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Trade-offs between fisheries, offshore wind farms and marine protected areas in the southern North Sea – Winners, losers and effective spatial management

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ABSTRACT

There is an increasing need for marine spatial planning in the North Sea given the multiple uses with competing objectives. Plans to increase renewable energy production by establishing offshore wind farms (OWFs) are likely to coincide with existing and planned marine protected areas (MPAs), with obvious incompatibilities relating to conservation goals. Both will restrict fishing activities to varying degrees, thus a framework is needed to assess possible trade-offs to differing stakeholders and ecosystem health. Using a spatially-explicit trophic model, ecosystem response to different types of spatial closures to fisheries was evaluated using a variety of indicators relating to ecosystem health and fisheries productivity in the southern part of the North Sea. Additionally, hypothetical MPAs designated with specific ecological objectives in mind were tested. Scenario outcomes suggest that closures may need to be accompanied with additional fisheries management measures to avoid unintended negative impacts outside the closed areas. Furthermore, size and placement of spatial closures are important factors influencing overall benefits and losses in terms of ecological health and fisheries yield. One particular hypothetical large-scale closure, designed with the goal of protecting areas with high biodiversity, performed better in terms of indicators and trade-offs than the more fragmented, currently planned and existing closures. Although model outcomes have to be treated with care, the spatially-explicit food web modeling approach will likely aid in providing a more holistic evaluation of trade-offs between conservation objectives and fishing activities, which should contribute to a more target-oriented framework for the evaluation of closed areas.

1. Introduction

Worldwide, human pressures on marine ecosystems have increased. This is particularly true for the North Sea where multiple human impacts cumulatively affect the ecosystem [52] and where especially fish stocks are heavily impacted by climate change [43]. Fisheries, aquaculture, ship traffic, renewable energy installations, oil and gas platforms, as well as tourism, are some of the most noticeable forms of usage of the marine environment [6]. The various uses compete for space with environmental protection and conservation interests, which can create conflicts among stakeholders and require the identification and quantification of usage trade-offs [46,66,84]. Solving such trade-offs is a key challenge addressed by the maritime spatial planning directive, which was legally adopted by the European Union (EU) in 2014 [34]). Important examples are trade-offs between fishing activities and increased ocean space requirements for offshore wind farms (OWFs), but also conservation measures like marine protected areas (MPAs).

The protection of marine ecosystems, especially through MPAs, has moved into the focus of international legislations and commissions (Appendix A Table A 1.1). Within the EU, multiple multinational legislative acts were enforced to regulate and protect the marine environment. The Habitat and Birds Directives address land-based as well as marine protection goals [28,33]. A Natura 2000 MPA network is being developed to protect the species listed in the annexes of both legal acts, promoting a network with good connectivity and including some degree of complete closure to other human activities [107]. In the marine realm, the Marine Strategy Framework Directive (MSFD) was adopted to protect marine ecosystems and to achieve a good environmental status by 2020 (for definition, see Table 1). Furthermore, the MSFD calls for the creation of MPAs in affiliation with the Natura 2000 areas [32]. In

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Table 1
Ecosystem attributes protected by MPAs differentiated between habitat types and species groups. Last column shows the fleets that were excluded from fishing in the spatial management scenarios.

Type of ecosystem attribute	Ecosystem attribute	Gears and associated Ecospace fleets excluded
Habitat	Sandbanks/Mud flats and Sand flats	Bottom contacting gear (demersal trawl, beam trawl, shrimp trawl, Nephrops trawl, dredges)
Habitat	Reefs	Bottom contacting gear (demersal trawl, beam trawl, shrimp trawl, Nephrops trawl, dredges)
Species	Birds	Drift and fixed nets, Gears using hooks
Species	Cetaceans	Drift and fixed nets, Gears using hooks
Species	Benthic community	Bottom contacting gear (demersal trawl, beam trawl, shrimp trawl, Nephrops trawl, dredges)
Species	Fish (including lampreys)	Bottom contacting gear (demersal trawl, beam trawl, shrimp trawl, Nephrops trawl, dredges)
Species	Mammal protection site	Drift and fixed nets, Gears using hooks

May 2020, the EU adopted the new Biodiversity Strategy 2030 as part of the European Green Deal, which requires an increase of EU-wide protected areas and stricter protection measures of at least 30% of the marine area [26].

Implementation of MPAs in the North Sea has been a stepwise process. By the end of 2018, 496 MPAs were part of the OSPAR MPA network, primarily within territorial waters. At that time, MPAs covered 6.4% of the OSPAR area and 18.6% of the Greater North Sea. Even though these numbers suggest a certain progress, an ecologically coherent implementation of the MPA network is lacking [115,27]. The ecological coherence of an MPA is generally evaluated based on five principles; depiction of features, representativity across biogeographical regions, resilience through replication, management and the ability to support ecological connectivity [27]. This concept of ecological coherence has been addressed and analyzed in several EU waters through projects or legal bodies [14,40,1,67,10] and while there is a good representation of biogeographical regions, there is a strong need to create tools that are able to assess ecological coherence [115]. Without looking beyond individual MPAs and focus on the network concept, ecological coherence might not be achievable [42]. One of the five principles included in ecological coherence is ecological connectivity, which was defined by Taylor et al. [108] as the support or obstruction of animal movement between resources. Currently, spatial connectivity among the Natura 2000 areas is seen as insufficient because MPAs are still too patchy and often lack a common planning process, including the use of protected areas that span country boundaries [80]. Additionally, most OSPAR MPAs have, for example, publicly-documented management information, although only a small percentage of these measures have been implemented [27]. For example, Belgian Natura 2000 sites are designated and management plans are finalized, yet their implementation and enforcement is still blocked by legal issues [42].

OWFs are an important measure for climate change mitigation through substitution of fossil fuel-based energy production. Presently, European OWFs have an installed capacity of 22,072 MW, a majority (77%) comes from the North Sea. In 2019, 502 grid-connected offshore wind turbines were installed in 10 OWFs, and 99% of the turbines were built in the North Sea [117]. While restricting or rearranging fishing activities, OWFs have the potential to affect ecosystem structure and functioning in diverse ways [39,74]. Positive impacts can include increased nursery areas for key species supporting the fish community [105,59,89,91]. In contrast, OWFs can potentially have a negative impact on the seabird community through collisions or as habitat loss by avoidance [15,17,45]. Furthermore, they also likely modify the ecosystem through structural changes by adding hard substrate that can potentially increase the abundance of epifauna, like the bivalve *Mytilus edulis*, which in turn may impact ecosystem functioning [100].

Modern ecosystem-based management (EBM) reconciles the multiple interests people have in using and protecting the ocean and has more recently expanded to include information on potential trade-offs in the equitable use of the marine space [35,76]. Indicator systems are important means for the evaluation of goals within EBM and may include single species measures, group indicators (e.g. pelagic vs. demersal, or invertebrates vs. fish species), ecosystem-level indicators (e.g. trophic-level based indicators) or conservation-based indicators (e.g. species richness or conservation status; [48,13,24,94,85]). An impact assessment of spatial management measures with regard to such indicators requires specialized tools, such as trophic models that consider spatially-explicit ecological processes. The models can be "end-to-end" ecosystem models [3,50,89] and multi-model ensembles [99] that can be utilized to support spatial management [104].

In this work, an existing spatially-explicit trophic model of the southern part of the North Sea was used to evaluate the potential impacts of existing and planned MPAs and OWFs on the system. Various biomass and catch-based indicators are compared among different closure types to identify changes in the state and functioning of the ecosystem. Based on the intention of achieving an MPA coverage of up to 30%, as it is proposed in the new Biodiversity Strategy 2030 (EU, 2019), two additional hypothetical protected areas that exclude fisheries were created, to test additional spatial management scenarios that are not impacted by national constraints as in the current Natura 2000 framework. One of the two additional closure scenarios is based on the core distribution of endangered species on the International Union for Conservation of Nature's Red List of Threatened Species (IUCN red list), while the second aims to protect an area with high biodiversity. Furthermore, the necessity of additional effort reductions to counteract the effect of spatial effort re-allocation due to closures was tested. By comparing scenario outcomes relative to each other, the model results provide insights into possible trade-offs and win-win situations in the spatial implementation of conservation measures, fisheries management and renewable energy production in general.

2. Material and methods

2.1. Modelling approach

One modelling software that is increasingly being used to evaluate anthropogenic influences on ecosystems is Ecopath with Ecosim (EwE) that comprises three interdependent modules representing (i) the static, mass-balanced Ecopath model, (ii) the temporal simulation component Ecosim, and (iii) the spatial implementation Ecospace [20,22]. All three model components were developed for the southern part of the North Sea in previous studies ([103,102,88], see Appendix B Figure B1 for food web diagram), representing International Council of the Sea (ICES) management areas 4b and 4c. Sixty-eight functional groups were defined, with a focus on commercially important higher trophic level species. Nonetheless, the 35 multi-species and 30 single-species functional groups comprise all trophic levels: mammals and birds (4), elasmobranchs (8), fish (35), crustaceans (4), benthic invertebrates (8), zooplankton and phytoplankton (6). Additionally, three groups represent particulate / dissolved organic matter and fishery discards. Different life stages with specific trophic needs were implemented for several commercially-important species applying the multi-stanza approach [114]. Exploitation is depicted by twelve fishing fleets, which represent the most prominent fisheries in the southern part of the North Sea.

The EwE Ecospace component is a two-dimensional model to test spatial management measures. It is partitioned into grid cells in which temporal dynamics derived from Ecosim are executed [113,22]. To determine the spatial distribution of functional groups, habitat suitability's and dispersal rates can be defined for model grid cells. Dispersal rates represent the fractions of biomass of each functional group dispersing at a given rate (km/year), with a base dispersal rate of 300 km/year [22]. For the model of the southern part of the North Sea, five different dispersal rates were chosen (1000–600–300–30–3) based on the functional group's life form (for a detailed list see Appendix A of [88]). Furthermore, a habitat foraging capacity model (HFCM) allows for the definition of the degree of habitat suitability in each cell. Habitat suitability for the most important and common functional groups were driven by distribution maps created with species distribution models. These were implemented and updated in 5-year steps during simulation of historical years to account for shifts in species distribution over time. Distributions of functional groups for which insufficient data existed to create species distribution models were driven by habitat properties based on sediment structures, distance to coast and water depth [88]. Model spin-up (burn-in period) was conducted for 10 years, executed for the historical period fitted to observational data until 2010 and closures were effective from 2011 onwards. The model was then run to equilibrium for 40 years followed by an additional 10 years which were used for evaluation. For a more detailed description of the Ecospace model, including the implemented species distribution maps, please see [88] and for the changes that were applied to the model for this study the Supplementary material.

2.2. Spatial closures and scenarios

2.2.1. Existing and planned closures

Ecospace allows for implementation of MPAs to test varying management strategies. Thirty-three currently designated MPAs that are located within the study area were incorporated (retrieved from [109], Fig. 1). Details on these MPAs, including their legal basis (e.g. Habitat Directive, Birds Directive, MSFD), are listed in Annex I. Specific management objectives were applied to each MPA individually, which can be fleet and time specific closures (year-round closures or just certain months). In practice, the majority of the defined MPAs lack enforced management and some completely lack definition of their restrictions. Hence, closure restrictions were defined based on the ecosystem attributes they aim to protect (Table 1) and translated these into the exclusion of specific gears in the MPAs: 1. all bottom-contacting gears (to avoid seafloor disturbance), 2. all static gears posing a threat to birds and mammals (drift/fixed nets and gears using hooks) or 3. both gear groups. Fleets fishing mainly within the water column, i.e. pelagic fisheries or low impact gears like pots were allowed to continue fishing, as they do not pose an imminent danger to the protected ecosystem attributes listed in Table 1. A detailed list of the protected ecosystem attributes for each MPA is presented in Appendix A Table A1.2 which is based on the European Nature Information System (EUNIS) and the declared protected habitats and species of the Habitats Directive [28].

The closure of OWFs is regulated differently within the Exclusive Economic Zones of Denmark, Germany, Belgium, Netherlands and the United Kingdom (UK). While Belgium, Germany and the Netherlands prohibit fishing within the OWFs [77], Denmark and the UK exclude trawling but allow other gears. Yet, it is reported that fishers generally try to avoid the OWFs [49]. For testing the effectiveness of OWFs as marine protection sites, all OWFs were closed for the entire year for all fishing gears. Two developmental stages of OWFs based on data retrieved from OSPAR [2] were included, the currently operational OWFs (from here on referred to as OWF_{op}) as well as planned and designated OWFs (OWF_{pla}; status in the beginning of 2020, Fig. 1).

The implementation of OWFs not only causes changes in spatial usage. OWF installation also alters habitats and creates additional hard substrate. Hard substrate has the potential to act as an artificial reef, likely affecting the ecosystem and, in particular, benthic functional groups [29]. To consider these impacts in the model, specific habitat layers based on OWF_{op} and OWF_{pla} were developed, whereby each grid cell contains the percentage of gained hard substrate based on turbines relative to the entire area of each cell. Unfortunately, detailed information on the type of turbine substructures were not available for all OWFs. Yet, since the majority of turbine substructures in the North Sea are monopiles [110] the mean diameter of monopiles in the North Sea was used for calculating the lateral surface of the turbines [83]. Studies have shown that the biggest increase in biomass at the turbines can be found within 1 m above ground and among the riprap, i.e. protective rubble at the base that is used to prohibit erosion from scour [69]. The lateral surface of 1 m of the turbine was aggregated with the area covered by riprap, multiplied this with the number of turbines in the area and calculated the percentage gained in comparison to the area of the entire grid cell. Affinities to these new habitats were assigned to five functional groups: (i) *large crabs*, (ii) *epifaunal macrobenthos (mobile grazers)*, (iii) *shrimps*, (iv) *small mobile epifauna (swarming crustaceans)* and (v) *sessile epifauna*. This is in line with Lynam et al. [78], which was also used as reference for the affinity of these benthic groups towards different sediment types (see Appendix A1 Table 1.3 A for detailed information). Other possible impacts caused by the implementation of OWFs, such as noise and vibration, disturbance and damage caused during the construction phase or effects such as connectivity sprawls [12] have not been included in this analysis.

2.2.2. Additional ecological-based closures

Additionally, two hypothetical closures were tested to evaluate their impact on the ecosystem as compared to the presently-designated MPAs. A first closure had the aim to protect areas of highest ecosystem diversity (i.e. Kempton's Q). This version of the Kempton's Q index was modified for use with EwE output to represent diversity among functional groups rather than on species level [4]. Ecosystem stability is assumed to increase with higher diversity, therefore the core area of higher Kempton's Q indices should identify the region with higher stability that should be protected [81]. A second hypothetical closure aimed to protect highest biomass of endangered species, as defined by the IUCN "Red List", which includes species categorized as "near threatened", "vulnerable", "endangered" or "critically endangered" [64]. This category applies to 21 species within the southern North Sea model, most of which are birds, elasmobranches and some fish species (for a complete list see Appendix A, Table 2.1 A). Based on the spatial patterns derived from the equilibrium baseline run, regions with values in the 75th percentile were identified as core areas for the Kempton's Q- and IUCN-based MPAs. In order to meet the 30% protection goal formulated by the Biodiversity Strategy 2030, a buffer zone was created around the core areas until 30% of the model domain was achieved (Fig. 1). For both closures, bottom contacting gears and static gears were excluded, similar to the designated MPAs. Furthermore, operational OWFs were closed to all fisheries, as they are already present.

Kempton's Q index as a diversity indicator displayed a slightly fractured pattern. This fragmentation is especially apparent in the northwestern part of the study area near the British coast, where high biodiversity was found around the 50 m depth line (compare with depth isolines in Fig. 2). The area is located off the northern coast of England and the south of Scotland and spans all the way over to the Danish coast. In contrast, the biomass of the IUCN-endangered species showed highest concentrations in the German Bight and along the British coast below 53° N towards the English Channel, similar to the pattern of total biomass. Small demersal fish (e.g. *Cyclopterus lumpus*), bird groups (e.g. *Larus argentatus*) and turbot were the main contributors to high biomass concentrations of IUCN-endangered species in the German Bight while elasmobranch groups (e.g. *Raja clavata* and *Squalus acanthias*) and toothed whales (*Phocoena phocoena*) were the main contributors along the British coast. With the exception of one small outlying area, this core area is one continuous area that serves as outline for the IUCN-based closure. In conclusion, the two different conservation goals led to a high contrast in the location of closed areas in these hypothetical scenarios and are therefore suitable to test resulting impacts on model outcomes.

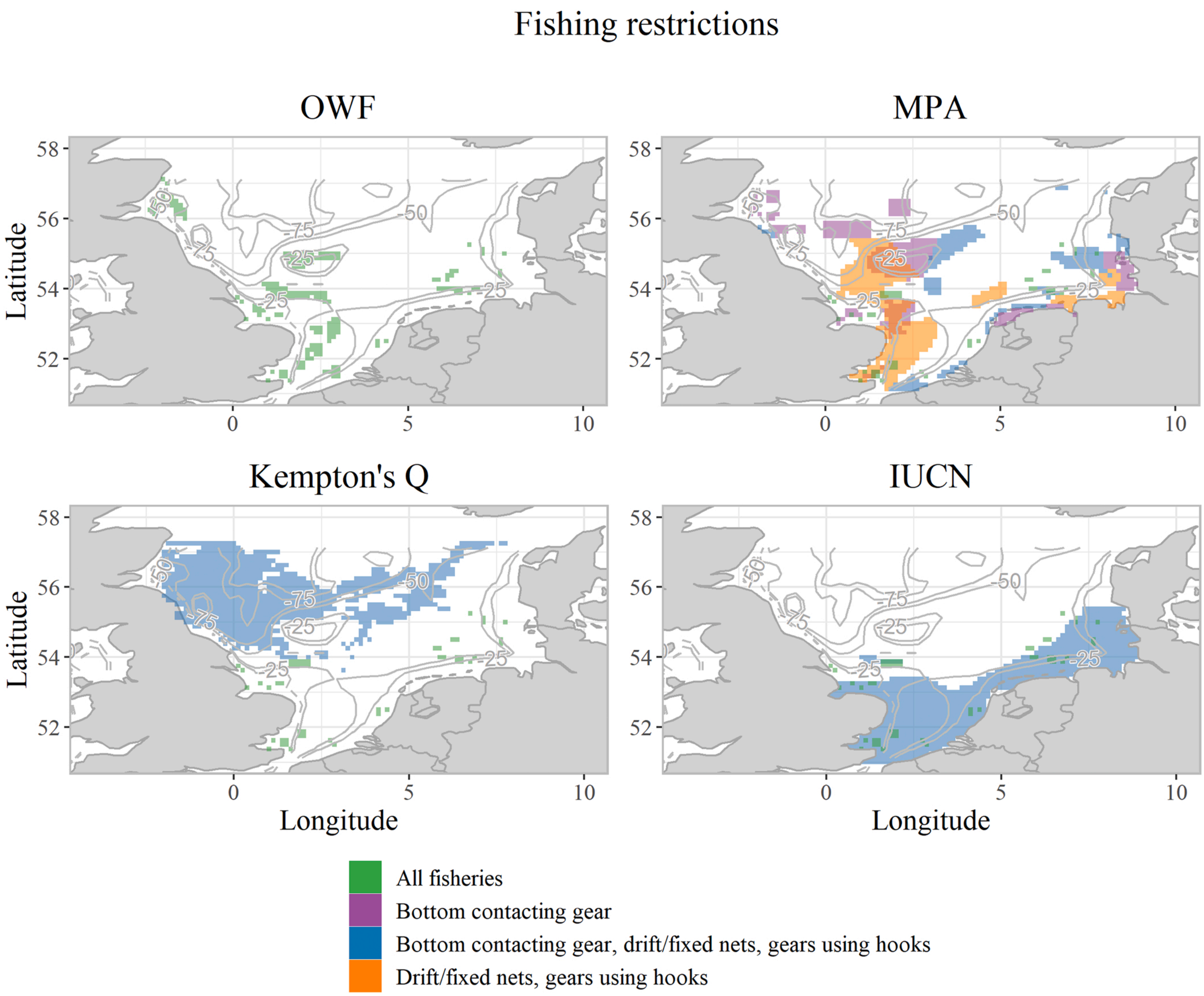


Fig. 1. Four different closures tested in this study. Each closing state contains the existing OWFs since they are currently the only closures that are in place. Color type depicts the type of fishery that was excluded. Green was closed for all fisheries, which affected up to 6.3% of the total area when operational and planned wind farms were closed. Marine protected areas were closed based on protected ecosystem attributes. Three types of exclusion: i) all bottom contacting gear (11.4% closure), ii) all bottom contacting gear as well as passive gears (6.7% closure), iii) all passive gears (14.4% closure). Kempton's Q and IUCN areas were closed for bottom contacting and passive gear.

2.2.3. Scenarios in fishing effort reduction

Closing areas to fisheries does not change the total effort in the model, but rather results in a redistribution to areas outside the closure. In Ecospace, effort is allocated via a "gravity model", where effort is distributed in relation to the net benefits gained by exploitation of a certain region. In particular, when closures are large enough, effort may increase substantially along the edges outside closed areas due to spill-over effects [22]. The effect of an additional effort reduction was tested in scenarios with closures > = 30% of the total area (MPA, Kemptons and IUCN, Table 2). In these scenarios, effort distributions for each fleet in the baseline reference run were overlaid with the three different types of area closures. Effort within cells in the baseline reference identified as impacted by closures was summed up and divided by the total fishing effort. Subsequently, the effort of 2010 was reduced by the percentage and implemented as effort for 2011 onwards in Ecosim (Appendix A3 Table 3.1 A). Overall, a set of seven scenarios combining closure types and effort levels was tested (Table 2).

2.3. Trait-based indicator approach to evaluate trade-offs

Spatial management scenarios were evaluated using a trait-based indicator approach [7]. The EwE ECOIND plug-in calculates ecosystem indicators based on functional groups included in the model [24]. Prior to applying ECOIND, traits needed to be assigned to the species in each model functional group. The entire species list includes the 410 species that were used to construct the functional groups in the original model [79]. This study focused on traits related to ecology, conservation and exploitation (Table 3). Biomass contribution of each species to the associated functional group were then calculated based on their mean relative occurrence in the ICES International Bottom Trawl Survey (IBTS) and the ICES Beam Trawl Survey (BTS) in the period 1991–1995. For functional groups that were not represented sufficiently within the surveys, like benthic or planktonic groups, the biomass contribution was kept equal for all species. Catches were assumed to have the same relative species contributions inside the functional groups as in the food web.

Indicators calculated via ECOIND can be split into five groups, four of

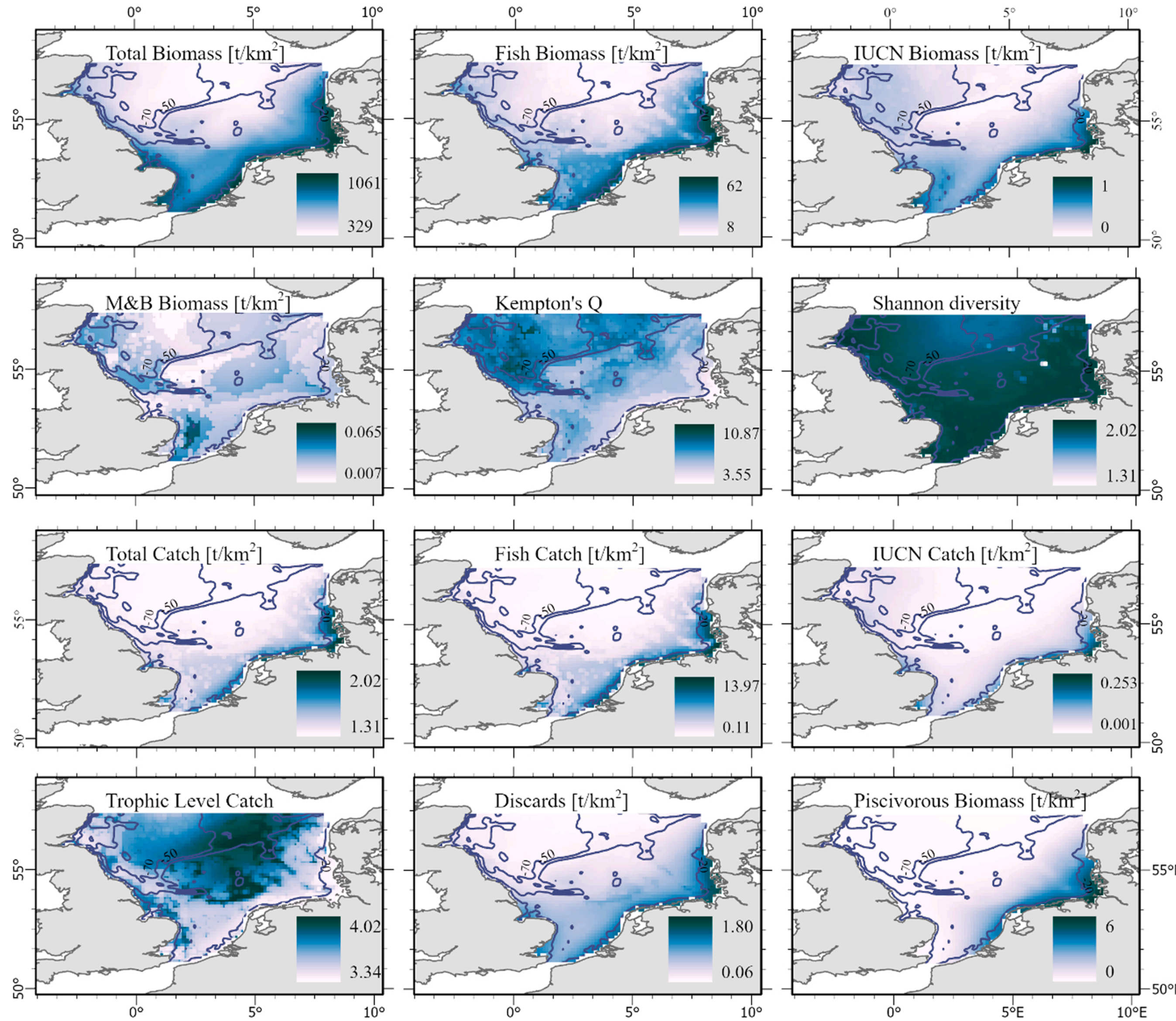


Fig. 2. Baseline spatial distribution of selected ecological indicators in the southern part of the North Sea according to the Ecospace model.

which were tested in this study: 1) *Biomass-based indicators* representing the standing stock in the ecosystem including total biomass, but also separate biomass per species group (i.e. fish and invertebrates), habitat (i.e. pelagic vs. demersal) and exploitation status (i.e. for commercial species); 2) *Catch-based indicators* representing catch and consequent discards, similarly to biomass for total catch or to subgroups like organism and ecology; 3) *Trophic-level based indicators* that refer to the position of the species in the food web; 4) *Species-based indicators*, of which some refer to the International Union for Conservation of Nature (IUCN) Red List of species at risk [65]. *Size-based indicators* were excluded in this study due to the course resolution of EwE with regard to length and age. Additionally, an indicator termed “Large Piscivorous” was calculated, consisting out of predators mainly feeding on fish. This includes birds, mammals, sharks and rays, as well as fish species with an $L_\infty > 50$ cm. We decided to use “Large Piscivorous” as indicator rather than a Large Fish Indicator ([99,44]) to include all piscivorous predatory species, including mammals and birds, and not to restrict it to fish species. Table A2.2 in Appendix A displays which species are included in the calculation of all ecological indicator, including the additional one.

For each biomass-based and catch-based indicator the mean values for the entire area, within closures, and outside closures were calculated and changes relative to the baseline run were evaluated. Furthermore, a comparison of effort distribution was conducted to assess the effect of each closure on fishing activity. Eventually, trade-offs in terms of positive and negative impacts (I_t) were evaluated. All indicators were analysed relative to the baseline and labeled according to its scenario (s) and trend (t) (i.e. $rel\_change_{s,t}$). Second, for each scenario and trend combination, these relative changes were summed up ($SRC_{s,t}$) and divided by the maximum relative change per trend ($maxSRC_t$):

$$SRC_{s,t} = abs\left(\sum_{i=1}^{n} rel\_change_{s,t}\right) \tag{1}$$

$$I_t = (SRC_{s,t})/\max(SRC_t) \tag{2}$$

3. Results

3.1. Characterizing the ecosystem in the southern part of the North Sea

Prior to assessing the effects of closures on the ecosystem, an overview of the composition and spatial structure of the food web in the southern part of the North Sea based on ecological indicators derived from the Ecopath mass-balance in 1991 was provided (Table 1.1B in

Table 2
Scenarios defined by closure type, excluded fisheries and gears, with different closure states and size of closure (abbreviations given in the text).

Scenario[a]	Excluded fisheries and gears	Size of the closure (% of study area)
0. Baseline	None	0
1. OWFop + OWF pla (OWF)	All fisheries	6.31
2. MPA + OWFop (MPA)	Bottom contacting gear, passive gear or both (MPA) + All fisheries in OWFop	31.6
3. Kempton's Q + OWFop (Kempton)	Bottom contacting and passive gear + All fisheries in OWFop	30.8
4. IUCN + OWFop (IUCN)	Bottom contacting and passive gear + All fisheries in OWFop	30.3
5. MPA + OWFop + Effort reduction (MPA_red)	Bottom contacting gear, passive gear or both (MPA) + All fisheries in OWFop + Effort reduction equal to effort before closure	31.6
6. Kempton's Q + OWFop + Effort reduction (Kempton_red)	Bottom contacting and passive gear + All fisheries in OWFop + Effort reduction to effort before closure	30.8
7. IUCN + OWFop + Effort reduction (IUCN_red)	Bottom contacting and passive gear + All fisheries in OWFop + Effort reduction equal to effort before closure	30.3

[a] All scenarios were run with a 10-year spin-up (burn-in period), followed by the model time period 1991–2010 and then run to equilibrium for 40 years followed by an additional 10 years used for evaluation

Table 3
Traits assigned to the species in the EwE model for the southern part of the North Sea.

Trait	Categories
Organism type	Mammals, birds, fishes, invertebrates, algae
Ecology	demersal (bathydemersal, benthic, benthopelagic), pelagic (bathypelagic, pelagic-neritic, pelagic-oceanic), land-based
IUCN status	Not evaluated, data deficient, least concern, near threatened, vulnerable, endangered, critically endangered

Appendix B includes all indicators). The vast majority of the total biomass (606.5 t/km^2) was composed of invertebrates (525.4 t/km^2) and only 3.6% was fish biomass (21.5 t/km^2). Over 90% of the total catch (5.9 t/km^2) in the system was composed of fish species (5.3 t/km^2).

Spatial patterns for the various indicators as derived from the baseline run without fisheries closures are presented in Fig. 2. Almost all indicators, including those shown in Appendix B, displayed lower values for deeper (i.e. central and northern) parts of the study area. Total biomass followed the depth pattern in the southern part of the North Sea with highest values at the coasts of Denmark, Germany, Belgium and the Netherlands, as well UK in the south towards the English Channel. Fish biomass was similarly distributed to total biomass, although patchier. Total catch was also concentrated along the coastlines following the biomass distribution.

3.2. Evaluation of closure scenarios– biomass-based indicators

Overall, excluding fisheries from pre-defined areas induced only small, but mostly negative changes in biomass-based indicators (Fig. 3). Negative impacts outside outweighed the positive effects inside the closed areas. Among all scenarios, the IUCN scenario seemed to have the strongest overall impact. Total biomass and Shannon diversity hardly changed in any of the scenarios, while the biomass of mammals and birds, as well as Kempton's Q and biomass of large, piscivorous predators, were most negatively affected. Inside the closed areas the most notable increase was for fish biomass in the IUCN scenario (approx. 25%). Next to the fish biomass, the biomass of IUCN-endangered species increased inside closed areas most noticeably. Outside the closed areas biomass-based indicators generally showed negative impacts, with the largest impacts in the IUCN scenario (decreases up to −15%). Only total biomass increased outside the closed areas due to food web effects (see below), with a maximum of 2% in the IUCN scenario. Changes in trophic levels of the community were small (maximum 0.5% change) (see plots in Appendix B2).

The overall negative impact of the scenarios on the biomass of mammals and birds is counter-intuitive, but relates to the inclusion of the functional group *surface-feeding seabirds*, which are a contributor to this indicator's biomass (8% in the baseline run). For this functional group, ~47% of the diet is based on discards. When reducing fishing opportunities, this decreases the availability of this food item, which results in a decrease in the biomass of surface-feeding seabirds. Biomass of large piscivorous decreases within the closures in the OWF and MPA scenario, while they increase in the IUCN and Kempton's scenario. The main driving functional group here is *Whiting (adult)*, which contributes ~27% to the biomass of the large piscivorous indicator.

Shifts in spatial distribution patterns were detected for most biomass-based indicators (for all indicators see Appendix B). The distributions of the overall biomass (primarily invertebrates) and fish biomass showed contrasting patterns, reflecting trophic effects associated with the decreased predation mortality of invertebrates by fish as their biomass was reduced (Fig. 4). While the fish species increased inside the closed areas and decreased outside, the total biomass increased outside the closed areas because of increasing invertebrate biomass. When closing areas to fisheries, the total effort is re-distributed among the remaining fishing areas. This leads inevitably to an increased effort outside the closed areas, which primarily decreases the biomass of commercially-targeted fish species. Within the closed areas, the reduced fishing pressure led to an increase in fish biomass and therefore biomass of large piscivorous predators.

Size, coherence and location of the closed areas influenced the spatial distribution of fish and total biomass (Fig. 4). While closing the areas only for OWFs already led to visible changes in distribution, adding MPA closures further influenced the magnitude of change. Moreover, the evaluation of the scenarios based on the designated MPAs revealed that only the MPAs that excluded bottom-contacting gears had noticeable impacts on biomass distribution patterns in contrast to MPAs closed to passive gears only (Fig. 4, compare to Fig. 1). Despite their similar overall spatial coverage, the impact of the two hypothetical scenarios varied. While the effect of the Kempton's Q scenario is comparable to the scenarios including MPAs, the scenario based on the distribution of IUCN-endangered species had a much stronger impact. Removing the fishing pressure in the area of the IUCN scenario led to a substantial increase in fish biomass inside the area and a strong decrease outside, due to the redistribution in effort, which in turn affected the invertebrate biomass visible via trophic cascades (increasing outside and decreasing inside the closed area).

3.3. Evaluation of closure scenarios – catch-based indicators

The closure of fishing areas led to an overall decrease in catch, despite the same fishing effort as in the baseline scenario (Fig. 5). For the entire study area, the strongest decrease was detectable for the IUCN scenario followed by the two scenarios Kempton and MPA. For the IUCN scenario, the total catch decreased in the entire area by around − 22% and discards decreased by − 47% compared to the baseline scenario. In the Kempton as well as the MPA scenario the total catch decreased − 8% and − 7%, respectively.

When splitting the model area into inside and outside closed areas, most depicted catch-based indicators increased outside the closed areas in most scenarios due to the effort displacement, with a few exceptions, especially with the Kempton scenario. The OWF scenario shows the most

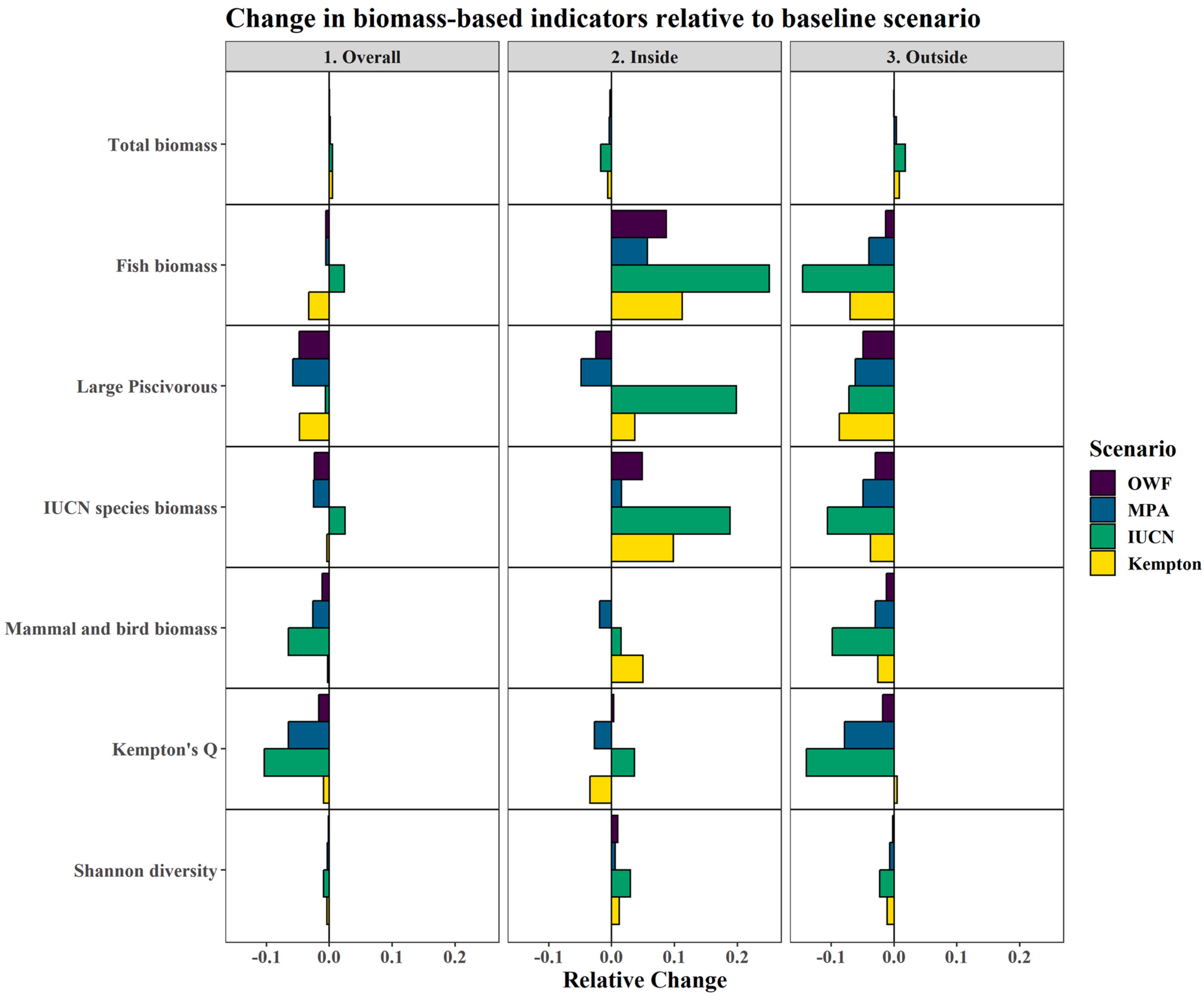


Fig. 3. Change in selected biomass-based indicators relative to the baseline scenario for the different closure scenarios. Left: Results for the entire southern part of the North Sea. Middle: Results inside areas with fishing restrictions in the different scenarios. Right: Results outside the areas with fishing restrictions in the different scenarios.

noticeable increase in total catch outside the closed areas (~ 6%), followed by the MPA scenario (~ 4%). Indicator values inside the closed areas reflected the applied closures, for example a 100% decrease for the OWF scenarios due to closures for all fisheries and much less in the IUCN scenario due to pelagic fisheries that are still allowed.

The distribution of fish catches shifted depending on scenario. Catch increased especially around the borders of the closed areas. Scenarios with larger area closures (closures including MPAs and the IUCN scenario) displayed areas with decreasing catch even outside the closed areas (Fig. 6). Overall, this shift in catches displayed the effect of effort reallocation. Outside the closed areas the fishing pressure increased, which led to a decrease in fish biomass, which in turn resulted in a lower catch in equilibrium. Furthermore, the effect of the closures applied to bottom contacting gears were more apparent than for the MPAs that only excluded passive gears.

The same shift is detectable when comparing the distribution of catch to one of the fleets using bottom-contacting gear, i.e. demersal trawls and seiners. Fishing effort is increasing around the edges of almost all closed areas, especially in the central and southern part of the modelled area. Again, the IUCN scenario stands out with the strongest shift in effort. In this scenario, the region with the greatest total catch values in the baseline equilibrium is being closed, therefore the area where the majority of the fleet effort was concentrated. Hence, this effort needs to be re-distributed, which resulted in the largest overall changes among scenarios.

These results showed that the size and location of the closed areas is crucial to reach an overall impact. While all three scenarios, MPA, Kempton's Q and IUCN close the fishing grounds up to 30%, the impact on catch-based indicators is quite diverse, highlighting that also the location and coherence of the closed areas is an important factor. Especially the difference between the IUCN and the Kempton's Q scenario is striking. While IUCN had a strong overall impact on the catch-based indicators, the effect of the closures in the Kempton's Q scenario had a much smaller impact. However, it has to be noted that all scenarios are subject to a different degree of closure and therefore a direct comparison is biased. While some areas are closed for all fisheries (OWFs), some are closed for all bottom contacting gears and static gears (Kempton's Q, IUCN and some MPAs) or just bottom contacting gears or static gears (some MPAs; compare Fig. 1).

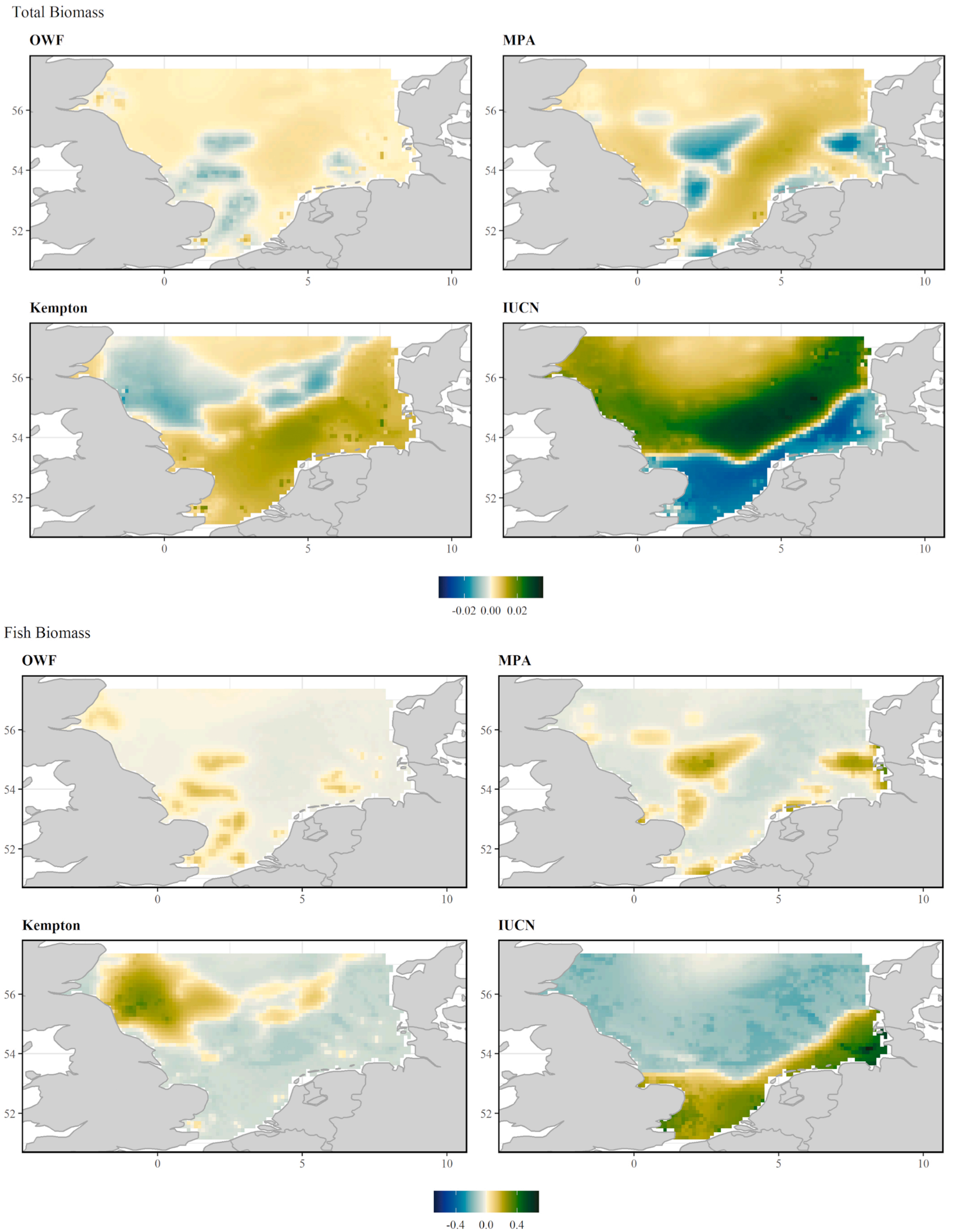


Fig. 4. Changes in total biomass (top) and fish biomass (bottom) by area. Changes are relative to baseline scenario with no closures with increases displayed by yellow to greenish colors and a decrease displayed by blue.

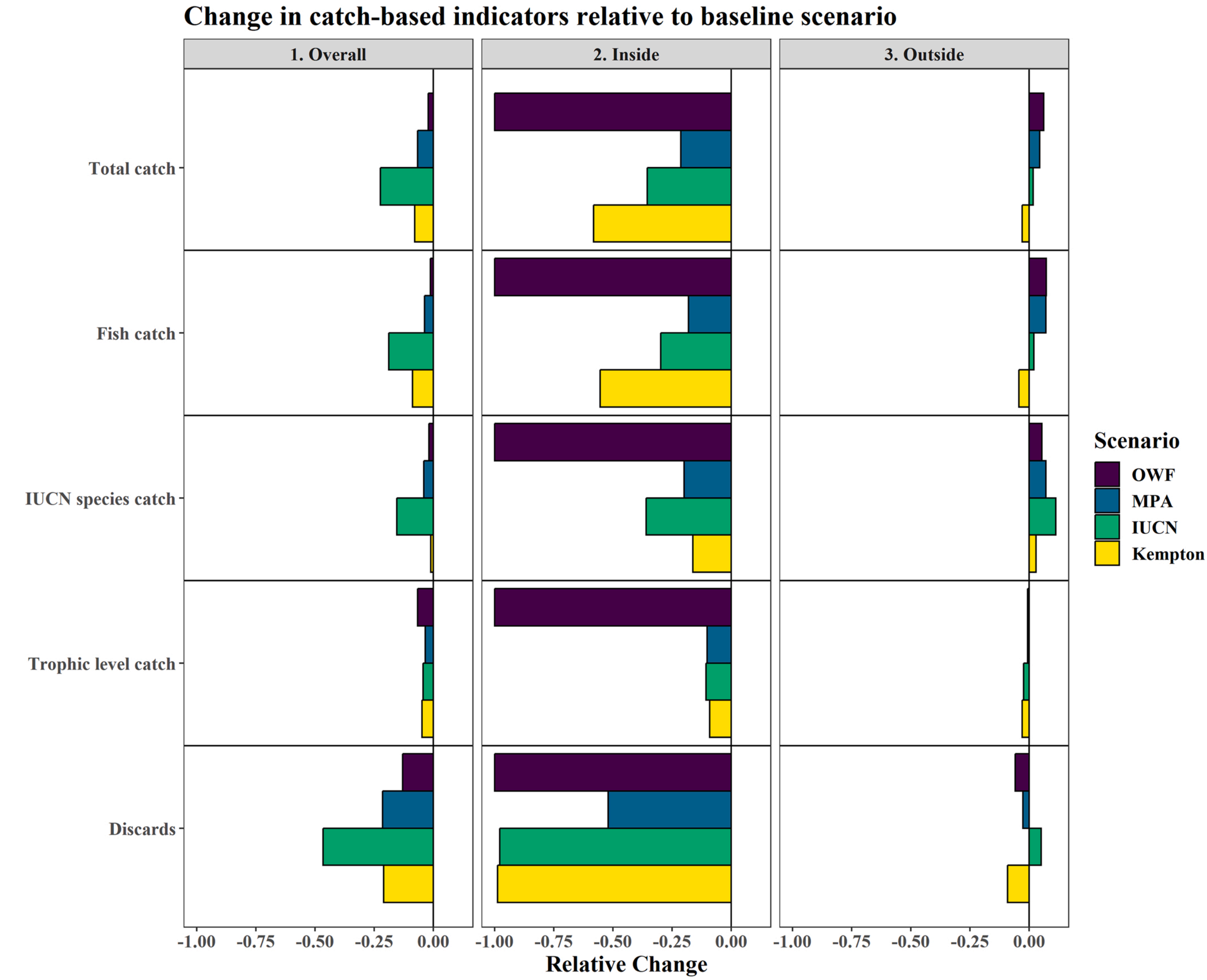


Fig. 5. Changes in selected catch-based indicators relative to the baseline run for the different closure scenarios. Left: Results for the entire model area. Middle: Results inside the closed areas. Right: Results outside the closed areas in the different scenarios.

3.4. Closure scenarios with additional effort reduction

Reducing fishing effort additionally to closing fishing areas led to an improvement of all indicators (Fig. 7). For the Kempton and MPA scenario, an overall reduction in fish biomass compared to the baseline scenario was turned into a small overall increase, while the overall increase in fish biomass in the IUCN scenario increased even further (~ 8%). In the case of the indicator describing the biomass of IUCN endangered species, the biomass now even increases slightly (~1%) outside of the closed areas in all scenarios. However, outside the closures the impact is still negative for the most indicators, even with the effort reduction.

For all scenarios, the reduction in fishing effort led to a decrease in all catch-based indicators compared to their associated scenarios (Fig. 8). Even outside the closed areas, all indicators now displayed a decrease in relation to the baseline run. The impact of the effort reduction was strongest for the IUCN as well as the MPA scenario. The overall strongest decrease compared to the baseline run was in discards in the IUCN scenario (−70%), while the total catch is reduced to − 32%. Therefore, positive impacts on certain food web components were traded off sometimes with strong negative impacts on economic components when reducing fishing effort.

3.5. Trade-offs

Fig. 9 shows a summary of the mean relative impacts in relation to the baseline run (Eq. 1) for all indicators across scenarios (including indicators only presented in Annex B2) enabling the comparison of the overall impact of all scenarios (Fig. 9). Evaluating the overall trade-offs, the great difference between the IUCN scenario, with and without effort reduction, and all other scenarios is again observed. While the IUCN scenarios led to the maximum sum in overall decreases compared to the baseline run, these scenarios also had the most positive impact on biomass and catch-based indicators. One important indicator that influenced this result is discards. For this evaluation, discard reduction is seen as a positive effect. Under both IUCN scenarios, the fisheries reduced their discards the most, from which the overall effect of these scenarios benefitted. The least overall negative impacts were calculated for the OWF and Kempton's Q scenario, but at the same time, the positive impacts were similar to the ones in the MPA scenarios.

Considering only the biomass-based indicators, the overall picture is changing especially in terms of negative impact. Now the IUCN scenario without the effort reduction seems to have the greatest negative impact. At the same time, the Kempton's Q scenario with effort reduction has the least losses and is equal in positive impact to the IUCN and MPA scenario

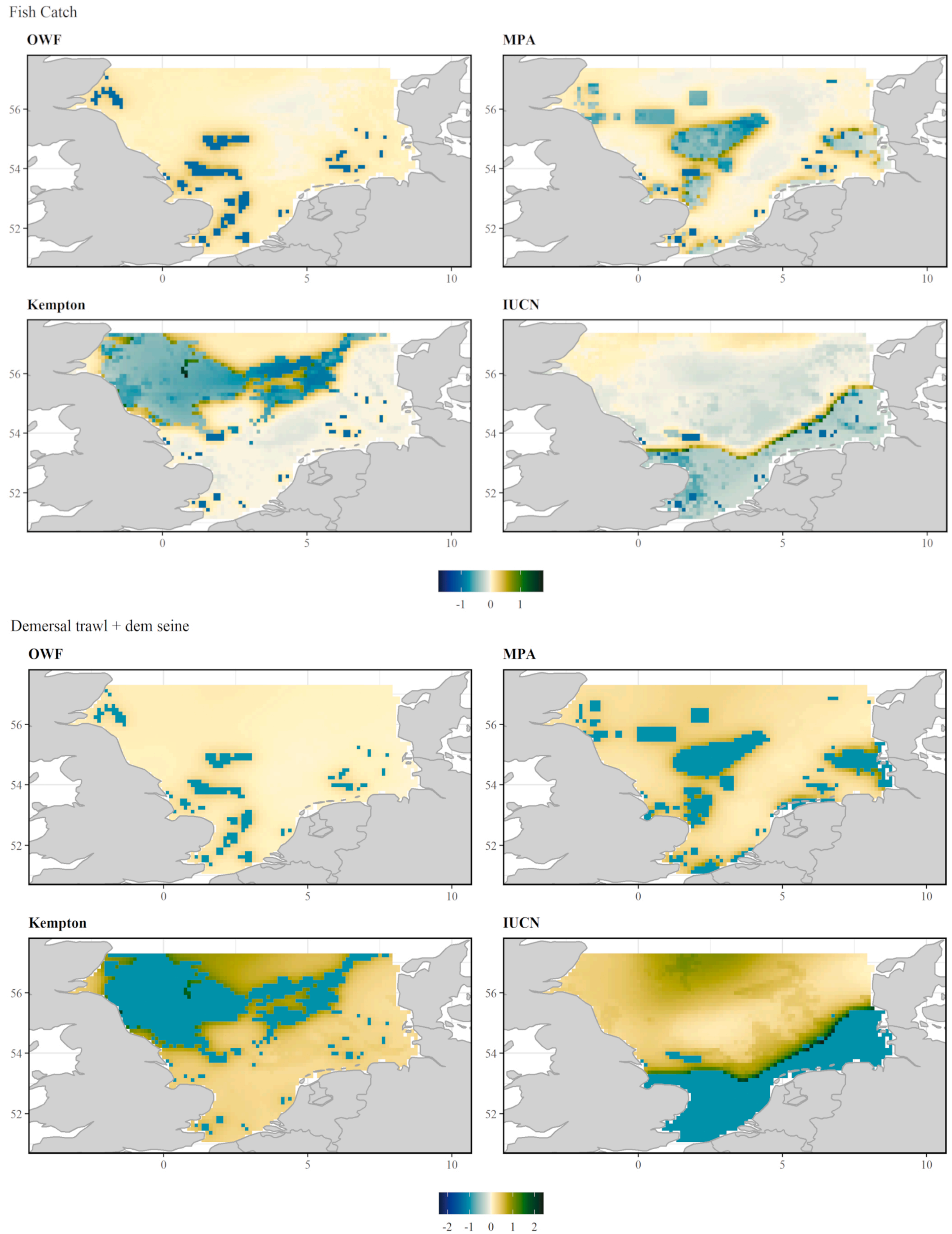


Fig. 6. Shift in fish catch distribution (top) and the distribution of effort of demersal trawlers and seiners (bottom). Changes are relative to baseline scenario with no closures with increases displayed by greenish to yellow colors and a decrease displayed by blue.

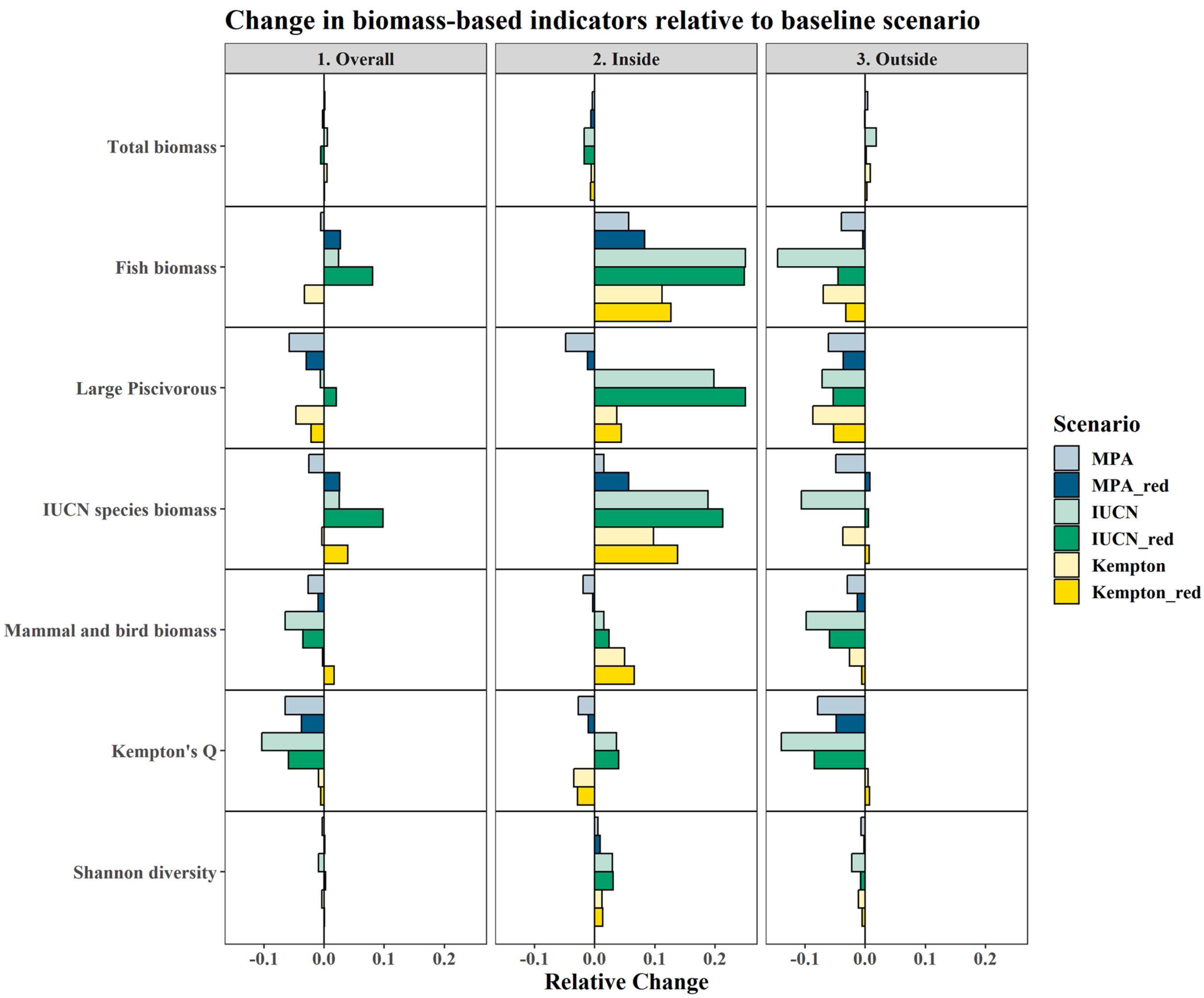


Fig. 7. Biomass-based indicators for the three scenarios that were run with an effort reduction. Darker colors display the effort-reduced scenarios, the lighter shaded colors the scenarios that were executed with the original effort as comparison.

with reduced effort, implying a positive overall impact on biomass-based indicators. Without the additional reduction in fishing effort, MPAs and OWFs did not seem to have much of a positive effect overall caused by trade-offs inherent in the food web, management decisions (e.g., amount of discards vs sea surface feeding seabirds and partial closures for certain gear types) and effort displacement. When assessing the losses and gains just for the catch-based indicators, all scenarios had only negative impacts on catch-based indicators and were therefore not displayed.

4. Discussion

A spatially-explicit ecosystem model for the southern part of the North Sea was used to evaluate the effects of closing specific areas to fishing in this complex ecosystem. The results show that the potential consequences and trade-offs of closures to fisheries are not always straightforward and depend upon many factors. Effort reallocation resulting from the closure, trade-offs within the ecosystem due to trophic interactions and trade-offs between conservation and economic goals complicate any spatial management approach. The impact of closed areas on the ecosystem and fisheries also highly depends on their size and location. Therefore, as a main lesson from this study one can conclude that it is most important to predefine management goals, to utilize tools that are able to predict possible outcomes and trade-offs from closure scenarios and to select suitable indicators that are able to measure progress.

4.1. Caveats and remarks

The EwE spatial modelling approach allows for the evaluation of the impacts of closures to fisheries on the full ecosystem [21,31,5,96,113]. Even though the southern North Sea model is focused on commercially-exploited fish species, all other ecosystem components from phytoplankton up to mammals are represented. The rich data availability for the southern part of the North Sea allowed parameterizing the distribution for over half of the functional groups with survey-based species distribution models.

Spatial modelling of a large number of functional groups however required a number of assumptions. For example, dispersal rates were entered based on life history traits rather than exact rates causing potentially some uncertainty about movement patterns. Moreover, fishery exclusions in MPAs in the model reduced catch but do not consider effects of reduced seabed disturbance and a possible recovery of the physical habitat, likely having positive impacts on the benthic

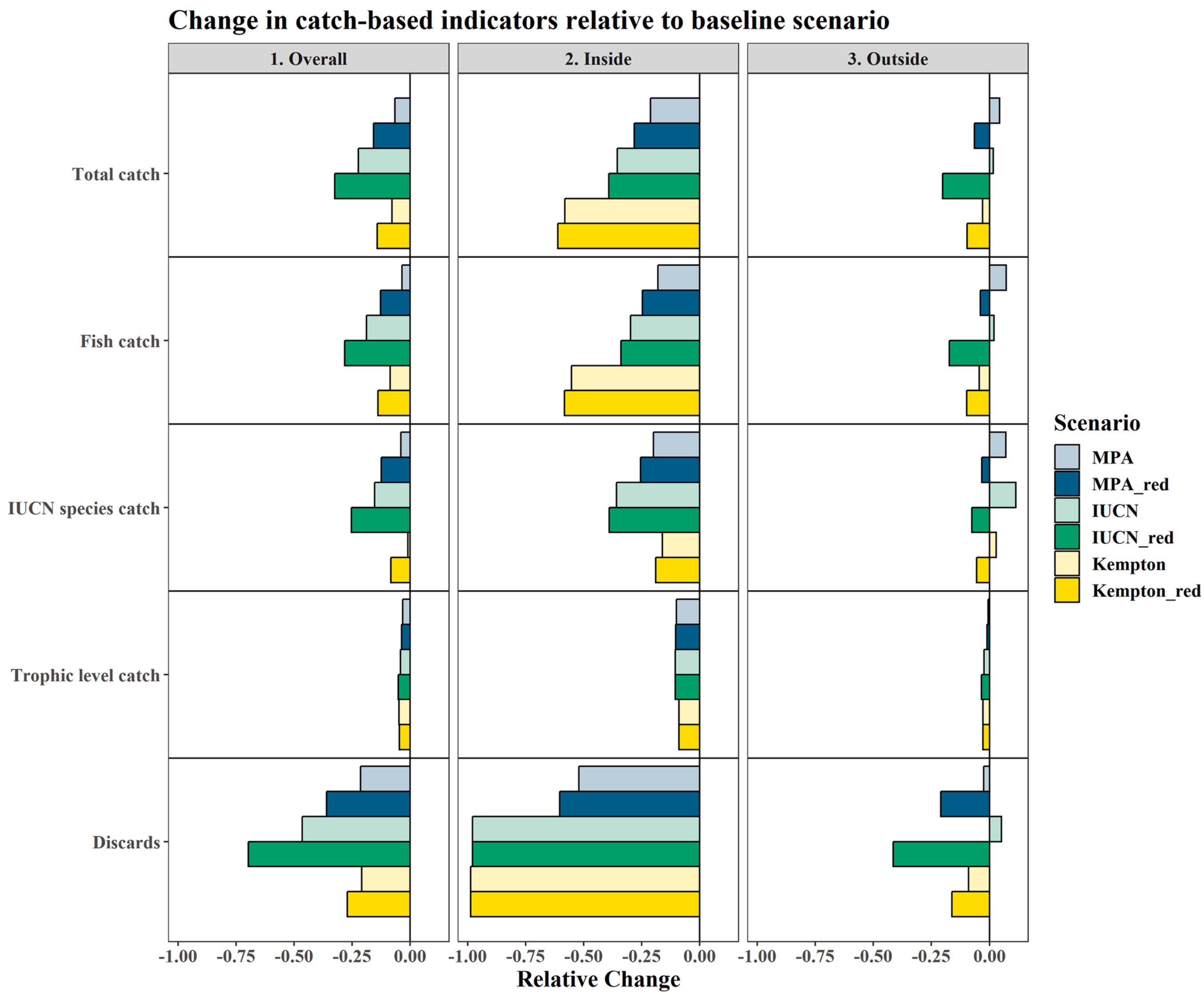
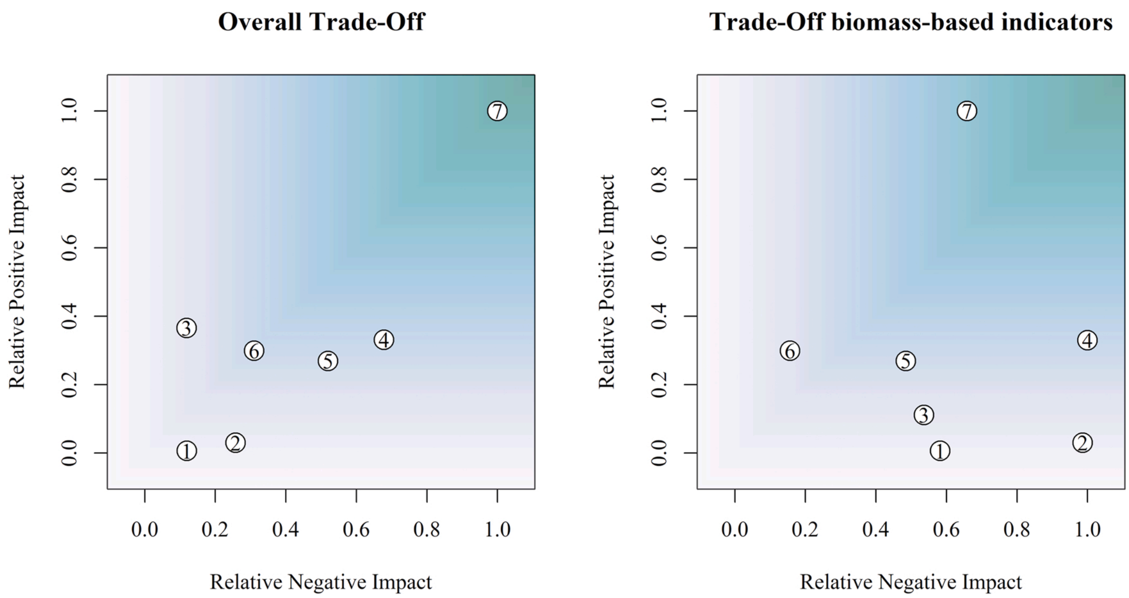


Fig. 8. Catch-based indicators for the three scenarios that were run with an effort reduction. Darker colors display the effort-reduced scenarios, the lighter shaded colors the scenarios that were executed with the original effort as comparison.

community within MPAs [71]. Modelling results were hence considered as conservative regarding the effect of closed areas and potentially underestimate spillover effects. Globally [25,55,68], as well as locally in the southern part of the North Sea [9,90], the impact of trawling on the epibenthic community as well as the impact of trawling on local habitats supporting the epibenthic communities has been studied [41,92,93]. However, the interconnection of trawling disturbance, seabed structures and traits of individual epibenthic species is not yet completely understood and only recently indicators aiming to categorize the vulnerability of epibenthic species to trawling have been suggested [8,57]. Furthermore, recovery rates due to the exclusion of fishing rates are highly dependent on prior amount of fishing pressure, depth of seafloor intrusion, habitat type as well as species and community types and can be highly variable [68,70,97]. Given the high variability among the recovery rates [54,86], including recovery rates per functional group per gear type for the entire study region is a complex endeavor. However, examples exist where closed areas had a positive impact on the local biomass of benthic species. One example being the closed areas in Georges Bank on the U.S. and Canadian East coast [82]. In this region large-scale areas were closed in 1994 after being heavily fished in the decades before in order to protect groundfish spawning habitats [75,82]. These closures led to an increase in the haddock population, as well as yellowtail flounder and cod, but most substantial was the increase in sea scallops inside the closed areas (*Placopecten magellanicus;* [82,106]).

Furthermore, specific effects of OWFs such as the sensitivity of benthic organisms and other functional groups to noise and vibrations produced by turbines were not assessed, due to the main study focus on trophic impacts of fishing effort re-distribution on the ecosystem. However, noise during construction pilling and operations could have impacted the distribution of the mammal functional group, inducing a shift in distribution of two top predator groups, which could increase the biomass of prey species such as cod and whiting within the OWF closures, while the pressure on all remaining prey outside the closure would have increased [111]. However, the impact of noise especially during operation on other functional groups included in the model is not yet fully understood [29,95]. No assumptions of the effects during the construction phase were made. The construction of OWFs could lead to a change in the composition of the macrobenthic functional groups [23], while the removal of the turbines could have varying impacts on multiple functional groups due to shifts in communities [78]. Both activities imply a shift in predator-prey fields, which can lead to negative impacts (less prey availability) or positive impacts (more prey availability). Furthermore, artificial structures can cause a connectivity sprawl, creating barriers and possibly influencing the distribution of species and



1. OWF, 2. MPA, 3. Kempton, 4. IUCN, 5. MPA + Effort reduction, 6. Kempton + Effort reduction,
7. IUCN + Effort reduction

Fig. 9. Trade-offs for all scenarios (left) and losses vs. gains per biomass-based indicators (right). For both gains and losses were summed up per scenario and trend (decrease/increase) and the impact relative to the other scenarios was calculated. Color gradient in the back displays the overall degree of trade-offs, small impact (light green) to strong impact (dark blue).

impact detrital pathways [12].

Within this study, an exclusion of fishing gears based on the ecosystem attributes they are aimed to protect was assumed (see Table 1). Therefore, fishing of bottom-contacting gears was restricted from areas with sensitive habitat or areas that were created to protect sensitive benthic communities, to prevent seabed disturbance [53]. Furthermore, static-gears were prohibited in protected areas designed for seabirds and mammals [61]. Therefore, not all of the currently implemented MPAs are closed to the same fisheries, some are only closed for bottom-contacting gear or passive gear, and some restrict fishing for both types of gears and finally in OWFs all fishing is restricted. This variance in restriction of fishing gears exacerbates the interpretation of the closure effect. Lester and Halpern [73] have shown, that there is a significantly higher density of organisms within no-take areas compared to partially-protected areas, implicating that the effects of the closures presented in this study would be stronger if the MPAs would have been closed for all fisheries. Additionally, fully-protected areas with a surrounding buffer zone with partial protection could increase the effectiveness of the MPAs [112].

In total, results show trends and relative differences between scenarios rather than absolute values. The results presented in this study should hence be interpreted as general lessons and provide an overview of what may be considered in future management via closed areas.

4.2. *Ecological and economic implications*

When removing fishing pressure the expected outcome is generally an increase in biomass and a healthier ecosystem [51]. The present analysis largely supports this assumption for biomass-based indicators inside the closed areas. Yet, over the entire study area the positive effect of closures is substantially reduced, or even negative, which is in line with previous results showing increased biomass of targeted fish species in MPAs while declining over the entire North Sea [72]. Catch-based indicators were found to mainly decrease, with strong variability among the scenarios. The overall reduction in total catch is the result of lost fishing areas that is not compensated by the catch outside of the closed areas. Unlike biomass-based indicators that react to shifts in fishing and changes in predator-prey distributions, the influence of the closures on catch-based indicators is a direct effect, which explains a generally greater deviation from the baseline run.

This study demonstrated that a major consequence of closing fishing grounds are alterations in species distributions. Especially with large, continuous closed areas a redistribution through dispersal of fish biomass became apparent. Outside the closures increasing fishing pressure and hence reduced biomass of predatory fish species locally caused increased invertebrate biomasses. Such spatial shifts due to altered spatial fishing patterns and modified trophic interactions lead to trade-offs between management goals. Clearly, trophic interactions need to be better incorporated in the evaluation of fishing restrictions like MPAs, otherwise these food web effects might be missed and the positive impact of MPAs potentially overestimated [31,5,96].

An important result of the study is the effect of effort redistribution due to fishing restrictions on local fish biomass. In the scenarios without effort reduction fish biomass generally declined, despite of local increases inside the MPAs. Effort redistribution here increased the fishing pressure outside the closed areas and especially at the MPA borders as a consequence of spillover of fish biomass in the model. Therefore, the overall fishing pressure was reduced to test if this decrease would cause positive effects on the ecosystem. In two out of three effort reduction scenarios even reversed trends with increasing fish and IUCN-endangered species could be displayed. Therefore, the study shows that fisheries closures potentially need to be combined with other management tools (e.g., Total allowable catch, effort limits) to counteract the effects of effort redistribution if an improvement also outside the closed areas is set as management goal [16,30,56]. Since additional effort reductions led to further losses in catch, a strong trade-off between fisheries and conservation goals becomes apparent.

Model simulations furthermore revealed that closing fishing areas will not only lead to reduced catches but consequently to reduced discards, in turn reduced discards negatively impact parts of the ecosystem such as scavenging seabird species that largely feed on discards [11]. In the southern North Sea model, the prey of *surface-feeding seabirds* is composed by a large proportion of discards which reflects the feeding behaviour described in literature. However, the diet parameterization has a degree of uncertainty [101,87] and therefore the effects of lower discards on these birds might be overestimated. Nevertheless, field studies showed seabirds to suffer from reduced discard-based prey availability [98] and hence reducing discards represents another example of trade-offs within an ecosystem that should not be ignored when setting management goals.

4.3. Placement, size and connectivity of fishing exclusion

The results of the two theoretical scenarios demonstrated that the location of a conservation area is as important as the size. The closures had roughly the same extent and were closed to the same fisheries as the previously defined MPAs, yet the outcomes varied significantly. The location of the IUCN scenario covers sea areas along the Danish, German, Dutch and Belgium coasts, which is an area of high biomass and catch. However, the distribution of areas with high biodiversity (i.e. Kempton's Q) was more dispersed with a higher concentration along the British coast and along the 50 m depth contour towards the east. The extent to which the MPAs covered fishing grounds varied greatly: the IUCN scenario overlaps significantly with major fishing grounds in terms of catch, the biodiversity-based closure only partially coincides with fishing grounds in the coastal areas. Therefore, the exclusion of fishing in these highly productive coastal grounds covered by the IUCN-based scenario has a much larger impact on the catch but also on the responses of biomass indicators. The importance of coastal areas along the southern German Bight is not surprising since it is a highly productive region with high net primary production [58]. Fish communities in this area are dominated by flatfish, such as the commercially important plaice and sole [38].

While the impact of closing this highly productive region was strongest, the closure of the region with high diversity and less fishing led to overall best ratio of benefits to losses in terms of ecological health and fisheries yield indicators. Therefore, effective placement of a closure site might be achieved by selecting an area with potentially high protection and traditionally low fishing effort. Spatial simulations furthermore demonstrate that evaluating conservation areas solely on their size might therefore be deceptive and considering the location in relation to the management goals is crucial when designating MPAs or even networks of MPAs [71].

An interesting result of the present study is that the spatial patterns of the Kempton's Q index did not overlap with most other indicators for biomass and catch. For Kempton's Q, areas along the 50 m depth contour of the British coast and the central North Sea were found to be the most important regions. This contour can be seen as a boundary, roughly separating epibenthic and fish communities [18] but with a high degree of mixing between round fish and flatfish [63]. Furthermore, in this area species (especially elasmobranchs) that are associated with the northern part of the North Sea mix with species that mostly appear in the southern part of the North Sea. For example, haddock (*Melanogrammus aeglefinus*), norway pout (*Trisopterus esmarkii*) or starry ray (*Amblyraja radiata*) are at the edge of their southern distribution, while plaice (*Pleuronectes platessa*) and cod (*Gadus morhua*) have shifted northward in the past decades [19,38,37,62].

Not surprisingly, the effect of closed areas was dependent on their individual sizes, where especially large closures were significantly more effective than small ones, which is supported by other studies that have found that larger MPAs may be needed to reach conservation goals [36,51]. Yet even small OWF may have larger effects than observed due to the coarse resolution of the model. Even though no specific test for spatial connectivity was carried out, results showed closing large areas (IUCN and Kempton scenarios) performed better compared to the OWF and MPA scenarios that represent many small-scale closures distributed throughout the southern part of the North Sea. This result may be in part due to the importance of cohesion when designing closures.

4.4. Indicator selection

Ecological indicators based on a trait-based approach were used to evaluate ecosystem impact by the fisheries in this study. Within the MSFD, ecological indicators are defined in association with environmental targets allowing the observation of progress towards achieving a "good environmental status" (EC, 2008). In a complex ecosystem like the southern part of the North Sea, these indicators are an important tool to analyze spatial patterns and shifts in the ecosystem in a more easily interpretable way. Nevertheless, a remarkable difference in the magnitude of changes among the different indicator types could be detected. Among all tested indicators, the community-based indicators showed less than 9% change in all scenarios and biodiversity indicators displayed less than 14% change, with a stronger impact on Kempton's Q. The remaining biomass-based indicators varied up to 25% in relation to the baseline run, while catch-based indicators decreased by up to 70% outside and overall. Total biomass showed one of the weakest overall responses, even in the IUCN scenario. This can be explained by the composition of the total biomass in the system. Eight out of sixty-seven functional groups (excluding discards) make up 92% of the total biomass, including particulate and dissolved organic matter and many benthic groups. All these groups are not targeted by the fishery. Additionally, especially the benthic groups, have a comparatively low predation mortality rate, thus a small overall production rate. Therefore, cascading trophic effects caused by closures only led to an impact of smaller magnitude compared to other indicators. Within the large ecosystem model applied for this study, some indicators such as all trophic level indicators include a large number of species and functional groups. Changes that occur within these indicators may not be as apparent than indicators including only specific groups or species, like mammals and bird biomass (17 species) or biomass/catch of IUCN-endangered species (21 species). Therefore, it is important to consider the right aggregation level (single species, functional groups, trophic levels) and the sensitivity of an indicator to be able to monitor progress towards management goals.

This study revealed the importance of single species or functional groups on some of the indicators, which proved important in their interpretation. As discussed in the previous section, the reduction in *surface-feeding seabirds* had a great impact on some indicators, especially the more specific indicators like "mammals and seabirds" or "IUCN-endangered species". Initially, the overall reduction in biomass for these indicators was counterintuitive, since conservation areas are also meant to primarily protect these vulnerable groups [47]. Yet, after investigating possible drivers to this reduction, it became apparent that only one group, *surface-feeding seabirds* was mainly responsible (see Appendix Figure 2.5B). One approach to circumvent effects like these would be to separate species that benefit from increased fisheries due to discarding from those that are directly targeted or caught as bycatch. Otherwise, progress towards certain management goals could be overlooked or management actions are based on the wrong impressions.

5. Conclusion and outlook

The evaluation of spatial management options and their resulting trade-offs in the southern part of the North Sea revealed the strong impact of placement and size of the areas closed to the fisheries in an ecosystem context. In particular, the contrasting impacts of the IUCN closure in a highly productive and heavily fished area versus the Kempton's Q closure focused on biodiversity, revealed that in order to achieve low economic and ecological trade-offs, regions with low fishing impact and potentially high protection value need to be identified. Furthermore, this study illustrates the potential necessity of further fisheries management measures simultaneously to closures to reach conservation goals. Moreover, the importance of evaluating impacts of spatial management options on a larger scale is highlighted. Regional effects, especially in and around an area with fishing restrictions, can differ severely from the overall impact. Increasing the focus on the impact of fisheries on the benthic community in addition to commercial species would improve conclusions that can be made from a spatial modelling study such as this one. Additionally, the effects of climate change need to be included in the creation of possible management scenarios, since usefulness and effectiveness of MPAs varies under a changing environment [116,60].

CRediT authorship contribution statement

Miriam Püts: Conceptualization, Methodology, Formal analysis, Writing – original draft, Writing – review & editing. **Alexander Kempf:** Supervision, Conceptualization, Methodology, Writing – original draft, Writing – review & editing. **Mollmann Christian Möllmann:** Supervision, Writing – review & editing. **Marc Taylor:** Conceptualization, Methodology, Writing - original draft, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.marpol.2023.105574.

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