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3 **Operationalising risk-based cumulative effect assessments in the marine**
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37

38 **Abstract**

39 Ecosystem-based management requires cumulative effect assessments. However, how those
40 assessments should be operationalised and integrated into decision-making processes still lack a
41 comprehensive and transparent framework. A risk-based cumulative effect assessment (CEA) framework
42 (dividing the process into risk identification, risk analysis and risk evaluation) could structure complex
43 analyses of XXX and facilitate the establishments of direct science-policy links. Here, we shed light on
44 the current operationalisation of such a risk-based CEA framework at different spatial scales and in
45 diverse settings. Using 11 case studies in Europe, French Polynesia, and Canada at local, sub-regional,
46 and regional management scales, we show that a single recipe on how to conduct a CEA does not exist,
47 but the application of a standardised framework facilitates a consistent, coherent, and transparent
48 comparison of the key issues to operationalise such complex assessments. We illustrate the large
49 variation in CEA drivers, objectives, and assessment endpoints. We identify four key recommendations
50 to better strengthen the implementation of CEA into management: 1) Framing the context and setting
51 risk criteria; 2) Defining the roles; 3) Reducing and structuring complexity; 4) Communicating
52 assumptions and uncertainty. We reveal the need for more research on the effectiveness of existing
53 management measures to improve these measures or the development of new ones to reduce the
54 cumulative effects through the reduction of pressures generated by individual activities. Key to the
55 whole process is to consider that the risk evaluation comprises a trade-off analysis of the cost and
56 potential benefits of alternative or additional management measures, which should be clearly separated
57 from the provision of technical advice by scientists. Our study makes a strong case that CEA should be
58 well framed and recognised as a cross-cutting tool that could bridge different management objectives
59 and could be a strategic approach to integrate ecosystem management considerations across multiple
60 sectorial policies.

61

62 **Highlights**

- 63 • How to bridge the gap between theory and practice in ecosystem-based management
- 64 • Well-structured CEAs is a solution for effective EBM
- 65 • 11 case studies demonstrate a large variation in CEA drivers, objectives, and assessment
66 endpoints
- 67 • Framing the CEAs context and risk criteria, and defining clear roles are recommended for larger
68 impact
- 69 • Reducing/structuring complexity and communicating uncertainty are essential for CEAs

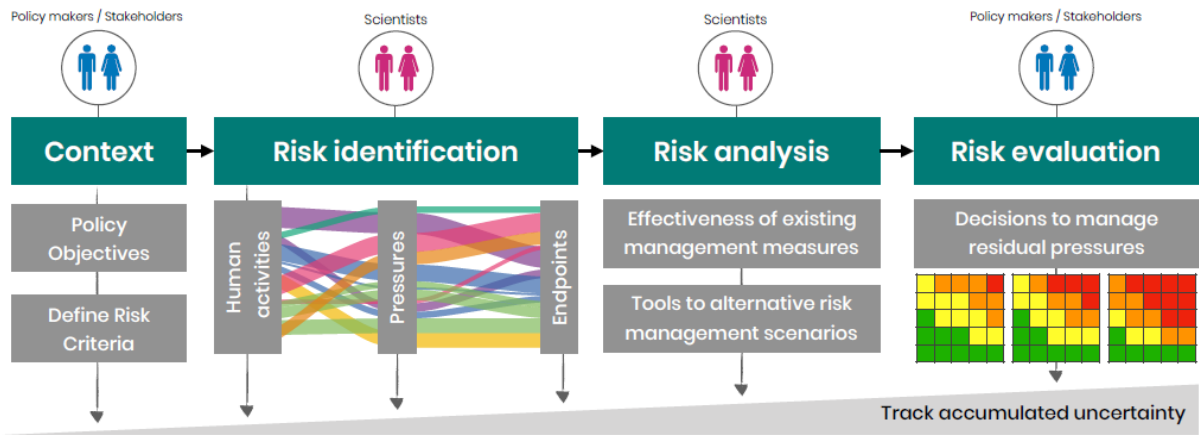
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73 **Graphical abstract**

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75

76 1. Introduction

77 Over the last decade scientific effort on the categorization and description of human pressures on
78 marine ecosystems has increased (Knights et al. 2015, Borgwardt et al. 2019) leading to a better
79 understanding of the globally increasing footprint and intensity of human activities at sea (Halpern et al.
80 2015). However, in front of increasingly rapid changes in direct and indirect pressures on biodiversity
81 and ecosystem services, pathways towards a sustainable future still remain uncertain (Lindegren et al.
82 2018) {O'Neill, 2017 #370; Harrison, 2019 #371}. In particular, better detecting the risk of current and
83 future changes of social-ecological systems is key to prevent the coupled human-nature systems to shift
84 into undesirable states (Bates et al. 2018, Rilov et al. 2019) (Hodgson and Halpern 2019).

85 In marine social-ecological systems, management frameworks exist that aim to explicitly avoid
86 undesired changes in the coupled systems. Marine ecosystem-based management (EBM) (Katsanevakis
87 et al. 2011) and integrated marine management (IMM) (Stephenson et al. 2019) can effectively inform
88 policies to meet sustainable development goals. A sound understanding of cause-effect pathways
89 describing the link from human pressures causing potential state changes of ecosystem components,
90 processes or functions should form the backbone for decision making in EBM or IMM (Stelzenmüller et
91 al. 2018) and help to implement appropriate programs, measures, procedures, and control actions
92 (Cormier et al. 2017, Stephenson et al. 2019).

93 Cumulative effects assessments (CEAs) aim to explore these causal pathways and should deliver advice
94 for the implementation of management measures for human uses to maintain or restore ecosystem
95 states while balancing conservation and restoration with social and economic objectives {Cormier, 2019
96 #339;Cormier, 2019 #342}. CEAs are defined as holistic evaluations of the combined effects of human
97 activities and natural processes on the environment and constitute a specific form of environmental
98 impact assessments (EIAs) (Jones 2016). As a consequence, CEA results should therefore directly inform
99 regulatory processes (Willstead et al. 2017), marine spatial planning (MSP) (Stelzenmüller et al. 2010,
100 Menegon et al. 2018, Liversage et al. 2019) or the implementation of environmental policies such as the
101 European Union Marine Strategy Framework Directive (MSFD 2008/56/EC). Although the numbers of
102 CEA case studies are increasing in the marine realm (Murray et al. 2015, Korpinen and Andersen 2016,
103 Menegon et al. 2018), the formal uptake of CEA results in management processes is yet to be evidenced
104 (Willstead et al. 2018).

105 Therefore, the operationalisation of CEA should be facilitated. Stelzenmüller et al. (2018) suggested a
106 risk-based CEA framework (dividing the process into risk identification, risk analysis and risk evaluation),

107 which structures complex analyses and facilitates the establishments of direct science-policy links,
108 highlighting the fact that CEAs should not only be scientifically driven {see also Cormier, 2018 #327}. By
109 applying standardized risk analysis along with a unified glossary and terminology, the framework
110 outcomes should allow, independent of the context, addressing the likelihood of exceeding accepted
111 ecological threshold and the risk of ecosystem state changes together with the potential effectiveness of
112 new management measures. Thus, this risk-based CEA framework can support the operationalization of
113 CEA as a strategic tool in EBM, being an integral part of the management process, where the roles of
114 scientists and decision-makers are clearly defined.

115 Here, we shed light on the challenges and opportunities of the operationalisation of such a risk-based
116 CEA at different spatial scales and in diverse settings. We identified eleven case studies in Europe,
117 French Polynesia, and Canada at local, sub-regional, and regional management scales. In each case
118 study, we applied the framework described by Stelzenmüller et al. (2018) to identify the main outcomes
119 and challenges for a better uptake of CEA into management and decision-making. Furthermore, we
120 provide some tailored tools that allow for the evaluation of the uncertainty around CEA assessment
121 results and facilitate the use of CEA outcomes for management advice. Based on the here compiled
122 knowledge state, we derived some key recommendations on how to overcome the main challenges for
123 the operationalization of the risk-based CEA framework. Ultimately, those should help scientists and
124 managers alike to foster the dialog between key players at the science-policy interface.

125

126 2. Comparative analysis of CEA case studies

127 We conducted a qualitative comparison among eleven CEA case studies (Figure 1, Appendix 1), which
128 either used the risk-based CEA framework to structure an assessment or used it as a lens for evaluating
129 existing CEA. We therefore designed a standardised questionnaire containing thirteen open questions
130 (see Appendix 2), which were answered for each of the eleven case studies. In the following sections we
131 present a synthesis of observed key outputs in relation to the context, knowledge and data, approaches,
132 and outcomes of the case studies and provide corresponding recommendations and solutions to
133 advance the operationalisation of CEA.

134

135 2.1. CEA drivers and objectives

136 With one exception, all case studies reported that CEAs were not commissioned by management bodies,
137 but had rather been initiated by scientists with the aim of producing meaningful results to inform a

138 respective management context. Only the Canadian case was initiated by a management body in the
139 course of the implementation of an integrated management plan. Across case studies, the targeted
140 management context spanned from implementation of regional policies such as the European MSFD,
141 Marine Spatial Planning (MSP) to sectoral regulatory processes and regional spatial management plans
142 (Figure 2). The selection of the general management goals or policy objectives determined the
143 ecosystem components, functions and processes for which the assessment of cumulative effects was
144 conducted. Based on the case studies, we also observed that targeted assessment endpoints were
145 broad, and comprised biological entities such as species (e.g. sea turtles, dolphins), ecosystem types
146 (e.g. coral reefs), ecosystems state (e.g. ecosystem health), and ecosystem services (e.g. sustainable
147 resource use) (Figure 2).

148 This diversity in assessment targets and assessment scales demonstrated clearly that the proposed risk-
149 based CEA framework is flexible and can be applied in different contexts. Further, the studied cases
150 exemplified the breadth of both assessment objectives and their strategic setting within specific
151 management processes, thus underlying the integrative setting of the framework between policy and
152 science.

153 As opposed to biological components, case studies reported also assessment targets in relation to the
154 effectiveness of conservation and management measures such as marine protected areas. The capacity,
155 functioning and the achievement of a Good Environmental Status (GES) of marine waters, as requested
156 by the MSFD, was targeted in two additional case studies (e.g. Aegean, Med LME). These are CEA
157 examples where an ecological assessment was integrated with the evaluation of cost-effective
158 management processes. Choosing GES as the endpoint of an assessment requires the consideration of
159 policy context, thresholds, and ecological state assessment. Two case studies targeted the broader
160 effects of sectoral plans of the energy sector (e.g. Belgian North Sea). Another example showing that
161 sector management needs to bring together not only the biological components and the relevant
162 human pressures with their management measures, but also has to take into account the inherent
163 complexity of the responsible authorities and sector policies (e.g. Adriatic Sea, see {Gissi, 2017
164 #304}{Cormier et al. 2019}).

165

166 2.2. Establishing cause-effect pathways

167 Applying the risk-based CEA framework entails the establishment of the linkages between human
168 activities, the corresponding pressure categories, and the effects on the respective ecosystem
169 components, processes and functions (Stelzenmüller et al. 2018) (Figure 3). Hence, well-established

170 cause-effect pathways play a central role enabling the identification of key activities that need to be
171 regulated to prevent an increased impact on the system. It is important to understand that a pressure
172 such as abrasion or siltation can be caused by different activities (Menegon et al. 2018) {Knights, 2015
173 #322} and that the pressure describes the actual mechanism of change or alteration to the ecosystem
174 component (Elliott et al. 2017). Thus, the general linkages between human activities and respective
175 pressure characteristic is an established concept (Borgwardt et al. 2019) when aiming to assess
176 environment effects. Measurable cumulative effects are caused by the amount of pressures (referred to
177 as residual pressures, see Figure 3), which still exists despite the implemented management measures
178 or restrictions (Cormier et al. 2018). This implies that, depending on the human activities and type of
179 measures, employed measures could be technically not capable and/or not effective enough at reducing
180 the pressure loads to levels, which are not deemed to cause adverse effects on ecosystem components.

181 We observed that all case studies identified general cause-effect pathways between an ecosystem
182 component at risk and the respective pressures generated by human activities or at least identified the
183 link to relevant human activities (see Figure 2). The total pressure load to which an ecosystem
184 component is exposed to will contribute to its overall vulnerability. Therefore, vulnerability estimates
185 are a result of the exposure of the ecosystem component to a given pressure and the sensitivity of the
186 ecosystem component to that specific pressure (Piet et al. 2015, Stelzenmüller et al. 2015). We noted
187 that this general concept of vulnerability has been embraced by most case studies. In other cases,
188 environmental vulnerability profiles were e.g. calculated as an aggregated product of the distribution of
189 essential nature values (habitat-forming benthic macroalgal and invertebrate species, benthic species
190 richness, birds and seals as top marine predators) and their sensitivity to disturbances (Aps et al. 2018).

191 One of the key issues encountered by case studies in relation to the establishment of the cause-effect
192 pathways were limitations with regard to both the quality of the human pressure data and the
193 confidence in the assumed causality. Reported data gaps related to ecosystem components and
194 functions (e.g. species richness, pelagic compartment, benthic habitats, non-commercial species,
195 movement patterns), data to develop modelling tools (e.g. water circulation, high resolution habitats,
196 artisanal and recreational fisheries), data on emergent activities (e.g. aquaculture and energy extraction
197 plans) and pressures (e.g., plastic pollution, noise, climate change) or the representability of data (e.g.
198 available information not capturing well inter-annual variation or different spatial scales). In fact,
199 reconciling data of different geographic scales (local to regional), seasonal dimensions (spawning,
200 secondary production) and temporal resolutions (past and current dynamics) seemed to be the main

201 challenge for most case studies. Further, several cases have been very explicit with regard to data needs
202 and identified knowledge gaps that should be addressed by future monitoring schemes, research
203 programs and initiatives aiming to provide standardised and accessible data systems.

204 In addition, the consideration of connectivity among the terrestrial, freshwater and marine realms, and
205 cross-realm pressures to ecosystems was also highlighted as a challenge due to data requirements from
206 different sectors (e.g. agricultural use of pesticides and fertilisers). This underlines the recently
207 described gap on knowledge and research on connectivity cross-realms {Giakoumi, 2019 #372}. Further,
208 not including climate change was also mentioned frequently as an important limitation of CEA case
209 studies contributing to the increase of uncertainties in the results. These observed challenges
210 correspond well with recent work that highlighted that uncertainty in the data resolution on human-
211 induced pressures can have significant effects on the interpretation of cause-effect pathways and
212 respective vulnerability assessments (Amoroso et al. 2018, Stock et al. 2018).

213 Working from the basis that uncertainty is part of any decision making process, and that it is key to deal
214 with uncertainty in a transparent and explicit manner in terms of the knowledge and data available in
215 risk identification, we developed a confidence matrix which facilitates a general communication of
216 uncertainty at this step (Figure 4).

217 Science advice of a CEA that underpins a regulatory process requires the highest confidence, as opposed
218 to scientific advice of CEA that supports policy processes (Figure 4). Hence, there is a greater need for
219 confidence in the established causal relationships between activity, pressure and effect at a regulatory
220 process since this entails technical advice on how to regulate human activities or requirements such as
221 environmental quality standards. Less confidence may be sufficient in a marine spatial planning context
222 when developing planning objectives for multiple activities. Here the confidence in estimated ecosystem
223 vulnerabilities forms the basis to identify the cumulative pressures that need to be addressed through
224 for instance a regional planning process. The matrix adopts the above rationale and recognises the
225 importance of human pressures data. It indicates that when data on pressures are of poor quality in
226 terms of e.g. a mismatch between spatiotemporal resolutions of pressure and ecosystem components
227 data, CEA outcomes should rather underpin strategic processes e.g. such as the development of policy
228 objectives. The eleven cases applied the confidence matrix and the aggregated results are shown in
229 Figure 4. Interestingly, none of the cases except for the Mediterranean LME case study reported a poor
230 quality of human pressure data, indicating that most of the case studies should be geared to advice e.g.

231 MSP processes. Hence, communicating the aggregated uncertainty should help building trust and
232 allowing to move towards a more transparent and informed decision-making process.

233

234 2.3. Understanding the need of risk criteria

235 Most case studies were confined to the risk identification stage, where the human activities, their
236 pressures and the respective vulnerable ecosystem components are being described. This is also in
237 accordance with the majority of the CEA published over the last years (Murray et al. 2015, Korpinen and
238 Andersen 2016, Menegon et al. 2018), which mainly identified priority areas to be of concern for
239 management processes. However, from the risk-based CEA framework perspective, all these scientific
240 initiatives were missing the essential ingredients that allow moving from risk identification to the
241 analysis of effectiveness of management measures to feed this advice in risk evaluation and risk
242 treatment processes. Hence, they were missing the identification of risk that would be tolerated.

243 To maximize the advice a CEA can deliver in the management process, the assessment should be
244 founded on established risk criteria reflecting the selected policy objectives. This is key to transparently
245 set the scope and context for the assessment, while delineating the level and resolution of information
246 needed to define cause-effect relationships. In alignment to the procedures of classical risk assessments,
247 risk criteria should be developed prior to initiating the CEA within the context and the scope of the
248 policies involved and in consultation with stakeholders (Rozmus et al. 2014). Thus, risk criteria should be
249 used to express different levels of state change and the overall risk of not achieving policy objectives.
250 Both need to be specified in relation to the identified management goals or policy objectives addressing
251 ecosystem components, functions and processes. We found that, in general, case studies responded to
252 national policies, informed MSP processes and have been well framed in the context of regional policies
253 (Figure 2). Some cases referred to the European Blue Growth policy {EU, 2017 #374} and designed the
254 CEA to assist the allocation of new uses while managing conflicts between them, and between uses and
255 the environment, according to the MSP Directive {EU, 2014/89/EU #136}. However, most of them did
256 not mention specific risk criteria in terms of definitions of effect sizes (e.g. defining the degree of change
257 of an ecosystem state due to a certain amount of exposure) or thresholds in relation to acceptable levels
258 of pressures remaining within management boundaries after considering existing management
259 measures (see also Figure 3). Examples of defined risk criteria can be found in Gimpel et al. (2013) or
260 Cormier and Lonsdale (2019). The former is an example for a semi-quantitative case where exact
261 thresholds were not known but the acceptable risk for an ecosystem component had been predefined

262 along clear criteria. A prerequisite for doing this is to well define the CEA context and the policy
263 objectives that are being addressed. Without risk criteria individual personal objectives and values
264 become the basis of debate of what is risky given the different perceptions of the level of risk and
265 individual tolerances to risk when making a decision (Cormier and Lonsdale (2019) (IEC/DIS, 2017)).

266

267 2.4. Accounting for the effectiveness of management measures and trade offs

268 Risk analysis means determining the actual consequences of cumulative effects; thus the consequences
269 that will occur when a state change of an ecosystem component, function or process has occurred. This
270 entails an analysis of the effectiveness of management measures that exist to regulate the pressures
271 (Figure 3). The case studies showed that with only a few exceptions, the existence and effectiveness of
272 management measures have not been considered as part of the CEA (Appendix 2). Further, we observed
273 quite some confusion across case studies on how to assess the effectiveness of management measures
274 and how to incorporate it within the respective studies. One exception is the Western Mediterranean
275 case, which considered the effectiveness of management measures. In this case, simulations of an
276 ecosystem model were used to vary the levels of protection of different marine protected areas located
277 in the study area. The case study used a marine protected area classification introduced by Horta e
278 Costa et al. (2016), which classifies marine protected areas according to the impacts of species and
279 habitats the uses allowed within their boundaries can have. Out of the five classes, only the two more
280 strictly protected prove to be effective (Zupan et al. 2018). A few case studies also mentioned the
281 importance to acknowledge the complex social-ecological dimensions in a CEA, hence pointing to the
282 fact that conflicts and trade-offs between human activities need to be analysed in relation to the risk of
283 cumulative effects. Hence, trade-off analysis might need to consider both positive and negative effects
284 of (cumulative) pressures since some human activities may counter-balance the effects of pressures,
285 while others may amplify them. Mechanistic models can be used to quantitatively identify such trade-
286 offs (Christensen and Walters 2004, Coll et al. 2008).

287 When cumulative effects are occurring in a given area, there are several factors at play that cannot be
288 managed by measures taken locally. Cumulative effects can also be related to natural variabilities, the
289 effect of climate change or pressures that are generated from outside the planning or management
290 area. In the latter case, regulatory options that can address these external factors often require cross-
291 jurisdictional or cross-boundary coordination in the implementation of management measures that can
292 reduce the pressures in each jurisdiction equivalently. This makes a strong case for the recognition of

293 e.g. climate change induced effects and contributions to cumulative effects in regulatory frameworks for
294 human activities and their pressures (e.g. limits to spatial use of an activity such as spatial allocation for
295 specific types of fishing gear or where dredging spoils can be disposed). This also requires the
296 consideration of such external effects in marine spatial planning processes, therefore complementing
297 conservation and restoration efforts. The effectiveness of the implemented measures is also influenced
298 by the level of conformity to the implementation specification of the measure, the compliance of those
299 that have to implement the measures and the reliability of the measures to perform adequately over
300 time (Cormier et al. 2018; Cormier et al. 2019). Due to the lack of studies and research designed to
301 determine by how much does a given measure contribute to the reduction of a specific pressure the
302 quantification of effectiveness remains challenging. This is why the residual pressures remaining in the
303 system after the implementation of a measure, or a suite of measures to manage specific human
304 activities, can be considered as a metric of measures' effectiveness (REF). An analysis of the
305 effectiveness of management measures must be able to characterize the amount of residual pressure as
306 an undesirable outcome of a measure or measures (e.g. level of contaminant reduction in an effluent,
307 the reduction of the spatial extent or frequency of sedimentation, etc.). From a methodological
308 perspective, for instance, modelling tools can be used to simulate different levels of effectiveness of an
309 action linked to different pressure levels and compare prediction with observational data (Coll et al.
310 2008) {Piroddi, 2015 #373}. Further, Cormier et al. (2018) present a modelling framework which allows
311 to quantify the residual pressure and how it contributes to the management effectiveness. A sound
312 understanding of the pressure-state relationship should then help assessing the contribution of the
313 reduction of the pressure to achieve the desired environmental state.

314

315 2.5. Providing scientific evidence for risk evaluation

316 The risk-based CEA framework considers risk evaluation as a process where management and
317 stakeholders evaluate what could be done to reduce the detected risks of cumulative effects. Risk
318 evaluation is where the decision is taken to maintain or improve existing measures or implement
319 additional ones. In other words, risk evaluation is where the results of the risk analysis are brought into
320 the policy realm of decision-making, which is actually the interface between the science and the policy
321 (Cormier et al. 2018). Up to this point, risk identification and risk analysis is primarily a scientific and
322 technical role in the provision of independent scientific advice without any value judgement such as
323 "serious", "harmful", "impacting", "severe", etc. (Figure 5; left). Thus only the levels of the likelihood of

324 the effect occurring and the severity of consequences are discussed in relation to the source of the risk,
325 as outlined by the cause-effect pathways. In risk evaluation, the scientific advice is provided to the
326 managers and stakeholders to underpin their decisions as to what to do in terms of management
327 measures required to reduce the risk considering the severity of those risks. Given that visualization is a
328 key communication tool to non-technical managers and stakeholders (Stelzenmüller et al., 2018), risk
329 matrices are typically used in risk evaluation as a graphical representation of the likelihood and
330 consequence combinations that are less to more tolerable given the policy context (Cormier and
331 Lonsdale, 2019). In risk evaluation tolerable refers to the likelihood or risk of not achieving stated
332 management objectives. Hence, it is important to note here that the use of such matrices goes beyond
333 the simple identification of the severity of the risk as commonly presented in ecological risk assessments
334 (Astles and Cormier, 2018). As shown in Figure 5, in a CEA context risk matrices show the combinations
335 of the levels of the likelihood of the effect of cumulative residual pressures occurring and the severity of
336 consequences are mapped to three different risk tolerance levels (high, moderate, low). In the examples
337 of Figure 5, the likelihood of the consequence of the existing management measures (EM) is compared
338 to the likelihood of the consequence for the proposed improvements to existing measures (PM). Thus,
339 improvements and additional measures should reduce the likelihood of the cumulative residual
340 pressures and/or the severity to a level that is tolerable in terms of reaching stated objectives given the
341 scientific, management and operational uncertainties. The different colour scheme of the three matrices
342 simply reflects different levels of risk tolerance. For instance, there are more red boxes for cases of low
343 tolerance to risk compared to matrices reflecting higher tolerances. This requires prior definitions; red
344 could mean that the likely consequences are not tolerable because the management measures are not
345 effective enough to reach defined objectives, while orange or yellow could mean that there are
346 uncertainties as to management measures will lead to the achievement of targets, which would imply
347 for instance extensive monitoring and review. Finally, green would imply that the management
348 measures are considered effective in the sense that policy objectives would be reached. Matrices should
349 avoid using qualifiers such as high, medium or low or 1, 2, 3 because they do not explicitly convey the
350 severity of the risks to managers and stakeholders (Baybutt 2018). If the risk of cumulative residual
351 pressures is assessed for many ecosystem components (e.g. species, functions, processes), each causal
352 concern should also have its own matrix because decisions regarding such risks would weight different
353 combinations of likelihood and consequences.

354 Given the iterative aspects of CEA, managers and stakeholders could submit new management options
355 to risk analysis that would then be analysed by scientific and technical experts. As explained above,

356 technically, scientists should not be part of the risk evaluation, but in practice they are often consulted
357 when it comes to actual decision making. Across our case studies, we identified examples where there
358 have been processes to clarify the roles of science and management (e.g. Canadian CS) up to cases
359 where roles have been mixed. Therefore, scientists should be prepared to develop and deploy tools in
360 risk identification and risk analysis to determine the effectiveness of various management options. In
361 risk evaluation, scientists can provide insight into uncertainties and assumptions involved in determining
362 the likely consequences of various management scenarios leaving the decision about the tolerability of
363 the risks not to meet the objectives to the managers and the stakeholders during the risk evaluation
364 phase. Some ready to use tools to provide informative results to managers and to help them find
365 alternatives or information about risk already exist (Stelzenmüller et al. 2018). This is linked to the
366 “being proactive” in the assessment of alternative scenarios of management advice and be ready to
367 present results in a science-policy context.

368

369 3. Unfolding uncertainty in CEA

370 The spatial and temporal distribution of ecological components, the causalities of multiple pressures
371 operating at various scales, their potential effects upon assessment endpoints, and the effects of
372 proposed management actions are fundamental pieces of information for a CEA. Throughout any
373 environmental impact assessment, including CEA, many assumptions and predictions often take place,
374 thus making it difficult to estimate the overall uncertainty of the analysis {Tenney, 2006 #305}. Thus
375 considering and treating the uncertainty that is inherent to the various steps of an environmental
376 impact assessment is critical for conveying a comprehensive understanding of the limitations and
377 accuracy of the generated outputs. Towards this direction, previous studies {Gissi, 2017
378 #304;Stelzenmüller, 2015 #50;Stock, 2016 #49} offered insights on the potential sources of uncertainty
379 linked with causality and data and proposed technical solutions on how to deal with it.

380 The risk-based CEA framework structures the evaluation of cumulative effects on ecosystem
381 components, considering the effectiveness of existing or proposed management measures, in relation to
382 levels accepted to reach policy objectives (Stelzenmüller et al. 2018). In this context, apart from dealing
383 with uncertainty in risk identification and risk analysis, it is also critical to disclose the degree and
384 sources of uncertainty associated with risk evaluation process where proposed management actions are
385 contrasted to the likelihood of achieving policy objectives. Hence, introducing an additional layer of

386 uncertainty which could influence the transparency throughout the decision-making process and
387 therefore affect capitalization of the outputs {Leung, 2015 #306; Tenney, 2006 #305}.

388 From our case studies, we observed that many cases acknowledged uncertainty, but often in an
389 unstructured fashion. Only for the Adriatic case a considerable effort was undertaken to assess the
390 sources of uncertainty in a transparent manner {Gissi, 2017 #360}. To unfold the dimensions of
391 uncertainty associated to the risk-based CEA framework and to offer a systematic guidance for
392 improving the treatment of uncertainty, we followed the approach presented in {Stelzenmüller, 2015
393 #52; Gissi, 2017 #360} and elaborated a Walker-type matrix {Walker, 2003 #193}. Building on {Walker,
394 2003 #193} we defined a total of eight uncertainty descriptors: context uncertainty, cause-effect model
395 uncertainty, data input uncertainty, statistical uncertainty, scenario uncertainty, recognized ignorance,
396 epistemic or knowledge-related uncertainty and inherent variability (Table 1 and Annex 3). These
397 descriptors represent uncertainty in a structured way, synthesise sources, causes and needs across the
398 three dimensions: location, level and nature.

399 We applied this approach to explore uncertainty in the eleven CEA cases (Annex 3) and found a great
400 variation of sources of uncertainty. The efficiency and adequacy of policies and measures identified at
401 local (e.g. through the managing authority responsible for the protection of sea turtle nesting habitats),
402 national (e.g. Belgium national policies on blue growth and offshore renewable sources) or international
403 scales (e.g. the CFP and the MSP in the case of fisheries in the western Mediterranean or as the
404 potential determinants for protecting habitats and key ecosystems in the Adriatic-Ionian sea) often
405 defined both the assessment endpoints and case study context uncertainty. Thus, results revealed that
406 policies and measures identified in the context setting are often not translated to clear operational
407 objectives with explicit criteria and targets.

408 Factors contributing to the uncertainty of the cause-effect modelling processes include the limited
409 knowledge of the spatiotemporal dynamics of ecological processes, the lack of a precise understanding
410 of the mechanism of cumulative effects (e.g. additive, synergistic or antagonistic), and the gaps and
411 incomplete information on the distribution, dynamics and magnitudes of pressures. To overcome these
412 gaps, inputs have been generated from models, expert assessments, and extrapolations from patchy
413 datasets. In some cases, efforts have been made to statistically quantify uncertainty by using sensitivity
414 analyses or by incorporating variability to ensure that model structure is adequate, such as in the
415 Western Mediterranean case. Still, in most of the case studies a further exploration and description of
416 sources of uncertainty through e.g. statistical tools was missing. In relation to management scenarios

417 that have been considered, sources of uncertainty comprised mainly the management measures tested
418 and the assessed or described magnitude of their future effectiveness and reinforcement. Following the
419 complexity of ecological, environmental and social-political dimensions involved in the CEA process and
420 the focus and spatial scale of the cases, contributors identified uncertainty, which they did not further
421 address. Examples of such recognized ignorance included the spatio-temporal variability of different
422 pressures (e.g. noise pollution, fishing pressure, tourism activities) but also the cumulative effect of
423 invasive species and climate change. Thus, environmental variability, the multi-dimensional interactions
424 at the ecosystems level or complex ecosystem responses due to climate change {Rilov, 2019 #324}
425 reflect some of these uncertainty types which are often acknowledged, but still hardly defined,
426 quantified or addressed.

427

428 4. Recommendations for the operationalisation of a CEA

429 Stemming from our analysis, we synthesize here four key recommendations to strengthen the
430 implementation of CEA into management through a risk-based CEA framework:

431 *1) Framing the context and setting risk criteria* – The operationalisation of a CEA requires a well-framed
432 context comprising the identification of the drivers, management objectives, and targets. In the absence
433 of clear objectives that address the avoidance or mitigation of cumulative effects, the CEA process
434 should still formulate or lay out the aspired objectives regarding the tolerance of cumulative effects.
435 Only then, risk criteria can be defined. Risk criteria need to be set prior to the assessment and requires
436 the involvement of stakeholders and decision-makers.

437 *2) Defining the roles* – A clear separation and allocation of the roles and expected tasks of decision
438 makers, various stakeholder groups and scientists is fundamental. This helps to build trust when sharing
439 and interpreting data and knowledge.

440 *3) Reducing and structuring complexity* – CEAs are context-dependent, resource intensive and complex.
441 There are unavoidable trade-offs among the level of complexity, available resources and timelines but
442 they should be reached in a transparent and well documented manner, as cause-effect pathways have
443 to be assessed for each identified link between human activities, pressures, and assessment endpoints.

444 *4) Communicating assumptions and uncertainty* – A cross-cutting issue in successful CEAs is a clear
445 communication of assumptions made throughout the process and types and levels of uncertainty.
446 Emphasis should be put on the selection of tools to present the different dimensions of uncertainty,
447 which accumulate along a CEA process.

448

449 5. Conclusions

450 Our analysis of the context, approaches, and implementation of eleven CEA case studies, which aligned
451 their analyses or evaluation to a risk based CEA framework, revealed the large variation in CEA drivers,
452 objectives and assessment endpoints. A single recipe on how to conduct a CEA does not exist, but the
453 application of a standardised framework facilitated a consistent and coherent comparison of the key
454 issues to operationalise such complex assessments. With this study we underline the urgent need to
455 differentiate CEA in light of the different clients or processes such as governance advice, marine spatial
456 planning or regulatory advice. Thus, laying out the context, assessment objectives and criteria, and roles
457 of those involved, is fundamental to allow for the take up of CEA outcomes in management processes.
458 We suggest that future CEAs should move towards this direction to maximize the advice a CEA can
459 provide in an EBM context. Further, we conclude that it is crucial to communicate uncertainty
460 throughout the various assessment steps in a transparent and structured manner, which helps build
461 confidence and trust in the derived scientific evidence. One of the reasons why CEA have not been
462 formally operationalised yet is their complexity and limitations of knowledge and evidence and the
463 difficulty in identifying which human activity and pressure should be reduced. Applying the risk-based
464 CEA framework together with a strategy of communicating uncertainty should help to overcome
465 bemoaning of imperfect knowledge on the sensitivity of ecosystem components to distinct pressures,
466 and embrace uncertainty around the scientific evidence. Our results underlined the need for further
467 research on the effectiveness of management measures to improve current practices or to develop new
468 ones to reduce the effects of specific human activities. Finally, risk evaluation comprises trade-off
469 analysis of the cost and benefits of additional management measures and should be clearly separated
470 from the provision of technical advice by scientists. We postulate that if the description and
471 quantification of uncertainty and trade-offs becomes a routine in CEA, then decision makers will more
472 likely understand the potential repercussions of their decisions. In summary, our study makes a strong
473 case that CEA should be well framed and recognised as cross-cutting tools that could bridge different
474 management objectives. We acknowledge that mismatches in governance structures can often not be
475 changed, but we suggest that CEA can be one strategic approach to integrate ecosystem management
476 considerations across multiple sectorial policies.

477

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482 7. References

- 483 Amoroso, R. O., A. M. Parma, C. R. Pitcher, R. A. McConnaughey, and S. Jennings. 2018. Comment on
484 "Tracking the global footprint of fisheries". *Science* **361**.
- 485 Aps, R., K. Herkul, J. Kotta, R. Cormier, K. Kostamo, L. Laamanen, J. Lappalainen, K. Lokko, A. Peterson,
486 and R. Varjopuro. 2018. Marine environmental vulnerability and cumulative risk profiles to
487 support ecosystem-based adaptive maritime spatial planning. *ICES Journal of Marine Science*
488 **75**:2488-2500.
- 489 Bates, A. E., B. Helmuth, M. T. Burrows, M. I. Duncan, J. Garrabou, T. Guy-Haim, F. Lima, A. M. Queiros,
490 R. Seabra, and R. Marsh. 2018. Biologists ignore ocean weather at their peril. Nature Publishing
491 Group.
- 492 Baybutt, P. 2018. Guidelines for designing risk matrices. *Process Safety Progress* **37**:49-55.
- 493 Borgwardt, F., L. Robinson, D. Trauner, H. Teixeira, A. J. A. Nogueira, A. I. Lillebø, G. Piet, M.
494 Kuemmerlen, T. O'Higgins, H. McDonald, J. Arevalo-Torres, A. L. Barbosa, A. Iglesias-Campos, T.
495 Hein, and F. Culhane. 2019. Exploring variability in environmental impact risk from human
496 activities across aquatic ecosystems. *Science of the Total Environment* **652**:1396-1408.
- 497 Christensen, V., and C. J. Walters. 2004. Ecopath with Ecosim: methods, capabilities and limitations.
498 *Ecological Modelling* **172**:109-139.
- 499 Coll, M., I. Palomera, S. Tudela, and M. Dowd. 2008. Food-web dynamics in the South Catalan Sea
500 ecosystem (NW Mediterranean) for 1978–2003. *Ecological Modelling* **217**:95-116.
- 501 Cormier, R., M. Elliott, and J. Rice. 2019. Putting on a bow-tie to sort out who does what and why in the
502 complex arena of marine policy and management. *Science of the Total Environment* **648**:293-
503 305.
- 504 Cormier, R., C. R. Kelble, M. R. Anderson, J. I. Allen, A. Grehan, and O. Gregersen. 2017. Moving from
505 ecosystem-based policy objectives to operational implementation of ecosystem-based
506 management measures. *ICES Journal of Marine Science* **74**:406-413.
- 507 Cormier, R., and J. Lonsdale. 2019. Risk assessment for deep sea mining: An overview of risk. *Marine*
508 *Policy*.
- 509 Cormier, R., V. Stelzenmüller, I. F. Creed, J. Igras, H. Rambo, U. Callies, and L. B. Johnson. 2018. The
510 science-policy interface of risk-based freshwater and marine management systems: From
511 concepts to practical tools. *Journal of Environmental Management* **226**:340-346.
- 512 EC. 2008/56/EC. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008
513 establishing a framework for community action in the field of marine environmental policy
514 (Marine Strategy Framework Directive).
- 515 Elliott, M., D. Burdon, J. P. Atkins, A. Borja, R. Cormier, V. N. de Jonge, and R. K. Turner. 2017. "And
516 DPSIR begat DAPSI(W)R(M) !" - A unifying framework for marine environmental management.
517 *Marine Pollution Bulletin* **118**:27-40.
- 518 Gimpel, A., V. Stelzenmuller, R. Cormier, J. Floeter, and A. Temming. 2013. A spatially explicit risk
519 approach to support marine spatial planning in the German EEZ. *Marine Environmental*
520 *Research* **86**:56-69.
- 521 Halpern, B. S., M. Frazier, J. Potapenko, K. S. Casey, K. Koenig, C. Longo, J. S. Lowndes, R. C. Rockwood, E.
522 R. Selig, and K. A. Selkoe. 2015. Spatial and temporal changes in cumulative human impacts on
523 the world's ocean. *Nature communications* **6**.
- 524 Hodgson, E. E., and B. S. Halpern. 2019. Investigating cumulative effects across ecological scales. *Conserv*
525 *Biol* **33**:22-32.
- 526 Horta e Costa, B., J. Claudet, G. Franco, K. Erzini, A. Caro, and E. J. Gonçalves. 2016. A regulation-based
527 classification system for Marine Protected Areas (MPAs). *Marine Policy* **72**:192-198.

528 Jones, F. C. 2016. Cumulative effects assessment: theoretical underpinnings and big problems.
529 *Environmental Reviews* **24**:187-204.

530 Katsanevakis, S., V. Stelzenmüller, A. South, T. K. Sørensen, P. J. S. Jones, S. Kerr, F. Badalamenti, C.
531 Anagnostou, P. Breen, G. Chust, G. D'Anna, M. Duijn, T. Filatova, F. Fiorentino, H. Hulsman, K.
532 Johnson, A. P. Karageorgis, I. Kröncke, S. Mirto, C. Pipitone, S. Portelli, W. Qiu, H. Reiss, D.
533 Sakellariou, M. Salomidi, L. van Hoof, V. Vassilopoulou, T. Vega Fernández, S. Vöge, A. Weber, A.
534 Zenetos, and R. T. Hofstede. 2011. Ecosystem-based marine spatial management: Review of
535 concepts, policies, tools, and critical issues. *Ocean & Coastal Management* **54**:807-820.

536 Knights, A. M., G. J. Piet, R. H. Jongbloed, J. E. Tamis, L. White, E. Akoglu, L. Boicenco, T. Churilova, O.
537 Kryvenko, V. Fleming-Lehtinen, J. M. Leppanen, B. S. Galil, F. Goodsir, M. Goren, P. Margonski, S.
538 Moncheva, T. Oguz, K. N. Papadopoulou, O. Setälä, C. J. Smith, K. Stefanova, F. Timofte, and L. A.
539 Robinson. 2015. An exposure-effect approach for evaluating ecosystem-wide risks from human
540 activities. *ICES Journal of Marine Science* **72**:1105-1115.

541 Korpinen, S., and J. H. Andersen. 2016. A Global Review of Cumulative Pressure and Impact Assessments
542 in Marine Environments. *Frontiers in Marine Science* **3**.

543 Lindegren, M., B. G. Holt, B. R. MacKenzie, and C. Rahbek. 2018. A global mismatch in the protection of
544 multiple marine biodiversity components and ecosystem services. *Sci Rep* **8**:4099.

545 Liversage, K., J. Kotta, R. Aps, M. Fetissov, K. Nurkse, H. Orav-Kotta, M. Rätsep, T. Forsström, A. Fowler,
546 M. Lehtiniemi, M. Normant-Saremba, R. Puntilla-Dodd, T. Arula, K. Hubel, and H. Ojaveer. 2019.
547 Knowledge to decision in dynamic seas: Methods to incorporate non-indigenous species into
548 cumulative impact assessments for maritime spatial planning. *Science of the Total Environment*
549 **658**:1452-1464.

550 Mauri, M., T. Elli, G. Caviglia, G. Uboldi, and M. Azzi. 2017. RAWGraphs: A Visualisation Platform to
551 Create Open Outputs. Pages 1-5 Proceedings of the 12th Biannual Conference on Italian SIGCHI
552 Chapter. ACM, Cagliari, Italy.

553 Menegon, S., D. Depellegrin, G. Farella, A. Sarretta, C. Venier, and A. Barbanti. 2018. Addressing
554 cumulative effects, maritime conflicts and ecosystem services threats through MSP-oriented
555 geospatial webtools. *Ocean and Coastal Management* **163**:417-436.

556 Murray, C. C., S. Agbayani, H. M. Alidina, and N. C. Ban. 2015. Advancing marine cumulative effects
557 mapping: An update in Canada's Pacific waters. *Marine Policy* **58**:71-77.

558 Piet, G. J., R. H. Jongbloed, A. M. Knights, J. E. Tamis, A. J. Pajmans, M. T. van der Sluis, P. de Vries, and L.
559 A. Robinson. 2015. Evaluation of ecosystem-based marine management strategies based on risk
560 assessment. *Biological Conservation* **186**:158-166.

561 Rilov, G., A. D. Mazaris, V. Stelzenmüller, B. Helmuth, M. Wahl, T. Guy-Haim, N. Mieszkowska, J.-B.
562 Ledoux, and S. Katsanevakis. 2019. Adaptive marine conservation planning in the face of climate
563 change: What can we learn from physiological, genetic and ecological studies? *Global Ecology*
564 *and Conservation*:e00566.

565 Rozmus, G., D. J. Smith, and D. A. Baum. 2014. Snares to LOPA action items. *Process Safety Progress*
566 **33**:183-185.

567 Stelzenmuller, V., M. Coll, A. D. Mazaris, S. Giakoumi, S. Katsanevakis, M. E. Portman, R. Degen, P.
568 Mackelworth, A. Gimpel, P. G. Albano, V. Almpnidou, J. Claudet, F. Essl, T. Evagelopoulos, J. J.
569 Heymans, T. Genov, S. Kark, F. Micheli, M. G. Pennino, G. Rilov, B. Rumes, J. Steenbeek, and H.
570 Ojaveer. 2018. A risk-based approach to cumulative effect assessments for marine
571 management. *Science of the Total Environment* **612**:1132-1140.

572 Stelzenmüller, V., M. Coll, A. D. Mazaris, S. Giakoumi, S. Katsanevakis, M. E. Portman, R. Degen, P.
573 Mackelworth, A. Gimpel, P. G. Albano, V. Almpnidou, J. Claudet, F. Essl, T. Evagelopoulos, J. J.
574 Heymans, T. Genov, S. Kark, F. Micheli, M. G. Pennino, G. Rilov, B. Rumes, J. Steenbeek, and H.

575 Ojaveer. 2018. A risk-based approach to cumulative effect assessments for marine
576 management. *Science of the Total Environment* **612**:1132-1140.

577 Stelzenmüller, V., H. O. Fock, A. Gimpel, H. Rambo, R. Diekmann, W. N. Probst, U. Callies, F. Bockelmann,
578 H. Neumann, and I. Kroncke. 2015. Quantitative environmental risk assessments in the context
579 of marine spatial management: current approaches and some perspectives. *ICES Journal of*
580 *Marine Science* **72**:1022-1042.

581 Stelzenmüller, V., J. Lee, A. South, and S. I. Rogers. 2010. Quantifying cumulative impacts of human
582 pressures on the marine environment: a geospatial modelling framework. *Marine Ecology*
583 *Progress Series* **398**:19-32.

584 Stephenson, R. L., A. J. Hobday, C. Cvitanovic, K. A. Alexander, G. A. Begg, R. H. Bustamante, P. K.
585 Dunstan, S. Frusher, M. Fudge, E. A. Fulton, M. Haward, C. Macleod, J. McDonald, K. L. Nash, E.
586 Ogier, G. Pecl, É. E. Plagányi, I. van Putten, T. Smith, and T. M. Ward. 2019. A practical
587 framework for implementing and evaluating integrated management of marine activities. *Ocean*
588 *& Coastal Management* **177**:127-138.

589 Stock, A., L. B. Crowder, B. S. Halpern, and F. Micheli. 2018. Uncertainty analysis and robust areas of high
590 and low modeled human impact on the global oceans. *Conservation Biology* **32**:1368-1379.

591 Visbeck, M. 2018. Ocean science research is key for a sustainable future. *Nature communications* **9**.

592 Willsteed, E., A. B. Gill, S. N. Birchenough, and S. Jude. 2017. Assessing the cumulative environmental
593 effects of marine renewable energy developments: Establishing common ground. *Sci Total*
594 *Environ* **577**:19-32.

595 Willsteed, E. A., S. N. R. Birchenough, A. B. Gill, and S. Jude. 2018. Structuring cumulative effects
596 assessments to support regional and local marine management and planning obligations.
597 *Marine Policy* **98**:23-32.

598

599

600 Table 1. Following the approach of {Stelzenmüller, 2015 #52; Gissi, 2017 #360} we expanded a Walker-type matrix {Walker, 2003 #193} with
 601 eight uncertainty descriptors: context uncertainty, cause-effect model uncertainty, data input uncertainty, statistical uncertainty, scenario
 602 uncertainty, recognized ignorance, epistemic or knowledge-related uncertainty and inherent variability. This allows assessing the dimensions of
 603 uncertainty associated to the risk-based CEA framework and offers a systematic guidance for improving the treatment of uncertainty.
 604

Uncertainty dimensions	Location - Identifies where uncertainty establishes within the methodological approach applied for the RB-CEA. Location can refer to the context, model and input.			Level - Encompasses statistical uncertainty, scenario uncertainty and recognised ignorance.			Nature - The nature of uncertainty can be distinguished as knowledge related and variability related uncertainty	
Uncertainty descriptors	Context	Cause-effect model	Input	Statistical uncertainty	Scenario uncertainty	Recognised ignorance	Knowledge related	Variability related
	Policy drivers for CEA (e.g. problem framing stage or boundaries determined by policies, legislations) and defined risk criteria (i.e. benchmarks) against which the evaluation of cumulative effects is being performed	Uncertainty in assessing cause-effect pathways can relate to (i) the description of causal relationships, (ii) externalities outside the CEA context	The data input relates to pressures and their related effects. It also comprises the data used for assessing the effectiveness of management measures	Uncertainty that can be statistically quantified	The range of possible outcomes of the management measures being considered to reduce pressures and the risk of cumulative effects	A fundamental uncertainty about the mechanisms and functional relationships considered in the CEA	Uncertainty which refers to the imperfection of knowledge; which may be reduced by conducting more research	Uncertainty related to the variability inherent in the studied system

605

606

607 **Figure legends**

608 Figure 1. Spatial distribution of the eleven local, sub-regional and regional case studies which applied the
609 risk based cumulative effects assessment framework (Stelzenmüller et al. 2018) either to evaluate an
610 existing CEA process or to structure a CEA. The latter represent cases where the CEA is still in progress.
611 Note that the exact spatial expansion of respective case study area is not shown.

612
613 Figure 2. Alluvial plot showing the frequencies of the relationships between the CEA drivers, number of
614 human activities, pressures, and the assessment endpoints considered by the eleven cases studies .

615
616 Figure 3. Representation of the logical chain between example human activities (e.g. fishing and
617 aggregate extraction), their corresponding pressures (e.g. selective extraction), sector specific
618 management measures (e.g. conservation measures, sectoral measures), cumulative residual pressures
619 (total selective pressure load in the system despite management measures) and measureable state
620 change of ecosystem components (e.g. benthic recovery, seabed recovery). Further the steps of risk
621 identification, risk analysis and risk evaluation are superimposed to indicate the required elements.

622
623 Figure 4. Confidence matrix for a transparent representation of the quality of pressure data and the
624 confidence of the applied causal pathways. The numbers correspond to the cases studies shown in
625 Figure 1. At the risk identification step the quality of the pressure data can be categorised ranging from
626 poor (spatiotemporal resolution showing a mismatch with spatiotemporal data on ecosystem
627 components), moderate (spatiotemporal resolution showing a partial overlap with spatiotemporal data
628 on ecosystem components), and rich (spatiotemporal resolution showing a sufficient overlap with
629 spatiotemporal data on ecosystem components). The assessment used to derive causality in the
630 relevant cause-effect pathways can be either based on expert knowledge, semi-quantitative, or
631 quantitative assessments. Dark blue implies that a rather low level of uncertainty of scientific evidence
632 should be provided to a regulatory process, middle blue implies that a medium level of uncertainty in
633 scientific evidence could still underpin a planning process and that scientific results with a rather high
634 level of uncertainty (light blue) would still be sufficient when advising the implementation of
635 environmental policies e.g. EU MSFD.

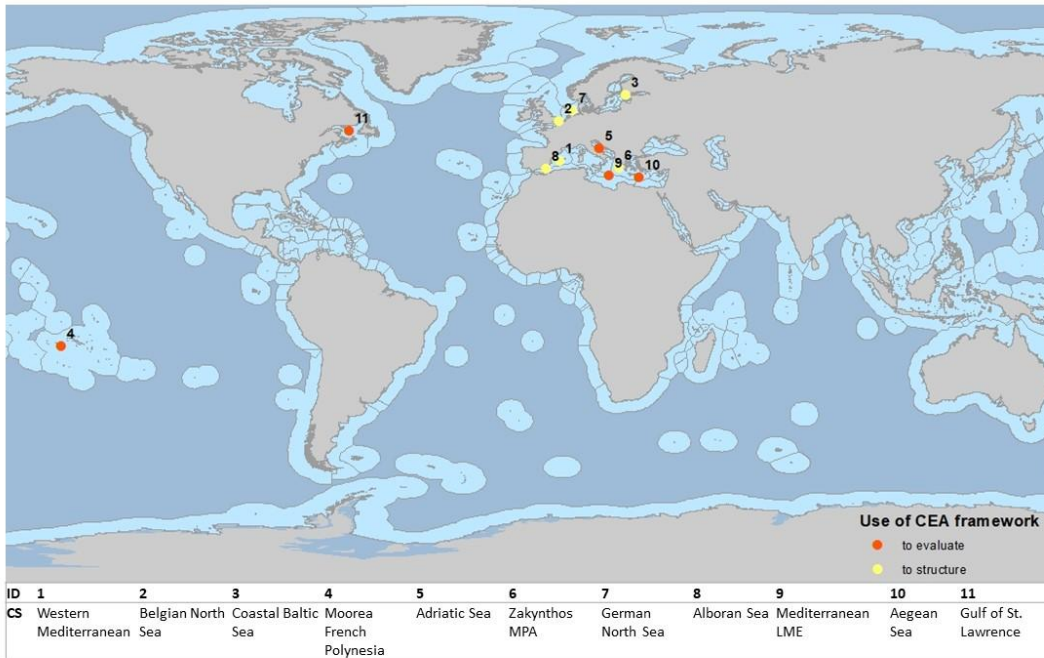
636

637 Figure 5. Risk matrix without tolerance levels (left) in risk analysis versus tolerance levels in risk
638 evaluation. The existing management measures (EM) is compared to the likelihood of a given
639 consequence for the proposed improvements to existing measures or additional measures (PM). Such
640 matrices are used as policy support tool to classify the combination of likelihood and consequences that
641 would be considered as undermining defined policy objectives (red colours) following a gradient of
642 colours (tolerance) to combinations that are considered as not undermining such objectives (green)
643 (Rozmus et al., 2014). These matrices compare the risk of existing management measures versus new or
644 improved management measures. These matrices can only inform managers and stakeholders as to how
645 “bad” things are or could be. They must be derived from the results of the risk analysis of the
646 effectiveness of the management measures given that the decision in risk evaluation is about choosing
647 the management strategy that would reduce the risks as low as reasonably practicable given that risk
648 can never be zero (Baybutt, 2014).

649

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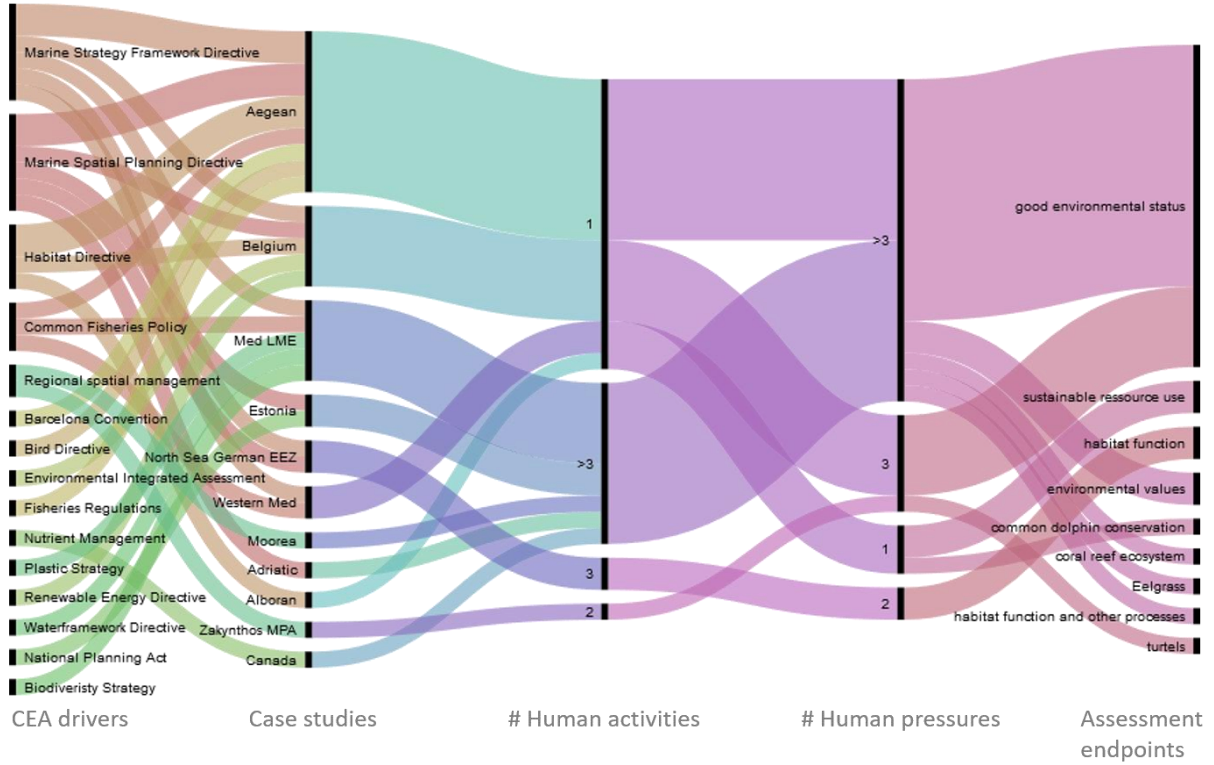
651 Figure 1



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654 Figure 2

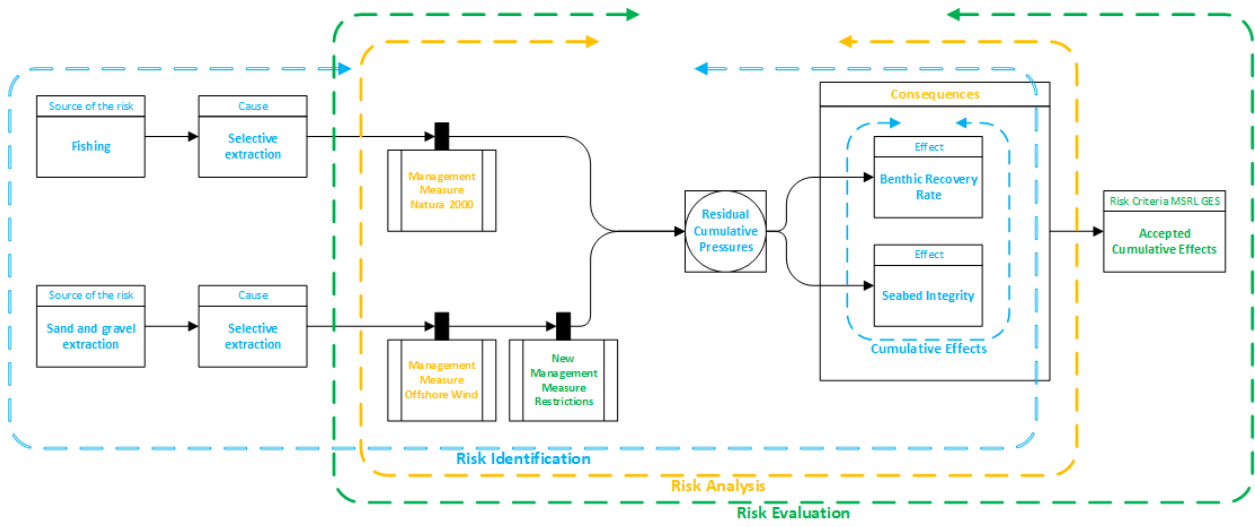


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658 Figure 3

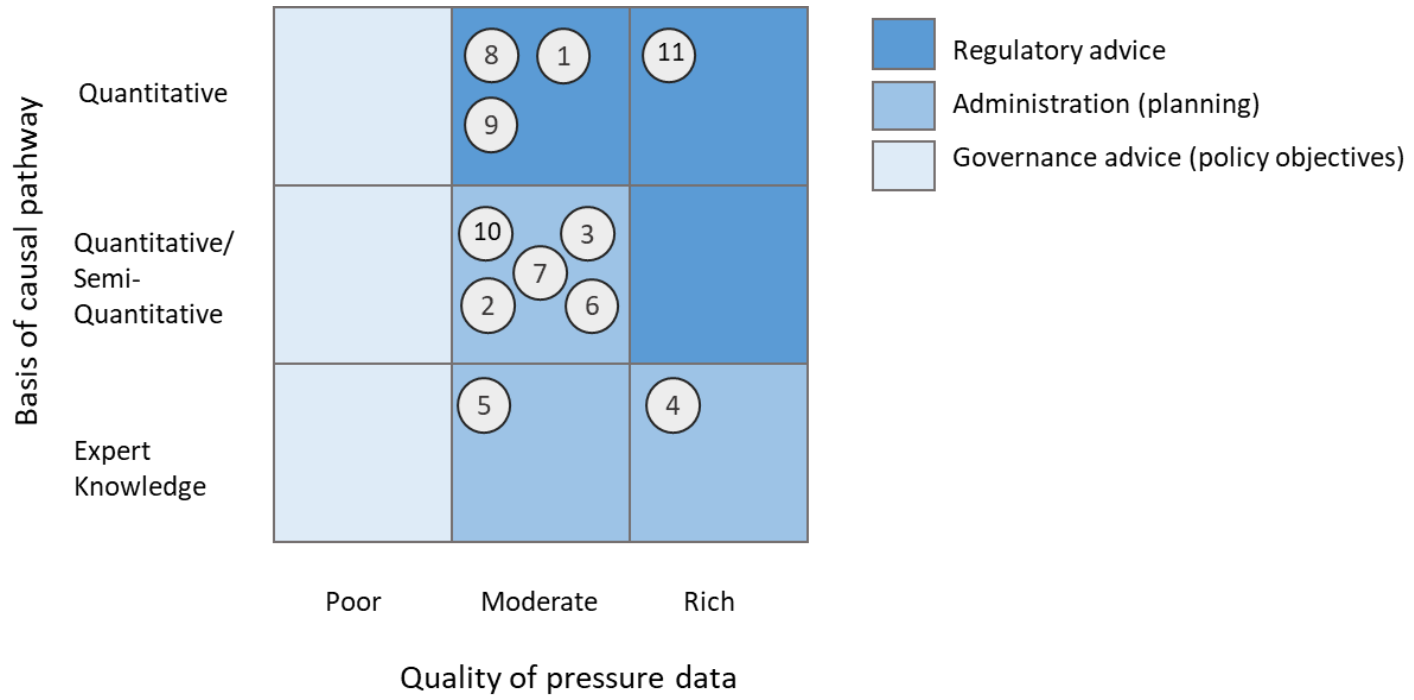


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662 Figure 4



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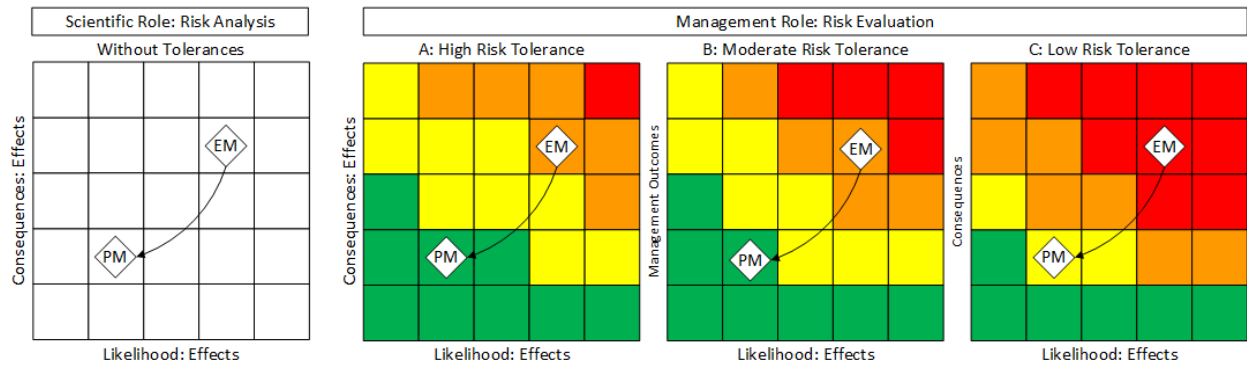
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667 Figure 5

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