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Relative impacts of multiple human stressors in estuaries and coastal waters in the North Sea–Baltic Sea transition zone

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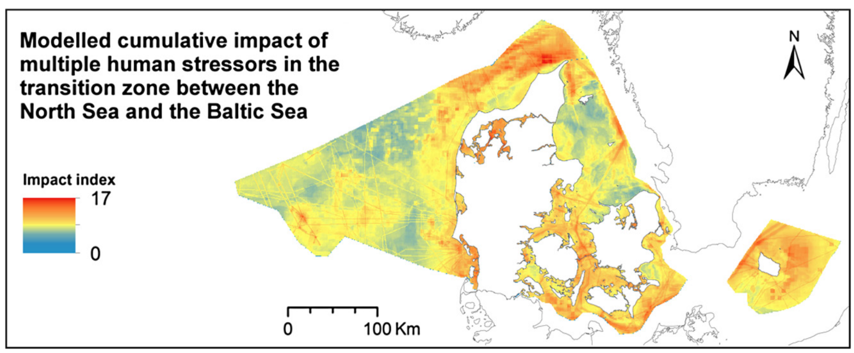
HIGHLIGHTS

- Relative impacts of multiple human stressors were studied in the North Sea/Baltic Sea.
- Top ranked stressors of the entire study area are Nutrients and Climate anomalies.
- Nutrients and Physical modifications have higher impact within fjord/estuarine systems.
- Fisheries, Contaminants and Noise have higher impact in offshore waters.
- Results provide evidence-based information in support of ecosystem-based management.

GRAPHICAL ABSTRACT



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ABSTRACT

The objectives of this study are 1) to map the potential cumulative impacts of multiple human activities and stressors on the ecosystems in the transition zone between the North Sea and Baltic Sea, for Danish waters 2) to analyse differences in stressor contribution between the European Union's Marine Strategy Framework Directive (MSFD, off-shore waters) and Water Framework Directive (WFD, coastal waters), and 3) to assess the local relative importance of stressors for 14 areas along a land-sea gradient, from inner fjords or coastal areas to offshore waters. The mapping of cumulative impacts is anchored in 35 datasets describing a broad range of human stressors and 47 ecosystem components ranging from phytoplankton over benthic communities to fish, seabirds and marine mammals, which we combined by means of a widely used spatial human impact model. Ranking of the stressor impacts for the entire study area revealed that the top five stressors are: 'Nutrients', 'Climate anomalies', 'Non-indigenous species', 'Noise' and 'Contaminants'. The gradient studies showed that some stressors (e.g. 'Nutrients', 'Shipping' and 'Physical modification') have a relatively higher impact within the fjord/estuarine systems whilst others (e.g. 'Fisheries', 'Contaminants' and 'Noise') have relatively higher impact in the open waters. Beyond mapping of cumulative human impacts, we discuss how the maps can be used as an analytical tool to inform ecosystem-based management and marine spatial planning, using the MSFD and WFD as examples.

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1. Introduction

Humans use and affect the marine environment in several ways, e.g. through fishing, shipping, extraction of materials, infrastructure, tourism and land-based pollution. This leads to multiple pressures caused by either direct activities (e.g. fishing, excavation) or indirect pressures (e.g. recreation, climate change), which all cause different kinds of stress upon the ecosystem components (e.g., overfishing or temperature effects on certain species) (Eigaard et al., 2017; Borgwardt et al., 2019; O'Brien et al., 2019). The combination of all activities and pressures acts upon marine ecosystems over different spatial areas and results in cumulative human impacts where the stressors (activities and pressures) occur together with ecosystem components in the same area (Halpern et al., 2008; Crain et al., 2008; Korpinen et al., 2012; Micheli et al., 2013; O'Brien et al., 2019).

The ecosystem health of Europe's seas is thus at risk from local to pan-European scales and several of the European Directives for the marine environment aim to mitigate this. This risk is irrespective of whether the objective is to reach Good Environmental Status (GEnS) *sensu* the EU Marine Strategy Framework Directive (MSFD; Anon., 2008), Good Ecological Status (GEcS) in coastal waters *sensu* the EC Water Framework Directive (WFD, Anon., 2000) or Favourable Conservation Status *sensu* the Habitats Directive (HD, Anon., 1992). The cumulative human impact needs to be addressed both to reach the various Directives' goals and to maintain sustainable ecosystems.

EU Member States bordering the European Seas are required to implement the MSFD through a wide range of activities including 1) an assessment of environmental characteristics and human pressures, 2) the setting of operational targets for GEnS regarding a total of 11 descriptors, and 3) the development and implementation of Programmes-of-Measures, with the ultimate goal to attain GEnS.

To assess characteristics, pressures and GEnS at the national level, the first generation of MSFD Initial Assessments was made by Member States in 2012. In 2018, follow up Initial Assessments were due, and this second generation of Initial Assessments was to some extent spearheaded by assessments at the regional level by Regional Seas Conventions (e.g. HELCOM for the Baltic Sea and OSPAR for the Northeast Atlantic Ocean). When assessing human impacts on their coastal and marine waters, Member States are required to evaluate not only the relevant individual human activities and pressures, but also their cumulative pressure (Anon., 2008; Article 1.3). The cumulative pressure should be kept within the levels to attain GEnS, ecosystem functions and resilience and preserving of marine goods and services (Anon., 2008; Article 8.1b). For the Danish EEZ, both the 2012 and 2018 Initial Assessment included detailed assessment of cumulative impacts (CIA) (see Naturstyrelsen, 2012; Miljø-og Fødevareministeriet, 2019).

The impact from human activities on marine ecosystems is determined by 1) the intensity, duration and characteristic, both in time and space, of the pressure that the activity is causing and 2) the specific ecosystem component's sensitivity to the pressure (Dailianis et al., 2018). The one-to-one relationship between a pressure and its impact on an ecosystem component is the most common focus of studies on environmental impact or impact risks (Borgwardt et al., 2019). This type of study can point to causal effects caused by the different pressures and provides important information for risk and environmental protection management. However, the cumulative effects of multiple stressors are not fully understood, in spite of many laboratory and field studies (Coté et al., 2016) and risk assessment studies (Doubleday et al., 2017; Borgwardt et al., 2019). Many approaches to assess cumulative effects use simple models that fill in knowledge gaps by means of expert judgment (Korpinen and Andersen, 2016), while more sophisticated, theoretically or empirically justified models are in development (e.g. Coll et al., 2016; Dafforn et al., 2016; Giakoumi et al., 2015; Teichert et al., 2016). However, they cannot, at present, incorporate all stressors that must be considered according to the European marine environmental directives (Hodgson and Halpern, 2019). Statistical models often suffer from small sample sizes and spatially clustered in-situ observations of overall ecosystem condition (Stock et al., 2018b). Hence, in this study, we explore how a relatively simple but widely used spatial model (Halpern et al., 2008, 2015) for mapping cumulative impacts of multiple human stressors could inform the implementation of the EU's coastal and marine environmental law.

Assessments of the environmental status in Danish marine waters are made on a regular basis. Both the environmental status and the temporal trends are well understood and well documented (HELCOM, 2010; Naturstyrelsen, 2012; OSPAR, 2010). Eutrophication – the effects of nutrient inputs and nutrient enrichment – is a nation-wide problem, and all Danish fjords and coastal waters are classified as 'eutrophication problem areas' (Naturstyrelsen, 2012). The only 'eutrophication non-problem area' is the open parts of the North Sea and the Skagerrak. With respect to marine biodiversity, all Danish marine waters are classified as being moderately to significantly impaired (Naturstyrelsen, 2012). A key driver behind the impairment is fishing activities (Andersen and Stock, 2013; Eigaard et al., 2017; ICES, 2019). Contamination due to inputs and occurrence of hazardous substances in sediments and biota has also been assessed in detail (Andersen et al., 2016), where offshore waters in the North Sea, Skagerrak and Kattegat are generally classified as non-problem areas, while many of the fjords and coastal waters are contaminated.

Beyond the creation of a national human impact map for Danish coastal and marine waters, we synthesize our results by ranking stressors and use the results as an analytic tool for studying the relative importance of key groups of human stressors along a land-sea gradient in 14 case studies. In order to evaluate the results based on the model of human impacts on marine ecosystems, we also investigate the uncertainty of the model results using recently developed simulation methods tailored for this model (Stock and Micheli, 2016; Stock et al., 2018a). Taken together, the aim is to demonstrate best practices and new ideas for analysing human impacts in the context of the European environmental law using the Danish marine waters as a test case.

2. Materials and methods

2.1. Study area

The study area consists of the eastern parts of the North Sea, southern parts of the Skagerrak, the western parts of the Kattegat, the northern and central parts of the Danish Straits as well as the south-western parts of the Baltic Sea (Fig. 1). Danish marine waters are parts of two neighbouring marine regions; the North Sea to the north and west and the Baltic Sea to the south-east. The hydrological conditions are variable, caused by a large part of the European drainage basin entering the North Sea as well as the large inflow of the brackish Baltic Sea waters into the Kattegat via the shallow Danish Straits and the Sound. Hence, the central Danish marine waters are affected by activities taking place, both on land and in the marine and coastal waters (HELCOM, 2018; Miljø-og Fødevareministeret, 2019).

2.2. Data sources

This study is based primarily on publicly available datasets for stressors (n = 35) and ecosystem components (n = 47) in Danish marine waters. In general, the newest published dataset that was

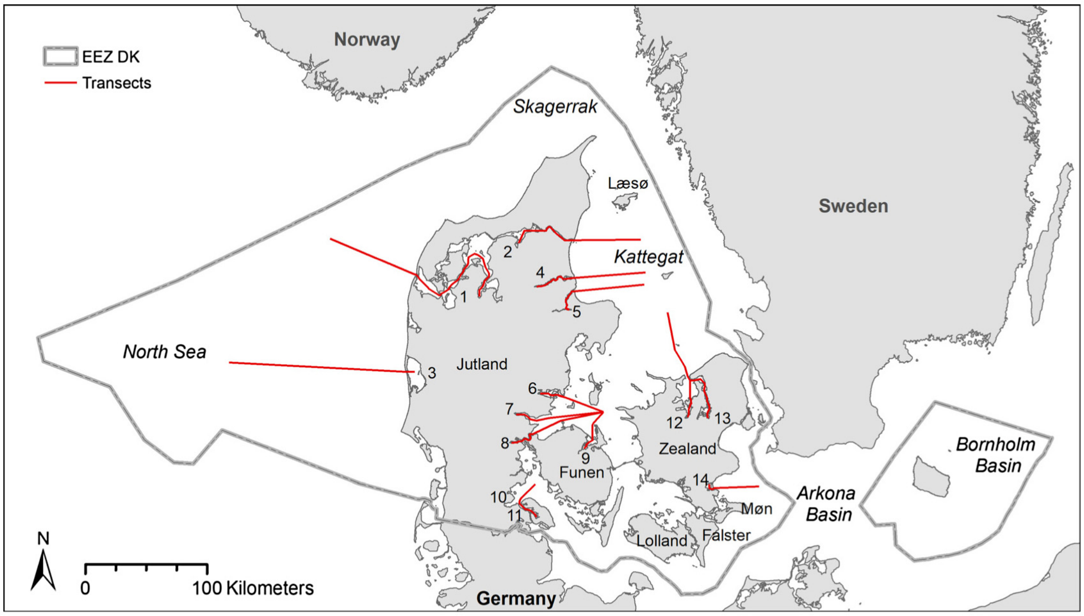


Fig. 1. Study area. Gradient studies have been undertaken in 14 case study areas and are highlighted by red lines. The numbers indicates the fjords; 1) Limfjord West, 2) Limfjord East, 3) Ringkøbing Fjord, 4) Mariager Fjord, 5) Randers Fjord, 6) Horsens Fjord, 7) Vejle Fjord, 8) Kolding Fjord, 9) Odense Fjord, 10) Aabenraa Fjord, 11) Augustenborg Fjord, 12) Isefjord, 13) Roskilde Fjord and 14) Præstø Fjord. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

available for each stressor and ecosystem component was used. All datasets cover one or multiple years, and seasonality is not considered. Table 1 summarises the origin of all stressor and ecosystem component data, while detailed descriptions of each dataset are provided as Supplementary Material.

Where possible, updates of the original stressor datasets from Andersen and Stock (2013) ('Oil and gas pipelines' and 'Recreational shipping' Table S1 #17, 20) were made with additional information to extend their spatial coverage to the whole study area.

Datasets from Mohn et al. (2015) (Table S1; physical constructions, noise, pollution, climate anomalies and shipping (#1–2, 5, 8–9, 11–13, 18, 21, 23) as well in Table S2; eelgrass, plankton communities, birds (#10, 12, 34–37) and marine mammals (#41–45)), were obtained by georeferencing pictures of data layers. The colour nuance of every grid cell was converted to a value, based on the colour scale and associated values given in the legends. Thereby, a quantitative representation of every map was achieved. All data processing was carried out in the statistical software 'R' using the packages "dplyr", "png", "tidyr", "data.table" and "raster". (Dowle and Srinivasan, 2017; Hijmans, 2017; R Core Team, 2013; Urbanek, 2013; Wickham et al., 2017; Wickham and Henry, 2018).

Data on benthic habitats from Populus et al. (2017) were provided as shapefiles. The original data contained 37 benthic habitat types, which were reclassified into the 8 broad-scale benthic habitat types used in this study (Table S2; #1–8). Since no estuary dataset existed, an estuary layer (Table S2; #9) containing the major Danish fjords, estuaries, and semi-enclosed bays was manually generated in GIS.

The remaining data layers used in this study were generated by the study by Andersen et al. (2017) or downloaded from the Danish Environmental Protection Agency's data portal (see Supplementary Material for details). One of these data layers is 'Oxygen deficit', which we consider a pelagic habitat and in Danish waters as a key response to nutrient enrichment and eutrophication (Andersen and Conley, 2009).

2.3. Mapping of potential cumulative impacts

Estimates of potential cumulative impacts of multiple human stressors were calculated as described by Halpern et al. (2008, 2015). The calculations were made using the software 'EcoImpactMapper' (Stock, 2016), an open-source Java program implementing the models developed by Halpern et al. (2008, 2015). In spite of its simplicity (Halpern and Fujita, 2013), this model is widely used for cumulative human impact assessments of marine ecosystems (Korpinen and Andersen, 2016). A conceptual model of the steps in the analysis is shown in Fig. 2.

The Halpern et al. model requires three kinds of input data:

Table 1
Origin of the stressor and ecosystem component layers used in the study. A detailed overview as well as maps of individual data layers are presented in Tables S1 and S2.

Source	Reference	Stressors	Ecosystem components	Total
DTU Aqua	Dahlskov et al. (2012) and Warnar et al. (2012)	12	21	33
HARMONY	Andersen and Stock (2013)	2	5	7
SYMBIOSE	Mohn et al. (2015)	11	11	22
EUSeaMap2	Populus et al. (2017)	–	8	8
RALAHA	Andersen et al. (2017)	10	2[1]	12
Total		35	47	82

[1] The dataset for 'contamination status' originates from the EMODnet Chemistry project and has been reanalysed for this study.

- $\boldsymbol{D_i}$, the spatial distribution of each stressor i, represented by a regular grid; for example, fishing intensity with a given gear type or climate anomalies. All stressors are normalized by log(x + 1)-transformation and rescaled so that the maximum value is 1.
- $\boldsymbol{e_j}$, the spatial distribution of each ecosystem component j, represented by a regular grid; for example, different kinds of soft-bottom habitats or fish species, either as presence-absence or continuous (e.g. probabilities of presence) data.
- $\mu_{i,j}$, so-called sensitivity weights, a numerical representation of the sensitivity of ecosystem component j to stressor i, based on expert interviews.

The intensities of the stressors were made comparable by log(x + 1)-transformation and rescaling to maximum 1.

For stressors with a point distribution or decay from a restricted area, effect distances were estimated based on expert interview, and the data layers were pre-processed by adding this effect according to the values listed in Table 2, assuming a linear decay function from the source and to the limit of the effect distance, similar to other studies in the North Sea and Baltic Sea regions (HELCOM, 2010; Korpinen et al., 2012; Andersen and Stock, 2013).

In the original model by Halpern et al. (2008, 2015) only the stressor data were normalized. However, these studies used only one kind of ecosystem component data (e.g. only presence-absence or only probabilities of presence). In contrast, our input data were diverse including presence-absence, probabilities of presence, population densities or concentrations. Hence, the ecosystem component data were also normalized by log(x + 1)-transformation and rescaling to maximum 1 in order to make the data layers comparable.

Based on these data, Halpern et al. (2008) calculate the dimensionless additive human impact index for each cell in the regular grid *(x,y)* estimated for n stressors and m ecosystem components. We calculated the human impact index I_{Mean} as follows (Halpern et al., 2009; Stock, 2016):

$$I_{Mean}(x,y) = \sum_{i=1}^{n}\sum_{j=1}^{m}\frac{1}{E_{Div}(x,y)}D_i(x,y)e_j(x,y)\mu_{i,j} \quad (1)$$

$$E_{Div}(x,y) = \sum_{j=1}^{m} e_j(x,y) \quad (2)$$

In this study we estimate the cumulative impact as the mean of the impact over all present ecosystem components, rather than the sum, because some ecosystem component datasets did not cover the whole study area. This model is also the most applied method in newer papers (e.g. Halpern et al., 2015), and avoids conflating the effects of high-intensity stressors with the number of ecosystem components in a given grid cell. Besides the impact indices, the contribution of each of the stressors to the total index was also calculated.

The spatial distribution of ecosystem components and stressors was summarized by calculating an ecosystem index (Eq. (2)) and an unweighted stressor index according to Stock (2016):

$$P(x,y) = \sum_{i=1}^{n} D_i(x,y) \quad (3)$$

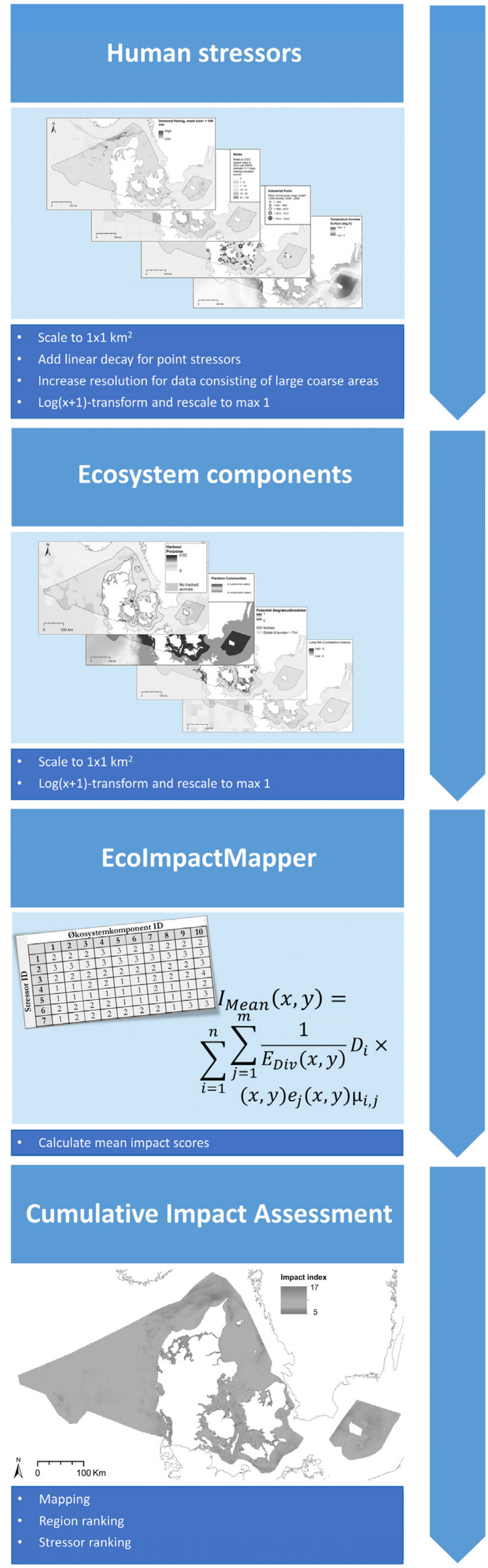


Fig. 2. Conceptual model of data pre-processing steps. Step 1a: Establishing a spatial data set on human stressors including scaling to assessment units, log-transformation and normalisation; Step 1b: establishing a spatial data set on ecosystem components, including scaling, log-transformation and normalisation; Step 2: Running the EcoImpactMapper software; Step 3: Mapping of spatial variations in mean impact scores based on step 2 including ranking of stressors. Based on Stock and Micheli (2016) and Riemann et al. (2019).

Table 2
Summary of experts' estimates of effect distances for stressors that have effects beyond the immediate areas where they occur, and where the spatial decay of stress was not inherent in the data.

No.	Effect distance per stressor	Median	Max	Min
1.	Bridges and coastal dams	1 km	25 km	0 km
4.	Dredged material disposal sites	5 km	10 km	0 km
5.	Dumped chemical munitions	1 km	5 km	0 km
6.	Industrial ports	5 km	10 km	0 km
7.	Marine aquaculture sites	5 km	5 km	0 km
9.	Military areas	10 km	50 km	0 km
15.	Offshore oil and gas installations	1 km	5 km	0 km
16.	Offshore wind turbines	1 km	5 km	0 km
20.	Recreational shipping	1 km	1 km	0 km
22.	Sediment extraction sites	1 km	10 km	0 km

2.4. Setting of sensitivity weights and effect distances

To estimate cumulative human impacts, each stressor's intensity must be linked to the responses of the ecosystem components. In Halpern et al. (2008, 2015) this is achieved by assigning a "sensitivity weight" to each combination of ecosystem component and stressor. The larger the sensitivity weight, the more sensitive is the ecosystem component to the stressor (Halpern et al., 2007; Teck et al., 2010), resulting in a larger contribution to the human impact index where the stressor and ecosystem component occur together. 12 experts, chosen through our professional network, were interviewed to estimate the sensitivity weight for each combination of stressor and ecosystem component, ranging from 1 (the stressor has a neglectable effect on the ecosystem component) to 5 (the ecosystem component is highly sensitive to the stressor). The median sensitivity weight from all responses for each ecosystem component and stressor was calculated and used in the model. Like Andersen and Stock (2013), the experts were asked to estimate the 'effect distance', i.e. the maximum distance from where a stressor is located to where it potentially might have an effect (in classes <1 km; >1 km; >5 km; >10 km; >25 km; and >50 km). Some stressor layers already represented the spatial extent of a stressor (e.g. underwater noise) whereas others were represented by the sources of the stressor (e.g. marine aquaculture sites). The effect distances were applied only to the latter by using the medians estimated by the experts and assuming a linear decay of stress from the source.

Empirical data for ecosystem components' responses to stressors only exist for very few of the 1645 combinations of stressors and ecosystem components included in this study. Therefore, as in previous cumulative human impact studies (De Lange et al., 2010; Andersen and Stock, 2013; Halpern and Fujita, 2013; van der Wal and Tamis, 2014; Giakoumi et al., 2015; Piet et al., 2015; HELCOM, 2018), as described above, we relied on expert judgement for estimating sensitivity, sometimes described as a Delphi technique (Hsu and Sandford, 2007; Rowe and Wright, 1999). Further, some risk assessments and evaluations of cumulative impacts have been based solely on the linkages between the human activities, the pressures they cause, and the expected effects on ecosystem components they cause, assessed by expert judgement (Borgwardt et al., 2019; Doubleday et al., 2017; Knights et al., 2015; Piet et al., 2015; Singh et al., 2017). Hence, one of the key assumptions in cumulative impact analyses is that the expert judgement is accurate. To fulfil this criterion, an experienced and broad range of experts was chosen. The sensitivity weights used in the model were taken from the medians of the replies.

2.5. Gradient studies

To further investigate the importance of spatial variation and local conditions for the different stressors, we examined transects from coastal and estuarine to open sea waters in 14 selected fjords (of which some are estuaries; Fig. 1). From positions located furthest inside the fjord and outwards to the open waters, we extracted the contributions of different stressor groups to the overall impact index at intervals of approximately 5 km. The lengths of the transects varied from approximately 25 km for Kalundborg Fjord to almost 200 km for the western Limfjord transect. At each 5 km point, the impact of each stressor was calculated as a percentage of the total impact at that position. General patterns were also investigated to study the importance of different stressors in a coastal to open sea gradient by combining the 14 transects. For this purpose, the results were adjusted from individual transects in two ways. First, the transect results were shifted in space so that the distance was measured from the mouth of the fjord and out, not the end of the transect (i.e. inner part of the fjord). This resulted in an alignment of the transects at the fjord mouths at 0 km, with negative distances indicating movement into the fjord and positive distances outwards into open waters. Second, the mean value per every fifth km was calculated for all percentage contributions in all transects for the stressor group in question. The percentage contributions were normalised to this average value before combining them.

2.6. Uncertainty analysis

The robustness of the impact index and stressor ranking for Danish marine waters was evaluated by means of Monte Carlo simulations with 1000 runs according to Stock and Micheli (2016). The uncertainty analyses included possible weaknesses in data quality (e.g. coarse resolution, missing input layers) and effects of model assumptions. Within each simulation run, we randomly modified the input data and the model as follows: 1) Randomly exclude up to 1/3 of stressor layers, 2) introduce a sensitivity weight error of +/− 0 to half of the original range of weights and chosen from a uniform distribution within this range, 3) vary effect distance of stress between 0 and 20 km (only applied to the stressors with an effect distance included), 4) use of different model calculations (i.e. antagonistic instead of additive model of stressor interactions), 5) use of ecological thresholds instead of linear ecosystem responses to increasing stress, 6) use of different stressor transformation methods (log, cumulative density function, or truncating at the 99-percentile), and 7) reduced analyses resolution from 1 km to 2 km cell side length. Please see Stock and Micheli (2016) for details regarding the implementation of the uncertainty analysis and ranges of the factors included.

3. Results

3.1. Expert interviews: sensitivity weights and effect distances

The median sensitivity weights provided by the expert survey ranged from 1 to 4.5 (Table S3). Thus, according to the experts, some stressors had negligible effects on certain ecosystem components whilst others had the potential to have much larger effects on specific ecosystem components.

The expert-derived effect distances for stressors applied to the relevant data layers are listed in Table 2. There was disagreement about the effect distances: For example, the estimates for military areas ranged from 0 km (only local impacts) to $\geq$50 km, the largest distance class in our survey. However, most experts estimated rather small effect distances: The median estimate for military areas was 10 km; the median estimates for marine aquaculture sites, industrial ports and dredged sediment disposal sites was 5 km; and the median estimates for all other stressors for which we requested such an estimate was 1 km. Hence, these are the distances we used in the cumulative impact model.

3.2. Spatial distribution of stressors, ecosystem components and cumulative human impacts

Several maps for the Danish EEZ were produced (Fig. 3) in order to summarize the spatial distribution of all stressors, all ecosystem components and the calculated cumulative human impact. A stressor index map calculated from Eq. (3) shows the spatial distribution of the intensity of the summed stressors for each grid cell (Fig. 3A). Grid cells with high values represent areas with many stressors occurring together at high intensities. The areas with the highest stressor index values were found in the north-eastern and southwestern most part of the North Sea, in the offshore parts of Skagerrak, in the north-eastern parts of the Kattegat and in the western part of the Danish areas of the Baltic Sea. The major commercial shipping routes (#23) between the North Sea and the Baltic Sea, as well as intense fishing activities are all located in these areas with high stressor index. The areas having the lowest stressor index values were in the open parts of the North Sea, the central and western parts of the Kattegat, north of the island of Zealand and southwest of Bornholm.

The ecosystem index (Fig. 3B) calculated from Eq. (2) shows the areas with a high density, high probability or presence of many ecosystem components. Most ecosystem components were found in the Kattegat and northern Great Belt, while areas with few ecosystem components were found around Bornholm and in coastal areas around Zealand.

Areas with high impact index values were found in most estuaries, fjord systems and coastal waters, apart from the coastal waters north of Zealand (Fig. 3C). The open waters in the southwestern part of the North Sea, in the Skagerrak, northern and central Kattegat, south of Lolland, Falster and Møn and northwest of Bornholm also had high index values. Low values were found in the open parts of the North Sea, south of Læsø as well as southwest and east of Bornholm.

3.3. Ranking of stressors

The 35 stressors were ranked according to their contribution to the cumulative impact index aggregated for the entire study area (see Table S4). In addition, we also ranked the stressors separately for each of the three regions North Sea, Kattegat and Baltic Sea. The stressors 'Nitrogen winter concentrations (DIN)', 'Climate anomalies', 'Non-indigenous species', 'Phosphorous winter concentrations (DIP)', and 'Marine litter' made the largest aggregated contributions to the cumulative impact index (Table S4). 'Noise' and 'Oil spills' also had large contributions. The ranking of top stressors was consistent within the different regions. In contrast, stressors having the least impact showed differences among the regions, as some of the stressor layers were not present in all regions, e.g. oil and gas installations. Together, the top 5 stressors contributed 68% to the total impact index for the model including all 35 stressors and 70% of the total for the model without 'Climate anomalies'. The top 10 stressors accounted for as much as 89% (incl. 'Climate anomalies') and 88% (without 'Climate anomalies') of the total impact index. Within the sub-regions, there was a similar pattern: the top 5 stressors accounted for a large fraction of the total impact (72%, 71% and 69% in the Baltic Sea, Kattegat and North Sea, respectively). In the North Sea and Baltic Sea, the top 3 stressors were identical to those for the Danish EEZ waters, whereas in the Kattegat, 'Phosphorous winter concentrations (DIP)' was ranked higher than 'Non-indigenous species'. The top 5 stressors in Danish waters all have in common that they are widely distributed and affect many of the ecosystem components sensitive to the stressors, whereas the stressors contributing the least to the total impact index act at local scales. The local relative importance of stressors has been investigated further in the land-sea gradient studies.

Since many of the stressors are similar or represent the same type of impact on the ecosystem, as a second step they were grouped together to represent the same type of pressure, with the stressors 'Climate anomalies', 'Non-indigenous species' and 'Marine litter' as separate groups. Using this classification, the pressure from 'Nutrients' had the greatest contribution making up about one third of the total impact index, followed by 'Climate anomalies', 'Non-indigenous species', 'Noise' and 'Contaminants' (Fig. 4A and B). When grouping the stressors, the impact from different fishing methods causing a pressure were clearer, contributing with 9.6% to the impact index. Furthermore, 'Fisheries' and 'Marine litter' were ranked equally. The group of stressors with the lowest combined impact was 'Physical modifications'. As there is a spatial difference in both stressor impact and presence and ecosystem complexity between coastal areas and offshore open sea waters, the relative contributions of stressor groups to the total impact within WFD coastal waters (Fig. 4C) and within MSFD offshore waters (Fig. 4D) were also analysed (without 'Climate anomalies'). The main difference is seen in the impact of 'Fisheries' contributing 2.8% in WFD coastal waters and 11.4% in MSFD open sea waters and 'Nutrients' contributing 46.4% in the WFD coastal waters and 33.2% in the MSFD marine waters. For a detailed information of stressor groups, see Supplementary Material Table S1.

3.4. Uncertainty analysis

The results from the Monte Carlo simulations for the stressor ranking, including 'Climate anomalies', showed that the 5 stressor layers 'Climate anomalies', 'Nitrogen winter concentrations (DIN)', 'Marine litter', 'Phosphorous winter concentrations (DIP)' and 'Non-indigenous species' were placed in the top 25th percentile of stressors in 83–100% of the 1000 simulation runs (Fig. 5). Thus, the simulations confirmed that these stressors would be found to be the most dominant under a wide range of model assumptions as well as possible errors in the input data. The stressors that were most consistently in the lowest-impact 25% were various physical modifications, 'Aquaculture', 'Longline fishery' and 'Mussel dredging' (85%–98% of simulation runs), the same ones as in the ranking according to the original model.

In the spatial results of the Monte Carlo simulations for the human impact index, about 33% of the most impacted 25% of the study area according to the original model (Fig. 5) were confirmed in at least 75% of simulation runs. However, while only some results were confirmed by the simulations, the general locations of high-impact areas according to the original model and the simulations agreed (Fig. 5A and C). For the least impacted 25% of the study area, a smaller proportion (19%) of the original results was confirmed in the simulations (Fig. 5B and C). While most low-impact areas in the Kattegat were confirmed by the

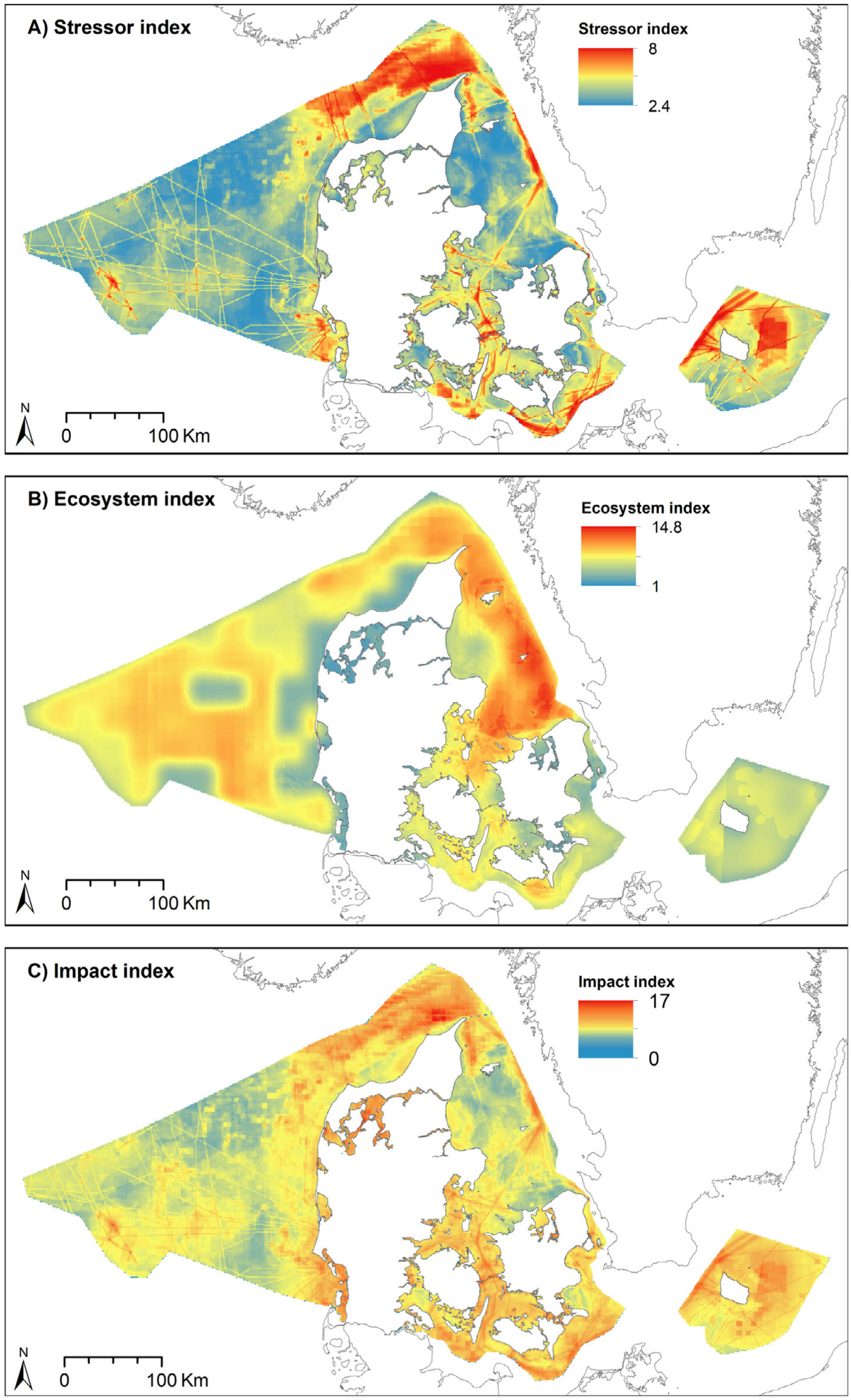


Fig. 3. Stressor index (number of stressors per assessment unit) (A), Ecosystem index (number of ecosystem components per assessment unit) (B), and Cumulative impact index (including 'Climate anomalies') (C). The estimated distribution of potential cumulative impacts is based on 35 stressors and 47 ecosystem components weighted by species sensitivity for Danish marine waters.

simulations, many low-impact areas in the offshore North Sea were not confirmed. This result does not necessarily imply that these areas are more impacted than estimated by the original model. Rather, it suggests that determination of areas with low modelled human impact is more sensitive to model assumptions and data quality.

3.5. Gradient studies

The relative contribution of each stressor to the total cumulative impact varied along each of the transects studied from the inner parts of the fjord to open waters (Fig. 6). Comparisons between the transects are difficult as they represent quite different

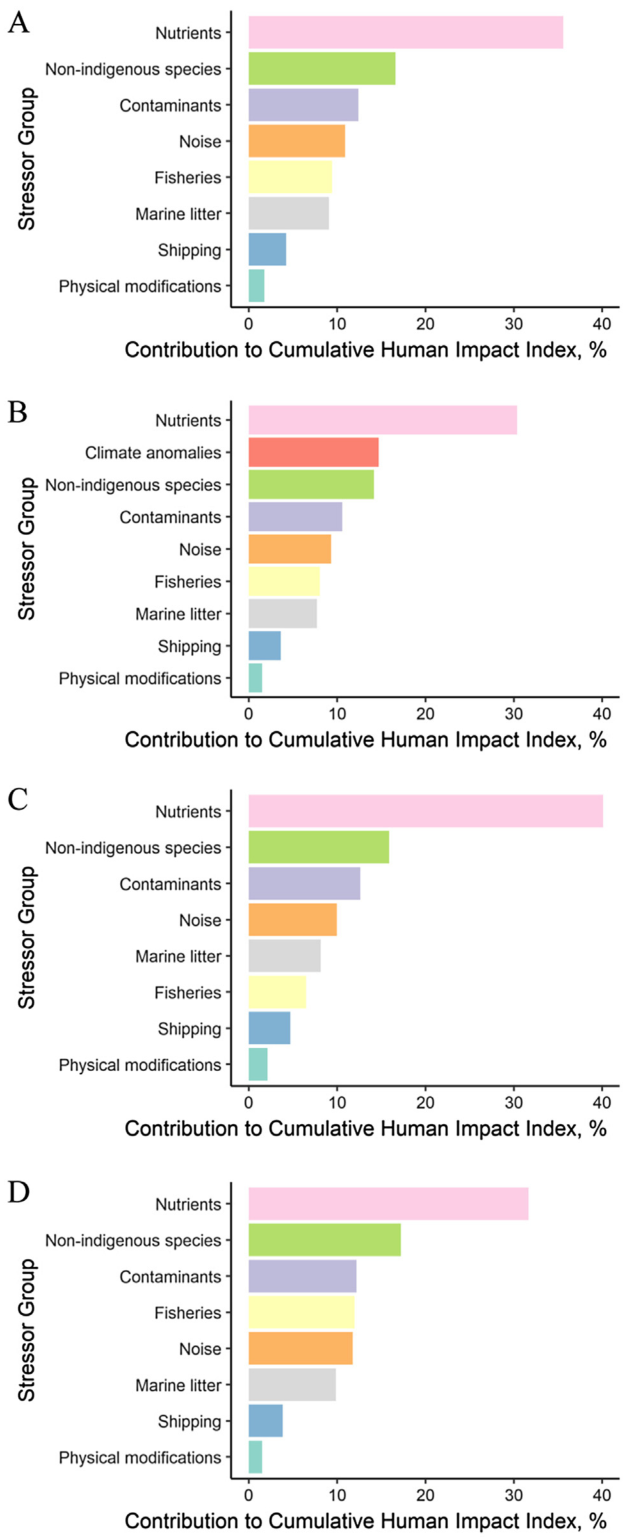


Fig. 4. Grouped stressor contribution. For all Danish marine waters (A), for all Danish marine waters including 'Climate anomalies' (B), for WFD coastal waters (C) and for MSFD offshore waters (D).

ecosystems. However, there are some clear trends regarding the stressor groups 'Nutrients' and 'Fisheries'. The relative contribution of nutrients to the total impact in each grid cell in the innermost parts of the selected fjords varied from 40% (Kalundborg Fjord) to above 75% (Mariager Fjord) of the impact index. Broadly speaking, nutrients accounted for a greater proportion of the impact index in the fjords than they did in open waters. As also might be expected, fisheries had greater contributions to the impact index in the open parts of the North Sea and Kattegat than in the inner parts of the fjord or Belt Sea areas.

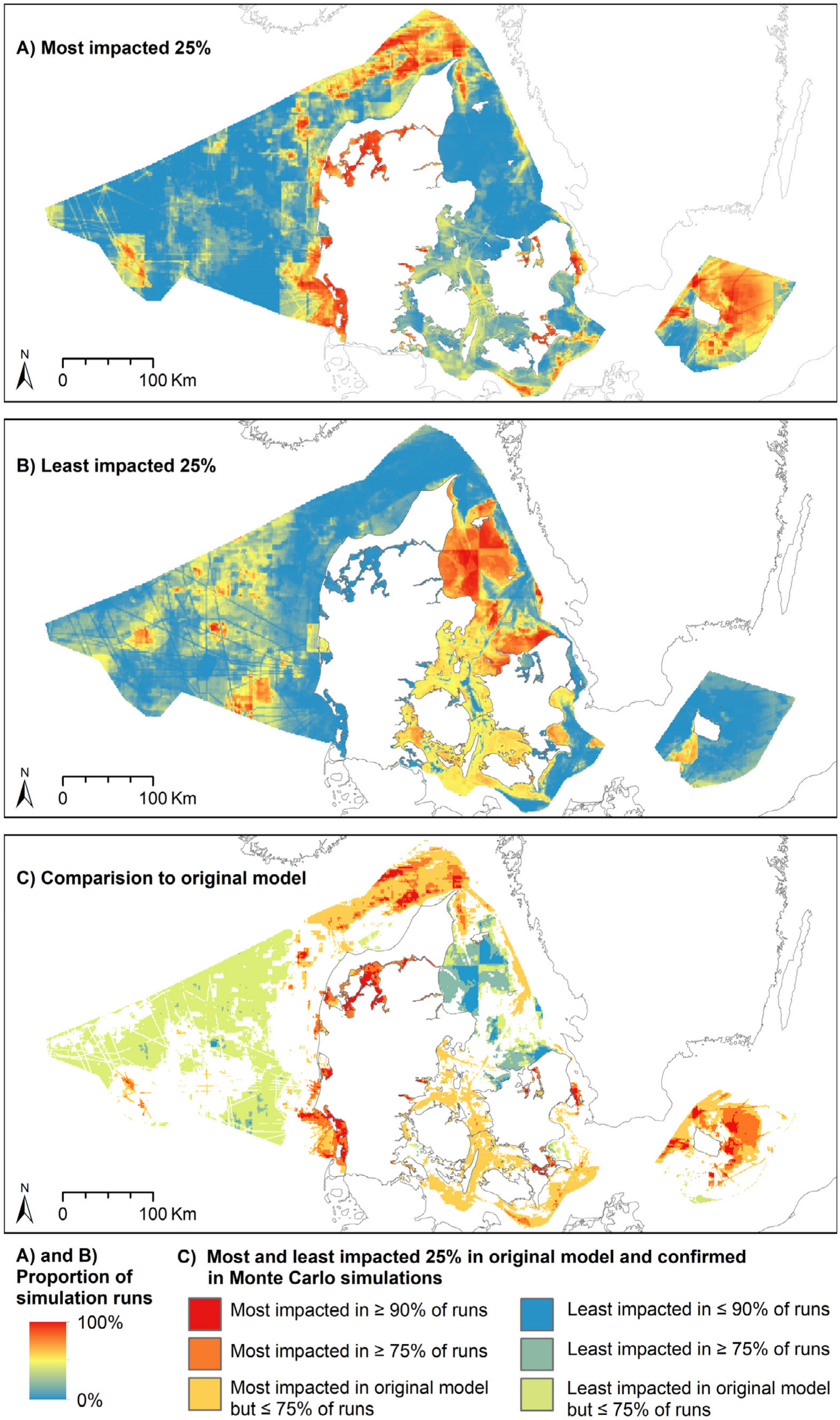
The general patterns for the transects were analysed by a local polynomial regression fitting (LOESS) for nine selected stressors groups (Fig. 7). The plots show the "combined transects" with the results for 'DIN' (Fig. 7A) and 'DIP' (Fig. 7B) separately. The results match the expectations that the impacts of nutrients are greater in the fjords than in open waters (e.g. Carstensen et al. (2006)). The impact of 'Fisheries' also matches expectations with lower impacts in the fjord and with higher impacts in coastal and especially offshore waters (Fig. 6F), although 'Mussel dredging' caused a local increase in relative contribution to the impact in the Limfjord west transect, approximately 40–50 km from the transect starting point. Both 'Noise' and 'Contaminants ' contributed more to the relative cumulative impact off-shore (Fig. 7G and I), whereas 'Non-indigenous species' decreased with distance from shore (Fig. 7D). The stressor groups 'Marine litter', 'Shipping' and 'Physical modifications' (Fig. 7C, E and H) had a rather constant contribution along the gradients.

4. Discussion

Assessment of human pressures or stressors in the marine environment has with the implementation of the MSFD gradually shifted from a focus on long-term temporal trends in individual stressors to include integrated assessments of cumulative impacts. This is a significant step forward, although the models applied at this stage are simple and do not consider synergistic or antagonistic effects. The most widely used models for spatially assessing cumulative effects of multiple human stressors are based on Halpern et al. (2008) and other, similar simple models (Andersen and Stock, 2013; Korpinen et al., 2012; Stock and Micheli, 2016). These models are generic, repeatable, spatially explicit and can represent several pressures acting upon several ecosystem components at one time.

There are other methods for assessing cumulative impacts or risk of impact, which are usually targeted towards a specified risk from specific human pressures based on expert elicitation (e.g. Singh et al, 2017; Borgwardt et al., 2019). These type of environmental risk assessments can be useful for guidance towards successful management and to identify species or habitats at risk of negative impacts from human activities. Although these methods may consider the effects of multiple pressures on some ecosystem components, they are not spatially integrated and the human activities are often treated separately, except in Borgwardt et al. (2019) where the impact risk is calculated based on estimated criteria and summed. However, studies focusing on the effects of combined multiple stressors are still rare (O'Brien et al., 2019).

The approach used for this study has some shortcomings and would benefit from further developments (Halpern and Fujita, 2013), but is considered not only useful but also fit-for-purpose and can inform management when incorporated into decision frameworks (Tulloch et al., 2015; Korpinen and Andersen, 2016) and under careful consideration of the involved uncertainty (Stock et al., 2018a). We have with this study addressed several of these limitations by presenting a comprehensive human impact map for Danish waters, based on robust and detailed national- to regional-scale datasets which cover most important human activities and relevant ecosystem components. Beyond the current standard methods for cumulative impact mapping, we have also applied recently developed uncertainty analysis methods to better distinguish robust model results from findings that are sensitive to model assumptions and data quality.



A) Most impacted 25%
N
0
100 Km
B) Least impacted 25%
N
0
100 Km
C) Comparision to original model
N
0
100 Km
A) and B)
Proportion of
simulation runs
100%
0%
C) Most and least impacted 25% in original model and confirmed in Monte Carlo simulations
Most impacted in ≥ 90% of runs
Most impacted in ≥ 75% of runs
Most impacted in original model but ≤ 75% of runs
Least impacted in ≤ 90% of runs
Least impacted in ≥ 75% of runs
Least impacted in original model but ≤ 75% of runs

Fig. 6. Relative contribution of stressor groups to the total impact, along transects from inner fjord (0 km) to open water for 14 selected fjord systems in Denmark. A dashed line indicates the location of the mouth of the fjord system.

4.1. Datasets

The stressor data represent both human activities and pressures that have the potential to cause stress upon the ecosystem. The scope of stressors can be global and exogenous (e.g., climate change), national or regional and endogenous (e.g., nutrient enrichment, abrasion and introduction of non-indigenous species) or local (e.g., infrastructure like bridges and coastal dams or industrial ports as well as local mussel dredging). This study includes stressors at all these spatial scales. We have included all ecologi-

Fig. 5. Results of the uncertainty analysis (Monte Carlo simulations) for the spatial human impact index. Panels A and B show the percent of simulation runs in which each grid cell was among the most (A) or least (B) impacted 25% of the study area. Panel C compares the most and least impacted 25% of the study area from the original map to the uncertainty analysis, where the red and dark blue represent areas where the original results were confirmed in at least 90% of the simulation runs and the orange and light blue areas show areas which were confirmed in at least 75% of the simulation runs. The yellow and green areas, in contrast, were identified as among the most or least impacted in the original model. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

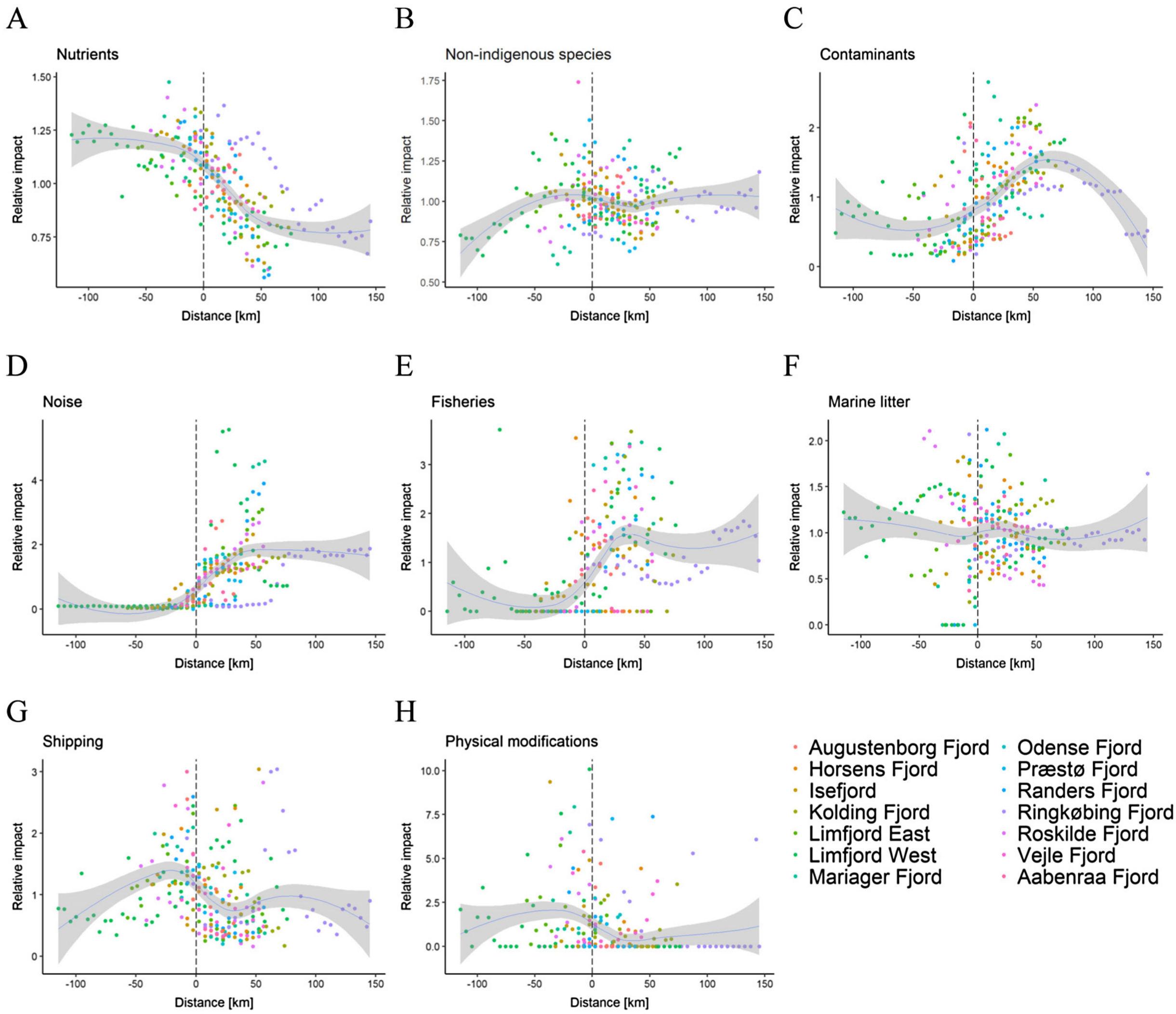


Fig. 7. Variation in relative contribution of stressors to the impact index. A) 'Winter nutrient concentrations', B) 'Non-indigenous Species', C) 'Contaminants', D) 'Noise', E) 'Fisheries', F) 'Marine litter', G) 'Shipping intensity' and H) 'Physical modifications' along a transect from fjord to open water. A local polynomial regression fitting (LOESS) is shown by a dark grey line. The 95% confidence interval is shown by the light grey shading.

cally relevant stressors according to the MSFD, as well as data representing climate change (i.e. anomalies in sea surface temperature). Seasonality is not included in the study due to the relatively stable climatic conditions in the Danish sea area, with relatively wet and warm winters and relatively cold, and sometimes also wet, summer periods. We acknowledge that seasonality would be interesting to include and study, but it's inclusion is not critical for this application of an informative and ecosystem-based cumulative impact tool.

A challenge for all cumulative impact assessments or multiple stressor analyses is the availability of high-quality and high-resolution datasets that accurately represent the pressure or ecosystem component (Dailianis et al., 2018). Hence, both the stressor datasets and the ecosystem datasets used in this study vary in quality. About 50% of the stressors are represented by a relatively good spatial coverage of the underlying data and an evaluation of the methods used, whereas 40% were critically evaluated and have been assessed to hold an acceptable quality. Given the objectives of this study, three datasets should merely be regarded as provisional 'Marine litter' (# 8), 'Non-indigenous species' (#14) and 'Recreational shipping' (# 20) (see Andersen et al. (2017) for details). The ecosystem component data is an improvement compared to earlier studies, e.g. the HELCOM HOLAS assessment (n = 13) (HELCOM, 2010; Korpinen et al., 2012) and the HARMONY project (n = 30) (Andersen and Stock, 2013). This study covers all ecologically relevant ecosystem components from phytoplankton over benthic communities to top predators like fish, seabirds and marine mammals, and most of the datasets (90%) are of high quality and robust. Some data layers originating from the HARMONY project (Andersen and Stock, 2013) have been included as these are considered important and no alternative data were available, although their spatial coverage was limited (see Andersen et al., 2017 for details).

4.2. Sensitivity interviews and risk assessment

Inaccurate setting of sensitivity weights may have an impact on the outputs (Halpern and Fujita, 2013; Jones et al., 2018). Also, the setting of effect distance of stressor layers covering a fixed area or represented as spatial data by points (ports, bridges, wind turbines, pipelines etc.) by expert judgement is a common procedure, and has been applied in other CIA studies (Ban et al., 2010; De Lange et al., 2010; van der Wal and Tamis, 2014) including the HARMONY project (Andersen and Stock, 2013) and the HELCOM HOLAS projects (HELCOM, 2010, 2018). Thus, the Delphi technique continues to be an important method for collecting information, with several applications and uses where scientists gather information from colleagues who are experts in the topic of interest. Perhaps most importantly, none of the alternatives to the currently used CIA methodology based on Halpern et al. (2008) have been as much developed, tested and applied. Furthermore, Stock and Micheli (2016) and Stock et al. (2018a) present sensitivity analyses suggesting that errors in expert judgment have similar or slightly smaller effects on broad-scale spatial patterns of modelled cumulative impact than other factors, such as the function used to transform stressor intensities, that are rarely questioned and considered standard procedures.

Regarding the setting of effect distances, we have used interviews in a similar way as HELCOM (2010), Korpinen et al. (2012) and Andersen and Stock (2013). Using linear decay for the effect distances is obviously a simplification but using actual data has not been an option, neither as part of this study nor in studies in the region initiated or led by e.g. HELCOM and national authorities. We acknowledge that actual data would be better than expert judgement and are convinced that region-specific information will emerge from other studies in the future.

4.3. Implications of the CIA model results

It is perhaps trivial to state that ecosystem components respond differently to different stressors. The different responses are not generic but will in some cases be temporally or spatially distinct. For example, seabirds are more sensitive to disturbances (e.g. different noise and shipping intensities) during the hatching period than the rest of the year. Also, seabirds might be more susceptible to contaminants during periods with low food availability, i.e. the winter period. These aspects are relevant and something that should be considered in future studies. Another feature not included in the model is the potential for recovery of an ecosystem component. Some stressors may have long-term effects (damming a fjord system) while others may only have temporary effects (a military exercise or the construction of a wind farm). Therefore, the results of this study should be seen as a snapshot of all stressors currently acting on the ecosystem components. In addition, some stressors can have negative effects on the ecosystem overall but have a positive impact on specific ecosystem components. For example, it is well documented that increases in nutrient loads lead to elevated nutrient concentrations and subsequently to a series of well-known eutrophication signals, e.g. accelerated growth of phytoplankton, increased sedimentation and in some areas decreased oxygen concentrations in bottom waters. This chain of effects is straightforward and is for most parts of the Baltic Sea and North Sea seen as a negative effect of nutrient inputs. However, the increased sedimentation of phytoplankton and detritus can, in some areas, give increased food supply to mussels at the seafloor and thus an increase in food availability for seabirds which feed on these. Due to the simplicity of the model, we cannot deal with indirect effects such as those illustrated in this nutrient-mussel-seabird example. However, where overlaps occur, the effects of the stressors can be synergistic and by mapping the potential of cumulative impacts, hotspot areas with intense human pressures can be identified.

The CIA method is well-documented, widely used and can be used for both spatial mapping as well as quantitative analyses (Halpern and Fujita, 2013; Korpinen and Andersen, 2016). Any application of the CIA methodology must, however, be accompanied by detailed descriptions of the datasets used, transparency in the setting of sensitivity weights and effects distances. These criteria have been fulfilled for this study. Further, the robustness of the main results of this study has been evaluated by means of the uncertainty analyses developed by Stock and Micheli (2016). These analyses broadly confirmed that our stressor ranking is insensitive to model assumptions and data quality.

The ranking of stressors contributing to the cumulative impact in the Danish marine areas corresponds to earlier studies, i.e. the HELCOM HOLAS project covering the Kattegat, Danish Straits and south-western Baltic Sea, and the HARMONY project covering the Danish parts of the North Sea, Skagerrak and Kattegat. Yet, this study is the first to include the stressors 'Climate anomalies', 'Non-indigenous species' and 'Marine litter', which were shown to contribute significantly to the total cumulative impact. However, the CIA results for those stressors should be considered a first estimation, as both the stressor layers for non-indigenous species and marine litter should be improved when more data becomes available. It should be noted that the ranking is an overall national-scale impact ranking (Fig. 3C) and that site-specific impact can vary between locations (Fig. 6). Therefore, stressors covering large areas with non-zero values (e.g. 'Climate anomalies') are likely to have a higher modelled impact across the whole region, than stressors present only in smaller isolated areas (e.g. 'Offshore wind turbines'). 'Climate anomalies' were excluded from the gradient studies as climate change is an exogenic stressor and acts on a larger scale than that of a certain case study (Elliott et al., 2015; Dailianis et al., 2018). Further, the exclusion is also justified by the WFD and the MSFD, which do not consider climate-change related pressures, either in the specific Initial Assessments or in the Programmes of Measures.

4.4. CIA methodology can support ecosystem-based management

Although both the WFD and the MSFD in principle are anchored in an ecosystem-based approach, they are also different regarding domain and ecosystem components included. A focal point of both is the assessment of pressures, which is worthwhile to discuss.

An ecosystem-based approach to environmental management usually 1) includes an emphasis on the protection of ecosystem structure, functioning, and key processes; 2) focuses on a specific ecosystem and the range of activities affecting it; 3) explicitly accounts for the interconnectedness within systems, recognizing the importance of interactions between many target species or key services and other non-target species; 4) acknowledges interconnectivity among systems, such as between air, land and sea; and 5) integrates ecological, social, economic, and institutional perspectives, recognising their strong interdependences (Christensen et al., 1996; McLeod et al., 2005). Given these criteria, our study contributes to the implementation of the Ecosystem Approach and to the execution of EBM for Danish marine waters, especially in the context of the MSFD, by providing an integrated view of available data describing human stressors and core ecosystem components, as well as current expert knowledge about the stressors' effects on specific ecosystem components. In addition, this study indicates that an ecosystem-based approach is not taken fully into consideration in the context of the WFD implementation, as several ecologically-relevant stressors in waters covered by the WFD are currently not considered by this directive, e.g. 'Fisheries', 'Mussel dredging', and 'Physical modifications'. The CIA approach assists in identifying areas that might be more problematic to manage (many human pressures, many stakeholders) as well as areas which are simpler to manage (few pressures; limited number of stakeholders). The approaches for integrating CIA in EBM management need to be incorporated in a structured and transparent way, see e.g. examples by Foley et al. (2017), Willsteed et al. (2018) and Dailianis et al. (2018), where common terminology and methods, setting of baselines, data access, filling data gaps and a larger incorporation of the latest research were identified as ways forward. If, as we suggest, there is an under-implementation of the ecosystem-based approach, then a potential way forward could be a closer coordination and harmonisation of the MSFD- and WFD-specific implementation processes.

On a final note, we would like to point out the need for ground-truthing, in the present study and in general, as this is important and justifies uses of the results in a management context. Ground truthing has been highlighted as an important part of the CIA methodology (Halpern and Fujita, 2013) but has not been an option due to the lack of detailed information about the environmental status of our study area. The constraints are threefold: 1) there are large spatial variation in the monitoring network covering the Danish EEZ, 2) the number of indicators used in offshore waters is limited, and 3) at present, there are no empirically measurable ecological indicators of overall ecosystem condition that would be comparable throughout our study area, from estuaries to the open sea. A ground truthing of the CIA method has previously been carried out for parts of the study area (Andersen et al., 2015) and we assume this important result is still valid, at least for the Kattegat and the western parts of the Baltic Sea. The bottleneck for ground truthing is the availability of integrated assessments of environmental status on the same scale as the CIA analyses; this is something we believe will be overcome soon, e.g. in the context of the upcoming 'Marine Messages II' (to be published by the European Environment Agency in 2020) or as part on CIA studies following up on HELCOM (2018).

5. Conclusions

Danish marine waters have an impaired status, an unfortunate situation documented by classifications of 'ecological status' of coastal waters *sensu* the WFD and 'environmental status' of marine waters *sensu* the MSFD. This is due to a wide range of sea- and land-based human activities affecting coastal and marine ecosystems. This study has therefore investigated potential cumulative impacts in Danish marine waters, as well as ranked the relative importance of key human stressors along a land-sea gradient. Evaluation of the spatial difference in stressor impact (i.e. ranking) indicates the root causes of the impairments documented in the context of the MSFD and WFD.

Based on the results of this nation-wide mapping of cumulative impacts of human activities in Danish marine waters, we summarize: 1) There are large spatial variations in the number of stressors in different parts of the Danish marine waters (Fig. 3A), 2) the number of ecosystem components (as an indicator of "ecosystem complexity") also varies greatly with high values in Kattegat and low values along the west coast of Jutland (Fig. 3B), and 3) the estimated cumulative human impacts, where the intensity of the stressors and the sensitivity of the ecosystem components are combined by means of specific sensitivity weights, also varies greatly (Fig. 3C). Highly impacted areas were found in the Wadden Sea, open parts of the Skagerrak, Limfjorden and other estuarine systems, the Danish Straits and along shipping routes in the Kattegat and western Baltic Sea, while areas with low estimated impacts were found in some offshore parts of the North Sea and Kattegat. Many of these broad-scale results were confirmed in the uncertainty analysis.

Regarding ranking of the stressors, based on a grouping of individual stressors in 9 groups, the relative importance was as follows: 1) 'Nutrients', 2) 'Climate anomalies', 3) 'Non-indigenous species', 4) 'Noise', 5) 'Contaminants', 6) 'Fisheries', 7) 'Marine litter', 8) 'Shipping intensity', and 9) 'Physical modifications'. Further, the most and least important stressor groups were confirmed to be robust to various model assumptions and data quality problems in the uncertainty analysis as well.

Based on 14 case studies in estuarine and fjord systems, we report the first ever analyses of the relative importance of stressors ('Climate anomalies' was as an exogenic stressor not included in these analyses) from land to open sea in Danish waters and conclude as follows: 1) Relative importance of key groups of stressors varies along a land-sea gradient, as expected; 2) some groups of key stressors are important in estuarine systems and coastal waters (e.g. 'Nutrients', 'Non-indigenous species', 'Contaminants' and 'Marine litter'), while others have a higher relative importance in offshore water (e.g. 'Fisheries' and 'Noise'); and 3) MSFD assessments are reaching towards a more ecosystem-based approach, while the current WFD practices concerning assessment of pressures can neither claim to be rooted in an ecosystem-based approach nor be taking the best available information about human activities and coastal ecosystem into account. Based on the results in this study, we believe we have identified a need for a closer coordination and harmonisation of the implementation of the MSFD and WFD, especially regarding pressure assessments where the methods and results of this study are MSFD-related but also relevant for the coastal waters covered by the WFD. Despite overlapping areas and threats, there are dichotomies in the implementation and reporting processes, especially regarding Initial Assessments and analyses of predominant stressors. Therefore, as both the MSFD and WFD are supposed to have an ecosystem-based approach, we suggest that future Initial Assessments under the MSFD and WFD should, where relevant, be based on the same data and methodologies, in particular the same approaches for mapping and assessing impacts of human activities.

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.

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Author contributions

JHA conceived the study and secured the funding. The datasets were provided by: ZAH, ETH, EK and CM. The software used (EcoImpactMapper) was developed by AS. ETH, EK, CM and AS did the analyses. A first draft of the manuscript was compiled by JHA, ETH and CM. All contributed to the editing and revisions of the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.135316.

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