of areas restricted to fishing, we used the Ocean Viewer portal (NSII 2025). Only areas banning the use of bottom trawling and/or bottom-contact longlines either permanently or seasonally (for more than half of the year) were considered for the analysis. This led to the selection of 38 fishery-restricted areas: 24 of these have a permanent ban on both fishing gears, 7 a permanent ban on trawling, 4 a permanent ban on longlines, whereas 3 have seasonal closures to either or both fishing methods. Respectively, we labelled these areas based on the banning regime: areas permanently closed to both bottom trawling and longlines, areas permanently closed to bottom trawling, areas permanently closed to longlines, and areas that have seasonal closures to both or to either fishing method. Mining, anchoring and aquaculture are not practised in all these areas.

5.2.2 Spatial prioritisation

To select priority areas for conservation we used the software Zonation 5 (Moilanen et al. 2022). Zonation selects priority areas by iterative ranking and by removing the raster grid cells that can be lost with the smallest aggregate loss of marine biodiversity (Virtanen et al. 2018). Two solutions were produced using the CAZMAX algorithm and a mean ranking error threshold of 0.001 (Moilanen et al. 2024): a constrained solution accounting for the presence of the fishery restricted area network, and an unconstrained one. In the constrained solution, areas of highest priority ranking are forced to be within the protected area network. We compared the constrained and unconstrained solutions to evaluate the performance of the current fishery-restricted area network in protecting the features considered in this analysis.

To favour certain features over others during the prioritization we applied different weights to each. Weights in Zonation can be of two different types: group weights and individual weights (Virtanen et al. 2018; Moilanen et al. 2024). Group weights correspond to the aggregate weight associated with a certain group of features, while individual weights refer to the weight associated with an individual feature within their group. Species with high sensitivity to fishing pressure were the group of features with the greatest aggregate weight (38.50%), followed by Arctic species (26.00%), then at-risk single-species layers (19.75%), species with high sensitivity to climate change (13.50%), and the commercially important single-species layers (7.25%). Following the logic outlined in Virtanen et al. (2018), we attributed larger aggregate weights to species-groups layers, as these are likely representatives of a variety of species with similar environmental requirements and sensitivities to the species included in each group. The layers of species sensitive to climate change were the only exception to this. This was deliberate, as in a previous analysis it was found that this group of species has been increasing in abundance during the period 1996-2024, related with a marked increase in Atlantic species in the region (Rozemeijer et al. 2025).

The individual weight of each feature composing the Arctic species-groups, the groups of species sensitive to climate change, and the groups of species sensitive to fishing was highest for species with a depth affinity <216m. The greater weight was allocated following the assumption that shallow-water species are the most impacted by anthropogenic pressures (Halpern et al. 2019). For commercial species, individual weights were allocated using the proportion of the total catch value that was associated to each species (2015-2024 average, SI 2026). For at-risk species, the highest individual weight value (4) was given to Endangered species, while lower scores were given to Vulnerable (2) and Near Threatened (1) species. If a species was both an at-risk species and a commercially important species, then it was included in the at-risk grouping. In order to avoid having the prioritisation selecting areas that have been degraded due to high levels of bottom trawling activity, we introduced the 2015-2024 average bottom trawling intensity as a log(1+X) transformed negative weighted layer. The weights used in the analysis are summarized in *Table C-1* of the Appendix.

5.3 Results

The evaluated area covered 491,629 km^2, of which 414,700 km^2 belong to the Icelandic EEZ, 58,892 km^2 to the Greenlandic EEZ, and 17,835 km^2 to the Faeroese EEZ. This amounted to 54.33%, 2.60% and 6.72% of each EEZ, respectively. The fishery-restricted areas that were selected spanned a total of 21,947 km^2, corresponding to 4.46% of the study area and 2.88% of the Icelandic EEZ. Six main biodiversity hotspot clusters were identified within the study area (Figure 5.2). The evaluated fishery-restricted area network covers different hotspots to a varying degree, but the proportion is in all cases relatively low (≤5%) (Table 5.1).

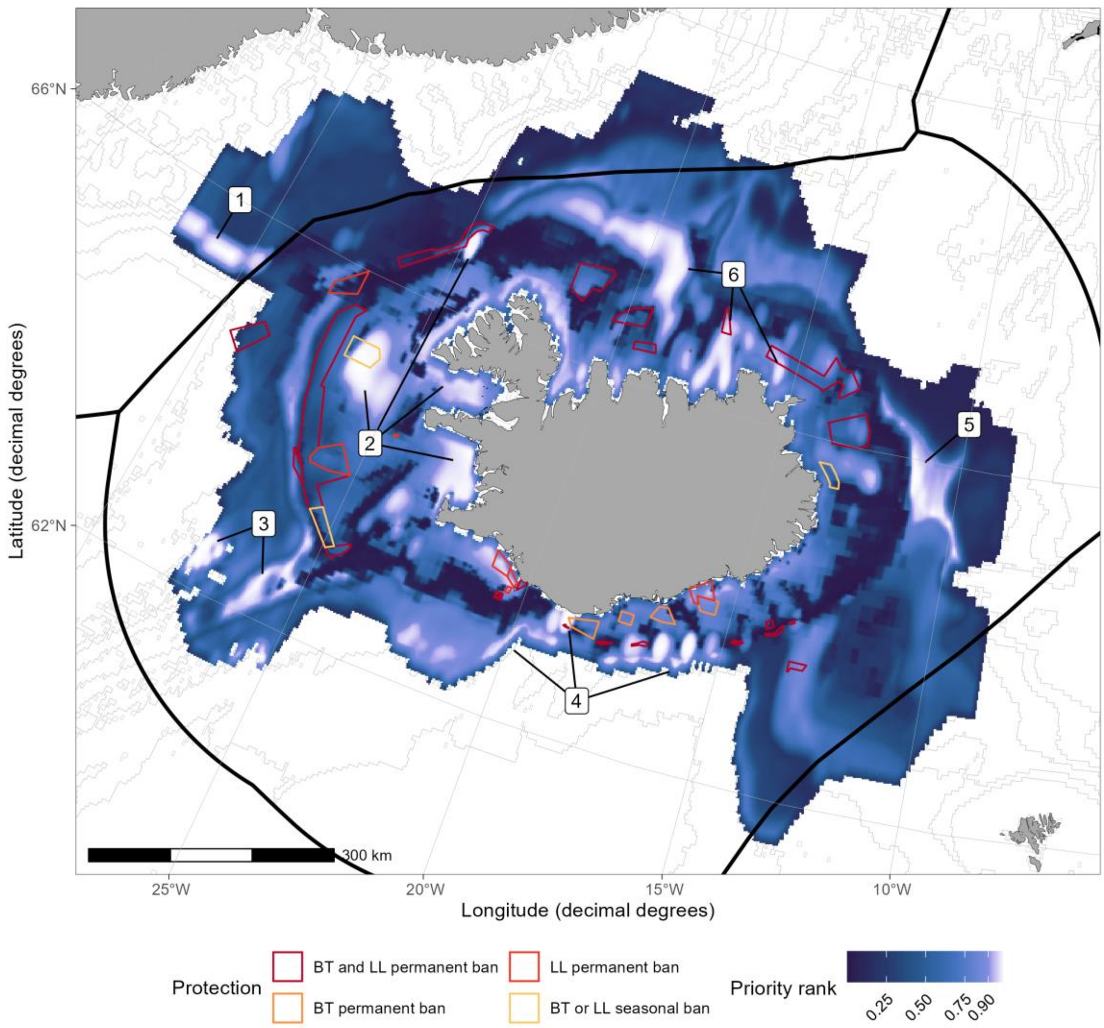

Figure 5-2. Seabed area covered by the analysis and locations of the considered fishery-restricted areas (yellow to red) within the Greenlandic, Icelandic and Faeroese EEZs (bold black line). Six main clusters of priority areas for conservation (white shades) were identified by the unconstrained solution: on the Greenlandic shelf break (cluster #1), offshore western Iceland (#2), Reykjanes ridge (#3), offshore southern Iceland (#4), shelf slope of northeastern Iceland (#5), and north of Iceland (#6). The outlines of the fishery protected areas considered are shown in shades of yellow and red, reflecting whether permanent or seasonal bans on bottom trawling (BT) or bottom-contact longlines (LL) are in place.

Table 5-1. Overlap between the considered fishery-restricted area network and areas for conservation identified by the unconstrained solution, at 3 levels of priority ranking. Areas of overlap are reported (in km²), as well as the proportions of priority areas for conservation that fall within fishery-restricted areas (%, in parentheses). BT stands for bottom trawling, while LL stands for bottom-contact longlines

Fishery-restricted area protection level	Area based on top 5% priority rank, km² (%)	Area based on top 10% priority rank, km² (%)	Area based on top 20% priority rank, km² (%)
BT and LL permanent ban	186 (0.75%)	561 (1.13%)	1334 (1.35%)
BT permanent ban	198 (0.80%)	560 (1.19%)	910 (0.92%)
LL permanent ban	44 (0.18%)	84 (0.17%)	748 (0.76%)
BT or LL seasonal ban	718 (2.9%)	766 (1.55%)	810 (0.82%)
Total	1146 (4.63%)	1999 (4.04%)	3802 (3.84%)

The considered features (5 groups of Arctic species, 10 commercially important species, 10 IUCN Red List species, 5 groups of species sensitive to climate change and 5 groups of species sensitive to fishing pressure) presented a broad variety of distribution patterns. For example, highly concentrated distribution patterns were found in the Arctic species with shallowest depth affinity (<216m), the commercial species anglerfish *Lophius piscatorius* and European plaice *Pleuronectes platessa*, as well as the Endangered species spiked dogfish *Squalus acanthias* and leafscale gulper shark *Centrophorus squamosus*; their very patchy distributions were indicated by convex performance curves (*Figure 5-3*A, B, C). Other features are more evenly distributed across the study area and hence tended to follow the diagonal of the priority rank fraction of covered distribution plots. This was the case of the species with high S_{CC} and S_{FP} scores (*Figure 5-3*D, E). Finally, few features have a widespread but sparse distribution, demonstrated by concave performance curves. This is the case of the Near Threatened species, spotted wolf-fish *Anarhichas minor* (*Figure 5-3*C).

Overall, the performance of the current fishery-restricted area network in protecting the considered features is relatively low; its average coverage is 6%. If a protected area network of the same extent as the existing fishery-restricted area network were to be established, following the area ranking given by the unconstrained solution, an average of 13% of the features would be covered (*Figure 5-3*F). The only group of features for which the average coverage by the constrained solution outperforms the unconstrained solution, is the commercial species group (*Figure 5-3*B).

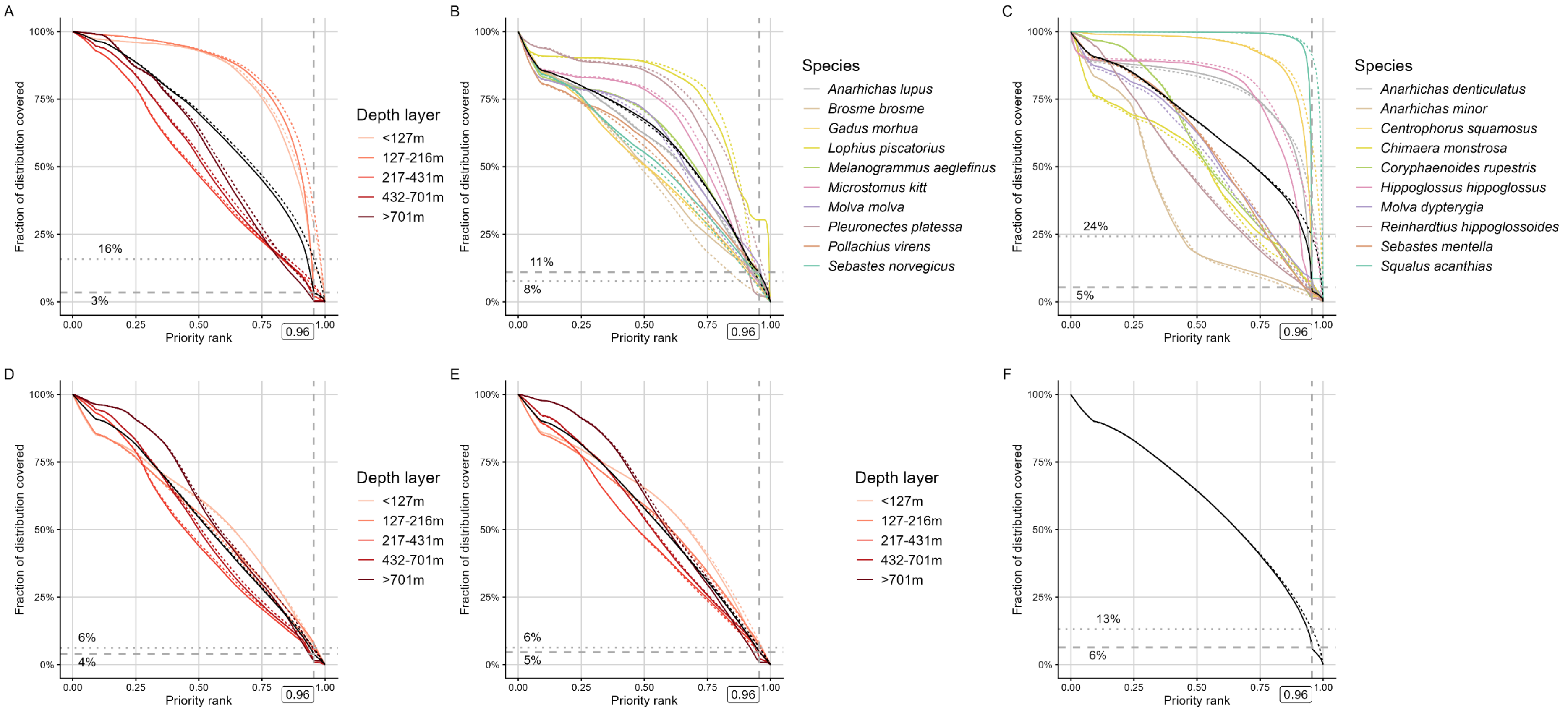

Figure 5-3. Performance curves of the constrained (solid line) and unconstrained (dashed line) solution for each group of features covered in this study (Arctic species, A; commercial species, B; threatened species, C; species sensitive to climate change, D; and species sensitive to fishing pressure, E), and the mean for all features (F). Black lines indicate the mean for each group of features. The considered fishery-restricted area network (vertical dashed line) covers a mean fraction of distribution indicated by the horizontal dashed line, while the dotted horizontal line indicates the mean feature coverage that would be obtained if an area with the same extent of the fishery-restricted area network was to be relocated following the area-prioritization obtained with the unconstrained solution.

The expansion of protection measures required to cover all areas with priority ranking ≥0.95 (i.e. top 5% priority areas) would span an additional 23,552 km^2, corresponding to 4.79% of the study area. Of this, 1,601 km^2 would be within the Greenlandic EEZ, of which it would comprise 0.07%, while 21,951 km^2 would fall within the Icelandic EEZ, of which it would comprise 2.88%. No areas relevant for the expansion of the MPA network were identified within the Faeroese EEZ (*Figure 5-4*). The combination of the existing fishery-restricted area network and its potential expansion would cover a total of 45.499 km^2, or 9.25% of the whole study area, and cover an average of 20.17% of all features.

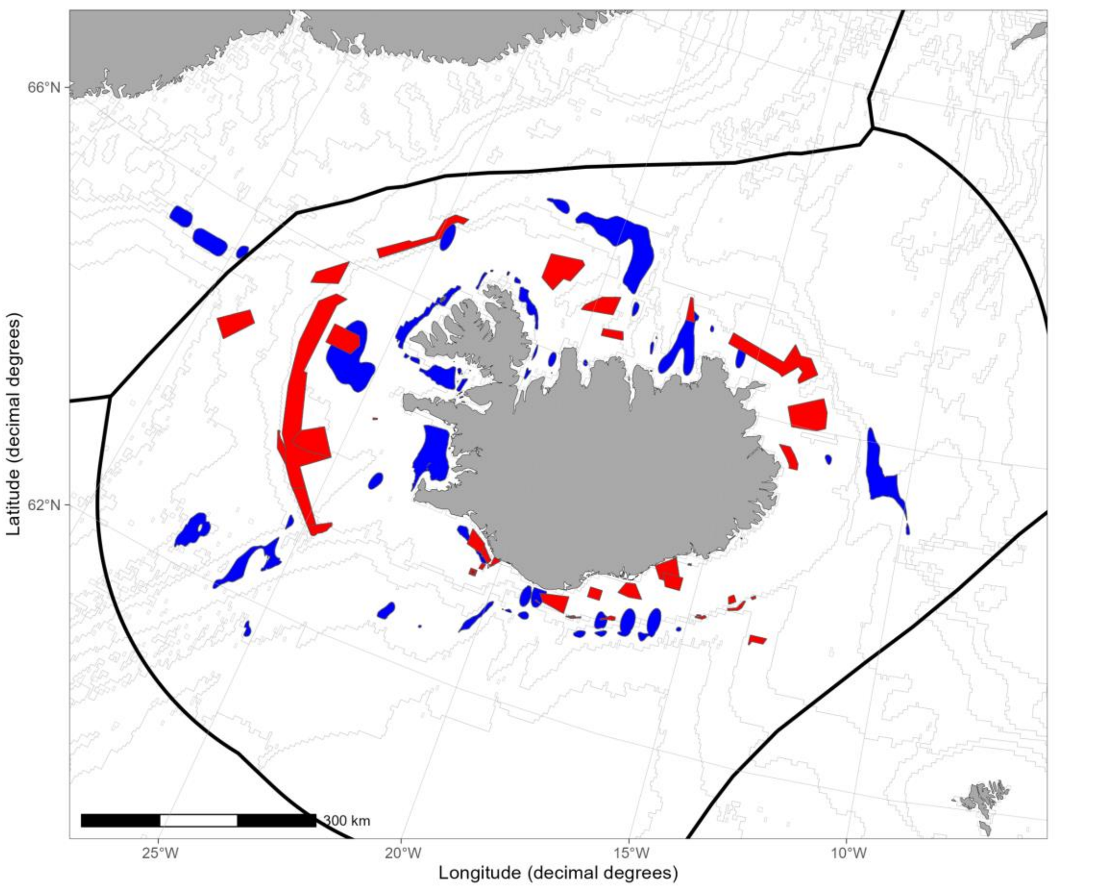

Figure 5-4. The present fishery-restricted area network (red) and the potential expansion (blue) that would be required to cover all areas with priority ranking ≥0.95 (i.e. top 5% priority areas). Black lines indicate Greenlandic and Icelandic Exclusive Economic Zones

5.4 Discussion and conclusions

Our analysis identified six main location clusters relevant for the conservation of groundfish biodiversity within the Greenlandic and Icelandic EEZs. Some of these areas, like the western Icelandic (#2) and the southern part of the north Icelandic (#6) cluster, are found at relatively shallow depths (<300m) on the continental shelf of Iceland. Others are found in deeper regions, such as the Reykjanes ridge (#3) and the shelf break southeast of Greenland (#1), and those to the south (#4), northeast (#5) and north (#6) of Iceland.

Considering that the distribution of species characterized by high S_{CC} or S_{FP} scores is relatively evenly distributed throughout the landscape, all these clusters were primarily defined by the concentrated distribution of some Arctic, threatened, and commercially important species. Due to the high weight associated during the prioritization, shallow-water (<217m) Arctic species (e.g., snakeblenny *Lumpenus lampretaeformis*) was the feature defining cluster #5 and #6, as well as the westernmost area within cluster #2. With the lack of Arctic species in the southwestern section of the study area, threatened and commercially important species were instead the features defining clusters #1, #2, and #4. For example, #1 was defined by co-occurring and relatively high biomass for northern wolffish *Anarhichas denticulatus*, blue ling *Molva dypterygia* and beaked redfish *Sebastes mentella* on the Greenlandic continental shelf break.

The overlap between the priority areas for the conservation of groundfish identified in this study and the network of fishery-restricted areas is rather limited. Furthermore, areas with a priority rank ≥0.95 overlapped with seasonal closures for bottom-contact fishing gear on a greater extent than permanent ones. Conversely, the overlap calculated for areas with a rank ≥0.80 was greater for such permanent closures and lower for seasonal. This indicates a certain "misalignment" between priority areas and the fishery-restricted area network, which are placed in locations with lower priority ranking that border areas with higher ranking. As a result, only a small portion of cluster #2, #4 and #6 are protected by fishery restricted areas, while cluster #1, #3 and #5 have no protection. In addition, the shallow-water portions of the clusters are the ones usually covered by fishery-management measures, leaving deeper-water priority areas on the shelf break and slope uncovered (e.g., cluster #4 and #6).

The bulk of the overlap among the fishery-restricted area network and areas with high priority ranking appears to be due to the closure mandated by the regulation 958/2019 article 8, which covers part of the cluster #2. This is a seasonal closure for bottom trawling and longline, in effect half of the year (15th of September – 30th of April) with the aim of protecting the Atlantic wolffish *Anarhichas lupus* spawning (Ólafsdóttir et al. 2024). As mentioned above, this is a hotspot for shallow-water (<127m) Arctic species. Other important areas for Arctic species are found within cluster #5 and #6, where also species with a depth affinity ≥127m are found. Arctic fish are vulnerable to climate change, with widespread shifts in their distribution patterns having been observed in response to changing seawater temperatures across the Arctic and subarctic (Frainer et al. 2021; Rozemeijer et al. 2025; Sólmundsson et al. 2025), including in deep-water regions such as the shelf break and slope of eastern Greenland (Emblemsvåg et al. 2020, 2022). Rendering permanent the closure overlapping cluster #2 and implementing protection measures on cluster #5 and #6 could increase the resilience of these species to climate change by reducing their fishing mortality.

Implementing conservation measures within cluster #1, #3, and #4, as well as the coastal section of #2, would instead protect a more limited number of species. Nevertheless, these species are relevant for the protection of marine biodiversity and the economy, as they include threatened, near threatened and commercially exploited species. For instance, within cluster #1 and #2 we can find high biomass levels of *M. dypterygia* and *S. mentella*, commercially important fish which are categorized as Vulnerable and Endangered by the IUCN, respectively. In Breiðafjörður and Faxaflói (cluster #2) a relatively high biomass of Atlantic halibut *Hippoglossus hippoglossus*, a species classified as Vulnerable, is present. Atlantic halibut is rare species around Iceland: catches declined from 1960 until 2012, when a fishing ban was introduced for the species (MFRI 2025). Finally, the priority areas in the southeast are largely defined by the occurrence of *C. squamosus*, a deepwater shark that is classified as Endangered and that historically has been commercially exploited in the northeast Atlantic (Finucci et al. 2024). The enforcement of measures to conserve these species could, in addition, also protect the habitat to which they are associated, possibly protecting a number of species that have not been considered in this study. For instance, *M. dypterygia* and *Sebastes* spp. are known to be associated with cold-water coral reefs (Ragnarsson et al. 2018; Ragnarsson and Burgos 2018), which three-dimensional structure sustain high levels of biodiversity (Mortensen & Fosså 2006; Soest et al. 2007; Weaver et al. 2015; Henry & Roberts 2016).

The fishery-restricted areas presented here have been designed to protect specific habitats (i.e., cold-water coral reefs) or to protect specific species during the spawning season (Ólafsdóttir et al. 2024). To enable a comprehensive protection of the entire groundfish community, it would be necessary to enforce a permanent ban on bottom-contact fishing gear in the most important locations identified in this analysis, i.e. all locations with a priority ranking ≥0.95. If such an extensive measure is not possible to implement, then the most urgent mismatches between protection measures and priority areas would be cluster #2, #5 and #6, as in these clusters Arctic species sensitive to climate change and fishing are found. In these areas permanent bans would be advisable, to protect the variety of shallow- and deep-water species that can be found therein and possibly at different times of the year. Concerning clusters #1, #3 and #4, protection measures in these locations can be tailored depending on the spatiotemporal distribution of species that defined them, i.e. to provide protection during months in which these species are present or spawning.

6 North Sea: epibenthic vulnerabilities and MPAs

6.1 Introduction

Marine protected areas (MPAs) represent an essential aspect of European and international strategies to protect marine biodiversity and counter biodiversity loss. The current MPA network in Europe is intended to provide adequate protection for sensitive habitats and species while maintaining ecosystem services (European Environment Agency 2024; European Commission 2023). However, the extent to which MPAs overlap with areas of high ecological vulnerability or risk remains uncertain. Addressing this question is critical for evaluating whether existing MPAs are fit for their purpose and for identifying priority areas where additional protection could yield the greatest biodiversity benefits.

In the North Sea, benthic invertebrate communities are exposed to multiple human and environmental pressures, including bottom trawling, eutrophication, and climate-driven warming. Traits such as trawling vulnerability and temperature vulnerability offer a way to characterize community-level risks (Beauchard et al. 2021; Polo et al. 2024). Also habitat properties including sediment grain size, organic matter content, and hydrodynamic energy modulate the resilience of benthic ecosystems to disturbance (Tiano et al. 2021, 2024). By combining community-weighted trait indices with environmental data, it is possible to identify both vulnerable communities and sensitive habitats, and to examine their distribution in relation to the MPA network.

This chapter addresses Task 4.3 of the B-USEFUL project by quantifying the match and mismatch between North Sea benthic vulnerability hotspots and existing MPAs. First, using data on benthic organisms sampled routinely during beam trawl surveys in the North Sea, we derived community- and habitat-level indicators of sensitivity, recoverability and vulnerability to trawling and temperature stress. These indicators were mapped across the North Sea, to identify areas where benthic communities exhibit elevated vulnerability. Second, we defined "sensitive habitats" by integrating biological vulnerability with key environmental characteristics known to be linked with higher disturbance susceptibility. Third, maps were overlaid with the harmonised MPA dataset covering the North-East Atlantic, including the United Kingdom, which was compiled for this work package (see Chapter 2 for details). This MPA dataset, assembled from multiple recent sources incorporates information on known pressure-specific regulations assigned to MPAs within the region. Finally, we evaluated the degree of spatial match and mismatch between high-risk benthic areas and designated MPAs. The resulting comparison provides insight into whether current MPAs sufficiently encompass high-risk benthic areas, and highlights potential gaps where future conservation efforts could be targeted to maximise protection.

6.2 Methods

6.2.1 Survey data

Benthic invertebrate data were obtained from the International Council for the Exploration of the Sea (ICES) Beam Trawl Survey (BTS), conducted annually across the North Sea in late summer (Quarter 3). All hauls containing invertebrate records from 2000–2024 were extracted from the ICES DATRAS database. Taxonomic identifications were harmonised, and abundance per haul was log-transformed to reduce the influence of highly abundant taxa and to stabilise variance across samples, a common approach in community ecology analyses (Legendre & Gallagher 2001), allowing greater emphasis of scarcer taxa (potentially of conservation value).

6.2.2 Community-weighted trait indices

Two independent community-level vulnerability indices were developed: *vulnerability to trawling* from Beauchard et al. (2021) and *vulnerability to temperature* which was derived from Polo et al. (2024). Trait values were assigned at the species level, based on the published trait databases and expert knowledge. For each survey haul (sampling location), community-level trawling vulnerability and temperature vulnerability were calculated. This was done by weighting species-specific trait values by the log-transformed abundance of each taxon and then summing these to derive a community-weighted trait score. These scores were re-scaled between 0–1 to facilitate comparison across hauls.

We also aimed at identifying the most "overall vulnerable communities" based on trawling vulnerability and temperature vulnerability combined. This was done by combining the re-scaled trait scores for trawling and temperature vulnerability into one '*combined vulnerability index*'. Only hauls within the upper 5% of combined-vulnerability values were then retained, to highlight areas where communities were most vulnerable to either fishing disturbance or thermal stress, or both (conform section 3.2 for the Mediterranean Sea). The 95th percentile threshold was chosen to highlight the most extreme vulnerability values while retaining sufficient spatial coverage and resolution.

6.2.3 Sensitive habitats

Next, we aimed at identifying the most "*sensitive habitats*" based on not only combined vulnerability (trawling and temperature vulnerability) but also key physical factors known to increase a sediment's or habitat's disturbance susceptibility (Tiano et al. 2021, 2024; De Borger et al. 2021). To select the most sensitive habitats, hauls were filtered according to the following abiotic and biotic criteria linked with high disturbance susceptibility:

(1) *Sediment grain size* in the lowest 25% quartile (fine sediments),
(2) *Organic matter content* in the highest 25% quartile,
(3) *Bottom current velocity* in the lowest 25% quartile, and
(4) *Community vulnerability score* in the highest 25% quartile of combined trawling and temperature vulnerability.

These abiotic criteria were selected because they influence benthic habitat susceptibility to physical disturbance and post-trawling recovery. Fine-grained sediments are less stable and recover more slowly, as resuspension and sediment reworking can persist longer than in coarser substrates. Low bottom current velocity reduces sediment transport and flushing, potentially slowing redistribution of sediments and organic matter and delaying recovery. High organic matter content can also support communities with traits linked to greater disturbance sensitivity (e.g., longer life spans, lower mobility, or bioturbation-dependent roles). Together, fine sediments, high organic matter, and low hydrodynamic energy are widely recognised as indicating benthic habitats with high sensitivity to physical disturbance and slower recovery (e.g. Tiano et al. 2021, 2024; De Borger et al. 2021).

Only the locations of hauls meeting all four criteria were classified as *sensitive habitats.* These were mapped separately by decade to evaluate persistence and spatial distribution of sensitive habitats through time.

6.2.4 Marine protected areas

Spatial data on existing MPAs were obtained from the newly compiled and harmonised dataset developed within WP4.3 of the B-USEFUL project (Chapter 2). This dataset integrates multiple recent MPA sources, including an updated classification of management regimes following Aminian-Biquet et al. (2024a), and was intersected with the marine domain to retain only marine portions of designated sites. The dataset covers the North-East Atlantic, including the United Kingdom (Sim 2025; albeit currently not yet including Norway). It includes polygon-level information on pressure-specific management categories (e.g. allowing fishing, dredging, infrastructure; see Chapter 2 for full details). For the present analysis, MPAs were spatially subset to the North Sea region (here delineated by 50°–60°N, 5°W–11°E).

6.2.5 Spatial analysis of overlap

The most vulnerable communities and sensitive habitats were overlaid with the North Sea MPA network using the sf package in R. Overlap was quantified by calculating the proportion of sensitive communities and habitats located inside MPAs, and by comparing vulnerability scores inside and outside MPA boundaries. In addition, pressure-specific management information associated with MPAs (e.g. fishing and dredging compatibility) was used to qualitatively assess whether sensitive benthic areas occurring within MPAs were subject to restrictions relevant to benthic disturbance. This allowed for the identification of both matches (sensitive areas within MPAs) and mismatches (sensitive areas outside MPAs), providing an assessment of the extent to which current MPAs encompass highly vulnerable benthic communities and habitats.

All plotting and statistical analyses were performed in R version 4.4.1 (R Core Team 2025).

6.3 Results

6.3.1 Spatial patterns in trawling vulnerability and temperature vulnerability

Community-weighted trait scores revealed distinct spatial patterns between trawling vulnerability and temperature vulnerability (*Figure 6-1*, *Figure 6-2*). Communities generally had higher scores for trawling vulnerability than for temperature vulnerability. There was a broader distribution of moderate to high trait scores across much of the central and southern North Sea. Temperature vulnerability showed more lower scores overall with a few localised hotspots of higher vulnerability such as the area near the Dutch coast. Both indices showed overlapping vulnerability in certain areas, notably the southern coastlines (both UK and Netherlands) and localised areas within the Dogger Bank.

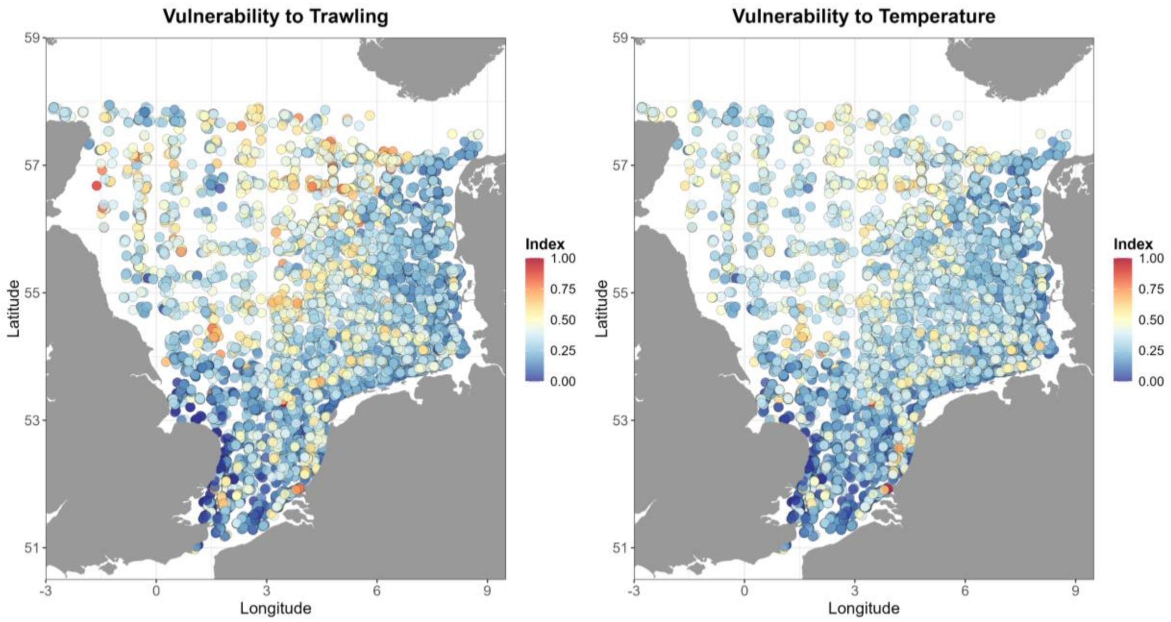

Figure 6-1. Community weighted trawling vulnerability scores (following Beauchard et al. 2021; left), and community weighted temperature vulnerability scores (following Polo et al. 2024; right) in the North Sea using BTS survey data from 2000-2024.

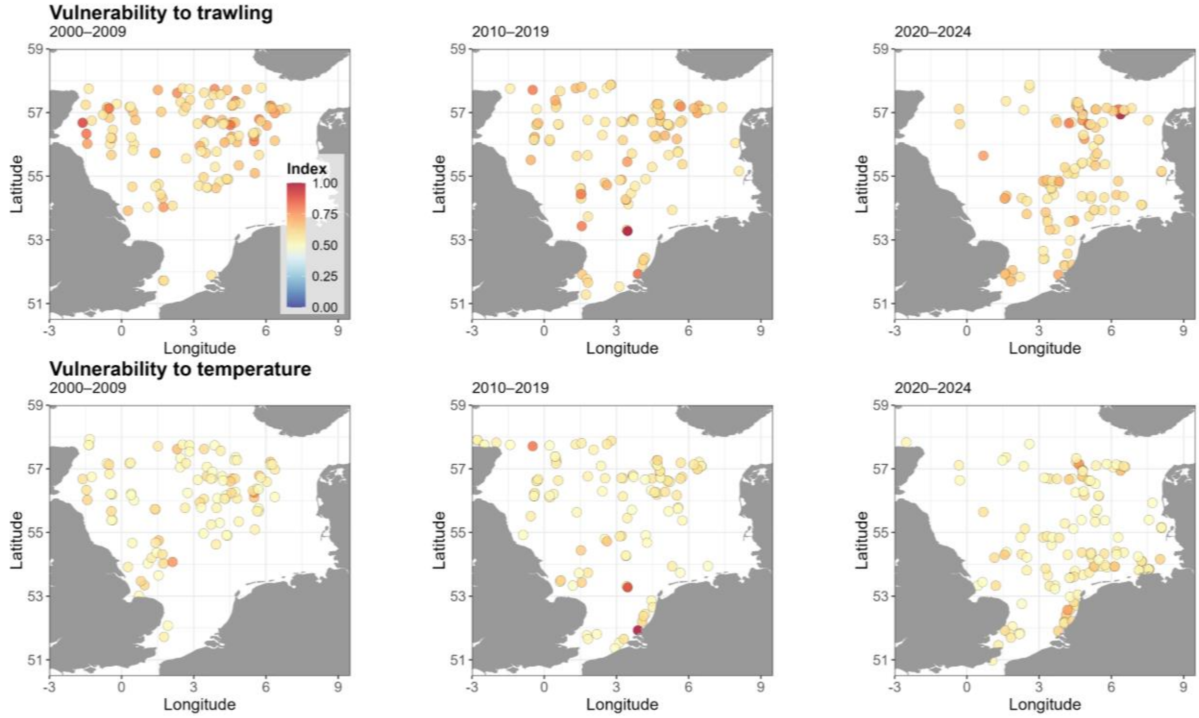

Figure 6-2. Upper 5% community-weighted scores for trawling vulnerability (Beauchard et al. 2021; top) and temperature vulnerability (Polo et al. 2024; bottom) in the North Sea, shown for the 2000-2009, 2010-2019, and 2020-2024 eras.

6.3.2 Hotspots of trawling vulnerability and temperature vulnerability

In order to distinguish the locations where benthic communities have the highest trawling vulnerability and/or temperature vulnerability, we restricted our analysis to the 5% highest community-weighted trait scores for either stressor (*Figure 6-2*). This revealed marked shifts in the locations of highest trawling vulnerability and highest temperature vulnerability over the past three decades. The most trawling-vulnerable communities were in the northern and north-western North Sea during 2000–2009, became more spatially dispersed during 2010–2019, then shifted towards the south and south-east by 2020–2024. Temperature vulnerability showed a similar temporal progression, with the most warm- vulnerability communities mainly in northern and western areas during 2000–2009, but by 2020–2024, mainly in southern and eastern areas. Hence for both trait groups, the spatial distribution of the top 5% of vulnerability scores became more prominent in southern and coastal regions in the most recent decade (*Figure 6-2*, *Figure D-1*) .

The overall most vulnerable benthic communities (*Figure 6-3*) were defined by combining trawling vulnerability and temperature vulnerability traits and selecting only the top 5% in combined vulnerability scores. The distribution of most combined-vulnerable communities shifted from a concentration in the northern North Sea during 2000–2009, via a more widespread pattern in 2010–2019, to a predominantly southern and eastern distribution in 2020–2024.

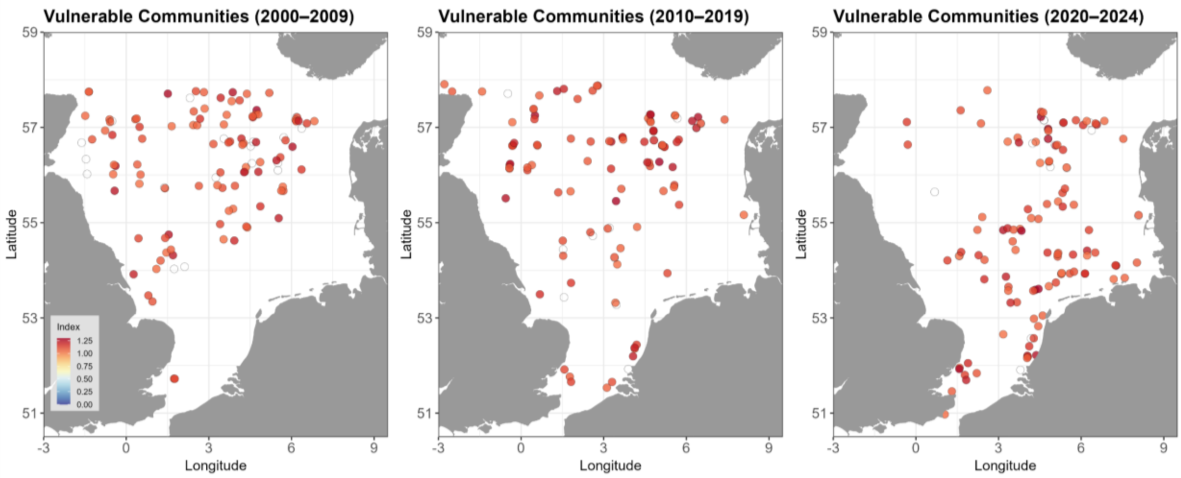

Figure 6-3. Upper 5% overall most vulnerable benthic communities, based on combined trawling vulnerability and temperature vulnerability, in the 2000s, 2010s and 2020s.

6.3.3 Hotspots of habitat sensitivity

The most sensitive habitats – filtered based on trawling vulnerability, temperature vulnerability as well as physical habitats characteristics (low grain size, low current velocity, high organic matter) – were found to be more localised (*Figure 6-4*). Sensitive habitats were primarily along the eastern UK coast north of the Dogger Bank and in the German Bight (around Heligoland). The extent of sensitive habitats in the western North Sea seems to have decreased over time.

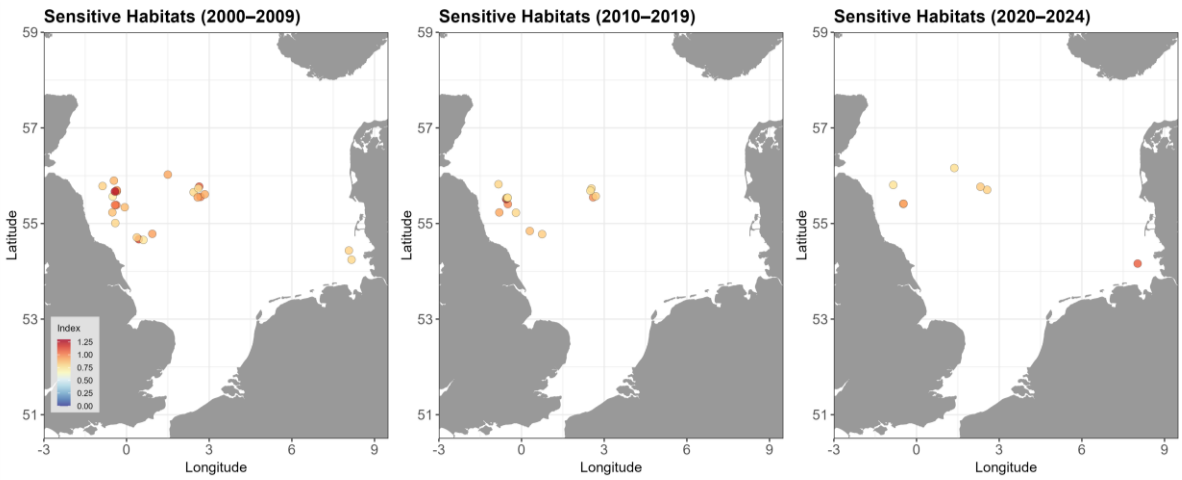

Figure 6-4. Sensitive habitats, filtered for known sensitive characteristics to seafloor perturbations (small grain size, low current velocity, high organic content) and high community vulnerability in the 2000s, 2010s and 2020s.

6.3.4 Match/mismatch of hotspots with MPAs

Out of a total of 47 highly sensitive habitats identified in this analysis for the North Sea, only ~8.5% (4 of a total of 47) were located inside MPAs. Three of these were located within the German Bight, and the fourth was located in the UK OSPAR MPA "Southern North Sea" (*Figure 6-5*). There was no significant difference in mean vulnerability trait scores of the community criterion (in habitat sensitivity definition, section 6.2.3) between habitats inside and outside MPAs, though the value was slightly higher inside MPAs (~0.89 outside MPAs and 0.91 inside MPAs; Mann–Whitney U test, $p > 0.05$).

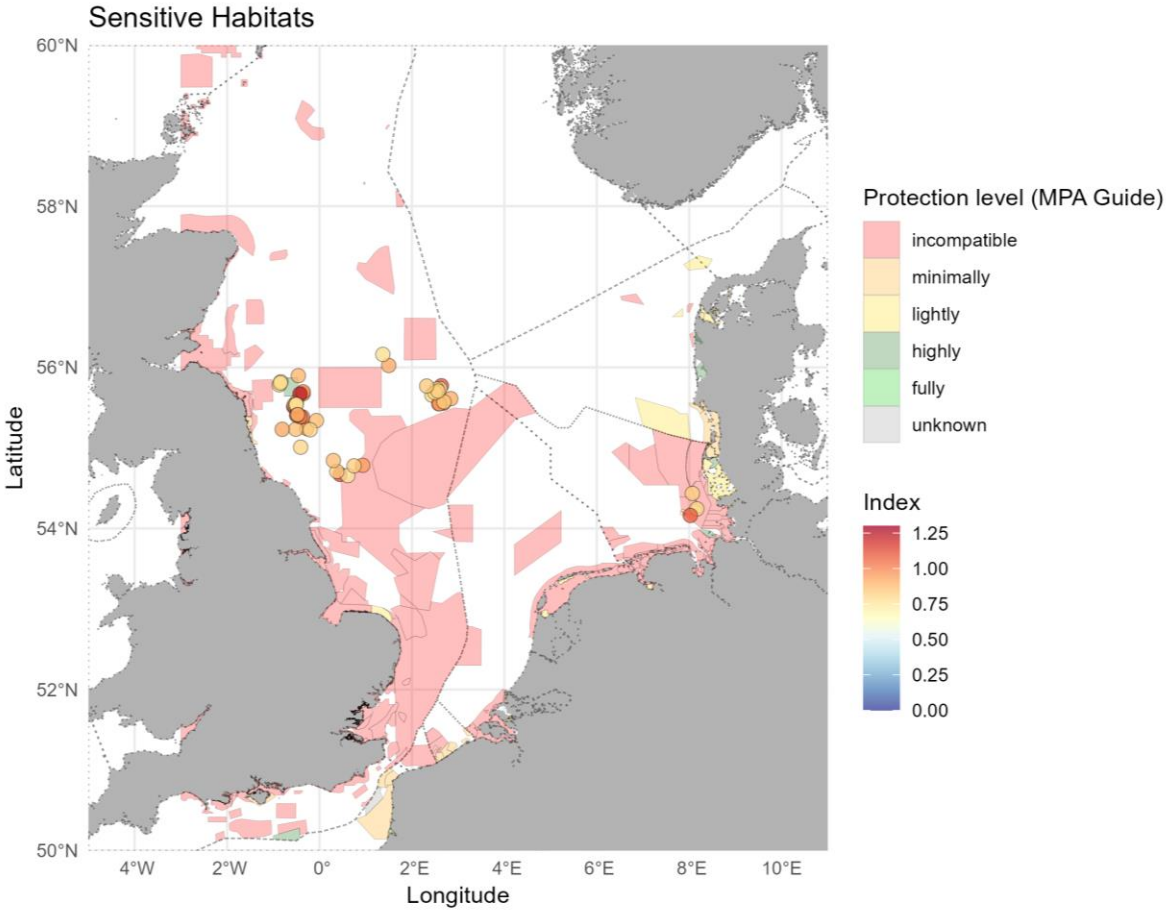

Figure 6-5. North Sea MPAs overlayed with highly sensitive habitats, filtered for characteristics known to be linked with susceptibility to seafloor perturbations (small grain size, low current velocity, high organic content), with colour within circles representing habitat sensitivity scores. Polygons represent official MPA boundaries colour-coded by their overall protection level (from minimal to full) or by areas classified as 'incompatible' for extractive activities to coincide with biodiversity conservation objectives under the MPA Guide framework (Oregon State University et al. 2023). Dotted lines specify individual country EEZ boundaries.

When considering vulnerable biological communities regardless of habitat type (defined as those with highest 5% combined trawling vulnerability and temperature sensitivity scores), approximately 19% (n = 24 out of 126) were found inside North Sea MPAs (*Figure 6-6*). Of the vulnerable communities found within MPA boundaries, 33% occurred within the relatively large Southern North Sea OSPAR MPA, 25% within the Dogger Bank, and 12.5% within the Frisian Front.

At the country level, 63% of all vulnerable communities identified were found in UK-controlled MPAs, 29% in Dutch MPAs, and the remaining 8% in German MPAs in the North Sea. For vulnerable communities, the mean vulnerability trait score was only slightly higher inside MPAs compared to outside, with no statistically significant difference (1.17 inside vs. 1.16 outside; Mann–Whitney U test, $p > 0.05$).

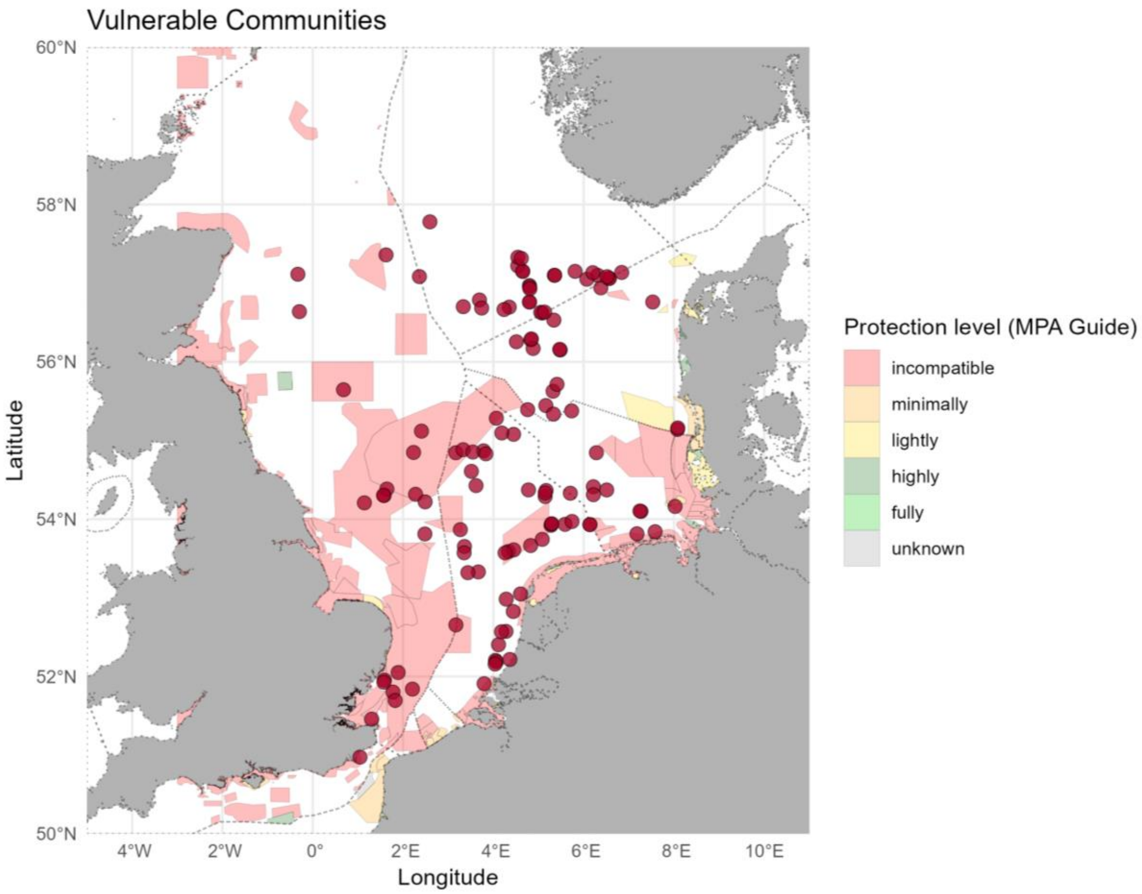

Figure 6-6. North Sea MPAs overlayed with the areas showing the top 5% biological community vulnerability in the most recent era (2020-2024). Polygons represent official MPA boundaries color-coded by their overall protection level (from minimal to full) or by areas classified as 'incompatible' for extractive activities to coincide with biodiversity conservation objectives under the MPA Guide framework (Oregon State University et al. 2023). Dotted lines specify individual country EEZ boundaries.

At the country level, 63% of all vulnerable communities identified were found in UK-controlled MPAs, 29% in Dutch MPAs, and the remaining 8% in German MPAs in the North Sea. For vulnerable communities, the mean vulnerability trait score was only slightly higher inside MPAs compared to outside but without any statistically significant difference (1.17 inside vs. 1.16; Mann–Whitney U test, $p > 0.05$).

Among the vulnerable biological communities located within MPAs (n = 24), management information showed that 62.5% of identified communities occurred in MPAs where fishing activities were deemed “incompatible” for the conservation of biodiversity, as defined by the IUCN (Oregon State University et al. 2023; *Table **6**-**1***). In comparison, 29% of communities were in MPAs which exhibited “light” protection from fishing and 8% with “minimal” protection. For dredging, 71% of communities were inside MPAs which were fully protected compared with 8% which were considered incompatible, and 21% which were lightly protected. Among the identified communities within MPAs, 66% were in areas incompatible with conservation, while 33% were in areas fully protected against mining operations. With regards aquaculture, 41% and 25% contained either fully or highly restricted aquaculture regulations. This indicates that vulnerable epibenthic communities are in MPAs of highly variable levels of protection concerning these important pressures, with an emphasis on low protection. Regarding protections against offshore development and infrastructure, 21% were considered fully protected, 8% highly protected, and 71% lightly protected. For the comprehensive ‘overall protection’ category, all 24 highly vulnerable communities located within MPAs were actually within MPAs deemed ‘incompatible with conservation objectives’. Protection levels against human activities found within North Sea MPAs are summarised in *Table 6-1*.

Of the four sensitive habitats found within North Sea MPAs, fishing was deemed incompatible in the Southern North Sea MPA, while exhibiting minimal protection in the Steingrund and Heligoland Seabird Protection Area, and light protection in the Eastern German Bight SPA. These sensitive areas inside MPAs received full protection from aquaculture developments, dredging and mining only in the Southern North Sea MPA compared to habitats found in the Steingrund, Heligoland Seabird Protection Area and Eastern German Bight SPA, which feature either light or minimal protection from dredging activity and no specified protections for aquaculture or anchoring. In contrast, the Southern North Sea MPA is only lightly protected against offshore infrastructure compared to the other three MPAs with sensitive habitats which feature full protection. In terms of overall protection, 3 out of the 4 sensitive habitats within MPAs deemed incompatible for extensive extractive activities while one (Steingrund) was categorized as minimally protected (*Table 6-1*).

Table 6-1. Protection levels of North Sea MPAs to human activities. Protection levels based on MPA guide classification (Chapter 2, Oregon State University et al. 2023). MPAs classified as: fully protected (no extractive activities), highly protected (low impact activities), lightly protected (moderate impact activities) or minimally protected (moderate to high impact activities), or incompatible with biodiversity conservation (when very impactful or industrial activities can occur within the MPA) (Aminian-Biquet et al. 2024a; Sim 2025).

Name of MPA	Anchoring	Aquaculture	Infrastructure	Fishing	Mining	Dredging	Overall
North Norfolk Sandbanks and Saturn Reef	lightly	highly	highly	incompatible	incompatible	lightly	incompatible
Dogger Bank (UK)	lightly	highly	lightly	incompatible	incompatible	lightly	incompatible
Southern North Sea (UK)	lightly	fully	lightly	incompatible	fully	fully	incompatible
Kentish Knock East	lightly	highly	lightly	incompatible	incompatible	fully	incompatible
Outer Thames Estuary	lightly	highly	minimally	incompatible	fully	incompatible	incompatible
Margate and Long Sands	lightly	fully	lightly	incompatible	incompatible	incompatible	incompatible
Swallow Sand	lightly	fully	highly	incompatible	incompatible	fully	incompatible
Sylt Outer Reef	NA	fully	fully	incompatible	incompatible	lightly	incompatible
Fulmar	lightly	fully	lightly	incompatible	incompatible	fully	incompatible
Southern Trench	lightly	fully	fully	incompatible	fully	fully	incompatible
Dungeness, Romney Marsh and Rye Bay	lightly	highly	highly	incompatible	incompatible	incompatible	incompatible
Schleswig-Holstein Wadden Sea	NA	highly	fully	incompatible	incompatible	lightly	incompatible
East of Gannet and Montrose Fields	lightly	fully	minimally	incompatible	incompatible	fully	incompatible
Cleaver Bank	NA	NA	lightly	lightly	incompatible	fully	incompatible
Frisian Front	NA	NA	lightly	lightly	incompatible	fully	incompatible
Dogger Bank (NL/DE)	NA	NA	fully	lightly	incompatible	fully	incompatible
Eastern German Bight SPA	NA	NA	fully	lightly	incompatible	lightly	incompatible
Southern North Sea (Danish)	NA	lightly	lightly	lightly	fully	NA	lightly
Haisborough, Hammond and Winterton	fully	highly	lightly	lightly	incompatible	incompatible	incompatible
Lower Saxony Wadden Sea	NA	highly	fully	minimally	incompatible	lightly	incompatible
Heligoland Seabird Protection Area	NA	NA	fully	minimally	incompatible	lightly	incompatible
Flemish Banks	NA	NA	lightly	NA	incompatible	lightly	incompatible
Brown Bank	NA	NA	lightly	NA	incompatible	NA	incompatible
Steingrund (Stone Ground)	NA	NA	fully	minimally	NA	minimally	incompatible

6.4 Discussion and conclusions

By combining habitat sensitivities (section 6.2.3) and trait-based community vulnerability indices (section 6.2.2), we were able to identify hotspots of risk for the epibenthos in the North Sea, and evaluate to what extent they are protected under the existing MPA network and regulations. Overall, the results suggest that the MPA network offers partial but incomplete coverage of vulnerable epibenthic habitats and communities. Only 8.5% of identified highly sensitive habitats and 19% of highly vulnerable communities were found inside North Sea MPAs. While this indicates that MPAs do capture some areas of elevated vulnerability, the majority of sensitive locations and communities remain outside designated protected zones. Notably, the German Bight and parts of the Southern North Sea OSPAR MPA encompassed several high-vulnerability communities. Many sensitive habitats along the UK coast and central North Sea fell outside the current MPA network.

There were no significant differences in vulnerability trait scores between sensitive habitats or vulnerable biological communities (no other habitat metrics) located inside and outside MPAs. This suggests that current MPAs may not be systematically targeting the most sensitive areas at a broad scale, or that ecological recovery resulting from relatively recent restrictions has not yet manifested in the survey data. Although MPAs overlap with some vulnerable benthic communities, their spatial footprint does not preferentially encompass the majority of the most recent vulnerability hotspots identified using trait-based indicators for epibenthos. However, it should be kept in mind that MPAs might be established for other species (groups) than epibenthos (see below).

Sensitive habitats characterised by fine sediments with low currents and high organic matter, conditions known to amplify trawling impacts (De Borger et al. 2021; Tiano et al. 2024), were disproportionately found outside MPAs. This indicates a gap in coverage for (epi)benthic ecosystems most susceptible to disturbance. Similarly, the shift in vulnerable community distribution over time from generally the northern to the southern North Sea within the 21st century highlights the importance of spatially dynamic threats, which static MPAs are not always well-equipped to fully accommodate (Cashion et al. 2020). The temperature vulnerability increased in those areas which had the highest temperature rise (south-eastern North Sea, alike the temperature rises described in Cornes et al. 2023), indicating a shift in species already adapted to those higher temperatures (Rozemeijer et al., 2025).

Importantly, incorporating information on pressure-specific management regimes within MPAs revealed that spatial overlap alone does not equate to effective protection from benthic disturbance. Among vulnerable biological communities located within MPAs, a substantial proportion occurred in areas where fishing activities were classified as incompatible with biodiversity conservation objectives (62.5%), and a considerable fraction was still located in MPAs offering only light or minimal fishing restrictions (37%) so offering minimal protection for vulnerable epibenthos. Of the vulnerable North Sea epibenthic communities located within MPAs, 71% occurred in areas classified as fully protected from dredging activities, indicating a relatively high level of regulatory restrictions for this pressure, but protection levels against offshore infrastructure and aquaculture varied widely among sites. Bottom fisheries and dredging represent an alike impact of removing the epibenthic species. However, it should also be considered that in *Table 6-1* fishing is not diversified in e.g. pelagic fisheries and bottom-contact fisheries nuancing potentially our results.

More importantly, the large majority of areas identified to contain vulnerable benthic communities (81%) or sensitive habitats (91.5%) were located outside of MPAs. These findings are consistent with recent pan-European assessments showing that many MPAs impose limited restrictions on activities most relevant to benthic ecosystems (Aminian-Biquet et al. 2024b).

Caution is required regarding the classification of MPAs as “incompatible” with extractive activities under the MPA Guide framework. This designation indicates that the presence of highly impactful human activities is considered inconsistent with stated conservation objectives, but it does not provide detailed information on the spatial extent, seasonal timing, or gear-specific restrictions that may apply within individual MPAs (Aminian-Biquet et al. 2025). In practice, many North Sea MPAs are subject to complex management regimes, where restrictions may apply only to specific zones, certain bottom-contacting gears, or limited periods of the year. Such nuance is not fully captured in the harmonised shapefiles used in this analysis, which simplify protection status to polygon-level categories. The subsequent step should incorporate this detailing on the actual management per MPA, its target groups and its subsequent match with the studied group. Given the enormous number of MPAs (Chapter 2) a limited approach should be chosen with a selected group of MPAs.

In addition, the underlying rationale for protection and assigning MPAs should also be taken into account. For example, in the Netherlands, potential MPAs were initially assigned based on the biodiversity of benthos, fish, birds and sea mammals and rare habitat characteristics. An area was selected if it had combinations of several animal groups or very specific habitat characteristics (Lindeboom et al. 2005). In many cases, epibenthos was not considered in these evaluations; consequently, areas important in terms of benthic fauna may not be included in Dutch MPAs. So original arguments to assign MPAs as well as the degree of measures to protect both target criteria and the measured groups of the research and application of the methodology matter in the final evaluation.

Moreover, MPA regulations are dynamic and have undergone substantial revisions in recent years, including new North Sea fisheries closures introduced since 2023. As a result, the protection classifications applied here may not reflect the most recent management measures in all cases. Our analysis therefore covers more general aspects of alignment between vulnerable communities and areas designated for biodiversity conservation, rather than a definitive assessment of effective protection.

It is also important to recognise that community-weighted vulnerability indices are inherently influenced by local species richness. Sites with higher richness have a greater probability of containing species with elevated trait-based vulnerability scores, which can increase community-weighted vulnerability values. This does not invalidate the spatial patterns identified but it explains some similarities between temperature and trawling related vulnerability trait characteristics while suggesting that vulnerability hotspots partly reflect areas of elevated biodiversity, and should be interpreted in that ecological context.

Nevertheless, from a management perspective, the mismatch between vulnerable communities and sensitive habitats and MPA coverage has several implications. Spatial planning for future MPAs or adaptive expansion of existing ones should prioritise areas of overlap between multiple stressors and sensitive communities, particularly in the southern and eastern North Sea (see Chapter 4). Given that the majority of vulnerable habitats and communities currently lie outside MPAs, achieving the EU Biodiversity Strategy target of protecting 30% of marine areas, and the aligned commitment to do so by the UK, will likely require both the expansion of the existing MPA network and a more strategic placement of protected areas based on ecological risk, rather than spatial coverage alone.

A complicating factor is that sites of highest vulnerability (and associated species richness) may shift spatially in time, as was observed here (*Figure 6-2*, *Figure 6-3*, *Figure 6-4*). Similar results were also seen for harbour porpoises (an important target species for Dutch MPAs). Their distribution pattern shifted in time (Rozemeijer et al., 2026) and thereby also the reason for assigning MPAs (Lindeboom et al., 2005). In general, MPAs have typically been established after comprehensive negotiations (e.g. Aminian-Biquet et al., 2024b) and are spatially fixed. Ideally a flexible approach should be applied addressing multiple criteria of both biological values and stakeholder interests.

Finally, the use of trait-based sensitivity indicators offers a transparent and scalable method for identifying vulnerability hotspots. This approach provides a robust scientific basis for aligning spatial conservation planning with ecosystem functioning, and for evaluating not only where MPAs are located, but how effectively they mitigate pressures on vulnerable benthic ecosystems. The match between target groups of the MPAs and measures (in time and space) also matters. The approach makes clear that both assigning and managing MPAs is a complex matter weighing the interests of stakeholders, management and nature. It adds additional previously unused information on vulnerability of species groups, where sites of highest vulnerability may shift in space and time. The approach both informs the evaluation and decision making and complicates it. However, it is arguably preferable to engage consciously and make active decisions in complex contexts than to remain comfortably ignorant.

7 Integrating climate change risk into benthic habitat conservation planning

7.1 Introduction

Climate change is reshaping marine habitats through rising temperature, ocean acidification, sea-level rise and deoxygenation, with each habitat type responding differently depending on its sensitivity (Brierley & Kingsford 2009). These changes alter species distributions, disrupt food webs, and threaten ecosystem services (e.g., Hodapp et al. 2023, Runting et al. 2017, Thompson et al. 2023).

A framework for assessing risks to marine benthic habitats and the biodiversity they support, posed by climate change and fishing pressure, was developed under B-USEFUL deliverable 4.2. It evaluates the sensitivities of 19 habitats to different stressors by integrating species-, community-, and habitat-scale responses (Section 8 from Rozemeijer et al. 2025). The assessment showed that three climate-change stressors—rising water temperature, increasing ocean acidity, and decreasing dissolved oxygen—affect most marine benthic habitats regardless of depth. Structurally complex habitats of biological origin (e.g., biogenic reefs, maërl beds, macroalgae forests, seagrass meadows, and aggregations that alter soft-sediment physiography) and habitats with high physical complexity (e.g., rocky reefs and seamounts) were highly sensitive to these stressors. This high sensitivity likely reflects the sessile or low-mobility species that form such habitats (e.g., gorgonians, corals, sponges, sea anemones and other small invertebrates), some adapted to narrow depth ranges and with limited capacity to escape or adapt (Hutchings et al. 2007, Turley et al. 2007, Butt et al. 2022). With the exception of abyssal plains (sensitive to lowered oxygen and warming), soft-sediment habitats were generally less sensitive, particularly in intertidal zones and deeper shelf and slope areas (Section 8 from Rozemeijer et al. 2025), likely because they include species spanning a wide range of tolerances, from highly sensitive to opportunistic and tolerant (Borja et al. 2000). The assessment also indicated that warming and ocean acidification were consistently linked with higher sensitivity scores, whereas reduced dissolved oxygen tended to be linked with lower scores. Habitats most sensitive to warming included macroalgae forests, followed by biogenic reefs, and mud volcanoes and cold seeps (Section 8 from Rozemeijer et al. 2025).

As ocean acidification reduces calcium carbonate available to build and maintain organisms' shells and skeletons (Byrne and Fitzer, 2019), habitats assessed as most sensitive to ocean acidification were those dominated by calcifying species, particularly biogenic reefs, as well as maërl beds, which are dominated by calcareous algae. Habitats formed by aggregations that change the physiography in soft sediment, and various rocky reef habitats had relatively higher sensitivity scores to increasing ocean acidity, reflecting their high biodiversity of calcifying species. Habitats assessed as least sensitive to ocean acidification include abyssal plains, mud volcanoes and hydrothermal vents, and deep-sea environments with species generally adapted to lower pH conditions (e.g., Gollner et al. 2010, Mullineaux 2014).

B USEFUL Deliverable 4.2 highlighted the need to combine ecological sensitivity assessments with spatial information on exposure to pressures to better identify higher risk areas. Building on this work, the present study integrates Deliverable 4.2 sensitivity outputs with spatial data on the distribution of these habitats (*Figure 7-1*). Combined with information on their exposure to climate change stressors (changes in temperature and pH over time), this is used to identify where habitat risk is elevated. These high-risk areas are then compared with the extent of Marine Protected Areas (MPAs), accounting for varying levels of protection (*Figure 7-1*), to estimate the share of vulnerable habitats currently under formal protection. The resulting risk maps and quantified overlap with MPAs provide insights to support more effective, habitat specific management under climate change.

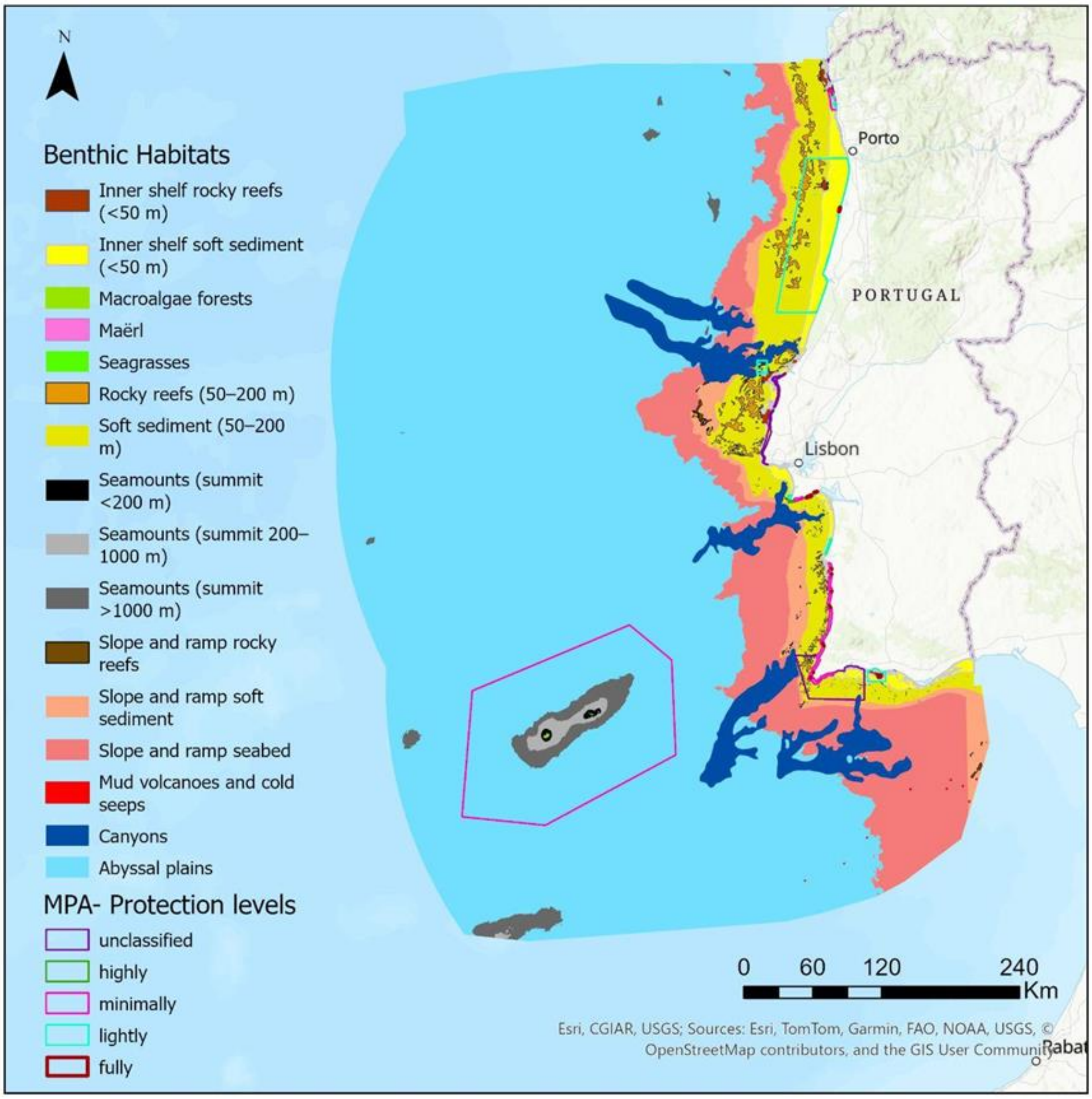

Figure 7-1. Habitat types in continental Portugal's EEZ subarea. Habitats are listed following Section 8 of D4.2 (Rozemeijer et al. 2025), but available data did not support mapping, so risk was not estimated; hydrothermal vents are absent. "Slope" and "ramp" seabed denote areas where data could not distinguish rocky reef from soft sediment. MPA limits are also shown by protection level: fully, highly, lightly, minimally, and unclassified but designated.

7.2 Methods

7.2.1 Climate change stressors

Data on temperature (T), dissolved oxygen (O_2) and pH were obtained from numerical model solutions provided by the Copernicus Marine Service: the Atlantic-Iberian Biscay Irish Ocean Physics Reanalysis (IBI_MULTIYEAR_PHY_005_002) and Ocean BioGeoChemistry non assimilative Hindcast (IBI_MULTIYEAR_BGC_005_003), implemented by the Atlantic-Iberian-Biscay-Irish Monitoring and Forecasting Centre (IBI MFC); and the Global Ocean Physics Re-analysis (GLORYS12V1, GLOBAL_MULTIYEAR_PHY_001_030) and Biogeochemistry Hindcast (GLOBAL_MULTIYEAR_BGC_001_029) implemented by Mercator Ocean International.

Monthly data cubes (latitude, longitude, depth) for January 1993–December 2014 (384 months) covering 34.5°N–44.0°N and 14°W–7°W were compiled from the above source products and processed to extract values at the deepest depth level. The same approach was applied to the three variables (T, O_2, pH) to generate bottom-value layers. Although bottom temperature is provided separately ("bottomT"), this method enables analysis of IBI bottom temperatures for depths >~300 m not included in the distributed bottomT layer and applies a consistent procedure across variables. Thirty-two-year trend maps for temperature, dissolved oxygen and pH (and p-values) were calculated using the Mann–Kendall (MK) trend test (Hirsch et al. 1982) implemented as the "Seasonal MK Test" in the pyMannKendall python module (Hussain et al. 2019). A p-value <0.1 was used as the threshold for significance.

The results using the higher resolution IBI models (both "PHY" and "BGC" at 1/36° horizontal resolution) and the coarser global ocean models (GLO "PHY" at 1/12° and "BGC" at 1/4° horizontal resolutions) were compared to check for the consistency in the trend maps.

There was relatively low confidence in the oxygen trends, as discussed in Section 7.3.1 (Climate Change Stressor Trends). This stressor was therefore excluded from the risk assessment, and the analysis focused on pH and temperature.

7.2.2 Habitat mapping

Benthic habitats within the study area were mapped according to the habitat classification in Stratoudakis et al. (2019), also adopted in B-USEFUL Deliverable 4.2 (Section 8 in Rozemeijer et al. 2025). This classification considered 19 habitats, from now on referred to as "selected habitats", spanning the intertidal to the abyssal zone and encompassing both hard and soft substrate habitats, as well as biogenic and geomorphological features.

Habitat mapping used data from multiple sampling and modelling approaches, including broad-scale predictive habitat models, bathymetric datasets and biological surveys (*Table 7-1*). Given this heterogeneity, two main approaches were applied: (1) substrate- and depth-based mapping using EUSeaMap 2025 (EMODnet Seabed Habitats 2025a) and bathymetry (Instituto Hidrográfico 2010, EMODnet Seabed Habitats 2023), and (2) integration of additional habitat-specific datasets. Procedures were tailored to each dataset, as summarised in *Table 7-1* and detailed in Appendix E. Of the 19 initial habitats, 14 were mapped; "slope and ramp seabed" was used where rocky reefs and soft sediments could not be separated.

Table 7-1. Data sources and procedures applied in habitat mapping by habitat type, with information on the score obtained for each of two climate change stressors analysed (temperature and pH) in B-USEFUL Deliverable 4.2. The references cited in the table are in the appendix E-2. Habitat mapping references.

Benthic habitats	**Data sources**	**Procedures**	**Rationale**	**Sensitivity (Temp)**	**Sensitivity (pH)**
Inner shelf rocky reefs (<50 m)	Instituto Hidrográfico 2010); EMODnet Seabed Habitats (2025a)	(1.1) Classifications correspondence and 50 meters depth limitation	A correspondence between selected habitats and EUNIS 2019 classifications for rocky and soft sediment habitats was generally possible. Since, for rocky and soft sediment habitats, selected habitats classification follows specific depth limits, the polygons originated through correspondence were merged and split by these depth limits.	2.8	3.9
Inner shelf soft sediment (<50 m)				1.2	2.8
Macroalgae forests	Cabral et al. (2025); DivGM IPMA (2024)	(1.2) Selection of rocky reef area above 85 meters depth on the Gorringe seamounts area, in accordance with expedition report (Cabral et al. 2025)	Although the entire rocky area of the seamounts was not sampled for macroalgae forests, they were observed at many sampling points. As these offshore areas are expected to have very high light penetration, it was assumed that the rocky area at depths of less than 85 metres contained macroalgae forests.	5.4	1.9
	EMODnet Seabed Habitats (2025c)	(2.1) Directly updated into map	As habitat extent data was available, it was used directly in the final selected habitats map.		
	Jacinto et al. (2021); MARSW project (2021); EMODnet Seabed Habitats (2025c); Meyer et al. (2025); Franco J et al. (ongoing)	(2.2.1) 250 x 500 meters grid cells containing observations, clipped by high-resolution rocky substrate area (when available)	Since the different datasets (Jacinto et al. 2021, EMODnet Seabed Habitats 2025c, Meyer et al. 2025, Franco J et al ongoing) were obtained through different methodologies, but all originated from points data, a common grid was used. As Franco J et al (ongoing) was the only dataset that used a grid, we used its grid of 250 x 500 metres and fitted it into the extent of the combined datasets. Since high-resolution substrate data was available for the area covered by EMODnet Seabed Habitats (2025c), the grid cells obtained in the aforementioned procedure were clipped using the high-resolution rocky area from MARSW project (2021).		

Benthic habitats	Data sources	Procedures	Rationale	Sensitivity (Temp)	Sensitivity (pH)
	Instituto Hidrográfico (2010); Cabral et al. (2023)	(1.2) Selection of rocky reef area above 50 meters depth on the Camões mountain area, in accordance with expedition report (Cabral et al. 2023)	Although the entire rocky area in the Camões mountain was not sampled for macroalgae forests, they were observed at many sampling points within one of its peaks. As these continental shelf areas are expected to have lower light penetration, it was assumed that the rocky area at depths of less than 50 metres contained macroalgae forests.		
Maërl	CCMAR (2008); Silva et al. (2025)	(2.2.1) 250 x 250 meters grid cells containing observations	Since the two datasets (CCMAR 2008, Silva et al. 2025) were obtained through different methodologies, but both originated from points data, a common grid was used. As Silva et al. (2025) followed a point grid, we used its 250 x 250 meters grid and fitted it into the extent of the combined datasets.	2.7	4.9
	Cabral et al. (2025)	(2.2.2) Observations with 2,5 meters buffer, in accordance with expedition report (Cabral et al. 2025)	The expedition report mentioned that these habitats seemed to extend 2.5 meters from the Remotely Operated Vehicle (ROV) observations.		
Rocky reefs (50-200 m)	Instituto Hidrográfico 2010); EMODnet Seabed Habitats (2025a)	(1.1) Classifications correspondence and 50-200 meters depth limitations	A correspondence between selected habitats and EUNIS 2019 classifications for rocky and soft sediment habitats was generally possible. Since, for rocky and soft sediment habitats, selected habitats classification follows specific depth limits, the polygons originated through correspondence were merged and split by these depth limits.	2.5	3.9
Soft sediment (50-200 m)				1.1	1.8
Seamounts (summit <200 m)	Pitcher et al. (2010); EMODnet Seabed Habitats (2023); EMODnet Bathymetry	(1.2) Areas selection and 200 meters depth limitation, based on available information and a slope raster, computed using bathymetry data	In this study, the selected habitats seamount depth limits (<200m; 200-1000m; >1000m) were used to distinguish between areas within each seamount (further details in this section text). Seamount identification was based on available information (Pitcher et al. 2010, DGRM et al. 2025, IHO-IOC 2026) and on the slope raster. Using this information, polygons from EUSeaMap 2025 were selected to represent the identified seamounts at	3.1	3.7

Benthic habitats	Data sources	Procedures	Rationale	Sensitivity (Temp)	Sensitivity (pH)
	Consortium (2024); DGRM et al. (2025); EMODnet Seabed Habitats (2025a); IHO-IOC (2026)	(EMODnet Bathymetry Consortium 2024).	their corresponding locations. As these polygons follow depth limits, only those extending down to around 2200 metres were considered. This depth limit was chosen because it corresponded to the summit of a "Tore Mount", which was the deepest seamount identified.		
Seamounts (summit 200-1000 m)		(1.2) Areas selection and 200-1000 meters depth limitation, based on available information and a slope raster, computed using bathymetry data (EMODnet Bathymetry Consortium 2024).		2.5	3.3
Seamounts (summit >1000 m)		(1.2) Areas selection and 1000 meters depth limitation, based on available information and a slope raster, computed using bathymetry data (EMODnet Bathymetry Consortium 2024).		3.3	3.5
Slope and ramp rocky reefs	Instituto Hidrográfico 2010); EMODnet Seabed Habitats (2025a)	(1.1) Classifications correspondence and 200 meters depth limitation	A correspondence between selected habitats and EUNIS 2019 classifications for rocky and soft sediment habitats was generally possible. Since, for rocky and soft sediment habitats, selected habitats classification follows specific depth limits, the polygons originated through correspondence were merged and split by these depth limits.	1.8	3.3
Slope and ramp soft sediment				1.2	2.0
Slope and ramp seabed		(1.1) Intermediate classification correspondence and 200 meters depth	The polygons selected here were assumed as “Slope and ramp” because they were located between the areas corresponding to” Slope and ramp rocky reefs” or “Slope and ramps soft sediment” and “Abyssal plains” (see below). Since substrate information	1.5	2.7

Benthic habitats	Data sources	Procedures	Rationale	Sensitivity (Temp)	Sensitivity (pH)
		limitation (further details above)	was not available in EUSeaMap 2025 for these polygons, (classified as “seabed” according to EUNIS 2019), an intermediate selected habitats classification of “Slope and ramp seabed” was assumed, following EUNIS 2019 options when substrate information is not available.		
Mud volcanoes and cold seeps	DGRM et al. (2025); EMEPC (2026)	(2.2.2) Observations with 1000 meters buffer	A 1000 meters buffer around the mud volcanoes observations was selected, to reflect the ecological influence of these structures, according to Levin et al. (2016).	4.4	1.6
Canyons	Harris et al. (2014); (IHO-IOC (2026)	(2.1) Directly updated into map, after canyons selection	Habitat extent was available, but it was based on a geomorphological perspective. Since the selected habitats classification was built around an ecological perspective, we only selected the larger canyon areas extending between the continental shelf, slope and ramp areas, and the abyssal areas. From an ecological perspective, these should be the most distinct canyon areas from the other habitats.	2.2	2.9
Abyssal plains	EMODnet Seabed Habitats (2025a)	(1.1) Classifications correspondence	As a direct correspondence between selected habitats and EUNIS 2019 classifications was not possible (“Na” values), a correspondence with MSFD BBHT classification was used instead. This meant that all “Abyssal” polygons that were not identified as “Seamounts” were assumed as “Abyssal plains”.	3.7	1.5

7.2.3 Risk assessment

A spatial risk assessment framework was developed, to allow evaluation of climate change risk associated with the selected key stressors, i.e., bottom temperature (T_{bot}) and pH. As noted above, dissolved oxygen was excluded from the analysis owing to low confidence in the oxygen trends data. Benthic habitat sensitivity results from B-USEFUL deliverable 4.2 to climate change stressors were integrated with stressor exposure data to estimate the potential risk to habitats (Section 8 in Rozemeijer et al. 2025).

After rasterisation of grid cells, those with non-significant ($p>0.1$) temporal trends in temperature and pH were excluded from the analysis. Risk was calculated by combining normalised exposure layers with rasterised sensitivity layers and subsequently merging the resulting stressor-specific risks to produce a combined climate change risk map.

The climate change risk assessment followed a spatial workflow integrating environmental exposure layers (bottom temperature and pH trends) with habitat sensitivity information obtained from Deliverable 4.2. The methodology consisted of four main steps: the preparation of input datasets, the exposure normalization, sensitivity rasterization and risk calculation and integration (*Figure 7-2*).

To allow comparison and integration of different environmental stressors, exposure layers were normalised to a common scale between 0 and 1. For temperature, the re-scaling of exposure was done using a min–max normalisation:

$$E_n = \frac{E - E_{min}}{E_{max} - E_{min}}$$

where:

E_n= normalised exposure

E= original exposure value

E_{min}, E_{max}= minimum and maximum values of the dataset

This process produced standardised exposure rasters ranging from 0 to 1, representing relative intensity of environmental change.

For pH (where lower values represent higher acidification), data were normalised using the transformation $1 - E_n$, ensuring that higher values correspond to greater exposure to acidification.

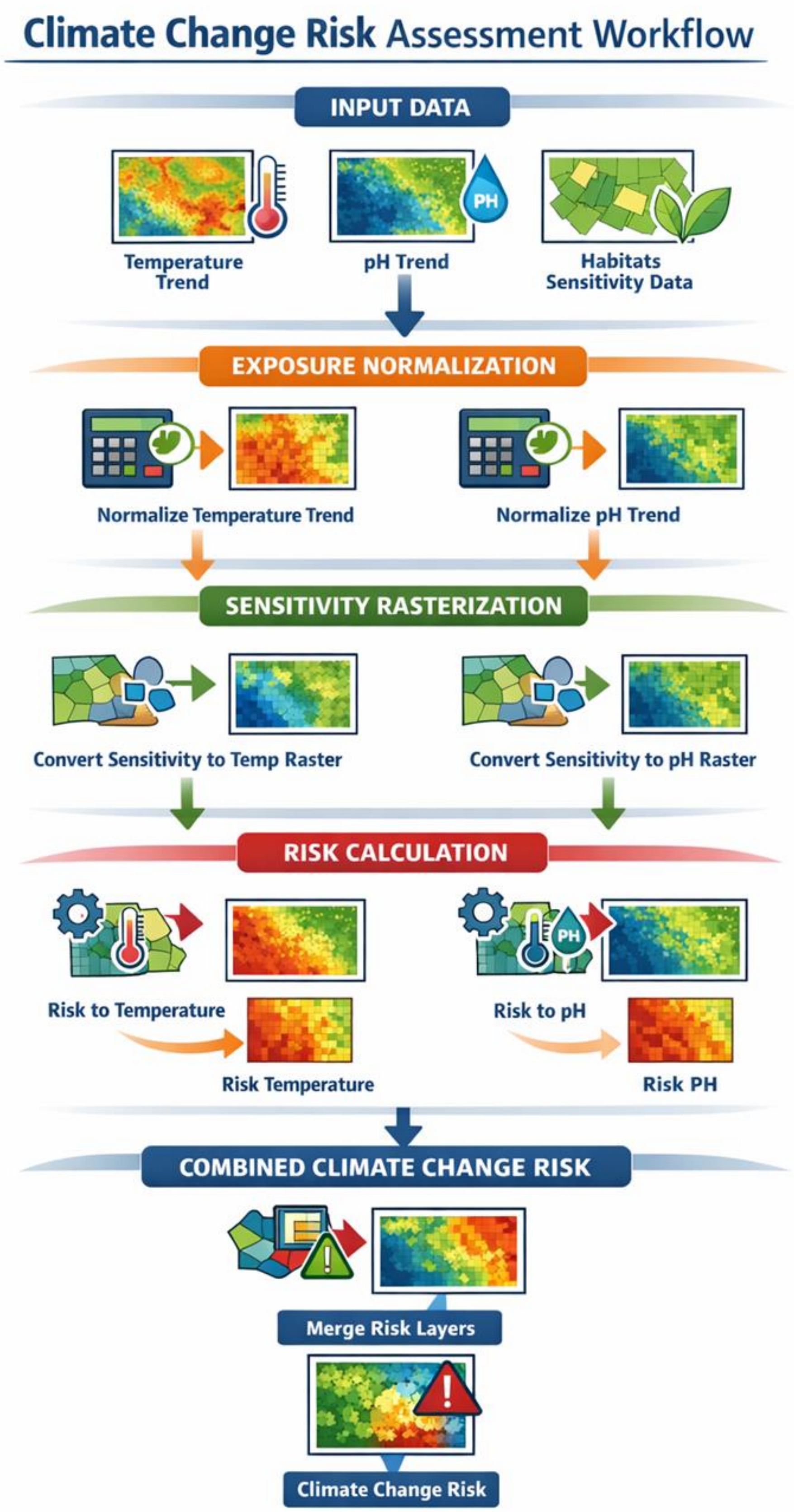

Figure 7-2. Risk-assessment workflow applied to each climate change stressor.

Habitat sensitivity information was converted into raster format to match the spatial resolution of the exposure datasets. Habitat polygons with sensitivity information previously normalised were rasterised at the same resolution of the stressor rasters, ensuring the alignment between exposure and sensitivity layers, enabling cell-by-cell risk calculations.

The risk associated with each stressor was calculated by combining the exposure and sensitivity layers using a multiplicative model, ensuring that risk is more elevated where both exposure and sensitivity are high:

$$Risk = Sensitivity_{norm} \times Exposure_{norm}$$

The final climate change risk was obtained by integrating the two stressor-specific risk layers, in a combined index:

$$Risk_{CC} = \frac{Risk_{Temp} + Risk_{pH}}{2}$$

The layers for temperature and pH were combined through raster overlay to produce a composite climate change risk map representing the cumulative potential impact of warming and acidification across habitats. The resulting map identifies spatial patterns of relative climate risk and highlights areas where sensitive benthic habitats coincide with strong environmental change.

7.2.4 MPA coverage estimation

Building on the framework proposed in the MPA Guide (Grorud-Colvert et al. 2021), which classifies Marine Protected Areas (MPA) according to both their stage of establishment and their level of protection, we selected the protected areas within the Exclusive Economic Zone of continental Portugal (PT), to evaluate the spatial overlap between protection and areas at high risk from pH and temperature stressors. For this purpose, only MPA and Special Areas of Conservation (SAC) at the Implemented stage were considered, as biodiversity and habitat benefits are expected only once management measures are in place (Stage of Establishment; Grorud-Colvert et al. 2021). Special Protection Areas (SPA) designated for marine birds were excluded, as their conservation objectives are limited to habitats in the water column.

MPA protection levels were spatially represented using polygon layers classified in accordance with the MPA Guide framework (Grorud-Colvert et al. 2021, Aminian-Biquet et al. 2024a, Aminian-Biquet et al. 2024b). Spatial overlaps among zones were resolved prior to analysis to ensure that each area was assigned a single protection level, applying a hierarchical rule that gives precedence to the highest level of protection. The methodology is detailed in Chapter 2 of this report. In addition to the sites assessed by Aminian-Biquet et al. (2024a), two areas were incorporated and classified following the MPA Guide framework (Grorud-Colvert et al. 2021). These included (1) Gorringe Bank, as its Stage of Establishment falls between Designated and Implemented, with the management plan in the final stage of approval and the corresponding decree-law forthcoming. As all extractive activities are prohibited except longline (by vessels >12m, previously assessed as having low impact on the seabed: see Section 8 in Rozemeijer et al. 2025), this area was classified as minimally protected. The other area added was (2) Algarve Reef Marine Natural Park – Pedra do Valado. With an Implemented Stage of Establishment, its strict protection zone was classified as fully protected, while the remaining area was considered lightly protected.

Here, individual protected areas were grouped into four categories of protection level: ‘fully protected’, ‘highly protected’, ‘lightly protected’, and ‘minimally protected’. In addition, ‘unclassified’ areas refer to zones for which insufficient information is available to determine their level of protection. Additionally, ‘incompatible’ areas comprise zones where permitted activities result in impacts deemed inconsistent with biodiversity conservation under the MPA Guide framework (Grorud-Colvert et al. 2021). The incompatible category does not apply to the current protected areas within the Portuguese continental EEZ subarea.

The spatial overlap between protection levels and risk areas (for each stressor: temperature and pH) was assessed using ArcGIS Pro (Coordinate system: ETRS89 Portugal TM06). First, MPA polygons were dissolved by protection level to obtain aggregated polygons for each category (fully, highly, lightly, and minimally protected). Risk rasters (excluding non-significant areas) were subsequently reclassified into five classes (very low, low, medium, high, and very high) using the Natural Breaks (Jenks) method. The reclassified rasters were converted to polygon format, and a dissolve operation was performed by risk class. These processed protection and risk layers were then used to quantify the spatial extent of overlap between protection levels and each risk class for pH and temperature stressors.

Prior to the overlay analysis, all polygon layers were validated and geometrically corrected to eliminate topology errors introduced during clipping and dissolve operations, thereby ensuring accurate spatial intersection estimates. For each stressor (temperature and pH), spatial intersections were then performed between the reclassified risk categories and each protection level (including unclassified areas), both for the Portuguese continental EEZ subarea as a whole and at the habitat level. The area of each resulting intersection was calculated in km² and subsequently used to quantify the extent of overlap, expressed in both absolute terms (km²) and as a percentage of the relevant spatial unit.

7.3 Results and Discussion

7.3.1 Trends in climate change stressors

To assess the stress of benthic habitats resulting from long term trends in the environmental conditions, the linear trends of bottom temperature (T_{bot}), dissolved oxygen (O_2) and pH were computed from oceanographic numerical model solutions provided by the Copernicus Marine Environment Monitoring Service (CMEMS) (cf. Section 7.2.1). Maps of the significant trends (90% significance level, i.e. estimates with p-value > 0.1 are masked) were computed over the 32-year period (1993-2024), using the solutions from the higher (IBI) and lower (GLO) spatial resolution models (*Figure 7-3*). Despite some resemblance in the spatial patterns from the two models (upper *vs* lower panels), there are some notable exceptions: (i) stronger trends in the lower resolution model (GLO), (ii) opposite signal trends in dissolved oxygen in the NE region, north of 40.5°N, at depths below 2000m between the coast and the Galician Bank (~42.5°N, 11.75°W), and (iii) opposite trends in temperature in the SE region along the slope (between the 1000m and 2000m isobaths, southward of Nazaré canyon ~39.6°N).

Temperature [°C / 10y]	O2 [mmol/m^3 / 10y]	pH [units / 10y]

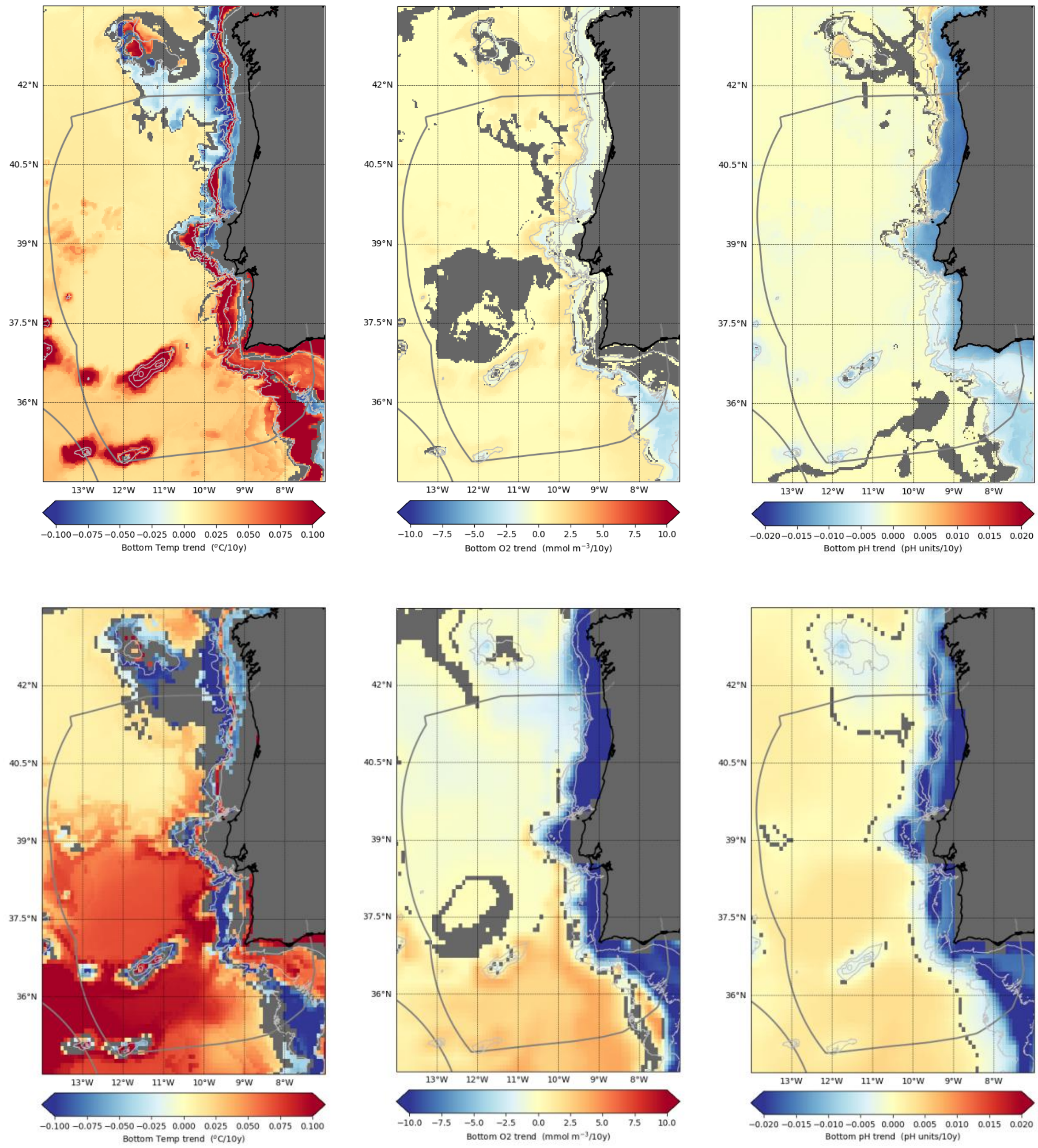

Figure 7-3. Trend maps (in units per decade) for temperature (°C/decade), dissolved oxygen (mmol O_2 m^{-3}/decade) and pH (units/decade), computed from regional (IBI, top row) and global (GLO, bottom row) hydrodynamic and biogeochemical models available from the Copernicus Marine Environment Monitoring Service (CMEMS). Grey areas represent grid cells with non-significant (p>0.1) temporal trends for a given stressor. The 200m, 1000m and 2000m bathymetric contours are represented in light grey. The offshore lines indicate the limits of the Exclusive Economic Zone subareas of continental Portugal and Madeira.

Since the oxygen trends remain uncertain and require further examination, this variable was not included in the analysis. Nevertheless, as presented in Deliverable 4.2 (Section 8 from Rozemeijer et al. 2025), among the three stressors related to climate change, the sensitivity scores assigned by experts to habitats in the face of dissolved oxygen were in general low, had the highest mean uncertainty, and the observed patterns of this stressor were largely consistent with those identified for the temperature stressor.

On the other hand, despite the need for further investigation on the relative disagreement in the temperature and pH trends between the numerical solution of the different resolution models, it was decided to use the outputs of the higher resolution (IBI) model to ensure maximum compatibility between the raster resolutions of the habitats and the stressor layers.

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7.3.2 Benthic habitats risk

The spatially detailed mapping of benthic habitat types and their sensitivity scores, and of changes in temperature and pH over the past 32 years, allowed us to spatially explicitly determine risk in each combination of habitat type and stressor. We highlight that risk could not be estimated for 7.86% of the EEZ area in the case of temperature, and for 4.90% in the case of pH, where the trend slope over the 32-year period was not significant (generally owing to high interannual variability). In the case of temperature, non-significant values of trend occurred along specific depth intervals of the EEZ, especially along a wide area of the inner continental shelf of the western coast of Portugal and along some narrow areas of deeper depths (*Figure 6-3*). In the case of pH, non-significant values of trend similarly occurred along some narrow areas of deeper depth, and in contrast also in a southern area of the abyssal plain (*Figure 7-4*). In areas with non-significant value of a stressor trend the risk from the stressor to benthic habitats could not be computed (*Figure 7-4, Figure 7-5*). This limitation is especially important in certain habitat types and should be taken into consideration when interpreting the results, especially in habitats of the continental shelf.

Temperature risk and pH risk differed between benthic habitat types and also present contrasting patterns of variation between the two stressors (*Table 7-2*).

Table 7-2.Temperature and acidification (pH) risk per benthic habitat type in the Exclusive Economic Zone subarea of continental Portugal: mean, standard deviation (SD), minimum, maximum, and rank of risk. Habitat types included in Deliverable 3.2 (Section 8 from Rozemeijer et al. 2025) are also included here for the purpose of alignment of the two deliverables, but available data did not allow mapping of those habitats and therefore risk was not estimated; hydrothermal vents are not present.

Benthic Habitat	Temperature risk					pH risk				
	Mean	SD	Min	Max	Rank	Mean	SD	Min	Max	Rank
Intertidal rocky reefs	-	-	-	-	-	-	-	-	-	-
Intertidal sof sediment (including gravel and cobbles)	-	-	-	-	-	-	-	-	-	-
Inner shelf rocky reefs (<50 m)	0,311	0,029	0,247	0,327	6	0,482	0,097	0,274	0,595	3
Inner shelf soft sediment (<50 m)	0,015	0,002	0,011	0,017	13	0,296	0,049	0,149	0,346	5
Macroalgae forests	0,713	0,000	0,713	0,713	1	0,088	0,009	0,077	0,098	11
Maërl	0,323	0,003	0,320	0,327	4	0,710	0,000	0,710	0,710	1
Seagrasses	-	-	-	-	-	-	-	-	-	-
Rocky reefs (50–200 m)	0,113	0,052	0,041	0,237	10	0,525	0,060	0,137	0,602	2
Soft sediment (50–200 m)	0,000	0,000	0,000	0,000	15	0,063	0,008	0,019	0,074	13
Aggregations that change physiography in soft sediment	-	-	-	-	-	-	-	-	-	-
Biogenic reefs (<200m)	-	-	-	-	-	-	-	-	-	-
Biogenic reefs (>200m)	-	-	-	-	-	-	-	-	-	-
Seamounts (summit <200 m)	0,327	0,042	0,260	0,388	3	0,365	0,054	0,233	0,407	4
Seamounts (summit >1000 m)	0,316	0,078	0,064	0,484	5	0,216	0,045	0,110	0,318	7
Seamounts (summit 200–1000 m)	0,211	0,022	0,146	0,268	8	0,198	0,067	0,101	0,336	8
Slope and ramp rocky reefs	0,060	0,024	0,029	0,098	11	0,272	0,060	0,085	0,340	6
Slope and ramp soft sediment	0,009	0,004	0,002	0,018	14	0,077	0,018	0,013	0,118	12
*Slope and ramp seabed**	*0,050*	*0,017*	*0,000*	*0,085*	12	*0,124*	*0,036*	*0,007*	*0,231*	10
Mud volcanoes and cold seeps	0,428	0,120	0,342	0,632	2	0,008	0,000	0,008	0,008	14
Hydrothermal vents	-	-	-	-	-	- -	-	-	-	-
Canyons	0,120	0,039	0,024	0,252	9	0,137	0,046	0,047	0,330	9
Abyssal plains	0,250	0,029	0,020	0,558	7	0,000	0,000	0,000	0,000	15

The highest risk from temperature increase was in macroalgae, mud volcanoes and cold sweeps, seamounts (summit <200m), maërl and seamounts (summit >1000m). The lowest temperature risk was in slope and ramp rocky reefs, slope and ramp seabed, inner shelf soft sediment (<50 m), slope and ramp soft sediment, and finally soft sediment (50–200 m). Intermediate values were found in inner shelf rocky reefs (<50 m), abyssal plains, seamounts (summit 200–1000 m), canyons and rocky reefs (50–200 m) (*Table 7-2*). At the large spatial extent of the whole study area, the spatial distribution of temperature risk (*Figure 6-4*) closely matched the spatial distribution of benthic habitats (*Figure 7-1*), with the notable exception of the northeast part of the study area. In this area, the abyssal plain exhibited substantially lower temperature risk than in the rest of this habitat. This pattern reflects the lower temperature trend slope observed in this area, which is likely associated with deeper depths found in this part of the abyssal plain. In what concerns habitat types with larger spatial extent, there was a contrast between higher risk in two of the habitat types of seamounts (summit <200 m and summit >1000 m) versus moderate risk in abyssal plains and canyons versus lower risk in soft sediments (inner shelf <50m, and 50–200 m) and slope and ramp (rocky reefs, soft sediment, seabed) (*Figure 6-4*). The other habitat types with higher risk (i.e., macroalgae forests, maërl, mud volcanoes and cold seeps) covered very small areas. There were however extensive areas, especially within specific depth ranges (especially shallower parts of the continental shelf) where temporal changes in temperature were not significant (*Figure 7-1*), and in which temperature risk was therefore not assessed (*Figure 7-4*).

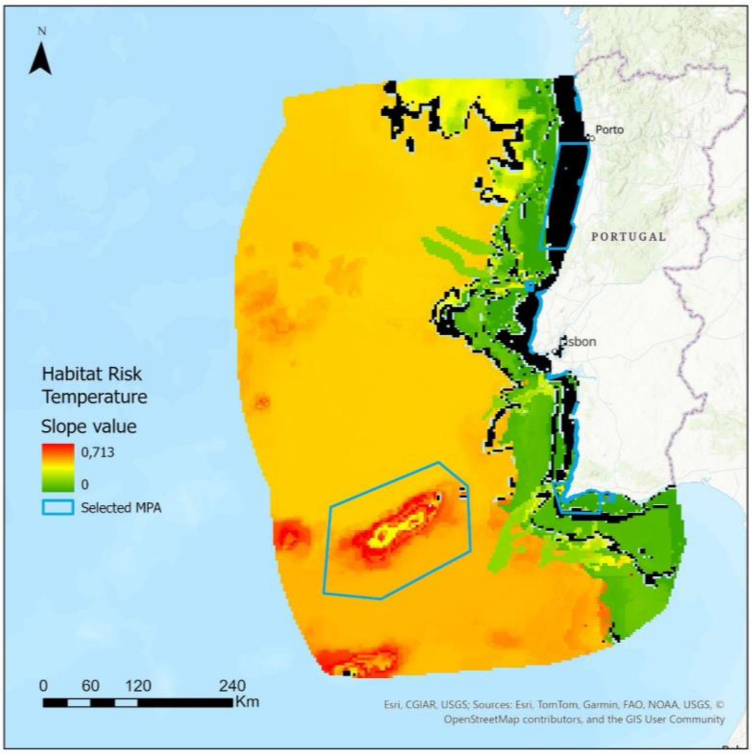

Figure 7-4. Temperature risk for benthic habitats in the Exclusive Economic Zone subarea of continental Portugal. Areas in black are the grid cells with non-significant (p>0.1) slopes in the trends of temperature (and represent in total 7.86% of the EEZ).

Comparing the patterns observed here for the habitats at higher risk from stressors analysed with previous results on habitat sensitivity alone (Section 8 from Rozemeijer et al. 2025), it is evident that macroalgae forests, mud volcanoes and cold seeps, and seamounts are of highest concern. These habitats combine high temperature sensitivity with exposure to warming making them particularly vulnerable, especially in the case of macroalgae forests and known mud volcanoes, since both also have small extents. Therefore, conservation strategies should prioritise actions that mitigate ongoing pressures and prevent imminent degradation of these two habitat types (Brooks et al., 2006). Among the remaining habitats at moderate risk from temperature stressor, the abyssal plains stand out. Although subject to lower exposure, these were classified as having moderate to high sensitivity. Therefore, a precautionary approach should also be adopted for certain areas of this habitat, which currently face relatively low levels of human impact, aiming to protect ecosystems before significant degradation occurs, and to maintain large-scale ecological processes (Brooks et al. 2006).

The highest risk from acidification (pH) was in maërl, rocky reefs (50–200 m), inner shelf rocky reefs (<50 m), seamounts (summit <200m) and inner shelf soft sediment (<50 m), whilst the lowest risk was in macroalgae forests, slope and ramp soft sediment, soft sediment (50–200 m), mud volcanoes and cold seeps, and lastly abyssal plains (*Table 7-2*). Intermediate acidification risk was the case for slope and ramp rocky reefs, seamounts (summit >1000 m), seamounts (summit 200–1000 m), canyons, slope and ramp seabed. The spatial distribution of the acidification (pH) risk across the whole study area (*Figure 7-5*) also mostly followed the spatial patterns of the habitats, but in a different manner than in temperature risk *Figure 6-4*).

There was higher risk in several habitat types of shallower depths (along the continental coast and seamounts), namely habitats with large extents such as inner shelf rocky reefs (<50 m), inner shelf soft sediment (<50m), rocky reefs (50–200 m), and others with smaller extents such as maërl and seamounts (summit <200 m). This contrasted with moderate and lower risk in other habitats. There were however large specific areas, especially in abyssal plains in the south, where temporal changes in pH were not significant (*Figure 7-1*) and in which acidification pH risk was therefore not assessed (*Figure 7-5*).

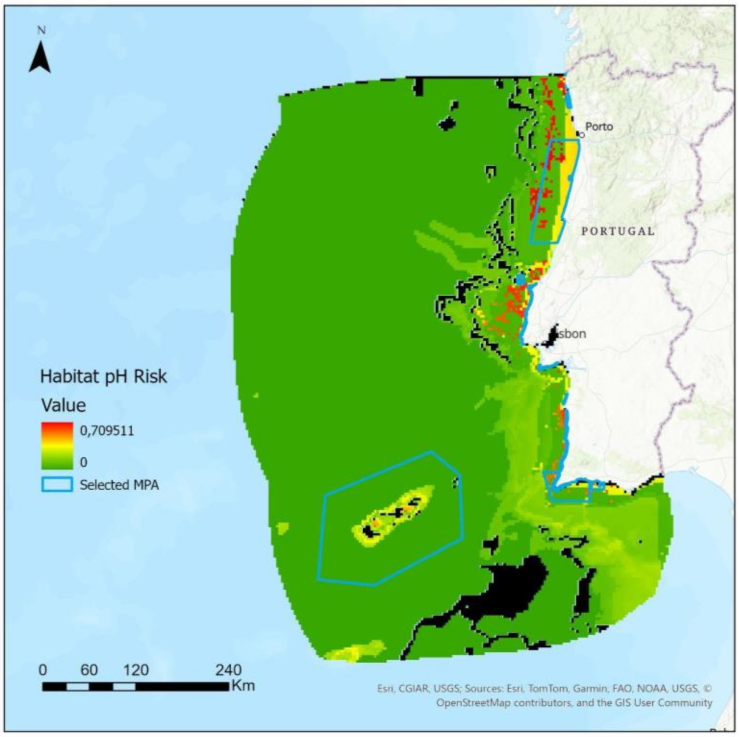

Figure 7-5. Acidification (pH) risk for benthic habitats in the Exclusive Economic Zone subarea of continental Portugal. Areas in black are the grid cells with non-significant (p>0.1) slopes in the trends of pH (and represent in total 4.90% of the EEZ).

Regarding acidification (pH) risk, observed patterns for habitats with highest sensitivity (as previously defined in Section 8 of Rozemeijer et al. 2025) show that the main concern relates to the maërl habitat. This combines rarity with high exposure and sensitivity, in addition to its high ecological value (Stratoudakis et al. 2019). Therefore, it is important to adopt an approach that prioritises its conservation across the largest possible extent of this habitat, particularly at sites where exposure to pH changes is lower. While less pH-sensitive than maërl, rocky reef habitats also show notable pH risk. However, given their wide distribution along the Portuguese continental shelf, conservation efforts should preferentially prioritise areas that may act as refugia from pH changes, in order to prevent imminent degradation and follow a precautionary approach (Brooks et al. 2006).

Combined climate change risk was only spatially explicitly determined as the mean of temperature and pH risk where exposure was significant in both stressors (temperature and pH) (*Figure 7-6*). The spatial patterns obtained therefore reflect the spatial patterns of both risks. In general, highest combined climate change risk is observed in seamounts and rocky reefs. Risk was intermediate for abyssal plains and canyons; and lower in various habitats of the continental slope, ramp and shelf. There were however fairly large areas of specific habitats and depth ranges where temporal changes in temperature or pH were not significant (*Figure 7-1*) and in which combined climate change risk was therefore not assessed (*Figure 7-6*).

Importantly, risks to some habitats previously identified as highly sensitive to temperature (aggregations in soft sediments) and/or to pH (biogenic reefs) could not be assessed here, due to a lack of spatial information on their distribution. It is urgent to map these habitats so that they can be included in any risk assessments, allowing the most appropriate strategy to be defined for prioritising their conservation in the context of global climate change.

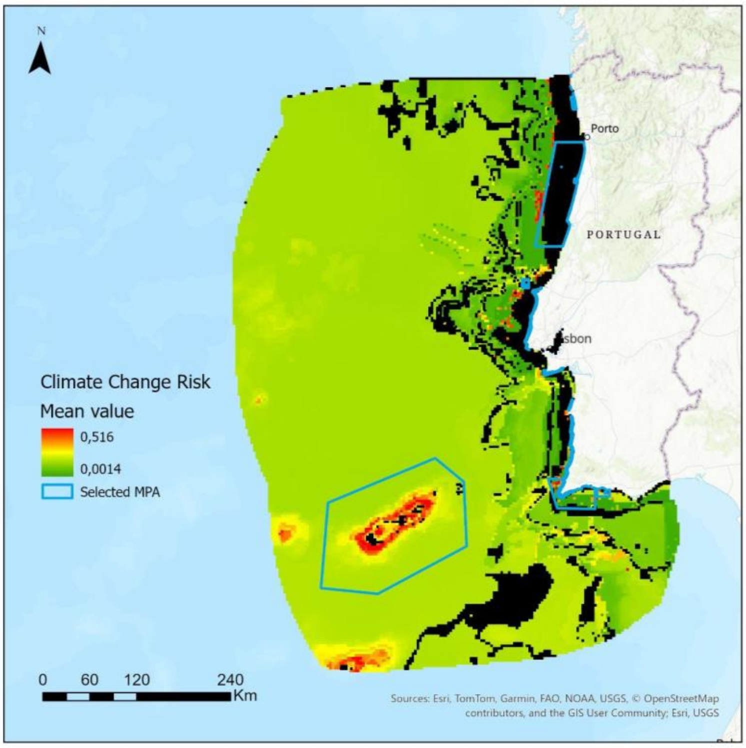

Figure 7-6. Climate change risk (mean of temperature and pH risks) for benthic habitats in the Exclusive Economic Zone subarea of continental Portugal. Areas in black are the grid cells with no-significant (p>0.1) slopes in the trends of pH and temperature.

7.3.3 Benthic habitat MPA coverage and conservation gaps

Marine Protected Areas (MPAs) are widely recognised as effective instruments for marine conservation, particularly in safeguarding habitats and the biodiversity they support (Lubchenco & Grorud-Colvert 2015; Edgar et al. 2014). Even under the pressures of climate change, a major ongoing management challenge, well-managed MPAs and MPA networks are expected to demonstrate greater ecological resilience, maintaining ecosystem functions and services over time (McLeod et al. 2009). This resilience arises from the restriction of direct human pressures such as overfishing, habitat destruction and pollution, allowing key habitats to recover and retain structural complexity, especially those composed by habitat-forming species like corals, sponges, kelp forests, seagrasses, among others (e.g., Mellin et al. 2016, Strain et al. 2019, Peleg et al. 2023). Biodiversity conservation through MPAs may also provide functional redundancy (i.e. several species playing a similar role in the ecosystem), ensuring that critical ecological roles such as herbivory, predation and nutrient cycling are maintained, even if some species are impacted by climate-induced disturbances (e.g., functionally similar species may have different thermal sensitivity) (e.g., Benedetti-Cecchi et al. 2024).

In the continental Portugal (PT) Exclusive Economic Zone (EEZ) subarea, 90.97% of the total seafloor area is effectively unprotected (i.e. unclassified but designated MPAs, and unprotected areas), while 9.03% is protected (*Table 7-3*). This is far from the global targets agreed under the Convention on Biological Diversity (CBD), i.e. to protect 30% of biodiversity and habitats by 2030. The target of 30% for the Portuguese MPA Network does include the EEZ subareas of the Azores and Madeira, alongside continental Portugal's EEZ subarea; still, this proportion of 9.03% of protection of benthic seafloor for the latter subarea might not be enough to ensure the essential representativity and replicability of an ecologically coherent MPA Network for Portugal (Stratoudakis et al. 2019). Given the distinct biogeographic characteristics of the EEZ subareas of continental Portugal, the Azores and Madeira, and the extended continental shelf, spatial planning should aim to protect approximately 30% within each of these regions individually to reach the overall well-balanced target of 30%. Safeguarding environmental heterogeneity across regions is particularly critical in the context of climate change, as discussed below.

Table 7-3.Total extent (km^2) of the continental Portugal (PT) EEZ subarea, and of MPAs within it, of different protection levels (fully, highly, lightly, minimally, unclassified but designated, and unprotected). Also presented are the area percentages of MPAs of all protection levels (fully, highly, lightly, minimally) relative to the total EEZ, and the area percentages of restricted MPAs (protection levels fully, and highly) relative to the total EEZ. The table also presents the percentage of the total area of the EEZ that has very high or high (according to natural breaks) risk from temperature and pH stressors (separately), as well as how much of this is covered by MPAs of each protection level - presented as percentage relative to the total area of the EEZ.

Benthic Habitats	ZEE continental PT (Km2)	% higher temperature risk	% higher pH risk
Total area	316854,05	30,48	1,15
fully	11,1	0,0002	0
highly	45,6	0	0
lightly	5299,5	0,003	0,002
minimally	23261,8	1,8	0,001
unclassified but designated	1519,1	0	0,001
Unprotected area	286716,9	28,7	1,141
Total unprotected (%)	91,4	-	-
Total protected (%)	9,03	-	-
Total restricted area (%)	0,02	-	-
Important: These estimates do not take Special Protection Areas (SPAs) into account			

To produce conservation benefits to habitats and the biodiversity these support, protected areas management plans must be implemented and have strict protection measures in place (Horta e Costa et al. 2016, Grorud-Colvert et al. 2021, Roessger et al. 2022), otherwise MPAs will provide little or no additional protection compared to outside areas (Horta e Costa et al. 2019, Grorud-Colvert et al. 2021), as is the case for most of the protected areas in the Portuguese continental EEZ subarea. Among all MPA protection levels, fully protected MPAs currently comprise the lowest percentage of the total EEZ (0.004%), followed by highly protected MPAs (0.014%); these two protection levels combined, referred to as restricted areas, only represent 0.02% of the total EEZ. Higher percentages of the total EEZ are classified as lightly protected (1.673%) and minimally protected MPAs (7.341%). This indicates that across most areas currently designated as MPA, various low- to moderate-impact activities remain permitted. Lightly protected MPAs generally allow activities such as small-scale fisheries and small-scale or low-density aquaculture (preferably unfed) (Grorud-Colvert et al. 2021, Aminian-Biquet et al. 2024a, Aminian-Biquet et al. 2024b). In minimally protected MPAs a broader range of uses is typically allowed, including most recreational and commercial fisheries (some bottom-contact gears included) as well as seafloor infrastructure, anchoring, tourism activities, and aquaculture (including fed or medium-density operations). Highly destructive activities are prohibited in minimally protected MPAs, such as industrial bottom trawling, dredging, seabed mining, oil and gas extraction (Grorud-Colvert et al. 2021, Aminian-Biquet et al. 2024a, Aminian-Biquet et al. 2024b). This study highlights the urgency of identifying areas of high conservation value where stricter protection should be implemented within current and new MPAs, in order to achieve the CBD target of 10% strict protection by 2030. It is also essential to implement monitoring plans to evaluate the effectiveness of existing management measures; to support the adoption of additional measures where needed; and to inform the planning and designation of additional MPAs. Such efforts are particularly important for areas that support sensitive species and habitats or those at high risk of impacts, and therefore of special concern.

The spatial distribution of temperature and pH risks to benthic habitats (classified as very high or high risks according to natural breaks) and their overlap with MPAs revealed spatial patterns of protection to these habitats (*Table 7-3*). Areas of very high or high temperature risk to benthic habitats represented 30.48% of the total EEZ, most this unprotected (28.67%). Small areas are minimally (1.81%) or lightly protected (0.0032%), while even smaller areas have restricted protection (0.0002% are fully protected and none highly protected). In contrast, very high or high pH risk to benthic habitats represented only 1.1450% of the total EEZ. Most of this area is unprotected (1.1415%) or are unclassified but designated MPAs (0.0005%), while only very small areas are minimally (0.0006%) or lightly protected MPAs (0.0024%), and none are restricted areas (i.e. fully or highly protected MPAs). These results underline the importance of integrating risk assessments into conservation planning of habitats to improve protection of high-risk areas, both in terms of MPA coverage and level of protection; this is in line with previous findings for global at-risk biodiversity (O'Hara et al. 2019). To this aim, spatial assessments of habitat risk inside and outside MPAs can improve understanding of the true potential of MPAs and MPA networks for effectively protecting marine biodiversity, and could support the selection of potential new areas for conservation.

Table 7-4.Overlap between MPAs in the continental Portugal EEZ subarea and benthic habitat types: total area per habitat type (in km^2); percentage area per habitat type relative to the total EEZ subarea; percentage area of any MPA protection level (fully, highly, lightly, minimally) relative to the total EEZ subarea; percentage of the restricted protected levels of MPAs (i.e. fully and highly) relative to the total EEZ subarea.

Benthic Habitat	Total Area (Km2)	% of habitat in ZEE	% of MPA coverage	% of restrict protection coverage	% of total habitat area with significant slope of high temperature risk with MPA coverage	% of total habitat area with significant slope of high pH risk with MPA coverage
Intertidal rocky reefs	-	-	-	-	-	-
Intertidal sof sediment (including gravel and cobbles)	-	-	-	-	-	-
Inner shelf rocky reefs (<50 m)	971,86	0,307	26,721	1,572	2,881	12,429
Inner shelf soft sediment (<50 m)	4803,09	1,518	44,844	0,818	0,103	0,619
Macroalgae forests*	63,60	0,020	30,881	0	25,750	17,792
Maërl	5,13	0,002	88,805	20	53,986	36,215
Seagrasses	-	-	-	-	-	-
Rocky reefs (50–200 m)	2827,16	0,893	19,683	0	0	15,370
Soft sediment (50–200 m)	15214,64	4,808	17,671	0,014	0	1,599
Aggregations that change physiography in soft sediment	-	-	-	-	-	-
Biogenic reefs (<200m)	-	-	-	-	-	-
Biogenic reefs (>200m)	-	-	-	-	-	-
Seamounts (summit <200 m)	153,40	0,048	98,158	0	82,601	66,569
Seamounts (summit >1000 m)	4026,99	1,273	58,851	0	54,356	0
Seamounts (summit 200–1000 m)	961,87	0,304	94,381	0	14,647	1,607
Slope and ramp rocky reefs	214,66	0,068	0	0	0	0
Slope and ramp soft sediment	6050,05	1,912	0	0	0	0
Slope and ramp seabed	28936,82	9,145	0	0	0	0
Mud volcanoes and cold seeps	28,27	0,009	0	0	0	0
Hydrothermal vents	-	-	-	-	-	-
Canyons	12571,44	3,973	0,139	0	0	0,043
Abyssal plains	239592,94	75,719	8,130	0	6,208	0

The total extent of different benthic habitat types within the continental Portugal EEZ subarea differs markedly (*Table 7-4*): very extensive habitats contrast with those of limited extent. Abyssal plains cover most of the EEZ (76% of the total subarea), followed by slope and ramp seabed (9%), soft sediment (50–200 m) (5%), and canyons (4%). Habitats that occupy very small extents, are especially the macroalgae forests (0.02%), mud volcanoes and cold seeps (0.009%), and maërl (0.002%). Other habitats have intermediate extents. Hydrothermal vents are not present, while some habitats could not be mapped due to unavailable data.

The percentage of each benthic habitat covered by MPAs of any protection level varies, as does the percentage of each habitat that simultaneously has MPA coverage and is at very high or high risk from a climate change stressor. The two stressors, temperature and pH, are considered separately (*Table 7-4*), as follows.

Some benthic habitats have high MPA coverage. Maërl, occupying a very small total area on the inner continental shelf, are 88% covered by MPAs, with 20.06% of the area at a restricted protection level. Maërl habitat is among those at higher risk from temperature and acidification (pH), with 54% of the area having both MPA coverage and being at very high or high temperature risk, whereas this percentage is 36% in the case of acidification risk. Given its rarity of habitat and high sensitivity to pH changes due to its calcifying-based structure (Section 8 in Rozemeijer et al., 2025), the entire extent of maërl habitat should be protected under highly restrictive management measures. Therefore, further efforts are needed to increase the level of protection of the 68.1% of maërl habitat currently under light protection and to protect the remaining known areas of this habitat, while also strengthening efforts to map and identify additional occurrences.

Seamounts (summit <200m, and 200-1000m) occupy a small total area and have high MPA coverage, respectively 98% and 95%. Seamounts (summit <200m) have high risk from temperature and pH, and 83% and 67% of its area have both MPA coverage and very high or high risk from temperature and pH, respectively. Seamounts (summit 200-1000m) are at intermediate risks from temperature and pH, and 15% and 2% of its area has both MPA coverage and very high or high risk from temperature and pH. In contrast, deeper seamounts (with summit >1000m) occupy a wider area with less high coverage by MPAs (58%). It is among habitats with higher risk from temperature and intermediate risk from pH, and 54% of its area has both MPA coverage and very high or high risk from temperature, but no overlap occurs in the case of (very) high pH risk. Regardless of depth, the high ecological value of seamounts (Stratoudakis et al. 2019), combined with their demonstrated vulnerability (current assessment), highlights the urgent need to implement more restrictive protection measures, which are currently absent.

On the inner continental shelf (<50 m), rocky reef and soft sediments are 26% and 44% covered by MPAs, respectively. MPAs with restricted protection levels occur almost exclusively in these two habitats, representing 1.57% and 0.82% of their total areas, respectively. These habitats are at intermediate and low risk from temperature, respectively, and only 3% and 0.10% of their area have both MPA coverage and very high or high risk from temperature. In contrast, they are at high risk from pH, and 12% and 0.619% of rocky reef and soft sediment habitats, respectively, have simultaneously MPA coverage and very high or high risk from pH.

At deeper shelf depths (50–200 m), MPA coverage decreases to 20% in rocky reefs and to 18% in soft sediment. These habitats are among those with high risk from pH, with 15% and 1.599% of their areas with both MPA coverage and at very high or high risk from pH, respectively. These habitats are among those with intermediate and low mean risk from temperature, but no overlap occurs. While soft-sediment habitats across all depths are generally considered lower priorities in terms of ecological value and climate sensitivity, rocky reefs rank among the habitats with highest ecological value and climate sensitivity, particularly to pH changes (Stratoudakis et al., 2019; Section 8 in Rozemeijer et al., 2025). The currently low proportion of this habitat under strict protection therefore represents a clear conservation gap that should be urgently addressed.

Macroalgae forests, which occur mostly in the inner continental shelf but also on seamounts (summit <200m), have 30% MPA coverage. This habitat is among those with higher temperature and lower pH risk, and 26% of its area is both covered by MPAs and at very high or high temperature risk, compared with 18% for pH risk. The macroalgae forests of the continental Portugal EEZ subarea are predominantly composed of kelp species (*Phyllariopsis* spp., *Saccorhiza polyschides, Laminaria ochroleuca,* and *Laminaria hyperborea*). These species maintain well-established populations along the Portuguese coastline, particularly up to 20 m depth, as well as in deeper offshore environments, including the Camões Seamount (~50 m) and the Gorringe Bank (~85 m). However, ocean warming is projected to drive poleward range contractions, placing Portugal’s southernmost populations at high extinction risk under midto-high emission scenarios (Assis et al. 2017). The convergence of climate vulnerability and critical ecosystem function (habitat-forming species), with expected cascading effects on trophic networks and changes in carbon cycling, justifies urgent conservation measures for macroalgae forests (Assis et al. 2017). These measures must include not only increased MPA coverage with a high proportion of restricted protection, but also the placement of these MPAs along a gradient of different temperatures (Assis et al. 2017, McLeod et al. 2009) and if possible, within climate refuges.

Mud volcanoes and cold seeps, in spite of their rarity and ecological uniqueness, are not covered by MPAs; they are at high risk from temperature and low risk from pH. The greatest threats to this habitat are oil and gas extraction, seabed mining, and bottom trawling (Levin et al. 2016); their protection priority is therefore inclusion within MPAs. Based on the known distribution of mud volcanoes, this could be achieved through implementation of a single MPA, covering the southeast corner of the continental Portugal EEZ subarea (Gulf of Cadiz).

Some deeper habitats are entirely lacking any MPA coverage, namely slope and ramp rocky reefs, slope and ramp soft sediment, and slope and ramp and seabed. These habitats are among those with low temperature risk and intermediate pH risk. Canyons, one of the largest habitats in Portuguese waters, have 0.139% MPA coverage, while this is 8& for the largest habitat – abyssal plains. Canyons are among the habitats with intermediate risks from temperature and pH, and 0.043% of its area has both MPA coverage and is at very high or high risk from pH, but no overlap occurs for (very) high temperature risk. Representativity and replicability are among the main principles of an ecologically coherent MPA network. In this sense, and despite their low-to-medium vulnerability, these habitats should be included in the national MPA network, as they encompass a set of species typically associated with, and characteristic of these habitats (Martins et al. 2013, Gomes et al. 2018, Moura et al. 2020).

Finally, abyssal plains habitats are at intermediate temperature risk and low pH risk, and 6% of its area has both MPA coverage and very high or high risk from temperature, but no overlap for (very) high pH risk. Abyssal plains were previously assessed as having low ecological value (Stratoudakis et al. 2019) but high temperature sensitivity (Section 8 in Rozemeijer et al. 2025). Following the same principles of representativity and replicability, conservation of abyssal plains should therefore be expanded. A new large MPA (Rei Dom Carlos MPA) is currently being Proposed/Committed, which would considerably increase the protection of this habitat (adding 48,770 km^2), beyond additional seamount habitats (adding 1424 km^2). Based on the proposed decree-law currently under public consultation, its level of protection would likely be classified as incompatible due to allowance of industrial fishing (vessels >12m). However, this classification is in need of revision to better reflect the characteristics of oceanic MPAs and the activities occurring within them, as these areas are only accessible to large fishing vessels (considered industrial in the MPA guide framework). This may require defining new criteria based on known impacts of such activities in deep-water environments. Since oil and gas extraction, seabed mining, and bottom trawling will be prohibited under the proposed decree-law, the MPA will still provide ecological benefits. Consequently, the level of protection will largely depend on the type and intensity of fishing activities permitted under management plans.

7.4 Discussion and conclusions

With current and future climate change, networks of MPA can promote climate resilience within biogeographic regions. In addition to representativity and adequate protection levels, the effectiveness of MPA networks strongly depends on ecological connectivity between protected areas. Connectivity processes, particularly those mediated by larval dispersal and the movement of juvenile and adult individuals, play a central role in maintaining population persistence and supporting recovery after disturbance. Designing MPA networks that facilitate dispersal among sites can enhance demographic connectivity and promote recolonisation of impacted habitats. In a context of rapid climate change, connectivity is also essential to enable species range shifts and maintain functional ecosystem processes across environmental gradients. Networks that include areas likely to function as climate refugia, where local environmental conditions buffer species from the most severe impacts of warming, may provide critical sources for recolonisation and long-term persistence of vulnerable populations. Furthermore, spatial configurations that incorporate “stepping-stone” MPAs can facilitate progressive dispersal and range expansion across fragmented habitats, thereby increasing the adaptive capacity of marine communities to ongoing environmental change. Therefore it is needed that the planning of MPA locations considers their connectivity and environmental gradients by enabling movements of individuals, or shifts in species distributions, between protected areas under exposure to different climatic stressors. This would buffer ecosystems against local extinctions and support faster recovery (McLeod et al. 2009, Rilov et al. 2019).

Two complementary strategies are commonly used to prioritise areas for conservation. **Reactive** approaches focus on regions where habitats and biodiversity are both highly irreplaceable and under immediate threat, aiming to mitigate ongoing pressures and prevent imminent degradation (Brooks et al. 2006, our risk approach). **Proactive** approaches prioritise areas of high irreplaceability that currently face relatively low levels of human impact, seeking to protect ecosystems before significant degradation may occur, and aiming to maintain large-scale ecological processes (Brooks et al. 2006, our sensitivity approach). Within the framework of systematic conservation planning, these two strategies reflect different ways of incorporating risk into prioritisation. Both are necessary for effective habitat and associated biodiversity conservation: reactive strategies address urgent conservation needs, while proactive strategies safeguard ecosystems in good condition and create opportunities for long-term protection (Brooks et al. 2006).

In this context, several criteria should be adopted when planning and designing the different MPAs within a network (McLeod et al. 2009, Rilov et al. 2019):

- ***Ensure habitat representativity*** across MPAs located along different climatic gradients, taking into account both current and projected species distributions, while promoting connectivity between habitats wherever possible.
- ***Include all distinct habitat types*** within the MPA network, while considering the natural size and distribution of each habitat as a basis for planning appropriate levels of representativity and replicability.

- ***Promote functional redundancy*** by prioritising the selection of habitat areas with the highest biodiversity value, supported by adequate protection measures and a well-balanced combination of strict and managed protection zones.
- ***Integrate information from risk maps and climate projections*** to identify and anticipate which natural areas are least exposed to climate change impacts and may function as climatic refugia, while ensuring their habitats are shielded from additional anthropogenic pressures through strict protection measures.

The first three are more or less aligned with our sensitivity approach. The fourth is in alignment with the risk approach. The present study applied a methodology of sensitivity, risk and the match between MPAs and hotspots for risk. It should be regarded primarily as a proof-of-concept exercise, and its findings must be interpreted with caution. With the presentation of the match and mismatches between MPAs and hotspots of habitats a discussion can be started on the need for adapting and assigning MPAs both in spatial measures or in (temporal) management like e.g. seasonal or permanent closure for bottom contact fisheries. Our methodology can be a powerful management tool.

Several limitations should be considered when interpreting the results of this analysis. First, the proxies used to represent anthropogenic and environmental pressures may not fully capture the local intensity or ecological impact of these stressors. For example, spatial datasets of fishing effort do not necessarily translate directly into seabed disturbance intensity, as the ecological impact depends on gear type, towing speed, and seabed characteristics. Similarly, regional warming trends may not accurately reflect the local exposure experienced by benthic communities, which can be influenced by fine-scale oceanographic processes. Finally, the assessment of overlap between vulnerability hotspots and MPAs does not explicitly account for differences in management effectiveness among protected areas. Many MPAs allow certain extractive activities or have limited enforcement, meaning that the nominal spatial overlap between sensitive habitats and MPAs may overestimate the level of effective protection provided.

Improvements can be defined on current status. Methodological scope and data availability impose limitations, including uncertainty in sensitivity scoring and assessing MPA protection level, gaps in habitat delimitation, and the use of linear trend slopes as a measure of stressor exposure. These various limitations may constrain the specificity and robustness of results. To improve upon the current outcomes, it is relevant to:

(1) Improve the spatial resolution of habitat mapping (e.g., maërl beds and macroalgal forests) and address current habitat data gaps, particularly in intertidal areas, biogenic reefs, and organism aggregations that modify soft-sediment physiography.
(2) Improve the current protection level classification (based on the MPA Guide) to enable the assessment of currently unclassified areas and adjust the scale to better reflect the characteristics of offshore areas, where fishing by large vessels (>12m) is automatically considered incompatible under the current framework.
(3) Refine the definition and accuracy of stressor exposure layers, particularly for bottom temperature. For example, relatively large areas of coastal habitats in northern Portugal had non-significant values of linear trend slope and in those areas a potential risk of temperature changes to benthic habitats was not assessed. In addition, the

accuracy of other stressor models for the study area should be explored and integrated to improve the characterization of exposure layers (e.g., other Copernicus products). In addition, the inclusion of Vessel Monitoring System (VMS) or Automatic Identification System (AIS) fishing effort data could provide a more accurate representation of the spatial distribution and intensity of bottom-contact fishing activities. Such datasets would allow a more direct estimation of seabed disturbance by accounting for the location, frequency and duration of fishing operations.

(4) Finally, complementary thermal exposure metrics should be explored, including non-linear patterns and shifts in seasonal temperature patterns, especially critical in upwelling-dominated systems such as the Portuguese continental shelf, as well as marine heatwave frequency and intensity, which represent acute thermal stressors increasingly relevant under ongoing climate change scenarios. Future analyses could integrate climate velocity metrics to better capture the spatial dynamics of climate-driven environmental change. Climate velocity estimates quantify the rate and direction at which temperature conditions shift across seascapes and therefore provide a useful indicator for identifying areas that may function as climate refugia or corridors facilitating species redistribution under climate change.

Risk hotspots and climatic refugia are distributed in complex mosaic patterns across species ranges and habitat patches, often remaining undetectable without a detailed understanding of biological vulnerabilities at both regional and local scales. This highlights the critical need for fine-resolution ecological assessments when designing spatially representative and climate-resilient MPA networks. Future studies should apply the same methodology to derive risk maps for fishing and other stressors and be integrated in the planning of the Portuguese MPA Network.

8 Risks from invasive species for Mediterranean MPAs

8.1 Introduction

The Mediterranean Sea is a global hotspot of marine non-indigenous species (NIS) (Edelist et al. 2013, Azzurro et al. 2022, Galanidi et al. 2023). Over a thousand marine species have been introduced in the region (Zenetos et al. 2022, Galanidi et al. 2023), either assisted (ship fouling or ballast waters, aquaculture escapees, aquarium releases, and other) or unassisted (dispersal through man-made corridors and canals, or unaided through natural dispersal) (Katsanevakis et al. 2013). Almost half of NIS have entered the Mediterranean through unassisted dispersal via the Suez Canal (Galanidi et al. 2023). These NIS are commonly called "Lessepsian" migrants/species (Por 1971).

A few of the Lessepsian NIS have become invasive in the Mediterranean, causing a variety of deleterious impacts on natural habitats and human activities (Katsanevakis et al. 2014, Galanidi et al. 2018). These harmful NIS create complications regarding biodiversity conservation efforts in the Mediterranean, especially in the eastern part of the basin. There is evidence that NIS may be favoured by the presence of Marine Protected Areas (MPAs) (Burfeind et al. 2013, Giakoumi et al. 2019, Kleitou et al. 2024), serving as 'safe islands' from where they proliferate further (Icarella et al. 2020, Dimitriadis et al. 2024). Additionally, harmful NIS like the siganids *Siganus rivulatus* and *Siganus luridus*, two intensive algal grazers, can cause profoundly adverse effects on MPAs, by fundamentally modifying the ecosystem structure and triggering abrupt changes, leading to complex cascading effects (Galanidi et al. 2018, Giakoumi et al. 2019, Dimitriadis et al. 2021, 2024). Climate change is projected to exacerbate this problem, by increasing the susceptibility of Mediterranean MPAs to biological invasions (D'Amen & Azzurro 2020, Icarella et al. 2020). Given the above, NIS can severely compromise the role of MPAs as a tool for marine biodiversity conservation, up to the point of reconsidering an area's protection status that no longer serves the goal of protecting the native biota (Bax et al. 2003, D'Amen & Azzurro 2020).

In the present study we used Generalised Additive Models (GAMs), developed in our previous work (Rozemeijer et al. 2025), to derive Lessepsian NIS hotspots. Specifically, we examine:

1. How much Lessepsian NIS have penetrated MPAs in the Central-Eastern Mediterranean;
2. The overlap between hotspots of Lessepsian NIS and vulnerable habitats important for conservation (fish nursery grounds);
3. The implications of the presence of Lessepsian NIS within future MPA designations by looking at a real example in the Aegean and the Ionian Sea.

8.2 Methods

We used benthic data from the Mediterranean International Trawl Surveys (MEDITS); monthly modelled data on climatic variables from the Copernicus CMEMS Mediterranean Sea Physics Reanalysis (Escudier et al. 2021) and the Mediterranean Sea Biochemistry Reanalysis (Teruzzi et al. 2021) products; and fishing effort data (Kavadas et al. 2015, 2025), from 1999 to 2021. These data were used to construct Lessepsian NIS presence/absence GAMs, as described in detail in Chapter 9 of Deliverable 4.2 (Rozemeijer et al. 2025). From the total inventory of ~300 species of Indo-Pacific origin included in the updated 2nd CIESM Atlas of Exotic Species in the Mediterranean (Golani et al. 2021, 2025), the MEDITS dataset includes 17 Lessepsian species (16 fish and 1 crustacean species), which are only occasionally caught by bottom trawls. To account for the low catchability of these NIS in MEDITS, we modelled the presence of Lessepsian NIS in general by considering hauls with at least one Lessepsian NIS as a presence. We then computed the occurrence probability of Lessepsian NIS across the Central-Eastern Mediterranean over a 0.1° hexagonal grid covering the entire area up to a depth of 1000 m.

Afterwards, for each model, we estimated the probability that maximised the agreement between observed and predicted presences/absences after accounting for chance, i.e. the probability that Cohen's kappa was maximised (Cohen 1960, Manel et al. 2001). Using that probability as the threshold for presence, we converted the models' probability predictions to presence/absence, and calculated the Getis-Ord Gi* statistic in order to identify current (as of 2021) local areas of spatial dependence (i.e. Lessepsian NIS hotspots) (Getis & Ord 1992, Ord & Getis 1995). The analysis was run for 10000 simulations on Queen's contiguity spatially weighted neighbours, and the grid cells that presented Gi*>0 and p-values≤0.01 (for either GAM model) were assigned as hotspots. Those grid cells were then grouped together, trimmed to remove land using high resolution coastline information (EEA 2017) to produce a single layer for subsequent analyses. The MPA shape files, described previously on Chapter 2, were categorised per each GFCM Geographical Sub-Area (GSA) and then filtered to keep only the European Central-Eastern Mediterranean sub-areas (GSAs 15, 16, 17, 18, 19, 20, 22, 23 and 25). MPAs shared between two GSAs were split accordingly. We then calculated the Lessepsian NIS penetration into MPAs (area overlap between NIS hotspots and MPAs) for each protection level (fully, highly, lightly, minimally, unclassified and incompatible) and for each GSA.

Additionally, we explored the implications of the presence of NIS for the preservation of vulnerable habitats (fish nurseries). Firstly, we used the Central-Eastern Mediterranean juvenile fish hotspots identified during our previous work (Hidalgo et al. 2026) to calculate their overlap with Lessepsian NIS. Sixty-one unique species, including 60 juvenile and 44 adult life-stages, were modelled through Hierarchical Modelling of Species Communities (HMSC – a joint-species distribution modelling framework) (Ovaskainen & Abrego 2020). The model's spatial predictions for each year (1999–2021), calculated on the same 0.1° hexagonal grid, were then used to identify the juvenile fish hotspots through Emerging Hot Spot Analysis (Baeza-González & Kamakura 2025, Esri 2026). Next, we estimated the overlap between the juvenile fish hotspots identified, and hotspots of Lessepsian NIS. To do so, we only considered intensifying, persistent, and consecutive hotspots.

Finally, we explored implications of Lessepsian NIS hotspots for future MPA designations in the Central-Eastern Mediterranean Sea. This was done by calculating the overlap between NIS hotspots and two new MPAs that are currently under implementation in GSAs 22 and 20 (Greek Ministry of the Environment and Energy 2025).

All plots and analyses were made in R, version 4.5.2. Manipulation of the shape files and area calculations were made using the 'sf' R package. GAM predictions were made with the 'mgcv' package. Presence thresholds were estimated using the package 'PresenceAbsence'. Gi* statistics were calculated through the 'sfdep' R package.

8.3 Results

The overlap between current Lessepsian NIS hotspots and MPAs in the Central-Eastern Mediterranean is shown in *Figure 8-1*. Most MPAs in the sub-basin were assessed as minimally or lightly protected (41.3% and 31.8%, respectively, of total MPA area) (*Figure 8-2*). Only 1.0% and 1.1% were either fully or highly protected, respectively, whereas 18.6% were MPAs of unclassified protection level and 6.1% were MPAs categorised as incompatible with conservation targets. The latter also comprised the MPA category most compromised by the presence of NIS within the sub-basin (11.88%) (*Figure 8-2*). For the other MPA categories, NIS penetration was highest in lightly protected (7.35%), followed by unclassified (5.07%) and minimally protected (4.89%) MPAs. Fully and highly protected MPAs were almost unaffected by NIS penetration (0.13% and 0%, respectively), reflecting that these areas are predominantly located outside the current extent of Lessepsian NIS (*Figure 8-1*, *Figure 8-2*).

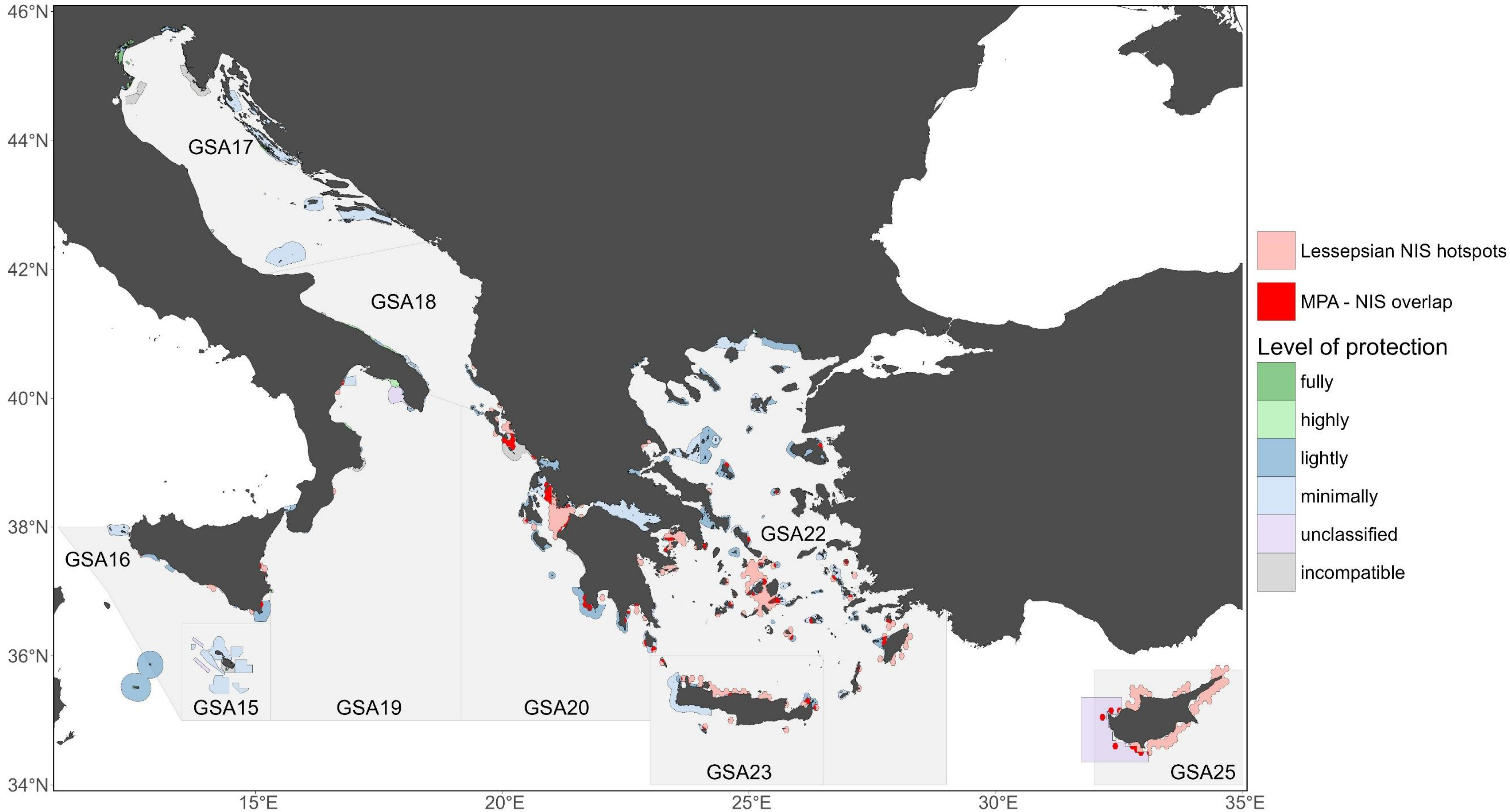

Figure 8-1. Central-Eastern Mediterranean Marine Protected Areas (MPAs) network, colour-coded by their protection status. Lessepsian non-indigenous species (NIS) hotspots as well as their overlap with MPAs are shown in pink and red, respectively. The borders and names of each European Geographic Sub-Area (GSA) are also denoted.

Interestingly, at the GSA level, MPAs in GSA 20 were the most compromised by NIS (15.53%), even though the relative NIS hotspot coverage was proportionally 3.8 times greater in GSA 25 and otherwise similar to the neighbouring GSAs 22 and 23, while the relative MPA coverage of the sub-area was similar to that of GSA 22 (*Figure 8-3*). NIS have also penetrated GSA 22 (9.06%), 23 (8.42%) and 25 (5.44%) MPAs, while GSAs 19 and 16 were affected at much lower levels (2.79% and 0.86%, respectively). MPAs of GSAs 15, 17 and 18 remain uncompromised by Lessepsian NIS, but these sub-areas have almost no active NIS hotspots presently (*Figure 8-3*).

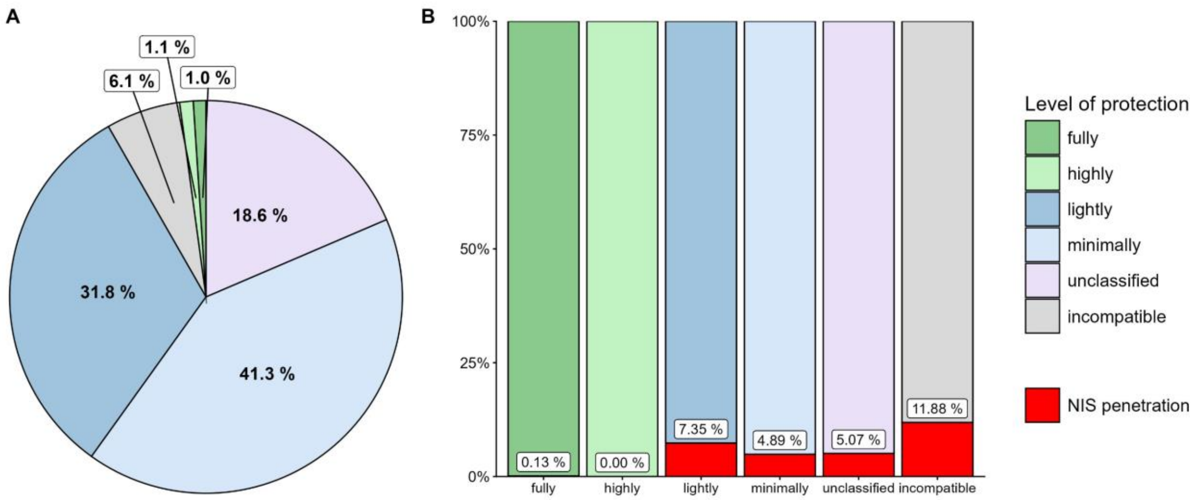

Figure 8-2. A) Percentage of total MPA area covered by each protection level and B) percentage of overlap between Lessepsian non-indigenous species hotspots and MPAs (NIS penetration) by protection status, in Central-Eastern Mediterranean.

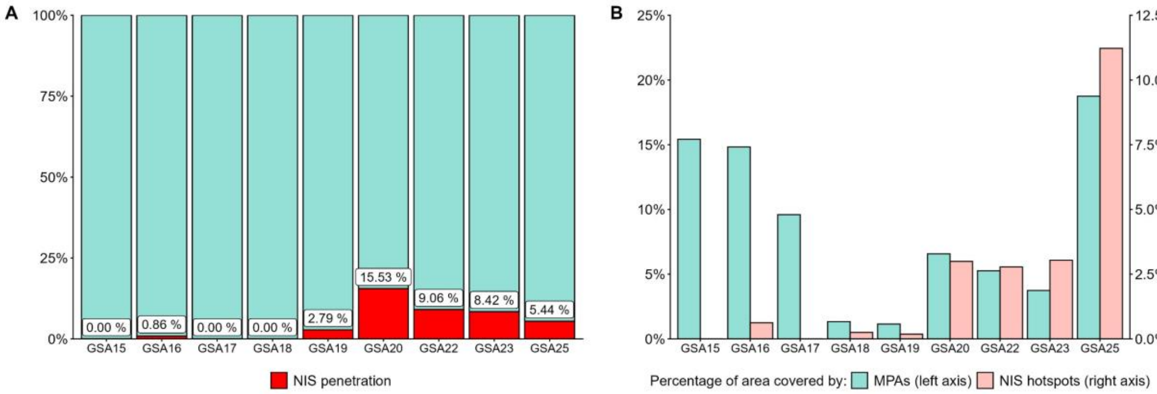

Figure 8-3. A) Percentage of overlap between Lessepsian non-indigenous species hotspots and MPAs (NIS penetration, red) by Geographic Sub-Area (GSA) and B) percentage of total sub-area covered by MPAs (cyan, left axis) along with the percentage of the total sub-area covered by NIS hotspots (pink, right axis), per GSA.

A significant overlap between Lessepsian NIS hotspots and juvenile fish hotspots was observed in the Central-Eastern Mediterranean, covering 28.94% of the total fish nursery grounds area (*Figure 8-4*). Considering the new GSA 20 and 22 MPAs, the newly introduced areas will already be compromised by Lessepsian NIS, as 9.36% and 10.97% of them will overlap with NIS hotspots, respectively (or 12.36% and 12.46% if the existing MPAs are included) (*Figure 8-5*). 5.73% and 11.97% of the new GSA 20 and 22 MPA networks, with the newly introduced areas and the existing MPAs, will contain juvenile hotspots, but most of them are already overlapping with NIS hotspots (*Figure 8-5*).

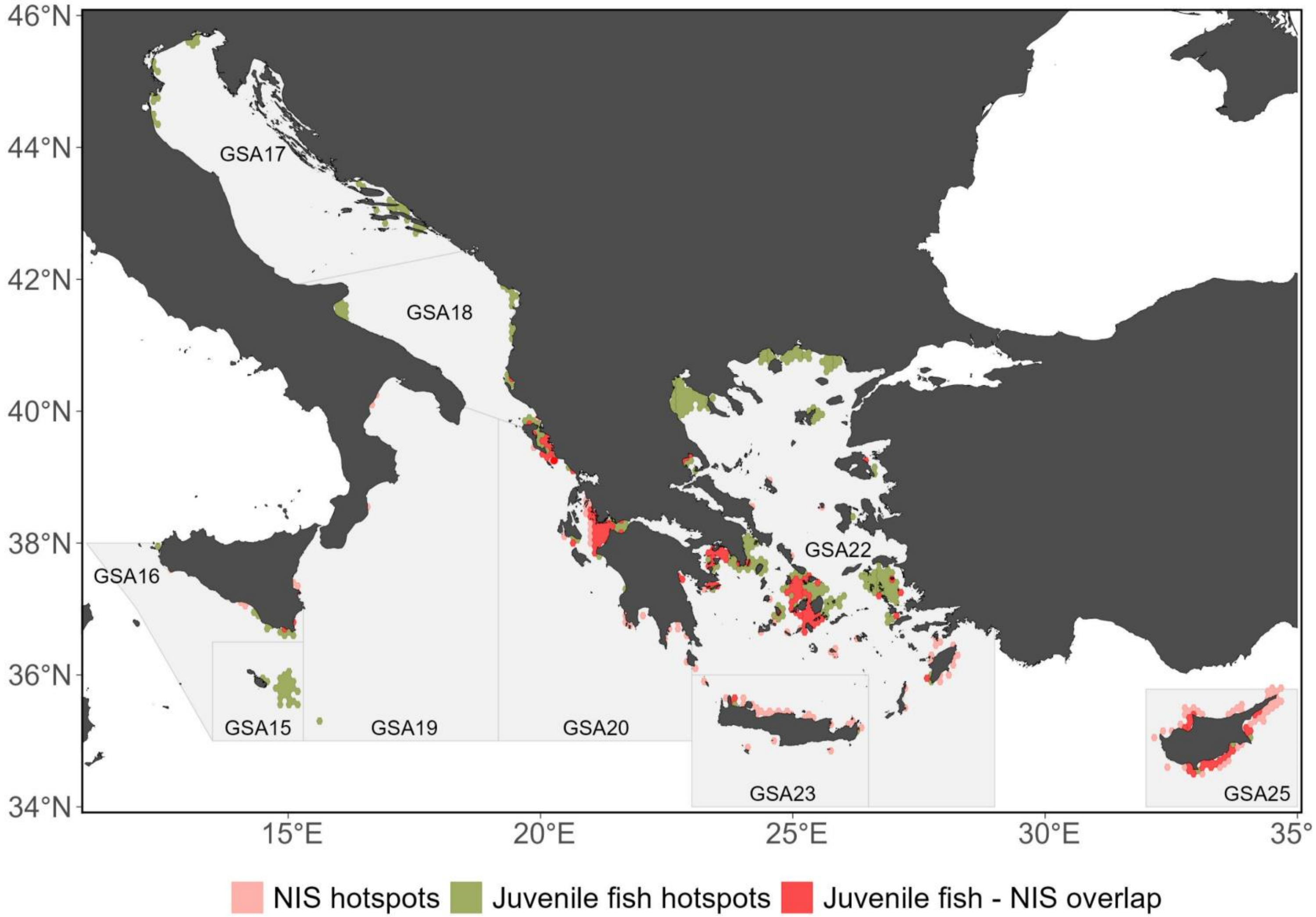

Figure 8-4. Lessepsian non-indigenous species (NIS) hotspots (pink) along with juvenile fish hotspots (green) in the Central-Eastern Mediterranean. Juvenile fish – NIS hotspots overlap is shown in red. The borders and names of the European Geographic Sub-Areas are also denoted.

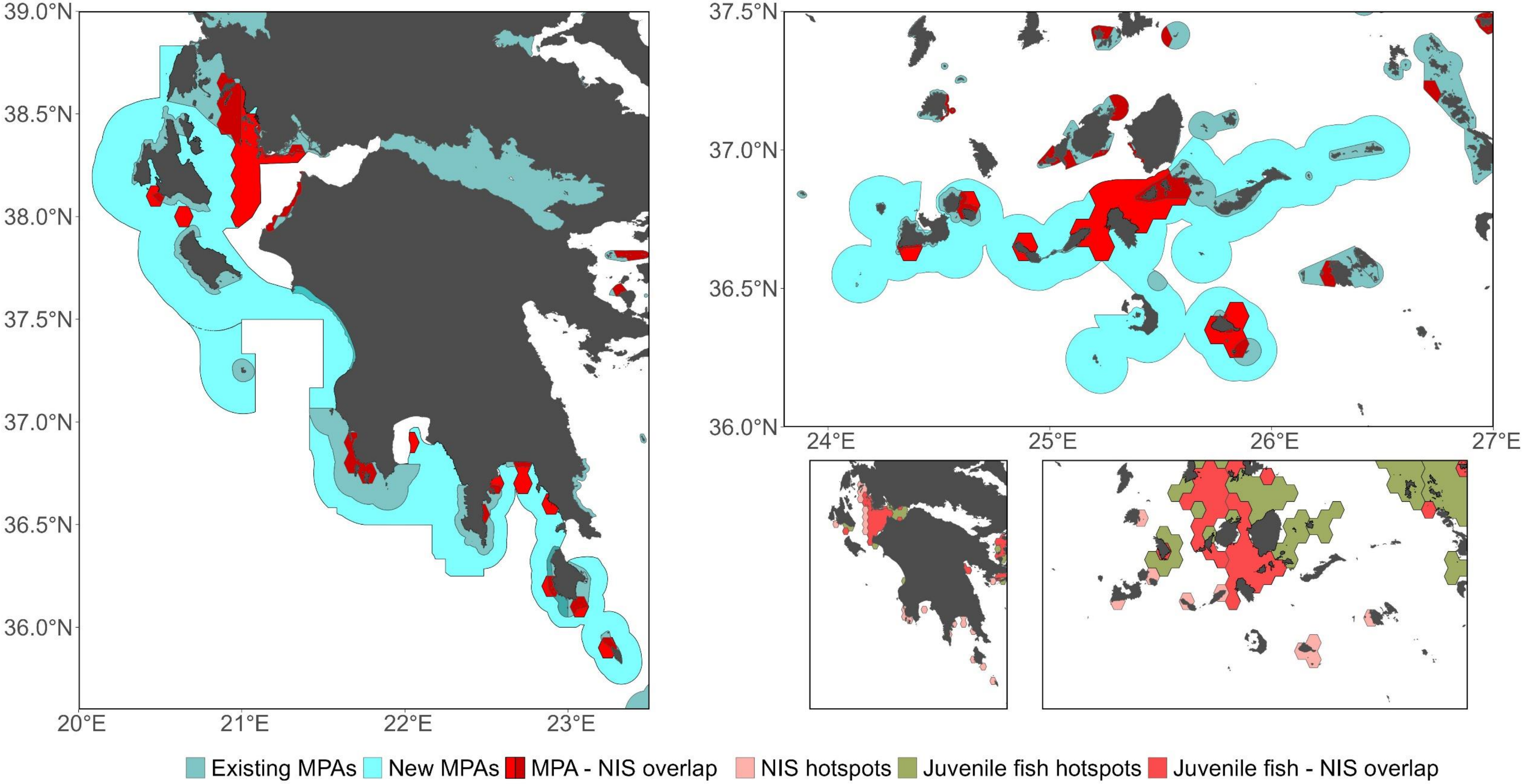

Figure 8-5. Lessepsian non-indigenous species (NIS) hotspots overlap (bright red, dark red) with the new, under implementation, MPA networks in GSA 20 (SE Ionian Sea, left) and GSA 22 (SC Aegean Sea, top right). Newly protected areas are denoted in cyan and the existing MPAs with dark cyan. Lessepsian NIS hotspots cover 9.36% and 10.97% of the newly protected areas in GSA 20 and 22, respectively (or 12.36% and 12.46% with the existing MPAs included). In the bottom right the Lessepsian NIS hotspots (light red), along with the juvenile fish hotspots (olive green) and their overlap (red), are also shown for the two areas.

8.4 Discussion and conclusions

The spread of Lessepsian non-indigenous species throughout the Mediterranean has, especially in its eastern half, raised important complications regarding the management and effectiveness of marine protected areas in the region. We found that many MPAs in the area are being compromised by the presence of Lessepsian NIS. Geographic Sub-Areas mostly affected are those in the south-east of the region (GSAs 20, 22, 23 and 25, comprising the eastern Ionian Sea, Aegean Sea, Crete and Cyprus). We also found that fish nursery grounds in the region overlap significantly with Lessepsian NIS hotspots, and may be impacted by these. Finally, two newly designated MPAs in the Aegean and Ionia Seas (GSAs 22 and 20) are already compromised due to overlapping Lessepsian NIS hotspots (10.97% and 9.36% of the newly protected areas, respectively).

In the Central-Eastern Mediterranean, 6% of the total area under any kind of protection (as defined by the criteria set out in Chapter 2) has been compromised by Lessepsian NIS, however the extent varies widely by protection status. As expected, MPAs of lower protection status tended to have higher level of penetration by NIS, and this was highest for the MPAs classed as incompatible with conservation objectives (11.88% overlap – *Figure 8-2*). Indeed, high anthropogenic disturbance can facilitate the spread of NIS, as in the case of lionfish *Pterois miles* in the Eastern Ionian Sea (Samourdani et al. 2024). Fairly high NIS penetration levels were also found for lightly protected (7.35%), unclassified (5.07%), and minimally protected MPAs (4.89%). The difference between lightly and minimally protected MPAs may well be linked with the generally westward migration path of Lessepsian species, and the MPAs they meet over that path, specifically along the Greek GSAs (22, 23 and 20). These areas might offer very little resistance to the spread of these species (Guidetti et al. 2014), or may even facilitate it (Burfeind et al. 2013, Giakoumi et al. 2019, Kleitou et al 2024).

By contrast, fully and highly protected MPAs were almost unaffected by Lessepsian NIS (0.13% and 0% overlap). Indeed, well preserved and adequately protected MPAs, with top-heavy food chains, may be better at resisting invasions or the negative impacts of harmful NIS (Guidetti et al. 2014, Holmes et al. 2019, Dimitriadis et al. 2024), although outcomes may differ widely across space and taxonomic groups (Giakoumi et al. 2017). Nevertheless, fully and highly protected MPAs only represent 2.1% of the total protected area in the Central-Eastern Mediterranean and are located outside the present extent of Lessepsian NIS. In the near future, climate change is projected to affect almost every MPA in the Mediterranean, by increasing their suitability to harbour invasive species (D'Amen & Azzurro 2020). Additionally, as shown in our previous work, Lessepsian NIS are projected to continue to rapidly spread westward and northward (Rozemeijer et al. 2025). It might therefore be preliminary to draw conclusions about the efficacy of fully or highly protected MPAs in preventing the spread of Lessepsian NIS.

We also found regional differences regarding the penetration of Lessepsian NIS into MPAs. The eastern Ionian Sea (GSA 20), a sub-area that has yet to bear the full brunt of Lessepsian invasions, emerged as the most compromised, with 15.53% of its total protected area being covered by NIS hotspots. This is, perhaps, our most worrisome find. Significantly adverse effects from the presence of NIS in this sub-area's MPAs have already been recorded, notably severe habitat deterioration due to overgrazing by *Siganus* species (Giakoumi et al. 2019, Dimitriadis et al. 2021, 2024). These habitats are important for life-cycle maintenance, and indeed we found a significant overlap between juvenile fish hotspots and NIS hotspots in the area, particularly in the Gulf of Patras and south of Corfu Island. Additionally, in our previous work (Hidalgo et al. 2026) we found that the eastern Ionian Sea was the area with the highest net losses in juvenile fish species richness. All these combined, and considering that the Ionian Sea is the pathway for Lessepsian migrants towards the north that has yet to receive the level of xenobiotic pressure the eastern areas are exposed to, marks the need for urgent measures to alleviate NIS impacts in the area. The Greek government has designated a new MPA network in the south-eastern Ionian Sea (that is currently under implementation), spanning from Antikithira Island (which is inside GSA 23) up to Lefkada Island (*Figure 8-5*), but the newly designated area will already be compromised by Lessepsian NIS. A robust plan for the management of these species is needed; otherwise, conservation goals will be also compromised. In addition, a significant part of juvenile fish nursery grounds in the area will not be covered by any protection status. Perhaps NIS management should also be extended to include these areas (Gulf of Patras, Corfu Strait) to preserve these fragile habitats.

Several MPAs within GSAs 22 (Aegean Sea), 23 (Crete) and 25 (Cyprus) have also been compromised by Lessepsian NIS. The level of NIS penetration is lower in GSA 25 than in the other two GSAs; this might be due to its MPA surface lying in waters deeper than the typical niche for Lessepsian species. Given that Lessepsian species are already well established in these areas, effort should probably be shifted towards the control and continuous monitoring of NIS populations to minimise their impact on MPAs. Targeted removals have been shown to be effective tools for the control of some NIS, notably lionfish *Pterois* sp. (Ulman et al. 2022, Kleitou et al. 2024). Incentivising hunting these species, either as a resource for food and/or other products or as sport/ecotourism (recreational fishing), will also be important for a long-term sustainable management plan (Kleitou et al. 2021, Ulman et al. 2022, Christidis et al. 2024). We expect that MPAs located in the northern Aegean Sea, currently unaffected by Lessepsian NIS, will face the same management complications sooner rather than later (Rozemeijer et al. 2025). Therefore, having a successful ready-to-go plan developed and implemented in currently compromised MPAs, will prove beneficial elsewhere in the future, possibly basin-wide. Finally, regarding the newly designated MPA in the south-central Aegean Sea, like in the case of the Ionian, much of the area will already be compromised by Lessepsian NIS. NIS management planning should also be considered to be expanded to include the quite extensive juvenile nursery grounds in the area.

In conclusion, we have shown that non-indigenous species pose a serious threat to the management of marine protected areas in the Central-Eastern Mediterranean. Much of the protected areas in the sub-basin are compromised by Lessepsian migrants, threatening conservation targets. MPAs in the eastern Ionian Sea (GSA 20) are most impacted, calling for prioritisation of this area for NIS management and impact mitigation. Long-term sustainable management plans that include targeted removals and population monitoring of NIS should be implemented in the MPAs of the Eastern Mediterranean, included in newly designated areas that might be already compromised by Lessepsian species. These management plans should be considered to be expanded in areas hosting vulnerable habitats such as fish nurseries, that are currently unprotected.

9 General discussion and management implications

9.1 Brief synopsis of key findings

This study has applied an innovative methodology to map hotspots of biodiversity, marine communities' sensitivity, vulnerability, and risk to fishing pressure and climate change across Europe's regional seas. The study then assessed to what extent there are matches or mismatches between these hotspots, and MPAs and/or other forms of spatial protection. Importantly, the degree of mismatch was found to be surprisingly high. The mismatches were either due to hotpots falling outside of MPAs or if inside MPAs, due to insufficient levels of protecting measures. Hotspots may also move through space with time as a result of climate driven range shifts of species. Overall, this urges for basin- and area-specific measures to be taken.

9.2 Usefulness of the approach

9.2.1 Strengths and weaknesses

The approach demonstrates that both the designation and management of MPAs are inherently complex, requiring a balancing of stakeholder, management, and conservation interests. It further incorporates additional, previously unused information on the sensitivities, risks and vulnerabilities of species groups and in particular on the locations where these may be highest. Moreover, these sensitivity or risk hotspots may shift over space and time as a result of climate driven range shifts of species, thereby both informing and complicating evaluation and decision-making. However, it is arguably preferable to engage consciously and make active decisions in complex contexts than to lack the information to base decisions on.

We have mapped biodiversity hotspots describing sensitivities and risks to two key pressures – climate change and fishing (Polo et al. 2025). However, other important drivers are also impacting marine life, including pollution, shipping traffic, recreational activities, oil and gas exploration, aggregate dredging, and offshore windfarms etc. (Kenny et al. 2009, 2018; Tiano & Rozemeijer in Lindegren et al. 2025, Piet et al., 2023). An extension of the present work, not attempted here, would be to include sensitivity and risk maps to these other anthropogenic pressures (Piet et al. 2023), and so obtain more integrated maps of the risks to marine biodiversity, and to what extent these match with the MPA network.

9.2.2 System behaviour and species composition

Incorporating additional information can add to the resolution of our vulnerability/risk methodology: system behaviour knowledge, species composition and biodiversity indices. Evaluating all these sources will generate more comprehensive information to underpin management decisions.

In principle the risk assessment may be simplified to a 2 × 2 matrix: with the combination of sensitivity being either low or high and pressure being low or high, yielding a matrix where risk (sensitivity and pressure combined) may be either low, high, or intermediate (in two matrix squares). In general, an increase in pressure may be expected to lead to reducing sensitivity: e.g. due to fishing pressure the more fishing-sensitive species are disproportionately affected compared to the more robust and resilient species remain (but less clear-cut e.g. for epibenthos, Rozemeijer et al., 2025), resulting in more intermediate levels of risk. In other cases, a combination of high pressure and high sensitivity is observed, resulting in high risk. For example, within the North Sea, temperature vulnerability of the epibenthos was found to be highest in the south, the part of the basin showing the greatest rate of warming (Holt et al. 2012). This resulted in particularly high temperature risk to epibenthos in the southern North Sea (Chapter 6; Rozemeijer et al., 2025; Lindegren et al., 2025)). Higher temperature sensitivity in warmer areas was also observed in Icelandic groundfish communities (Chapter 5). This seems contradictory, but could be the result from an interaction between climate change and fishing pressure, and/or other drivers (Rijnsdorp et al. 2009). Information on system behaviour may account for these patterns, such as the influx into the shallow south-eastern North Sea of relatively warm water masses from the English Channel (Cornes et al. 2023). This has been accompanied changes in composition of other species groups towards more warm-affiliated species (Hiddink & Ter Hofstede 2008; van der Kooij et al. 2016; Rozemeijer et al., 2025, Lindegren et al., 2025). The same pattern is seen in the Icelandic waters where oceanic water movements increase temperatures and Arctic and Boreal species are replaced by more temperature-tolerant Atlantic species (Chapter 5; Rozemeijer et al. 2025).

In the deeper waters of the northern North Sea, highly climate-sensitive communities of demersal fish were observed whereas the climate risk was not particularly high in these areas (*Figure B-3*). This area is more connected to the North Atlantic Ocean and has a lower rate of warming than the southern North Sea. This combination of lower climate pressure × higher climate sensitivity resulting in intermediate climate risk.

In summary, information on both pressures and individual species with their particular sensitivities adds to understanding of the risk patterns observed, and may help in addressing to what extent risks may be reduced.

9.2.3 Adding biodiversity to the equation

In general, biodiversity and sensitivity/vulnerability are often correlated: hotspots of species richness often have high prevalence of sensitive species (e.g. Trew & Maclean 2021). However, important differences also remain, and biodiversity and sensitivity hotspots do not necessarily overlap; hotspots may also depend on the type of biodiversity metric use (e.g. α-, β- or γ-diversity: Thompson et al. 2021). Since biodiversity is often the core asset to be protected, it may be valuable to incorporate it explicitly in the estimation and evaluation of hotpots.

Chapter 3 presents an exploration of this approach, where hotspots have been defined based on sensitivity, species richness and risk combined; although it does not compare the outcomes of different methodological choices. Other chapters rely solely on sensitivity, vulnerability, and risk-based approaches. A useful next step would be to investigate how these two approaches - sensitivity alone versus sensitivity combined with biodiversity - differ in practice, and what implications these differences may have for evaluating protection needs.

9.3 Protecting sensitivity versus reducing risk

In the present approach, the focus is on identifying two new hotspot types – sensitivity/vulnerability hotspots, and risk hotspots – and evaluating to what extent these overlap with MPAs and other protective measures. In various cases, clear spatial differences between sensitivity/vulnerability hotspots, and areas of high risk were observed. For example for the North-East Atlantic study area, *Figure B-1* and *Figure B-2* show marked differences between areas of high sensitivity to fishing pressure (i.e. Irish Sea and West of Ireland) and areas of high fishing pressure risk (e.g. northern Bay of Biscay). This raises questions on how to allocate conservation measures: should hotspots of sensitive/vulnerable communities be prioritised, or should priority be given to the areas at highest fishing pressure risk? Reducing pressure is expected to enhance sensitivity and support recovery, though the extent of this effect is hard to predict at this stage.

Discussion of this issue remains limited: weighing the merits of protecting highly sensitive areas versus reducing high pressures elsewhere is still an emerging topic. The International Maritime Organization (IMO) provides one relevant example through its designation of Particularly Sensitive Sea Areas (PSSAs) – areas vulnerable to damage from international shipping. To be identified as a PSSA, an area must (1) have specific ecological, socio-economic, or scientific attributes; (2) be vulnerable to damage from international shipping; and (3) have appropriate protective measures that can be adopted by the IMO to protect the attributes of the area from the vulnerability to damage by international shipping (IMO 2005). This is a management strategy focussing, within the pressure x sensitivity risk matrix, on the cell of high sensitivity x high pressure, rather than on the matrix cell with low sensitivity x high pressure. This is a typical no-regret situation in management options, where managing high pressure x high sensitivity is prioritised. This would lead to low priority given to low sensitivity x high pressure and high sensitivity x low pressure areas. Here typically a "blind spot" is noted with regards scoping and prioritising limited available resources in management effort and funding. Our tool is able to show that ecological quality in areas can improve where pressures are reduced, broadening the dialogue towards better evaluation and assignment of resources.

A similar debate arises in the field of disaster risk reduction (e.g. hurricanes, volcanic eruptions, major industrial hazards). Here, the challenge is to balance the need to protect (limited) vulnerable individuals from potential harm, while mitigating systemic risks to prevent disasters or negative outcomes (larger scale risk reduction)[1]. This raises an ethical discussion about how to allocate limited resources between areas of high vulnerability but low risk or low population numbers, and those areas of lower vulnerability but higher risk. This dialogue refers to impacts on human lives and wellbeing, versus just assigning fundings for area management, adding a dominating aspect not included in allocating funding to different marine areas (so a different discussion). Likewise, the IPCC (2014, 2022, 2023) urges to pay attention to areas with low exposure to pressures but potentially high impacts (due to high sensitivities), though without specifying how trade-offs between low pressure–high impact (sensitivity) and high pressure–low impact (sensitivity) areas should be weighted.

[1] Risk management to protect the vulnerable from climate disasters | PreventionWeb assessed 19-03-2026

Given limited funding set aside for protection and the spatial implications for stakeholders, a clearer discussion is needed on what to protect and why. An approach that is based on a pressure x sensitivity risk matrix, can help to better prioritize and clarify decisions. For example, it may generate a clearer perspective on the evaluation concerning current biodiversity hotspots with a high sensitivity but low risk, versus biodiversity "coldspots" with potential to recover once pressures are reduced, versus the needs of local users of these areas. Clearly, more in-depth evaluations are needed to shed further light on this complex issue of whether to prioritise the protection of highly vulnerable areas or the mitigation of risks in high pressure areas.

9.4 Some considerations on the level of protection of MPAs

Our study confirms 2*Figure 2-1*existing concerns on a lack of consistent levels of protection provided by MPAs (Lindegren et al. 2018; Claudet et al. 2020; Aminian-Biquet et al., 2024a, b). Many MPAs have been assigned here (in line with Aminian-Biquet et al., 2024a, b; Sim 2025) to those categories where the degree of protection and management are considered incompatible to serve protection of the species groups examined. However, it should be borne in mind that many MPAs were not originally intended for protection of the species groups considered in this report. For example, many large MPAs in the UK's EEZ within the North Sea were designed for harbour porpoises. Likewise, important MPAs in the Netherlands were assigned for e.g. seabird and marine mammal conservation. As a result, the protection measures in place may not be appropriate for the demersal fish communities covered here, or for epibenthic fauna.

Our results emphasise that there are wide discrepancies in the levels of protection that MPAs provide to different groups of marine organisms. Many hotspots of biodiversity, sensitivity or risk across European basins are not well protected. It is acknowledged that the levels (management) and areas of protection may be unlikely to change appreciably in the near future, owing to the following factors:

- Interest and area use by many different stakeholder groups often overlap. The establishment of MPAs is therefore the outcome of a complex negotiation process balancing the priorities of both stakeholders and environmental conservation. This can lead to suboptimal outcomes for either party.
- Once an MPA has been designated, the resulting agreement is typically highly conservative, as any subsequent renegotiation would require substantial time and effort (Aminian-Biquet et al. 2024a, b).
- Once an MPA is designated, the resulting agreement is typically highly conservative, as renegotiation would require significant time and effort (Aminian-Biquet et al. 2024a, b).
- MPAs may be designated for a wide range of purposes, e.g. ranging from protecting seabirds or marine mammals to conserving cold-water corals, each requiring distinct measures to manage human activities.
- Wide differences in the size and extent of MPAs can make protective measures more or less costly or effective.

- Some MPAs are situated near densely populated coastal areas or major shipping routes, whereas others are located in more remote regions, with implications for the pressures imposed on marine life.

Policy managers should recognise the complexity of nature values that are dynamic across both time and space. Our results highlight the tension between these inherent ecological dynamics and the more structured, conservative management approaches that stakeholders often prefer to have predictable conditions for use of the marine environment.

In addition, our results emphasise that establishing MPAs mainly driven by the 30 x 30 target may not mean that 30% of marine life is appropriately protected. A more thorough means through ecological criteria is offered with our approach to support this evaluation. Purely spatial management may be insufficient, in addition also needed: temporary closures, additional management options, bespoke measures for specific pressures

Furthermore, our results emphasise that establishing MPAs primarily to meet the 30 × 30 target does not necessarily ensure that 30% of marine life is adequately protected. Our approach, which relies on ecological criteria, provides a more robust basis for evaluating protection levels. Solely spatial management is unlikely to be sufficient; additional measures such as temporary closures, supplementary management tools, and bespoke interventions for specific pressures are also required.

On the other hand, sometimes our categorisations may also underestimate the level of protection in some MPAs. The polygons and classifications used do not always reflect the most recent information on spatial extent, seasonal measures, or gear-specific restrictions (Aminian-Biquet et al. 2025b). In practice, many North Sea MPAs are subject to complex fine-scale regimes in which restrictions may apply only to particular zones, specific bottom-contacting gears, or defined periods of the year. Recent developments under the North Sea Agreement illustrate this complexity. Since 2023–2025, Dutch MPAs such as the Frisian Front, Central Oyster Grounds, Klaver Bank, and parts of the Dogger Bank have been partially closed, with several areas now largely off-limits to bottom-trawling. Additional seasonal closures—such as half-year restrictions on gillnets and seine nets for moulting guillemots in the Frisian Front—and the introduction of Real-Time Closures based on catch data further add to the nuanced management. *Table **6**-**1***Such nuances are not fully captured in the harmonised shapefiles used in this analysis, which simplifies protection status to polygon-level categories. The next step should therefore integrate more detailed information on actual management measures within each MPA, their target groups, and how these align with the species or pressures examined. Given the very large number of MPAs in European waters 2a focused approach using a selected subset of MPAs is advisable.

9.5 Management implications per basin

At the heart of B-USEFUL is the intention to provide tools for policy making on marine biodiversity. Below, we present an overview of important take-home messages for biodiversity policy makers and area managers, emerging from this report's theme around biodiversity, sensitivity and risk hotspots, and their overlap with protected areas in European waters. First, we highlight key management advice at the basin or area level; next, the management implications beyond basins or areas are presented, valid more widely across European regional seas and beyond.

9.5.1 Mediterranean Sea

(1) Which biodiversity hotspots are appropriately protected?

- Only a very limited portion of biodiversity hotspots overlap with protected areas. Hotspots of species richness and climate sensitivity have slightly better coverage in the Western Mediterranean, where the overall extent of MPAs is larger. In spite of this, most of this coverage is in areas with relatively low levels of protection, underlining the limited benefit for conservation purposes.

(2) Where are the most "urgent" mismatches between hotspots and (a lack of) protection?

- The most urgent mismatches occur in hotspots of climate change risk, fishing pressure risk, and fishing pressure sensitivity, where the majority of the area remains outside protected zones.
- Species richness and climate change sensitivity hotspots are somewhat better protected by the existing MPA network and can therefore be considered of medium priority.

(3) At basin level: To improve protection, could policy recommendations either build on existing MPAs, or are fully new management measures needed?

- Policy efforts could focus on increasing the level of protection and management effectiveness within existing MPAs at the basin level to gain stronger ecological benefits – especially in the Western Mediterranean Sea, where MPAs currently have wider spatial coverage despite their relatively low protection levels.
- Expanding the current MPA network by establishing new protected areas is recommended – especially in the Eastern Mediterranean Sea, where the existing network is still relatively limited in spatial extent (much more than in the West).
- Including Fishery Restricted Areas (FRAs) within the MPA network could also be beneficial, as these areas are already subject to fisheries management measures, so any additional closure or strengthening of protection may be of lower economic impact, while still enhancing biodiversity conservation.

(4) At the European level: what broad-scale recommendations are emerging from this case study, relevant to biodiversity protection across European basins?

- Expansion of strictly protected areas, which are currently underrepresented in the Mediterranean basin, particularly in the deep strata.
- Strengthening protection levels and management effectiveness within the existing network, where coverage is broader but largely dominated by low-stringency categories.
- Rearrangement of the actual MPA network to better align protection levels with biodiversity and risk hotspots.

- Strengthening transboundary cooperation to support the achievement of international targets, including the 30×30 target, while safeguarding and enhancing ecosystem resilience through effective biodiversity conservation measures.

9.5.2 Northeast Atlantic

(1) Which biodiversity hotspots are appropriately protected?

- Very few biodiversity hotspots are currently adequately protected within the current MPA network in the area encompassing the Greater North Sea, Celtic Seas and Bay of Biscay and Iberian Coast.

(2) Where are the most "urgent" mismatches between hotspots and (a lack of) protection?

- The highest priority, and most urgent, mismatches are in the Northern North Sea and Skagerrak Strait where communities are highly vulnerable to warming but largely unprotected. These are major climate-sensitive hotspots with very limited MPA coverage. The Bay of Biscay and Iberian Coast also have large climate-risk hotspots driven by strong exposure to warming, but strict protection is almost absent.
- Lower-priority mismatches are those in the Celtic Sea and northern Bay of Biscay where there is some foundation for strengthening protection.

(3) At basin level: To improve protection, could policy recommendations either build on existing MPAs, or are fully new management measures needed?

- Improve the effectiveness of existing MPAs, upgrading minimal restrictions to higher protection levels (including restrictions on high-impact fishing gears, and/or extractive industries).
- Establish new protections in regions such as the northern North Sea and Bay of Biscay.
- Expand MPA boundaries and improve protection in the Celtic Sea.

(4) At the European level: what broad-scale recommendations are emerging from this case study, relevant to biodiversity protection across European basins?

- Area-based targets alone (e.g., "30 by 30") are insufficient if protection does not cover ecologically vulnerable areas. Policy at the European and global level should prioritise representativity and ecological effectiveness, not just total protected area.
- Protection strategies should combine strict protection in key biodiversity hotspots and climate-adaptive management across larger ecosystems.

9.5.3 Icelandic waters

(1) Which biodiversity hotspots are appropriately protected?

- A limited extent of the biodiversity hotspots identified during the analysis are currently protected by the fishery-restricted area network.
- The hotspot that is best represented in the network is located offshore western Iceland (within cluster #2, see *Figure 5-2*); however, this hotspot is protected only for half of the year by a seasonal closure to bottom-contact fishing gear.

(2) Where are the most "urgent" mismatches between hotspots and (a lack of) protection?

- The most urgent hotspots are the ones defined by high biomass levels of several Arctic fish species, which are the most sensitive to climate change and fishing pressure in the region. These hotspots are located north and west of Iceland.
- Other hotspots in the south and west are instead defined by fewer species, i.e. the hotspots defined by high biomass levels of at-risk or commercially important species.

(3) At basin level: To improve protection, could policy recommendations either build on existing MPAs, or are fully new management measures needed?

- Fully protected MPAs could be placed where Arctic species are present.
- Some of the locations where Arctic species are present are already largely covered by seasonal closures, while others are partially covered by permanent ones. Hence, it could be possible to either render the closures permanent or to expand the areas.
- Regarding the other biodiversity hotspots, measures tailored to the behaviour of at-risk or commercially important species could be enforced (e.g., fishing bans enforced only during the spawning season or during migrations). This would likely generate completely new closures (e.g., on the Reykjanes Ridge).

(4) At the European level: what broad-scale recommendations are emerging from this case study, relevant to biodiversity protection across European basins?

- While benefits for biodiversity can be obtained from measures aimed at conserving commercially important fish stocks, attention should be paid to the protection of marine groundfish communities sensitive to climate change and fishing pressure.
- In the North Atlantic and southern Arctic Ocean, these communities often include relatively high abundance of Arctic species. To achieve the biodiversity targets, MPAs should be placed also where high abundance of Arctic species is found.
- If only a limited number of vulnerable species defines a biodiversity hotspot, as may be the case in (sub-)Arctic waters, enforcing seasonal protection measures that are tailored to specific species' behaviour may be sufficient for their conservation – unless species are non-migratory, or are tightly linked with vulnerable habitats (e.g., cold-water coral reefs).

9.5.4 North Sea epibenthos

(1) Which biodiversity hotspots are appropriately protected?

- Two different epibenthic hotspot types were investigated: based on vulnerable communities and sensitive habitats. Of the most sensitive habitats identified for the North Sea, only 8.5% were located inside MPAs. Of the top 5% most vulnerable epibenthic communities, some 19% were located inside North Sea MPAs, generally offering low levels of protection.

(2) Where are the most "urgent" mismatches between hotspots and (a lack of) protection?

- For sensitive habitat hotspots, not one receives the protection needed since they are either outside MPAs or in an MPA classed as 'incompatible,' due to fishing and/or dredging activities being allowed. Also, for the vulnerable epibenthic communities no real protection is offered, with all 126 hotspot sites in areas subject to fisheries and dredging.

(3) *At basin level: To improve protection, could policy recommendations either build on existing MPAs, or are fully new management measures needed?*

- Epibenthos (and benthic and demersal communities) are especially vulnerable to bottom contact fisheries and aggregate dredging. Given the difficulties in establishing MPAs and/or altering their boundaries, a stepwise approach is suggested:
    - First, increase level of protection from bottom contact fisheries and estimate the attributed value.
    - Second, consider protection from dredging in specific MPAs, especially those with gravelly substrate (rare) and where sensitive habitats were defined in this study.
    - Third, additional areas could be established with appropriate levels of protection.
    - Each step should carefully be evaluated on its merits in protecting epibenthos and other bottom related communities

(4) *At the European level: what broad-scale recommendations are emerging from this case study, relevant to biodiversity protection across European basins?*

- This study demonstrated that hotspots of epibenthic vulnerability (and likely other biodiversity hotspots) can shift over time, within decades. Shifting biodiversity hotspots are challenging to accommodate in rigid marine spatial planning and regulations. It seems wise to have warning signals in place and apply this methodology across European basins, to assess potential shifts in vulnerability hotspots over time. From monitoring results an urgency can be derived to apply additional spatial-temporal management measures (e.g. temporary area closures) to hotspots in need of protection.

9.5.5 Benthic habitats in Portuguese waters

(1) *Where are the most "urgent" messages and key policy recommendations regarding risk to sensitive benthic habitats and their protection?*

- In continental Portugal's EEZ subarea, most seafloor habitat (90.97%) is unprotected, with only 9.03% within MPAs. Although Portugal's 30x30 CBD target also covers the Azores, Madeira and extended continental shelf, protection in the continental EEZ subarea alone is likely insufficient for an ecologically coherent MPA network. Coherence requires habitat representativity and replicability across subareas- ensuring the 30% designated in each subarea includes all habitat types and environmental gradients. A regionally balanced approach would better maintain biodiversity heterogeneity and resilience under current and projected climate-change stressors.
- Most MPAs currently allow low- to moderate-impact activities, while restricted areas cover only 0.02% of the EEZ. This highlights the urgency to identify high-conservation-value areas for restricted protection within existing and new MPAs to meet the CBD

target of 10% restricted protection by 2030, especially for sensitive habitats in high-risk areas.

- Key recommendations by benthic habitat type:
    - **Maërl:** Rare, pH-sensitive habitat with limited extent; its full area should be covered by highly restrictive measures. This requires upgrading protection across the 68.1% currently lightly protected and extending protection to the unprotected 11%. Further mapping is needed to identify additional occurrences.
    - **Seamounts (summit <1000 m)** – Ecologically important but high-risk, climate-sensitive habitats (temperature and pH). Most are within MPAs but lack strict protection; strengthen measures within existing MPAs.
    - **Rocky reefs (all depths on the continental shelf area)**: Highest ecological value and high climate risk, especially from pH change. Limited restricted protection creates a major conservation gap that should be addressed urgently.
    - **Macroalgae forests** (coastline to 20 m depth, Camões Bank, Gorringe Seamount): High temperature-risk habitat, of high ecological value. Expand MPA coverage (coastline to 20 m; Camões Bank), add strict protection zones in new and existing MPAs, and protect across temperature gradients and potential climate refugia (based on modelling projections).
    - **Mud volcanoes and cold seeps**: Rare, ecologically unique habitats with no current MPA coverage and high temperature risk. Key threats also include oil and gas extraction, seabed mining, and bottom trawling. Identify additional occurrences. Prioritise inclusion in MPAs and map more sites.
    - Several deeper-water habitats (e.g., **slope and ramp rocky reefs, soft sediments, and seabed**) currently lack MPA coverage. **Canyons and abyssal plains** have very limited coverage (<8%). Despite low or intermediate climate vulnerability, they support distinct communities, so MPA coverage should be strengthened to ensure representativity and replicability, prioritising areas in best condition or under higher pressure (e.g. from fishing).
- Overall, for all MPAs in the continental Portuguese EEZ subarea, monitoring is essential to assess management effectiveness, guide improvements, and inform the planning and designation of additional MPAs.

(2) *At the European level: given the overview of the protection of sensitive habitats across European basins, can a general advice be defined?*

- With climate change, well-designed MPA networks strengthen ecological resilience by ensuring connectivity and covering environmental gradients, enabling species to shift between areas with different climatic exposures and supporting faster recovery. MPA planning and designing **in each country's MPA network** should therefore:
    - **Ensure habitat representativity along climatic gradients** (and biogeographical, where applicable), reflecting current and projected species distributions and promoting connectivity where possible.
    - **Include all distinct habitat types**, using their natural size and distribution to set appropriate representativity and replicability targets.

- **Promote functional redundancy,** prioritising habitat areas with the highest functional biodiversity, backed by appropriate protection and a balanced combination of strict and managed zones.
- **Integrate the information from risk maps and climate projections** to identify climate refugia (least exposed to climate change) and protect their habitats from additional human pressures through restricted protection measures.
- **Improve the spatial resolution of habitat mapping.** At European level, mapping for small-extent habitats (e.g., maërl, biogenic reefs, mud volcanoes and cold seeps, macroalgal forests, and soft-sediment biogenic aggregations) is often coarse or absent. Better mapping is needed to support new MPA designation and appropriate management. This is essential not only to achieve CBD 30x30 targets but also deliver under the Marine Strategy Framework Directive and the EU Nature Restoration Law.

9.5.6 Mediterranean non-native species

(1) Which biodiversity hotspots are appropriately protected?

- Currently, MPAs of GSAs 15, 16, 17 and 18 (Malta, South Sicily, North and South Adriatic Sea, respectively) remain almost unaffected by Lessepsian migrants, but this can change rapidly in the near future as these NIS continue spreading northward and westward. A combination of continuous monitoring and when needed appropriate protection measures is advised (active adaptive management).
- ‘Fully’ and ‘highly’ protected MPAs remain fairly uncompromised, but this is most probably due to their locations being outside the path of the Lessepsians rather than an inherent biotic resistance to invasions. For the moment, continuous monitoring to allow early detection is advised, in order to be able to react promptly and effectively when needed (anticipated adaptive management).

(5) Where are the most “urgent” mismatches between hotspots and (a lack of) protection?

- Management actions should be prioritized in GSA 20 to minimise Lessepsian NIS impacts. GSA 20 MPAs (eastern Ionian Sea) are the most compromised by NIS between the Central-Eastern Mediterranean sub-areas (for existing MPAs, new MPAs, and juvenile hotspots), and it has yet to suffer the full scale of the Lessepsian invasion.
- Permanent management plans for NIS population control should be implemented in the Eastern Mediterranean MPAs (GSAs 22, 23 and 25, Aegean Sea, Crete and Cyprus respectively), included in newly designated MPAs that may be already compromised by NIS.
- NIS management should be extended to include locations with vulnerable habitats (e.g. spawning/nursery grounds) that are currently unprotected.

(2) At basin level: To improve protection, could policy recommendations either build on existing MPAs, or are fully new management measures needed?

- New management plans should be devised and implemented specifically for NIS population control. Drawing from successful management examples, these plans should contain monitoring of their populations and regular targeted removals of harmful NIS.

- In the long-term, stakeholder and public involvement to incentivise the sustainable exploitation (harvesting) of NIS as new resources will be key. Amongst others, collecting catch data will also facilitate the regular assessment of these species to effectively monitor their stocks in the basin.
- Fisheries can be employed for NIS population control outside MPAs, if the proper incentives for fishers are created. Artisanal fisheries operating in lightly or minimally protected areas could employ highly selective gears and métiers that target these species. Research on the development of new NIS-selective fishing gears is highly recommended.

(3) *At the European level: what broad-scale recommendations are emerging from this case study, relevant to biodiversity protection across European basins?*

- Currently, a specific MPA management framework for NIS is lacking in the EU. Member states will benefit greatly by a ready-to-deploy comprehensive and yet specifically adjustable NIS management framework.
- Continuous research to fill the knowledge gap regarding the biology and the impacts of these frequently understudied species is essential to devise successful site and species-specific management plans.

9.6 Generalised management implications at the European level

At the European level, important broad-scale recommendations emerge from each of the above case studies. This section synthesises these recommendations in a general perspective valid for biodiversity protection across all basins.

- We have an informative methodology demonstrating important biodiversity and risk hotpots, so far applied to limited groups of species. It is strongly advised to apply this methodology on all relevant species groups across sea basins.
- It has become apparent that the current MPA network insufficiently protects biodiversity and risk hotspots: across sea basins a mismatch between important areas and protection was noted. Better alignment with established MPAs is recommended. This implies expanding the total area of MPAs across basins.
- In addition, in many MPAs the levels of protection were found to be relatively low. This calls for strengthening of protection levels and management effectiveness within the existing and future network. Especially expansion of strictly protected areas is needed, which are currently underrepresented in all European basins.
- Biodiversity or sensitivity hotspots can shift over time, which is challenging to accommodate in rigid marine spatial planning and regulations. Appropriate and regular monitoring and risk assessment will generate management warning signals that could indicate potential shifts in vulnerability. Such information could underpin decisions on additional spatial-temporal management measures (e.g. temporary area closures) to hotspots in need of protection.
- Across all basins, there is a need for strengthened monitoring to support system understanding and informed decision-making.

- Special attention is needed for highly vulnerable habitats and species groups, particularly deep strata and Arctic areas.
- Non-indigenous species require adaptive management Europe-wide and especially in the Eastern Mediterranean. Drawing from successful management examples, these plans should contain monitoring of their populations and regular targeted removals of harmful NIS, especially where these are entering MPAs.
- As marine basins are typically shared between countries, it is advisable to strengthen European collaboration in both monitoring and management. Transboundary cooperation will support the achievement of (inter)national targets, including the 30×30 target, through effective biodiversity conservation measures.

9.7 Perspectives

For the first time, we have mapped hotspots of biodiversity, its vulnerabilities and risks across European regional seas using a comprehensive approach. We have also assessed to what extent these hotspots are sufficiently protected through Europe's MPA network, or are lacking adequate protection levels. This has greatly enhanced our ability to evaluate the appropriateness and effectiveness of the current MPA network for biodiversity protection. Our findings indicate that biodiversity or sensitivity hotpots for many of the species groups examined are not well protected, urging for a revaluation of the spatial designations and existing management measures.

Looking ahead towards a more thorough reassessment, we recommend that the approach is extended to assess the hotspots of biodiversity, vulnerability and risks for the many other marine taxonomic groups inhabiting the seas of Europe. A temporal analysis should be included in that approach to assess for any shifts in space and time of both hotspots and the pressures. This should include the engagement of human stakeholders and a consideration of their interests. Combining all these aspects, this would allow for a more holistic assessment, integrating the needs for human society and the needs and risks to marine life when assessing how to optimise the placement and management of the European MPA network.

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A. Appendix: Mediterranean Sea

Table A-1. Fishery-restricted areas (FRAs) considered along with their zone (A, B, C, Core, Buffer), and the corresponding pressure levels of major human activities (mining, dredging and dumping, anchoring, infrastructure, aquaculture, fishing, and non-extractive activities). Scores (0–4) indicate the degree of impact associated with each activity, while the final column reports the protection level based on the MPA classification (Highly, Lightly, Minimally, Incompatible).

FRA	Zone	Mining	Dredging & Dumping	Anchoring	Infrastrucure	Aquaculture	Fishing	Non-extractive Activities	Protection level (MPA based)
Pomo/Jabuka Pit	A	0	0	1	0	0	1	2	**Highly**
	B	0	1	2	1	0	2	2	**Lightly**
	C	0	1	3	1	0	2	2	**Minimally**
Bari Canyon	A	0	0	2	0	0	1	2	**Highly**
	B	0	1	3	1	0	2	2	**Minimally**
Otranto channel	A	0	0	2	0	0	1	2	**Highly**
	B	0	1	3	1	0	2	2	**Minimally**
Gulf of Lion	A	0	0	2	0	0	1	2	**Highly**
	B	0	1	3	1	0	2	2	**Minimally**
Lophelia reef off Capo Santa Maria di Leuca	core	0	0	1	0	0	2	2	**Lightly**
East of Adventure Bank (Strait of Sicily)	core	0	0	2	0	0	2	2	**Lightly**
	buffer	0	1	3		0	4	2	**Incompatible**
West of Gela Basin (Strait of Sicily)	core	0	0	2	0	0	2	2	Lightly
	buffer	0	1	3	1	0	4	2	**Incompatible**
East of Malta Bank (Strait of Sicily)	core	0	0	2	0	0	2	2	**Lightly**
	buffer	0	1	3	1	0	4	2	**Incompatible**

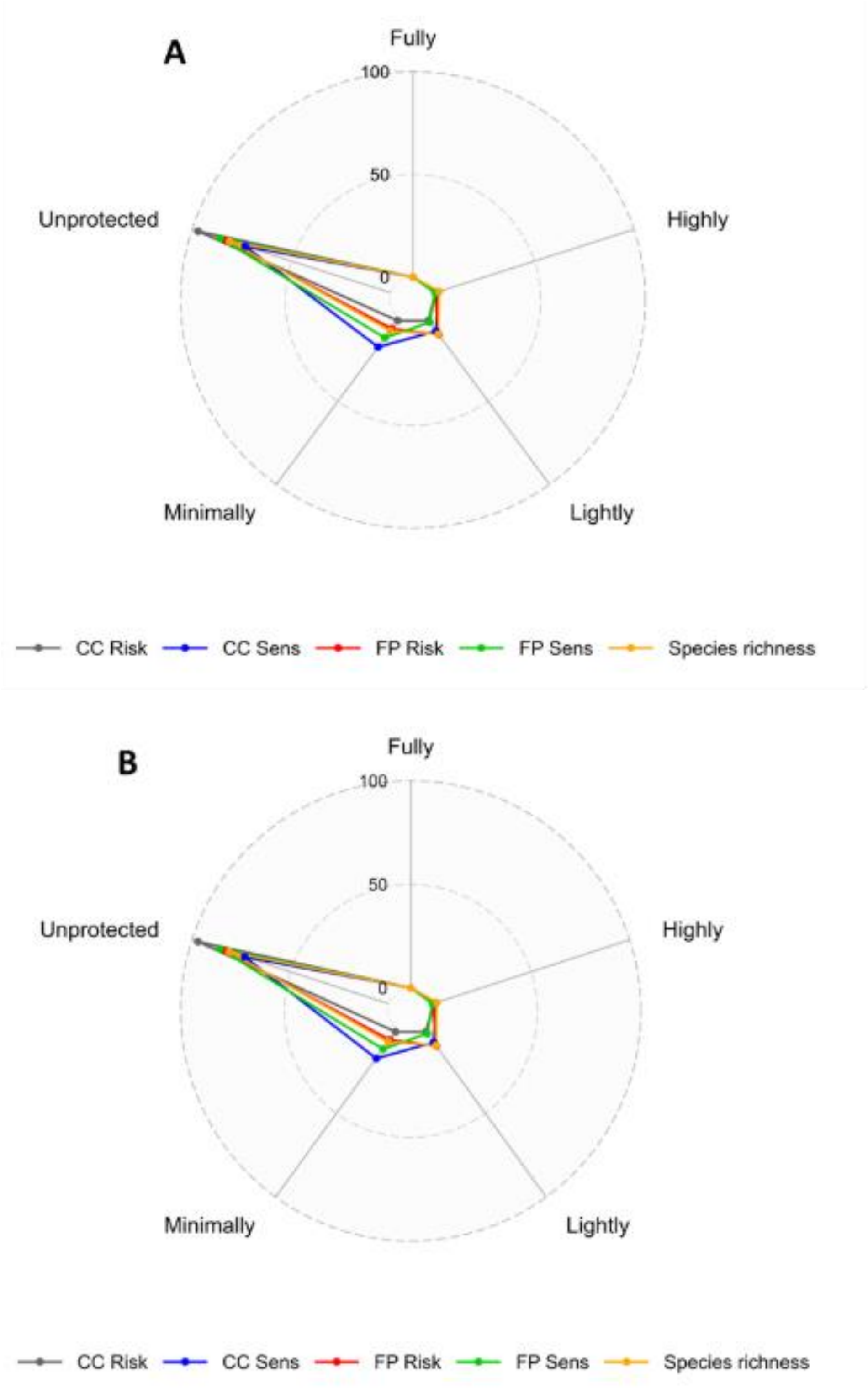

Figure A-1. Radar plot showing the hotspot surface distribution across protection levels under MPA-only scenario (A) and MPA + FRA scenario (B).

B. Appendix: Northeast Atlantic

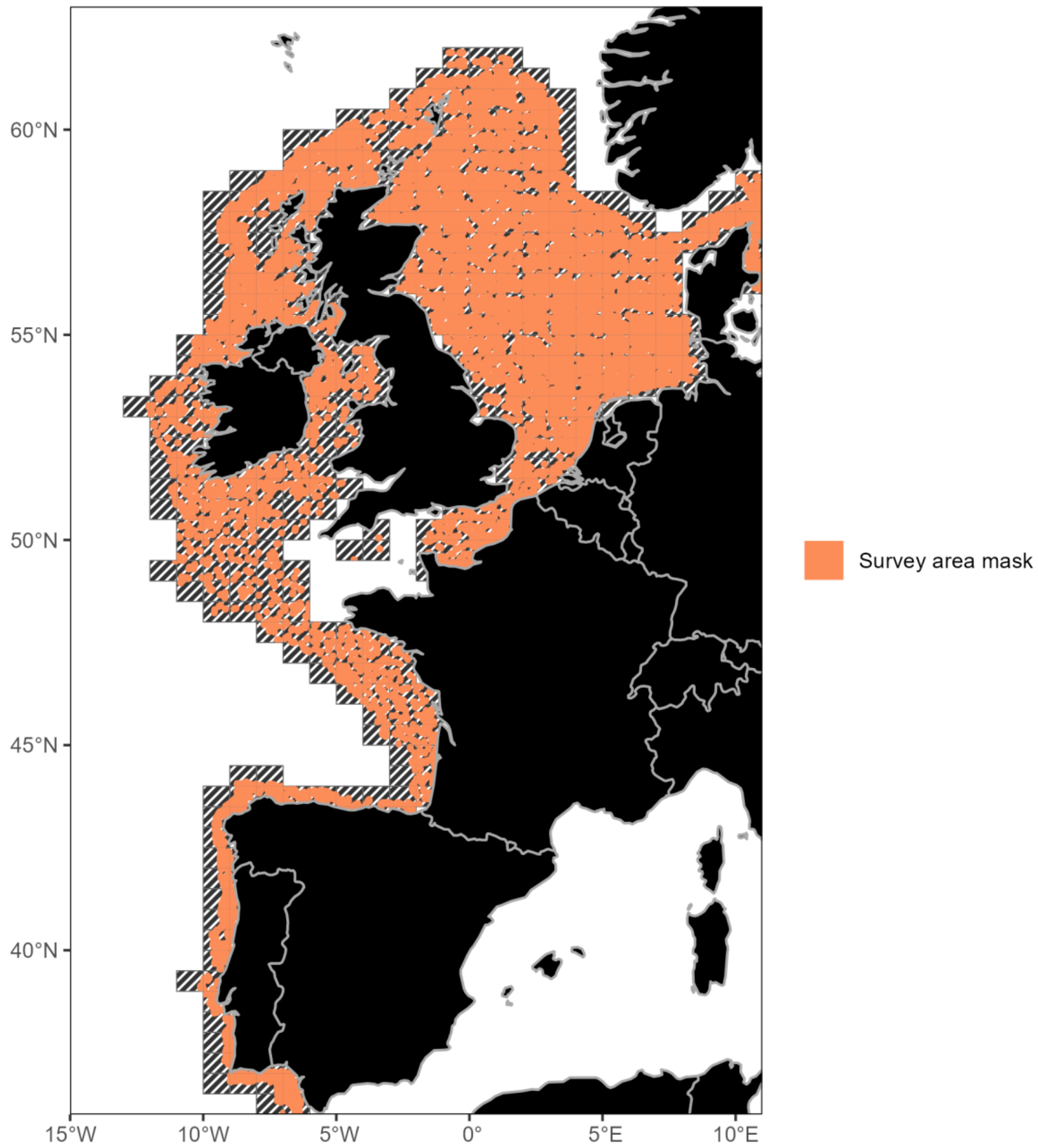

Figure B-1. Mask of survey area used to crop grid cells of hotspot values. Point data of hauls plus a 10 km buffer were used to create the mask. Black and white striped area represents the ICES grid cell study regions used to combine all species trait information from hauls within the cell and calculate the corresponding indices (sensitivity, risk, hotspots) at the ICES rectangle scale. Orange shows the survey data from FISHGLOB.

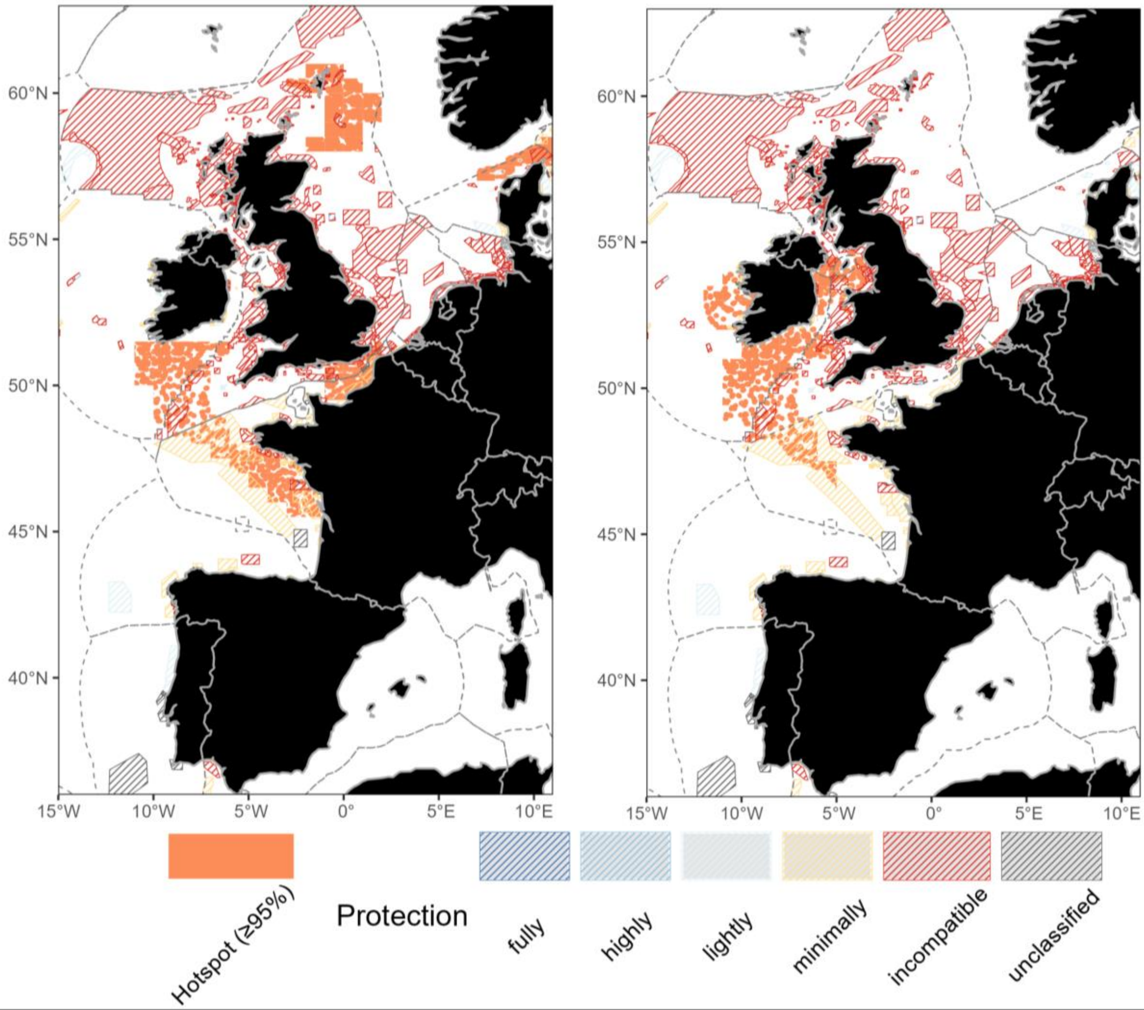

Figure B-2. Hotspots of community-level risk (left) and sensitivity (right) to fishing and their overlap with marine protected areas. Hotspots of sensitivity are shown in orange. Hashed areas indicate MPAs and colours represent the level of protection (from fully protected to unclassified). EEZs marked with dashed grey lines. Cells are classified as hotspots at 95% confidence represent areas where high sensitivity/risk values are spatially clustered more strongly than expected by chance.

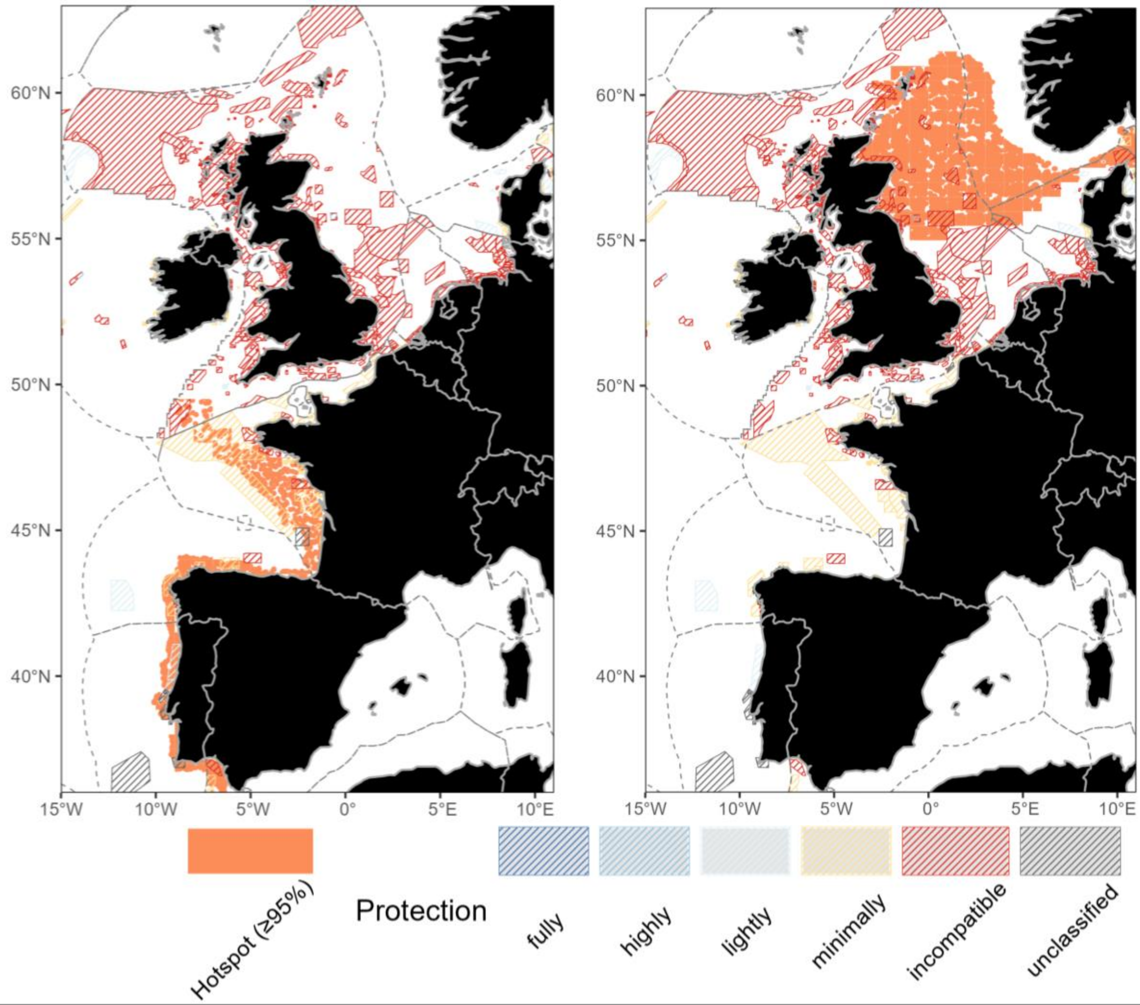

Figure B-3. Hotspots of community-level risk (left) and sensitivity (right) to climate change and their overlap with marine protected areas. Hotspots of sensitivity are shown in orange. Hashed areas indicate MPAs and colours represent the level of protection (from fully protected to unclassified). EEZs marked with dashed grey lines. Cells are classified as hotspots at 95% confidence represent areas where high sensitivity/risk values are spatially clustered more strongly than expected by chance.

C. Appendix: Icelandic waters

Table C-1. Weights associated with each feature during the prioritisation.

Group	Group's weight	Feature	Individual weight	Effective weight
Arctic species	26.00	D_{aff} <127m	30	9.750
		127≤ D_{aff} <216	20	6.500
		217≤ D_{aff} <431	10	3.250
		432≤ D_{aff} <701	10	3.250
		D_{aff} ≥701	10	3.250
Species sensitive to climate change	13.50	D_{aff} <127m	30	5.063
		127≤ D_{aff} <216	20	3.375
		217≤ D_{aff} <431	10	1.688
		432≤ D_{aff} <701	10	1.688
		D_{aff} ≥701	10	1.688
Species sensitive to fishing pressure	38.50	D_{aff} <127m	30	14.438
		127≤ D_{aff} <216	20	9.625
		217≤ D_{aff} <431	10	4.813
		432≤ D_{aff} <701	10	4.813
		D_{aff} ≥701	10	4.813
Commercial species	7.25	*Anarhichas lupus*	1.3	0.094
		Brosme brosme	0.2	0.015
		Gadus morhua	63.2	4.582
		Lophius piscatorius	0.2	0.015
		Melanogrammus aeglefinus	13	0.943
		Microstomus kitt	0.5	0.036
		Molva molva	1.1	0.080
		Pleuronectes platessa	1.9	0.138
		Pollachius virens	8.6	0.624
		Sebastes norvegicus	10	0.725
At-risk species	19.75	*Anarhichas denticulatus*	4	2.926
		Anarhichas minor	1	0.731
		Centrophorus squamosus	4	2.926
		Chimaera monstrosa	1	0.731
		Coryphaenoides rupestris	4	2.926
		Hippoglossus hippoglossus	2	1.463
		Molva dypterygia	2	1.463
		Reinhardtius hippoglossoides	1	0.731
		Sebastes mentella	4	2.926
		Squalus acanthias	4	2.926
Marine pressures	-5.00	Bottom trawling intensity	1	-5.000
Total:	**100.00**		**Total:**	**100.000**

D. Appendix: Epibenthic vulnerability in the greater North Sea

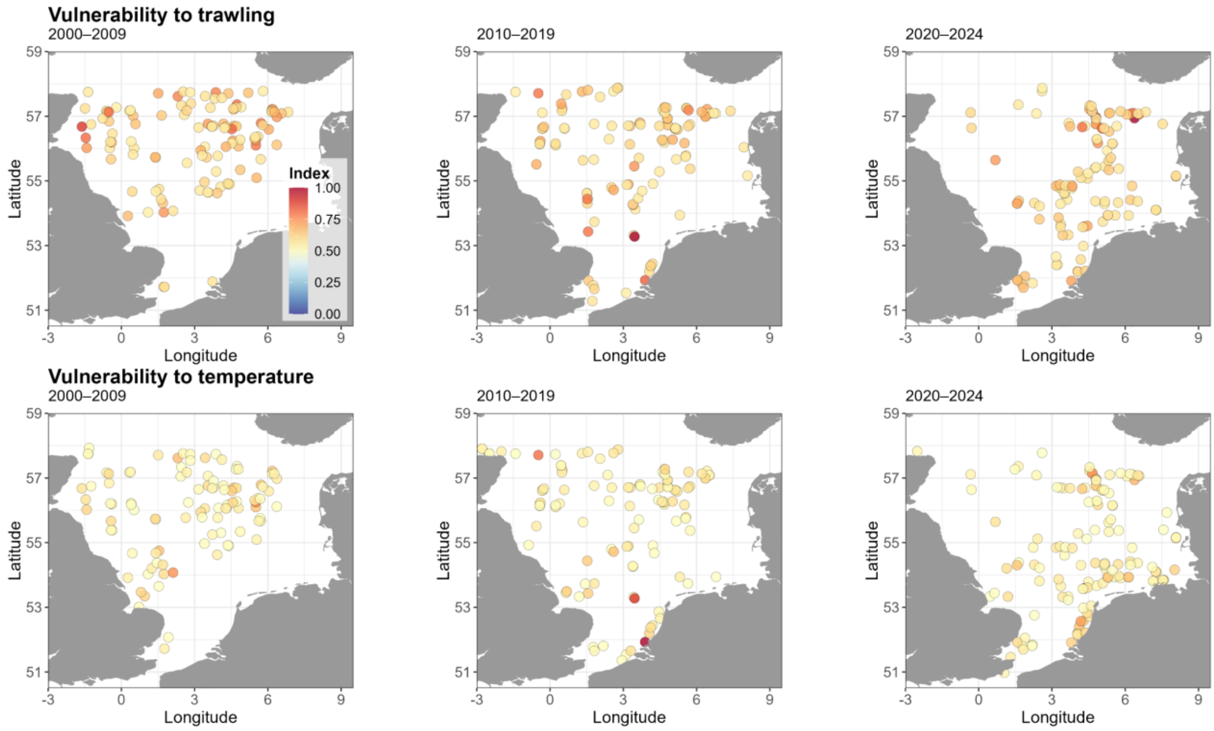

Figure D-1. Upper 5% community weighted vulnerability scores for bottom trawling (Beauchard et al., 2021; top), and temperature (Polo et al., 2024; bottom) in the North Sea from the 2000-2009, 2010-2019, and 2020-2024 eras.

E. Appendix: Benthic habitats in Portuguese continental waters

This appendix provides maps describing trends in bottom temperature and pH trends in the Exclusive Economic Zone (EEZ) of continental Portugal.

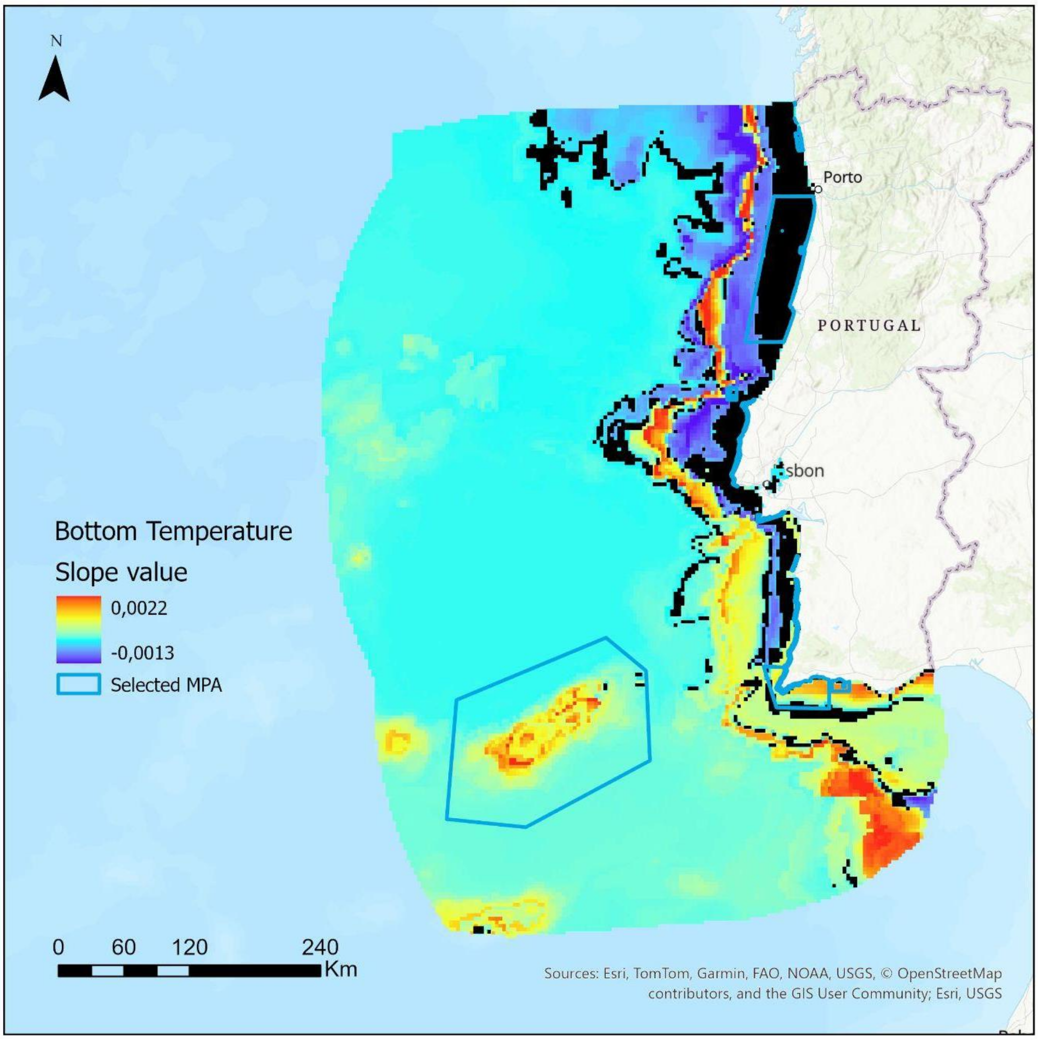

Figure E-1. Bottom temperature trend input data (change over 32-years in °C/decade) in the EEZ of continental Portugal, overlapped with locations of MPAs. Areas in black represent grid cells with non-significant (p>0.1) slopes in the temperature trends.

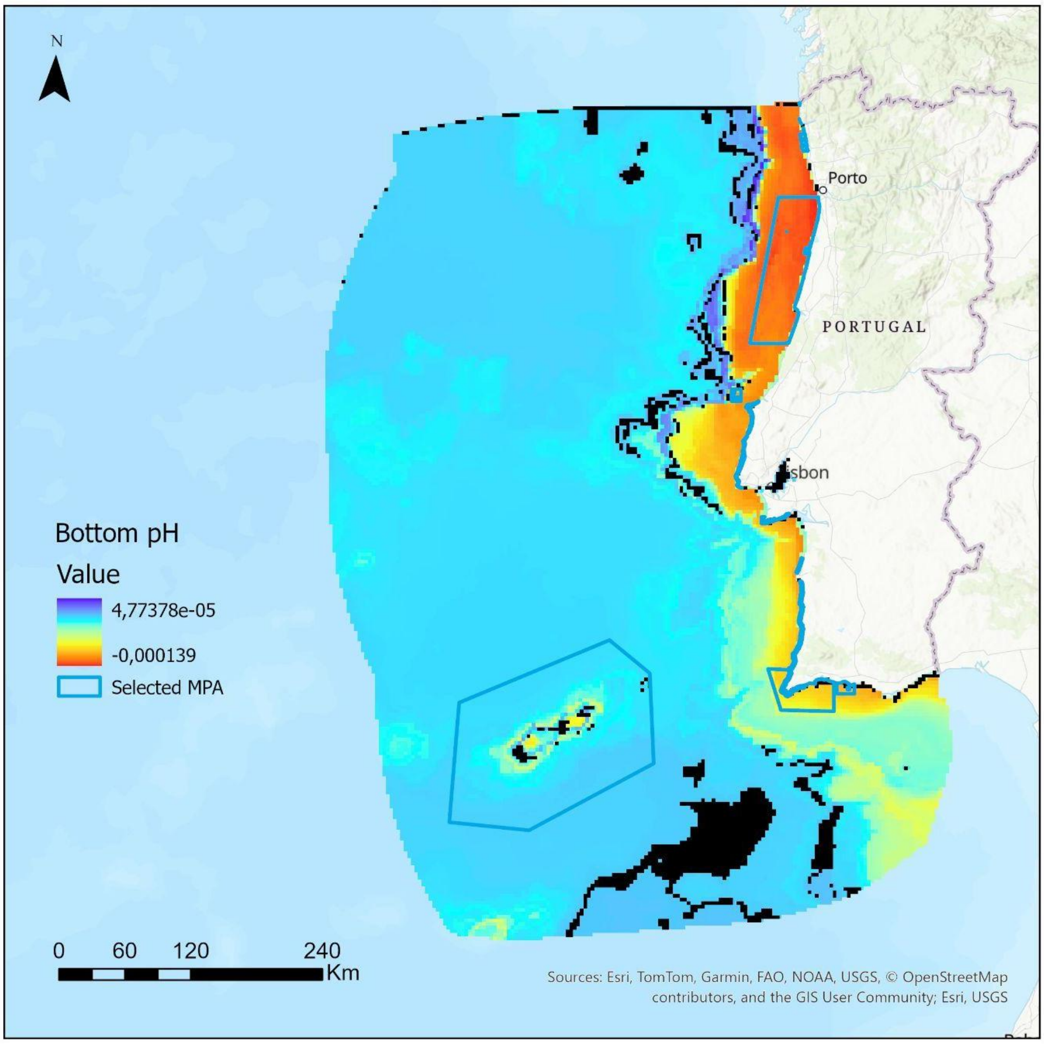

Figure E-2. Bottom pH trend input data (change over 32-years in units/decade) in the EEZ of continental Portugal, overlapped with locations of MPAs. Areas in black represent grid cells with non-significant (p>0.1) slopes in the pH trends.