VALUING THE IMPACT OF A POTENTIAL BAN ON BOTTOM-CONTACT FISHING IN EU MARINE PROTECTED AREAS

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CONTENTS

Executive summary 4
1. INTRODUCTION 7
2. LITERATURE REVIEW 9
  Marine Protected Areas 9
    What is a Marine Protected Area? 9
    Are they all the same? 9
    How much of the world's oceans consists of MPAs? 10
    MPAs in Europe 11
    Associated socio-economic MPA costs and benefits 14
Ecosystem services and natural capital 18
  General overview 18
    MPA context 21
  Bottom-contact fishing 21
    Gear types 21
    Current Activity 25
    Impacts in MPAs 26
3. Ecosystem services cost-benefit analysis model 30
  Estimating bottom-contact fishing in European MPAs 30
  Establishing the impacts of bottom-contact fishing 36
    Valuing the economic impact of ecosystem services 38
  Incorporating costs and displacement 41
  Results 42
4. Case studies 47
  Case study 1: 'Cod box' in the southern Kattegat, Sweden 47
    Background to the Marine proected area 47
    Changes observed as a result of the MPA 48
    Model simulation 51
  Case study 2: Danish Natura 2000 sites in the Kattegat 52
    Context 52
    Data 52
    Applying the model to Kattegat sites 54
5. Conclusion and recommendations 57
Appendix A 60
References 61

EXECUTIVE SUMMARY

Marine Protected Areas (MPAs) are one tool to help combat the global decline of marine foundation species and critical habitats, by allowing seabed habitats and species to recover from human impacts. They represent part of major international conservation commitments to build resilience and turn the tide against anthropogenic overexploitation of the world's oceans. Despite the ability of MPAs to deliver conservation outcomes, without management and enforcement many of Europe's MPAs designated under Natura 2000 have not managed or restricted human activity such as fishing (in particular, fishing using heavy bottom towed gear – trawls and dredges – which have been shown to negatively impact benthic habitats and species). Evidence shows that 59% of the 727 MPAs designated in 2017 still permit trawling and that trawl fishing effort within these sites was 46% higher than outside the MPAs.

The environmental and socio-economic benefits of MPAs (eg through tourism, fisheries catches, and revenue) have been documented in the EU and globally, but trade-offs exist between these objectives in many instances. In previous cost-benefit analyses (CBAs), monetisation typically focused on the cost side rather than the benefit side. Research referenced throughout this report has also shown that costs to industry in impact assessments are frequently overestimated, while long-term benefits to society are chronically undervalued and presented as part of a narrative but not as monetised benefits. This creates an imbalance in decision-making where short- term financial costs are given primacy over long-term societal gains, some of which are far too important to be ignored in decisions that affect them. Despite the prominence of un-monetised benefits provided by healthy, well-functioning marine ecosystems, these are often not captured in financial terms.

It is within this context that the New Economics Foundation (NEF) undertook this research, looking at the public as well as private costs and benefits of MPA designation and the management of bottom towed fishing gear (such as trawls and dredges). This research developed an ecosystem services model using a benefit transfer approach to conduct a wider CBA of the impacts of fully restricting bottom towed fishing in the EU's offshore Natura 2000 sites. The available data meant only marine Special Areas of Conservation (SACs) designated to protect habitats and species of importance at the European scale (sandbanks, reefs, and submarine structures made by leaking gases) were analysed. The following ecosystem services were covered in the model:

- Provisioning: fisheries and food production, aquaculture, and other biological resources.
- Regulating: natural hazard protection, climate regulation, clean water, and sequestration of waste.
- Supporting: nutrient cycling.
- Cultural: tourism and recreation (nature watching, sea angling, scuba diving, and unique natural attractions in coastal areas).

In terms of the lived experience and economic impact of MPAs, it is mainly the provisioning services (ie seafood caught within and outside the MPAs) and cultural (tourism and natural attractions) that have immediate and financial or employment benefits. The others (regulating climate change, cleaning water, and remediating waste or those essential supporting services like nutrient cycling) are not captured in the economic exchange, but without them the ocean would cease to function. It is the value of these services that needs to be understood in ecological and geochemical terms. Our current decision-making model under-acknowledges these scientific issues and prioritises short-term economic and financial impacts: improving decision-making incrementally relies on being able to monetise and value what nature does for society to ensure decision-makers consider this reality. There are ethical issues around valuing nature (what type of value and to whom); there are also technical difficulties (eg uncertainty or lack of data); and there are other societal questions (short- vs long-term costs and benefits and how to weigh up trade-offs that result from these choices). Choosing to present a CBA means monetising benefits to make them visible, but there are other tools and methods that acknowledge some of the limitations and are deliberative; there is a need to push decision-making towards these. This requires radical reform to the worldview, tools, and approaches used to determine how we can best conserve our marine environment for current and future generations.

Using a CBA approach, we were able to estimate the annual net benefit from a potential bottom-contact fishing ban in European MPAs in terms of the estimated cumulative value for each ecosystem service across several stages of a 20-year period following implementation of a ban.

When ecosystem service benefits and costs/displacement are brought together, the net benefit of a mobile bottom-contact fishing gear ban (trawls and dredges) across European MPAs showed that the costs of implementing a ban outweigh the benefits in the terms of annual net impact for the first two years. However, from year three onwards, there is an annual net benefit, which rises sharply up to year 5, as the ecosystem service impacts become increasingly more pronounced. The benefits for many of the ecosystem services increase until year 13, where the habitat reaches a theoretical maximum of annual ecosystem service value. By year 13, the highest annual net impact value is observed, €615 million. From this point, there is a very gradual decrease in annual net ecosystem service, which is the result of the discount rate used (3.5%). This discount rate is commonly used by the UK government’s impact assessments, usually at 3.5%, to account for the time value of money. The rate used is greater than the rate of inflation (2%). Discounting the future therefore impacts the CBA results and biases decisions against the long-term (putting ecosystem recovery and future generations at a disadvantage compared to short-term costs to industry and public finance).

The cumulative ecosystem service benefits, cumulative total costs, and cumulative net benefit across the 20-year period, show that from year 4 to year 5 there is a sharp increase in cumulative net impact, with a more considerable increase from €34 million to €390 million, as the effects of protecting seabed habitats lead to improved ecosystem services as habitats recover from fishing pressure. By year 10, we see a net impact of €2.7 billion, and a value of more than treble that by year 20, €8.5 billion.

For the period between year 13 and year 20, we see an average annual cost benefit ratio for a potential ban on bottom-contact fishing in MPAs of €3.41 returned for every €1 spent, a positive return despite the inclusion of very conservative estimates with regard to potential displacement (75% displaced to seabed habitat 90% quality of protected areas).

	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10
Ecosystem benefit (€ million)	119.7	358.9	724.7	1,224.6	1,867.0	2,546.4	3,251.6	3,983.4	4,742.2	5,528.7
Total costs (€ million)	304.0	603.6	898.9	1,189.9	1,476.7	1,759.3	2,037.9	2,312.3	2,582.9	2,849.5
Net impact (€ million)	**-184.4**	**-244.8**	**-174.3**	**34.7**	**390.3**	**787.0**	**1,213.8**	**1,671.0**	**2,159.4**	**2,679.3**
	Year 11	**Year 12**	**Year 13**	**Year 14**	**Year 15**	**Year 16**	**Year 17**	**Year 18**	**Year 19**	**Year 20**
Ecosystem benefit (€ million)	6,342.5	7,184.2	8,054.4	8,912.0	9,757.2	10,590.1	11,411.0	12,220.0	13,017.2	13,802.9
Total costs (€ million)	3,112.2	3,371.1	3,626.3	3,877.8	4,125.6	4,369.8	4,610.5	4,847.7	5,081.5	5,311.9
Net impact (€ million)	**3,230.3**	**3,813.1**	**4,428.1**	**5,034.3**	**5,631.6**	**6,220.3**	**6,800.5**	**7,372.2**	**7,935.7**	**8,491.0**

To get a clearer sense of the benefits and costs associated with an MPA bottom-contact fishing gear ban and to contextualise the relevance and possible utility of the ecosystem services CBA developed for this research, two case studies where a ban has been implemented or proposed were briefly explored. These case studies describe the context and observed/expected impacts before using available information to estimate impacts adopting the ecosystem services CBA model. This process highlighted some of the utility of using the model at a site level as well as the challenges of capturing the unique context of each protected area.

Insights and recommendations stemming from this research include greater efforts to acknowledge the long-term ecosystem services benefits arising from a ban of mobile bottom-contact fishing in MPAs; an emphasis on the importance of ecosystem service valuation as a tool to improve societal choices while acknowledging their limitations; an understanding of the tensions between short-term costs and long-term benefits and different notions of 'value' (eg between directly observable socioeconomic impacts and non-cashable valuations); and the need for more and better data to build on this research's indicative values.

1. INTRODUCTION

Human activity is seriously impacting the world's oceans. Foundation species and critical habitats are in decline, and with that the ability of these ecosystems to contribute to human health and wellbeing or climate change mitigation. Marine Protected Areas (MPAs) are one tool to help combat this decline. They allow seabed habitats and species to recover from human impacts and are part of major international conservation commitments. Where they are effectively implemented and managed, MPAs have been effective in increasing biodiversity, biomass, and average sizes of fish and shellfish.

Larger MPAs have been more effective than smaller MPAs in terms of achieving conservation outcomes. Despite the ability of MPAs to deliver conservation outcomes, without management and enforcement many of Europe's MPAs designated under Natura 2000 have not managed or restricted human activity such as fishing (eg fishing using bottom towed gear – trawls and dredges – which have been shown to negatively impact benthic habitats and species). This research has found that 59% of the 727 MPAs designated in 2017 still permit trawling and trawl effort within these MPA sites was 46% higher than outside the MPAs[1].

Environmental and socio-economic benefits of MPAs (eg through tourism, fisheries' catches, and revenue) have been documented globally and in the EU, but trade-offs exist between these objectives in many instances. The benefits of MPAs relative to their costs and the distribution of the benefits between different stakeholders is hugely varied and little primary data and research exists that takes a macro perspective. In cost-benefit analyses (CBAs) identified, monetisation was typically focused on the cost side rather than the benefit side. Despite the prominence of un-monetised benefits, these CBAs typically found that the benefits of MPAs exceeded their costs.

It is important to consider the worldview, framing tools and methods for describing and evaluating the benefits of nature conservation by using an ecosystem services valuation approach. There are tensions around monetising the value of nature as well as concerns about the role of financial valuation in the development of markets for nature (often instead of regulation or taxation, for instance). Here, a CBA that explicitly values nature's services is used to demonstrate the long-term value of conserving the seabed in offshore EU MPAs through managing the impacts of bottom towed fishing gear. It is recognised that costs to the fishing industry in the short term are real, in the sense that there may be a loss of income, as are the costs to designation and management. Benefits in terms of future catches, tourism, and any associated jobs created are also real, although they take time to develop. However, the function of healthy oceans and seabed habitats that are intact and able to fulfil functions which we depend on to survive (from supporting fisheries through habitat provision, to sequestering carbon and recycling nutrients) are to some degree of infinite value to society. Despite this reality, demonstrating benefits in economic terms is important given the limited tools used to make decisions at the EU level: impact assessments rely on using CBA and while the field of valuation is developing rapidly in academia, policy has yet to catch up.

It is within this context that the New Economics Foundation (NEF) undertook this research, exploring the costs and benefits of MPA designation and the management of bottom towed fishing gear (such as trawls and dredges). This research is presented in the following structure. First, it provides a literature review of relevant subject areas, covering MPAs, ecosystem services/natural capital, and the impacts of bottom-contact fishing. Next, it describes the ecosystem services CBA model, detailing the methodology before presenting the research results. This section is followed by a brief exploration of case studies of relevant MPAs to contextualise the use of the CBA model. The final section provides concluding remarks and recommendations.

2. LITERATURE REVIEW

Marine Protected Areas

What is a Marine Protected Area?

Human activities impact on the natural world in many ways. In the case of the marine environment, human activity, such as fishing or aggregate extraction for the construction industry, has caused significant damage to marine habitats and species.[2,3,4] Marine and coastal habitats and biodiversity are impacted through over-exploitation,[5] pollution,[6] land-use change, and invasive species, leading to losses in productivity and diversity.[7,8,9] Climate change[10,11] and overfishing[12] are the two most significant challenges to the structure and functioning of marine ecosystems.[13,14,15] Global declines of foundation species (such as seagrasses, corals, kelp, and oysters) have been widely documented and their loss often reduces their beneficial flows (from carbon sequestration[16] to waste detoxification and recreation[17]) to humans, impacting wellbeing.[18]

Marine Protected Areas (MPAs) are one of the policy and management tools which have been introduced globally to help protect the marine environment as a result, by limiting some or all human activity in certain key areas of conservation importance.[19]

MPAs are conservation tools which involve the protective management of natural areas according to specific management objectives. MPAs can be protected for a number of reasons: economic resources (fisheries, minerals, tourism revenue, etc.); biodiversity conservation (particular habitats and species); and species protection for rare or endangered species, or species of national importance. They are designated through creating boundaries, or zones, which allow or restrict certain uses (eg fishing using particular fishing gears, or recreational fishing, or all extractive uses) within that boundary.[20]

MPAs are intended to meet major international commitments, including the Convention on Biological Diversity (CBD, eg article 10)[21] to achieve the Aichi targets[22] (eg Target 11), as well as the UN Sustainable Development Goals (eg SDG 14, life below water).[23]

Are they all the same?

The international literature on MPAs is widespread and diverse.[24] Furthermore, there are multiple types of MPAs, ranging from multi-use MPAs (which have few or no restrictions in place) to highly protected marine reserves or no-take zones (NTZs).

The International Union for Conservation of Nature (IUCN) describes nine different types of MPAs, varying in focus and level of protection.[25] On the more highly protected end are strict nature reserves or marine reserves, where there are strict controls on human activity; wilderness areas, where nature is protected from any modern infrastructure; and national parks, where biodiversity and ecosystems are largely protected, subject to limited infrastructure for visitors and associated codes of conduct.[26] Similarly, specific habitats around natural monuments may be protected or certain species targeted by a Species Management Area (often within a larger MPA), or an area may be declared an NTZ and subject to a ban on all extractive activity. Other classifications such as Protected Seascapes and Protected Areas with Sustainable Use of Natural Resources allowed for more extensive human activities, though an overall aim of conservation remains in place. UNESCO World Heritage Sites, though seldom used in marine areas, are also considered a type of MPA. It is noteworthy that according to the current IUCN categories of MPAs,[27] none of the current management measures set for EU MPAs would mean these sites meet the IUCN definition of an MPA.

A 2014 global analysis of MPA coverage found that many of the largest MPAs with strong protection (eg NTZs) are located in remote areas with limited commercial activity. Although these residual MPAs are valuable, the tendency towards remote locations raises questions over whether MPAs are being designated in locations needed to avert the overall decline in marine biodiversity, or whether the process is being driven by a need to minimise negative socio-economic, and therefore political impacts of MPAs. The difference in ecological benefits of an MPA in a densely populated area, compared with one in a very remote area, also underlines the need to look beyond narrow targets for area covered by MPAs and consider designating MPAs in areas where they offer the highest gains in biodiversity relative to the status quo.[28]

How much of the world's oceans consists of MPAs?

While progress over the last decade has been notable, with some organisations claiming that 5%[29] and others that 7% of the ocean is protected, currently only 3.6% of the ocean is covered by MPAs which are being implemented (ie not 'paper parks'), and within that only 2% are documented as being implemented *strongly* or *fully protected*.[30,31]

A 2014 analysis of a global MPA database showed ~3.5% of the world's seas under national jurisdiction have been designated as MPAs, whereas only 0.5% have been protected within NTZs or marine reserves.[32] A significant proportion of the ~6,000 global MPAs are found within tropical latitudes and have traditionally focussed on protecting coral reef systems. It is very important to differentiate these tropical MPAs from MPAs in temperate regions, such as the EU (where 1.8% of the marine area are MPAs with management plans, whereas 12.4% are designated for protection).[33] In light of the socio-economic and political differences between the two groups, we should not derive our approaches to temperate marine conservation from lessons learned in a tropical context.

A global review of studies of 124 NTZs/marine reserves revealed that fishes, invertebrates, and seaweeds had the following average increases inside the reserves: biomass increased an average of 446%; density increased by an average of 166%; body size of animals increased by an average of 28%; and species diversity increased by an average of 21%.[34] Another global review of evidence found that the effects of protection of temperate areas, in terms of improvements to biomass and density of organisms, are on the same scale or above those found in tropical settings.[35] The authors also documented that the size of the NTZ seemingly did not influence the scale of response to protection for any of the biological variables considered, namely the numerical density or biomass/area of organisms, the individual organism size, or the species richness/area when controlled for tropical/temperate systems. Indeed, their findings suggested that small reserves could result in notable increases in the average size of individuals.[36]

One key example of the differences between temperate and tropical contexts is that tropical coral reef systems, although they represent less than 1% of global oceans[37], have a very high diversity and biomass of marine life (they are effectively oases in a low-diversity environment).[38] For this reason, the need for, benefits of, and impacts of MPAs[39] are much more easily identified and clearly defined in coral reef systems. These impacts[40] are also frequently linked to fishing communities or eco-tourism operations, which are more likely to benefit from changes in management of those areas (in terms of fisheries,[41] biodiversity,[42] and tourism revenue).

In contrast, temperate MPAs[43] do not necessarily have the equivalent fisheries spill-over benefits,[44] community linkages, or traditional tenure systems that characterise many successful tropical coral reef MPA case studies.[45] There are, however, clear benefits of using MPAs and marine reserves (ie NTZs) in a temperate/European context in terms of species and habitat conservation as well as ecosystem recovery, especially in conjunction with wider fisheries management systems.[46,47,48]

MPAs are sometimes presented, or thought of, as a fisheries management tool, mainly due to their historical use in the tropics. On the contrary, MPAs should be viewed as a tool which enables spatial management of all (extractive) industries, rather than a focus on fisheries alone. Wider fisheries management measures (effort or quota restrictions, minimum sizes, and technical gear requirements or licensing) are all more effective tools than MPAs for managing fisheries of finfish, whereas shellfish fisheries for crab, lobster, and scallops (tentatively called ‘spat factories’, and ‘brood stock’ areas) can be developed as evidenced by relatively small closures (eg Isle of Man, South Devon MPAs, South Arran MPA).[49]

MPAs in Europe

A number of EU and UK regulatory drivers are important for improving the condition of the marine environment, including the EU Marine Strategy Framework Directive (MSFD), the EU Water Framework Directive (WFD)[50] and the EU Habitats and Birds Directives.[51] MPAs in Europe aim to meet the international conservation requirements as described in the previous section, but they also contribute to key EU targets and policies, for example achieving the Good Environmental Status (GES) required by the EU Marine Strategy Framework Directive (MSFD).[52]

The primary means at EU level to designate MPAs are as European Marine Sites (EMSs), also known as Natura 2000, comprising Special Protection Areas (SPAs) for birds and Special Areas of Conservation (SACs) for marine habitats and species. Socio-economic concerns are not permitted to influence the designation of these areas. Only scientific considerations are to be taken into account in designation to meet the requirements for the Natura 2000 network[53] (SPAs and SACs) designated under the Birds Directive and the Habitats Directive.[54] These directives provide a cornerstone for European designation of MPAs and are led by biological and ecological criteria rather than by stakeholder consensus.[55] The EU Habitats and Birds Directives recently underwent thorough reviews and have repeatedly been shown to be fit for purpose.[56]

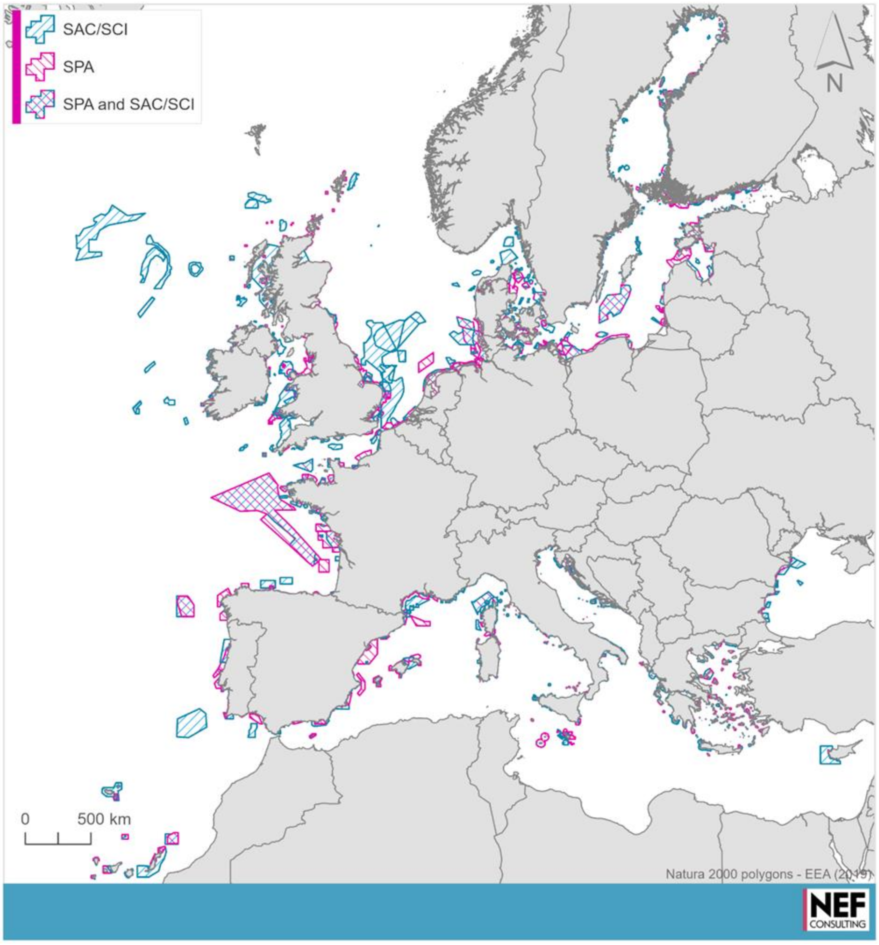


Figure 2.1. Marine Natura 2000 sites.

The primary purpose of MPA networks as per the seventh EU Environment Action Programme is clear: "They must act as sanctuaries, with the primary purpose of safeguarding marine life." In contrast with this aim, the first 20 years of setting up MPA networks have had no significant positive impact on conservation objectives at the European scale, mainly as a result of failing to agree, implement, and enforce management plans that restrict some activities (such as bottom towed fishing). In part this is due to lobbying from industry over the concerns over loss of fishing grounds and short-term costs. There has, however, been evidence of positive impact on species density, size, biomass, and richness from certain European MPAs at the local level, but only in areas where significant restrictions of human activities have occurred.[57]

There are gaps in the coverage of EU-level MPA initiatives, for example certain habitats are omitted from the nine specified by the Habitats Directive and species such as commercially exploited fish are not covered. There have been efforts at national level to fill these gaps in coverage. There are large differences in coverage of coastal versus offshore areas. Not all species or habitats within the boundaries of European MPAs receive equal protection, with those habitats and species that are listed typically receiving higher protection.[58]

Under Aichi Target 11, the EU aimed to have 10% of coastal and marine areas covered by "effectively and equitably managed, ecologically representative and well-connected systems" of MPAs by 2020.[59] This percentage of coverage had been achieved as of the end of 2016, when 10.8% (approx. 625,000 km^2) of the area of Europe's seas had been designated as MPAs, but the network was not fully representative, with lower coverage of MPAs in the central and eastern

Mediterranean seas (2.6% to 5.8%) and Macaronesia (3.3%). More extensive MPA coverage was recorded in the Black Sea, Celtic Sea, Baltic Sea, western Mediterranean, and especially in the Greater North Sea. Coverage improved between 2012 and 2016 in nine out of ten European seas (all except for the Aegean-Levantine Sea). The highest proportional increases during this period were in the Black Sea, Celtic Seas, Bay of Biscay and Iberian Coast, Macaronesia, and the Adriatic Sea, with the proportion of these seas covered by MPAs having effectively tripled in four years.[60] The area of European seas covered by MPAs has increased further since 2016, with notable recent additions including four MPAs designated in Scotland in December 2020, covering 13,596 km^2 (equivalent to 0.2% of the 2016 total).[61]

Coverage of MPAs is higher within near-shore areas (less than 1 nautical mile from shore: 44.3% coverage by area) than in territorial (1–12 nautical miles from shore: 24.2% coverage) and offshore (12 or more nautical miles from shore: 7.2% coverage) waters (Figure 2.2). Nearly all of the European sea areas not covered by an MPA (5.17 million km^2 or 89.2% of all seas) are located in offshore (4.42 million km^2 not covered by MPAs) or territorial waters (649,000 km^2 not covered by MPAs).[62]

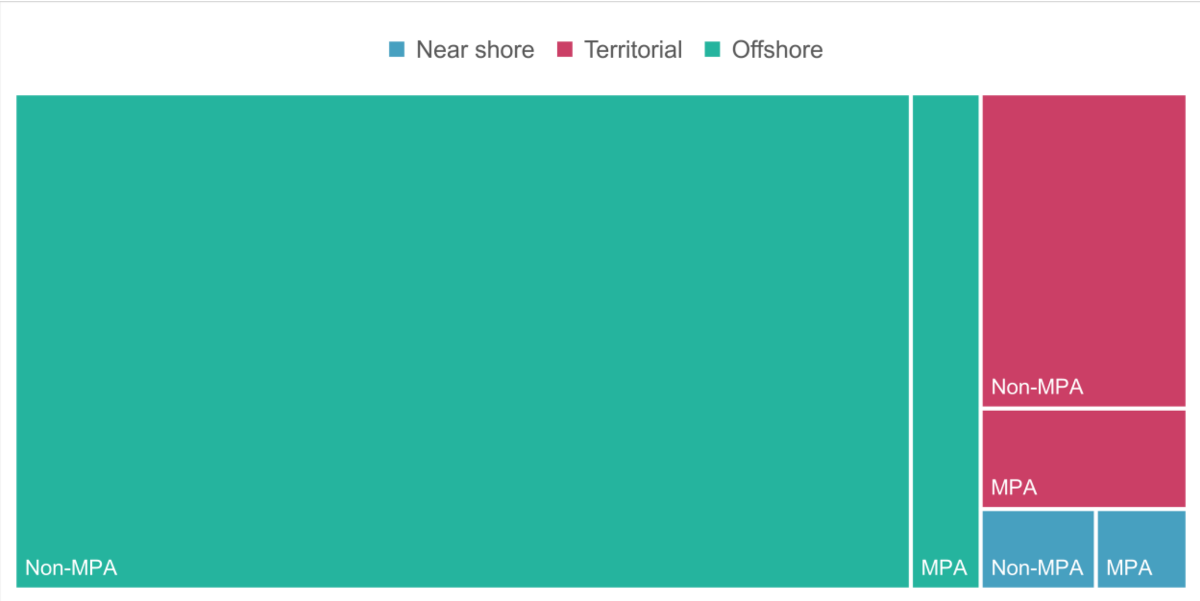


Figure 2.2. Representative area of MPAs and other zones of Europe's near-shore, territorial, and offshore waters as of 2016. [63]

As of the end of 2016, the coverage of different habitats by depth within Europe's MPAs was not sufficient for the MPA network to be representative under the Aichi target.[64] In particular, coverage was below target levels in mud habitats in deeper biological zones and in most habitat types in the deepest (bathyal) zone. Although larger MPAs (with an area of at least 100 km^2) have been shown to be more successful in achieving conservation outcomes than small MPAs,[65] the majority of Europe's current MPAs are very small in size.[66] More than half of Europe's MPAs as of the end of 2016 had an area less than 5 km^2.[67]

Despite recent increases in their spatial extent, some of Europe's MPAs have been found to provide little protection in practice.[68] The European Court of Auditors found that of 21 long-standing Natura 2000 MPAs in Europe 43% had little or no specific restrictions on fishing activities, the MPA network was not ecologically representative, and MPAs may still be used for other harmful industrial activities such as mining, dredging, and industrial discharge.[69] An analysis of 727 European MPAs in 2017 found that trawling was still occurring in 59% and that trawling intensity on trawled sites within MPAs was 46% higher than on trawled sites outside MPAs.[70]

A major factor contributing to ineffective MPAs is a widespread lack of management plans and/or regulatory management measures. As of 2019, although 12.4% of Europe's marine area was covered by MPAs, only 1.8% of total marine area was covered by MPAs with a management plan.[71] Out of 23 marine EU member states, 11 had not reported any MPA management plans by 2019 and a further 8 had plans covering less than 10% of their marine areas.[72] Both the European Environment Agency[73] and the European Court of Auditors[74] recently called for stronger management of Europe's MPAs.

A 2015 review of progress on MPAs during the preceding two decades made a number of recommendations for the future, calling for the implementation of "a modern, holistic approach to MPA design, management and evaluation, if EU MPA networks are to reach their potential in protecting marine biodiversity". It was proposed that an ecosystems-based approach, as introduced in the Common Fisheries Policy (CFP) and the MSFD, would be a more holistic way to design and manage the European MPA network.[75] An ecosystems-based approach considers the whole ecosystem, with the goal of maintaining it in a healthy, productive, and resilient state, as opposed to more traditional targeting specific species, sectors or activities.[76]

Associated socio-economic MPA costs and benefits

Ecological benefits of MPAs

The primary aims of an MPA are to realise ecological benefits for the habitats and species that they protect. When successfully implemented, MPAs lead to more fish and bigger fish. They can protect habitats (which can recover in some cases) and reduce bycatch and harm caused to non-target fauna.[77,78,79] There is also evidence, when managed and implemented effectively, of increased biodiversity within MPAs.[80,81,82]

Research into this ecological effectiveness suggests that it depends on where an MPA is located, how many MPAs there are, how big they are, what kind of protection they entail (blanket fishing prohibition versus prohibition of certain fishing gears), how well managed and enforced they are, as well as the mobility of the relevant fish, shellfish, or other species of conservation interest in and out of the MPA.[83,84]

When assessing the effectiveness of an MPA in terms of ecological benefits, it is important to consider what is happening outside the MPA zone. There can be potential benefit to fishers from increased catch in adjacent areas (known as the spill-over effect) but fishing effort which is displaced from the MPA can also have a negative impact on sustainability outside of the MPA.[85] Recent estimates of bottom trawling footprints showed an over 50% footprint in some European seas and highlighted the environmental impacts of intensively trawled areas.[86]

A 2014 study looking at the conservation performance of 87 MPAs worldwide found that the species richness of large fishes and the biomass of large fishes and sharks were significantly higher in MPAs with no or restricted fishing than in fished sites.[87] On the other hand, no statistically significant increase was identified in total fish biomass; species richness of all fishes; or biomass of groupers, jacks, or damselfishes[88] for MPAs, relative to fished sites. From the data, five characteristics were identified that have contributed to successful conservation in MPAs: having no-take status (no fishing permitted), having good enforcement, being old (>10 years in age), being large (>100 km^2), and being isolated (with deep water or sand forming a barrier between the MPA and fished areas).[89] MPAs with three or more of these features (41% of those sampled) performed well relative to fished areas in terms of total fish biomass and biomass of large fishes and sharks (with performance improving the more features present). MPAs possessing only one or two of the features were not significantly different ecologically from fished areas.[90]

An earlier meta-analysis of the effects of marine protection, encompassing 19 no-take marine reserves from around the world, found that species richness was significantly higher inside marine reserves than outside (a difference of 11%).[91] Overall fish abundance (of both target and non-target species) was found to be higher in marine reserves than outside, but the difference was not statistically significant, and there was considerable variation between different reserves.[92]

The impact of MPAs also depends on the mobility of the fish species targeted for protection. There is evidence that MPAs increase the density, biomass, and individual size of certain fish with limited mobility. For species with high mobility, the impact of the MPA will depend on the proportion of the area that they move within in a typical year that is also protected within an MPA.[93]

Model simulations[94] suggest that MPAs are as effective, or more effective, relative to effort restrictions in their ability to protect vulnerable benthos while minimising deviation from the optimal fisheries yield.

Socio-economic benefits of MPAs

Broadly, MPAs reallocate resources between different users over space and time.[95] This can raise distributional issues, due to the inherently concentrated costs (in the short term and accruing to a few stakeholders) and diffuse benefits (longer term and for a large number of stakeholders) that an MPA entails. Additionally, it may take time for the benefits to manifest themselves (eg spill-over effects on fishers' revenue).

As part of a recent European Commission study into the socio-economic benefits of MPAs, an extensive literature review and a gap analysis were conducted[96] and case studies produced for ten European MPAs.[97] The literature review found that the evidence base for the economic benefits of MPAs was limited and relatively narrow in scope.[98] Evidence tended to focus on the benefits to maritime tourism and artisanal fisheries, with no evidence available for the benefits to other blue economy sectors.[99] Geographically, most of the available evidence was based on the Mediterranean and northeast Atlantic areas.

A number of **economic benefits to commercial fishing** were identified. In some cases,[100] the MPA led to improvements in the quantity and quality of **fish resources** in the area, which in turn led to an **increased catch and revenue** for local fishers. This benefit typically accrued to smaller, artisanal fishers once larger competitors had been banned from fishing in the MPA. Other studies have documented the accrual of benefits to be closely associated with **gear type**.[101] The literature also contained evidence of spill-over gains, where those fishing in waters adjacent to an MPA saw an increased catch, though this depended on the size of the MPA, the mobility of the local fish species, and other factors.[102] Only one study calculated the net change in catch from both the MPA zone and adjacent areas, finding a net gain. The case studies also found economic benefits to fishers from **MPA labels and direct selling initiatives**, either through a price premium received for their catch or improved access to a larger market for their catch.[103] The literature review noted the potential for MPA designation to create displacement of fishing activity both within MPAs (ie more intensive fishing using the still-permitted types of gear) and outside (ie more intensive fishing in an adjacent site).[104]

There were also various economic benefits observed for the **tourism sector**. There was evidence in the literature and case studies that MPAs led to an increase in the **number of visitors** to an area. This increase in demand had different drivers in the case studies, including the improvement in environmental quality created by an MPA, the marketing power that an MPA contributed directly, and the role of MPAs in coordinating efforts across the tourism sector.[105] There was evidence in the literature that this increase in visitor numbers can lead to increased revenues and job opportunities locally. Both sources emphasised the importance of good enforcement of the MPA regulations in facilitating the economic benefits to tourism.

Beyond fisheries and tourism, there were benefits for **other economic sectors** identified in the case studies. In some cases, the MPA led to **direct employment** in its management and monitoring functions, as well as for sectors delivering ecological restoration (eg conservation non-governmental organisations (NGOs), artificial reef design and construction, eco-engineering). **Output** sometimes increased in the aquaculture and bio-economy[106] sectors (albeit this was partly driven by direct financial support provided to operators by the MPA). Some MPAs contributed to the local economy by operating facilities directly (eg hostels, museums, guided tours) or by stimulating additional economic activity locally through research or inward investment.[107]

There was a lack of evidence in the literature on the benefits of MPAs relative to their costs and the distribution of the benefits between different stakeholders. In the limited amount of **CBA** evidence that was identified by the literature review, a large part of the benefits came from non-market outcomes, but monetisation was typically focused on the cost side rather than the benefit side. Despite the prominence of un-monetised benefits, these CBAs typically found that the benefits of MPAs exceeded their costs.[108]

Costs of MPAs

There are costs incurred when designating, implementing, managing, and monitoring an MPA, both for the public (either at the national or European level) and for relevant stakeholders such as the local fishing sector.

MPAs require public funding to be designated and implemented. They can in some instances deliver tangible economic benefits, whether through increased fisheries biomass, increased tourism revenue, or wider ecosystem service provision, but this is limited to those MPAs that are accessible to the public (divers, fishers, or anglers for example). The ongoing management of an MPA usually requires costs such as communications and liaison work, public awareness efforts, and signage to be incurred by the public.[109].

There are public sector costs associated with **enforcement** of an MPA. Co-management can lower the costs and increase the compliance with management measures, but a clear risk-based enforcement strategy, which is well communicated is essential if management is to lead to improved outcomes as a result of designating an MPA.

Monitoring costs (to the public) include collecting essential baseline data. There is also a need for annual or multiannual monitoring plans that are consistent and geared towards monitoring the condition of the site, as well as the extent of habitat or species recovery as a result of an NTZ or management that impacts marine industries (such as energy, fishing, or aquaculture). There is potential for citizen science initiatives to be used to involve local people and reduce monitoring costs.[110]

There is also likely to be an **opportunity cost** to certain economic sectors due to the designation of an MPA. Economic growth and extractive industries are frequently given primacy in political decisions, but the interests of the marine environment and that of future generations need to be considered in the designation and management process. There is wider value in having a healthy marine ecosystem and ignoring this in the interest of short-term economic return risks further degrading our marine commons. The most significant costs to the private sector are likely to fall on the fishing industry via partial loss of income for some fishers, potential increased steaming distance to access other fishing grounds, and gear conflict (as they have to move onto other grounds where they come into contact with other vessels/fleets).[111]

Ecosystem services and natural capital

General overview

Natural capital refers to the stock of renewable/non-renewable resources, which combine to yield flows of benefits to humans.[112] The elements of nature that directly or indirectly produce benefits for people, which can be material or non-marketed and include myriad examples – ecosystems, biodiversity/species, climate regulation, fresh water, erosion control, land, minerals, the air and oceans, as well as natural processes and functions – are all covered by the concept of natural capital.[113,114]

The functions and products from nature that can be turned into human benefits with varying degrees of human input are referred to as ecosystem services.[115] These can be seen as the beneficial *flows* that stem from the natural capital *stocks* and supply a public need covering economic, social, environmental, cultural, or spiritual benefits. This utilitarian concept was developed with the aspiration of becoming the political lever to reduce biodiversity and habitat loss, making the benefits we derive from nature visible in economic decision-making.[116] These beneficial flows are dynamic and interact with each other. They represent the benefits people derive (including economic goods and services), directly or indirectly, from ecosystem functions, which sustain and fulfil human life.[117] Therefore, they evolve in time and space, as do the ecological processes and resources. The wider processes are value-neutral, but the goods and services are valued in a societal sense even if they are not mediated through markets.[118,119] How the value of these benefits is described can be qualitative or quantitative (including monetary).[120]

Crucially, ecosystem services influence human wellbeing, among many others including secure and adequate livelihoods, food, shelter, clothing, health, a healthy physical environment, good social relations, security, and protection against natural and human induced disasters.[121] Humans are part of global ecosystems that drive ecosystem change both directly and indirectly, impacting human wellbeing. The impact of economic, cultural, and social factors influences people, who in turn shape ecosystems, together with natural forces.[122] The effect of ecosystems on our wellbeing has been evident in the context of the COVID-19 pandemic. Contact with nature was found to have reduced the likelihood of reporting symptoms of depression or anxiety among people coping with the lockdowns of spring 2020 across nine countries.[123]

The links between these flows and wellbeing were described by the Millennium Ecosystem Assessment (MEA, 2001–2005)[124,125] which first drew global attention to the concept[126] and has helped conceptualise these interactions between ecosystems and people.[127] The MEA examined the consequences of changing ecosystems for human wellbeing involving 1,300 global experts to provide the scientific basis for action to improve the conservation and the sustainable use of ecosystems – including the provision of clean water, food, timber, fuel, forest products, flood control, and other natural resources.[128]

The main findings of the MEA were:

- Between 1950 and 2000, ecosystems were impacted and changed faster than ever before in human history, largely as a result of human activity.
- Sixty percent of the 24 ecosystem services examined were being degraded. Irreversible biodiversity loss has been one major consequence.
- Any benefits derived from exploiting nature came at the cost of significant degradation of ecosystem services, resulting in higher risks of irreversible change and increasing poverty.
- The long-term impacts for future generations were shown to be a severely depleted resource/natural capital base.
- Non-linear changes such as new diseases, water quality decline, fish stock collapse, and coastal dead zones (waters with low oxygen content) were identified, together with regional climate shifts.
- Significant policy changes were urgently needed.

The MEA raised the question of how changes in ecosystems impact human wellbeing and how to communicate to decision-makers, as the economic value of non-marketed services was almost non-existent and costs of the depletion were not tracked in national economic accounts.[129] These still do not feature in the UK's Gross Domestic Product (GDP), although the Natural Capital Committee advice to government on the 25-Year Environment Plan[130] makes recommendations of using a natural capital accounting approach to the environment.

The accepted high-level classification of 'functional grouping' divides ecosystem services into four categories:

1. Provisioning services (products obtained from ecosystems.

2. Regulating services (those benefits obtained from the regulation of ecosystem processes).

3. ***Cultural services*** (any nonphysical benefits that humans obtain from ecosystems).

4. ***Supporting services*** (those necessary for the production of all other ecosystem services)[131,132].

Figure 2.3 illustrates the linkages between ecosystem services provision and human wellbeing, according to the MEA.

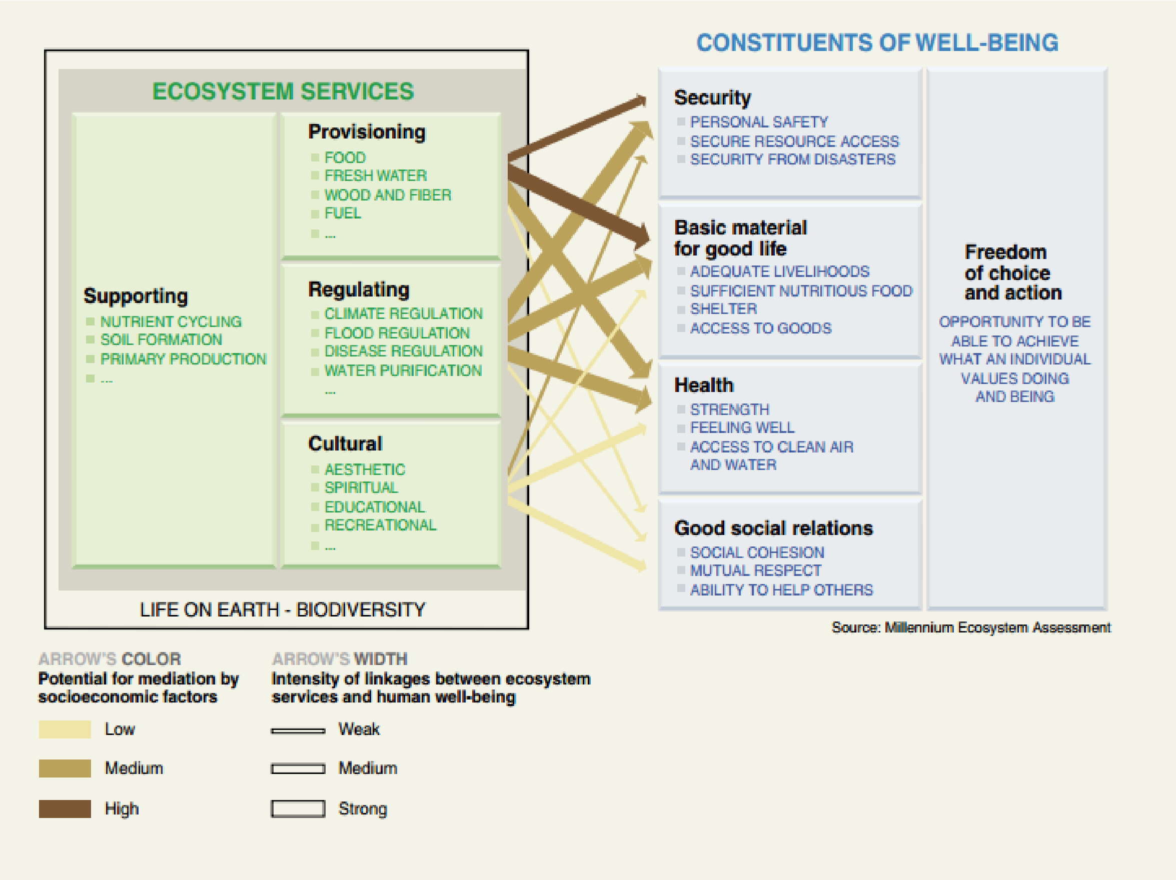


Figure 2.3. From ecosystem services to human wellbeing.[133]

The economic valuation of ecosystem services involves expressing a value for these services in monetary terms, to bring hidden costs and benefits to view – and to the attention of decision-makers (via CBA in the form of impact assessments).[134] All investment decisions and interventions involve trade-offs and the valuation of ecosystem services is a step towards more inclusive decision-making, by making these trade-offs explicit and comparable in monetary terms. A full valuation of the wide array of services provided by marine ecosystems would enable decision-makers to better understand and compare trade-offs.[135]

Valuation should support decision-making regarding policy-making, regulation, and management.[136] The valuation of ecosystem services is considered widely to be a tool to improve societal choices through presenting the costs of ecosystem degradation and the benefits of restoration. Understanding the importance of action (or inaction) is a requirement for improved management.

Valuations have been described in three categories: *decisive*, *technical,* and *informative*. While valuation is considered an important contribution to decision-making, distributional aspects (who wins and who loses as a result of decisions) are often absent. These distributional impacts may also be unclear or change over time, but need to be presented, discussed, and acknowledged as part of the process.[137,138]

Natural capital and ecosystem services are concepts used to communicate society's dependence on nature and to develop economic theory and practice to capture myriad externalities (causing environmental degradation), which arise from human activity.[139,140] This approach has its challenges,[141,142] as it remains a broad concept, with few applied examples of best practice. In truth, many of nature's benefits cannot be valued in monetary terms.[143]

MPA context

As part of the UK's National Ecosystem Assessment Follow-on in 2014,[144] a list of ecosystem services provided by marine ecosystems was established, drawing on the available data and literature on the services provided and their contributing factors.[145] The key ecosystem services to consider, which can also be generalised in the context of evaluation of MPAs, were outlined as follows:

Provisioning

- Fish and shellfish (capture fisheries) driven by primary productivity in the water column, and secondary productivity of seabed habitats, spawning and nursery grounds for key fish species, and crustacean and mollusc production.

Fish and shellfish (aquaculture).

- Other biological resources including the production of ornamental materials and fertiliser, and output from biotechnology and biofuels.

Regulating

- Natural hazard protection as provided by offshore sand banks, seagrass beds, areas of saltmarsh, shingle, and mudflats, all of which aid sea defence
- Climate regulation driven by the North Sea carbon pump, seabed depth, and carbon sequestration in saltmarsh and seagrass beds.
- Clean water and sediments driven by the breakdown and sequestration of waste in the coastal water column, saltmarsh, coastal coarse sediment and rock habitats, and coastal mud habitats and muddy sands.

Supporting

- Nutrient cycling via the coastal water column, secondary producers and vegetation in certain habitats, and soft sediments.

Cultural

- Tourism and recreation including activities such as nature watching, sea angling, scuba diving, and unique natural attractions (eg picturesque coastline).

Bottom-contact fishing

Gear types

Trawl and dredge gears (bottom towed fishing gear) used in Europe

Fish and shellfish species caught in European fisheries are caught using several different fishing techniques including pots, nets, hooks, and trawls. For certain species they can be caught by multiple gears, which have differing impacts on the target and non-target species and the wider marine ecosystem.

By way of illustration, Figure 2.4 shows the range of fishing gears used by the UK fleet. The main distinction is between active and passive gears. Active gears include trawls and dredges which are towed, whereas passive gears are those which are fixed or drift (these include fixed nets, drift nets, pots, and traps as well as hook and line). Active and passive gears vary primarily in their selectivity, survivability of non-target catches, fuel use, and impacts on the seabed, with active gears having a higher environmental impact. Pots and traps are typically employed in shellfish fisheries (eg crab, lobster, whelk) as are dredges (eg scallops). On the other hand, nets, hook and line, and trawls are used in finfish fisheries, both demersal (seabed, eg cod or sole) and pelagic (water column, eg mackerel or herring).

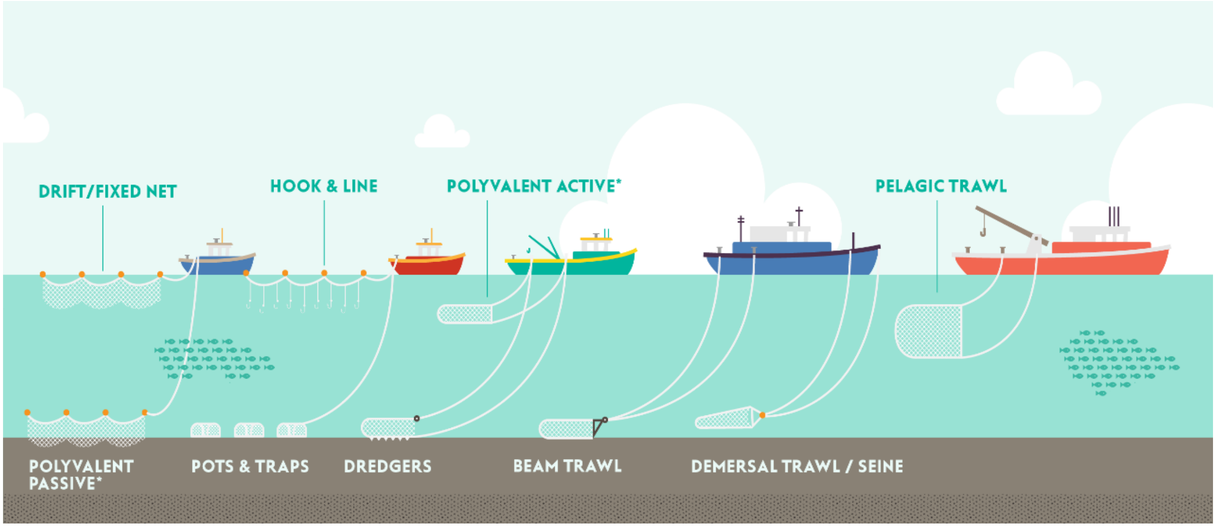


Figure 2.4. UK fishing gears in the Data Collection Framework classification.[146]

Many species can be caught by either active or passive gears. For example, *Nephrops* (langoustine) are caught by trawls as well as creels (pots). The rates of bycatches and seabed impacts are very different for these two fisheries.[147] Similarly, trawls and nets can both catch cod in the North Sea, with the two gears having different environmental impacts, from fuel use to bycatch.[148]

There are further distinctions between similar gear types in terms of their footprint and the mesh sizes and panels used, which in turn can determine their impact. Impact can also be affected by the fishery (mixed or single species) and the location (grounds/habitat) where the fishing takes place how the gear is towed, the weight of the gear, etc.[149] Figure 2.5 illustrates how towed demersal mobile gear (trawls, seines, and dredges) are employed in the water, thus showing the footprint (ie area of contact with the seabed) of the different gears.[150]

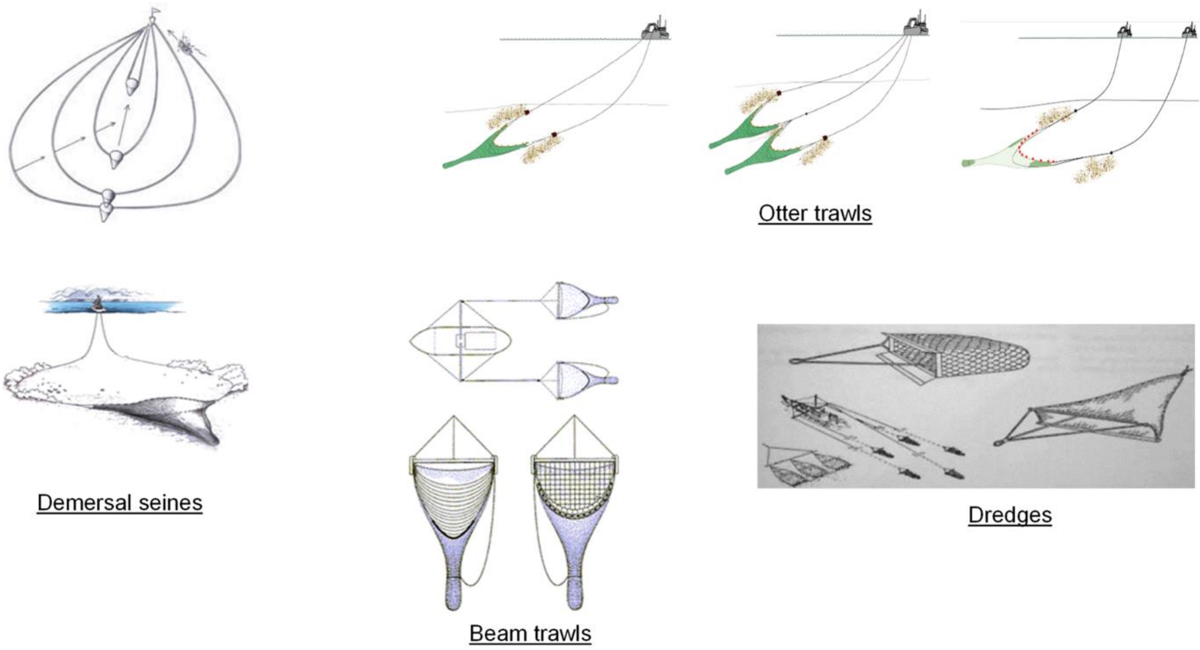


Figure 2.5. Towing principles of the four main high-impact demersal gear groups identified: demersal seines (left), otter trawls (top right), dredges (bottom right), and beam trawls (bottom centre).[151]

Different types of **trawling** gear vary in terms of their footprint (area covered per hour) and their penetration into the sediment. The most commonly used trawling gear types include otter trawls, demersal seining, beam trawls, and dredging.[152] Demersal otter trawls consist of conical nets that are held open and dragged along the sea floor. Demersal seining uses a net that is gradually drawn closed by retractable ropes. The seine may be put out from an anchor point (Danish or anchored seining) or towed behind a moving vessel as the ropes are winched in (Scottish seining or fly shooting). Beam trawling and dredging are used to target species found on the seabed or partially buried in the sediment, meaning that they use gear designed to penetrate and disturb the sediment (eg via tickler chains, teeth, or shearing edges). The first trawl pass is likely the most detrimental to the seafloor in absolute terms although repeated trawling limits the ability of benthic habitats to recover.[153]

An industry survey (collecting 1,132 responses from 13 European countries) recorded 14 distinct groupings based on gear type and target species (Table 2.1).[154] These groupings varied in the proportion of impact that occurred at surface and subsurface (2 cm or more into the seafloor sediment) levels. Dredging and beam trawling (sole and plaice) were found to have 100% of their impact at subsurface level, whereas seining and some otter trawling (sprat, sandeel, cod, plaice, pout, and benthic fish) tended to have over 90% of their impact at surface level.[155]

Table 2.1. Fourteen groupings of gear and target species identified in a European industry survey[156]

Gear	Typical target species	Typical ground gear informed in questionnaire
Otter trawl	Sprat or sandeel	Cookie
Otter trawl	Bentho-pelagic fish	Cookie, Roller
Otter trawl	*Nephrops* or shrimp	Bobbin, Roller, Chain
Otter trawl	*Nephrops* and mixed demersal	Bobbin, Roller
Otter trawl	Shrimp	Chain
Otter trawl	Cod or plaice or Norway pout	Bobbin, Cookie
Otter trawl	Benthic fish	Rockhopper, Bobbin
Otter trawl	Individual species not informed	Bobbin, Roller, Cookie
Beam trawl	Brown shrimp	Bobbin
Beam trawl	Sole and plaice	Chain
Beam trawl	Thomas' Rapa whelk	Chain
Dredge	Scallops, mussels	Sheering edge
Danish seine	Plaice, cod	Cookie
Scottish seine	Cod, haddock, flatfish	Roller, Chain

Bottom towed fishing gear also includes **dredges**: weighted rectangular bags, which are dragged across the seabed to catch shellfish (most commonly scallops, mussels, or oysters).[157] The upper part of this bag is mesh, while the underside is metal (the ‘chain belly’) to withstand constant abrasion as it is dragged across the seabed. The individual dredges (metal ‘teeth’ to rake the surface and detach the target species) are then attached to a dredge beam (a heavy steel tube with wheels at either end). A steel wire bridle and towing warp are also connected. These beams are then balanced either side of the vessel in pairs, and a typical scallop dredger in the UK may tow up to 16 dredges (8 per side) when fishing offshore, although 4–6 per side is the norm for inshore vessels.[158] Scallop fisheries are non-quota and have no EU long-term management plan under the CFP. There is no regional or national UK-wide management plan for the species, but management via technical regulations and effort regulations exist. In Scotland there are area-specific Fishery or Marine Conservation Orders which exclude scallop dredging in Lamlash Bay and the South Arran MPA for example, while in England Inshore Fisheries and Conservation Authority (IFCA) byelaws set scallop dredge vessel size limits, working hours, dredge limits, and permit systems.[159,160]

Current Activity

Spatial patterns and extent of trawling in Europe

Data from logbooks and vessel monitoring systems[161] give a relatively accurate picture of the extent of trawling activity in Europe from 2010 to 2012,[162] while more recent work published in 2018 has updated those findings using similar approaches.[163] Some trawling was found to occur across large parts of the European continental shelf and its slope. At relatively shallow depths (0 to 200 m), trawling affected 28%–99% of each management area in the northeast Atlantic and 57%–86% in the Mediterranean management areas using data up to 2015.[164,165] The most heavily trawled areas relative to landings were found on the Portuguese and Italian coasts. The trawling footprint did not vary much across different seabed habitats.[166]

This trawling footprint is not evenly spread across European waters, however. For all the management areas, habitat types, and gear types assessed, there existed some areas of intensive trawling and other parts subject to lower intensity effort.[167] Data for various management areas around Europe shows that 90% of the fishing effort in each place is concentrated in zones covering between 17% and 63% of that management area.[168] These hotspots are thought to relate to depth gradients (eg along the northeast Atlantic continental shelf and the Greek and Italian coasts), the location of mud patches (eg in the Celtic and Irish Seas) and transient patterns in the distribution of target fish species (eg in the case of beam trawling for flatfish).[169] Grid cells in which trawling frequency at shallow depths was less than once per year were estimated to cover between 1% and 47% of the management areas assessed (ranging between 10% and 30% for most of the areas).[170] The concentration of trawling in relatively small proportions of the seabed underlines the potential for well-targeted restrictions to have a significant effect on the industry's negative impacts. On the other hand, the concentration of trawling in certain areas suggests that protecting those areas would come with a significant short-term political cost in terms of lost revenue for fisheries.

Seabed integrity, a measure of the proportion of taxa within a habitat that have been impacted by trawling (based on their lifespan relative to the time between each trawl pass) was found to vary substantially.[171] The lowest seabed integrity values were found in the Adriatic and Iberian Portuguese management areas, and in general most grid cells either had high (less than one-sixth of taxa impacted) or low (more than five-sixths of taxa impacted) levels of seabed integrity. This was driven by trawling intensity: only those grid cells that are trawled less than once every ten years are expected to exhibit high seabed integrity (this is due to an estimated 17% of biomass being made up of taxa with a life span of more than 10 years).[172]

A similar mapping of EU fishing activity for 2014/2015, based on Automatic Identification System (AIS) vessel tracking data, found that a high intensity of trawling was occurring in nearly all of the continental shelf area in Mediterranean countries.[173] The extent and intensity of trawling in the Adriatic was found to be especially severe relative to the rest of the EU, while the intensity of trawling in the northeast Atlantic was more variable, with coastal areas having higher intensity than offshore areas.[174]

AIS vessel tracking data from the Mediterranean for the years 2012–2014[175] show that much of that sea's bottom trawling activity is concentrated in northern and central areas, including the northern Aegean Sea and the Spanish, French, and Italian coasts. Approximately threequarters of fishing activity in the years 2012–2014 occurred at depths of less than 200 m, with the remainder occurring in depths between 200 m and 800 m.[176]

Impacts in MPAs

Impact of different types of bottom towed fishing gear

Trawling

There are inherent challenges in measuring the effects of bottom trawling.[177] It can be difficult to isolate the causative effect of trawling when environmental conditions such as currents, temperatures, and storms[178] are constantly changing. The field of research has also suffered from a lack of data on species abundance and composition prior to when trawling began, as well as practical issues when studying offshore habitats. Nonetheless, the following **direct effects**[179] have been regularly identified due to trawling.

Trawling **directly leads to mortality of** marine organisms, including its target catch and other species (eg cetaceans)[180] that are caught incidentally. The latter category, **bycatch** of undersized or non-target fish species, occurs under all types of bottom trawl gear and can be significant in extent relative to the targeted catch.[181] A global survey of bycatch and discards in the 1990s and early 2000s found that 8% of the weight of global fishing catch was discarded, but that this discard ratio was higher for shrimp trawling (62.3%), dredging (28.3%), and demersal finfish trawling (9.6%), while midwater trawling (3.4%) had a lower-than-average discard ratio.[182] Mortality from trawling also includes incidental killing of other species by the fishing gear and increasing the vulnerability of other organisms to predators when the seafloor sediment is disturbed.

Trawling can similarly cause an increase in **food availability**, as fish and other fauna that experience increased mortality can serve as food for scavengers. The impact of trawling across the food web can vary. In some cases, species less exposed to mortality from trawling face less competition for their food supply from other species who are more exposed, and trawling may even stimulate increases available feed if those less exposed species serve as a food source for fish. In other cases, where a key food source for fish is also a species more likely to be killed by trawling, there can be a negative effect on fish numbers.[183]

By disturbing and destroying fauna on the seafloor, trawling leads to **loss of habitat**. Because certain species are more or less vulnerable to and able to recover from the impacts of trawling, it can reduce **habitat complexity**. Trawling gear has driven a decline in several endangered and threatened species in the USA, including the smalltooth sawfish[184] as well as some cod stocks and marine habitats in EU waters.[185,186] There is also extensive evidence that deep-sea trawling damages and removes habitat-forming species such as corals and sponges.[187] Repeated trawling can cause changes in **community structure** and can cause a reduction in **productivity** of the habitat.

The impact of trawling varies depending on gear type, timing, and extent of historic trawling, pre-fished composition of the benthos and the habitat, and between species. There is also evidence of spatial variation and seasonal variation in the impact of trawling.[188]

The impact of trawling is usually more severe for long-lived benthic organisms, which take longer to recover afterwards[189] and reductions in the biomass of these organisms have been recorded in areas subjected to frequent trawling, where there has been a shift towards shorter-lived species.[190]

There is also evidence of a number of **indirect effects** of trawling (occurring as a consequence of the direct effects).

Trawling can have an effect on the behaviour and biomass of the **predators, prey, and competitors** of the target species that is caught, with resulting changes to community structure.[191]

Trawling causes changes in the **flow of materials and energy through ecosystems** (eg impact on nutrient cycling), affecting the balance of primary production and consumption and secondary production in ecosystems. Bottom trawling near Oslo has been found to have reduced the density of four species[192] important to the nitrogen cycle of the seabed.[193] These fauna greatly speed the process of breaking down and remineralising organic matter on the sea floor (increasing the pace of this by two to ten times), which then feeds new algae. The role played by these macrofauna in bioturbation is “one of the most important functions that regulate process rates and pathways during organic matter mineralization in marine environments”[194]. Two species in particular were most important to the flux rates of nutrients in a study in the Kattegat in 2008: the burrowing shrimp that trawlers were targeting (whose population fell by 65% in the area) and a species of sea urchin. Extrapolating this reduction to other trawled areas nearby suggests that trawling could be responsible for a 39% reduction in the release of silicate and a 63% reduction in the uptake of nitrogen.[195] Furthermore the impact of bottom trawling on water turbidity through re-suspension of sediments has been documented.[196] This suspension of sediments has been found to reduce the survival rate of eggs and larvae of various marine organisms, including corals,[197] cod,[198] and herring.[199]

A further indirect impact of trawling is its effect on the **storage of carbon**, which has major implications for climate change. Offshore shelf sediments constitute a significant global carbon store that is compromised when bottom trawling re-suspends sedimentary organic carbon. The subsequent process of remineralisation leads to changes in the depth and rate of organic carbon burial. A modelling study looking at this process under a scenario of increased trawling and climate change over the coming 25 years estimated that costs of up to $12.5 billion could be incurred via release of this ‘blue carbon’.[200] Although some of this damage is driven by climate change more broadly, there is potential to avoid a portion of the carbon release through stricter regulation of bottom trawling.

The effects of natural and manmade disturbances on seabed habitats are subject to non-linearity and feedback loops, meaning that individual disturbances can have a major impact on biodiversity and ecosystem function depending on their timing and context. The impact of these disturbances is sometimes characterised by tipping points: thresholds beyond which ecosystems can rapidly collapse.[201] This was observed in the eastern Scotian shelf in the northwest Atlantic, where the ecosystem has failed to recover to its previous, cod-dominated state, despite fishing having ceased in 1993. This case of overfishing is an example of the indirect impact of fisheries on biodiversity. There followed a collapse in cod populations, leading to increases in abundance of smaller fish species and predatory crustaceans, which in turn is likely to have knock-on effects for their prey (eg sedentary worms and bivalves). In this way, fishing has caused significant changes in the functional characteristics and biodiversity of the system.[202] In some instances of overfishing, a perverse outcome of the reduction in biodiversity has been to favour species that are more economically valuable. For example, in the Firth of Clyde in Scotland, a reduction in bottom-fish populations due to overfishing has favoured the more economically valuable catch of langoustines and scallops.[203]

Research on a larger geographical scale suggests a significant impact from bottom trawling. One estimate of the current state of benthic biomass in the Baltic Sea using a model of trawling impact and hypoxia (a separate issue caused by global warming and pollution from agriculture and sewage), which is validated using empirical data, suggests that biomass is reduced by at least half in 14% of the Baltic Sea, with biomass reductions of between 10% and 50% in a further 8% of the area.[204] The parts of the Baltic Sea that are subject to bottom trawling and in which biomass is reduced by at least 10%, are estimated to account for 9% of the total area of that sea.[205]

Dredging

Dredging is known to have significant impacts on the seabed.[206,207] Scallop dredges are considered to be the most damaging to non-target benthic communities and seafloor habitats, especially for slow-growing organisms like maerl or *saballari*a that form biogenic reefs. The ecology of soft sediments can also be severely impacted by scallop dredging in comparison to the lower vulnerability of species adapted to high-energy environments. Rocky reefs too have been shown to suffer damage from scallop dredging, with incremental damage based on fishing intensity. Benthic epifauna are the most vulnerable to scallop dredging, reducing the capacity to support biodiversity, negatively impacting recruitment species, including scallops. Mobile species can also be affected by dredging as considerable levels of by-catch is caught.[208,209] These impacts have also been shown to cause considerable conflict between scallop fisheries and fisheries for other species.[210] Dredging has been shown to be incompatible with the conservation objectives of EMSs in the UK where it damaged eelgrass (*Zostrea*) beds in the Solent.[211] Recent research has shown that adapting the design and weight of dredges can drastically reduce the negative impacts on the seabed and bycatch, as well as reduce fuel use.[212]

Bottom towed gears and seabed impacts

A global meta-analysis of 122 experimental studies on the effects of bottom fishing found that on average, one fishing gear pass reduced the abundance of benthic invertebrates by 26% and reduced species richness by 19%.[213] These ecological metrics were predicted to take more than three years to recover after bottom fishing occurred. The negative impact of trawling, in terms of reductions in benthic community abundance and species richness, was significantly greater for gears that penetrate deeper into the seafloor sediment. There was significant variation between the time taken for species richness to recover following fishing from different gear types, from a few days to more than three years.[214]

A similar analysis of 46 comparative and experimental studies[215] found a very close correlation between the penetration depth of a type of trawling gear and the percentage of biota that were removed per trawl pass. Otter trawls and beam trawls had a lower penetration depth into the sediment (2.4–2.7 cm on average) and removed 6% and 14% of organisms per trawl pass, respectively. On the other hand, towed dredges (penetration of 5.5 cm and a removal rate of 20% of biota) and especially hydraulic dredges (penetrating 16.1 cm and removing 41% of biota) had a more severe impact on macrofaunal community biomass per trawl pass. Higher rates of trawling frequency (more passes per annum) corresponded to greater reductions in community biomass and numbers.[216,217]

3. ECOSYSTEM SERVICES COST-BENEFIT ANALYSIS MODEL

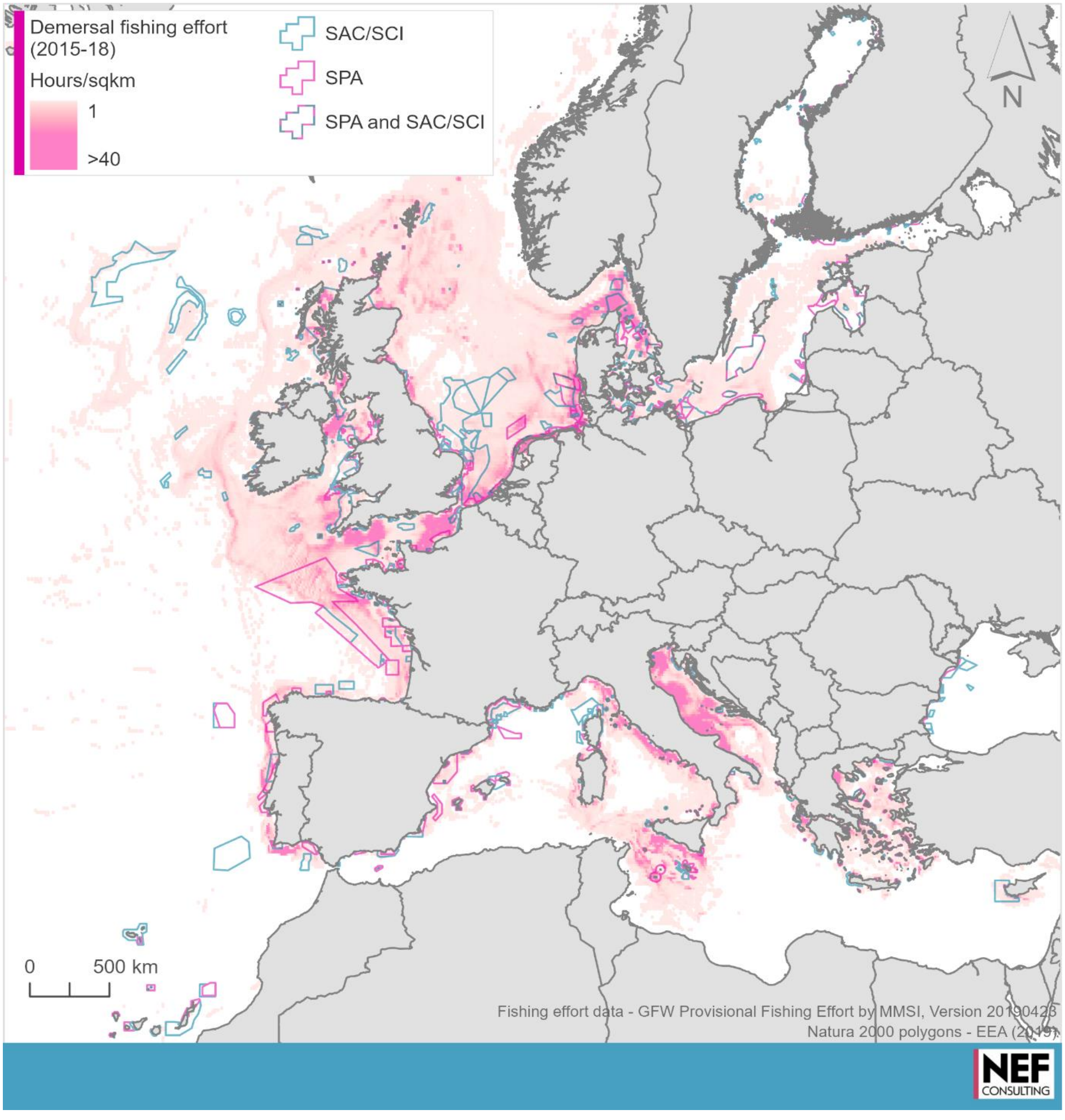
The literature review has outlined the numerous ways in which a ban of bottom-contact fishing can impact socioeconomic outcomes as well as ecosystem services (both positively and negatively). In this section, we present the model developed to estimate the value, in monetary terms, of a potential bottom-contact fishing ban across Europe's Marine Protected Areas (MPAs). Like most ecosystem service valuation studies, the model we have developed takes a benefit transfer approach, defined by researchers[218] as "the use of research results from pre-existing primary studies at one or more sites (often called study sites) to predict welfare estimates, such as willingness to pay (WTP), for other, typically unstudied sites (often called policy sites)".

As with all benefit transfer studies, there are limitations around measurement and generalisation error.[219] Acknowledging these limitations, we transparently present all our sources and modelling assumptions for both impact and financial proxies. Given the diversity of habitats/ecosystems, the geographical range of these MPAs, and the different fishing practices, species, and gear types, it is not possible for the model to capture the complexity of impact (and subsequent value) within this project's scope. Instead, the model estimates indicative values, both the type and extent, using the best scientific and economic data available. This section outlines the current status of the cost-benefit analysis (CBA) and how it was developed, detailing its theoretical underpinnings, the data used, and the assumptions made.

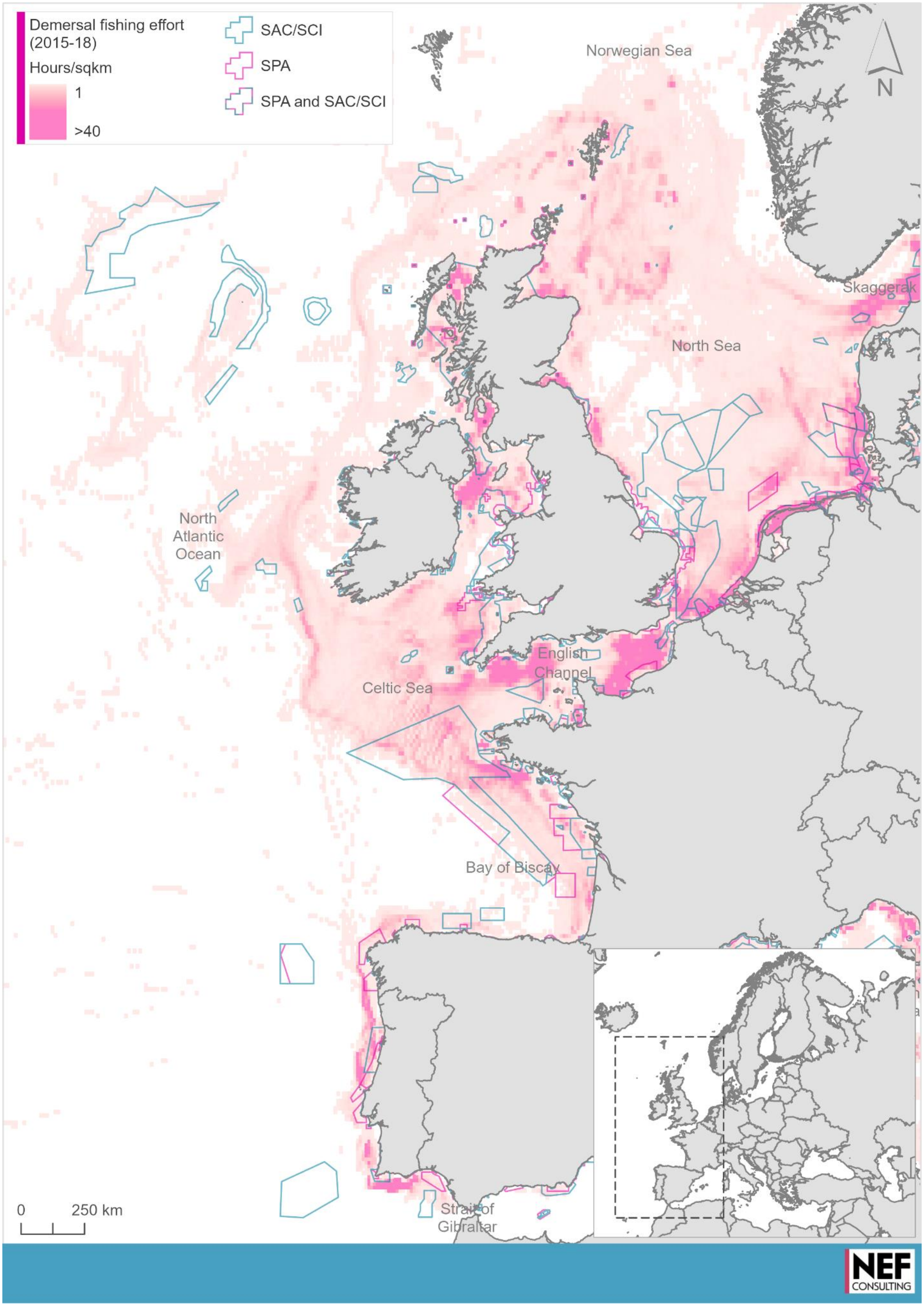
Estimating bottom-contact fishing in European MPAs

The first aspect to consider for the CBA was geographical scope, for example which MPAs to include in the model estimations. Considerations included the data availability and representativeness to reflect Europe's wide diversity of MPAs and management regimes. In consultation with Seas at Risk, it was decided that offshore Special Areas of Conservation (SACs) Natura 2000 sites (European Marine Sites – EMSs)[220] met the practical and representativeness criteria. Natura 2000 data was obtained from the European Environment Agency.[221] Once the geographical scope was outlined, it was then required to determine the proportions of different seabed habitats across these SAC-designated offshore areas. This is important, as different seabed habitats experience varying levels of fishing activity and intensity (therefore impacts as a result of bottom-contact fishing are also varied), so the model needed to reflect types of seabed found in European MPAs. Proportions of seabed type were determined through combining data from Natura 2000 with data from EMODnet broad-scale seabed habitat map for Europe (EUSeaMap).[222] The categorisation used for seabed habitat is the Marine Strategy Framework Directive's (MSFD) classified benthic habitats.[223]

With the area and seabed type across MPAs determined, the next stage required mapping the extent to which bottom-contact fishing was currently taking place in offshore SACs and over which seabed habitats. Working with Marine Conservation Society (MCS), 2015–2018 provisional fishing effort data from Global Fishing Watch (GFW) was used to estimate this bottom-contact fishing activity (see Figure 3.1 for maps visualising this activity). For this model, any EU vessel that is registered as using demersal trawl, dredge, and/or demersal seine gear, in the absence of pelagic trawl gear (which does not have the same seabed impacts), was used. It was necessary to use EU vessels as GFW categorises vessels as 'trawlers' but does not specify whether they are demersal or pelagic. As such, by removing any vessel registered using pelagic trawls as their main/subsidiary gear, this avoids including any pelagic activity in the data. The overall result is a grouping together of effort from all the gears to produce a single 'demersal fishing effort' layer. Combining this data with the Geographical Information Systems (GIS) data for Natura 2000 and EMODnet as outlined, a baseline for likely suspected trawling activity within MPAs and the type of habitat that this corresponded to could be made (Table 3.1). Key limitations with the use of this data include vessel size (only information for vessels >15 m, as these are the vessels that have an Automated Identification System (AIS) installed, was available), and nationality (only EU vessels, for the reasons described, were used – for instance, in the case of the Mediterranean non-EU fleets are also active).



(a)


Demersal fishing effort (2015-18)
Hours/sqkm
1
>40
SAC/SCI
SPA
SPA and SAC/SCI
N
Norwegian Sea
Skaggerak
North Sea
North Atlantic Ocean
English Channel
Celtic Sea
Bay of Biscay
Strait of Gibraltar
0 250 km
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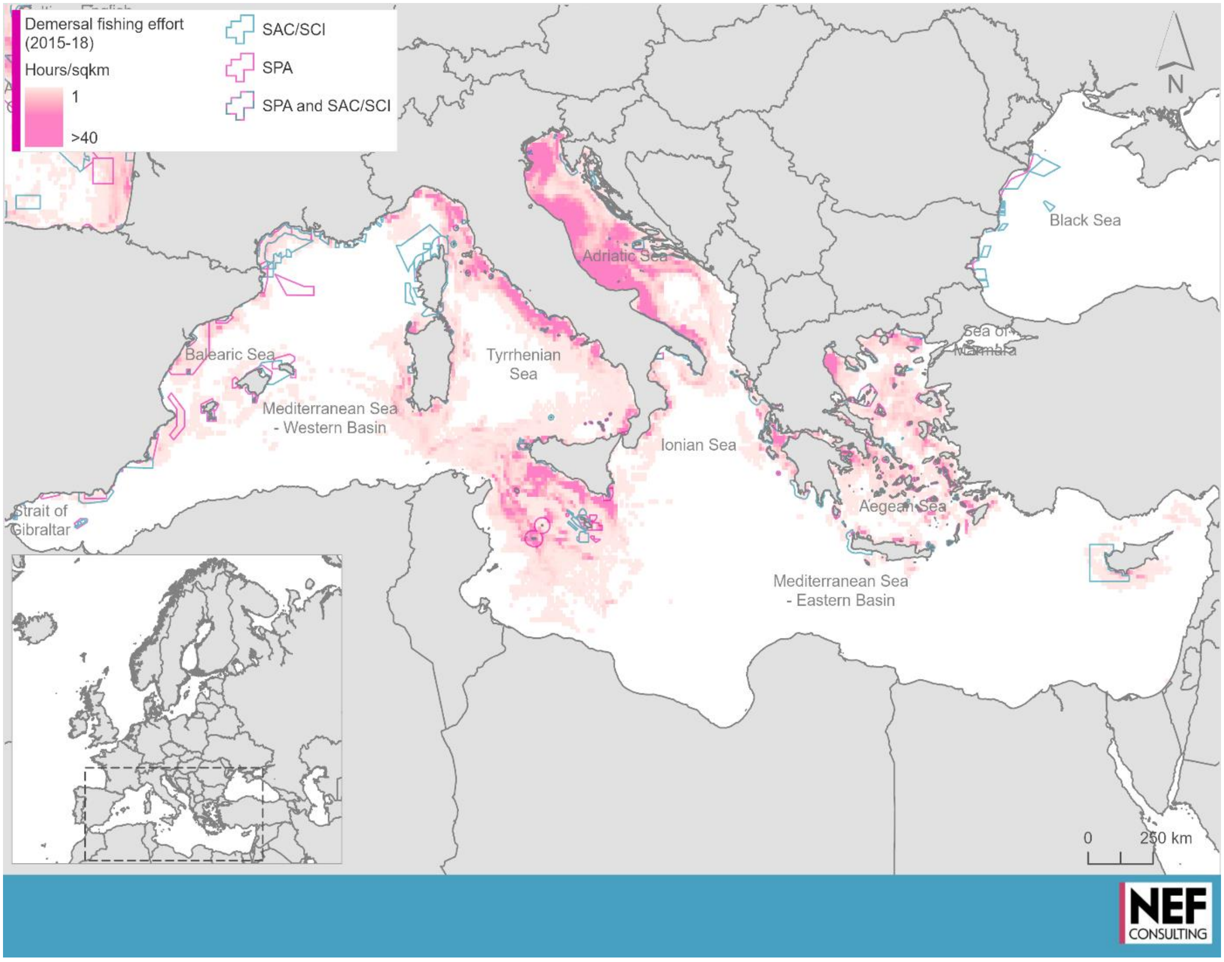
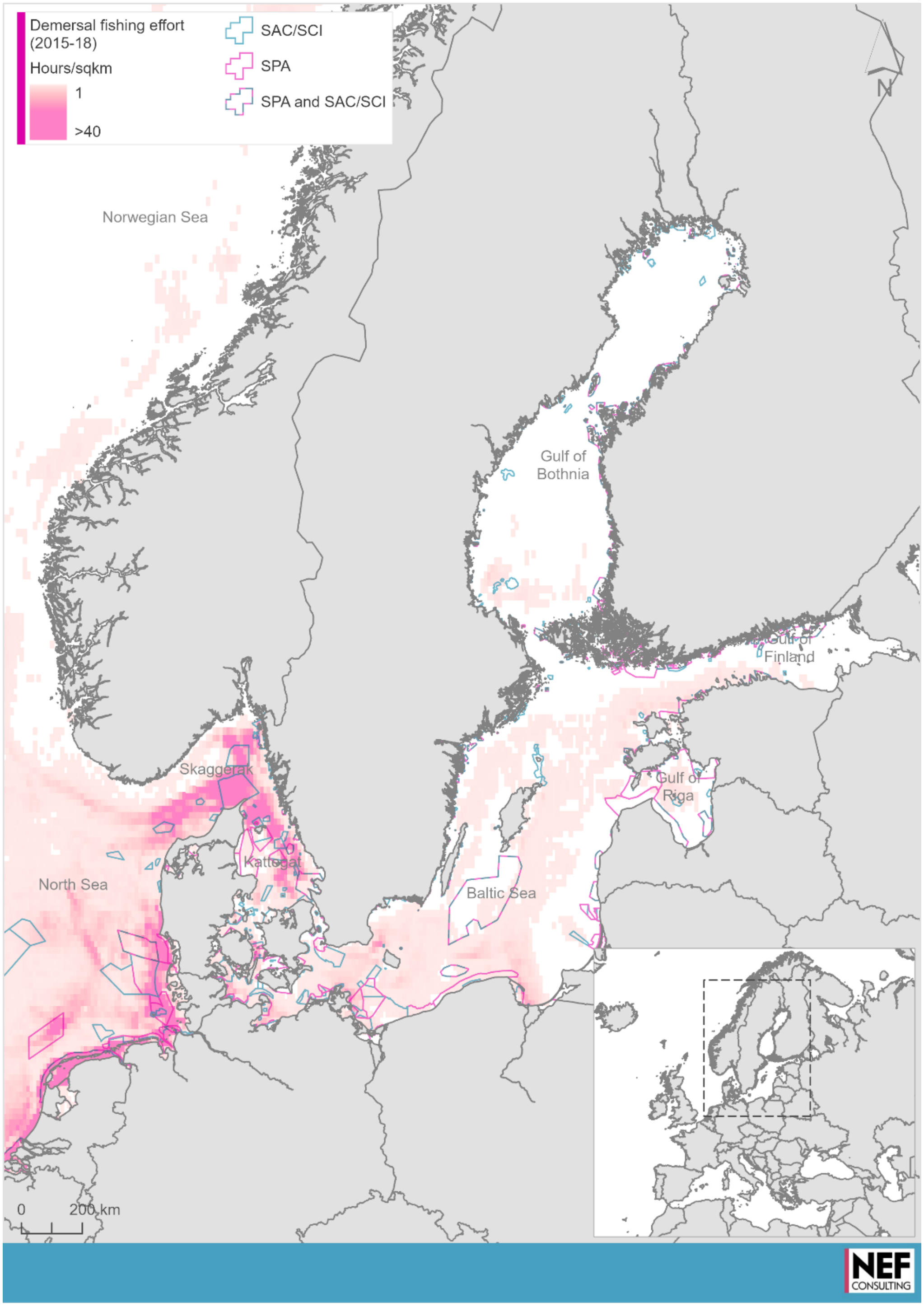
(b)


Demersal fishing effort (2015-18)
Hours/sqkm
1
>40
SAC/SCI
SPA
SPA and SAC/SCI
N
Black Sea
Adriatic Sea
Sea of Marmara
Balearic Sea
Tyrrhenian Sea
Mediterranean Sea - Western Basin
Ionian Sea
Strait of Gibraltar
Aegean Sea
Mediterranean Sea - Eastern Basin
0
250 km
NEF CONSULTING

(c)



(d)

Figure 3.1. Map of offshore Special Areas of Conservation (SACs) Natura 2000 sites and estimated demersal fishing effort across European waters (a) Europe (b) northeast Atlantic (c) Mediterranean (d) Baltic.

Table 3.1. The area of each MPA where bottom-contact fishing is undertaken, as categorised by MSFD benthic habitat type.

	Area of MPA experienced suspected bottom-contact fishing (km^2)				
MFSD benthic habitat	**2015**	**2016**	**2017**	**2018**	**2015-18 average**
Abyssal	746	1,149	746	2,600	1,310
Circalittoral coarse sediment	6,715	5,985	6,715	7,193	6,652
Circalittoral mixed sediment	3,248	2,718	3,248	3,154	3,092
Circalittoral mud	2,639	2,538	2,639	2,495	2,578
Circalittoral mud or Circalittoral sand	113	157	113	492	219
Circalittoral mud or Offshore circalittoral mud	1,191	713	1,191	426	880
Circalittoral rock and biogenic reef	558	464	558	648	557
Circalittoral sand	44,115	41,659	44,115	45,160	43,762
Infralittoral coarse sediment	1,420	1,350	1,420	1,469	1,415
Infralittoral mixed sediment	243	199	243	80	191
Infralittoral mud	203	193	203	288	222
Infralittoral rock and biogenic reef	179	165	179	197	180
Infralittoral sand	8,436	7,674	8,436	6,940	7,872
Lower bathyal rock and biogenic reef	3	11	3	76	23
Lower bathyal sediment	2,207	2,627	2,207	3,468	2,627
Lower bathyal sediment or Lower bathyal rock and biogenic reef	7	22	7	543	145
Na	343	351	343	336	343
Offshore circalittoral coarse sediment	14,947	14,118	14,947	14,860	14,718
Offshore circalittoral mixed sediment	1,154	1,146	1,154	1,128	1,145
Offshore circalittoral mud	11,866	9,471	11,866	9,021	10,556
Offshore circalittoral rock and biogenic reef	1,078	1,095	1,078	1,132	1,096
Offshore circalittoral sand	37,464	36,021	37,464	39,330	37,569
Upper bathyal rock and biogenic reef	203	238	203	452	274
Upper bathyal rock and biogenic reef or Lower bathyal rock and biogenic reef	131	14	131	14	72
Upper bathyal sediment	12,391	14,298	12,391	16,372	13,863
Upper bathyal sediment or Lower bathyal sediment	11,613	9,984	11,613	9,598	10,702
Upper bathyal sediment or Upper bathyal rock and biogenic reef	477	1,031	477	1,356	835
Grand Total	**163,691**	**155,388**	**163,691**	**168,829**	**162,899**

There is likely trawling activity across 162,899 km^2 of the offshore SACs, which represents 29% of the total area (556,489 km^2).

Establishing the impacts of bottom-contact fishing

Incorporating the type and extent to which ecosystem services are impacted by bottom-contact fishing is a central component of the model. As demonstrated in the literature review, bottom-contact fishing impacts the marine environment and socioeconomic activity in diverse ways. For the model, it was necessary to outline a set of ecosystem services that were both broad enough in scope to capture the key impacts of bottom trawling but that also had sufficient data available for (a) the extent to which the ecosystem service was impacted and (b) monetary values per annual unit of change (eg $/ha/year). Two sources were used as the basis of the ecosystem services selected in the model: *the Ecosystem Services Valuation Database (ESVD)* (updated in June 2020)[224] and *The Marine Bill - Marine Nature Conservation Proposal: Valuing the Benefits (MNCP)*, prepared on behalf of Defra by researchers from SAC Ltd and the University of Liverpool. [225] The MNCP presents in detail the type/extent of impact estimated from a scenario that includes “restriction of bottom fishing gears either spatially or temporally and technical conservation measures” in the UK context.[226] Table 3.2 presents the type/extent of impact categorised by ecosystem services and seabed habitats [relevant for MPAs in this study, therefore no shallow inshore waters are included as there are shallow areas where no trawling activity takes place and importantly for this study, inshore vessels are not mandated by law to have an in Vehicle Monitoring System (iVMS) or AIS] and therefore fishing effort that can be overlaid on marine habitats was not available. For each ecosystem service and seabed type, a code ranging from Very Low to Very High is provided in terms of the % impact estimated over the 20 years. Specifically, it states the increase (or, in fact, lack of decrease) in ecosystem services relative to the baseline status quo scenario (which is likely to be a deterioration). The report provides a range for these categories, here we use an average of that range. Hence, the impact coding was adopted as follows: Very High (VH), 95%; High (H), 70%; Medium (M), 30%; Low (L), 5%; and Very Low (VL), 0.5%.

Table 3.2. Impact estimates of conservation measures including restrictions on bottom-

Ecosystem Service	Aphotic reef	Oceanic cold water coarse sediment	Oceanic cold water mixed sediment	Oceanic cold water mud	Oceanic cold water sand	Oceanic warm water mud	Oceanic warm water sand	Photic reef	Shallow moderately tide stress coarse	Shallow weak tide stress coarse sediment	Shallow strong tide stress mixed sediment	Shallow moderately tide stressed mixed	Shallow weal tide stress mixed sediment	Shallow mud	Shallow sand	Shelf strong tide stress coarse sediment	Shelf moderately tide stress coarse sediment	Shelf weak tide stress coarse sediment	Shelf strong tide stress mixed sediment	Shelf moderately tide stress mixed sediment	Shelf weak tide stress mixed sediment	Shelf mud	Shelf sand
Resilience and resistance	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Biologically mediated habitat	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Nutrient recycling	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Gas and climate regulation	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Bioremediation of waste	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Option use values	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Non-use / bequest values	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H
Leisure and recreation	H	VL	VL	VL	VL	VL	VL	H	L	H	M	M	M	L	L	L	L	L	L	L	L	VL	VL
Food provision	M	VL	VL	VL	VL	VL	VL	VL	L	L	L	L	L	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL
Raw materials	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL
Disturbance prevention and alleviation	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL
Cultural heritage and identity	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL	VL
Cognitive values	H	H	H	H	H	H	H	H	M	H	M	M	M	M	M	M	M	VH	H	H	H	H	H

By using the impact coding described by Moran et al.,[227] **we are making several assumptions.** First, that the impacts described here are broadly similar across all European waters. Second, that the status described by Moran et al. is similar to a bottom-contact fishing gear ban and is also similar to the European MPA context. Third, that the baseline used is the expected deterioration in UK waters during the 20 years following ca. 2008, with the assumption that the same deterioration would apply in European waters over the period approx. 2021–2040 in the absence of mobile bottom-contact fishing ban.

The time profile of the impact is applied using the same approach as in the source paper,[228] with the ecosystem services impact of a ban building up gradually for some habitats and ecosystem services as these habitats recover (eg rising at a constant growth rate for 5/6/12.5 years) and occurring instantaneously for others. After this initial build-up period, the level of ecosystem services sits at its maximum for every remaining year until the end of the 20-year period covered by the model. The ecosystem services expected to improve immediately upon protection in some habitats include leisure and recreation, food provision, raw materials, and cultural heritage and identity, albeit in all cases the magnitude of this immediate impact is very low.[229]

For example, a given ecosystem service for a given habitat may eventually increase by 70% 'High' impact coding) but have a build-up period of 5 years. In this case, it will grow by 11.2% per annum from the status quo to achieve a level 70% higher than the status quo by year 5. From year 6 to year 20, the same high level (70% above the status quo) will be achieved each year. The profile of increased ecosystem services per year relative to the status quo is shown in Figure 3.2, using the example of a 'High' impact (70% increase) over ramp-up periods of 5, 10 and 12.5 years.

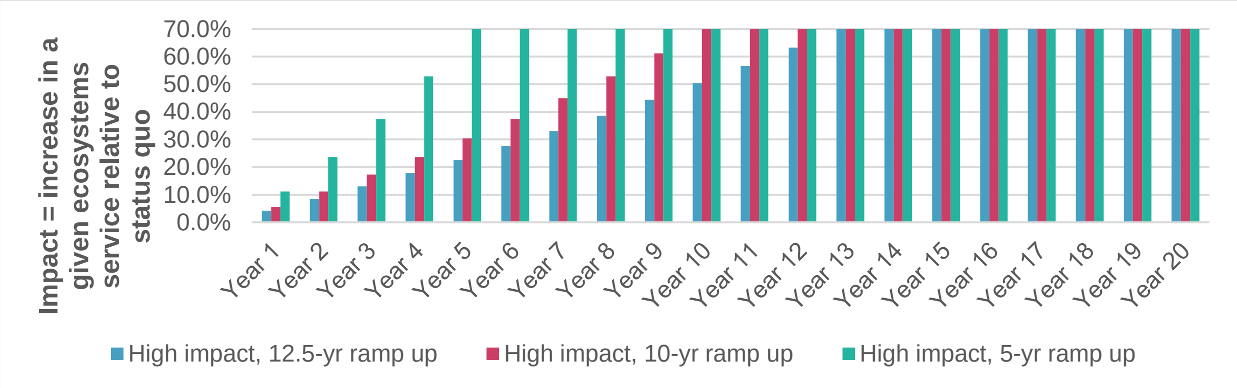


Figure 3.2. Ecosystem services increase by year, relative to status quo, under three different time profiles.

Valuing the economic impact of ecosystem services

The ESVD consists of over 600 studies and more than 4000 value records distributed across all biomes, services, and geographic regions for the economic benefits of ecosystems and biodiversity, as well as the costs of their loss using valuation mentions such as contingent valuation and replacement-cost methods. This database was updated as recently as June 2020. As well as consisting of up-to-date financial proxies for a wide range of ecosystem services (mostly from academic peer-reviewed literature) the database standardises proxies for each ecosystem service in easily comparable units: Int $/ha/year (2020 prices). Here, the assumption is that one ha/year/$ is for the perfect hectare of ecosystem service quality. We assume in the estimation of impact in the cost-benefit model, that a % of this increases in quality when mobile bottom-contact fishing is banned. For this study, the database was queried for the following: ecosystems categorised under 'Open oceans and/or Open seas' (one of the ten biomes covered by the ESVD), and location within Europe. Table 3.3 provides a list of the ecosystem services returned (that possessed monetary values). [230,231,232,233,234,235] The valuation methods used for these financial proxies include Contingent Valuation, Damage Cost Avoided, Market Prices, Net Factor Income, Production Function, Replacement Cost, Travel Cost, and Value Transfer.[236]

Table 3.3. Financial proxies for ecosystem services related to 'Open Oceans/Seas' biome (as categorised by the ESVD) within Europe.[237].

Ecosystem service	Value (€ per hectare per year)*
Biodiversity protection	€7.24
Climate regulation	€34.65
Carbon sequestration	€57.12
Cultural values	€1.91
Fish	€35.24
Food [unspecified]	€3.72
Hunting/fishing	€17.02
Marine leisure and recreation.	€683.23
Nutrient cycling	€23,267.93
Prevention of extreme events	€1.85
Raw materials (the extraction of marine organisms for all purposes, except human consumption)	€7.31
Recreation	€377.66
Recreational use values	€17,501.04
Waste remediation	€180.41

**xe.com 1 EUR = 1.207 USD - Jan 18, 2021, 15:31 UTC*

For the purposes of the model, these ecosystem services needed to align with the ecosystem services outlined in the MNCP (Table 3.2). Table 3.4 presents the alignment of these ecosystem services.[238,]

Table 3.4. Alignment of ESVD ecosystem services with MNCP ecosystem services.

Ecosystem service type (MNCP)	Ecosystem service (MNCP)	Ecosystem service (ESVD - Open Ocean/Sea EU)	Economic value per ha2 per year (€)* *2020 prices
Regulating	Resilience and resistance	Prevention of extreme events	€1.85
Regulating	Disturbance prevention and alleviation	Prevention of extreme events	€1.85
Regulating	Biologically mediated habitat	Biodiversity protection	€7.24
Supporting	Nutrient recycling	Nutrient cycling	€157.44 [239]
Regulating	Gas and climate regulation	Climate regulation	€34.65
Regulating	Gas and climate regulation	Carbon sequestration	€57.12
Supporting	Bioremediation of waste	Waste remediation	€180.41

Provisioning	Option use values		
Provisioning	Non-use/bequest values		
Provisioning	Leisure and recreation	Marine leisure and recreation.	€683.23
Provisioning	Leisure and recreation	Recreation	€377.66
Provisioning	Leisure and recreation	Recreational use values	€17,501.04[240]
Provisioning	Leisure and recreation	Hunting / fishing	€17.02
Provisioning	Food provision	Fish	€35.24
Provisioning	Food provision	Food	€3.72
Provisioning	Raw materials	Raw materials (the extraction of marine organisms for all purposes, except human consumption)	€7.31
Cultural	Cultural heritage and identity	Cultural values	€1.91
Cultural	Cognitive values		N/A

Similarly, the MSFD benthic habitat types used in the EMODnet data required alignment with the habitat types described in the aforementioned MNCP study. Table 3.5 presents this alignment.

Table 3.5. Alignment of MSFD benthic habitats with MNCP habitat types.

MFSD benthic habitat type	MNCP habitat type*
Circalittoral coarse sediment	Oceanic coarse sediment
Circalittoral mixed sediment	Oceanic mixed sediment
Circalittoral mud	Oceanic Mud
Circalittoral mud or Circalittoral sand	Oceanic Mud / Oceanic Sand
Circalittoral mud or Offshore circalittoral mud	Oceanic Mud / Shelf Mud
Circalittoral rock and biogenic reef	Aphotic Reef
Circalittoral sand	Oceanic Sand
Offshore circalittoral coarse sediment	Shelf coarse sediment
Offshore circalittoral mixed sediment	Shelf mixed sediment
Offshore circalittoral mud	Shelf Mud
Offshore circalittoral rock and biogenic reef	Photic Reef
Offshore circalittoral sand	Shelf Sand

** Where more than one MNCP habitat aligns with MFSD, it is split evenly across habitats.*

Incorporating costs and displacement

The final stage of the model incorporates costs and fishing effort displacement to estimate the net impact of a potential bottom towed fishing gear ban. The process of banning bottom trawling in MPAs involves public costs, both in the administration of setting up and enforcement of the bans but also in lost economic benefit (private costs) for fishers when prevented from fishing in these prohibited, but often very productive fishing grounds designated as MPAs. In reality, fishing activity is often displaced to other areas outside MPAs. It is necessary to consider the amount of effort displaced and the ecosystem quality of the areas it is displaced into, compared to the reduction in effort in the protected areas. These factors are incorporated into the model to provide the net benefit value, presented in monetary terms, of a mobile bottom-contact trawling ban. Over time, benefits to the commercial fishery could accrue outside the MPAs through MPA spill-over benefits but this is not part of our analysis. Data from other studies would suggest this is a likely benefit to industry after some years of closures of MPAs to bottom towed fishing gear.

To estimate administration costs, we used estimated costs per hectare per year taken from the UN's *Catalysing Ocean Finance* report.[241] In this report, they describe the annual operational cost of MPAs representing 10% of the world's oceans as $21,191,857,538. If the total area of the world's oceans is taken as 361,100,000 km^2, this represents an annual cost per hectare of $5.87 (€4.86). When applied to total area of MPAs considered in this study (556,489 km^2), the annual costs are estimated as $326,586,295 (€270,576,881). One important thing to consider when estimating MPA costs is the high variability of costs in relation to MPA size. For example, the report describes a variation as large as $293,639/km^2 for MPAs with an average size of $0.5 -5/km^2 for those of an average size of 300,000km^2. For Natura 2000 sites included in this study, it is not possible to know if management plans spread across several smaller sites or if they are managed separately. For this reason, a broad average is used.

To estimate the costs of lost bottom-contact fishing activity, a rough estimation of fishing value per hectare of European sea was made. The estimated total landings value in the EU in 2018 (€7,800,000,000)[242] was divided by the total area of European seas (2,000,000,000 ha).[243] This provides an estimated value of €3.90 per hectare per year. The annual estimated fishing value from Natura 2000 sites in this study is estimated by multiplying this value by total area estimated to be trawled annually within these areas (12,828,429 ha) and then multiplied by the estimated % of catch by mobile bottom-contact fishing in Europe (61%).[244] This provides a value of €30,518,833.

We estimate that displaced activity will be approximately 75% (we have chosen a high estimate based on impact assessments for MPAs in the UK[245]) as some of the forgone landings will be recovered from fishing other grounds. Furthermore, the displaced bottom towed fishing gear activity will instead be undertaken on seabed habitat that is of lower ecosystem services quality in terms of biodiversity and biomass, which in relation to fishing reflects the catch per unit effort – CPUE – of the ground inside an MPA versus the area displaced to, which is assumed as less productive in terms of catch. The assumption is that MPA-designated areas are likely to have higher ecosystem service value as a starting point (hence the need to protect them) but are also more likely to be targeted by higher fishing effort due to the likely higher CPUE. We have conservatively estimated a 90% quality in comparison to designated areas.

Results

Bringing together the information described, we were able to estimate the annual net benefit from a potential bottom-contact fishing ban in European MPAs. Table 3.6 presents the estimated cumulative value for each ecosystem service that was assessed across several stages of a 20-year period following an implementation of a ban on bottom towed fishing gear in these offshore Natura 2000 MPAs (SACs).

Table 3.6. Cumulative ecosystem service benefits over 20 years resulting from a mobile bottom-contact fishing ban in offshore Natura 2000 MPAs (SACs) by ecosystem service type.

		€ million			
Ecosystem service type	**Ecosystem service**	**1-year impact**	**5-year impact**	**10-year impact**	**20-year impact**
Regulating	Resilience and resistance	1.6	25.8	77.0	193.5
Regulating	Biologically mediated habitat	6.2	100.9	301.3	757.0
Supporting	Nutrient recycling	135.3	2194.9	6554.3	16466.6
Regulating	Gas and climate regulation	39.4	639.7	1910.2	4799.0
Supporting	Bioremediation of waste	155.0	2515.2	7510.6	18869.1
Provisioning	Leisure and recreation	29.6	260.5	641.4	1351.7
Provisioning	Food provision	0.7	5.2	12.0	24.1
Provisioning	Raw materials	0.3	1.7	3.7	7.3
Cultural	Cultural heritage and identity	0.1	0.6	1.1	2.0
	Total impact	368.2	5744.5	17011.5	42470.4

** Discount rate used is 3.5% per annum in accordance with UK Treasury Green Book. Inflation is applied in all future years at a rate of 2.0%, based on the IMF's forecast for the European economy, 2021–2025, as contained in the World Economic Outlook October 2020.*[246]

The majority of value comes from two supporting services, bioremediation of waste and nutrient cycling, with each accounting for approximately 40% of the impact over the 20-year period. In the first year, this amounts to €290 million in value and cumulatively €35 billion across a 20-year period for those two supporting services together. The next highest value was for the regulating service of gas and climate regulation with approximately 11% of the total. Cultural services produced by a bottom-contact fishing ban is of very small value, only approximately 0.1% of all the value found at each stage. This is due to a combination of the low financial proxy per hectare value (€1.91) and the minimal impact predicted across habitats within the MNCP impact coding (0.2% per year). Similarly, food provision services produce a notably small value (€700,000). This is to do with the minimal impact predicted across habitats within the MNCP impact coding (0.2% per year).

Table 3.7 presents estimations of costs and displacements across the 20-year time period. In the first year of impact, the estimated annual fishing value lost within MPAs is €31 million, of which 75% is displaced elsewhere, so a total of approximately €8 million is lost. This displaced activity reduces the total ecosystem services impact outlined in Table 3.6 by €249 million. Estimated administration costs in the first year are €304 million across all the MPA sites.

Table 3.7. Cumulative costs and displacement values across the 20-year period following a ban on bottom towed fishing gear in all offshore SACs.

	Year 1	Year 5	Year 10	Year 20
Estimated annual fishing value lost within MPA (€ million)	30.9	149.9	289.3	539.2
Total fishing value lost (net of displacement) (€ million)	7.7	37.5	72.3	134.8
Ecosystem services impact of displaced bottom-contact activity (€ million)	248.5	3,877.5	11,482.8	28,667.5
Estimated annual administration costs (€ million)	296.3	1,439.2	2,777.1	5,177.1
Total costs (€ million)	**304.0**	**1,476.7**	**2,849.5**	**5,311.9**

When ecosystem service benefits and costs/displacement are brought together, the net benefit of a mobile bottom-contact fishing ban across European MPAs can be estimated. Table 3.8 and Figure 3.3 present the annual net impact value for each year across the 20-year period. As can be seen, the costs of implementing a ban outweigh the benefits in terms of annual net impact for the first two years. However, from year 3 onwards, there is an annual net benefit, which rises sharply up to year 5, as the ecosystem service impacts become increasingly more pronounced. The benefits for many of the ecosystem services ramp up until year 13, where the habitat reaches a theoretical maximum of annual ecosystem service value. By year 13, the highest annual net impact value is observed, €615 million. From this point, there is a very gradual decrease in annual net ecosystem service, the result of the discount rate (3.5%) being greater than the rate of inflation (2%).

Table 3.8. Annual net impact value for 20-year period of mobile bottom-contact fishing ban (€ million).

Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10
-184.4	-60.4	70.5	208.9	355.6	396.8	426.8	457.3	488.3	519.9
Year 11	**Year 12**	**Year 13**	**Year 14**	**Year 15**	**Year 16**	**Year 17**	**Year 18**	**Year 19**	**Year 20**
551.1	582.8	615.0	606.1	597.4	588.7	580.2	571.8	563.5	555.3

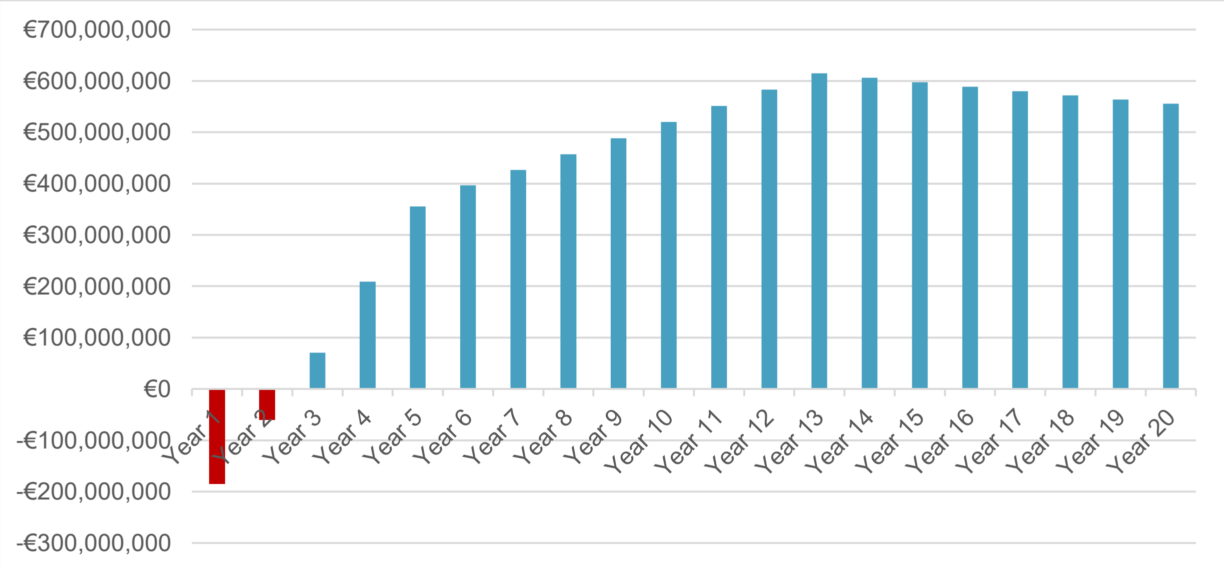


Figure 3.3. Annual net impact value for a 20-year period of mobile bottom-contact fishing ban.

Table 3.9 presents the cumulative ecosystem service benefits, cumulative total costs, and cumulative net benefit across the 20-year period, with Figures 3.4 and 3.5 presenting these values visually. The first three years witness a loss, as costs of implementation and lost fishing activity are greater than ecosystem service benefits gained. However, from year 4 to year 5 there is a sharp increase in cumulative net impact, with a considerable increase from €34 million to €390 million, as the effects of protecting seabed habitats lead to improved ecosystem services. By year 10, we see a net impact of €2.7 billion, and a value of more than treble that by year 20, ie €8.5 billion.

From year 13 onwards, where using the MNCP impact trajectories for each seabed habitat, we see a theoretical maximum improvement reached for many ecosystem services, with the annual net impact value increases stabilising from this point. For the time period between year 13 and year 20, we see an average cost-benefit ratio for a potential ban on bottom-contact fishing in MPAs of 3.41:1 (€3.41 returned for every €1 spent), a positive return despite including very conservative estimates regarding potential displacement (25% loss of fishing landings, displacement of 75% catch to an area that is 90% quality of protected areas).

Table 3.9. Cumulative ecosystem services benefits, costs and net impact across a 20-year period.

	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10
Ecosystem benefit (€ million)	119.7	358.9	724.7	1,224.6	1,867.0	2,546.4	3,251.6	3,983.4	4,742.2	5,528.7
Total costs (€ million)	304.0	603.6	898.9	1,189.9	1,476.7	1,759.3	2,037.9	2,312.3	2,582.9	2,849.5
Net impact (€ million)	-184.4	-244.8	-174.3	34.7	390.3	787.0	1,213.8	1,671.0	2,159.4	2,679.3
	Year 11	**Year 12**	**Year 13**	**Year 14**	**Year 15**	**Year 16**	**Year 17**	**Year 18**	**Year 19**	**Year 20**
Ecosystem benefit (€ million)	6,342.5	7,184.2	8,054.4	8,912.0	9,757.2	10,590.1	11,411.0	12,220.0	13,017.2	13,802.9
Total costs (€ million)	3,112.2	3,371.1	3,626.3	3,877.8	4,125.6	4,369.8	4,610.5	4,847.7	5,081.5	5,311.9
Net impact (€ million)	3,230.3	3,813.1	4,428.1	5,034.3	5,631.6	6,220.3	6,800.5	7,372.2	7,935.7	8,491.0

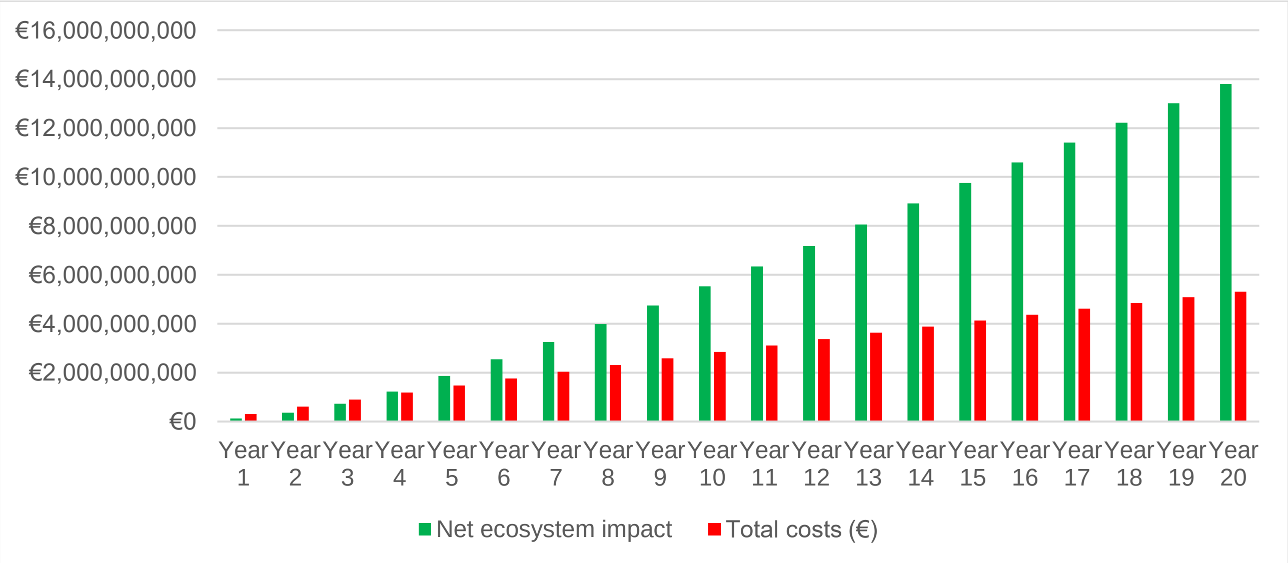


Figure 3.4. Cumulative ecosystem services benefits, and costs across a 20-year period.

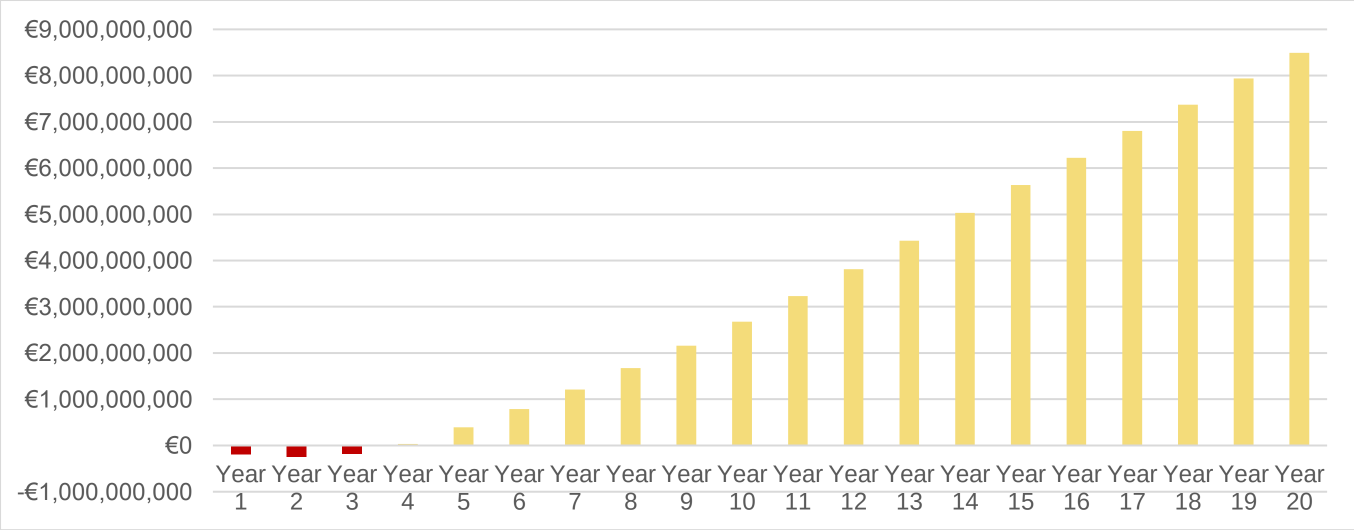


Figure 3.5. Cumulative net impact across a 20-year period.

4. CASE STUDIES

To get a greater sense of the benefits and costs associated with an MPA bottom-contact fishing gear ban and to contextualise the relevance and possible utility of the ecosystem services cost-benefit analysis (CBA) developed here, we briefly explore two case studies where a ban has been implemented or proposed. We describe the context and observed/expected impacts before using available information to estimate impacts using the ecosystem services CBA model.

Case study 1: 'Cod box' in the southern Kattegat, Sweden

Background to the Marine proected area

Established in 2009, the 'cod box' Marine Protected Area (MPA) in the southern Kattegat consists of a no-take zone (NTZ) and three adjoining areas that are subject to various restrictions to fishing gear use at certain times of the year.[247] The area in question was fished extensively over the previous century using otter trawls, which targeted cod during spawning season and *Nephrops* (langoustine, a mud-dwelling crustacean shellfish similar to a small lobster) and various other finfish during the rest of the year.[248] The creation of the MPA was a joint effort of the Swedish and Danish governments in response to a reduction of over 80% in the Spawning Stock Biomass (SSB) of Kattegat cod stocks between the 1970s and the 2000s, which the previous approach of reducing total allowable catch had failed to halt.[249] Thus, the MPA is used as a fisheries management tool, rather than a conservation tool. However, embedded in the MPA are several Natura 2000 areas and a Swedish marine nature reserve.

The MPA, shown in Figure 4.1, is subject to the following restrictions: in **area 3** all fishing is forbidden (hereafter the NTZ), while in **area 2** fishing throughout the year is only permitted using selective gears with a very low catch of cod and all trawling is banned during cod spawning season (between 1 January and 31 March). In **area 1**, only selective gears are allowed between 1 January and 31 March each year, and in **area 4** only selective gears are allowed between 1 February and 31 March. The selective gear types allowed, which were assessed to minimise bycatch of cod, included trawl gear with a Swedish *Nephrops*-grid and an 8-m long 70-mm square mesh cod-end installed, and cod-ends with a 300-mm square mesh panel installed 3–6 m from the codline.[250]

There have been ongoing restrictions applied to fishing in the Kattegat that are not directly related to the MPA, including requiring the use of more selective trawling gear in the whole area for *Nephrops* fisheries (since summer 2020)[251] and increasing use of remote electronic monitoring (cameras on trawlers) by the Danish Fisheries Agency.[252] A marine nature reserve, Skånska Kattegatt, was established in 2020 covering the NTZ to ensure continued protection and recovery of benthic habitats and to safeguard important habitats for fish and shellfish species such as cod and *Nephrops*.[253]

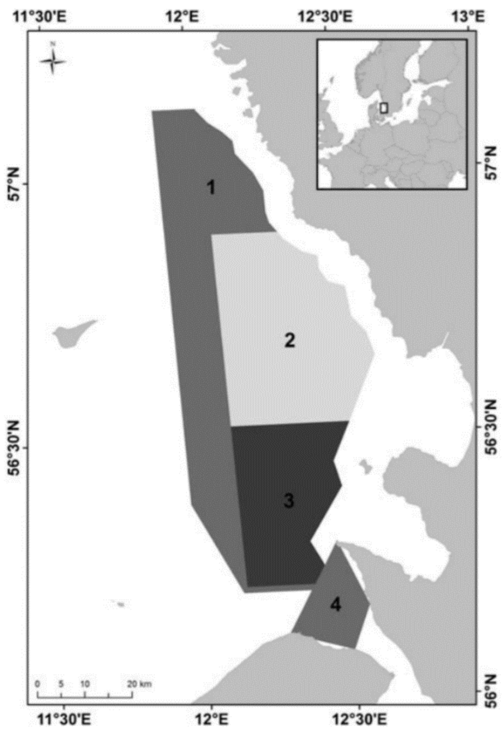


Figure 4.1. Location of MPA zones in the southern Kattegat.[254]
Key to areas: 1: partial gear restrictions (see above); 2: partial gear restrictions (see above); 3: no-take zone; 4: partial gear restrictions (see above).

Changes observed as a result of the MPA

Observed benefits

Fishing effort

The MPA has successfully reduced fishing effort within the NTZ. A reduction in trawling intensity has been observed within the NTZ relative to outside.[255] The median trawling intensity observed at sites in the NTZ fell from twice per year in 2009, to approximately once per year in 2010 and zero times per year in 2011 and 2014, compared with between two and six times per year at sites outside the MPA during the same three years.[256]

Some spatial displacement of fishing effort appears to have occurred due to the restrictions. Modelled estimates of fishing effort relative to 2008 levels, based on data from VMS and logbooks, indicate that by 2012–2015 a reduction of 95%–98% had occurred within the NTZ (Area 3).[257] There was a less pronounced reduction in fishing effort in Area 2 (reduction of between 8% and 66% from 2011 to 2015), while some fishing effort was displaced into Area 1 (where effort rose by 39%–95% over the years from 2009 to 2014) as intended by the restrictions.[258] In other areas of the Kattegat not covered by the restrictions, fishing effort fell only gradually between 2009 and 2015 (never below 84% of its 2008 level).[259]

The period after the MPA was created coincided with a reduction in total fishing impact on cod, with the MPA being just one of several causes. There is no direct fishing of cod in the Kattegat at present but a significant quantity of cod is caught each year as bycatch in *Nephrops* fisheries.[260] Total fishing impact on cod in the Kattegat fell by 70% between 2007 and 2017,[261] driven in part by a reduction in the total fishing effort of Kattegat

Nephrops fisheries over a similar period (2010–2016 for the Danish fleet and 2007–2016 for the Swedish fleet).[262] The NTZ was found to have reduced the impact of Danish fisheries on medium and large cod, but the switch to more selective fishing gears was found to have been the main driver of reduced impact for both the Danish and Swedish fleets.[263]

There have been some violations of the NTZ by fishers, estimated to have occurred 10 to 20 times per year.[264] Most notably, some illegal fishing by the Danish fleet occurred in 2010 within the NTZ, although during the period 2011–2017 there was almost no fishing effort within the NTZ.[265] Monitoring compliance has been challenging due to a lack of powers for Swedish authorities over Danish fishers and the challenge of conclusively proving ex post that certain fish were trawled within the MPA.[266]

In contrast with the adjacent Öresund area,[267] **recreational fishing in the MPA is extremely limited** due to the low cod stocks, meaning that there is unlikely to be any economic benefit of this kind arising from the MPA.[268]

Ecological recovery

The cod population has not recovered fully since the implementation of the MPA, suggesting that the restrictions introduced in 2009 were not sufficient in themselves to ensure a recovery in cod stocks. There was an initial positive trend in spawning-stock biomass (SSB) between 2009 and 2015, but this has reversed in the past five years and SSB fell to a historically low level in 2020.[269] This collapse in cod biomass has been attributed to the removal of fishing effort regulation, which has reduced the incentive to use selective gears that minimise bycatch of cod by *Nephrops* fisheries.[270,271,272] Relative mortality among cod in the Kattegat as a whole fell steadily from 2009 to 2015, although it has since begun to rise, reversing that trend and offsetting nearly all of the improvement.[273]

Projected benefits

Due to the very low level of cod stocks in the Kattegat at present, future improvements have the potential to yield substantial increases in fishing productivity. Prior to the establishment of the MPA, annual landings of cod in the adjacent but separate Öresund area (most of which has had a trawling ban in place since 1932) were ten times the equivalent figure for the Kattegat, despite the Öresund area being one-tenth of the size of the Kattegat.[274] Catch per unit effort for cod in the Kattegat was typically 20 to 50 times lower than in Öresund during the 2000s.[275] Catch per unit effort in the Kattegat (as measured by the annual catch of cod in kg per km^2) improved considerably by 2015 after improvements in cod biomass, to a level 7–18 times higher than the equivalent figure before the MPA was designated (2008).[276] Although this improvement cannot be solely attributed to the MPA, it indicates the substantial efficiency benefits for fishers if cod stocks were given sufficient time and protection to recover fully (through a combination of regulations on gear types, fishing effort, and spatial restrictions).

It is likely to take a long time for biodiversity to recover in the NTZ, though some small changes were recorded in the first few years after the MPA was created. Over the past century the Kattegat has lost most of its original species richness due to various factors, including trawling impact.[277] Benthic communities in the Kattegat are dominated by burrowing brittle stars (*Amphiura filiformis* and *Amphiura chiajei*), which are relatively more resistant to the impact of bottom trawling compared with other species such as clams, bivalves, and ostracod.[278] Trawling intensity has been found to have a significant impact on the number, richness, and diversity of species and habitat quality in the Kattegat, with an increasingly negative impact on species number, diversity, and richness as trawling passes per year increase from zero to five.[279] Between 2009 and 2014 there was a decrease in the abundance of the two dominant species of brittle stars in the NTZ, compared with no change in the parts of the Kattegat still being trawled.[280] This difference was not highly statistically significant, but may be evidence of the effects of increased predation from recovering flatfish and *Nephrops* populations.[281] Later studies commissioned by Skåne County Administrative Board showed greater abundance and densities of sea pens (*Pennatula phosphorea, Virgularia mirabilis)* within the NTZ as well as findings of ocean quahog (*Arctica islandica*) and northern horsemussel (*Modiolus modiolus*), indicating the ecological recovery of benthos sensitive to trawling.[282]

It is possible that a recovery of cod stocks as a consequence of protective measures (including the MPA) could reduce eutrophication in the Kattegat.[283]

Costs

Because the MPA is relatively farther from the coast compared with others in Sweden, monitoring is carried out by the Swedish Coast Guard rather than the Skåne County Administrative Board.[284] The costs to the Coast Guard were estimated by survey at 30% of the salary for one full-time job (the time spent monitoring) and increased diesel fuel costs.[285] For the County Administrative Board, the initial cost in the year the MPA was established was found to be 1% of the salary for one full-time job and the ongoing cost was equivalent to five full-time positions per annum.[286] In light of a lack of reliable data, however, there was significant uncertainty surrounding these cost estimates, especially in the case of the County Administrative Board's ongoing cost.[287] Nonetheless, the total cost of the MPA is estimated at SEK48,000 for the initial year before it was established and SEK2,544,000 per annum thereafter.[288]

Given that the NTZ is closer to the coast than the other parts of the MPA and the wider Kattegat, and given that vessel monitoring system data indicates that very little fishing has occurred in the NTZ since 2010, there is likely to have been an additional fuel cost to fishers as a consequence of the MPA. This is acknowledged in a 2012 evaluation of the MPA but has yet to be accurately quantified.[289] For this reason we cannot incorporate fuel costs into our estimates of displacement.

Other considerations

Migration of cod between the Kattegat and the North Sea has been found to be significant, meaning that the cod caught in the Kattegat originate from multiple populations.[290] Return migration of cod to the North Sea is thought to explain some of the reduction in cod in the Kattegat that is not accounted for in fishing catch data.[291] This interaction between populations makes ecological management of the Kattegat cod population more complex and suggests that the future success of such measures will depend in part on management of cod stocks in the North Sea.

Gear conflict is most commonly reported as being due to mobile gear (trawls) passing through an area in which static gear (pots or fixed nets) are located. Removing trawl effort can provide opportunities to increase potting or static netting effort (where environmental conditions, eg depth and currents allow). Research has shown that in the case of *Nephrops,* trap-caught carry a higher price and number of jobs supported per tonne landed.[292]

Displacement of static gear *into* the MPA where trawling is excluded, may take place, but this has not been included in the modelling.

Model simulation

We apply the CBA model to a hypothetical improvement in the condition of the NTZ habitats, particularly considering it has now been protected as a marine nature reserve. The NTZ (Area 3) is approximately 647 km^2 in area.[293] We estimate from EUSeaMap resources that this area of seabed is made up of approximately 83% shelf mud, 15% shelf sand, and 2% shelf coarse sediment.[294]

The model suggests that if this area were to recover ecologically due to a continued ban on bottom trawling, there would be a significant increase in annual ecosystem services relative to the status quo, with this increase widening over the initial 6-year period after the ban before remaining at its maximum level. The net present value of this increase is estimated at €2.89 million during the first year, reaching €47.0 million in the first 5 years, €125.9 million in the first 10 years and €267.5 million over the full 20-year period covered by the model (Table 4.1). Based on the data outlined on the Swedish administration of the MPA, there would be an ongoing cost of €260,000 per annum and an initial set-up cost of €4,915, both reflecting labour costs, which compares very favourably to the large annual benefits.

Given that bycatch of cod occurs in *Nephrops* fisheries in the Kattegat, some constraints on these fisheries may be worth exploring in the short term to hasten the recovery of cod stocks. The value of *Nephrops* landings in the Skagerrak and Kattegat combined was estimated at €11,989,000 in 2013/2014,[295] while the Kattegat accounted for 31.2% of landings of *Nephrops* in the two areas in 2014,[296] suggesting the revenue from *Nephrops* in the whole of the Kattegat (not just the NTZ or MPA areas) was approximately €3.74 million in 2013/2014. Given that the increase in ecosystem services in a recovery scenario would top out at approximately €17.73 million per annum in a given year (after the initial 6-year ramping up period), some short-term restrictions on *Nephrops* fisheries could be accommodated in cost-benefit terms without driving the MPA into negative net-benefit territory.

Table 4.1. Ecosystem service benefits and administrative costs of the Kattegat NTZ, assuming ecosystem recovery (all figures in net present value, with 2% inflation per annum and discount rate of 3.5%) (€ millions).

	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10
Benefits	2.9	5.9	9.1	12.6	16.5	16.2	16.0	15.8	15.6	15.3
Cumulative benefits	2.9	8.8	17.9	30.6	47.0	63.3	79.3	95.1	110.6	125.9
Administrative costs	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cumulative administrative costs	0.3	0.5	0.7	1.0	1.2	1.5	1.7	1.9	2.1	2.4

	Year 11	Year 12	Year 13	Year 14	Year 15	Year 16	Year 17	Year 18	Year 19	Year 20
Benefits	1.5	1.5	1.5	1.4	1.4	1.4	1.4	1.4	1.3	1.3
Cumulative benefits	14.1	15.6	17.1	18.5	19.9	21.3	22.7	24.1	25.4	26.7
Administrative costs	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cumulative administrative costs	0.3	0.3	0.3	0.3	0.3	0.4	0.4	0.4	0.4	0.4

Case study 2: Danish Natura 2000 sites in the Kattegat

Context

Denmark has designated 97 marine Natura 2000 sites in its territorial waters of the western Baltic, Kattegat, Skagerrak, and the North Sea. A total of 65 sites have been designated for the protection of reef structures. In general, the conservation status of reef structures in the Danish Natura 2000 sites are classified as unfavourable due to physical disturbances and the high nutrient content in the water column. In recent years, the Danish government has presented proposals to the European Commission for fishery management measures on three sites: Store Middelgrund, Schultz og Hastens Grund samt Briseis Flak, and Havet omkring Nordre Rønner.[297] The measures include the prohibition of fishing activity with mobile bottom-contacting gear in areas mapped as reefs. The reef structures found in these sites are protected from fishing activity impacts by placement of buffer zones.297

Data

In its management plan proposal document297 and their base analysis documents for each site,[298] the Danish Environmental Protection Agency presents information useful for incorporation in the CBA model. The 'basisanalyse' documents provide maps of fishing intensity with bottom trawling fishing gear in these areas over a 2013–2018 period as well as 2018 specifically. Figure 4.2 presents these maps with fishing intensity present in 100 m x 100 m square. Using these maps, we can visually estimate (albeit crudely) the total area of the MPAs affected by mobile bottom-contact fishing (Table 4.2).

Figure 4.2. Map of fishing intensity with bottom trawling fishing gear in 2018. The map shows the fishing intensity of bottom trawling fishing gear in 2018. (a) Store Middelgrund; (b) Havet omkring Nordre Rønner; (c) Schultz og Hastens Grund samt Briseis Flak.

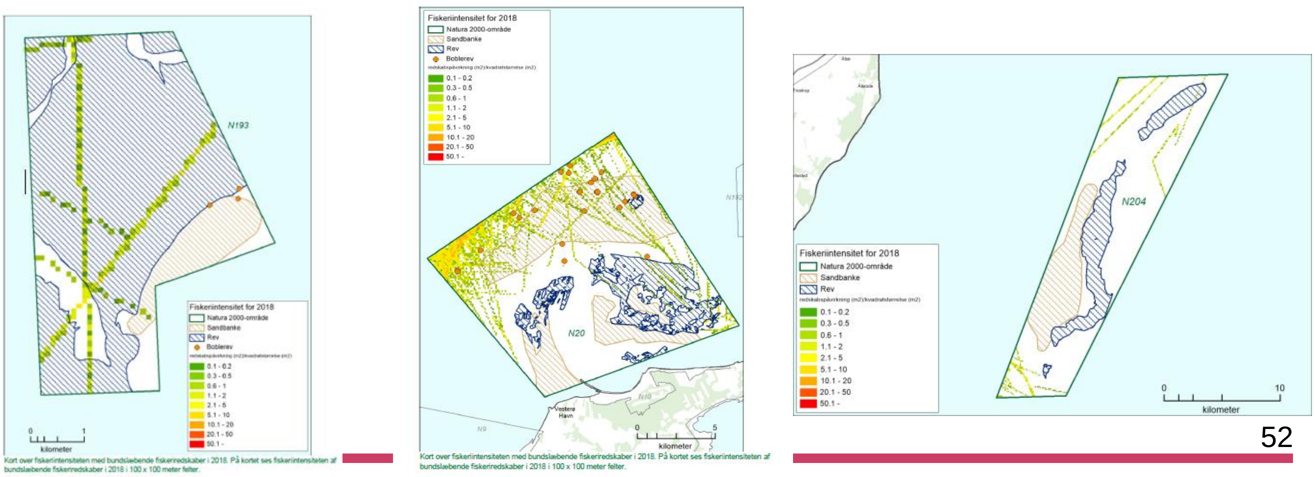


Table 4.2. Areas of seabed habitat bottom-trawled in 2018 within three Kattegat MPAs. Note: To implement values into the CBA model, sandbanks were categorised as 'shelf sand'; stone reefs categorised as 'photic reef', and areas non-identified in the basisanalyse maps categorised as 'oceanic mud'.

Store Middelgrund			
Seabed habitat type	**Area / ha**	**Estimated % trawled annually**	**Area trawled / ha**
Sandbanks	1651	10	165.1
Stone reefs	207	2	4.1
Bubbling Reefs	3	0	0
Oceanic mud	286	2	57.2
		Total area trawled	**226.4**
Havet omkring Nordre Rønner			
Seabed habitat	**Area / ha**	**Estimated % trawled annually**	**Area trawled / ha**
Sandbanks	5963	10	596.3
Stone reefs	2564	50	1282
Bubbling Reefs	26	90	23.4
Oceanic mud	10070	30	3021
		Total area trawled	**4922.7**
Schultz and Hastens Grund and Briseis Flak			
Seabed habitat	**Area / ha**	**Estimated % trawled annually**	**Area trawled / ha**
Sandbanks	3257	0	0
Stone reefs	3223	2	64.5
Bubbling Reefs	0	0	0
Oceanic mud	14353	5	717.7
		Total area trawled	782.11
	Total area trawled annually / ha		
Sandbanks	**761**		
Stone reefs	**1351**		
Oceanic mud	**3796**		

The Danish government's recent proposal to the European Commission for fishery management measures of these areas provides mean annual landed fishing values from Danish, Swedish, and German vessels between 2011 and 2015. These values (Table 4.3) can be used in the model as lost fishing costs.

Table 4.3. Mean annual landing value per Kattegat site (2011–2015).

	Mean annual landing value (2011-15)
Store Middelgrund	€10,850
Nordre Roenner	€76,607
Schultz and Hastens Grund and Briseis Flak	€8,385
Total	**€95,842**

Lastly, both the 'basisanalyse' documents and the management plan proposal describe how the allocated protected areas are of higher ecological quality than surrounding areas. We assume a high displacement value as used in the overall European seas estimation of 75%[299]; however we use a lower ecological quality percentage for areas (50%) where displaced fishing takes place. The documents do not provide annual management costs, so we used the cost/hectare value used for the European seas (€4.86).

Applying the model to Kattegat sites

Using the data described, the model is used to estimate the net impact of a potential ban in these sites. Tables 4.4 and 4.5 and Figures 4.3–4.5 present these estimations. The first two years witnessed a net impact loss as costs of implementation outweigh the initial ecosystem services benefits. As ecosystem services improve as seabed habitats are protected from damage associated with bottom-contact fishing, the net impact becomes increasingly beneficial. By year 10, we see an estimated cumulative benefit of €4.3 million and €12.3 million by Year 20. By Year 10, there is a cost-benefit ratio of €2.11 for every €1, a ratio that grows larger by Year 20 to €3.22 for every €1 the ban costs. This ratio is higher than the €1.46 observed for the whole European seas. This is primarily down to the greater proportion of reefs found in these areas, a seabed habitat that is more susceptible to experience ecosystem service improvements from the prevention of bottom-contact fishing than other habitats such as oceanic sand or shelf sand.

Table 4.4. Annual net impact value for a 20-year period of mobile bottom-contact fishing ban for three Danish Natura 2000 sites in the Kattegat (€ million).

Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10
-0.08	0.06	0.21	0.37	0.54	0.57	0.61	0.65	0.68	0.72
Year 11	**Year 12**	**Year 13**	**Year 14**	**Year 15**	**Year 16**	**Year 17**	**Year 18**	**Year 19**	**Year 20**
0.76	0.80	0.84	0.83	0.82	0.81	0.80	0.78	0.77	0.76

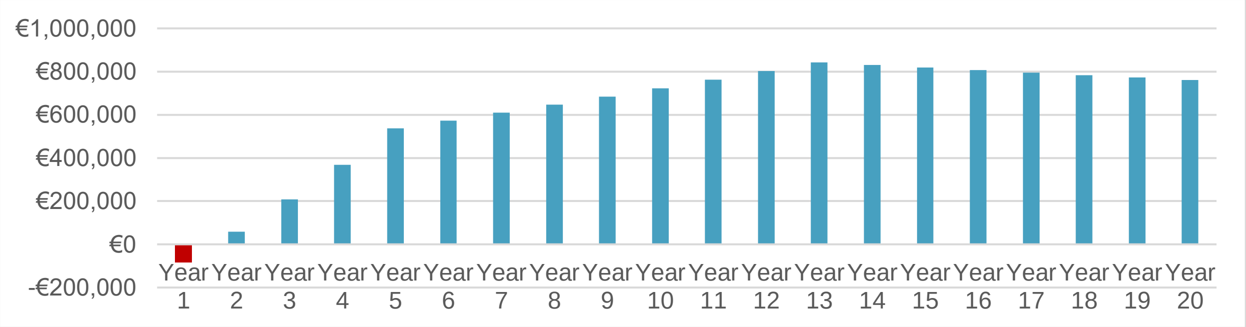


Figure 4.4. Annual net impact value for a 20-year period of mobile bottom-contact fishing ban for three Danish Natura 2000 sites in the Kattegat.

Table 4.5. Cumulative ecosystem services benefits, costs and net impact across a 20-year period for three Danish Natura 2000 sites in the Kattegat.

	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10
Ecosystem benefit (€ million)	0.1	0.4	0.8	1.4	2.1	2.9	3.7	4.6	5.5	6.4
Total costs (€ million)	0.2	0.4	0.6	0.9	1.1	1.3	1.5	1.7	1.9	2.0
Net impact (€ million)	-0.1	0.0	0.2	0.6	1.1	1.7	2.3	2.9	3.6	4.3
	Year 11	**Year 12**	**Year 13**	**Year 14**	**Year 15**	**Year 16**	**Year 17**	**Year 18**	**Year 19**	**Year 20**
Ecosystem benefit (€ million)	7.3	8.3	9.3	10.4	11.3	12.3	13.3	14.3	15.2	16.1
Total costs (€ million)	2.2	2.4	2.6	2.8	3.0	3.1	3.3	3.5	3.6	3.8
Net impact (€ million)	5.1	5.9	6.7	7.6	8.4	9.2	10.0	10.8	11.5	12.3

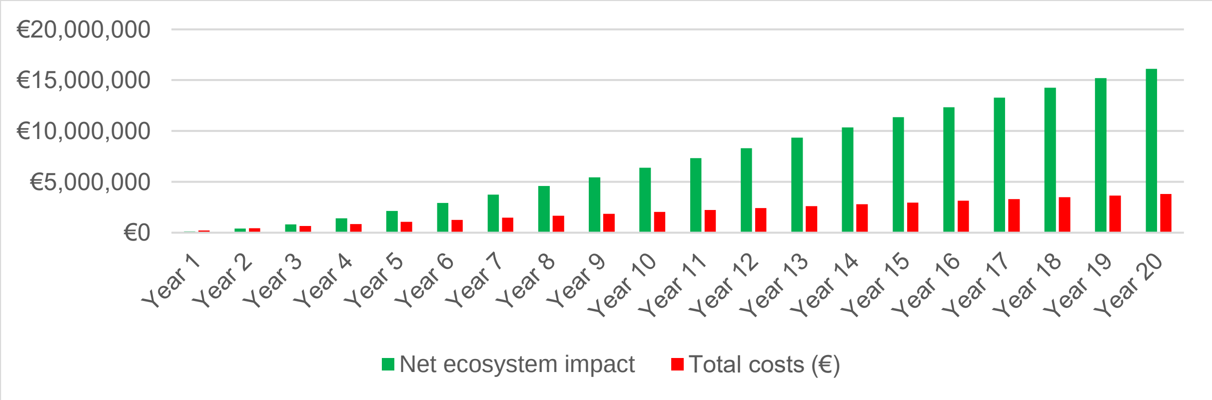


Figure 4.5. Cumulative ecosystem services benefits, and costs across a 20-year period for three Danish Natura 2000 sites in the Kattegat.

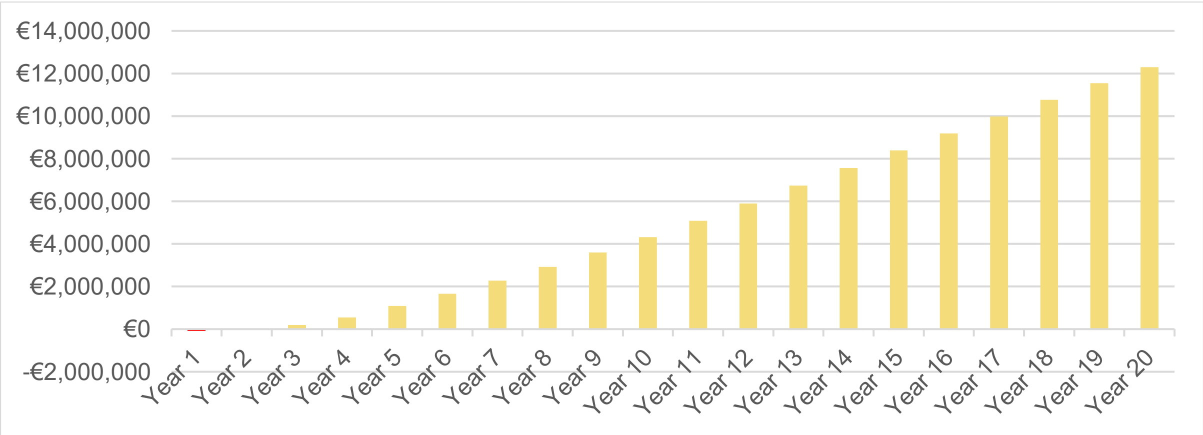


Figure 4.6. Cumulative net impact across a 20-year period for three Danish Natura 2000 sites in the Kattegat.

5. CONCLUSION AND RECOMMENDATIONS

This research has sought to develop a generalised model to capture and value the long-term ecosystem services benefits associated with a ban on bottom-contact fishing in economic terms. Historically these wider environmental and societal benefits are often under-acknowledged in the traditional cost-benefit analysis (CBA) that is required for impact assessments. What this research shows is the importance of understanding the value of improvements in ecosystem services and how this might be better incorporated into the policy and spatial management of fisheries and conservation of the marine environment. Current decision-making tools that rely on CBA must be able to better reflect the wider and long-term value to society of taking decisions around nature conservation and be transparent assumptions as well as about who costs and benefits accrue to. Issues around the distribution of costs and benefits, the acknowledgement of uncertainty and data gaps, as well as the difference between short-term and long-term outcomes must also be addressed transparently. While there is utility in developing a model of this kind, it is important to recognise the *indicative* nature of its findings. A political focus on the short-term financial impacts of decisions about nature adds biases towards decisions that put private gain ahead of public benefits.

Ultimately, the complexity and diversity of MPAs, seabed habitats, ecosystem impacts, and fishing activity (as well as the technical aspects of fishing gear configuration) means that no model is ever going to achieve a complete account of ecosystem service/socioeconomic impact. Nevertheless, the estimations here are a useful guide for discussing and acknowledging overlooked ‘value’ generated for society overall from a bottom-contact fishing ban in the long term. Looking forward, it is possible for the assumptions and parameters underpinning the model to change as better, more up-to-date information becomes available or when there is data for a context-specific site, as undertaken in the case studies presented in Section 4.

Aggregating accurate data from all the MPAs in Europe (once management measures have been implemented and impacts have been documented and verified) would make it possible to ground-truth and update this model.

Key insights and recommendations stemming from this research cover various different themes.

Strengthening the Marine Protected Areas evidence base

- Marine Protected Areas (MPAs) can play an important role in helping to restore marine habitats and supporting environmental, social, and economic benefits to people. MPAs are not the only tool to manage fisheries (and in many instances they are not as effective versus input or output controls including quota and effort limits), but in cases where seabed habitats underpin important ecosystem service benefit flows, then MPAs that restrict the types of fishing that negatively impact those services can be effective. Restricting bottom towed fishing gear allows these benthic habitats to recover and the benefits to accrue over time, whereas the costs to industry are short term and immediate. This presents a problem for decision-makers, who need to consider the long-term benefits to society but are trapped in a short-term economic paradigm that biases them against the long term. Revealing these costs and benefits is crucial to good policy-making in the best interests of society.

- The economic costs of implementing a trawl and dredge ban outweigh the benefits in terms of annual net impact early on; however, by year 5 the beneficial ecosystem service impacts become increasingly more pronounced. The highest annual net impact value is observed after 13 years (€615 million). Decision-making on this basis leads to short-term costs to some groups (fishers) for long-term benefits (to society), focussed on future generations of fishers and citizens, which will not necessarily be comparable with the immediate cost to fishing vessels and businesses that are excluded from these areas.

- MPAs are only one tool in the box. Fisheries management can focus on fishing effort (days or hours at sea), tonnage (quotas), temporal limits (fishing seasons or set hours), and spatial limits (gear restrictions or MPAs for instance). Therefore, MPAs should not be considered the primary fisheries management tool. One case study in the Kattegat has shown, a combination of fisheries management tools is needed where multiple outcomes (eg cod stock recovery, bycatch reduction, and habitat conservation) are sought. Global and European meta-research comparing different fishing gears across a range of objectives shows that for the same resource (eg cod stock), the more passive the gear, the better the overall performance. These studies include general comparisons of gear types, gears in deep-water, Vulnerable Marine Ecosystems, gears in the Russian Barents Sea fishery, gears in the UK cod fishery, gears in the UK seabass fishery, and also two studies that cover Nephrops fishing in Scotland, Portugal, and Sweden.[300,301,302,303,304,305]

Developing the field of valuation studies for EU MPAs

- Ecosystem service valuation is considered widely to be a tool to improve societal choices through presenting the costs of ecosystem degradation and the benefits of restoration. The distribution and time horizon of the costs and benefits are a crucial part of informing decision-making. Discounting the future forces myopic thinking, which biases decisions against the long term due to short-term economic concerns.

- All investment decisions and interventions involve trade-offs. The valuation of ecosystem services is a step towards more inclusive decision-making by making these trade-offs explicit and comparable in monetary terms. A full valuation of the wide array of services provided by marine ecosystems would enable decision-makers to better understand and compare trade-offs, but these are reliant on high-resolution data (both ecological and financial). Beyond this, there is, however, the infinite and unvaluable existence of nature. This cannot be captured (nor should it be) by economic tools.

- CBA modelling undertaken indicates there are long-term benefits of ecosystem services value arising from a bottom-contact fishing ban in Europe's offshore Special Areas of Conservation (SACs), particularly from supporting services such as bioremediation of waste and nutrient cycling as well as regulating services like carbon sequestration. These benefits are essential for a functioning ecosystem and society, and while the benefits can be valued and monetised, allowing the continued degradation poses a risk to society that goes way beyond an economic impact. Frequently, methodological concepts such as 'replacement costs' are used (ie what would it cost to replace these services with human labour) but that does not mean these functions *are actually replaceable* by human labour.

- For some ecosystem services changes, directly observable socioeconomic impacts may take place (eg increases in fish populations and landings by other gears, or increased biodiversity leading to increased tourism via angling or diving for instance). For others there is no cashable benefit which occurs (eg improved nutrient cycling); however in the absence of improved management the alternative situation (eg marine dead zones) would have economic impacts. Distinguishing between those impacts that can be captured in rents versus those that are valued to indicate their importance is a key distinction to be aware of when advocating the value of a mobile bottom-contact fishing ban.

- The approach undertaken here represents a low-cost, rapid benefits-transfer approach to produce indicative values. To strengthen insights, findings should not be used in isolation rather alongside deliberative work with stakeholders, scientists, managers, and interested parties.

Assessment tools for socio-economic impacts of MPAs

- MPAs throughout Europe's offshore marine area vary in terms of habitats and species. Displacement, tourism benefits, and such are all hard to deduce from this macro picture in a way that is practical at the individual site level. Therefore, there is a need for more studies and standardised data collection and valuation methods, as shown by the case study examples. Fisheries and marine management authorities, in conjunction with scientists and industry partners, need to monitor the impacts over time and develop tools to assess the actual ecosystem services changes. These can then be compared to the modelled benefits to determine the levels of 'goodness of fit' or to ground-truth predicted impacts through CBA modelling.

- Trade-offs exist at many levels, and tools such as Multi-Criteria Decision Analysis (MCDA) have been effectively used to reveal these trade-offs. Using MCDA and incorporating deliberation with representatives from sea-users and other members of society may provide a step towards changing decision-making.

- Regarding displacement, there is a need to ensure this is evidence based and a need to agree methods for assessing it. For example, if it is about compensation, then fishers will give high estimates, as previously seen in the case of windfarm development and if too low then it might not actually reflect the impact it has on fishing businesses. Another example relates to potential displacement of static gear into the MPA where trawling is excluded, understanding any economic benefits as well as the potential impacts on cetaceans are important to determine.[306]

APPENDIX A

Descriptions of valuation methods (extracted from the Ecosystem Services Valuation Database).

Valuation method	Approach	Application to ecosystem services
Contingent Valuation	Ask people to state their willingness to pay for an ecosystem service through surveys	All ecosystem services
Damage Cost Avoided	Estimate damage avoided due to ecosystem service	Ecosystems that provide storm, flood or landslide protection to houses or other assets
Market Prices (Gross Revenue)	Prices for ecosystem services that are directly observed in markets	Ecosystem services that are traded directly in markets
Net Factor Income (Residual Value; Resource Rent)	Revenue from sales of ecosystem-related good minus cost of other inputs	Ecosystems that provide an input in the production of a marketed good
Production Function	Statistical estimation of production function for a marketed good including an ecosystem service input	Ecosystems that provide an input in the production of a marketed good
Replacement Cost	Estimate the cost of replacing an ecosystem service with a man-made service	Ecosystem services that have man-made equivalents
Travel Cost	Estimate demand for ecosystem recreation sites using data on travel costs and visit rates	Recreational use of ecosystems
Value Transfer (Benefits Transfer)	Estimate the ecosystem services value for a "policy site" using existing information from a different "study site(s)".	All ecosystem services

Source: Brander, L.M., van Beukering P., Balzan, M., Broekx, S., Liekens, I., Marta-Pedroso, C., Szkop, Z., Vause, J., Maes, J., Santos-Martin F. and Potschin-Young M. (2018). Report on economic mapping and assessment methods for ecosystem services. Deliverable D3.2 EU Horizon 2020 ESMERALDA Project, Grant agreement No. 642007)

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