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A roadmap towards quantitative cumulative impact assessments: Every step of the way

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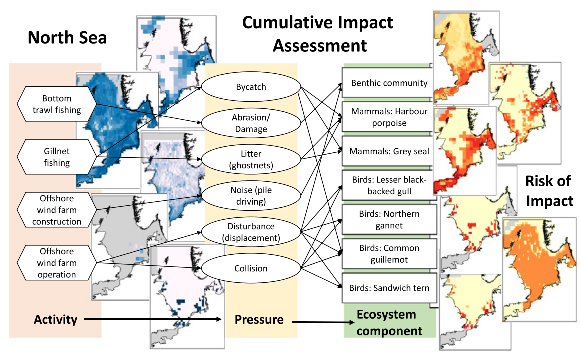
HIGHLIGHTS

- Provides a step-wise approach towards a fully quantitative Cumulative Impacts Assessment (CIA).
- Clarifies several concepts used as part of risk-based approaches.
- Reveals methodological issues to be explicitly considered in CIAs and how to resolve them.
- Introduces generic confidence criteria to guide the development and application of the CIA.
- Application of the approach in a North Sea case study provides proof of concept.

GRAPHICAL ABSTRACT



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ABSTRACT

Currently most Cumulative Impacts Assessments (CIAs) are risk-based approaches that assess the potential impact of human activities and their pressures on the ecosystem thereby compromising the achievement of policy objectives. While some of these CIAs apply actual data (usually spatial distributions) they often have to rely on categorical scores based on expert judgement if they actually assess impact which is often expressed as a relative measure that is difficult to interpret in absolute terms. Here we present a first step-wise approach to conduct a fully quantitative CIA based on the selection and subsequent application of the best information available. This approach systematically disentangles risk into its exposure and effect components that can be quantified using known ecological information, e.g. spatial distribution of pressures or species, pressure-state relationships and population dynamics models with appropriate parametrisation, resulting in well-defined assessment endpoints that are meaningful and can be easily communicated to the recipients of advice. This approach requires that underlying assumptions and methodological considerations are made explicit and translated into a measure of confidence. This transparency helps to identify the possible data-handling or methodological decisions and shows the resulting improvement through its confidence assessment of the applied information and hence the resulting accuracy of the CIA.

To illustrate this approach, we applied it in a North Sea CIA focussing on two sectors, i.e. fisheries and offshore windfarms, and how they impact the ecosystem and its components, i.e. seabirds, seabed habitats and marine mammals through various pressures. The results provide a "proof of concept" for this generic approach as well as rigorous definitions of several of the concepts often used as part of risk-based approaches, e.g. exposure, sensitivity, vulnerability, and how these can be estimated using actual data. As such this widens the scope for increasingly more quantitative CIAs using the best information available.

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1. Introduction

The development and application of cumulative (or combined) effect assessments (CEAs) and/or cumulative impact assessments (CIAs) is gaining considerable attention in scientific literature (e.g. Halpern and Fujita, 2013; Goodsir et al., 2015; Stelzenmüller et al., 2015; Judd et al., 2015; Korpinen and Andersen, 2016; Willsteed et al., 2017; Stelzenmüller et al., 2018). The terms CEA and CIA are often used interchangeably within the literature (Judd et al., 2015; Korpinen et al., 2019; Lonsdale et al., 2020) but sometimes a distinction is made. Judd et al. (2015) explains the distinction as follows: "Human activities exert pressures which have effects which may lead to impacts on receptors". Korpinen et al. (2021) follow Goodsir et al. (2015) to use combined effects when only additive effects are included while cumulative impacts "fundamentally refer to the sum of synergistic, antagonistic and additive effects on the focal environmental aspect". In this paper we will work from these definitions and consider impact as the change in state of the receptor, i.e. focal ecosystem component, as the consequence of some pressure-induced effect. Following Elliott et al. (2020) this effect may be additive, synergistic, antagonistic (compensatory), or masking but in this study we only consider addition. A choice that is in line with Judd et al. (2015) and discussed in Section 4.6.

Environmental risk assessment concepts have been used to provide a clear structure for CEA (Judd et al., 2015). Ecological (or Environmental) risk assessment (ERA) is considered a powerful framework for assessing anthropogenic changes to the environment (Gibbs and Browman, 2015; Judd et al., 2015; Stelzenmüller et al., 2018). An ERA comprises hazard identification, exposure assessment, effect assessment and risk characterisation, and is an integrated part of the risk management procedure. These four steps of the risk assessment process were first elaborated by the US national research council (National Research Council, 1983) and were adopted by the EU (EC, 2003) and their Regional Sea Conventions (OSPAR, 2003). A risk assessment approach has often been used to assess the cumulative effects of multiple human pressures on the marine ecosystem in the context of marine (or maritime) spatial planning and/or ecosystem based management (Stelzenmüller et al., 2010; Fock, 2011; Knights et al., 2015; Stelzenmüller et al., 2015).

To assess the cumulative impacts of all the human activities on the ecosystem and its components we use a linkage framework together with a risk-based approach (Knights et al., 2015). The basic elements of the linkage framework are activities, pressures and ecosystem components and how these are connected: activities can cause a range of pressures which, in turn, may impact one or more ecosystem components (Knights et al., 2013; Tamis et al., 2016; Piet et al., 2017). Different perspectives to risk-based approaches can be distinguished (Fletcher, 2005; Campbell and Gallagher, 2007; Astles et al., 2006; Kaikkonen et al., 2021), e.g. (1) based on a likelihood-consequence approach for estimating the risk of a rare or unpredictable event, often calamities (Williams et al., 2011), or (2) based on an exposure-effect approach which is considered more suitable when assessing an existing and (more or less) continuous pressure, often as part of normal operations or business as usual (Smith et al., 2007; Knights et al., 2015). These exposure-effect approaches (e.g. Bax and Williams, 2001; Stobutzki et al., 2001; Knights et al., 2015) were usually based on qualitative descriptors to assess risk to habitats and species from ongoing human activities. The application of such qualitative risk assessments, however, proved challenging to guide ecosystem-based management (e.g. Buhl-Mortensen et al., 2017) as they lack the accuracy to translate realistic mitigation measures into an advice with the level of detail that decision-makers require (Piet et al., 2015; Piet et al., 2019). Therefore in this study we will consider both the likelihood-consequence and exposure-effect approaches and show how these apply to a fully quantitative and systematic approach to calculate cumulative impact consistently across a selection of impact chains (i.e. representing the link activity-pressure-ecosystem component). With this exercise we aim to clarify part of the terminology often used as part of CEA/CIA and illustrate potential issues or choices to be made in the process of selecting appropriate data sources to be used for CEA/CIA. Moreover, because these selected impact chains cover different sectors, pressures and ecosystem components, this study can be considered a proof of concept for a single generic approach to assess cumulative impacts. This systematic approach consists of a step-wise process that applies and calculates several of the known concepts often applied in risk-based approaches and explicitly includes how available knowledge and data quality in each step contributes to the overall confidence in the assessment. This can identify knowledge gaps and drive the advancement of science and allow the piecemeal incorporation of better information once this becomes available in each iteration of the CIA development process. Ultimately, this should result in a transparent and rigorous CIA approach capable of using the best sources of, both qualitative and quantitative, information (and thus with the highest confidence scores) and allows prioritization of anthropogenic threats based on metrics with a solid science base that are intuitive and can be easily communicated to the recipients of advice. While the methodology in this study calculates both the effect caused by the pressure and the impact on the ecosystem component we will use CIA for brevity throughout this study as this is what is ultimately assessed.

2. Method

This methodology follows the general guidance for a CIA process as described by Judd et al. (2015), but without considering its actual application, i.e. the identification of management options. Thus, we here focus on the steps taken to quantify the risk or potential threat from human activities to the marine ecosystem, providing a detailed elaboration of all relevant risk aspects while explicitly considering confidence. In fact, the piecemeal iterative process to improve CIA introduced in this paper is driven by the availability of relevant information (Breen et al., 2012; Knights et al., 2015) and the confidence we have in the quality of that information.

2.1. Assessment

The linkage framework used for this CIA only considers direct linkages between stressor and receptor, i.e. impact chains, not indirect effects such as through the foodweb. Here we present a process that gradually improves a CIA one impact chain at a time through the use of the best knowledge available to estimate its impact on a specific ecosystem component and hence its contribution to the cumulative effects on that component. The combined impact chains allow an assessment of the cumulative effects on a single or on multiple receptors.

Our risk-based approach is essentially an exposure-effect approach where we systematically broke down respectively exposure and effect into methodological units (representing part of the impact chain, e.g. pressure) that can be improved (in this study fully quantified) through a step-wise process that assesses the quality of the available information and guides its application to improve the estimation of cumulative impacts.

2.2. Practical example: North Sea

For demonstration purposes we focussed on two sectors operating in the Greater North Sea in the recent past (Baseline year 2016), namely fisheries because it poses the greatest risk to Europe's regional sea ecosystems (Knights et al., 2015) and renewable energy (more specifically offshore wind farms) because it is an emerging sector that is already one of the major parties in marine spatial planning and receiving much attention regarding potential impacts on marine ecosystems (e.g. Willsteed et al., 2018). Within these two sectors we distinguished four activities: bottom trawl fishing; gillnet fishing; and offshore wind farm construction and operation. The pressures descriptions apply commonly used terms (Borgwardt et al., 2019) often with an addition (between brackets) of what it actually represented in this study: "Extraction of flora and/or fauna" (i.e. Bycatch), "Abrasion/Damage", "Litter" (i.e. Ghostnets), "Noise" (i.e. Piledriving), "Disturbance" (i.e. Displacement) and "Death or Injury by Collision". The ecosystem is considered at the most basic structural level, consisting of the following biotic ecosystem components: Birds, Mammals and Benthos. Each of these may be split up into increasingly smaller ecosystem components depending on the aim of the CIA (as identified in a scoping exercise), in practice often driven by legislative requirements but not beyond the species level. In our attempt to use quantitative information we used indicator species to represent the ecosystem components. Being aware of the limitations of and critiques on using indicator species (Zettler et al., 2013) we used more than one species if data availability permitted, which was not always the case. In this study we used Common guillemot (*Uria aalge*), Lesser black-backed gull (*Larus fuscus graellsi*), Northern gannet (*Morus bassanus*) and Sandwich tern (*Sterna sandvicensis*) to represent sea birds and Harbour porpoise (*Phocoena phocoena*) and Grey seal (*Halichoerus grypus*) to represent marine mammals. For benthos we considered the whole community. We only considered ecosystem state in terms of its structure without any inferences on the functioning of the ecosystem components important to assess the capacity to supply ecosystem services as this was considered outside the scope of this study. All elements and their relations are presented in Fig. 1, adding up to a total of 16 chains. An overview of the used data is provided in the Supplementary material (SM Tables 2.2.1 and 2.2.2). As this study aims to provide a roadmap with a methodological focus only a selection of these chains is presented here for demonstration purposes. Results of all other chains are provided in the Supplementary material (SM Table 2.2.3).

Practicality dictates that choices had to be made regarding the spatial resolution of the available information which needs to be combined into one assessment. Such choices consider among others the different levels of detail at which the incoming datasets are available, the need (or wish) to have a sensible level resolution in the outcome, the (estimated) importance of a dataset in determining the results. Finally it needs to balance the loss of (some) resolution from detailed datasets while also doing justice to the coarser resolution at which other datasets are collected. For this study the analysis has been performed at a spatial resolution of 0.25° longitude by 0.25° latitude.

2.3. The step-wise process

The step-wise process is the process that describes for each impact chain the quantification of the entire impact chain consisting of Activity, Pressure and Ecosystem component, and includes a confidence assessment (Fig. 2). This is captured in different steps that cover the main aspects of ERAs, i.e. Exposure (Table 1), Effect (Table 2), translated into an effect and subsequently (in a next step), Impact (Table 3), always explicitly considering Confidence (Table 4).

2.4. Exposure

Information on the spatiotemporal distributions of the ecosystem component abundance and the pressure magnitude can be used to calculate different endpoint indicators that capture the exposure aspect of CIA as well as other commonly used concepts such as co-occurrence and severity:

- Overlap: This may have a spatial and/or temporal dimension calculated as the proportion of space (e.g. extent as proportion of the total area, e.g. based on number of grid cells) and/or fraction of time in which both the ecosystem component and the pressure occur together. This is essentially the most basic measure of co-occurrence.
- Likelihood (of encounter): Proportion of the ecosystem component co-occurring (e.g. per grid cell) with the pressure. This only requires information on presence/absence of the pressure, not magnitude. Here the focus is on the ecosystem component and reflects the chance that the ecosystem component encounters the pressure.
- Magnitude: The proportion of the pressure co-occurring (e.g. per grid cell) with the ecosystem component. This only applies where the ecosystem component is present and does not require information on its density. Here the focus is on the pressure and reflects the magnitude that the ecosystem component encounters.
- Severity: Expresses the degree to which the ecosystem component is likely to be affected by the magnitude of the pressure and is calculated

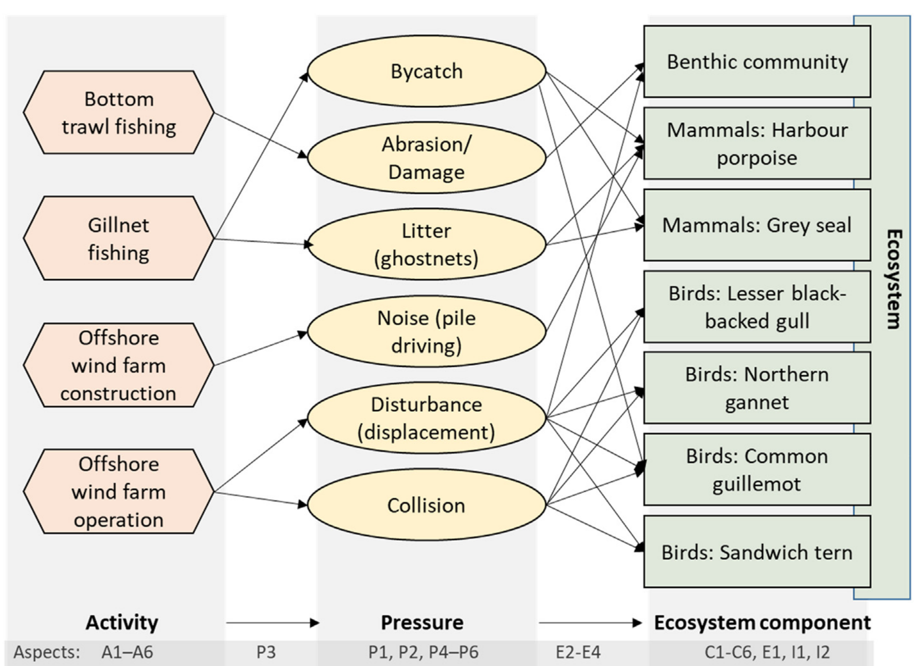


Fig. 1. Linkage framework indicating the elements and their relations selected in this study. Numbering of the exposure (A, P, C), effect (E) and impact (I) aspects corresponds to that used in Tables 1 to 3 and Fig. 2.

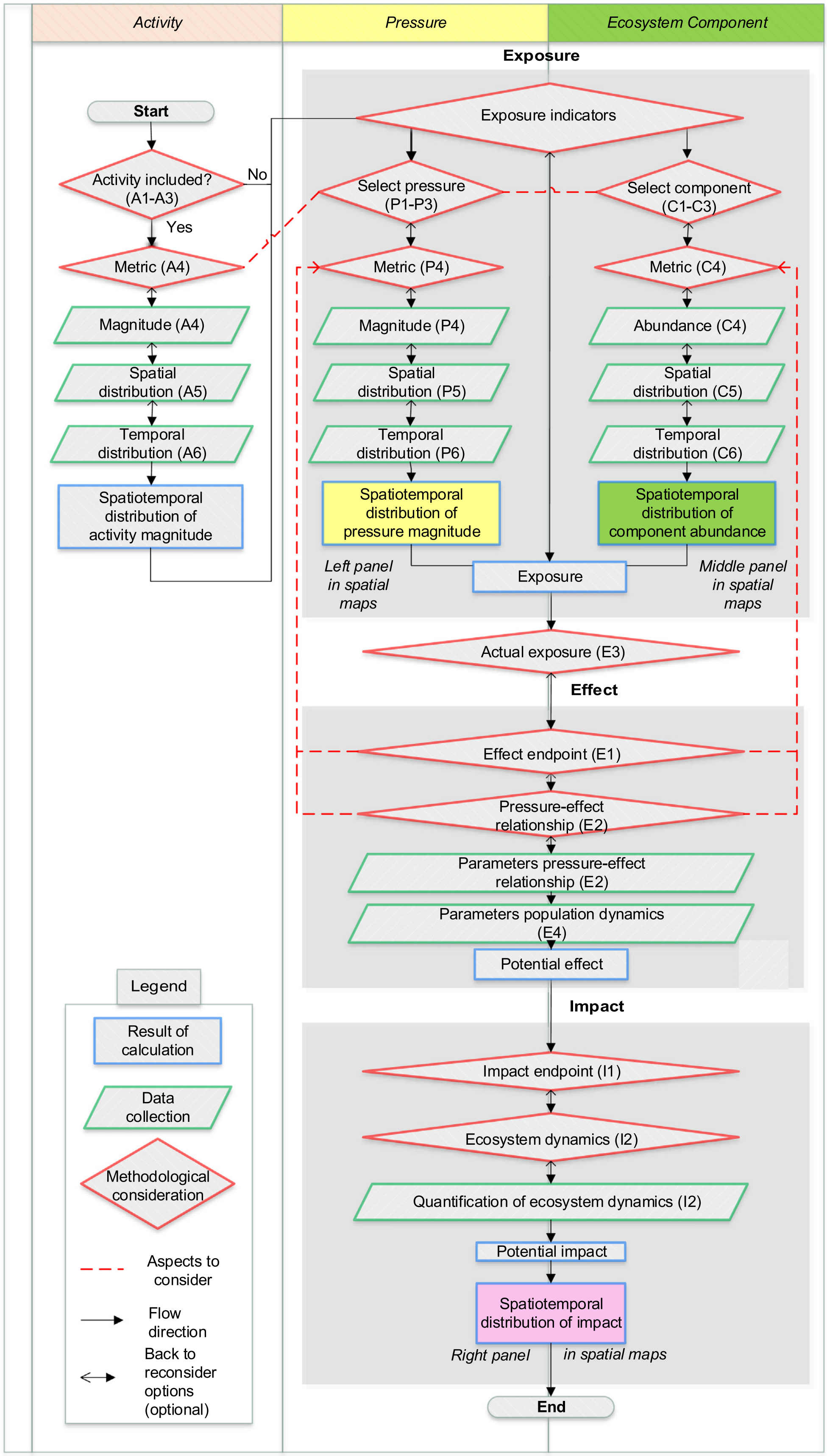


Fig. 2. Flow scheme of stepwise process and its considerations in the Cumulative Impact Assessment elaborated in this study. The different aspects of risk, i.e. Exposure (see Table 1) and Effect (see Table 2), and how these result in an Impact (see Table 3) are indicated where the letter- and number codes, e.g. A1 or P4, refer to the considerations that are covered throughout this paper. Examples of spatiotemporal distributions are presented in Fig. 3 and Fig. 5.

Table 1
Considerations applying to the Exposure aspect of risk including the calculation of its metrics. These may apply to each of the elements, Activity (A), Pressure (P), and Ecosystem Component (C), as represented by a code corresponding to the stepwise process (see Fig. 2). Note that the order of presentation here does not necessarily follow the work flow as shown in Fig. 2.

Code	Considerations and elaboration
A1 P1 C1	A1 only applies if there is a specific need to address the activities within the assessment. Information is required to assess the activity (A1–A5). A (quantitative) activity-pressure relationship is useful or might even be essential to inform decision makers in the process of regulation and planning of marine activities and/or to provide options for management. If there is no specific need, the activity could be excluded provided there is information on the pressure and activity-pressure relationship. Also consider if and/or to what extent the activity should be disaggregated into increasingly more detailed (sub)activities. Ranging from sector (i.e. windfarms) to sector-specific activities. For example, offshore windfarms as a sector could be disaggregated into construction, operation, decommissioning. For fisheries there could be even more disaggregation levels from the fishing sector as a whole into increasingly more detailed fishing métiers defined as "part of the activity of a fleet taking place in a given area, with a specific gear and targeting a specific (ensemble of) species" (ICES, 2003). This is also depending on the aim of the CIA and should be appropriate to the strategic level (policy, plan, program, regional, sectoral, project). It is important to avoid the use of different aggregation levels within the CIA as this can bias the outcome of the assessment (Piet et al., 2017). P1 can be based on a reporting of the pressure itself or as a derivative of the activity. In case of the latter the issue of dispersal applies, resulting in an increased extent through the application of a buffer around the activity (Lonsdale et al., 2020). Sources of information could be sector/industry or government (at request) or publicly available databases such as Eurostat, ODIMS or EMODnet (online search). The use of the information (A2-A5, P2-P5, C2-C5) may come with additional data requirements, e.g. preferred metric and format, sufficient spatial/temporal scale, —resolution, precision and accuracy, driving the selection of appropriate source(s).
A2 P2 C2	If the information is not available in the exact format as required for the assessment, data processing may be required. For example data transformation to match the geographic coordinate system and resolution of use, e.g. from ICES rectangles to the World Geodetic System 1984 (WGS84) with a grid size of 0.25 × 0.25 degrees.
A3 P3 C3	This aspect specifically addresses the confidence based on the quality of the information, see Table 4. The selection of appropriate activities (A), pressures (P) or ecosystem components (C) is part of the scoping exercise. The selection of appropriate ecosystem components also determines the assessment endpoint (I1,Table 3) and may involve choices on what can be considered an appropriate organization level, i.e. organism-, population- or community-level, and/or a consideration of several species per component that together cover the whole spectrum from best case to worst case (see Table 2).
A4 P4 C4	What is the best metric (e.g. amount, frequency of occurrence, concentration) to express the magnitude of the element? This is primarily determined by the availability of information. Other considerations may include: for A4 the metric should allow the use of an activity-pressure relationship (see A1), i.e. x amount of activity leads to y amount of pressure; and for P4 and C4 the metric should allow application of an appropriate pressure-effect relationship (see Table 2).
A5 P5 C5	Is the spatial distribution of the element available in an appropriate resolution? The resolution is considered appropriate when it matches the resolution of the other elements (i.e. related pressures and ecosystem components) and is sufficient to meet the aim of the study. A lower resolution might compromise the ability to provide an answer to the research question whereas a higher resolution might lead to unnecessary data demand. To what extent is the study area covered? The spatial distribution should preferably cover the entire study area, or at least the most important/representative parts (Table 2). How is the spatial distribution of the element used? See magnitude precision and accuracy (Section 2.7 on confidence). For example, for offshore wind farms spatial information has highest precision and accuracy because exact locations of wind farms are known from a reliable source.
A6 P6 C6	Is the temporal distribution of the element available in an appropriate resolution? Are relevant temporal issues considered, e.g. seasonality in the spatial distribution of an activity or ecosystem component? To what extent is the time period covered? The temporal distribution should preferably cover the entire time period, or at least be sufficiently representative for the intended time period.

Table 2
The quantification considerations applying to the Effect aspect represented by a code corresponding to the stepwise process (see Fig. 2). For further detail see Table 1.

Code	Considerations and elaboration
E1	The chosen organization level: organism-, population- or community-level should be representative of the ecosystem component and allow processing in the subsequent steps.
E2	Different types of relationship between ecosystem component and pressure may apply, e.g. dose-response (linear, logistic) or binary (effect vs no effect). The shape of the pressure–effect relationships of the involved pressures are often unknown and assumed to be linear (Halpern and Fujita, 2013; Judd et al., 2015). Information on the sensitivity of the ecosystem component to the pressure is required to quantify the relationship, e.g. parameterise the slope of the relationship. The quality of the information available determines the level of confidence (see Table 4).
E3	When a pressure and ecosystem component co-occur (in time and space) it can be assumed that the ecosystem component is fully exposed to the pressure magnitude. However, in some cases it is known that the ecosystem component is likely to actively avoid or is attracted to the pressure. For example, the actual chance of birds colliding with offshore wind turbines can be expected to be less than the co-occurrence data suggest. Taking this into account is challenging, as this depends on (knowledge of) the animals' behaviour, see e.g. Aarts et al. (2016).
E4	The chosen organization level (E1): organism-, population- or community-level determines the required parameters for the growth and depletion processes. Any change in the population growth rate requires a change of the rate of at least one of the underlying demographic processes: reproduction, survival, emigration or immigration. Pressures affecting the demographic processes (E2) need to be translated into a change of population growth rate, e.g. using demographic parameters. The quality of the information available determines the level of confidence (see Table 4).

as the proportion of the ecosystem component exposed to the proportion of the pressure load. This, therefore, requires spatiotemporal information on pressure magnitude and ecosystem component density. This is the most elaborate indicator that reflects the chance the ecosystem component encounters a certain pressure magnitude.

In this study the above exposure endpoint indicators were estimated quantitatively by collecting spatially explicit information on the ecosystem component abundance and the pressure magnitude on a, for the present study, predefined spatial grid with rectangular 0.25 by 0.25 degree grid cells of the study area (Aspects A3, A4, P5, P6, C5 and C6 in Table 1 and Fig. 2). If spatially-explicit information is lacking the worst-case (and most risk-averse) assumption is 100% exposure but of course any other percentage can be assumed depending on how risk-averse the CIA needs to be. Knights et al. (2015) resolved this by applying a semi-quantitative score for spatiotemporal overlap based on expert judgement.

Table 3
The quantification considerations applying to the (Risk of) Impact represented by a code corresponding to the stepwise process (see Fig. 2). For further detail see Table 1.

Code	Considerations and elaboration
I1	An appropriate metric (e.g. numbers or biomass) to capture ecosystem state should be chosen and that can be easily understood by those contributing the information required and developing and applying the CIA as well as the recipients of advice. The appropriate organisational level is mostly determined by the purpose of the study. Policy objectives, such as those from Marine Strategy Framework Directive (EC, 2008), Birds Directive (EC, 2009) and Habitats Directive (EC, 1992), often require population- or community-level impact assessments.
I2	The approach to capture the ecosystem components dynamics needs to be sound and with a solid scientific basis that may include sensitivity and life-history characteristics and their parametrisation. We applied semi-chemostat dynamics as this allows for a population growth rate that not only depends on reproduction but also immigration into the grid cell which was deemed more appropriate and allows for recovery within that grid cell even if completely depleted.

Table 4
Confidence classification criteria for aspects and elements, i.e. Activity (A), Pressure (P), and Ecosystem Component (C), addressed in this study. Aspect codes corresponding to those in Fig. 2 and Tables 1–3 are extended to reflect aspects that needed to be distinguished for classification of confidence. Note that a reliable source is considered as any source that has competence in the field of interest. This includes but is not restricted to peer-reviewed literature or (broadly recognized as) authoritative (inter)national data portals.

	Aspect	High (1)	Moderate to high (0.8)	Moderate (0.6)	Low to moderate (0.4)	Low (0.2)
A2 P2 C2	Data processing	No data processing required	Some processing required, but only format change. No data transformation	Some processing required, including minor data transformation	Processing required, including data transformation	No spatial information. Single point value
E3	Actual exposure	The actual exposure in the grid cells where C and P co-occur is well known and fully quantified	The actual exposure in the grid cells where C and P co-occur is well known but issues with quantification	The actual exposure in the grid cells where C and P co-occur is not precisely known but based on assumptions from a reliable source	The actual exposure in the grid cells where C and P co-occur is not precisely known and unfounded assumptions were required	The actual exposure in the grid cells where C and P co-occur is unknown (but assumed to be 100%)
A4b P4b C4b E1b I1b	Metric suitability	Best possible representation	Is a proxy based on well-known relationship and covering the relevant pressure properties	Is a proxy based on founded assumptions and covering much of the relevant pressure properties	Is a proxy based on unfounded assumptions, covering only some of the relevant pressure properties	No metric used
A4c P4c C4c	Estimation metric Magnitude Abundance	Exact data from a reliable source, not based on assumptions and/or modelling	Data from a reliable source, not based on assumptions and/or modelling	Data from a reliable source, based on founded assumptions and/or modelling	Data, based on unfounded assumptions and/or modelling	No metric used. Single point value (presence/non-presence)
A5a A6a P5a P6a C5a C6a	Spatial / Temporal resolution	Resolution exactly represents the element	Resolution is appropriate	Resolution is slightly lower	Resolution is much lower	Not used. Single point value
A5b A6b P5b P6b C5b C6b	Spatial / Temporal coverage	Extent covers the entire relevant area and/or representative time period	No complete coverage, but sufficient to be representative (covering >80%, including the main parts)	Covers a substantial part of the area / time frame (covering appr. 50-80%)	Only a small part of the area / time frame (<50%) is covered	Not used. Single point value

Table 4 (*continued*)

E2c	**Parameters P-E**	The parametrisation of the relationship between magnitude and effect is known from a reliable source and is well established	The parametrisation of the relationship between magnitude and effect is known from a reliable source	Parametrisation of the relationship is estimated, based on data from a reliable source	Parametrisation of the relationship is estimated, based on assumptions from a reliable source	Parametrisation of the relationship is estimated
E4	**Parameters population dynamics**	Parameters are from reliable sources and with little variation	Parameters are from reliable sources and large variation	Parameters are from less reliable sources and little variation	Parameters are from less reliable sources and large variation	Parameters are based on (unfounded) estimations
I2	**C dynamics approach suitability**	Best represents the dynamics of the ecosystem component, based on a reliable source, suitable for study aim and endpoint	Is a proxy based on a reliable source, suitable for study aim and endpoint	Is a proxy (partly) based on unfounded assumptions, suitable for study aim and endpoint	Is a proxy with poor suitability	No ecosystem components dynamics used
E2c	**Parameters P-E**	The parametrisation of the relationship between magnitude and effect is known from a reliable source and is well established	The parametrisation of the relationship between magnitude and effect is known from a reliable source	Parametrisation of the relationship is estimated, based on data from a reliable source	Parametrisation of the relationship is estimated, based on assumptions from a reliable source	Parametrisation of the relationship is estimated
E4	**Parameters population dynamics**	Parameters are from reliable sources and with little variation	Parameters are from reliable sources and large variation	Parameters are from less reliable sources and little variation	Parameters are from less reliable sources and large variation	Parameters are based on (unfounded) estimations
I2	**C dynamics approach suitability**	Best represents the dynamics of the ecosystem component, based on a reliable source, suitable for study aim and endpoint	Is a proxy based on a reliable source, suitable for study aim and endpoint	Is a proxy (partly) based on unfounded assumptions, suitable for study aim and endpoint	Is a proxy with poor suitability	No ecosystem components dynamics used

Table 5
The impact chains and their methodological options that were considered in this study with their calculated endpoint indicators. For chain 2 the methodological options differed in terms of the spatial distribution which was based on quarterly (2I) or yearly (2II) data, for chain 5 the options differed because the pressure distribution was based on location of ship wrecks (5I), caught litter (5II) or location of set nets (5III), for chains 13–16 the options varied depending on whether displacement was considered (I) or not (II).

ID	Elements			Exposure				Impact	
	Activity	Pressure	Ecosystem component	Overlap	Likelihood	Magnitude	Severity	Total	Local
1	Bottom trawl fishing	Abrasion	Benthic community	72.3	95.0	81.3	0.049	4.44E+00	0.9
2I	Gillnet fishing	Extraction	Common guillemot	27.5	39.7	36.5	0.014	1.18E−05	2.8
2II	Gillnet fishing	Extraction	Common guillemot	27.5	39.7	36.5	0.014	1.27E−05	2.8
3	Gillnet fishing	Extraction	Harbour porpoise	24.5	40.8	25.3	0.014	3.96E−02	6.4
4	Gillnet fishing	Extraction	Grey seal	47.7	50.6	99.9	0.051	2.12E−01	2.2
5I	Gillnet fishing	Litter	Harbour porpoise	42.9	68.5	66.9	0.034	2.31E−03	3.3
5II	Gillnet fishing	Litter	Harbour porpoise	46.0	84.7	73.9	0.053	3.71E−03	1.0
5III	Gillnet fishing	Litter	Harbour porpoise	24.5	40.8	25.3	0.014	9.49E−04	6.5
6I	Gillnet fishing	Litter	Grey seal	62.5	66.2	100.0	0.050	4.88E−03	1.6
6II	Gillnet fishing	Litter	Grey seal	64.3	68.0	100.0	0.050	5.09E−03	0.1
6III	Gillnet fishing	Litter	Grey seal	47.7	50.6	99.9	0.051	5.20E−03	2.3
7	Offshore wind	Noise	Harbour porpoise	1.2	1.1	41.2	0.006	1.26E−02	13.3
8	Offshore wind	Habitat loss	Benthic community	5.3	6.6	95.1	0.053	7.44E−04	6.5
9I	Offshore wind	Disturbance	Common guillemot	5.3	13.8	95.3	0.094	2.53E+00	4.7
10I	Offshore wind	Disturbance	Lesser black-backed gull	5.3	15.0	95.3	0.149	4.79E−01	2.8
11I	Offshore wind	Disturbance	Northern gannet	5.3	10.7	95.3	0.094	3.89E−01	2.2
12I	Offshore wind	Disturbance	Sandwich tern	4.3	23.2	79.9	0.220	2.66E+00	10.6
13I	Offshore wind	Collision	Common guillemot	5.3	13.8	95.3	0.094	3.69E−01	4.7
13II	Offshore wind	Collision	Common guillemot	5.3	13.8	95.3	0.094	4.09E−01	4.7
14I	Offshore wind	Collision	Lesser black-backed gull	5.3	15.0	95.3	0.149	4.82E+00	2.8
14II	Offshore wind	Collision	Lesser black-backed gull	5.3	15.0	95.3	0.149	5.19E+00	2.8
15I	Offshore wind	Collision	Northern gannet	5.3	10.7	95.3	0.094	2.34E+00	2.2
15II	Offshore wind	Collision	Northern gannet	5.3	10.7	95.3	0.094	2.54E+00	2.2
16I	Offshore wind	Collision	Sandwich tern	4.3	23.2	79.9	0.220	4.35E+00	10.6
16II	Offshore wind	Collision	Sandwich tern	4.3	23.2	79.9	0.220	4.74E+00	10.6

When selecting data to estimate the spatiotemporal distributions of the pressure and the ecosystem component it essentially boils down to: what information is available with sufficient quality; and what is feasible to improve the quality? Here the choice of an appropriate metric is important. Whereas abundance or density is an obvious choice for the ecosystem component this is not always this straightforward for the pressure metric (see Table 1 and Fig. 2).

Often information on the spatiotemporal distribution of the pressure may be lacking. The first option is then to derive this from that of the activity (Elliott et al., 2020), possibly complemented with a pressure-specific dispersal based on the buffer according to Lonsdale et al. (2020). This may also have advantages if the CIA is to be applied to inform operational management which is often sector-specific (Cormier et al., 2017). If adequate quantitative information on the spatial distribution of the pressure exists then steps A1 up to A5 (Table 1 and Fig. 2) could be effectively skipped for the impact chains containing that pressure. For example for the pressure ‘physical damage’ caused by bottom trawl fisheries (chain 01, Table 5), information on the swept area ratio, i.e. the summed area contacted by a fishing gear within a grid cell over one year divided by the surface area of the grid cell (ICES, 2019), was readily available from ICES (2018) and information on the occurrence of the fishing vessels was therefore not necessary. The bottom trawl fisheries comprised four fishery types each with its specific depletion rate for fishing on sublittoral sediment: otter trawl (0.06); beam trawl (0.14); towed dredge (0.20) (Hiddink et al., 2017); demersal seine (0.016) (Rijnsdorp et al., 2020). For other pressures the metric was assumed identical to that of the activity, such as for bycatch in gillnet fisheries (chains 02I, 02II, 03 and 04, Table 5), where both can be expressed as hours fished per square kilometer. Another example is the pressure ‘ghost nets’ (chains 05I, 05II, 05III, 06I, 06II and 06III from Table 5) where kilometer lost nets could be a sensible metric for the pressure. However, both the amount and spatiotemporal distribution of ghost nets is not well known. Different assumptions on the fraction of set nets that are lost and its spatial distribution are used (and evaluated for their respective confidence later on) to obtain a proxy of the pressure, which is described in the illustrated example in Section 3.1. Criteria based on spatiotemporal coverage and resolution can be used to select the best quality data (Aspect P5 and P6 in Table 1 and Fig. 2). For the choice of metric, however, it is not only the data availability and quality that should be considered. The metric should also be suited to express the relationship with the effect on the ecosystem component (Aspect P4a in Table 1 and Fig. 2), as described in Section 2.5.

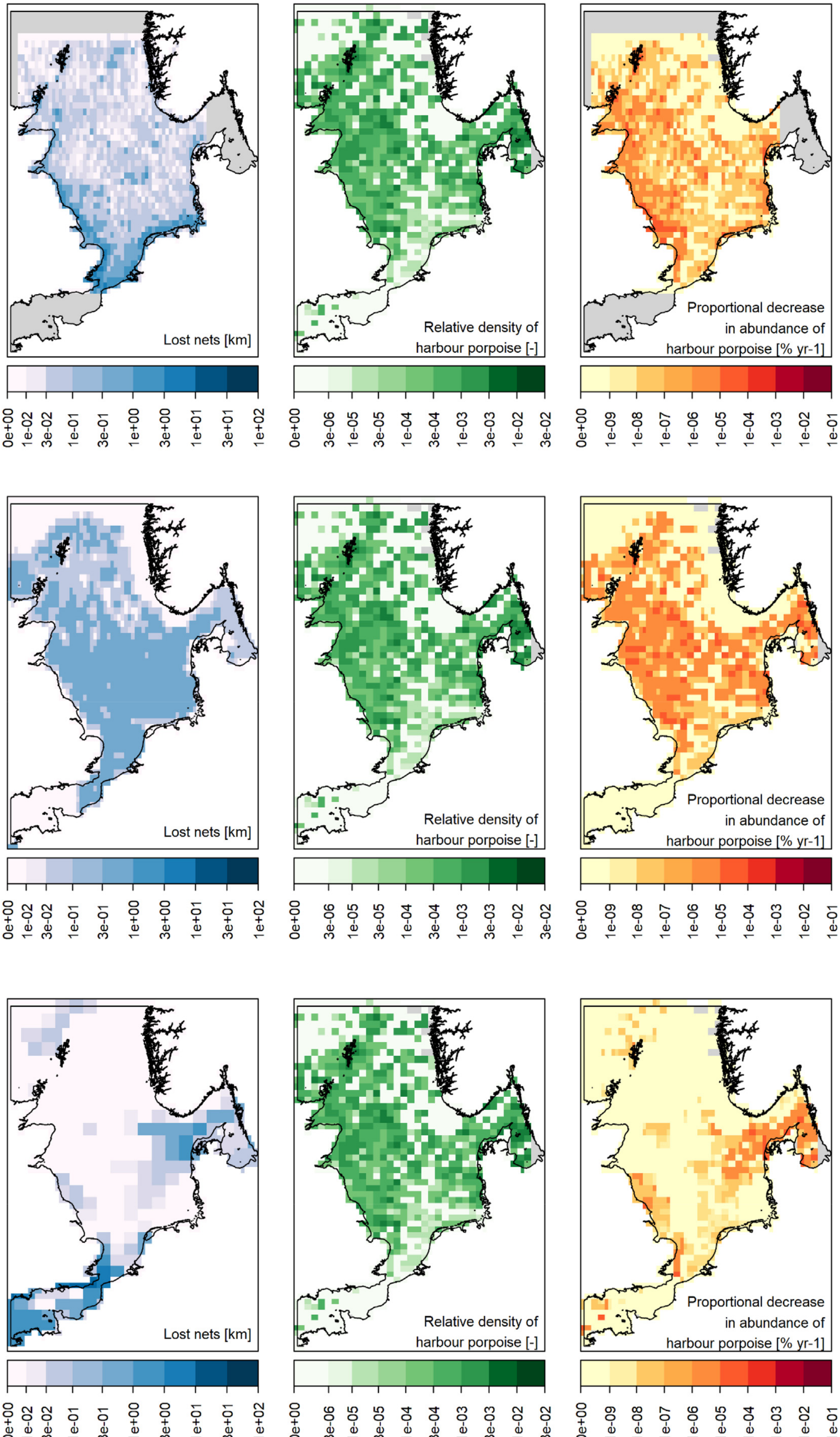
For each ecosystem component it is assumed that the study area, i.e. Greater North Sea as specified in the main policy framework the Marine Strategy Framework Directive (EC, 2008), contains the entire population of the indicator species and that its distribution is stationary according to a single (or several in case of e.g. seasonality) fixed spatial distribution map of the species. The assumption of stationarity may apply for some ecosystem components (e.g. sessile benthic species (chains 01 and 08 from Table 5) but not for mobile species such as birds. Therefore the sensitivity of the outcome of the CIA to such assumptions was studied by using quarterly information on spatial distribution for the Common guillemot compared to yearly averaged distribution (chain options 02I and 02II, Table 5). Without such information, a single fixed time-averaged spatial distribution is the default option.

The spatial distribution of the pressure and the ecosystem component are shown in respectively the left and middle panels of Fig. 3 and Fig. 5.

2.5. Effect

When the ecosystem component is exposed to the pressure, the severity of the effect of this pressure on the ecosystem component needs

Fig. 3. Spatial distribution maps (grid size = 0.25 × 0.25 degrees) for the impact chain “Gillnet fishing - Litter (ghost nets) – Harbour porpoise” distinguishing three methodological options based on locations of respectively shipwrecks, marine litter and set nets (ID = 5I, 5II and 5III) from top to bottom) described in Table 5. The panels show from left to right the pressure (expressed as km nets lost, based on different assumptions), ecosystem component (Harbour porpoise relative density, same in all cases) (expressed as fraction with total sum = 1)) and impact (proportional decrease in abundance (% yr^{-1}) caused by the pressure).


Lost nets [km]
0e+00 1e-02 3e-02 1e-01 3e-01 1e+00 3e+00 1e+01 3e+01 1e+02
Relative density of harbour porpoise [-]
0e+00 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02
Proportional decrease in abundance of harbour porpoise [% yr-1]
0e+00 1e-09 1e-08 1e-07 1e-06 1e-05 1e-04 1e-03 1e-02 1e-01
Lost nets [km]
0e+00 1e-02 3e-02 1e-01 3e-01 1e+00 3e+00 1e+01 3e+01 1e+02
Relative density of harbour porpoise [-]
0e+00 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02
Proportional decrease in abundance of harbour porpoise [% yr-1]
0e+00 1e-09 1e-08 1e-07 1e-06 1e-05 1e-04 1e-03 1e-02 1e-01
Lost nets [km]
0e+00 1e-02 3e-02 1e-01 3e-01 1e+00 3e+00 1e+01 3e+01 1e+02
Relative density of harbour porpoise [-]
0e+00 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02
Proportional decrease in abundance of harbour porpoise [% yr-1]
0e+00 1e-09 1e-08 1e-07 1e-06 1e-05 1e-04 1e-03 1e-02 1e-01

to be estimated. This requires an estimate of the magnitude of the pressure and a relationship with some characteristic of the ecosystem component that translates this into an effect. We call this a pressure-effect relationship but in other specific contexts this is also referred to as a pressure-state relationship in case of fisheries (Jennings, 2005) or dose-response relationship for contaminants (Ritz, 2010).

The pressure-effect relationship and its scope for parametrization depends on the choice of the pressure metric, the effect metric and selected representant of the ecosystem component (Aspects P4 and C4 respectively in Table 1 and Fig. 2). The parametrization and definition of the effect is based on two processes which, depending on the organization level i.e. organism-, population- or community, may be named differently but should represent respectively growth and depletion which ultimately determine how the ecosystem component will be impacted by the pressure. These different organization levels are often the cause of confusion (Suter et al., 2005) as different names and parameters may apply. For example at the organism-level this can be e.g. respectively survival (at some specific stage) and reproduction (pups per female), while at the population-level it can be e.g. respectively mortality and reproduction rates and at community-level, e.g. depletion or growth rates. Effect is defined as a certain degree of change in one, or both of the processes.

To describe the pressure-effect relationship (Aspect E2 in Table 2) we may consider both the type and the parameterization of this relationship. The state-of-the art knowledge of pressure-effect relationships varies considerably per pressure. For toxicants, exposure-effect relationships are well known and assumed to follow a log-normal distribution with the EC50 (concentration causing 50% effect to the exposed test population after a standardised exposure time) as midpoint and the standard deviation of the distribution as representative for the slope of the relationship (Smit et al., 2001). For most pressures, however, information on these relationships is lacking and, often strong, assumptions need to be made, on the parameterization but sometimes even on the type of relationship (e.g. linear, exponential). Examples are provided below to illustrate potential issues that may occur, the proverbial “bumps in the road” as one goes “every step of the way”.

For the pressure ‘ghost nets’ (chains 05I, 05II, 05III, 06I, 06II and 06III from Table 5) kilometer lost nets was suggested as a sensible metric for the pressure (see Section 2.4). Besides the net length, there are many factors determining the effect of the pressure ghost fishing, such as gear type, location (including the depth, substrate material, degree of protection from wave energy, presence of features where the gear can become entangled), gear design and materials (FAO, 2016). Quantitative relationships between the pressure magnitude and the degree of ecosystem effects are not well known (FAO, 2016). Here, we quantify the relationship between the pressure ‘ghost nets’ (in km net length) and the effect (mortality) on marine mammals using assumptions and available knowledge. First, a linear pressure-effect relationship is assumed (Halpern and Fujita, 2013; Judd et al., 2015). This relationship is then parameterized by describing the slope of the relationship, i.e. the mortality rate. Annual mean entanglement rates by fishing gear were reported for grey seals at Cornwall UK (Allen et al., 2012). Mortality rates of entangled seals are increased (Allen et al., 2012) but by how much is unknown and we assumed this to be 50% (half of the entangled seals die). Based on the entanglement rate of 0.043 (fraction of population getting entangled (Allen et al., 2012)) and the 50% mortality assumption, the mortality rate is estimated at 0.02. This can be considered a mortality rate that applies both at population-level as well as organism-level (Suter et al., 2005). Because the effect for this chain is mainly based on assumptions, the confidence is very low which is reflected in the confidence assessment in Section 3.4.

Several alternative metrics to express the pressure may exist. Underwater noise from pile driving (chain 07 from Table 5), for instance can be expressed in terms of amplitude ratios at specific frequencies (in decibels), or pulse block days (EU, 2017). Although for the first metric a better pressure-effect relationship exists (Thompson et al., 2013; Finneran and Jenkins, 2012; Miller et al., 2014), the latter is readily available for specific periods and is therefore used in the present study.

2.6. Impact

Usually CIAs apply risk-based approaches determining the risk that the ecosystem is impacted and policy objectives are not achieved (Breen et al., 2012; Knights et al., 2015). This process of quantification also allows (or even forces us) to better define this concept of risk into a more tangible assessment endpoint which can be better understood by those contributing to the CIA and more easily communicated to the recipients of advice.

The process to estimate the two components of risk, i.e. exposure and effect, required us to incorporate spatiotemporal distributions of the pressure and ecosystem component, and in case of co-occurrence (Section 2.4) how this translates into an effect on the (population) dynamics of the ecosystem component (Section 2.5) which then determines the impact, i.e. change in state of the ecosystem component (this section). If this involves several species representing specific ecosystem components this may first require the aggregation of species-specific impacts into that of the ecosystem component. If possible we used at least one but preferably more indicator species to represent the ecosystem components. We also provide an example where the ecosystem component, i.e. “Sublittoral sediment”, is represented by habitat-specific benthic communities rather than one or more species as this was deemed more appropriate.

Ultimately the impact on ecosystem state (i.e. the ultimate assessment endpoint) should be expressed in one common metric to allow aggregation across impact chains as is required for the CIA. Here we propose the equilibrium abundance relative to an undisturbed situation as the assessment endpoint.

To calculate this assessment endpoint we assumed semi-chemostat dynamics to model the abundance of the ecosystem component because these allow a potential inflow, for example from an unmodeled juvenile stage, or immigration from outside the modelled area. This has the advantage that it is possible to recover from a locally extinct population. Semi-chemostat dynamics are given by:

$$\frac{dN}{dt} = r(K - N) - dN \tag{1}$$

where, N is the abundance, r is the maximum population growth rate, d the depletion rate and K the carrying capacity (=abundance in an undisturbed or pristine situation).

This can be solved into

$$N_t = \frac{(d+r)^*N_0 - K^*r^*\left(1 - exp^{\,(d+r)^*t}\right)}{(d+r)^*\, exp^{\,(d+r)^*t}} \tag{2}$$

By setting (1) to equilibrium ($dN/dt = 0$) and rearranging, we find an equilibrium abundance of

$$N_{eq} = \frac{r}{(r+d)} {}^*K \tag{3}$$

with the special attribute that at $N_{eq} = ½\, K$ applies $r = d$. We consider this the baseline $N_{eq,B}$. Both d and r can be affected by the pressure. When $d = 0$, the equilibrium abundance N_{eq} is present at carrying capacity K. Setting K at 100% the potential impact is 100-N_{eq} indicating the potential loss of the ecosystem component if this pressure continues at that magnitude ad infinitum. Thus the potential loss relative to an undisturbed situation (i.e. carrying capacity) is determined by a growth rate and depletion rate.

For the seabirds and marine mammals the population growth- and depletion rates, r and d, could then be estimated from M the species-specific proportion annual mortality (yr^{-1}) per life-stage using parameters from the literature (Supplementary material, SM Table 2.6.1) calculated according to

$$M = \left(\frac{(1-Cj)^{*}Tj + (1-Ca)^{*}Ta)}{(Tj + Ta)}\right. \quad (4)$$

where: $Cj =$ Proportion annual survival (yr^{-1}) in the juvenile stage, $Ca =$ Proportion annual survival (yr^{-1}) in the adult stage, $Tj =$ average duration (yr) of the juvenile stage and $Ta =$ maximum duration (yr) of the adult stage. Which allows the calculation of the annual, thus $t = 1$ in (2), population depletion rate assuming there is no recovery ($r = 0$) as:

$$d = Ln\left(\frac{1}{1-M}\right) \quad (5)$$

Assuming an equilibrium situation at N_{eq} and hence $r = d$ allows the calculation of r. Any change in d or r caused by a (change in) the pressure resulting in a change (C), can then be calculated as

$$\Delta d = \Delta r = Ln\left(\frac{1}{1-(M-C)}\right) \quad (6)$$

Here $C > 0$ represents decreased survival or reproductive capacity and thus $(M - C) < 1$.

Depending on the (change in) pressures causing for example a change in mortality (affecting d) or in reproductive capacity (affecting r) a new equilibrium abundance can be calculated as

$$N_{eq,I} = \frac{(r + \Delta r)}{((r + \Delta r) + (d + \Delta d))}{}^{*}K \quad (7)$$

The impact I (proportion additional yearly mortality, yr^{-1}) can be calculated as $I = (N_{eq,B} - N_{eq,I}) / N_{eq,B}$. This can be considered as the potential impact, assuming the ecosystem component is present.

For the benthic community we used the Population Dynamics (PD) approach from Rijnsdorp et al. (2020) which estimates the potential impact of bottom trawling (I) in terms of the reduction in the benthic biomass (B) relative to the carrying capacity (K) of the habitat (Hiddink et al., 2019; Pitcher et al., 2017). This PD approach is based on empirical estimates of gear-specific depletion rates from a meta-analysis by Hiddink et al. (2017) and a recovery rate that is a function of the habitat-specific benthic community composition in terms of longevity and which is estimated from a meta-analysis (Hiddink et al., 2019).

The actual impact can be calculated per grid cell by multiplying the pressure-induced potential impact I with the normalized density of the ecosystem component (i.e. proportion of population) in each spatial grid cell where pressure and ecosystem component co-occur. Underlying assumption for this approach of simply combining the potential impact distribution with the ecosystem component distribution is that the co-occurrence in a specific grid cell implies full exposure to the pressure magnitude. This results in an impact density map where the assessment endpoint (i.e. % change in equilibrium abundance relative to an undisturbed situation) per grid cell is shown. The total impact is obtained by summation of the values of all grid cells. An example of the calculation is presented in the Supplementary Material (SM Table 2.6.2). Note that for this proof of concept, impacts are calculated for each separate chain. As the assessment endpoint for each chain consistently represents a change in equilibrium abundance of the ecosystem component, a fair comparison of impacts between the chains becomes possible (keeping in mind the quality of the information and assumptions used for each chain).

2.7. Confidence

Classification criteria for confidence assessment were developed (Table 4) using the following generic hierarchy levels (but with deviations if needed): best possible; well-known; founded assumptions; unfounded assumptions; not used. These classification criteria were applied to the aspects for all elements and their relations (Fig. 1). For each chain, individual scores were used to produce an average confidence score per Activity (A), Pressure (P), and Ecosystem Component (C). Additional detail is in Supplementary material (SM Table 2.7.1).

3. Results

For each selected impact chain, with in some cases different methodological options, we have applied the step-wise approach to calculate potential endpoint indicators relevant for CIA (Table 5). These CIA endpoint indicators include the four exposure endpoint indicators (see Section 2.4) and the total impact (see Section 2.6). The Local Impact, calculated in addition to Total Impact, is not intended as a potential endpoint indicator but considered informative to inform management (notably Marine or Maritime Spatial Planning, MSP) as it provides the % contribution of the single most-impacted grid cell to the total impact. Theoretically every indicator should have a value between 0 (no potential impact) and 100 (potential extinction).

As proof of concept we show for two of the impact chains, i.e. impact of marine litter from fisheries, i.e. ghost nets, on harbour porpoise and impact of windfarm collisions on seabirds, how the application of the step-wise approach leads to the result presented in Table 5. The other impact chains are documented in the Supplementary material (SM Figs. 3.1.1–3.1.19).

3.1. Impact of ghost nets on harbour porpoise

This example (Figs. 3 and 4) shows three options to assess the impact of the pressure ghost nets (km nets lost) on harbour porpoise. The distribution of harbour porpoise is identical for each option and based on data from the Atlas of Cetacean distribution in north-west European waters (Reid et al., 2003). Although more recent data exist from the SCANS-III surveys, this was not readily available and thus not included in this study. The available information on the pressure magnitude and spatial distribution is not available for the preferred metric (km nets lost). Instead, the total net loss in the study area was estimated by taking the average of reported values for nets lost in the North East Atlantic (FAO, 2016): 0.354% of the nets set. The amount of nets set in the North East Atlantic was not reported. Therefore, the gill net fishery effort in the Dutch sector of the North Sea (7919 km net length/yr (Scheidat et al., 2018)) was used, resulting in approximately 28 km net length lost per year. This was extrapolated to the Greater North Sea according to the ratio between fishing effort for the Dutch sector of the North Sea (2122 days/year (Scheidat et al., 2018)) versus that in the Greater North Sea (44,165 days/year (ICES, 2015)) giving an estimated total of 583 km nets lost per year in the study area as a proxy for the total pressure load. For the spatial distribution of the pressure we considered three options based on different assumptions allowing the use of different sources of spatial data:

I) loss of nets is caused primarily by entanglement in shipwrecks, thus using locations of ship wrecks. This gives a patchy distribution with the highest densities along the coasts of England, Belgium, the Netherlands and Germany (option 5I, Fig. 3 top row)

II) occurrence of lost nets is identical to that of marine litter caught by fishermen, thus using marine litter data. This gives a more even distribution spread across the entire study area (option 5II, Fig. 3 centre row)

III) lost nets remain (more or less) at the same locations as they were set, thus using the locations of set nets. Now the pressure occurs in about half of the study area, especially Southern Bight, English Channel, Skagerrak, Jutland Bank, Fisher Banks and, Northwest of the Shetlands (option 5III, Fig. 3 bottom row).

This shows that both the exposure indicators, total impact as well as its spatial distribution, differ markedly depending on the chosen option.

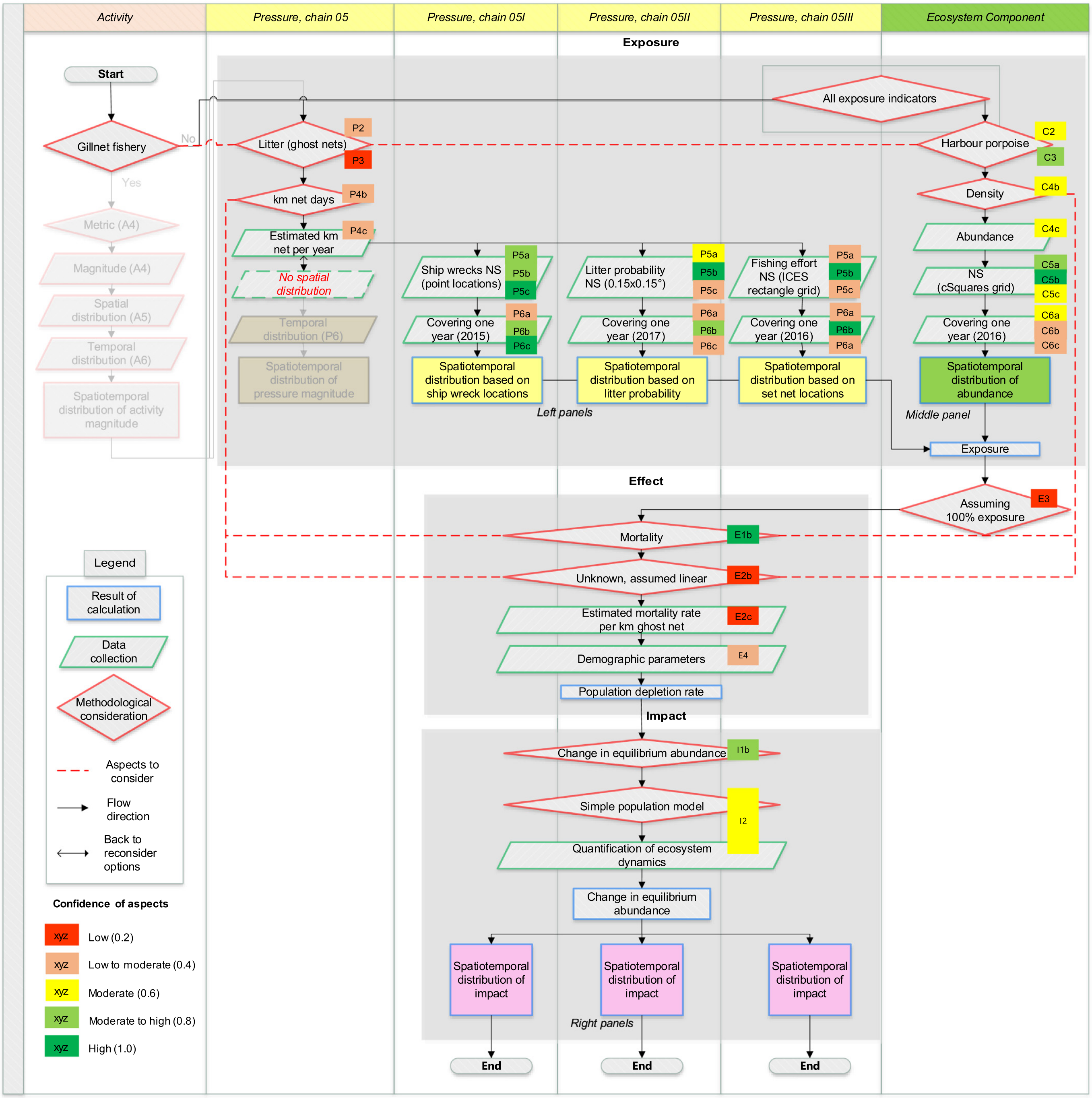


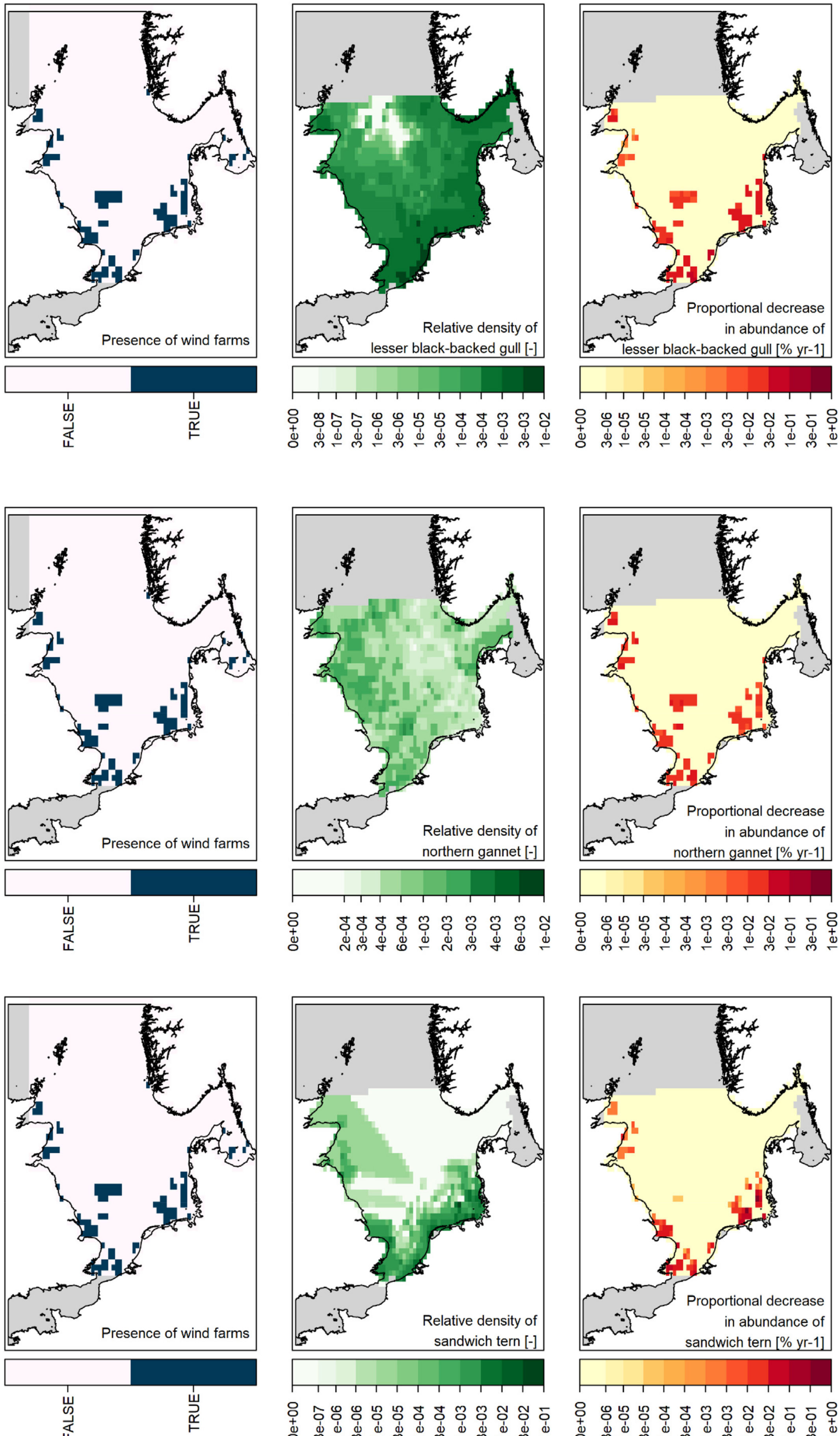
Fig. 4. Flow scheme of steps and decisions for the chain "Gillnet fishing - Litter (ghost nets) – mammals-Harbour porpoise" (see Fig. 1) with ID = 5I, 5II and 5III (see Table 5). Confidence of aspects are expressed in colors (for explanation see Table 4). The letter- and number codes refer to the aspects described in Fig. 2 and Tables 1–3.

There is a two-fold difference in overlap and even four-fold in total impact between options 5II and 5III caused only by underlying assumptions and hence the data used for the pressure distribution. The confidence varies slightly between options, with highest confidence for pressure distribution based on ship wreck locations (option 5I, Fig. 4). This confidence information could be used as a criterion for selecting an option. Note, however, that the spatiotemporal pressure distribution is not the weakest link (and thus with lowest confidence) but rather the activity-pressure relationship (P3), the pressure-effect relationship (E2) and the degree of exposure within the pressure-ecosystem component overlap (E3).

3.2. Impact of windfarm collisions on seabirds

This example illustrates the consequences of species selection to represent the ecosystem component (Figs. 5 and 6). Species selection

Fig. 5. Spatial distribution maps (grid size = 0.25 × 0.25 degrees) for the impact chains "Offshore windfarms - Collision – Birds" (see Fig. 1) distinguishing three methodological options based on respectively Lesser black-backed gull, Northern gannet and Sandwich tern (ID = 14 l, 15I, 16I from top to bottom) described in table 5. The panels show from left to right the pressure (presence wind farms, ecosystem component (species relative density (expressed as fraction with total sum = 1)) and impact (proportional decrease in abundance (% yr^{-1}) caused by the pressure).


Presence of wind farms
FALSE
TRUE
Relative density of lesser black-backed gull [-]
0e+00 3e-08 1e-07 3e-07 1e-06 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02
Proportional decrease in abundance of lesser black-backed gull [% yr-1]
0e+00 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02 1e-01 3e-01 1e+00
Presence of wind farms
FALSE
TRUE
Relative density of northern gannet [-]
0e+00 2e-04 3e-04 4e-04 6e-04 1e-03 2e-03 3e-03 4e-03 6e-03 1e-02
Proportional decrease in abundance of northern gannet [% yr-1]
0e+00 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02 1e-01 3e-01 1e+00
Presence of wind farms
FALSE
TRUE
Relative density of sandwich tern [-]
0e+00 3e-07 1e-06 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02 1e-01
Proportional decrease in abundance of sandwich tern [% yr-1]
0e+00 3e-06 1e-05 3e-05 1e-04 3e-04 1e-03 3e-03 1e-02 3e-02 1e-01 3e-01 1e+00

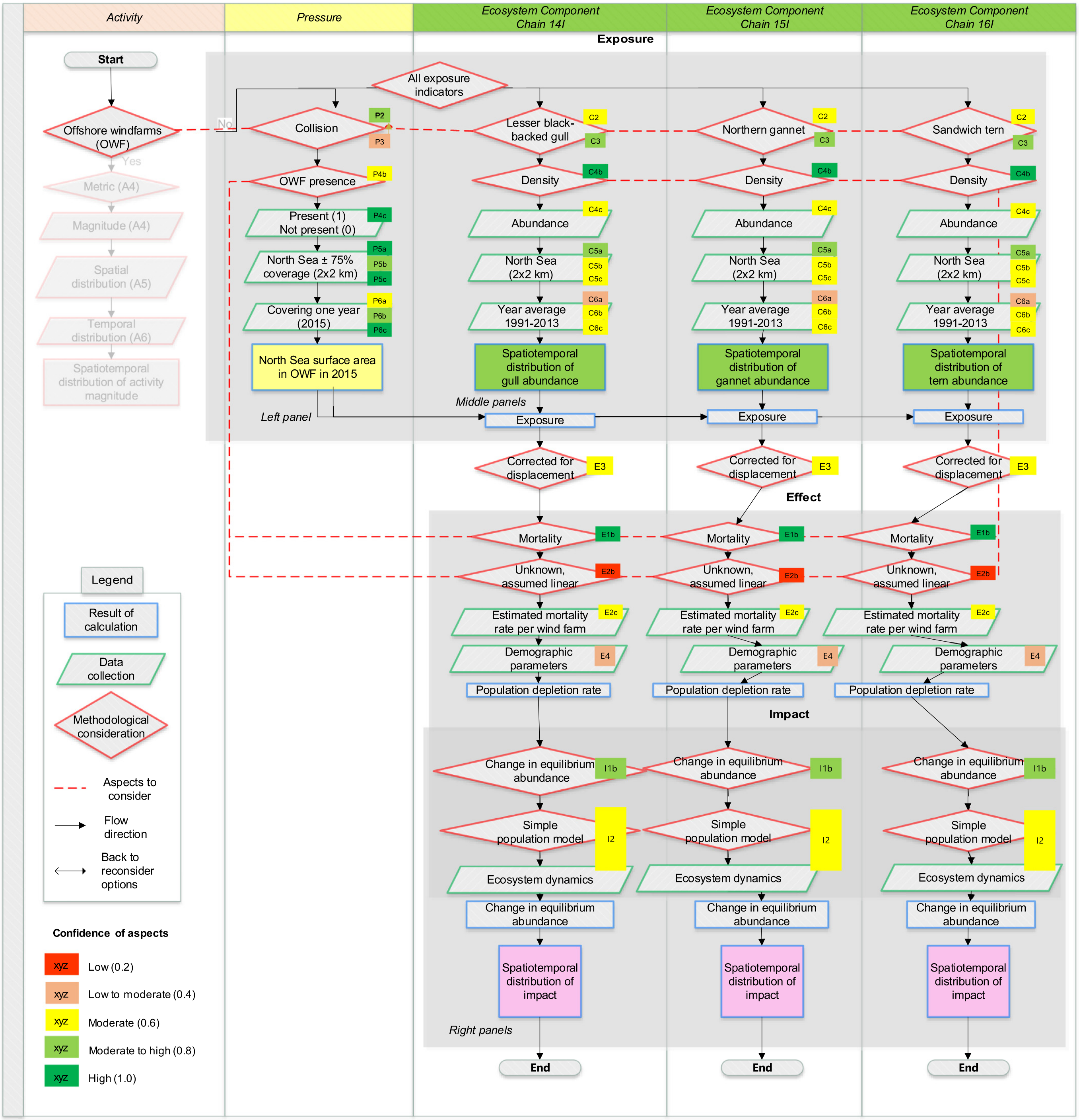


Fig. 6. Flow scheme of steps and decisions for three indicator species relating to the impact chain "Offshore windfarms - Collision – Birds" (ID = 14I, 15I, 16I, see Table 5). Confidence of aspects are expressed in colors (for explanation see Table 4). The letter- and number codes refer to the aspects described in Fig. 2 and Tables 1–3.

may be determined in the scoping phase where for example species with a specific conservation status may be selected but in practice it is often determined by the availability of data. If data availability allows the selection of several species to represent an ecosystem component, the outcome of the assessment can be assumed more representative for that ecosystem component. This, however, requires a consideration of the sensitivity of the assessment outcome to the selection of the species or the aggregation across species. For example, when there are two species, i.e. one moderately sensitive the other very sensitive, then aggregation may imply a choice between representing an average situation versus worst case. We illustrated this for the ecosystem component 'Birds' using two sensitive species (top and centre rows, Fig. 5) and one moderately sensitive species (bottom row, Fig. 5). Our CIA shows the most impacted species (Impact = 4.82%) is the Lesser black-backed gull (top row, Fig. 5). This could be expected as this species is considered sensitive for collision with offshore wind turbines and is an abundant species for offshore and coastal areas. However, the other sensitive species, Northern gannet, is least impacted (Impact = 2.34%) despite it being more sensitive to collision than the Sandwich tern which is relatively more impacted (Impact = 4.35%). The explanation lies in the higher exposure of the two species (best captured by the Likelihood and Severity indicators) that mostly occur close to land as do

Table 6
Spearman rank correlation results of the different endpoint CIA indicators shown in Table 5 (and SM Table 3.3.1). In the lower left the *p*-values are given, in the upper right the correlation coefficients. Significant correlations ($p \leq 0.05$) are in bold.

	Overlap	Likelihood	Magnitude	Severity	Impact
Overlap		**0.79**	0.20	**−0.58**	**−0.49**
Likelihood	**0.00**		−0.08	−0.35	−0.27
Magnitude	0.34	0.71		0.36	0.28
Severity	**0.00**	0.09	0.08		**0.74**
Impact	**0.01**	0.19	0.17	**0.00**	

the locations of the wind farms. Because of the coastal distribution of the Sandwich tern, it's moderate sensitivity eventually leads to higher impact than the more sensitive Northern gannet, which occurs further offshore. This illustrates that the species selection process is not straightforward as the selection of the most sensitive species does not always imply a worst case outcome. In this case the confidence scores also do not help as all species are considered equally suitable for representing the ecosystem component birds (C3) and data is taken from the same source so data processing (C2), metric, magnitude (C4) and spatiotemporal aspects (C5 and C6) are all identical (Fig. 6). The weakest link with lowest confidence of these chains is the pressure-effect relationship (E2).

3.3. Ranking of chains

As the purpose of a CIA is usually the prioritization of the chains in terms of their threat to ecosystem health we tested whether the choice of assessment metrics and/or methodological option explored in this study would have consequences on the outcome of the CIA (see Table 5) in terms of the ranked order of the chains (in Supplementary material SM Table 3.3.1). This shows that the choice of assessment indicator had a significant effect on the outcome of the assessment and, depending on the endpoint indicator chosen, the choice for a specific methodological option as well.

An analysis of the correlations between the different endpoint indicators to test if the indicator of choice would have affected the outcome of the analysis (Table 6) shows that our most elaborate endpoint indicator, Impact, is significantly correlated with Severity (correlation coefficient = 0.74), not significant with Magnitude but at least with a similar ranked order (correlation coefficient = 0.28) while an inverse rank order is observed for Likelihood (correlation coefficient = −0.27) and Overlap (correlation coefficient = −0.49), in case of the latter even significantly.

3.4. Confidence

The confidence assessment provides an overview of the quality and adequacy of the information that was available for the CIA (Fig. 7). It also shows which of the methodological options can be assumed to be the most adequate. After selection of the best methodological options this CIA would consist of these impact chains: 1, 2I, 3, 4, 5I, 6I, 7, 8, 9, 10, 11, 12, 13I, 14I, 15I, 16I.

4. Discussion

In this study we introduce a step-wise risk-based approach for CIA. The approach essentially disentangles "Risk" into its "Exposure" and "Effect" aspects that can be further improved (here fully quantified) through a step-wise process that assesses the quality of the available information and guides its application to improve the estimation of threat (or risk of impact). This now also provides a solid basis to estimate other known concepts often applied in risk-based approaches and their usefulness as proxies for what we consider the ultimate measure of threat, i.e. abundance of the main ecosystem components relative to an undisturbed situation. The approach requires that underlying assumptions and methodological considerations and choices are made explicit. This helps to identify the possible improvements of data and methodological choices and shows the resulting improvement in the quality (as reflected by the confidence score) and thus expected performance of the CIA. This approach can help address issues on the use of existing knowledge that are considered to hamper the reliability and hence applicability of CIA, e.g. use of dose-effect relationships (linear and non-linear), interaction mechanisms of multiple stressors (additivity, synergy, antagonism), refinement of space and time dimensions and explicit incorporation of uncertainty (Crain et al., 2008; Halpern and Fujita, 2013; Korpinen and Andersen, 2016; Piggott et al., 2015; Stelzenmüller et al., 2015; Stelzenmüller et al., 2018; Gissi et al., 2017).

We illustrate this step-wise approach for CIA and how it can use existing knowledge through a North Sea case study involving two sectors, i.e. fisheries and offshore wind farms, that differ considerably in how they impact the ecosystem, i.e. through pressures such as bycatch,

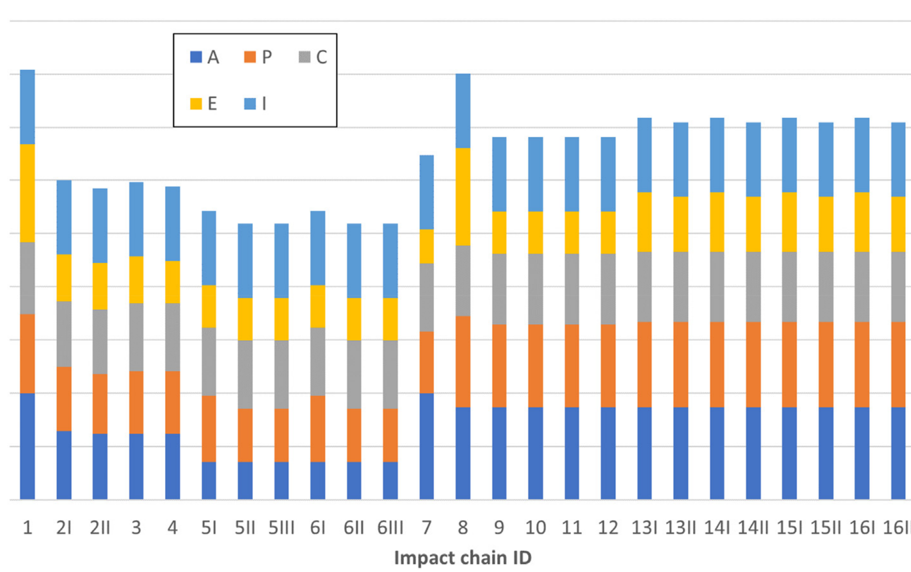


Fig. 7. Confidence assessment showing for each chain (and options) the aggregated scores distinguishing the main elements and steps (Activity (A), Pressure (P), Ecosystem Component (C), Effect (E) and Impact (I)). More detail is provided in the Supplemental material (SM Table 3.4.1). Numerical scales are omitted as relative differences are more important than exact values of the scores.

physical damage or litter (ghost nets), underwater noise, habitat loss and collision, and which includes different receptors, i.e. seabirds, benthic invertebrates and marine mammals. This as a "proof of concept" that, depending on the availability of information, it is possible to develop a fully quantitative CIA covering different sectoral human activities, pressures and ecosystem components and better understand its strengths and weaknesses.

4.1. Step-wise CIA method

The step-wise method is essentially based on a risk-based (exposure-effect) approach (e.g. Bax and Williams, 2001; Stobutzki et al., 2001; Knights et al., 2015) for which the existing qualitative approaches were found to be insufficiently satisfactory to guide ecosystem-based management (Piet et al., 2019) or Marine Spatial Planning (Lonsdale et al., 2020). This quantitative approach therefore fulfils the need for direct quantitative assessments of the effects of multiple stressors and the way in which they interact on ecological communities (Brown et al., 2014; Guarnieri et al., 2016; Lonsdale et al., 2020). In developing more quantitative risk-based approaches there are several studies that quantify the exposure aspect but only few attempt to do this for the Effect aspect, e.g. Stelzenmüller et al. (2018) and Fock (2011), only covering a single sector, i.e. fisheries, and hence not providing a generic approach that includes all aspects of risk. Several studies (Crain et al., 2008; Halpern and Fujita, 2013; Piggott et al., 2015; Korpinen and Andersen, 2016; Stelzenmüller et al., 2018) stressed the importance of explicitly including what we call here the pressure-effect relationships. Below we will discuss in more detail how this approach quantifies the exposure and effect aspects.

The chosen spatial resolution (0.25° longitude by 0.25° latitude) in this study was a compromise between conflicting needs and the actual availability of spatial information. Reviewing the presented results it can be seen that in none of the cases the sometimes coarser resolution of either the pressure or ecosystem component affected the outcomes of the CIA.

4.2. Exposure

There are several concepts used as part of risk-based approaches that essentially cover how we interpreted the exposure aspect, e.g. spatial overlap, likelihood of encounter but also severity as all these indicators reflect increasingly more elaborate degrees of co-occurrence calculated from the combination of spatially explicit information of the pressure and the ecosystem component which differ in that they do or do not consider information on respectively magnitude and/or density. As spatiotemporal distribution of pressure magnitude is not always available a common approach is to scale the pressure values linearly such that the highest value is equal to 1 or relative to a known theoretical maximum value (Korpinen and Andersen, 2016).

There is broad agreement that mapping co-occurrence of ecosystem components and pressures can lead to more quantitatively transparent and robust assessments of ecological risk (Martin et al., 2018) which provides information on a meaningful scale for decision-making (Helle et al., 2020). In environmental or ecological risk assessment, exposure refers to the concentration of substances in the environment (De Lange et al., 2010; EC, 2003; Stelzenmüller et al., 2018). Consequently, exposure can be expected to be better represented by indicators which include the magnitude of the pressure (i.e. Magnitude or Severity).

Using severity as an exposure indicator may appear contentious as several studies (Knights et al., 2015; Borgwardt et al., 2019) consider it as complementary to exposure because it is supposed to reflect the consequence of exposure. However, these studies then refer to the combination of exposure and severity as the risk of impact or impact potential which is similar to the definition of exposure as "the degree of change that it is projected to experience" (e.g. Cabral et al., 2015; Zolkos et al., 2015; Weißhuhn et al., 2018). Our use of severity as one of the potential endpoint indicators for exposure is thus in agreement with that definition.

Assuming Impact is the most accurate assessment endpoint, then comparison of the rankings of the other potential (and less information-heavy) endpoint indicators in this limited (in terms of impact chains included) proof of concept shows that severity is the best proxy followed by magnitude. Likelihood and overlap which do not consider the magnitude of the pressure appear very poor proxies of the potential threat of a pressure to an ecosystem component.

This nicely illustrates how the actual calculation using quantitative information helps to better define concepts that were previously used in mostly qualitative approaches but without a rigorous definition of what those concepts are meant to represent. As such this quantitative approach can guide qualitative risk-based approaches towards an improved application, definition and estimation of their concepts used to assess risk.

In addition to these more conceptual issues there are several methodological issues to consider when estimating exposure, e.g. spatiotemporal scale or the use of pressure or activity data. Our examples have illustrated how choices in the selection and application of data may influence the outcome of the assessment. In addition we want to discuss the choice of the spatial scale as this was found to differ considerably between existing CIA studies where spatial distributions of pressures were considered (Halpern et al., 2008; Stelzenmüller et al., 2015; Korpinen and Andersen, 2016; Stelzenmüller et al., 2018) but which is known to greatly determine the outcome of the assessment (Piet and Quirijns, 2009). The chosen spatial resolution is often dependent on the available sector- or pressure-specific data where for example fisheries data is available at higher resolutions than pollution (Halpern et al., 2012) and/or related to the scale of the assessment unit where regional sea assessments have a coarser resolution than e.g. national assessments (Korpinen et al., 2019).

4.3. Effect

Multiple choices can be considered for pressure metrics. Suitability and quality of the available information are both relevant for these choices. In our examples we choose to elaborate some options, for illustration purposes, but others may be equally or even more important. For the pressure metric of the chains representing the impact of gillnet fishing on marine mammals and seabirds we used fishing hours but tonnage of fish landings could also be used. The latter option is applied by Bisack (2003), Fock (2011) and Scheidat et al. (2018) and will result in comparable results concerning the number of bycatch victims for the North Sea when included in our CIA approach, but requires a different parameterization of the pressure-effect relationship.

The need to include pressure-effect relationships (otherwise referred to as driver-response, dose-effect, dose-response (Halpern and Fujita, 2013) or driver-response relationship (Hunsicker et al., 2016) is stressed by many authors but, for lack of an appropriate method, not included in any CIA - if accepting the assumption that CIA relates to 'all impacts of all activities' (Lonsdale et al., 2020) thus excluding single-sector studies. In our study, the default assumption was a linear relation as recommended by Halpern and Fujita (2013) and Judd et al. (2015) if the type of pressure-effect relationship was not specified. This was countered by Hunsicker et al. (2016) who conducted an extensive literature review and robust statistical analysis on single driver–response relationships in marine pelagic systems and found that non-linearities are the most common, comprising at least 52% of all driver–response relationships. They also found many non-linear relationships where the assumption of linearity could result in an underestimation of impact (Hunsicker et al., 2016).

4.4. Impact

According to this approach impact is the preferred indicator to assess the importance of anthropogenic threats on the ecosystem because it incorporates all relevant aspects, i.e. exposure of an ecosystem component to a pressure with specific magnitude, the effect it has on the ecosystem component and the sensitivity of that ecosystem component. Zacharias and Gregr (2005) define vulnerability as the probability that an ecosystem component will be exposed to a pressure to which it is sensitive. Thus impact as calculated in this study can also be considered as an appropriate indicator of vulnerability.

CEA and CIA are used interchangeably within the current literature (Korpinen et al., 2019; Lonsdale et al., 2020). Yet, the step-wise approach introduced here allowed us to make a distinction between the two as we differentiate between (cumulative) effects and impacts. The effect is the immediate consequence of the magnitude of the pressure on some aspect (e.g. mortality, reproduction) of an ecosystem component (e.g. at the level of an organism or population). In practice this is often determined by whatever pressure-effect relationship is known. In contrast (and ideally), the impact should be expressed as an assessment endpoint that is (1) identical for each impact chain allowing aggregation across impact chains into a cumulative impact assessment and (2) intuitive so that it can be easily communicated to the recipients of advice. As this should be the ultimate goal we propose to use CIA only if the assessment includes all the steps described here. This distinction between a CEA and a CIA resembles the difference between midpoint and endpoint level results within life cycle assessment (LCA), a framework to quantify the potential environmental impacts of a product over its full life cycle (ISO, 2006). For example, Middel and Verones (2017) integrated noise pollution impacts on marine ecosystems into the LCA framework. Their midpoint represents directly affected animals whereas the endpoint represents the impact expressed as the number of affected Harbour porpoises within a population (Middel and Verones, 2017). To calculate impact in terms of the assessment endpoint, i.e. equilibrium abundance relative to undisturbed, we assumed semi-chemostat dynamics to model the abundance of the ecosystem component. The main advantage over the logistic growth model applied by Pitcher et al. (2017) in a risk assessment approach for trawling impact, is that it avoids the assumption of a fully closed population, where there is no return from an extinct population, i.e. $N = 0$, and hence considered an unstable equilibrium. Using this equilibrium biomass as a concrete assessment endpoint has an advantage to the risk-based approaches commonly applied to assess cumulative effects (e.g. Knights et al., 2015) because it can be intuitively understood by stakeholders and translated more easily (albeit not necessarily one-to-one) into policy advice such as specific indicators for ecosystem status or the capacity to supply ecosystem services. While the applied population modelling approach might appear the best approximation of real-world populations, we acknowledge that specific circumstances may cause the outcome to be different from what is actually observed. There are many different forces that drive ecological dynamics (density dependence, environmental and demographic stochasticity, and climatic forcing—as well as their often complex interactions) adding complexity to assessments of population dynamics (Bjørnstad and Grenfell, 2001). Trying to incorporate realistic population dynamics into one broad generic approach covering multiple stressors and ecological components is a considerable challenge and may not always be feasible.

4.5. Validation

A first validation comparing some of our results with values found in literature shows good correspondence for the Lesser black-backed gull affected by collision from wind farms: 11% according to our estimate and 9.4% according to Brabant et al. (2015). However, for Harbour porpoise affected by impulsive noise from wind farms a considerably lower impact than our estimate of 1.1% of the porpoise population is found in literature. Middel and Verones (2017) estimated a potential disappeared fraction per year of the harbour porpoise population at 2.73×10^{-13} per kwh. This was based on a midpoint of 288.518 affected animals per year, which would be 0.11% of the North Sea population (Middel and Verones, 2017). Our impact estimate of physical damage from bottom trawling on benthos was based on the same approach and parameters based on a global analysis of observed depletion and recovery of seabed biota after bottom trawling disturbance by Hiddink et al. (2017) and is therefore expected to be representative for the overall benthic community. This, however, is likely to differ considerably between species with life-history traits that make them more or less vulnerable to bottom trawling (Hiddink et al., 2019).

4.6. Aggregation

Once the impact on the assessment endpoint (change in equilibrium abundance) is estimated for each impact chain some aggregation method is required across impact chains to complete the CEA. For a qualitative (or semi-quantitative) approach where the assessment endpoint was estimated from categorical scores, Piet et al. (2017) considered the issue of aggregation and concluded there is no one-size-fits-all aggregation method distinguishing between the guidance prior to management interventions, e.g. by Knights et al. (2015), using minimum, median and maximum aggregated risk scores, and evaluation of their performance afterwards, e.g. Piet et al. (2015); Piet et al. (2019), who chose summed risk. According to Judd et al. (2015) the CIA should start under the assumption that the different pressures and effects act in an additive fashion, but should include at least a qualitative appraisal of the likelihood and potential consequences of specific interactions (e.g. synergisms/antagonisms). If synergism is likely to be the dominant process than the calculated (contribution to) risk associated with those pressures can be considered a minimum and in case of antagonism a maximum.

This approach where each unique impact chain (in terms of the combination activity-pressure-ecosystem component) causes a specific change in equilibrium abundance appears ideally suited for addition across all unique impact chains as suggested to be the preferred aggregation method for CIA (Piet et al., 2019; Judd et al., 2015). Here we did not consider synergism or antagonism as quantitative information of sufficient quality -to distinguish between alternative mechanisms and additivity - does not exist for any combination of the impact chains considered in this study. As for some of the impact chains we had more than one species to represent the ecosystem component this provides the option to consider worst-case versus best-case assessment outcomes by selecting the species with respectively highest and lowest vulnerabilities. For the overall CIA the difference was approximately 4-fold for the marine mammals and even 10-fold for the seabirds which stresses the relevance of selecting a representative species or better several species.

4.7. Confidence

A simple addition was used for the aggregation of the confidence scores into an overall confidence assessment. Alternatives include multiplication of scores or taking the minimum score across all aspects per impact chain. Multiplication was rejected as it would require the confidence scores for each of the aspects per impact chain to be independent. Taking only the minimum assumes that (the confidence in) the impact chain is as strong as its weakest link but was rejected as it assumes this one aspects determines the overall confidence. Thus addition was considered the aggregation option with least assumptions.

The outcome of the confidence assessment was used to select the preferred methodological option for those impact chains for which several options were available. A risk-averse alternative that avoids the issues in estimating an overall confidence score would be to simply apply a precautionary approach and always use the impact chain with the highest impact independent of the outcome of the confidence assessment.

4.8. Conclusion and way forward

This approach and step-wise process applied to a limited but diverse selection of impact chains and applied to the North Sea has shown that it is possible to perform a fully quantitative CIA that can be further developed and expanded using available information and an explicit assessment of confidence to improve its accuracy and performance. Even though more information is available to allow further quantification of (parts of) several more impact chains than were presented here, we need to acknowledge that we are still far from having a fully quantified, comprehensive and operational CIA and therefore consider this as a first step of a gradual but systematic process that transforms existing comprehensive but qualitative CIAs (often based on expert judgement) into gradually more quantitative CIAs that apply the best available information and can prioritize the main threats as part of strategic advice to guide ecosystem-based management.

CRediT authorship contribution statement

Gerjan Piet: Conceptualization, Methodology, Writing – original draft, Writing – review & editing, Supervision, Funding acquisition. **Jacqueline E. Tamis:** Conceptualization, Methodology, Data curation, Formal analysis, Visualization. **Joey Volwater:** Software, Formal analysis, Visualization. **Pepijn de Vries:** Methodology, Software, Formal analysis, Visualization. **Jan Tjalling van der Wal:** Formal analysis. **Ruud H. Jongbloed:** Conceptualization, Methodology, Validation, Investigation, Data curation, Writing – original draft, Writing – review & editing, Visualization.

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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Appendix A. Supplementary material

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