Australia's National
Science Agency

Review of Cumulative Effects Assessments (CEAs)

Background to a New CEA process

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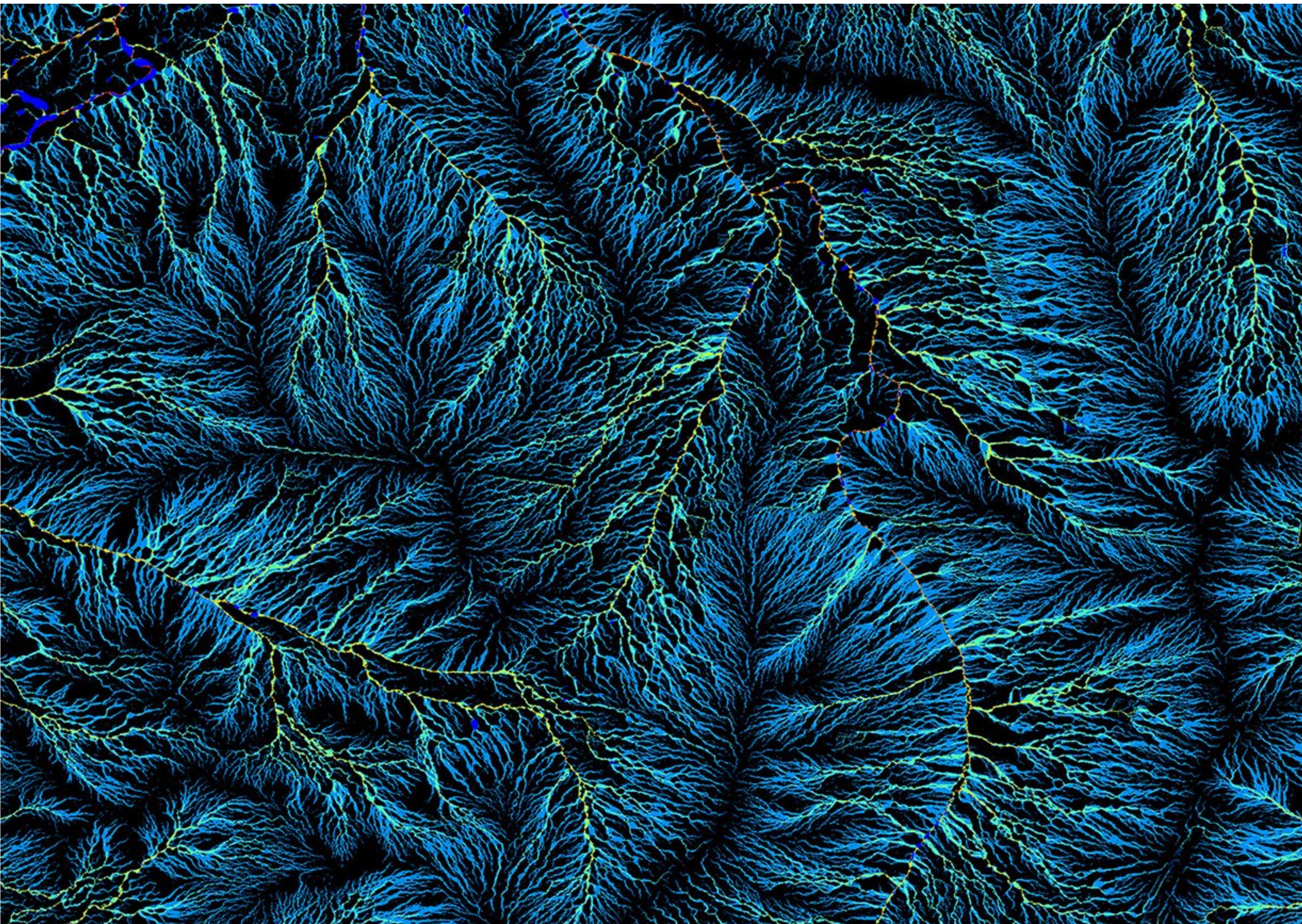

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1 Acknowledgements

We would like to acknowledge funding from the Fisheries Research and Development Corporation (FRDC) and CSIRO. We would also like to acknowledge input from Skipton Woolley, who commented on early versions of the cumulative effects assessment method

2 The Purpose of this Report

This report is part of a Fisheries Research and Development Corporation research study into the development of a cumulative effects assessment for fisheries around Australia. The Australian Fisheries Management Authority (AFMA) has an explicit goal of ensuring that fishing (by Commonwealth commercial fisheries) does not reduce any species populations to/below a level at which the risk of recruitment failure is unacceptably high (Fisheries Management Act 1991). This is explicitly linked to the potential cumulative effects of multiple fisheries, both commonwealth and state and to pressures on stocks from other sources outside of fisheries, including climate. As impacts from other sources increase, it may become increasingly difficult to achieve fisheries objectives. This risk directly motivated the current project – to ensure they are operating sustainably Australian fisheries of all kinds need to understand their cumulative footprint on the species, habitats and ecosystems they interact with and they need to understand how that nests into other large scale pressures, such as climate change.

Previous modelling of Australian ecosystems for the purposes of updating harvest strategies (Fulton et al 2014) or to explore climate effects on Australian ecosystems and fisheries (Fulton and Gorton 2014; Pethybridge et al., 2020) indicates strong non-linear responses, as observed in real world systems around the world (Crain et al., 2008). This indicates that it will be important to attempt to explicitly consider non-linear effects in the assessment method developed for this study. This represents a significant challenge as AFMA is also seeking a spatial assessment (given the extent of their jurisdiction, knowing where to prioritise any mitigations will be important). While spatial additive assessment frameworks are now widely available (e.g. Halpern et al., 2008a; Stelzenmüller et al., 2018), and have previously been applied in Australia (e.g. Hayes et al., 2021; Jones et al., 2018), assessments considering non-additive effects have been rarely attempted beyond a few system-level modelling studies (the scale of which is not yet feasible at a national scale).

To aid the development of the new assessment method this document was created to provide a concise review of cumulative effects assessment techniques available globally; before drawing on that understanding to propose a hierarchical method of creating cumulative assessments, sequentially building from readily available and qualitative information to more complex and quantitative methods as system-type demands (i.e. only moving to non-additive methods where necessary) and as data allows.

3 Introduction

3.1 The Need for Ecosystem Approaches and Cumulative Effects Assessments

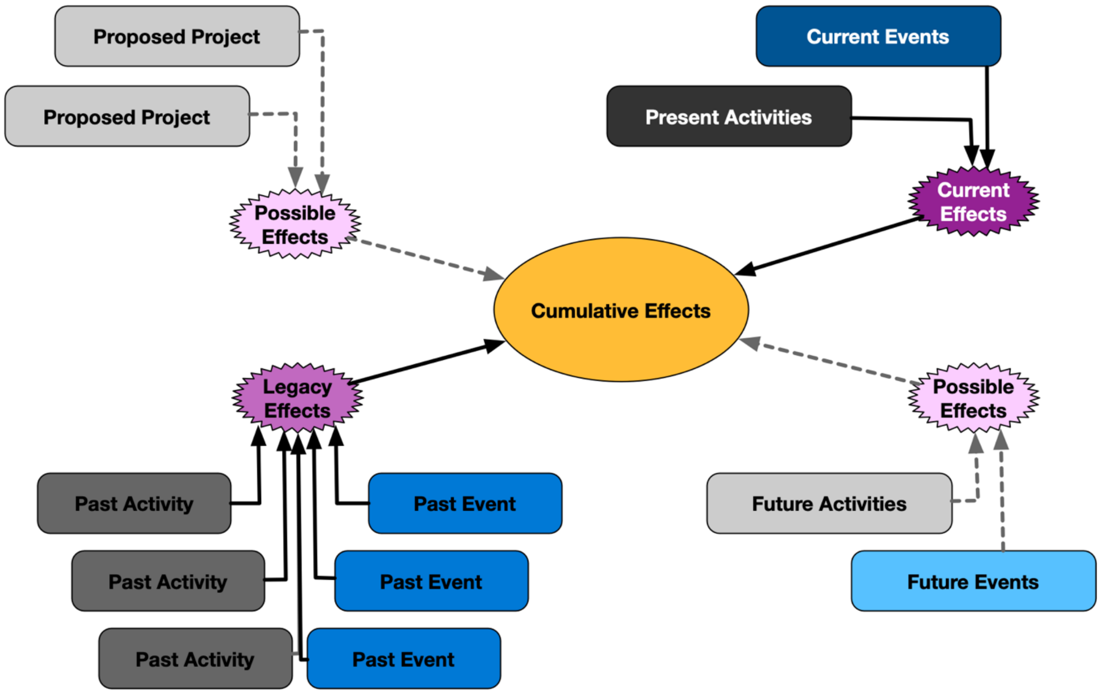
There is a general acceptance globally that the move towards an ecosystem approach to management – such as ecosystem-based fisheries management EBFM (Pikitch et al., 2004) or ecosystem approach to fisheries EAF (FAO, 2003) – is needed to ensure the sustainability of fisheries stocks and the associated livelihoods of those that rely on fisheries for food and income. As use of the ocean by society increases, the potential for direct conflicts between the differing sectors using the ocean and associated impacts increases (Wenhai et al., 2019). Within Australia, there has been growing recognition of the potential for impacts, their interactions and likely accumulation to significantly limit the ability to achieve management objectives (Smith et al., 2017). Overlapping with human activities, the impacts of climate change may increase the potential for impacts to interact and accumulate in marine systems.

The United Nations Fisheries and Agriculture Organisation defines EAF as "striving to balance diverse societal objectives, by taking account of the knowledge and uncertainties about biotic, abiotic and human components of **ecosystems** and their interactions and applying an integrated **approach** to **fisheries** within ecologically meaningful boundaries" (FAO, 2003). This definition links all the potential drivers of change to the sustainability of fisheries. To implement this requires an understanding of how and where human activities and natural events interact with different ecosystem components (including commercially important species) and the identification of avoidance and mitigation measures that can reduce the potential impacts on the system (Halpern et al., 2008a; Levin et al., 2009, Ban et al., 2010; Curtin and Prellezo, 2010). To do this collectively requires an understanding of cumulative effects.

3.2 What are Cumulative Effects

A cumulative effect is a simple concept already reflected in phrases such as "the straw that broke the camel's back". Put simply, every activity has some effect on the system around it (e.g. a single fisherman, an individual jetty etc). As the number of activities increase, via their accumulation through time in the same spatial region and/or expansion over greater spatial areas, the combination of the effects generated on the system results in the occurrence of detectable (and potentially undesirable) change. A mosaic of activities across neighbouring or interconnected areas (e.g. connected by water movements or even migratory animals) can have a compound effect over quite large areas. Figure 1 provides some schematics of how cumulative effects can come about. Panel (a) describes the concept behind cumulative effects, (b) an example from the Great Barrier Reef region where

(a)



The dark colours represent current activities/events/effects, the medium colours past activities/events with lingering effects and the very lightest colours future (as yet unrealised) activities/events/effects (the dashed lines would become solid once realised).

(b)

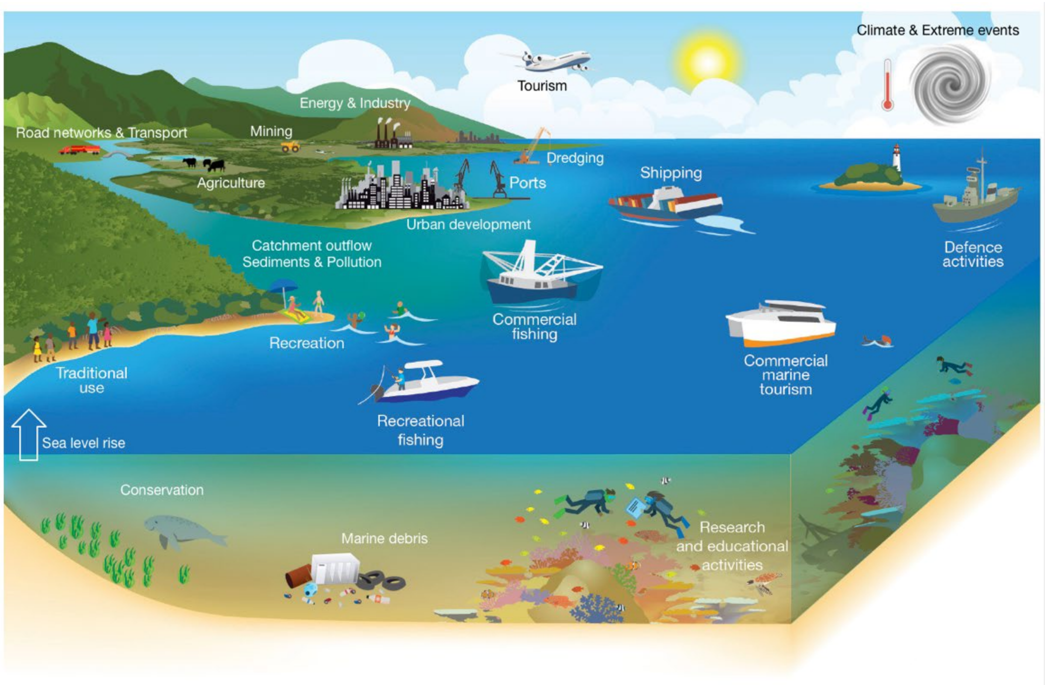


Figure 1: Schematics to convey the concept of cumulative effects. (A) General concept, (B) Schematic of the Great Barrier Reef region showing multiple activities that together could have a cumulative effect, (C) Schematic of a New Zealand watershed showing cumulative effect on the life history of a native fish (over page).

(c)

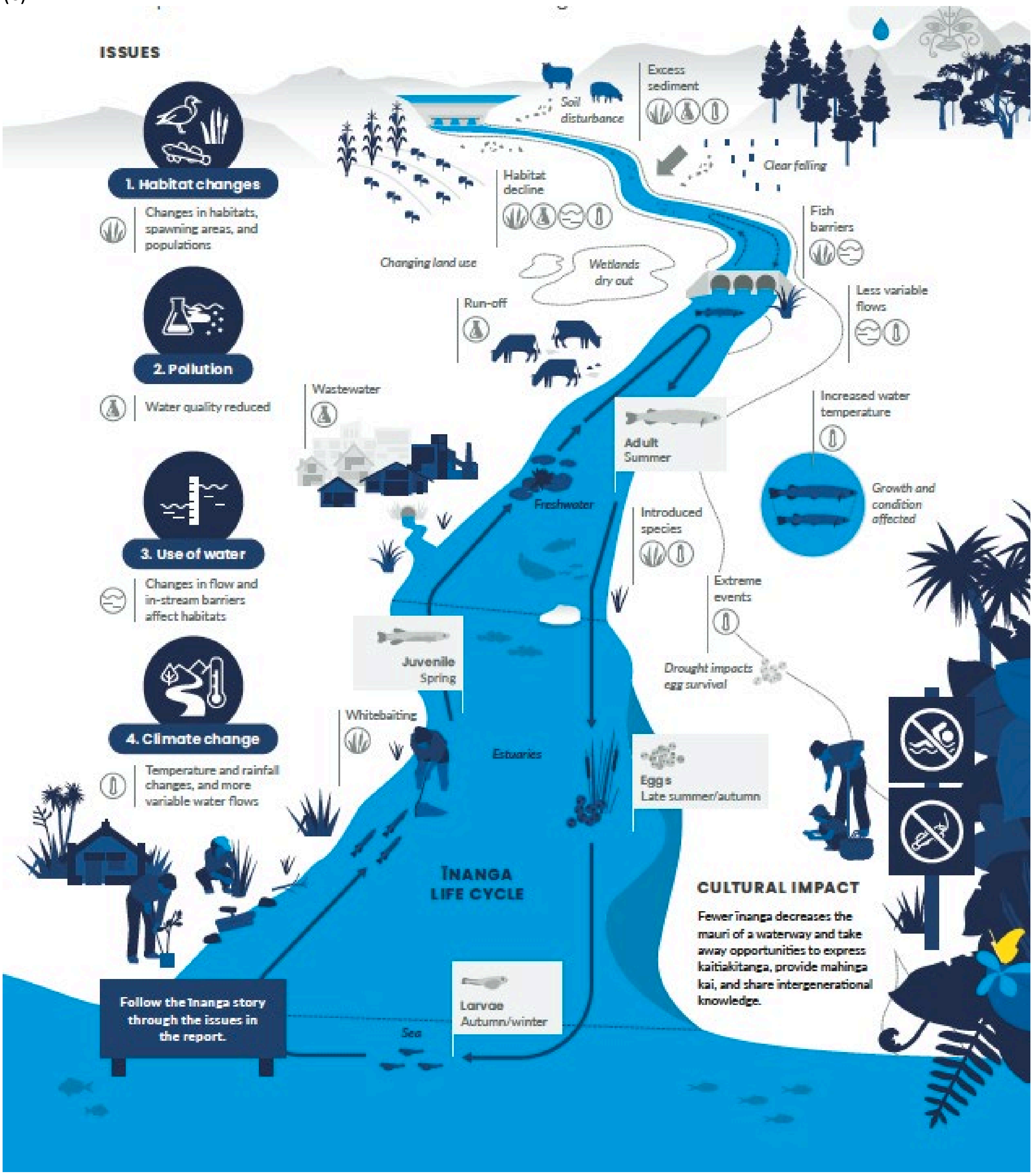


Figure 1: Continued – image from Ministry for the Environment & Stats NZ (2020).

multiple activities or events within one geographic region co-occur (setting up the conditions for cumulative effects to be realised) and (c) shows an example for a native New Zealand fish where different events through the life of the fish, as it moves about, can accumulate to have a deleterious outcome. Examples spanning a single watershed, as in the New Zealand example (or for salmon in the northern hemisphere) are some of the easiest examples to illustrate the concept.

Note historically the literature tends to refer to cumulative impacts, but as not all outcomes are necessarily deleterious for all species or system components, the more neutral term of effects is more commonly used today.

When are effects cumulative?

Cumulative effects can result from interacting activities across spatial areas but also via sequential or overlapping activities in a single location through time, where the effect of one activity has not dissipated before another activity occurs. The most complicated cumulative effects arise from direct and indirect effects of the diverse set of activities that happen within in a region over time. As marine and coastal areas become crowded (as on land) cumulative effects are increasingly likely.

Cumulative effects can be of many forms (Figure 2), but can be simply classified as additive or nonlinear. In physical systems additive effects are often seen (MacDonald 2000), but in ecological systems nonlinear outcomes are more common (Crain et al., 2008; Piggott et al., 2015; Côté et al., 2016). Nonlinear effects occur when the outcome is not simply the same as the result of each individual pressure added together. One pressure may dominate (mask) others; pressures may have a combined (synergistic) effect that is greater than the sum of the individual effects; or the interaction of the pressures may see an outcome that is less than the sum of the individual effects (antagonistic effects) (Figure 2). The form of the cumulative effects is important for directing management interventions, as synergistic effects represent the potential for cost effective intervention (larger outcomes than for the same spend for additive cases), while antagonistic will need careful handling over the long term as intervention will likely require multiple steps and can see things become worse before they improve (Brown et al., 2013). The later highlights an aspect of cumulative effects that often gets little attention – that management activities can themselves generate a nonlinear response.

(a)

	Single Pressure	Multiple Pressures
Occurring Once	Not Cumulative	Potential cumulative effects overlapping in space
Occurring Multiple Times	Potential cumulative effect through time	Potential cumulative effect overlapping in space and time.

(b)

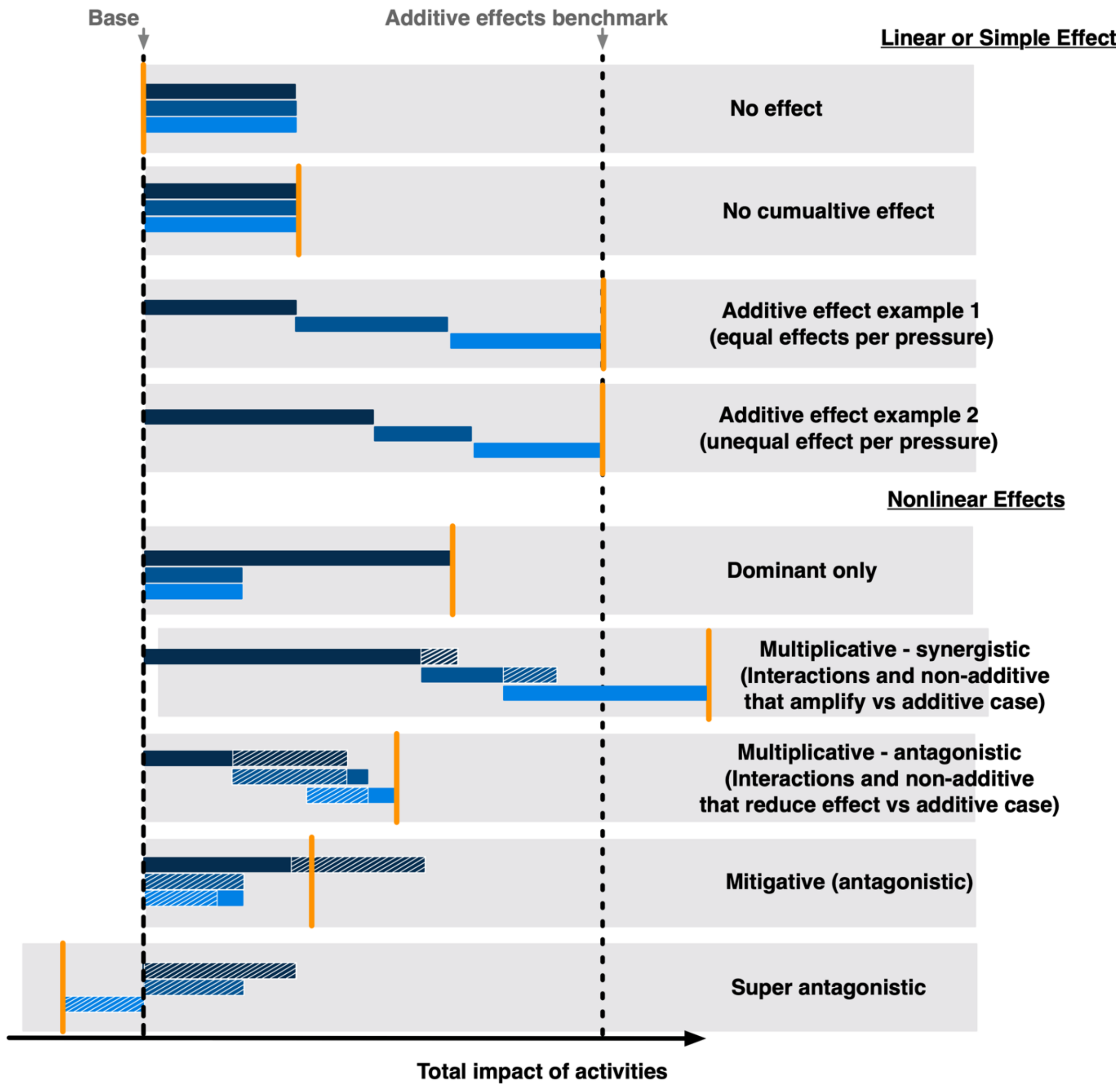


Figure 2: (A) Matrix of conditions leading to cumulative effects; (B) schematic showing different types of effects – non-cumulative and cumulative (additive and non-linear). Non-linear effects are marked by interactions (hashed areas on each bar) meaning the outcomes do not simply add up to the linear sum of the individual effects (the base of no effect and the benchmark of additive effects are shown as vertical black dotted lines so the levels resulting from other effects are clearer). Panel B is modified from Halpern et al (2008b)

3.3 A short History of Cumulative Effects Assessments in Australian Waters

Globally, the examination of cumulative effects in marine environments has expanded rapidly since the first assessments in the 1990's, with increasing attention as the use of marine resources has also expanded and the impacts of climate change have become more apparent. In systems that are more sensitive to change, and systems where there are significant overlapping uses, cumulative effects are becoming more apparent.

Within Australia, cumulative effects are recognized as something to be accounted for by the Environmental Protection and Biodiversity Conservation Act (EPBC)(1999), which calls for consideration of direct, indirect and offsite impacts (specifically upstream, downstream and facilitated impacts). While the EPBC Act allows for broad-scale strategic assessments, it does not indicate what these should entail, nor does it explicitly address cumulative effects in all their forms (Dales, 2011; Dunstan et al., 2020).

Nonetheless, over the past decade there has been an increasing number of cumulative effects assessments (CEAs) in Australia. These have been completed by development proponents (as part of project-related environmental impact assessments), or by government agencies (federal and state) as part of strategic or regional assessments. Notable examples include exercises completed under the NSW Marine Estate Management Strategy (MEMA, 2018) or under Victoria's Marine and Coastal Policy (DELWP, 2020); as well as Parks Australia's Monitoring Evaluation Reporting and Improvement (MERI) framework (Hayes et al., 2021). Regional assessments include those for: Spencer Gulf (Gillanders et al., 2016); Gladstone Harbour (Eco Logical Australia, 2019); and the Great Barrier Reef (Dunstan et al, 2020). Model-based assessments have also been under taken for Gladstone Harbour (Fulton et al 2017) and for the Gascoyne (Fulton et al., 2015) and Kimberley (Boschetti et al., 2020) regions of Western Australia. Apart from the model-based studies the majority of these assessments have taken an additive approach, producing cumulative maps of pressures on marine ecosystems, such as that shown in Figure 3.

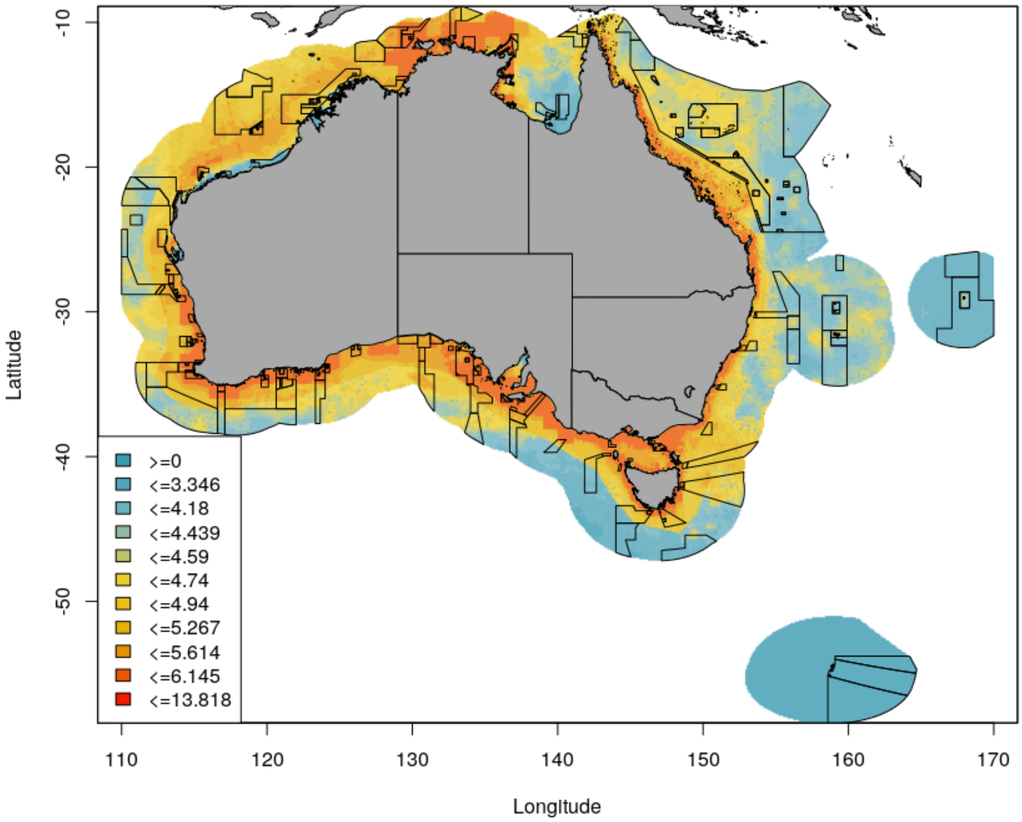


Figure 3: National cumulative pressures map for the Australian EEZ, showing the zone boundaries of the Australian Marine Parks overlayed on a pressure map that is the sum of 39 standardised activity layers (derived from 109 standardised pressures from the period 2011 to 2015). The map should be interpreted as showing the relative intensity of anthropogenic pressures in the Australian EEZ. (From Hayes et al, 2021; refer to that publication for details on the estimation method).

Within Australia, the Great Barrier Reef Strategic Assessment (GBRMPA, 2014) contained the first formal government use of CEAs and provided a framework so that these assessments could be implemented in a structured way that could be repeated through time. The strategic assessment ultimately led to the Reef 2050 Cumulative Impact Management Policy 2018 and the Net Benefit Policy 2018, both of which identify the management of cumulative effects as the core process to improving environmental outcomes for the Great Barrier Reef. Guidance on the implementation of these policies was further detailed by Anthony et al (2013) and more recently as guidelines for CEAs (Dunstan et al 2020). The Guidelines identify a framework for cumulative effects analysis with 5 key steps

1. Understanding Pressures
2. Understanding Values
3. Conceptual models/understanding of ecosystems
4. Zones of Influence, and
5. Risk and Uncertainty

This framework can be applied to other regions and across many types of pressure (whether from climate change or direct human use). In a fisheries context, that means across fished systems, considering all the many types of fisheries active in the area and any other broad scale pressures, such as climate change.

Terrestrial CEA implementations in Australia may not have considered as many sectors as some of the marine examples, but those bioregional assessments undertaken to investigate potential impacts of current and future coal industry development (Barrett et al., 2013) have taken some innovative approaches to combining many data and modelling approaches into a single assessment (Figure 4); thereby providing a generalised approach that could be tailored to specific locations and levels of resources and data availability. This approach steps through five clear steps, which share much with the guidelines put forward by Dunstan et al (2020):

- Contextual information: context and background against which qualitative and quantitative assessments of impact and risk are generated.
- Model-data analysis: evaluates and synthesises information from data and models to develop a quantitative description of the hydrologic relationship between the proposed activities and associated effects on anthropogenic or ecological receptors.
- Impact analysis: reports and records the direct, indirect and cumulative impacts and associated uncertainties of the effects of the activities on system components (values).
- Risk analysis: a scientific assessment of the likelihood of the effects, including the propagation of uncertainties from models and data.
- Outcome synthesis: a synthesis of outcomes used to support scientific advice on management actions.

This understanding of what assessments had already been done in Australia provided a background to a more in-depth global review described in the following sections.

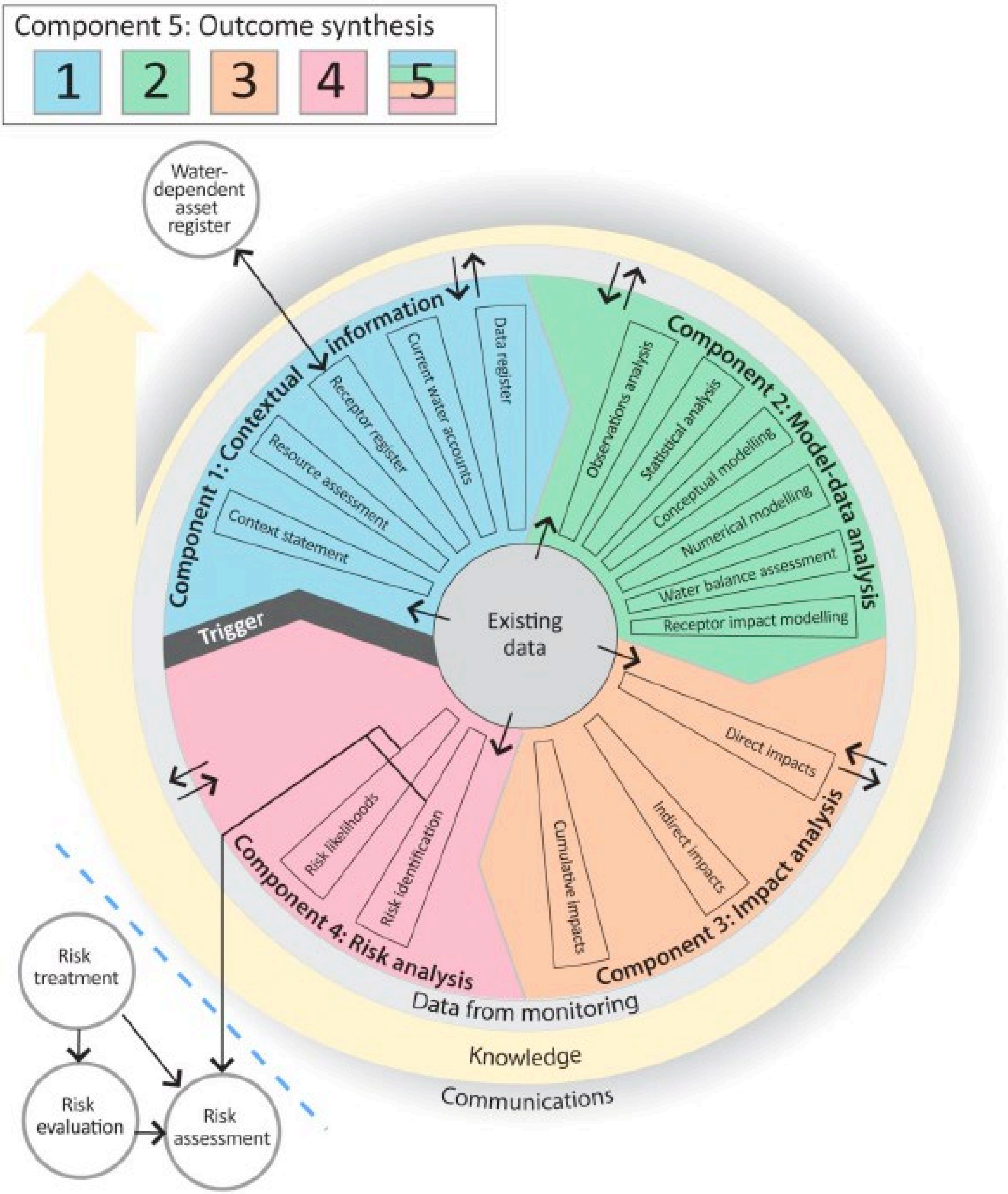


Figure 4: Schematic diagram of the of the bioregional assessment method, which comprises five components, each delivering new information into the while building on prior assessment components. The smaller grey circles indicate activities external to the actual assessment. (Figure from Henderson et al., 2013)

4 Methods

Below we lay out the approach taken in reviewing the relevant literature. The understanding drawn from familiarity with the various methods covered by the review became the heart of a series of expert driven method development meetings held through 2019-2020. The outcome of those meetings is the proposed method presented later in this document. To ensure the method developed in this project is feasible now and into the future, dedicated effort was put into making sure it maximised the use of existing and familiar information and processes – such as that coming from Ecological Risk Assessment (ERA) procedures.

The body of the review was drawn from existing review documents known to the research team and literature searches to identify publications outside the reference databases of the research team. Review documents included were specifically aimed at synthesising past CEAs, such as Korpinen and Andersen (2016) and the cumulative effects chapter of the Second World Ocean Assessment (United Nations 2021), and at identifying different approaches to CEAs (e.g. Hodgson and Halpern 2018) and prioritised methodological challenges (Hodgson et al 2019).

A literature search was undertaken following the PRIMSA protocol. On the 7th of January and again on the 19th of November 2020 a search was carried out using the Web of Science, Google Scholar (https://scholar.google.com/) and Semantic Scholar (https://www.semanticscholar.org/) based on the search phrases; (cumult* OR compound* OR combin*) AND (effect* OR impact*) AND assess* and for the entire time period available. While this set of search terms did go beyond the marine and environmental sectors it provided useful insight into historical precedent, motivation and grounding of more recent methods. Once that foundation was summarised, the body of the review focused on marine and coastal CEAs (on the assumption that this would include assessments directly relevant to fisheries or could be extended to fisheries easily). This stage of the review was constrained to publications from 2000 onwards, as this coincided with the initiation of increasing interest in the marine sphere, as well as a review and refinement of methods used in many of the disciplines using CEAs. In total more than 65 papers were reviewed in drawing together material for this report.

5 Review of Cumulative Effects Assessments

5.1 Historical applications of Cumulative Effects Assessment

Researchers and risk assessors have recognized the need to address cumulative effects since at least the 1970s, but progress has been slow because of insufficient knowledge, inadequate understanding, technological limitations, and scarce funding (Callahan and Sexton, 2007). Through the 1980s and 1990s ecological risk assessment became a recognised means of integrating science, policy, and management to address this aspect of decision making and planning (MacDonald, 2000; Hope, 2006). However, these assessments were still largely done on a sectoral basis and did not attempt more cumulative considerations. Figure 5 shows the timeline of the development of CEAs across the environmental sciences and associated biophysical and medical sciences.

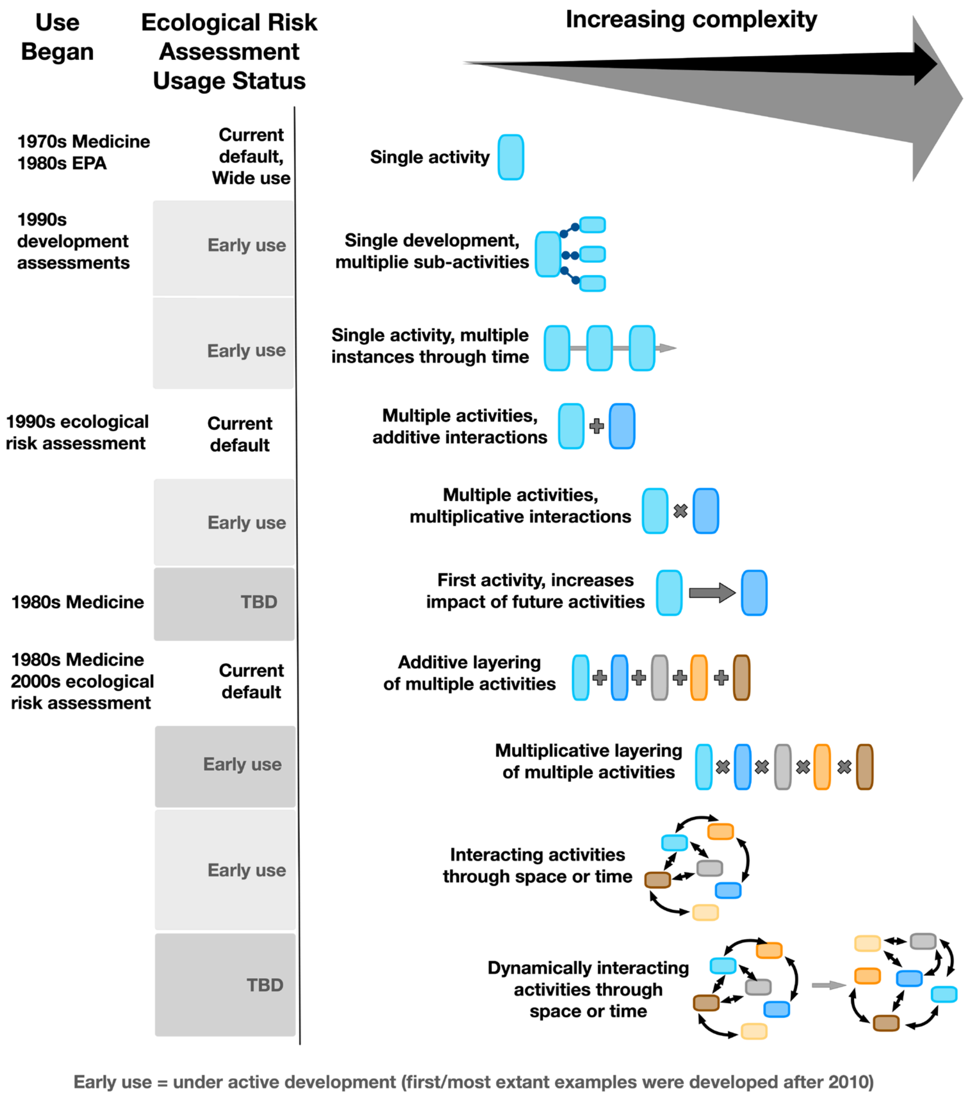


Figure 5: Timeline of development of different approaches of cumulative effects assessments (this captures key developments across a number of related disciplines of relevance to environmental assessments).

The early development of risk assessment methods occurred in parallel in many field, with that in the health and medical science sector leading that of environmental assessments in many ways. In the health sector the methods developed focused primarily on estimating the likelihood of an event. However, limited data availability put estimating the likelihood of events in this way out of reach for ecological assessments in the 1970s and 1980s (Callahan and Sexton, 2007). Risk assessment tables were the dominant means of addressing the issue or assessing risk, particularly in ecotoxicology, through the 1970s and early 1980s. Such tables listed the hazards associated with the stressor, exposure factors and associated risk ratings (with the qualitative risk ratings defined based on combinations of severity and likelihood; e.g. see Figure 6) (Cox, 2008).

		Severity				
		Negligible	**Minor**	**Moderate**	**Significant**	**Severe**
Likelihood	**Very Likely**	Low	Medium	High	Extreme	Extreme
	Likely	Very Low	Low	Medium	High	Extreme
	Possible	Very Low	Low	Medium	High	High
	Unlikely	Very Low	Low	Low	Medium	High
	Very Unlikely	Very Low	Very Low	Low	Medium	Medium

Figure 6: Example of the common risk table approach.

Through the 1980s-early 2000s the field as a whole coalesced around standardising assessment approaches into tiered hazard assessments (Bascietto et al., 1990; Hope, 2006). These begin with a broad qualitative assessment of whether exposure pathways exist between the pressure/stressor and the ecological components of interest (a hazard analysis), from there the assessment progress through more quantitative assessments using available data to create weight-of-assessments in later tiers (Hope, 2006). The first quantitative assessment typically involves numerical estimates of exposure and consequence (ecological responses) to generate “hazard quotients” (which is used as a quasi-index of risk, even though such quotients are not true estimates of risk). During such screening assessments, precautionary and conservative assumptions are used (e.g. no data equates to high risk by default). If the screening and hazard quotients suggest a risk exists, another more quantitative tier may be undertaken to assess the true extent of risk (e.g. via formal exposure modelling, empirical data collection, laboratory tests etc (ASTM, 2003). This hierarchical approach to considering risk remains at the heart of many risk frameworks today and hazard analysis remains the first step of any comprehensive assessment. In many CEAs analysis of hazards is in fact the only assessment made, such as the many expert-based assessments described in Stelzenmüller et al. (2018).

In 1983 the United States Environmental Protection Agency (US EPA) released its seminal “Red Book” guide detailing industry standard practices for assessing risk to the environment at the time (NRC 1983; Holling and Mefe 1996). The guide outlined risk assessments that estimated the probability of adverse effects occurring as a result of a specific human activity (NRC, 1983). The outputs of these assessments were then used to set directives (or commands) that were focused on controlling the release of pollutants into the environment by particular industries. The guide recommended the development of quantitative assessments for ecological endpoints (e.g. species, habitats, or other aspects of the environment that are valued and should be protected; Hope 2006) following a standardised and explicit framework (NRC, 1983). Although this recommendation did not constrain assessments to clearly defined effects (Hope, 2006). Oak Ridge National Laboratory (ORNL) was commissioned by the US EPA to develop the recommended framework for ecological effects/risk assessments – learning from the health assessment approaches described in the "Red Book" they produced a method similar in form to approaches applied for human health, but more suited to assessing ecological risk (Suter et al., 2003). These methods included scoping steps (where the spatial, temporal, sectoral and ecological scope of the assessment is defined), as well as exposure, effects and risk assessment steps (Figure 7). However, while the Red Book simply called for a hazard identification step (Figure 7), the ORNL method has 3 preliminary steps (Suter, 2008) - definition of the "endpoints" (or system components or values of interest), identification of source terms (i.e. determination of where, when, and how the pollutant, or other stressor, entered the environment) and a description of the broader environment (I.e. the environmental context).

The application of the ORNL approach through the 1980s and 1990s saw risk-based approaches become standard in many aspects of planning, impact mitigation and decision making (Hope et al, 2006); with considerable effort put into adding flexibility (so the assessment method could be tailored to the conditions specific to individual assessments) and allowing for the inclusion of nonstandard data in the approach (Suter, 2008), which had previously involved the use of quite specific algorithms. A US EPA update to approved assessment methods released in 1998 explicitly called for consideration of human dimensions in the planning and execution of ecological risk assessments (Cooper, 1998).

During the 1980s and 1990s, expansion of urban and industrial developments drove increasing recognition that the methods used for risk assessments would have to address

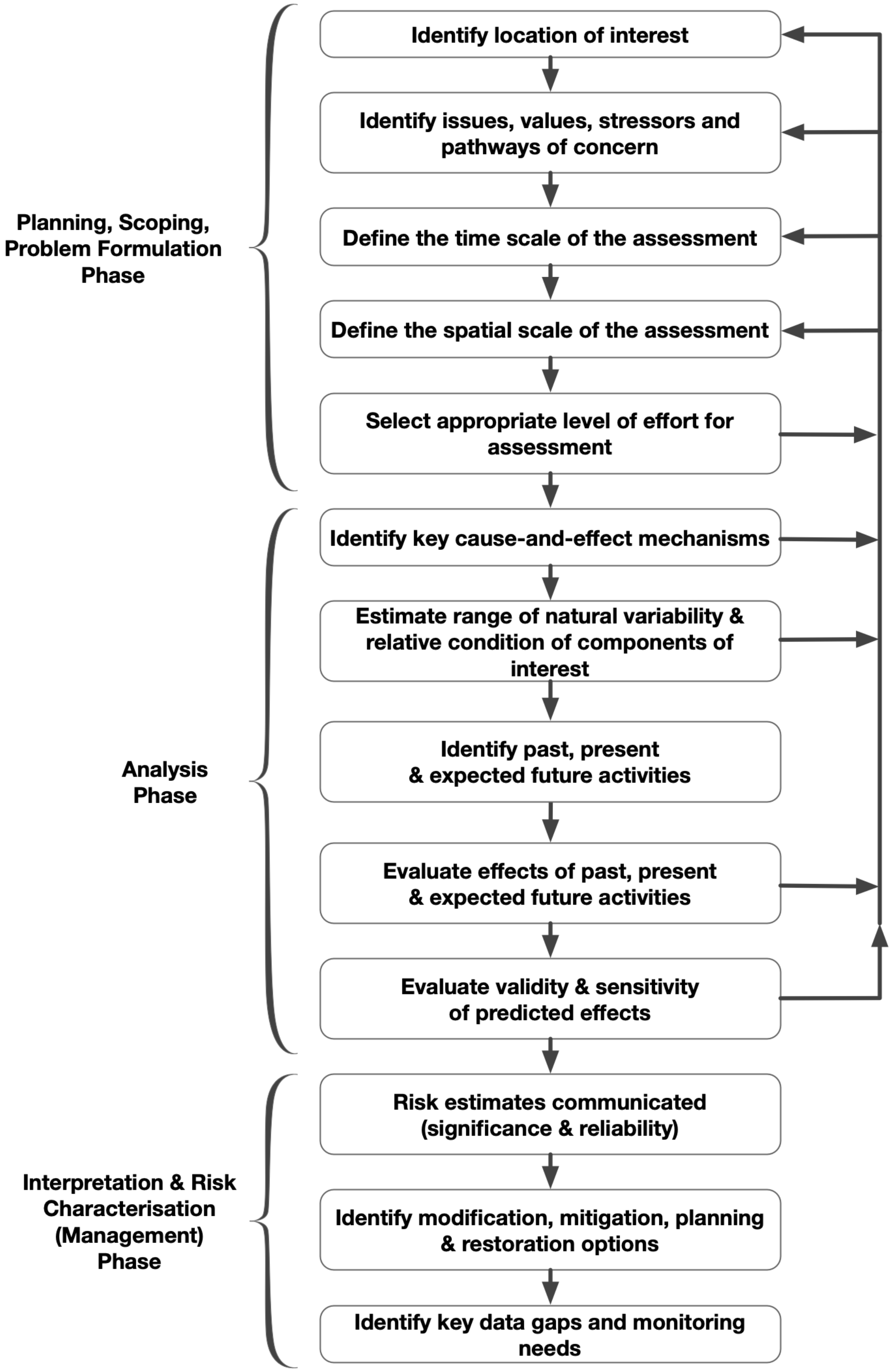


Figure 7: Typical steps in a risk assessment and risk management approach (based on EPA, 2003; MacDonald, 2000)

more complicated questions and trade-offs, involving more species and ecosystem processes, more industries and the various ways they can effect the environment; including through time, over multiple spatial scales and activities (Landis and Weigers, 1997). This led to a revision of the US EPAs guide (US EPA 2003) resulting in the EPA “Silver Book” (NRC, 2009), which introduced new cumulative risk assessment methods that takes an additive approach (i.e. stressor 1 plus stressor 2, see Figure 2b). The Silver Book recommended that cumulative risk assessments be undertaken (lest risk assessment become operationally irrelevant for decision makers simultaneously juggling many uses/stressors/demands), but that the decision-making aspects of the guideline focus more on discriminating amongst risk management options. This thereby narrowed the scope of cumulative risk assessments to stressors that could be influenced by management interventions (directly or indirectly)(Sexton and Linder, 2010). The Silver Book also called for databases to facilitate assessments along with straightforward and versatile analytic tools that could be used by researchers, industry proponents and other stakeholders to undertake screening-level cumulative risk assessments (Sexton and Linder, 2010), effectively level 1 or 2 of the assessment hierarchies outlined above.

Beyond the narrow scope of the ecotoxicological and medically oriented quantitative assessments developed by the US EPA and equivalents around the world, there have been few strictly quantitative advances in risk assessment methods.

5.2 Recent Marine Cumulative Effects Assessments

Interest in CEAs rose sharply from 2000 onwards, especially in marine systems. More broadly, the entire field of CEAs across many disciplines saw review and refinement of the methods applied (e.g. the release of the USEPA cumulative risk assessment guidelines in 2003 and the “Silver Book” in 2008). This renewed interest was driven in the main from an increasing intersection and intensification of human activities in the marine environment, so the need was more clearly realised. In addition, it was becoming evident that low likelihood, but potentially high impact events could no longer be ignored (for example events such as cyclone Katrina), leading to demand for new risk assessment and modelling methods (Lee et al, 2012). The US EPA (amongst others) recognised that a transition from methods focusing on single (primarily chemical) stressors and a limited number of end points (values), sources and pathways of exposure to more holistic approaches was needed. Such approaches needed to have the capacity to assess the combined effects of cumulative exposure to multiple stressors (of multiple types) from multiple sources, acting via multiple and potentially interacting pathways (Callahan and Sexton 2007).

The foundational focus of (environmental) cumulative risk assessments have remained on evaluating the degree to which human activities undermine the achievement of management objectives related to particular ecological components (Samhouri and Levin, 2012); though more often it has default to considering effects on ecological components in general as many CEAs don't link back through a management response to adjust activities in order to attain the management objectives (United Nations, 2021). Moreover, the scale of the ecological focus of the many studies undertaken over the past two decades has varied, from assessments of single stressors or a limited number of species and habitats (e.g. Korpinen et al, 2012; Marcotte et al, 2015; Coll et al, 2016) through to broad whole-of-system approaches, where all existing stressors and their effects across a suite of ecological components present in the marine environment are included in the assessment (Halpern et al. 2008a; Stelzenmüller et al, 2018).

Regardless of scope, marine CEAs have increasingly been seen as reflecting the interactions between science, policy, and management for a range of environmental issues (Knights et al, 2015). The complexity of these marine socioecological systems (natural and human parts of the system together) further entrenched hierarchical approaches to CEAs. Qualitative methods (typically expert based) have been used as screening tools for rapid or "broad but shallow" (strategic) assessments (e.g. Fletcher, 2005; Breen et al., 2012), while richer but more targeted quantitative methods (e.g. Samhouri and Levin, 2012), have been used to inform more tactical decision-making (Knights et al, 2015).

Foresighting - or expert-based description of system structures, connections, causal mechanisms and potential outcomes (Cook et al 2014) - is still the most widely used approach for complex risk assessment situations, as it is a rapid method that can cross scales and scope with vagary (Callahan and Sexton 2007). While such exercises can be immensely insightful, particularly if they draw on a diversity of knowledge types that have a deep understanding of the system (Aminpour et al., 2021) they tend to under-estimate variability, bifurcation points or the consequences of non-linear interactions of multiple variable types, whose alignment may see step changes in behaviour (Surowiecki, 2004; Kahneman, 2011). While experts can vary significantly in the level of risk associated with a situation there is a strong tendency to systematically under estimate risk, as shown by risk and "noise" audits (Kahneman et al., 2021).

Stelzenmüller et al. (2018) reviewed 154 cumulative effect studies and found that spatial data on the distribution of activities, implemented usually through Geographic Information Systems (GIS), was a core component of nearly all assessments. Stelzenmüller et al. (2018) also noted that expert-derived information (i.e. expert elicitation) was a common (though not universal) feature of assessments. This is at least in part due to the availability of resources for conducting assessments, but also because of incomplete and heterogeneous data availability and quality. However, the need for repeatability and transparency for planning purposes has seen a growing number of research and assessment groups work on novel integrative methods, such as a combination of qualitative data and qualitative modelling (Stelzenmüller et al, 2018).

The main classes of methods used in cumulative effect assessments are summarised in Table 1, highlighting strengths and weaknesses known to be associated with the various methods. Hierarchical approaches remain an effective way of dealing with issues of scope and available information. In particular, hierarchical approaches provide a cost and resource effective means of dealing with the complexity of multi-sector, multi-activity, multi-value assessments because they use simple less data-intensive methods for scoping and initial screening and move to more data-intensive methods if follow-up assessments with quantified risks or effects are required for specific decision making (Callahan and Sexton, 2007; Samhouri et al., 2014; Holsman et al., 2017). More quantitative approaches (i.e. additional tiers of the hierarchical approach) however are likely to become increasingly needed for decision making and planning purposes, as a CEA that results in “nothing more than a cursory checklist covering a smorgasbord of issues.... may satisfy legal requirements, but it typically provides minimal guidance to managers or decision makers” (MacDonald 2000).

On a technical note, while managers and non-technical audiences consider all forms of CEAs, risk assessments, vulnerability-based assessments, and more qualitative or semi-quantitative impact assessments, are what is formally known as hazard analyses. They identify points of potential risk, but do not yet provide true insight into the degree of real risk as there is no absolute measure of risk produced by the assessment. Indeed, recent validation exercises suggest that for at least some ecosystem values (e.g. seagrass), the most commonly used CEA approaches do not reflect realised risk (Stockbridge et al 2021). Two general approaches are currently employed to estimating risk in extending these kinds of hazard analyses to actual risk assessments. The first continues the likelihood-consequence approach (e.g. Astles et al., 2006; Williams et al., 2011), particularly for rare, or as yet unobserved, events (Knights et al, 2015). The second, exposure-effect analysis, is typically employed for assessing ongoing pressures (Smith et al., 2007).

Whether considering hazards or true risk, the complexity and scale of cumulative effects, and the paucity of supporting knowledge on causal relationships, means that to date no single method has had the capacity to cover all aspects of cumulative effects (Hodgson and Halpern, 2018). In many instances the complexity of CEAs remains a major stumbling block, leading to simplifications in one or more dimensions to make assessments tractable. These simplifications might include the spatiotemporal extent, or the range of activities included, or assumptions pertaining to non-linearity and additivity (Korpinen and Andersen, 2016). In attempting to account for or rectify issues of complexity another significant challenge associated with ease of use (the validity of the underlying method) can become evident. Even twenty years ago it was recognised that methods originating from (often time and capacity stretched) regulatory agencies, were simpler (and thus more widely used), but often had significant gaps in logic or lacked validation; while methods developed by researchers were theoretically grounded and validated, but were also complex and thus rarely used beyond specialist circles (MacDonald 2000). This balance of extent and ease of use likely explains the broad adoption of simple additive GIS-based methods (e.g. Halpern et al 2008a; O’Hara et al 2021). This is despite wide appreciation that these methods do not address non-linear interactions, nor non-additive or indirect effects (MacDonald, 2000; Hodgson and Halpern, 2018; Hodgson et al 2019), which make up a sizeable portion of all marine effects (Crain et al., 2008; Brown et al., 2013; but see Stockbridge et al. 2020). Concerningly, given the risk and management context of the assessments and the reliance on qualitative or data sources of mixed pedigree, uncertainty with respect to both the

Table 1: Summary of major types of cumulative effects assessments. We have not stated any specific strengths or weaknesses here as that perception may be determined by the specific circumstances of an assessment; instead, we have tried to tease out all the facets of an assessment that may be important in deciding whether a method is appropriate or not (so the potential user can judge strengths and weaknesses for themself). Note that, due to the width of the table, the table has been split into two sections, with each "type" (row) is repeated in each section, with an example reference for each type of assessment given in the second section of the table.

Type	Description	Interaction Type	Response Type	Scale & Resolution	Method Type
Expert elicitation (effects and interaction matrices or lists)	Experts, ideally from diverse background, are asked to contribute their knowledge on potential threats, connections (impact pathways) etc.	Typically, independent, additive or facultative only	Mainly direct; depending on participants indirect effects may also be captured.	Typically, a non-spatial snapshot, but spatial variants possible. Non-dynamic, would need to repeat for different time periods	Elicited / conceptual. Explicitly defines links between pressures and values/effects.
Qualitative system modelling	Signed diagraphs showing the positive/negative connections between different system components (pressures and values). The outcome of those interactions can be explored via press perturbation analysis (where a pressure is increased or decreased and the model predicts the direction of change of the values)	All possible (dependent on outcomes of press perturbation analysis)	Mainly direct (indirect and non-linear possible via press perturbation analysis)	Non-spatial snapshots (would need to repeat for other locations, time periods)	Elicited / conceptual. Explicitly defines links between pressures and values/effects.
Cumulative impact and structured decision-making (CISDM) framework	Uses qualitative and probabilistic (BBN) modeling to provide a systems-level understanding of how cumulative stressors affect habitats and ecosystems of interest (originally the Great Barrier Reef). The modeling provides information on precautionary spatial and temporal boundaries ('zones of influence') for the assessment of development proposals. The zones are integrated with a structured decision-making process to support informed choices regarding a range of possible intervention/management/development scenarios.	All possible (dependent on outcomes of press perturbation analysis). BBN's cannot represent feedbacks directly.	Mainly direct (indirect and non-linear possible)	Can be explicit if information for mapping available, but as originally applied was implicit (though was applied "per region" originally so had some high-level spatial resolution)	Elicited / conceptual foundation, but can be based on quantitative modelling if available. Explicitly defines links between pressures and values/effects.
Exposure or usage maps	Simple overlays of spatial maps of usage or environmental properties; often treated additively. May be encompass all pressure or only those considered relevant to a particular value. May also be combined with multi-criteria decision analysis if using it in combination with marine spatial planning.	Additive	Linear and direct	Spatial snapshot	Empirical

Type	Description	Interaction Type	Response Type	Scale & Resolution	Method Type
Weighted cumulative pressure mapping (OHI, ODEMM)	Related to exposure maps, but goes further to consider relative contribution of affect. A spatial map of the presence (or intensity) of an activity is multiplied by the vulnerability (or impact weighting) for the value of interest and then the cumulative effect calculated by summing across uses per spatial cell. This is the most commonly applied CEA approach; more than half of all publications are variants of the original Halpern et al. (2008) and there are also a significant number of CEAs using analogous methods developed independently.	Additive	Linear and direct	Spatial snapshot (can extend over a longer period dependent on when data were collected)	Empirical
Statistical modelling - hierarchical statistical assessments	Use of statistical methods to model exposure (spatial overlap) and response relationships between pressures and values; resulting outputs are measures of absolute risk	Have often been applied independently or additively, but if sufficient information exists other interaction types can be represented	All possible with sufficient available data (linear and additive easiest)	Typically, a spatial snapshot (spatial time series possible)	Empirical
Bayes Nets (BBN)	BBNs are networks of causal linkages with each node of the network associated with a transition matrix of conditional probabilities that indicate the likelihoods of an increase, decrease or no change in the state of the node (given the state of connected nodes).	All possible (dependent on interaction of transition matrices); cannot represent feedbacks directly.	Mainly direct (indirect and non-linear possible)	Non-spatial	Depending on data sources – conceptual, empirical, mechanistic
ERA/ERA-like	A tiered assessment approach that aims to rapidly identify potentially vulnerable species, prioritizing them for more rigorous assessment and management responses. Tier 1 (the entry tier) consists of an expert driven Scale Intensity Consequence Analysis (SICA); for any species considered to be risk under Tier 1 a Tier 2 analysis is carried out – for data poor species this is a Productivity-Susceptibility Analysis (PSA), but for species with catch and distributional data a Sustainability Assessment for Fishing Effects (SAFE) estimate of fishing mortality is made. Further (Tier 3) quantitative analyses are possible for species considered high risk under Tier 2 assessments, these are usually in the form of a formal fisheries assessment (population dynamics) model. Variants on this approach estimate absolute risk, allowing for cumulative pressure to be ascertained additively.	Additive	Linear and direct	Snapshots (depending on the method these may be spatial)	Empirical

Type	Description	Interaction Type	Response Type	Scale & Resolution	Method Type
IEA	A system-wide iterative assessment that requires ongoing stakeholder engagement, scoping, identification of indicators for monitoring/assessment, ecosystem assessment (status/trends/performance per indicator vs management objectives), risk assessment (regarding likelihood each indicator will reach/remain at an unacceptable level), uncertainty assessment, evaluation of management options, monitoring and evaluation through time. There is strong overlap with CEA in terms of methods that could be used in the assessment steps. IEA is directed at integrated management (which could reduce cumulative effects if the exist), whereas CEA is focused on the cumulative aspects first and foremost.	Depends on methods used (only additive in the simplest methods); all interaction types possible (if sufficient information available to base model detail on).	All possible, depending on underlying model formulation (only linear and direct for the simplest methods)	Depends on the method used (the simplest are non-spatial snapshots, higher-level methods are spatiotemporally dynamic)	Mechanistic
Quantitative - MICE models	Small number of interacting species (e.g. trophic connections, habitat dependencies, technical interactions within fisheries), potentially also with environmental driver and/or multiple fleets. Used to consider management questions. Involves robust quantitative (statistical) fitting of model variables in line with best practice fisheries assessment methods	All interaction types possible (if sufficient information available to base model detail on)	All possible, depending on the underlying model formulation	Typically, only time dynamic	Mechanistic
Quantitative - whole of system models	Consider key/representative parts of the entire system in a variety of ways (either as age structured populations, biomass pools, size- or trait-based bins); to date primarily focused on climate and fisheries relevant system dynamics, but can represent the footprint of other human uses too.	All interaction types possible (if sufficient information available to base model detail on)	All possible (is dependent on model formulation but non-linear responses are common)	Time dynamic, many models are also spatial	Mechanistic

Type	Description	Interaction Type	Response Type	Scale & Resolution	Method Type
Bioregional assessments	A multistep CEA process involving • define, characterise and explain conceptual models that establish causal pathways describing the chain of interactions and events connecting the activity to potential effects on system components (anthropogenic and ecological) • generate quantitative, semi-quantitative or qualitative analyses of the likelihood of effects • assessments of the likelihood of risks to system components due to proposed developments • provide information on the level of confidence of scientific advice on these effects • identify monitoring programs, assessment review frequency and additional risk assessment studies that could be undertaken outside of the main bioregional assessment process to help minimise any effects Originally undertaken for coal seam gas and coal mining developments on water resources.	All interaction types possible (if sufficient information available to base model detail on)	All possible (is dependent on model formulation)	Conceptual models non-spatial, but dynamic models are time dynamic, models may also be spatial	Elicited/conceptual and mechanistic (different ones used in different steps of the process)
Meta-analysis based	Use meta-analysis to synthesize the ecological consequences of anthropogenic threats to species or habitats	All possible (as based on observations)	Direct (shape can be linear or non-linear though shape not reported in the original application)	As defined by observations but reported in aggregate	Empirical
Potential recovery rate	Pressure assessment generated by collating information on anthropogenic pressures on system component of interest (originally seabed) and linking with maps of the distribution of the component or proxies (e.g. sediment types). Then consider cumulative effects possible under 4 scenarios - single greatest, additive, antagonistic, and synergistic	All (via scenarios)	Direct, linear and simple non-linear possible	Typically, a spatial snapshot (spatial time series possible)	Empirical

Type	Values Considered *(geochemical/physical, species, habitat, economic, social, cultural),*	Data Demands	Uncertainty Handling	Updatable/Testable, Validated	Predictions or Attribution Possible?	Key Reference
Expert elicitation (effects and interaction matrices or lists)	All possible though often physical and ecological only	Low (quantitative data unnecessary), but best served if the participants are drawn from diverse and experience rich backgrounds	Qualitative via IPPC style confidence rating. Structural uncertainty often poorly considered. Some consideration of domain bias is possible via clustering.	Method is easy to apply, but objective repeatability difficult (growing literature on Delphic processes helps)	Predictions possible within a foresighting framework, but often not undertaken. Cannot perform objective attribution	Singh et al. (2017)
Qualitative system modelling	All possible	Low (quantitative data unnecessary), but best if supported by knowledgeable participants and background literature or response data	Not in isolation (needs to be linked with BBN)	Press perturbation outcomes testable, but best if linked with BBN (as then updatable)	Qualitative (directional) predictions and (some) attribution possible (more effective if linked with BBN)	Dambacher et al. (2009)
Cumulative impact and structured decision-making (CISDM) framework	All possible, but has typically been ecological	Mixed. Can begin with qualitative, but best if can integrate quantitative data (though that can become data intensive quite quickly if multiple values considered)	Explicit uncertainty index produced – combination of uncertainty regarding model input values, observational/expert error, distribution of conditional probabilities used, and structural uncertainty	Produces testable outputs with associated likelihoods and the model content/ parameterization can be updated with new information.	Generates predictions and attribution is possible if sufficient data is available on which to base the analysis	Anthony et al. (2013); Uthicke et al. (2016)
Exposure, usage and vulnerability maps	Species or habitat typically	Reasonably high – requires quantitative GIS maps for each pressure and value, as well as weightings per pressure for each species	Poor or non-existent	Poor or non-existent	No	Tuda et al. (2014)
Weighted cumulative pressure mapping (OHI, ODEMM)	Species or habitat	Reasonably high – requires quantitative GIS maps for each pressure and value, as well as weightings per pressure for each species	Poor or non-existent	Poor or non-existent	No	Halpern et al (2008a)

Type	Values Considered *(geochemical/physical, species, habitat, economic, social, cultural),*	Data Demands	Uncertainty Handling	Updatable/Testable, Validated	Predictions or Attribution Possible?	Key Reference
Statistical modelling - hierarchical statistical assessments	All possible, but has typically been applied to species or habitats	High – the more available empirical data the more robust the resulting estimates	Explicit	A strength of this approach, as it can generate predictions based on a likelihood with an associated quantified uncertainty	Generates predictions and attribution is possible if sufficient data is available on which to base the analysis	Hunsicker et al (2016); Teichert et al (2016); Dunstan et al ref; Woolley et al (2016)
Bayes Nets (BBN)	All possible	Can be constructed from a variety of data sources (from purely expert based to quantitative). Amount of data required depends on data source type and model complexity	Explicit	Produces testable outputs with associated likelihoods and the model content/ parameterization can be updated with new information.	Produces simple one step predictions of change. Attribution is possible in many cases	Anthony et al. (2013)
ERA/ERA-like	Species or habitat	Hierarchical, with low data demands at lower assessment levels and more data required as more quantitative (higher levels) applied	Improving (originally rarely considered, but now uncertainty around base data is being represented more explicitly)	Yes, if applied iteratively over time	Not typically	Hobday et al. (2011)
IEA	Species or habitat typically	Can be applied hierarchically, but typically high due to the number of components involved	Explicit	Is intended to produced testable outputs and to be updated with new information	Depends on the method employed, the higher-level methods allow for prediction and attribution	Holsman et al. (2017)
Quantitative - MICE models	All possible, but species or habitat typically (economic also common, but sociocultural rare to date)	Requires reasonably good quality time series of values and pressures of interest (for statistical fitting)	Explicit	Produces testable outputs and can be updated with new information.	Prediction standard, attribution amongst explicitly represented pressure possible	Plagányi et al. (2014)

Type	Values Considered *(geochemical/physical, species, habitat, economic, social, cultural),*	Data Demands	Uncertainty Handling	Updatable/Testable, Validated	Predictions or Attribution Possible?	Key Reference
Quantitative - whole of system models	All possible, but greatest focus has been on physical, species, habitat and economic	High – requires data for all modelled aspects	Possible, can be constrained by computational cost of running alternative parameterisations or model structures	Produces testable outputs and can be updated with new information.	Prediction standard, attribution possible	Fulton et al. (2017); Coll et al. (2014); Blanchard et al (2014)
Bioregional assessments	All possible (anthropogenic and ecological)	Mixed. Can begin with qualitative, but best if can integrate quantitative data (though that can become data intensive quite quickly if multiple values considered)	Explicit	Produces testable outputs with associated likelihoods and the model content/ parameterization can be updated with new information.	Prediction standard, attribution possible	Barrett et al (2013)
Meta-analysis based	All possible (though dictated by what is present in the literature; typically effects on species or habitats)	High (as there must be sufficient to undertake a meta-analysis)	Explicit (ranges of response in meta-analysis)	Can be updated with new literature as it becomes available	Produces simple one predictions of response shapes. Attribution is possible as effect defined per pressure	Claudet and Fraschetti (2010)
Potential recovery rate	All possible (though dictated by what is present in the literature; typically effects on species or habitats)	High – requires data for all effects considered	Some – via considering different scenarios for the form of the interaction type	Testable. Updates possible once new data becomes available	Produces simple predictions which can be tested via comparing with independently estimated recovery rates	Foden et al (2011)

inputs and outputs of CEAs has often not been adequately addressed (Stelzenmüller et al, 2018). Jones et al (2018) is a rare exception, using a method that explicitly recognising uncertainty in expert elicitation.

In many cases, assessments fail to directly link back to specific management actions and therefore there are few instances where possible risk mitigation is considered explicitly under alternative management arrangements (Stelzenmüller et al., 2018). The weak link to action included in current assessments is reflective of a lack of consideration by many management and policy frameworks of multiple activities and potential impacts from multiple sources. Most frameworks remain siloed into individual industries or domains and therefore struggle to consider and incorporate the trade-offs that are often necessary when multiple activities are allowed (Turschwell et al., 2020). Moreover, assessments linked to management and regulatory processes are typically constrained to specific activities within a specific jurisdiction, with no account for dispersal of effects beyond that area. Cumulative assessments and management must move beyond these constraints to consider cases where there is spatial separation of the activity and the effect (Stephenson et al 2019). With the increasingly crowded and contested nature of coastal and ocean spaces, the need for effective systems-level planning and management is increasing – as evidenced by the 2020 commitment by the nations participating in the High Level Panel for Ocean Sustainability international initiative, which pledged to have 100% of their national waters under management plans within 5 years (HLPO 2020). Without appropriate assessment methods and information sources, such initiatives will struggle to make evidence-based decisions, with management processes faltering as a result (Vince et al., 2015).

6 Common Steps in the CEA Process

In addressing the many challenges inherent in CEAs, it is useful to identify what has and has not been effective. When considering the general form of the assessments, several common elements exist across the various approaches, especially where the output is intended to advise management or planning processes (e.g. Halpern et al, 2008b; Kappel et al, 2012; Jones et al, 2018; ICES, 2019). In general terms, these elements can be characterised as (United Nations, 2021):

- definition of extent and content (i.e. stressors, spatiotemporal domain and ecosystem components to be included)
- information on the spatial extent (and potentially intensity) of activities or other stressors;
- information (where available) on potential responses by ecosystem components (or some index of the resistance and recovery potential of the ecosystem components); and
- information on mitigation or management measures that might be implemented to reduce the realised extent or magnitude of the stressors.

The Driver-Pressure-State-Impact-Response (DPSIR) framework (Smeets and Weterings, 1999; Elliott et al, 2017) allows these elements to be considered and provides a structure from which quantitative models for estimating effects can be developed. It also provides an easily understandable and widely-recognised conceptual approach which is useful for the purpose of communication with policy makers and other decision makers. That is that drivers (underlying natural and human-caused forces) exert pressures (immediate factors) on the environment that lead to changes in the state of the environment and if great enough impacts on that environment and potentially social-ecological systems more broadly (Elliot et al 2017). To be operational, a CEA should also explicitly include an evaluation of the effectiveness of management measures (Cormier et al, 2018; Stelzenmüller et al, 2018), particularly in, first, quantifying the effects of any management measures on pressures and their resulting impacts and, second, identifying how management measures might be modified to further reduce those pressures and resulting impacts (the reponse). As mentioned above most CEAs to date lack linkages to management measure that might regulate activities and therefore focus on the Pressure – State – Impact elements (typically a scientific and technical exercise). In addition, many do not consider Drivers which requires an understanding of wider socio-economic factors and government/policy mechanisms. As a result, many CEAs provide limited linkages between planning processes and regulatory frameworks that might identify where a precautionary approach might need to be implemented or improvements to management process are needed (ICES 2019). Further, most widely accepted CEA methods consider that the provision of ecosystem services and estimates of socio-cultural effects are outside the remit of a CEA (ICES 2019).

7 A proposed process for cumulative effects assessments

Based on the ideas in previous work (e.g. Jones et al., 2017; Dunstan et al., 2020; United Nations, 2021) we propose the following process to complete a CEA consistent with a full DPSIR approach (steps summarised below, with a schematic given in Figure 8). The steps in the assessment can be completed by different actors since they cross the boundary between science and policy in many instances. For CEA to be truly effective, it requires the collaboration of both science and policy, and could be extended to include other stakeholders (e.g. community). We have identified where potential actors (i.e. either science or policy) can lead or participate in the process.

1. **Scope (Policy leadership with science input)**: Define the scope – both the geographic extent of the assessment, but also what values (ecosystem components, whether ecological, social, economic or cultural) and activities/stressors are to be considered. The scope of the CEA may also consider the desired objectives from the system.

2. **Understanding Pressures (Science leadership with policy input).** Given the scope of the assessment, map the spatio-temporal extent of the stressors in relation to the ecosystem components (Elliott et al., 2020). Identifying a tangible expression of the potential cumulative impacts involves confirmation that the components of the system defined in the scoping phase and the stressors potentially placing pressure on those components (i.e. identifying where overlaps imply realised pressures). This requires that disturbances and activities potentially placing pressures on the marine system in the area of the assessment be identified, and the nature of pressure (e.g. direct, indirect, continuous, pulse) and their spatio-temporal distribution be mapped and/or quantified. Many activities or disturbances concentrated in a small area over a short time can result in pressures or stressors that accumulate due to a crowding effect. An area may be resilient against some level of disturbance, but if that level is exceeded faster than the natural recovery rate, then the disturbance could exceed an ecological or societal threshold for a valued component (Johnson, 2016). Pressures can disperse from the activity area, resulting in a lagged effect on areas outside the immediate footprint of the activity. As a result, the extent, dispersal, frequency and persistence of pressures associated with an activity need to be accounted for when assessing exposure to risk (Borgwardt et al, 2019) both in terms of movement beyond the area in scope but also those external to the area in scope. All potential stressors therefore within and adjacent to the area of the assessment should be considered in order to identify already occurring dispersed risks and potential emerging risks.

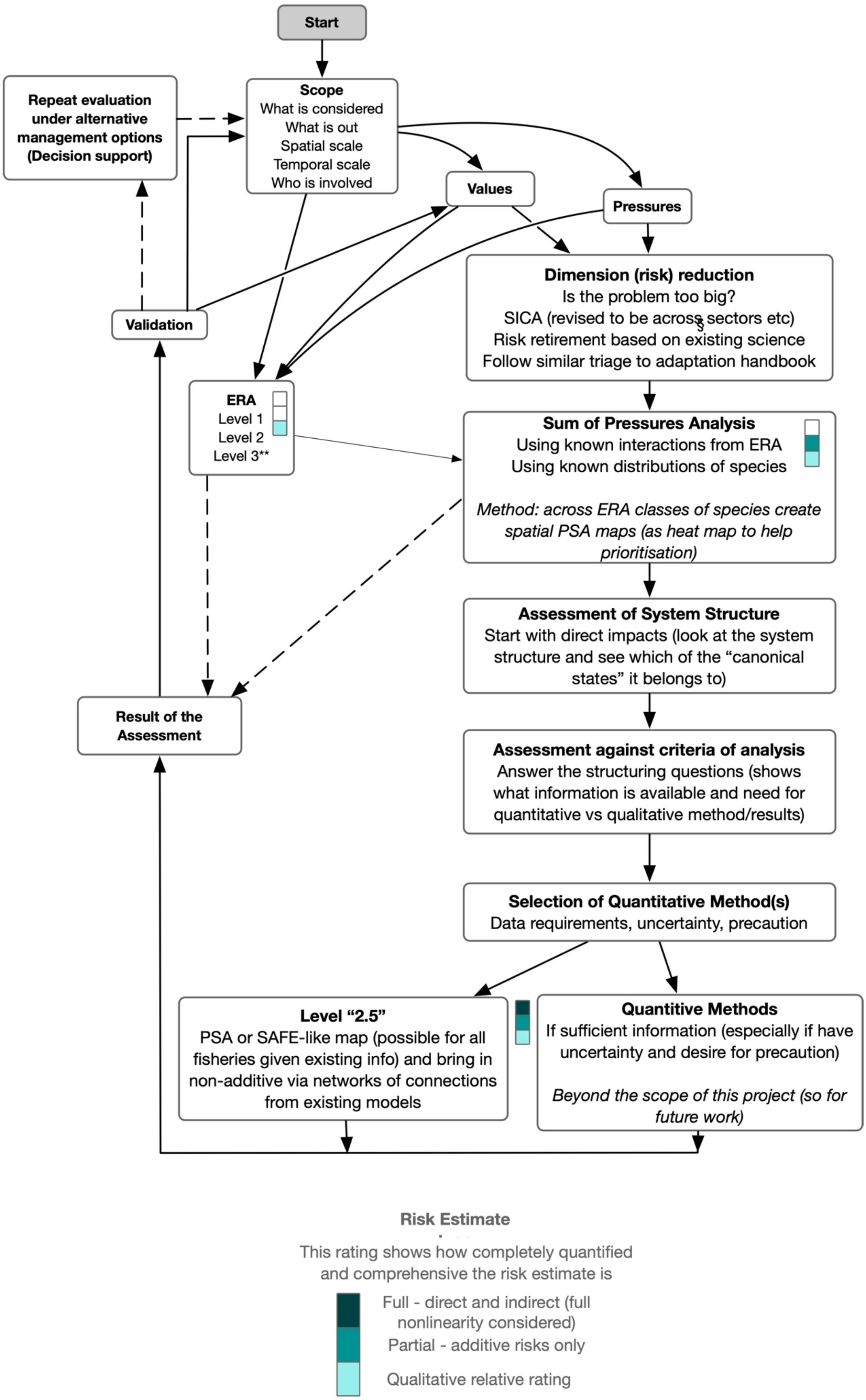


Figure 8: Proposed method - Steps to complete a CEA analysis

3. **Understanding Values (Science leadership with policy input).** Given the values identified in the scoping step, map the spatiotemporal extent of values in the system. The values identified define the key parts of the ecosystem that will be assessed – the impact endpoints. They are often defined in a policy context (e.g. habitat, targeted species, or threatened, endangered and protected species (TEPs)). Where possible the ecosystem(s) that ecological, economic, social and cultural values are associated with should be identified, as this can act as a proxy for the values of interest. Two classes of information support the mapping of values: 1) Direct observations that provide the known spatio-temporal extent of values (*e.g*., scientific surveys, citizen knowledge, historical collections and knowledge); 2) Inferred understanding of the spatio-temporal extent of values (Likely, Potential, Unlikely) based on either expert knowledge or modelling of the distributions from known occurrences. Because of the varying nature of values and stressors and their measurement, the data or information that might be available to contribute to the mapping process, it is unlikely that a single approach will be appropriate in all circumstances. Rather, the approach undertaken should be appropriate for the data available (including its complexity), capture the spatio-temporal components of the data appropriately, and address any uncertainties, biases or assumptions associated with those data.

4. **Priorities (Policy leadership with science input).** An option at this point is to combine the outputs of the pressure and values mapping process through simple overlays or via weights based on the sensitivity of ecosystem components to those stressors to provide a prioritisation map that represents the area of highest overlap. This process will only provide the area of highest overlap between individual values and the direct effects of pressures – it will not be able to quantify indirect effects. As a result, it could be unreliable in identifying endpoints where there is strong nonlinearity in effects on values (as has been found for seagrass in Spencer Gulf, Stockbridge et al 2021). An analysis at this level is particularly useful to prioritise where further work including more detailed analysis should be conducted.

5. **Conceptual Models of system structure (Science leadership with policy input).** Conceptual approaches (e.g. qualitative or quantitative models identifying impact pathways) can then be used to link the values identified and the various potential activities and stressors through the ecosystem in the assessment area (e.g. Dambacher et al, 2009; Anthony et al, 2013). This step elicits how components and processes in the marine environment are related, how natural and anthropogenic pressures can affect the system, and where knowledge gaps and key uncertainties in the system occur. Ideally, consideration of the nature of potential interactions between pressures caused by multiple stressors is included, recognizing that interactions may be nonlinear and that they may be synergistic, antagonistic or masking in nature (see above). Understanding how values and pressures interact might be carried out initially using qualitative models that allow for the identification of the direction, nature and extent of interactions.

6. **Zone of Influence (Science leadership with policy input).** Once the pathways for the effects of pressures on values are understood, the scale of the effect on the value can be

quantified, so that the level of exposure resulting from different stressors are integrated across their individual spatial extents - their “zones of influence” (e.g. Figure 9; Anthony et al, 2013). The definition of this zone of influence can draw from a wide range of sources, including expert elicitation, quantitative estimation and meta-analysis.

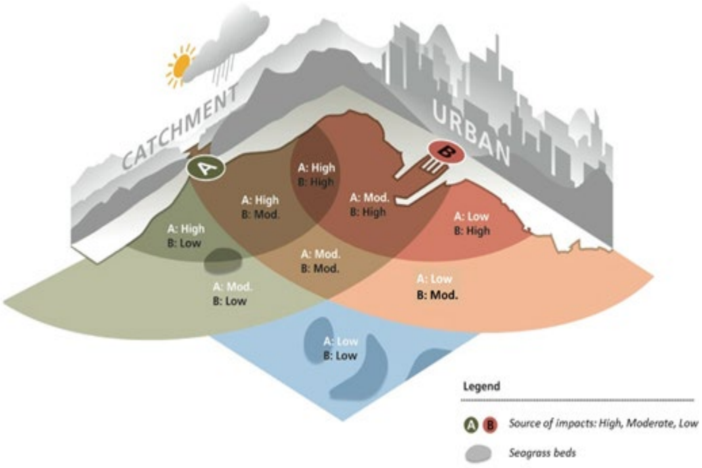


Figure 9: A conceptual model illustrating zones of influence for two examples of point sources: (A) river run-off from catchments and (B) urban or port development. The probabilities of change for each ecosystem value and the amount of ecosystem value potentially impacted (with those probabilities accounting for uncertainties) are calculated within each zone of influence. From Anthony et al (2013).

7. **Assess risk and uncertainty (Science leadership with policy input).** The risk to the value by the pressure and associated uncertainty can then be estimated, while noting that, often, the limited understanding of both the value and the pressures is, in itself, also a source of uncertainty. For example, often the spatial and temporal patterns of pressures are not fully known, nor are the responses of particular values to pressures that might vary through space and time (Stock and Micheli 2016). Identification of sources of uncertainty and their influence on assessment results can be challenging in itself, so appropriate sensitivity analyses that explore the influence of all stressors and their interactions should be carried out (Stock and Micheli 2016). Estimation of risk needs to be capable of capturing the complexity of the system components, the interactions with the activities and the uncertainties associated, and then incorporate the relevant spatial and temporal distributions of any consequences, both positive and negative in nature (e.g. Gregory et al, 2012; Stock and Micheli, 2016). Where information on mitigating measures exists, these maps can be further modified to reflect residual pressure only (i.e. what remains after management actions have been accounted for; Halpern et al, 2008b; Kappel et al, 2012; Jones et al, 2017; ICES, 2019). The complexity of this essential process is dictated by the number of ecosystem components and stressors being considered, as well as the connectivity between all. There are a wide variety of different analytical approaches that can be used in this step (as summarised above in Table 1), all with different strengths and weaknesses. It is difficult to determine exactly which approach will provide the best output without specific understanding of the type of system being assessed and the associated pressures and values. There is no single approach that can deal with all the complexities that might be present within a system. Understanding the assumptions an approach might be making and where it is making those assumptions is important for ensuring a robust

CEA. To assist in the identification of an appropriate approach and the associated assumptions, we have developed a series of questions to help guide the decision (Table 2). These questions will also guide any additional analysis including whether multiple assessments using different approaches (that have different assumptions) might be appropriate.

Table 2: Questions to test assumptions for CEA

Question	Guidance
Can the method estimate the spatial distribution of cumulative effects?	If not, then if the expected spatial distribution of impacts is large then additional analysis may be necessary to identify all stressors and their potential effects.
Can the method identify alterations to ecosystem components and processes such as nutrient cycling, predation, habitat modification, sedimentation, light penetration?	If not, then the absence of understanding of key processes may mean that ecosystem responses are not well characterised.
Does the method imply the link between multiple pressures and values or is this explicitly described in the approach?	If implied, then additional information will be necessary to ensure that the pressures cause a change in the values
Can the proposed methods assess the indirect effects caused by the pressures on values?	If not, caution must be taken to ensure indirect effects (mediated through the system being assessed) that may change the magnitude and direction of change in values are accounted for.
Can the method assess facilitative effects (where one effect makes another effect more likely) of multiple pressures on values?	If not, caution will need to be taken to ensure that pressures that facilitate effects on values from other pressures are accounted for.
Can the method distinguish between masking, antagonistic, additive and synergistic links between multiple pressures and values?	If not, the full effects of pressures on values may not be properly estimated.
Are non-linear links between pressures and system components able to be identified?	If not, the inflection points and transitions in effects may not be well estimated.
Can the method distinguish between the effects of a single pressure acting sequentially?	If not, the assessment may not capture the full effects of pressures acting on a value through time.
Can the method distinguish between the effects of multiple pressures acting simultaneously or sequentially?	If not, the assessment may not capture the full effects of pressures acting on values through space and time.
Can the method estimate emerging or future effects on values?	If not, it will not be possible to estimate any future risks on values by pressures
Can the method produce an estimate of uncertainty in the likelihood and consequence of effects?	If not, additional caution is necessary as the estimate of risk may be not be accurate.
Can the method incorporate temporal variation and time lags?	If not, the assessment may not capture the full effects of pressures acting on values through time.

8. **Validation (Policy leadership with science input).** Finally, where possible, the networks of interactions, maps of risk and cumulative effects should be empirically tested (though, in practice, this has occurred relatively rarely; see Halpern and Fujita (2013)). In order to facilitate such validation, risk assessments need to be reported in such a way that they can be testable; that is, measured and mapped in the environment. This requires that a formal framework for monitoring the system is developed including the identification of priorities for monitoring. The monitoring framework should ensure that the priority values and pressures are monitored and are linked to the desired objectives of the system. While rarely undertaken, such validation is important for verifying the efficacy of assumptions used to simplify the CEA process; if those assumptions are not met there is no guarantee the method is representing the system of interest (for example, mismatches between CEA estimates and real-world outcomes was found for seagrass by Stockbridge et al, 2021).

9. **Evaluation of management options/performance (Science leadership with policy input)**
   This may be considered an optional step in one sense, as it is not absolutely needed for completing an assessment. However, in taking the outcome of a (validated) assessment and transforming that into meaningful management decisions, it is useful to understand how the outcomes of the assessment might change under alternative management options. Such as management evaluation step is well known in some resource management sectors (e.g. fisheries; Smith 1994; Sainsbury et al., 2000), but is not widely applied within the context of cumulative effects as yet. Taking such an approach would, however, open up new decision support options at strategic scales, including the application of adaptive management, which has been shown to be an effective means of dealing with uncertainty and injecting precaution into complex systems (Benidickson et al., 2005). Links to management, especially adaptive management, will likely also become easier once repeat assessments begin to occur (e.g. Halpern et al 2015, 2017) and insights can be drawn on changes in cumulative effects through time (i.e. trends). The impact on management will be greatest, when there are not only status and trends considered by the assessments, but also when there are traceable connections between effects and relevant human activities, both locally and distally (Kopinen and Anderson, 2016)**.**

   Estimating the economic value of benefits and costs (in monetary terms) generated by alternative management options that reduce cumulative effects on ecosystems relative to a counterfactual enables a welfare analysis typically conducted through a social cost-benefit analysis (CBA). A social CBA provides information on (1) whether any alternative management option generates a social net benefit (gain in social welfare) or a social net cost (loss of social welfare) compared to a counterfactual, (2) the relative welfare gain/ loss of alternative management options, and (3) the distribution of benefits and costs across different stakeholders ('winners and losers') (e.g., Boardman et al. 2018). A rigorous Cost-Benefit Analysis would include the economic value of benefits that are generated by ecosystem use (e.g., fishing profits; recreational fishing benefits) as well as that of non-use benefits (e.g., the knowledge of the continued existence of a TEP species enjoyed by the general public) (e.g., Hanley and Barbier 2009). The value of non-use benefits can be estimated using well-established non-market valuation methods (e.g., benefit transfer, discrete choice experiment, travel cost approaches) (e.g. Bateman et al., 2002; Hensher et al., 2015; Pearsons 2017).

8 Discussion

The global shift toward ecosystem based and integrated approaches (e.g. Stephenson et al 2021) means that taking a system perspective has a much more central role in resource and environmental management. Australia's growing marine industries and population, and its struggles with deteriorating ecosystem status (Samuel, 2020), means it is increasingly widely acknowledged that the multitude of interacting pressures is constraining Australia's collective capacity to achieve its management and societal objectives (Smith et al., 2017). The understanding provided by CEAs with respect to delineating the magnitude of the problem and the identification of potential mitigation measures (Levin et al., 2009) is a key part of addressing the challenges facing sustainable industries and conservation.

This dual need for understanding and a systemic management perspective has been a key motivator behind the clear increase in the number of CEAs conducted over the last couple of decades, with applications in regional marine assessments, planning and regulatory processes (Halpern et al, 2015; ICES, 2019). However, despite their increased use, CEAs are largely lacking at both national and regional scales from areas outside Europe and North America (Korpinen and Andersen, 2016; United Nations, 2021). This likely reflects the capacity to undertake assessments (both in terms of capability and information available for assessments). It could also reflect different approaches to development and strategic planning in different locations driven by differences in policy frameworks – for example contrast the European approach, based on collaborative identification of indicators and assessment process in support of the Marine Strategy Framework Directive, with the more directed statements of the most recent five-year plan in China, which has accelerated transitions in planning perspectives in that country.

The availability of data for use in assessments is considered a universal challenge in all jurisdictions, despite ever increasing instrumentation and broader monitoring of the marine environment by many nations. This is because the challenge is not simply about data collected, but also about data being made (i) publicly accessible, (ii) searchable, and (iii) provided in useable formats (i.e. the degree to which FAIR principles have been enacted; Wilkinson et al., 2016). There are collective efforts to address some of these shortcomings (e.g. movement of government agencies to make their data collections available), but this takes time and resources, meaning it is rolled out heterogeneously at best. Moreover, even with current and near future collection technologies it will be hard to provide detailed data about all aspects of marine and coastal socioecological systems. This is why it has been considered for a long time that "qualitative approaches may be the only practical means to overcome the problems of complexity and data deficiencies and provide some insight into the nature and magnitude of cumulative risks [effects]" (Callahan and Sexton, 2007). Despite limitations in data, in the last 10-15 years assessments including quantitative approaches have become more commonly used.

Of those CEAs that have been undertaken over the past two decades, over half took quite similar/identical methodological approaches (based on the method of Halpern et al 2008). These GIS based approaches bring together the spatiotemporal footprint of exposure (e.g. pattern of intensity of usage by a sector or physical environmental properties) and the distributions of values to create vulnerability maps. By adjusting these maps – based on sensitivity of the value to that pressure, or management arrangements – residual vulnerability per pressure and then a measure of the anticipated cumulative effect of those pressures can be generated (Halpern et al 2008a; Kappel et al 2012; ICES, 2019).

The maps generated by spatial analyses resonate with planners and other groups interested in marine and coastal issues, as they find such maps intuitively easy to interpret. Unfortunately, it is less clear whether the interpretation of results on the basis of maps produced by some methods are actually straightforward. O'Hara et al (2021), found that the final "risk" hotspots as defined by maps were heavily conditioned by the available data on vulnerable species, rather than the threat layers. This meant that such outputs would not be ideal for forward looking planning processes. Such overlay-based approaches also currently struggle to move beyond additive layering of the pressures and system values of interest (Clarke Murray et al 2014). European management agencies have been trying to address these weaknesses by using an emerging framework that combines the conceptual structuring of cause-effect pathways with a quantitative assessment of effects (Cormier et al, 2018). This approach puts an explicit focus on (i) linkages and (ii) the effectiveness of management measures in reducing human pressures and the resulting prevailing cumulative pressure load on distinct ecosystem values (United Nations, 2021).

Non-linear responses – either in the form of connections between system components or their responses are not easily dealt with by many existing methods (Halpern and Fujita, 2013; Clarke Murray et al 2014; Stelzenmüller et al., 2018; United Nations 2021). Currently non-linearity is most commonly dealt with via some form of (often process-oriented) modelling. Qualitative models in the form of signed diagraphs are one of the least logistically intensive. Despite being considered as qualitative, these methods are actually semi-quantitative, drawing on information from diverse data sources and knowledge types to construct a network of relationships between key system components (Dambacher et al., 2003) with the outcomes of perturbations identified using signed matrix algebra. Bayesian networks, which can also be built from a diverse of information types (either directly or via leveraging qualitative models), can be used to explore probabilistic outcomes of perturbations and cascading effects (Hosack et al., 2008; Anthony et al., 2013). Where sufficient quantitative data is available, additional methods, such as statistical approaches that use well understood relationships between variables to explore combined pressures, changing distributions or zones of influence are appropriate (Anthony et al., 2013). Process models applied across entire life cycles or socioecological systems can also be appropriate (e.g. Fulrton et al., 2017). Common features of these tools are that they can (i) encompass multiple stressors, scales and their interconnections; and (ii) have the capacity to express multiple potential endpoints or system structures. Learning from assessments done in the context of ecosystem-based fisheries assessments or climate adaptation, it is clear that using a range of approaches also provides a means of handling model structural uncertainty, a key but often unacknowledged source of uncertainty (Pethybridge et al., 2020).

All methods are currently challenged by the incorporation and propagation of uncertainty with relatively few published CEAs addressing major underlying uncertainties (Halpern and Fujita, 2013). Further, few studies currently validate their assessment outputs or consider temporal components to human activities (many of those reviewed assumed that activities were long lasting and overlapped in time), such as accounting for historical impacts that have already modified the marine environment, or presenting trends. As previously outlined, few consider management decisions and how assessment outcomes might change under different management arrangements (as recommended for assessments intended to inform ecosystem-based management; Holsmann et al 2017). Marrying CEAs with management strategy evaluation (MSE) is one way that this weakness can be addressed – and thus then recommended step 9 in the proposed CEA approach described above. MSE involves modelling each part of the adaptive management system – the natural world, human users, monitoring, assessment and decision-making processes. In this way the robustness of the management system can be assessed against all possible sources of variation, error and uncertainty (Smith, 1994; Sainsbury et al., 2000; Fulton et al., 2011).

The few applications of CEA (or CEA-like) approaches within Australia (as summarised in the short history presented above in the introduction) highlight why these methodological gaps are concerning and need addressing. The work done in NSW (MEMA, 2018) and Victoria (DELWP, 2020) in particular highlight what can be achieved via taking a staged approach. While work both on the Great Barrier Reef (Dunstan et al., 2020) and in Spencer Gulf (Stockbridge et al 2021) resulted in the recognition that validation (or at the very least experimental or field confirmation of response functions) is critical for well-informed decision making. These studies also highlighted that non-linear changes are common and dynamic, with ecological thresholds and responses (especially to multiple pressures) likely to change over ecologically and management relevant time frames - both through acclimation, which can ameliorate effects, or interactions that can amplify responses (United Nations, 2021). A final key insight from these Australian case studies – and the broader review of assessment approaches undertaken here – is that an assessment must be contextually based (i.e. must be tailored to the scope and characteristics of the system of interest). This is because the system structure and management needs both informs relevant scales, values for inclusion and other dimensions of the assessment, which ultimately dictate the complexity of a CEA.

8.1 Linking CEA to Systems of Environmental Economic Accounts

Linking CEA to broader system considerations means it can complement other management and reporting processes. For exampling linking CEA to economic indicators – market or non-market – goes beyond simple performance metrics (Step 9 above); it can be linked into the relatively new domain of ecosystem accounting. The UN-System of Environmental Economic Accounting-Ecosystem Accounting (Un-SEEA-EA) is a framework that facilitates an integration of ecosystem and economic data, highlighting the relationships between ecosystem assets, the ecosystem services flow they generate, and the resulting benefits enjoyed by humans (UNCEEA 2021). Data is organized by means of five accounts: (1) ecosystem extent account (physical terms), (2) ecosystem condition account (physical terms), (3) ecosystem services flow account (physical terms), (4) ecosystem services flow account (monetary terms), and (5) ecosystem asset account (monetary account). This accounting scheme is becoming an expected part of national accounts and is likely to be a standard part of international reporting in the future.

Information generated by a CEA could be integrated into the ecosystem condition and physical ecosystem services flow accounts, and therefore implicitly into the monetary ecosystem services flow and ecosystem asset accounts. The ecosystem condition account could be expanded by integrating data on multiple stressors, cumulative impacts, and the results of the risk assessment. Since these accounts are updated for every accounting period and hence the record changes over time, they also capture effects on ecosystem assets and services accumulated through time and facilitate the monitoring of temporal changes in stressors and risk. The monetary ecosystem accounts apply the concept of exchange value, which allows for estimation of the contribution of ecosystem assets to measures of economic activity (e.g., value-added). However, since exchange value excludes consumer surplus, using monetary ecosystem services and asset accounts to inform decisions on management changes may result in a net loss to society. Assessing the impact of management changes on social welfare (typically through a cost-benefit analysis) requires the application of welfare values, which - by definition - include both producer and consumer surplus. Hence, ecosystem accounting should be used jointly with other approaches and tools, such as CEA, for application in management decisions.

9 Conclusions

This report takes the lessons drawn from reviewing how CEAs have been conducted and has used them to formulate the key functional steps a CEA must contain in order to deliver an approach tailored to Australian needs, one which can address the non-linear nature of cumulative effects likely in the fished ecosystems along Australia's coastlines. The process begins with a firm foundation regarding the scale at which the assessment is being conducted, the values and specific management objectives being addressed and the data available for undertaking a CEA. These steps shape the format of the assessment outputs produced and their suitability for informing planning and management.

For CEA outputs to be of most utility, however, assessments need to (i) be verified observationally; and (ii) incorporate the extent, spatial and temporal variability in data and associated uncertainty. This is because most CEAs lack experimentally or observation-based relationships between the magnitude of a stressor and the ecosystem components, so the veracity of outputs need to be verified and uncertainty transparently presented. This is required not only to ensure the CEA outputs are robust, but also to highlight knowledge gaps and prioritise future data collection and research efforts that can improve assessments by reducing uncertainties. Ideally, in this way, assessment and associated data collection can explicitly describe the causes and consequences of deleterious effects and elucidate causal pathways of risk (Nicol et al, 2019; United Nations, 2021). Moreover, the knowledge gaps and hypotheses generated during the assessments should feedback and inform future observational regimes, in turn progressively informing future assessments (and so on so that the two processes strengthen one another; Addison et al., 2018; Newman et al., 2019).

For a nation with Australia's geographic extent, relatively limited population density and budgetary constraints, any CEA method applied would need to: (i) make best use of available data; (ii) be able to incorporate a broad range of data sources (including First Nation's knowledge and citizen science); (iii) be readily updateable as new information becomes available or new stressors arise; (iv) provide outputs that can directly interface with management decision-making processes. In addition, the method should be easily implementable (both in terms of skills and time to make sure it sees repeated use. These insights and criteria led us to propose a CEA process, founded on a DPSIR framework, that can be completed collaboratively by different actors (across science, industry and policy (Figure 8). Application of this approach should put jurisdictions, such as Australia, in a good position to deliver CEA across broad spatial and temporal extents, across different ecosystem uses and components and in such a way it can directly inform management decision making.

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