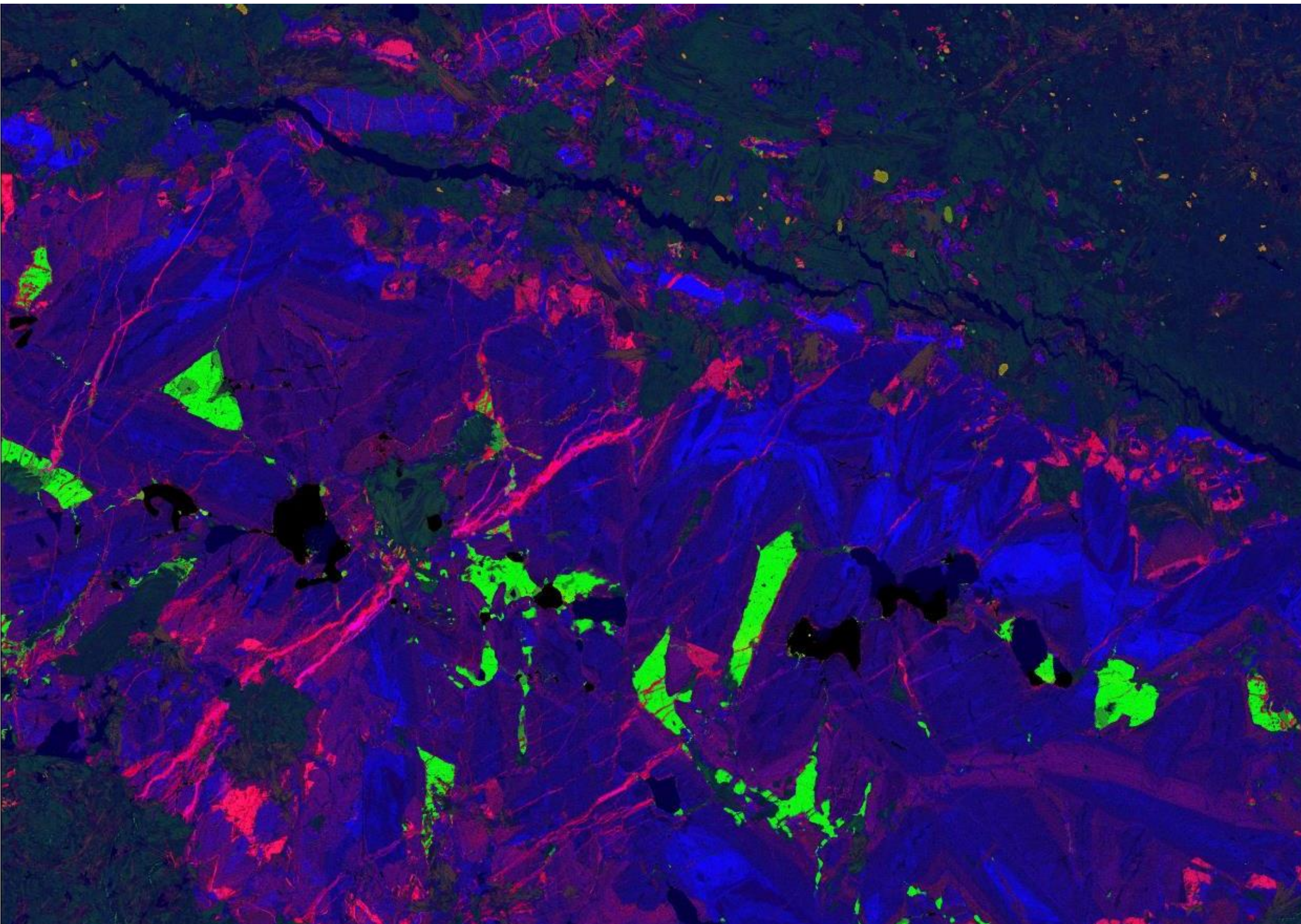
Ecosystem-Based Management Framework for Polymetallic Nodule Collection Activities

Project: Integrated Ecosystem Assessment and Ecosystem-Based Management Framework for Polymetallic Nodule Collection Activities

June 2025

Dambacher JM, Dunstan PK, Fulton EA, Clark MR, Hosack GR, Hyman J, Leduc D, O'Hara TD, Parr JM, Rowden AA, Schlacher TA, Stewart RA



Prepared by the CSIRO Research Consortium for The Metals Company Australia Pty Ltd.

The 2021–2025 consortium, led by CSIRO, includes Museums Victoria, Griffith University, University of the Sunshine Coast and NIWA.

Citation

Dambacher JM, Dunstan PK, Fulton EA, Clark MR, Hosack GR, Hyman J, Leduc D, O'Hara TD, Parr JM, Rowden AA, Schlacher TA, Stewart RA. 2025. *Ecosystem-Based Management Framework for Polymetallic Nodule Collection Activities*. CSIRO Marine Laboratories, Hobart.

Copyright

Important disclaimer

Acknowledgement of Country

Acknowledgements

The analysis and writing of this report was funded by The Metals Company Australia Pty Ltd through a contract to a research consortium that includes the Commonwealth Scientific and Industrial Research Organisation (CSIRO, Australia), National Institute of Water and Atmospheric Research (NIWA, New Zealand), Griffith University, University of the Sunshine Coast, and Museums Victoria (Australia).

Contents

Figures

1 Introduction

Ecosystem-based management is an interdisciplinary approach that balances ecological, social and governance principles at appropriate temporal and spatial scales in a distinct geographical area to achieve sustainable resource use. Scientific knowledge and effective monitoring are used to acknowledge the connections, integrity and biodiversity within an ecosystem along with its dynamic nature and associated uncertainties. EBM recognizes coupled social-ecological systems with stakeholders involved in an integrated and adaptive management process where decisions reflect societal choice.

Long *et al.* (2015)

Ecosystem-based management (EBM) seeks the sustainable use of natural resources through an integration of the physical and biological components of an ecological system with the activities and well-being of humans – that is by taking a systemic social-ecological (or socioecological) perspective. It is an adaptive and evidence-based approach, the operationalization of which depends on a scientific understanding drawn from processes such as integrated ecosystem assessments (IEAs) of relevant ecological and socio-economic indicators in relation to specific management objectives (Levin *et al.* 2009). In marine environments, applications of EBM have come from an appreciation that a systemic view is needed if the many drivers and interconnected parts of the system are to be recognised and accounted for, and for any associated management trade-offs to be transparent. The desire for truly systemic and cross-sector management was one of the drivers for the development of IEAs and more comprehensive marine spatial planning; however, this form of management is still becoming established and is motivated by different mixes of drivers in different locations[1]. Individual sectors, such as fisheries, have made greater advances in taking an ecosystem-based perspective. Looking at fisheries is informative in understanding nuances in the development of the concept. Looking “down” from the entire system to the activity of interest, fishing, is ecosystem-based fisheries management (EBFM), looking “out” from a single activity to the ecosystem is an ecosystem approach to fisheries management (EAFM). EAFM is by far the more commonly used approach as it has developed and expanded out of a need to broaden the focus and scope of fisheries management from a single commercial fish species to that of the entire ecosystem and its attendant human socio-economic system, with current innovations and challenges considering the management of activities and trade-offs posed by the use of multiple competing gears (*e.g.,* nets vs trawl), kinds of operator (small versus large, commercial vs subsistence. This sector-specific view becomes more challenging as ocean spaces become more crowding, which is why management is now attempting to look across multiple, and sometimes competing, resource- and use-sectors – *i.e.,* aquaculture, coastal development, cultural, energy (oil and gas, marine power and wind), fisheries, seabed mining, recreation and shipping (Link and Browman 2017, DePiper *et al.* 2017, Smith *et al.* 2021).

In some respects, the application of EBM to deep seabed mining in the Clarion Clipperton Zone could be described simply as an extension of a well-established approach, as there are few other resource- or use-sectors to consider (albeit potential impacts to fisheries and cultural uses for Pacific Island Nations are being considered) and proposed methods to collect polymetallic nodules will be limited to a one-time pass of a collector vehicle over the seabed surface. There are, however, novel and significant challenges. Benthic ecosystems of the NORI-D lease are remote, deep and difficult to observe, nutrient inputs are extremely low and recovery times can be exceptionally long for some impacted components of the benthos (Jones *et al.* 2017). As nodules form over millions of years, they are effectively a non-renewable resource, and the key EBM principle of sustainability (Long *et al.* 2015, Kirkfeldt 2019) applies not to that standard fishery

[1] This was the conclusion of a 2022 workshop on EBM that drew on examples from around the globe, the results of which are the subject of a paper in review: Haugen *et al*. Marine ecosystem-based management: challenges remain, yet solutions exist, and progress is occurring.

goal of maintaining a sustained yield (Long *et al.* 2017) but rather to that of conserving biodiversity through preserving essential or representative habitats (Vierros 2008). Moreover, as the first-ever commercial-scale operation of seabed mining that is being considered for approval by the International Seabed Authority, regulatory procedures and protocols are still being developed or remain untested, with no track record or experience from which to chart or guide management objectives, decisions or actions. In this context there is a need for an EBM framework that is tailored to a socioecological system that is both data- and experience-poor and, accordingly, be particularly structured to be robust to high levels of uncertainty, be incentivised to fill key knowledge gaps, and be rapidly adaptive as system understanding and regulatory circumstances evolve.

2 EBM and the scientific foundation of IEAs

A sound scientific foundation for systemic EBM is IEAs, which formally implement EBM goals and objectives through a stepwise process that engages stakeholders, scientists, and managers. IEAs explicitly addresses cumulative impacts, uncertainty, and management trade-offs, and are an adaptive process that encourages learning-by-doing by monitoring key system indicators and using observed responses to test system understanding and evaluate management effectiveness (Fig. 1). IEAs include the eight steps of engagement, scoping, indicator development, ecosystem assessment, risk assessment, uncertainty assessment, evaluation of management options, and monitoring and evaluation (Levin *et al.* 2009, Smith *et al.* 2021). The final step, monitoring and evaluation, feeds back into the first four steps, thus creating an iterative process (Fig. 1).

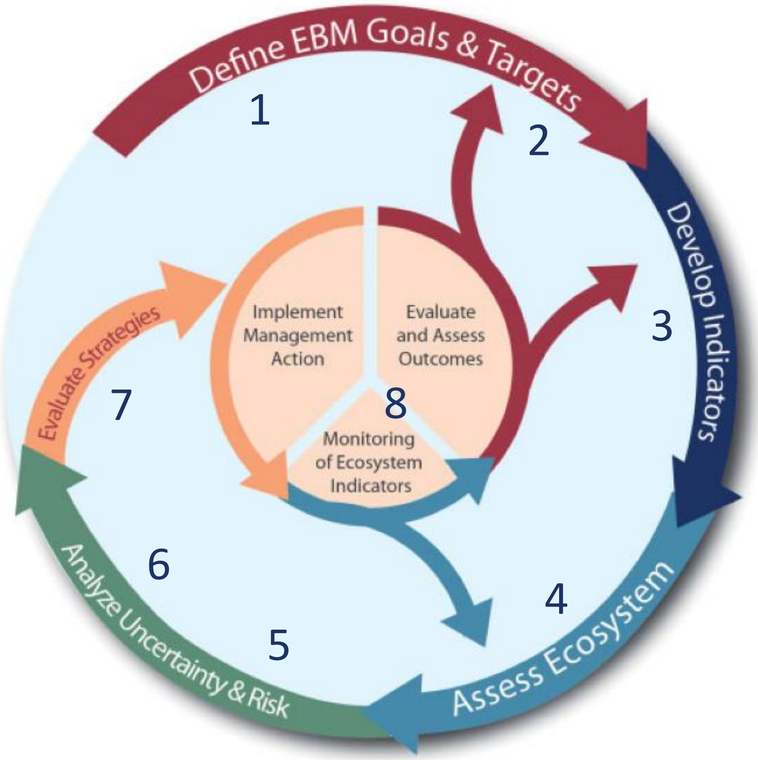


Figure 1. Conceptual diagram of Integrated Ecosystem Assessments (IEAs) that provide science to support Ecosystem-Based Management (EBM); and includes the eight steps of 1) engagement, 2) scoping, 3) indicator development, 4) ecosystem assessment, 5) risk assessment, 6) uncertainty assessment, 7) evaluation of management options and strategies, and 8) monitoring and evaluation; actual implementation of management actions occurs between steps 7 and 8 (Smith *et al.* 2021, adapted from NOAA's Integrated Ecosystem Assessment).

3 The Eight Steps of IEAs

3.1 Engagement

The purpose of this first step of an IEA is to initiate a meaningful dialogue between a triad of stakeholders, scientists and decision makers (Fig. 2) so that the values, knowledge, hopes and fears of concerned or potentially impacted communities are reflected in tangible management goals and targets, and to transparently and legitimately determine by what basis and to what extent these goals and targets can be judged to have been met or not (Röckmann *et al.* 2015). Smith *et al.* (2021) contend that engagement should be open and ongoing throughout the entire IEA process, with different nuanced approaches being required to reach across a spectrum of communities, stakeholders, and publics.

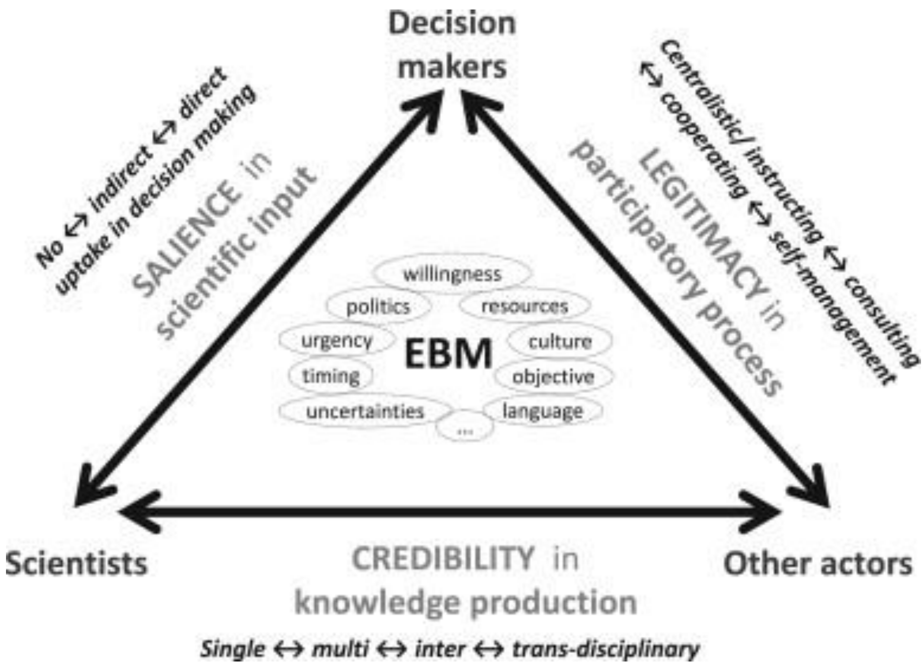


Figure 2. Engagement triad of Ecosystem-Based Management (from Röckmann et al. 2015).

Colvin *et al.* (2016) recognizes stakeholders based on their interests in, or physical proximity to, a project (Fig. 3) and describes eight different approaches that practitioners typically use to distinguish stakeholders from citizenry. Two are based on the practitioners themselves identifying stakeholders either through key informants and snowballing, or through the use of media. In five other approaches stakeholders are identified through selection criteria that include geographical footprint, interests, influence, intuition, and past experiences, with an eighth approach involving stakeholder self-selection. Each approach has its own strengths and potential pitfalls with long-term implications for trust building and conflict resolution. Thus, in awarding stakeholder status it is advised that a mix of approaches be used (*e.g.,* Forrester *et al.* 2015).

The process of identifying scientists to engage in the IEA process shares some of the basic elements for identifying stakeholder status, with a researcher's domain of interest and their geographic area of experience being essential criteria. As with the stakeholder identification, there are several approaches available for selecting scientists. Where the science domain associated with the EBM issue is well developed then it may be possible to choose key informants from a discrete and well-known set. But where a discrete set of key informants does not exist or is unknown, we suggest proceeding by either of two approaches that are based on the degree or ease to which individual scientists can be identified.

One approach takes advantage of scientists that regularly publish their research findings in peer-reviewed journals. Bibliographic details of these publications are readily retrieved and collated through internet literature searches (*e.g.,* PubMed, Scopus, Web of Science). Such searches can target domain-specific search terms and provide a well-defined set of scientists that can be ranked according to publication statistics such as number of peer-reviewed articles produced within a recent time window or citation indices (CSIRO unpublished methodology).

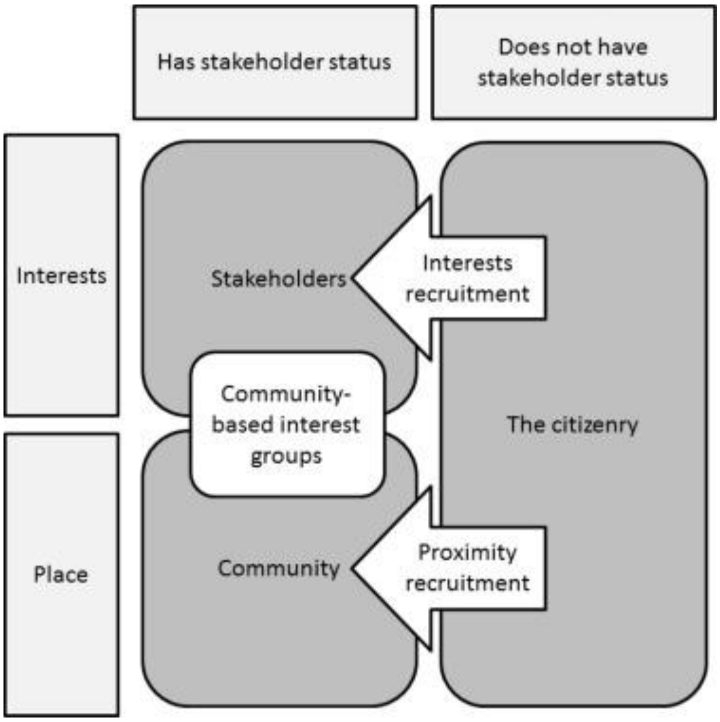


Figure 3. Identification of stakeholder status through interests in a project or proximity to its affected area (from Colvin et al 2016).

Another approach pertains to a population of scientists where some are identifiable and easy to reach, but the majority are not. In this situation their domain knowledge and experience, while highly valued, may not be sufficiently represented by a publication record. Here, snowball or respondent-driven sampling (Goodman 1961, 2011) can be an effective approach. It presupposes a network of professional relationships among scientists through which the total population can be reached by chain-referral sampling (Bagheri and Saadati 2015). A seed population is first contacted and asked if they would be willing to engage and also to identify other scientists they know with relevant domain knowledge and experience. This second set is given the same questions, with repetition leading to multiple waves of referrals. Iterations are continued until new recruits diminish, or a target size is reached with sufficient diversity in the science domains represented.

In either instance, to be relevant to the question at hand, it his highly advisable to draw on a diverse population of scientists that spans multiple backgrounds and disciplines. Maximising the utility of expert knowledge and resulting predictive skills underlines the benefits of having a diverse base; the more systemic the perspective the more disciplines required (*e.g.,* the work summarised in Tetlock and Gardner 2015).

3.2 Scoping

The purpose of this step is to determine the appropriate spatial and temporal scale of the assessment, describe valued components and services of the ecological system as well as potential pressures and threats from proposed activities and specify core management objectives (Levin *et al.* 2009). All participants in the engagement triad will come to the table with their own mental model of the way the socioecological system works and what may or may not be important, and through the engagement process they are tasked with developing a shared understanding of the system being assessed and to what end it should be managed. This shared understanding is best developed and documented through conceptual models.

3.2.1 Conceptual models

The primary goal of conceptual models is to explicitly record mental models so that different world views are made clear and can be evaluated, refined and communicated to technical and non-technical audiences (Maddox *et al.* 1999). The role of a conceptual model is to summarize the causal interactions of key ecosystem components and processes, the spatial and temporal scales at which they operate, and existing or potential stressors and pressures to the system. They also serve the purpose of identifying informative indicators and providing a causal narrative to guide and structure formal analysis of monitoring data.

Conceptual models come in many different forms including simple narrative descriptions, schematic diagrams, box-and-arrow flowcharts, or even cartoons that pictorially illustrate physical and biological processes and the effects of anthropogenic pressures. Even though there are many forms of conceptual models, they all hold common elements and can be constructed using a common set of steps. Gross (2003) systematically describes the main tasks required in constructing conceptual models.

1. Clearly state the goals of the conceptual models.
2. Identify bounds of the system of interest.
3. Identify key model components, subsystems, and interactions.
4. Develop control models of key systems and subsystems.
5. Identify natural and anthropogenic stressors.
6. Describe relationships of stressors, ecological factors, and responses.
7. Articulate key questions or alternative hypotheses and approaches.

N.B.: Gross (2003) makes the useful distinction between a control model and stressor model. Control models represent the key components, processes, and causal feedbacks of the system without inclusion of stressors or pressures. Stressor models, which can be based on the control model, are designed to illustrate, or predict, the cumulative impact of stressors and pressures on the system leading to changes in key ecological responses or indicators.

While an explicitly stated conceptual model is a summary of current understanding of, and assumptions about, an ecosystem, it is important to recognise that it does not represent ‘the truth’; is not final or unmodifiable; nor is it expected to be complete or include the entire ecosystem. Rather, it is intended to be a flexible construct that should evolve as understanding of the ecosystem increases. If an EBM program is successful, understanding and knowledge of the system will increase with time and necessitate model revision and refinement (*e.g.,* Cloern 2001, Groffman *et al.* 2004). Indeed, if an active adaptive approach is taken, then management needs to be specifically structured to generate new learning with the intention of restructuring management based on that learning (Walters and Holling 1990).

Dunstan *et al.* (2019) provide a checklist of considerations for using conceptual models in IEAs:

Questions	**Caveats**
Is the context of the conceptual model clearly defined?	
Does the conceptual model of the system capture the same temporal and spatial scales as desired for the assessment/of interest?	If no, caution needs to be taken to ensure that the spatial and temporal scale are appropriate to enable estimation of impact
Are the spatial and temporal limits of the system clearly identified?	If no, additional consideration should be given to defining the limits to ensure that the model captures the relevant parts of the system for management
Does the conceptual model include ecosystem components that adequately represent key species, habitats and processes (*i.e.,* resource flows, ecological relationships, and disturbance regimes)?	If no, potential ecosystem impacts from pressures may not be well described
Can you actually measure the outputs of the system, identify indicators and monitor the outcomes	
Does the conceptual model describe how the pressures, values and ecosystem components relate to each other and interact?	If no, potential ecosystem impacts from pressures may not be well described
Are the assessment endpoints (the ecosystem components that will be monitored) represented in the conceptual model?	if no, the direct impacts of pressures on ecosystem components they impact are not well described
Are there alternative ways that pressures could impact values or alternatives for how the ecosystem might be structured?	if yes, then each different conceptual model should be considered in the assessment

3.3 Indicator development

Proceeding directly from the development of conceptual models comes the critical step of developing indicators of the state of ecological and socio-economic system. Ideally, indicators should be informative with respect to how threats or pressures will impact valued components and processes of the system (Hayes *et al.* 2015). In this respect they should be linked to management objectives, respond to ecosystem changes in a readily interpretable and predictable manner, be easy to measure and have agreed responses when operational or management thresholds are crossed. The process of identifying, classifying and organizing informative indicators is based on the Driver-Pressure-State-Impact-Response (DPSIR) framework (Fig. 4). It is widely used as an overarching structure to organize and categorise different types of indicators for the integrated monitoring and management of complex ecological and socio-economic systems. This schema distinguishes distinct indicators for each of the five elements of the framework as well as for each of the four types of management responses.

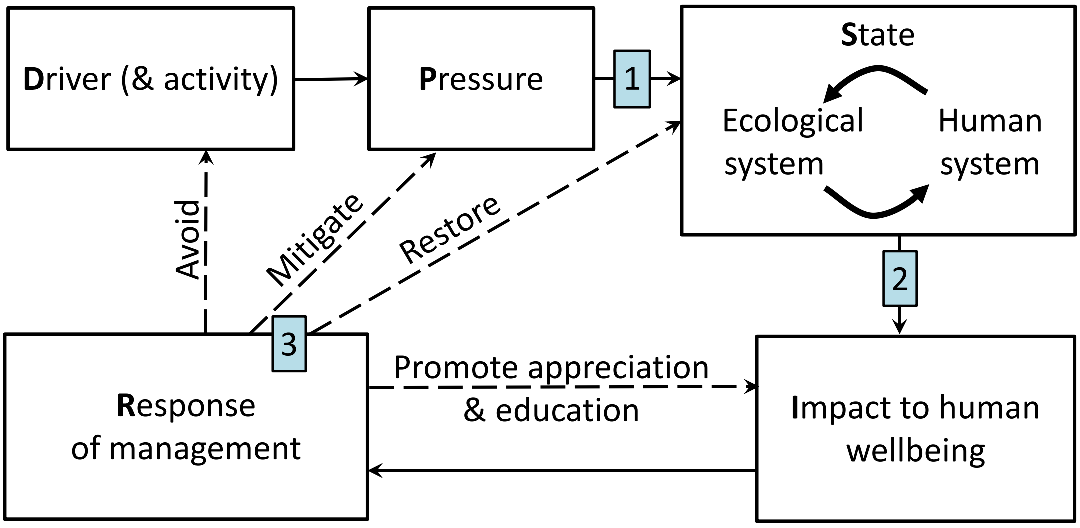


Figure 4. DPSIR framework for identifying and classifying indicators for integrated monitoring and management of ecological and socio-economic systems; key challenges to effective implementation include understanding and delineation of 1) pressure-state interactions, 2) state-impact interactions and 3) management response interactions (*i.e.,* avoid, mitigate, restore and promote appreciation and education).

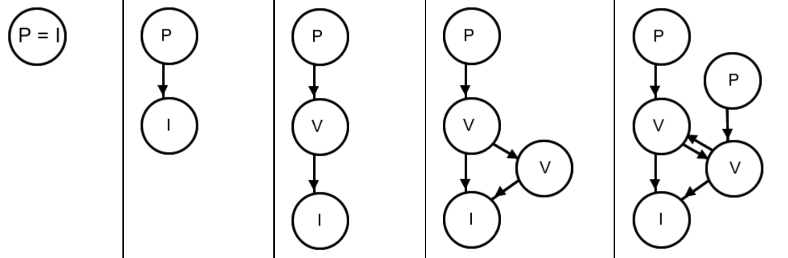
We identify three main challenges to the effective implementation of the DPSIR framework; all are problems of attribution. Firstly, the framework assumes that a system will respond predictably to a given pressure. It does not, however, specify how to deal with uncertainties in attributing observed or predicted effects to particular causes, and this challenge is especially difficult in complex systems when there are multiple stressors and pressures. Poorly defined causal attribution makes it very difficult to select informative indicators that need to measure the response of complex ecological systems reliably and consistently. Secondly, understanding impacts to valued components and processes of the system—described in DPSIR framework as human wellbeing—requires the assignment and mapping of human values to measurable system components or services. In many instances, however, such links are poorly known or articulated. Thirdly, the framework requires the linking of predefined management objectives and criteria to four possible types of management response (*i.e.,* avoid, mitigate, restore, and educate or promote). The challenge here is to identify informative indicators that measure whether management responses are effective. It is often the case, however, especially for a nascent activity such as deep seabed mining, that management or regulatory objectives have yet to be clearly stated or codified.

3.3.1 Meeting DPSIR challenges

Causal attributions for change in state

To meet the challenge of causal attribution we recommend using an array of modelling approaches (conceptual, illustrative, or mathematical) to understand and predict how the state of the ecological and human system will respond to pressures. Here, we emphasize the importance of applying analytical tools or models that are capable of addressing the relative level of complexity of the pressure-state interactions, and that also allow for the attribution of impacts to human wellbeing and effectiveness of management responses. In considering the spectrum of possible levels of system complexity, analytical approaches can differ remarkably in their ability to meet the needs of the DPSIR Framework (Table 1).

Table 1. Sufficiency of analytical tools and models to implement DPSIR framework with different levels of system complexity (adapted from Hayes et al. 2015).



Tools and models	Complexity of cause-effect relationships: None (1)	Simple (2)	Directed (3)	Diffuse (4)	Feedback (5)
1. Unstructured list	✓	✓			
2. Objective-indicator matrix	✓	✓			
3. Structured list		✓	✓		
4. Value-impact matrix		✓	✓		
5. Conceptual diagram or cartoon		✓	✓		
6. Influence diagram		✓	✓	✓	
7. Fuzzy cognitive map		✓	✓	✓	
8. Statistical model		✓	✓	✓	
9. Bayes net (6)			✓	✓	✓ (7)
10. Qualitative process model (8)				✓	✓
11. Quantitative process model (9)				✓	✓

(1) No cause-effect relationship; the pressure is the indicator; methods beyond objective-indicator matrix not required.

(2) Pressure directly impacts indicator variable; methods beyond statistical models not required.

(3) Pressure directly impacts a variable that has knock-on effects to indicator variable; methods beyond Bayes nets not required.

(4) Pressure indirectly impacts an indicator variable via multiple intervening variables.

(5) Multiple pressures simultaneously impact complex system with feedbacks between variables.

(6) Also known as Bayesian network or Bayesian belief network.

(7) With difficulty; standard Bayes nets are limited to acyclic graph structures. Dynamic Bayes nets include feedback but are difficult to parameterise and analyse, and thus are impractical for complex systems (but see Hosack *et al.* 2008 for Bayes nets based on qualitative process models).

(8) Also known as qualitative mathematical models or signed digraph models.

(9) Also known as numerical simulation models.

P	pressure acting on one or more variables
V	system variable: an element of the ecological or human system or benefit derived from that system that forms part of the cause-effect relationship but is not measured
I	indicator variable: a measurable indicator of a specific ecosystem element or benefit derived from the ecosystem

The sufficiency of the analytical tool or model is determined by 1) its ability to summarise the most important ecosystem descriptors, spatial and temporal scales of biological processes, current and potential threats to system, and feasible management interventions, 2) its capacity to identify indicators for monitoring by focusing on those aspects of the system that are most informative with respect to both the impact of potential pressures and the feasibility of management interventions, and 3) its suitability to facilitate the interpretation of monitoring results and the formulation of alternative courses of management. Due to these analytical demands, the number of suitable methods decreases as the complexity of the system's cause-effect relationships increase (Table 1). The simplest method, collating lists of candidate variables that satisfy selection criteria, works well in simple situations where an impact to an indicator is directly attributed to a specific pressure (*e.g.,* easily measured toxicants in a supply of drinking water). Statistical and mathematical process models become necessary in more complex systems with pressures that affect variables linked by causal pathways of interaction. Where there are multiple pressures on a system with complex feedbacks then only mathematical process models, either qualitative or quantitative, are sufficient to meet the needs of the DPSIR framework. The utility of models in this context is determined, in part, by the level of available information and the level of system complexity. This is a key issue given the high levels of uncertainty in our knowledge of the structure and function of deep-sea ecosystems, and in the nature and extent of impacts from deep-sea mining.

For applications centred in complex human and ecological systems, we advocate a strategy of model building (Levins 1966) that seeks to combine statistical approaches with qualitative process models and quantitative process models, recognising that each approach has inherent strengths and weaknesses. Taken in combination, each approach provides complementary and mutually informative results that lead to more robust understanding, prediction, and intervention in complex systems. As monitoring and management programs for complex ecosystems are typically data-limited (especially in large-scale marine ecosystems), we maintain that qualitative process models (also known as *qualitative mathematical models* or signed digraphs, Levins 1998) are an appropriate choice for initial phases of their design and implementation. Data requirements for qualitative mathematical models are far less than that of quantitative process models and statistical models, and they can be used to rigorously describe the main interacting physical and biological variables within a system, linking them to their surrounding ecosystem, to the activities and pressures of concern, describe and predict impacts to measurable components of human wellbeing and assess the likely success of potential management interventions. Their construction can be assisted by other analytical tools, including statistical analyses, and analysis of the structured lists and value-impact matrices, which can inform assessment of the relative importance of activities and pressures on the system. Model links are qualitative and represent only the 'sign' of the direct effects (*i.e.,* positive, negative, zero); while not quantitative or precise, qualitative mathematical models can provide rigorous means to assess a system's dynamics and its response to disturbances (Dambacher *et al.* 2002, 2003, 2007; *e.g.,* Fig. 5 left panel).

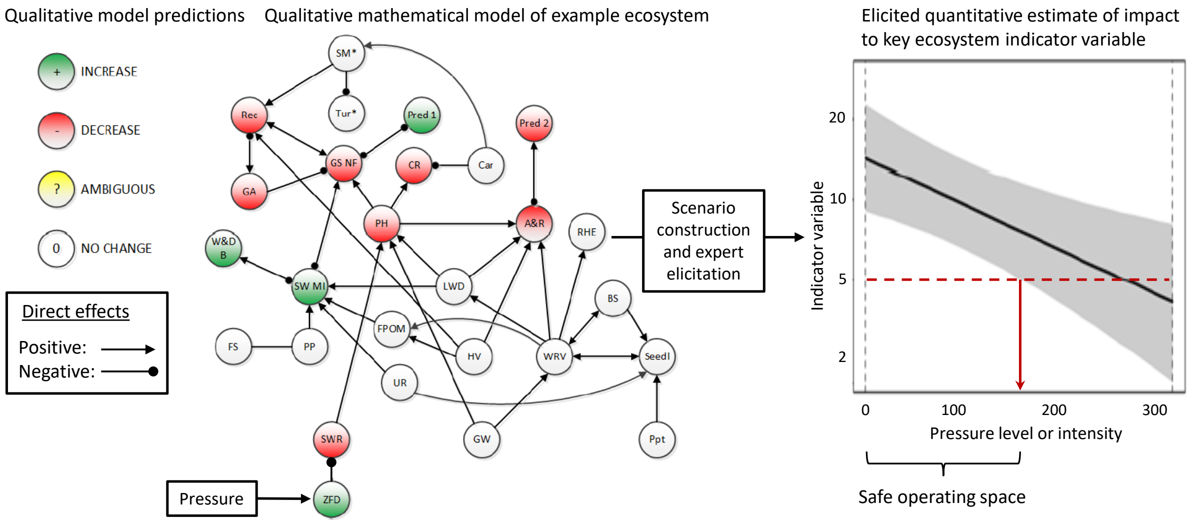


Figure 5. Illustration of a qualitative and quantitative assessment of cumulative impacts for an example ecosystem showing use of qualitative mathematical model to identify key indicator variables that can be used to predict cumulative impacts qualitatively, and then to focus expert elicitations for a quantitative estimate of impact with associated levels of uncertainty (greyed area). Dashed red line illustrates concept of a predefined 'serious harm' threshold based on an accepted level of uncertainty (lower limit of greyed area) and solid line with arrow illustrates concept of an upper limit to safe operational space for a proposed activity that generates a pressure on the system (adapted from Hosack *et al.* 2018).

Hosack *et al.* (2008) developed acyclic Bayes nets that incorporate the qualitative dynamics of complex systems through analysis of the feedback properties of qualitative mathematical models; Bayes nets of this structure (Fig. 6) perform the key IEA tasks of:

1. Qualitatively predict cumulative impacts to complex systems arising from both pressures and management interventions.
2. Diagnose the likely cause of changes observed in the system.
3. Test the validity of alternative models of the system.
4. Identify and rank informative system indicators.

In the first task the application of qualitative mathematical models that incorporate alternative management interventions (*e.g.*, via multiple alternative models with multiple sources of input) it is possible to undertake a formal qualitative analysis of management options and strategies (e.g., Dambacher *et al.* 2015, Trenkel *et al.* 2015) though this is more typically done with quantitative process models; see *Evaluation of management options* below for more detail).

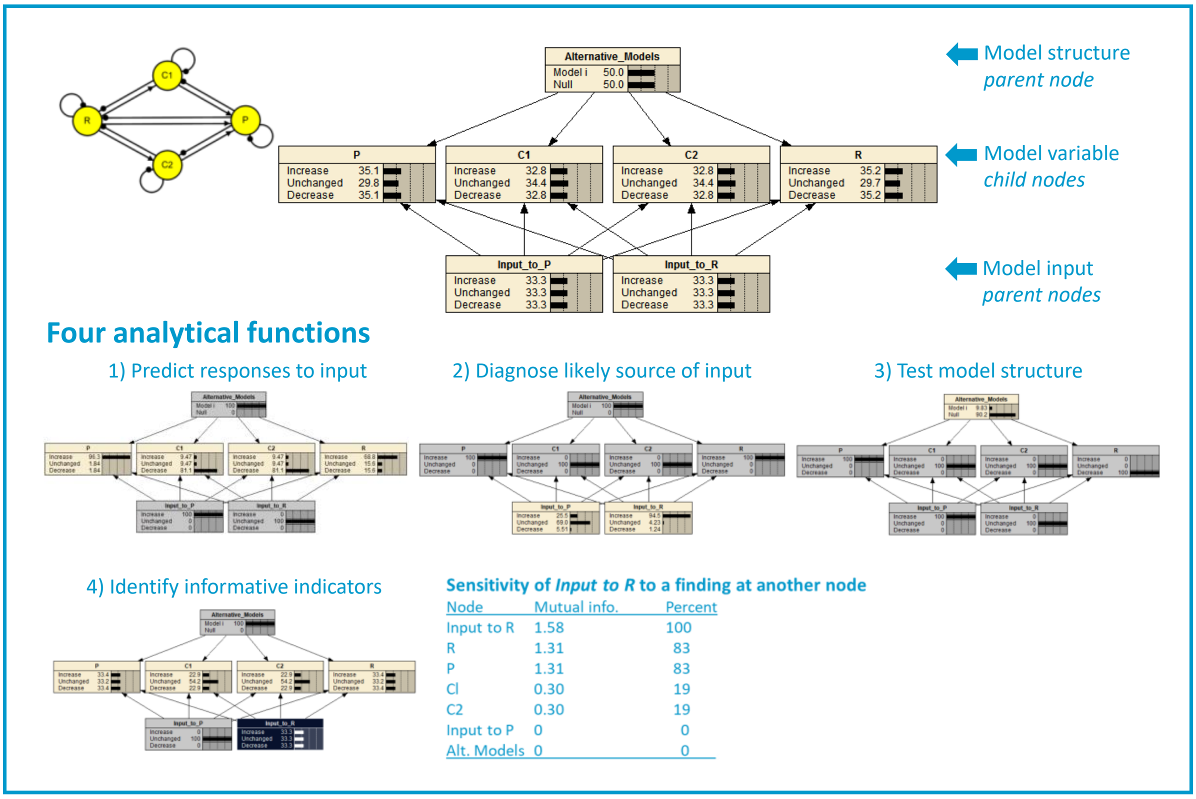


Figure 6. Representation of dynamics of qualitative mathematical models in Bayes nets and their use to perform key IEA tasks.

Attribution of values to system components

To address the second DPSIR challenge we advocate create an explicit mapping of system components to aspects of human well-being that are specifically defined through ecosystem functions and services. Descriptions of final provisioning, regulating, and cultural services will be adapted from the Common International Classification for Ecosystem Services (CICES) database (EC 2013, Czúcz *et al.* 2018). Descriptions of ecosystem functions will be derived from peer-reviewed literature (*e.g.,* Armstrong *et al.* 2012; Culhane *et al.* 2018; Le *et al.* 2017; Thornborough *et al.* 2019), as well as ISA recommendation for the guidance of mining contractors (ISA 2020). Beyond aiding identification informative indicators, ecosystem functions and services, where applicable, will be used as endpoints in assessing risk.

Development and attribution of management response

The regulation of seabed mining is a work in progress with Exploitation Regulations still being developed or in draft form (Jaeckel 2015, Blanchard *et al.* 2023). Regulation of seabed mining proceeds from Article 145 of UNCLOS, which requires ISA to adopt and implement appropriate rules, regulations, and procedures to, among other things:

a) the prevention, reduction and control of pollution and other hazards to the marine environment, including the coastline, and of interference with the ecological balance of the marine environment, particular attention being paid to the need for protection from harmful effects of such activities as drilling, dredging, excavation, disposal of waste, construction and operation or maintenance of installations, pipelines and other devices related to such activities; and

b) the protection and conservation of the natural resources of the Area and the prevention of damage to the flora and fauna of the marine environment.

Furthermore, the ISA is required to protect the marine environment from the harmful effects of seabed mining activities and make recommendations to avoid "serious harm" to the marine environment (UNCLOS Art165). The ISA is currently progressing a draft set of Regulations, Standards and Guidelines to regulate exploitation of mineral resources in the Area (ISA 2019). These form part of an overarching Mining Code which comprises rules, regulations and procedures covering both exploration and potential exploitation of mineral resources (https://www.isa.org.jm/the-mining-code/). An important component of the Mining Code is to prevent serious harm to the environment.

The definitions of the terms 'effects', 'impacts', and 'harm', however, as well as of the qualifying terms 'significant' and 'serious' are not used consistently in legislation or science and policy literature. There is currently no agreed operational definition of impact significance in national jurisdictions (Murray *et al.* 2018) or of serious harm for the seabed area beyond national jurisdictions (Levin *et al.* 2016), although there have been several recent attempts to progress this issue (ISA 2022a, Hitchin *et al.* 2023, Hiddink *et al.* in press). Leduc *et al.* (2024) propose a framework to develop operational definitions based on ISA documentation (ISA 2000, 2020, 2022b, c) as well as international and national criteria and approaches developed for the management of deep-sea resources (*e.g.,* ILC 2001, Mengerink 2008). This approach advances three levels of harm: detectable, significant and serious, with significant harm bounded by a lower threshold and an upper limit (Fig. 7). A two-step process is proposed that first assesses environmental effects and then assesses provisional levels of harm with corresponding thresholds and limits defining the boundaries between detectable, significant, and serious harm (Fig. 8).

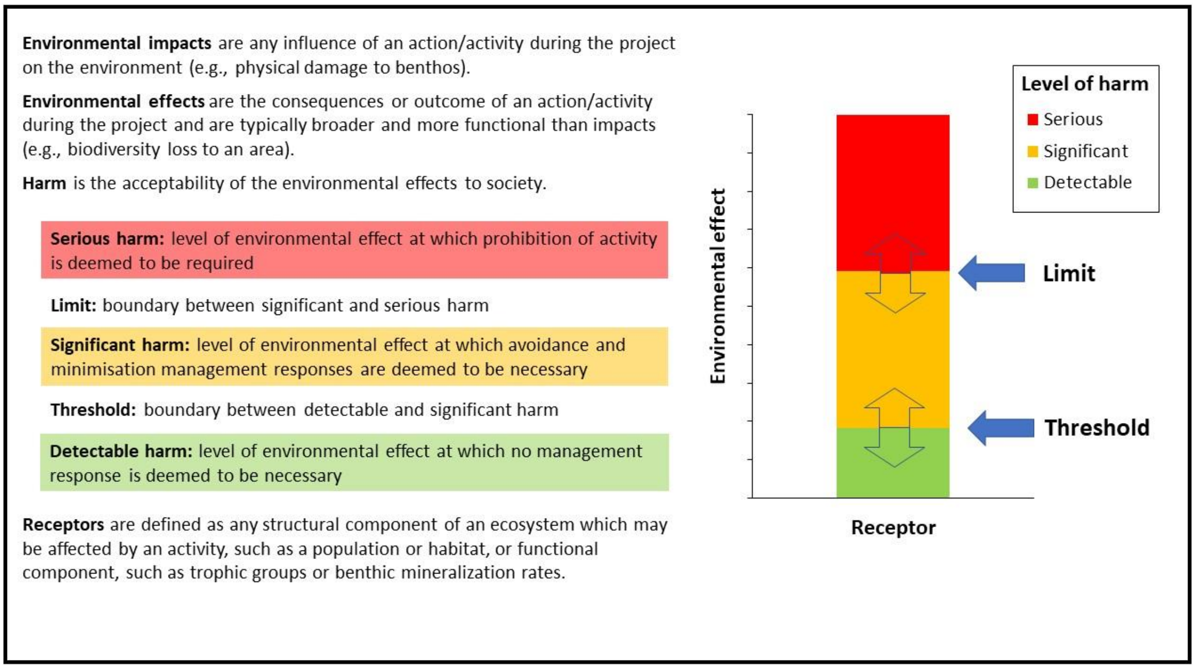


Figure 7. Schematic representation of environmental impacts or effects as well as detectable, significant and serious harm, and thresholds and limits, for an ecosystem component or indicator variable (from Leduc *et al.* 2024).

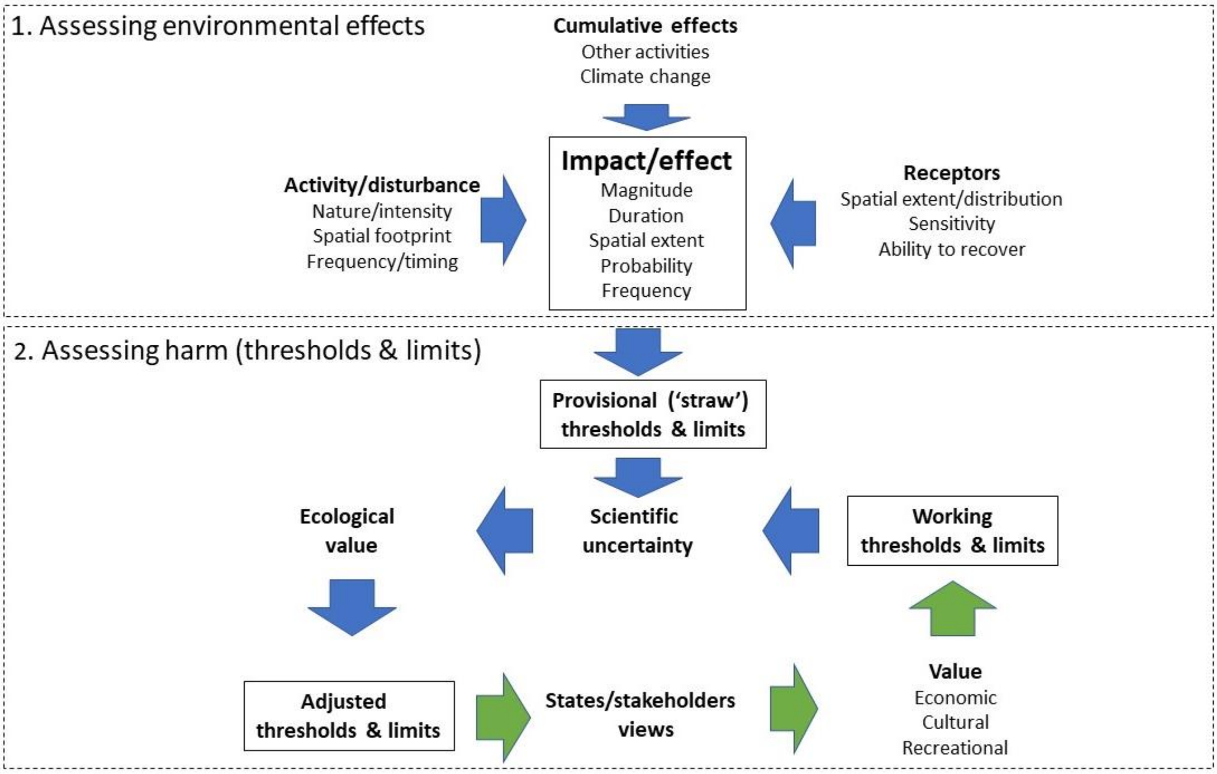


Figure 8. Schematic representation of proposed stepwise process for determining levels of environmental effects and harm on ecosystem components or indicator variables. Step 1. Identifying and measuring key attributes of environmental change. Step 2. Assessing the severity of harm and defining thresholds and limits of acceptable environmental change. Blue arrows: steps undertaken by scientific community; green arrows: steps undertaken by regulators/states/stakeholders (from Leduc *et al.* 2024).

The above approach will provide the key ingredients for creating effective and operational objectives for key indicator variables. In consultation with TMC, effective and operational objectives will adhere to the SMART properties and criteria put forward by ICES (2005):

1. Specific. Objectives should clearly specify the state to be achieved and be interpreted unambiguously by all stakeholders.
2. Measurable. Good objectives should relate to measurable properties of ecosystems and human societies, so that indicators and reference points can be developed to measure progress towards the objective.
3. Achievable. Good objectives should not conflict. Within an effective management framework, it should be possible to achieve all objectives. Good objectives should describe a state of the ecosystem, including the position and activities of humans within it, which accurately reflects the values and desires of a majority of stakeholders.
4. Realistic. Good objectives will be implementable using the resources (research, monitoring, and assessment and enforcement tools) available to managers and stakeholders. Good objectives should reflect the aspirations of stakeholders, such that the majority of stakeholders will strive to achieve them and ensure sustainable development.
5. Time bound. There should be a clearly defined time scale for meeting objectives.

3.3.2 Indicator selection

For indicator selection we will employ a structured process developed and tested by Hayes *et al.* (2015). This approach (Fig. 9) emphases that informative and fit-for-purpose indicators of complex ecological and socio-economic systems emerge as a result of a clearly defined process rather than as an *a priori* set. The process is based on a spatially explicit description of key components of the ecological and socio-economic systems and predicts how components will be impacted by stressors and pressures. The process does not require experts to agree on the system's structure or the activities that threaten the ecosystem. Rather it defines a set of alternative system models and pressure scenarios that accommodate epistemic uncertainty. Informative and robust indicators are identified based on their consistency with which they are predicted to respond across the set of alternative models and scenarios. This process for indicator selection is scientifically based in that it relies on indicators to be selected from a compressive set of model predictions. The validity of these models is subsequently tested through an integrated monitoring program that compares model predictions with observed responses in the systems.

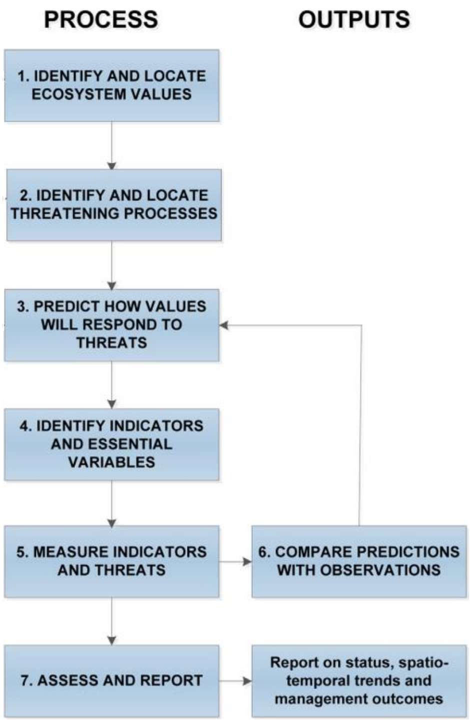


Figure 9. Process for identifying informative indicators for IEA (adapted from Hayes et al. 2015).

3.4 Ecosystem assessment

In this step the ecosystem is assessed through an analysis of the previously defined indicators to determine the status of the ecological and socio-economic system and how it might be changing through time. Where historical data is lacking then the first iteration of the EBM cycle will establish the all-important baseline for which future reporting will build upon. Statistical analyses of individual indicator results provide an assessment of whether management goals and prescribed targets are being met or not for key components of the system; integrating results from the full suite of indictors provides a whole-of-system assessment of status and trends. With time, repeated iterations of the EBM cycle provide a comprehensive record of system and its management (*e.g.,* NOAA Ecosystem Status Reports).

3.5 Risk assessment

A risk assessment is undertaken to determine the likelihood that indicator will reach or surpass a predetermined management threshold or limit due to any of the stressors and pressures on the system. For complex ecological and socio-economic systems, the risk assessment will necessarily require a comprehensive analysis of cumulative impacts. And where there are many components, processes and pressures that are interdependently linked through system feedback, then the assessment will require application of mathematical tools that are of commensurate rigor (Table 1). Within the context of a data-poor system, there is insufficient data to reliably parameterize statistical or quantitative process models to assess cumulative impacts, thus for the first iteration of the EBM cycle, we advocate the use of qualitative mathematical models (a.k.a., qualitative process models, loop models, signed digraph models) to assess the cumulative impact of stressors and pressures on the system (Fig. 5 left panel). Results from the qualitative models are used to focus expert elicitations for a quantitative estimate of cumulative impact for key indicator variables. The expert elicitation essentially parameterizes a dose-response type relationship (*e.g.,* Fig. 5 right panel) that is defined by a Bayesian general linearized model (Hosack *et al.* 2017). Such relationships provide the essential ingredient for assessing risk, the development and evaluation of management strategies, and, coupled with thresholds and limits derived from definitions of serious harm (Fig. 7 and 8), can be used to define a safe operating space for proposed activities (*e.g.,* Fig. 5 red arrow in right panel).

Intrinsic to the assessment cumulative impacts is a critically important spatial context that greatly influences the interpretation and implications of risk. Anthony *et al.* (2013) developed the concept of a zone of influence to support spatially explicit assessments of cumulative impacts. It is based on a mapping of the relative intensity or concentration of a given pressure in two- or three-dimensional space with respect to its potential to impact key indicators or values (Fig. 10).

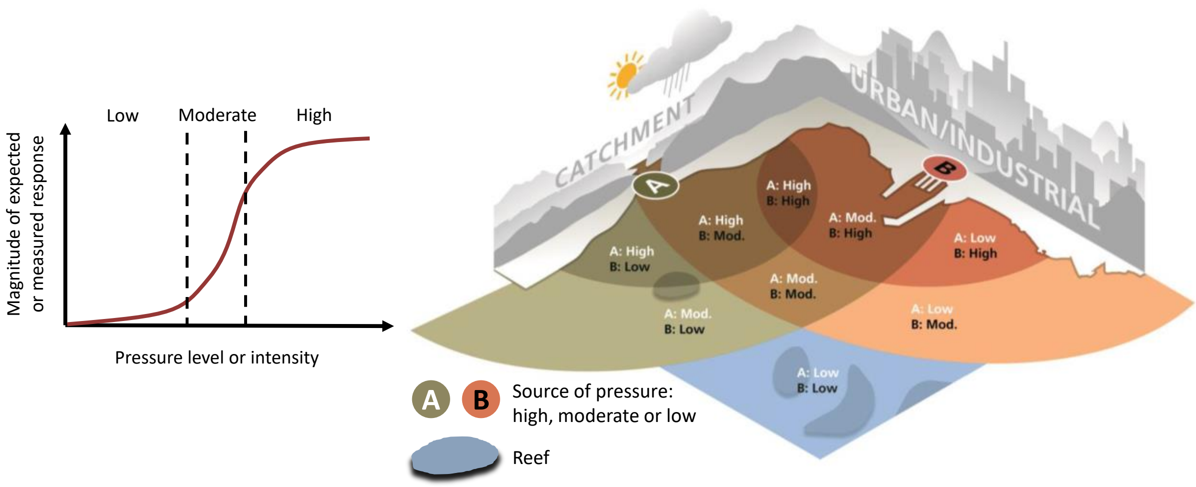


Figure 10. Zone of influence for difference sources of pressures overlapping with valued components of system creating a mosaic of pressure-state interaction intensities. Pressure intensity zones based on predefined thresholds or limits (dashed vertical lines) in dose-response type relationship; adapted from Anthony *et al.* (2013).

Zones of influence are areas that encompass valued component of the ecosystem and are defined by predetermined thresholds and limits in the intensity or concentration of a pressure and its direct effect on an indicator variable or valued component of the system (*i.e.,* Fig. 7 and 8). A fundamental aspect of assessing cumulative impacts is attribution, *i.e.,* distinguishing specific sources of a pressure of concern from existing or background levels, whether they are from anthropogenic or natural sources. A given pressure may be distinguished as having a discrete entry point into ecosystem, or it may directly impact multiple components of an ecosystem. The definition and delineation of zones of influence may require several iterations that consider the influence of a pressure on the system against the magnitude of thresholds for system components or values with respect to specific assessment and measurement endpoints.

Dunstan *et al.* (2019) provide a checklist of considerations for developing zones of influence:

Questions	Caveats
Are pressures linked to ecosystem components	
Is the response variable of the dose-response relationship clearly represented in the ecosystem's conceptual model?	If no, the conceptual model should be reconsidered to ensure that identified responses variables are represented
Is the zone of influence based on a well-defined dose-response type relationship (demonstrated and measured clear impact) relevant to the valued components of the ecosystem?	If no, care must be taken to ensure that the effect of pressures can be linked to values
Are threshold values sufficiently detailed to address the biology of the response variable?	If no, uncertainty about the threshold for a response should be considered
Do threshold values address a range of effects that are relevant to management concerns and desired future conditions of associated values?	If no, additional caution is necessary as the reliability of predictions cannot be determined.
Is uncertainty in the dose-response relationship adequately assessed and documented?	
If based on empirical data, does the dose-response relationship included error bounds?	If no, uncertainty about the threshold for a response should be considered
If based on modelling studies, is there documentation of variation in modelling results?	If no, additional evidence of the dose-response relationship should be sought
If based on expert opinion, is there documentation of the elicitation process and attendant level of uncertainty?	If no, additional evidence of the dose-response relationship should be sought
Does the zone of influence adequately address or document different sources of pressures relevant to the assessment?	
Is the granularity of the pressure data sufficient to address the pattern of distribution in the response variable of the dose-response relationship and the distribution pattern of valued components of the system?	If no, caution needs to be taken to ensure that the spatial and temporal scale are appropriate to enable estimation of impact
Are concentrations or intensities of existing pressures adequately differentiated from pressures associated with proposed projects and plans of management?	If no, care needs to be taken to distinguish the effects pressures from other potential sources of impact
Are anthropogenic sources of pressures adequately differentiated from natural or otherwise background levels of pressures?	If no, care needs to be taken to distinguish the effects pressures from other potential sources of impact

3.6 Uncertainty assessment

When we have robust control it is not crucial that we resolve our uncertainties about the real world. It is only in those cases where we are unable to achieve robustness that we are compelled to improve our understanding of how the world actually works. Our 'burden of proof' is met when we demonstrate that a controller is capable of meeting our objectives. We do not need to prove in advance that a given level of a practically reversible human activity is sustainable, it is sufficient to prove that our methods for managing that activity will be capable of curtailing it before its impacts exceed acceptable bounds.
de la Mare (2005)

Uncertainty is inherent in any IEA and can occur at several stages throughout the process: quantifying the amount of an individual pressure produced by an activity; the extent to which that pressure impacts the ecosystem; how single pressures interact with one another and how these interactions vary across space and time (*i.e.*, factors of exposure); and how ecological components are affected. Uncertainty can arise through the inherent vagaries of language (*i.e.*, linguistic uncertainty) inadequate knowledge (*i.e.*, epistemic uncertainty), low predictive ability of ecosystem behaviour, natural variability, measurement error, or changing policies or management objectives (*i.e.*, decision uncertainty), and all can be manifest in the calculation and communication of risk in an IEA (Halpern and Fujita 2013, Opdam *et al.* 2009, Stelzenmüller *et al.* 2015, Clarke Murray *et al.* 2014).

Identifying and documenting the various sources of uncertainty is a central requirement for IEAs as it underpins how managers and contractors should react to assessed levels of risk. Based on the precautionary principle, higher levels of uncertainty in an estimated level of risk should attract commensurately higher levels of management constraints and conditions (Fig. 11). Where this uncertainty can be reduced through acquisition of data and knowledge, then there is an economic incentive for contractors to increase their investment in research and monitoring, with any reduction in uncertainty potentially leading to a decrease in management constraints and conditions but also an increase in system knowledge (there is of course the possibility that the revised estimate of risk does not decrease, and management conditions are not reduced). Here we propose that the engagement triad explicitly consider an adaptive approach to risk management that facilitates identification of key uncertainties, encourages monitoring and research to resolve them, and supports sufficiently rigorous yet flexible management arrangements to encourage investment in targeted research and monitoring that meets the needs and concerns of contractors, regulators and stakeholders.

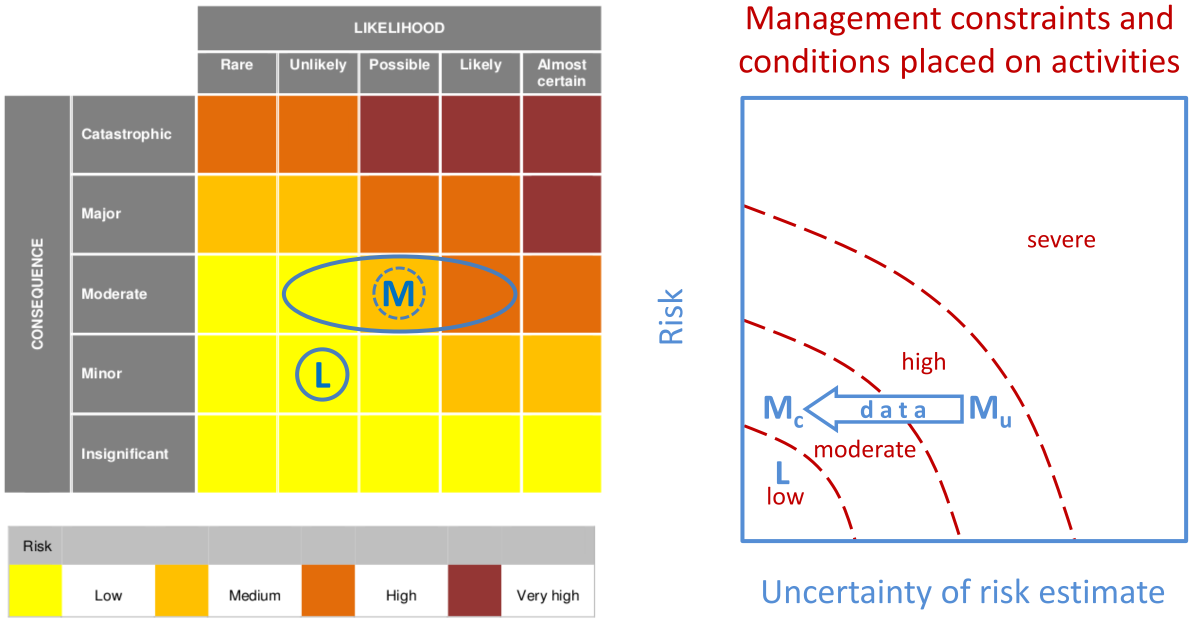


Figure 11. Illustrative example of risk estimates for activities with different levels of uncertainty that are mitigated through different levels of management constraints and conditions. An activity with an estimated level of risk that is low (L) and certain would have a relatively low level of management constraints and conditions placed upon it, while an activity with a moderate but uncertain level of estimated risk (M_u) would operate under a higher level of management constraints and conditions (*i.e.*, here the higher uncertainty is associated with poorly defined likelihood where the solid-line circle around M in the left panel spans three levels including low, medium and high risk). Where uncertainty in the estimate of risk likelihood is reduced (*i.e.*, dashed-line circle around M) to an acceptable level through acquisition of data from targeted research and monitoring, then the same level of estimated risk that is deemed more certain (M_C) can be mitigated through a moderate level of management constraints and conditions (*i.e.*, M_u versus M_c in right panel).

3.7 Evaluation of management options

This penultimate step uses system models to evaluate alternative strategies and options that can be used to meet management objectives. For complex ecological and socio-economic systems, this requires exploring and assessing management options and strategies through the mathematical abstractions of qualitative or quantitative process models (Table 1). Management strategy evaluation (Smith 1994, de la Mare 1996) can proceed from either a qualitative process modelling approach (*e.g.,* qualitative mathematical models; Dambacher *et al.* 2015, Trenkel *et al*. 2015) or quantitative process models (*e.g.,* Atlantis; Fulton *et al.* 2014; Ecopath with Ecosim, Coll and Libralato 2012). Either approach attempts to replicate the essential dynamics of an ecological and socio-economic system and understand how it might react to different management options.

The key ingredients for this step include clearly defined management objectives, a set of performance criteria related to each objective, a set of alternative management strategies or options to meet the objectives, and a means to calculate the performance criteria for each strategy (Smith 1994). The main goals of this step include identifying trade-offs, revealing unintended consequences, and identifying options with the greatest chance of success.

3.8 Monitoring and evaluation

Based on the best option or strategy identified in the previous step, a management action is selected and implemented (Fig. 1 inner cycle), and in this final step of the IEA there is a concurrent implementation, or continuation, of an integrated monitoring program the purpose of which is to provide feedback on the status and trend of the ecological and socio-economic system to determine whether management objectives are being achieved. The feedback of monitoring results into the first four steps of the IEA is the essential fuel that drives the adaptive capacity of EBM. Within the engagement triad (Fig. 2), the reporting of monitoring results to stakeholders establishes transparency and credibility, and reporting to managers and regulators determines and maintains the relevancy of science within the decision-making process. Analyses of measured responses of system indicators strengthens the scientific basis of EBM, as provides the means to test the validity of the conceptual and mathematical model on which the IEA depends (Fig. 10, Hayes *et al.* 2015) and to determine the information content and utility of the indicators being monitored so that status and trend assessments can be adapted, focused, and improved over time.

Hedge *et al.* (2017) developed a framework and guidance for integrated monitoring that is consistent with the IEA steps and approach articulated here; it describes nine essential functions:

1. Clearly defining the purpose of the monitoring program and the monitoring objectives.
2. Compiling and analysing relevant information on existing monitoring programs.
3. Developing (and refining) conceptual models.
4. Developing (and refining) overall design for integrated monitoring.
    a. Selecting and prioritising indicators.
    b. Selecting monitoring programs for integrated monitoring.
    c. Developing (and refining) sampling design for integrated monitoring.
5. Developing and refining monitoring protocols.
6. Managing data.
7. Analysing data.
8. Reporting and communication.
9. Reviewing and auditing.

4 A strategy for data- and experience-poor environments

We have set out a framework for IEAs that is intended to meet the challenges of a data- and experience-poor socioecological system in which deep seabed mining intends to operate. While the challenges remain daunting, the tools that are brought to bear have a proven track record to address high levels of system complexity and uncertainty through a scientific approach that is scalable in its scope, transparent in its assumptions, rigorous in its formulation and testable in its predictions.

Our approach is parsimonious and adaptive, by first employing qualitative analytical tools to model complex and data poor ecological systems, it allows for iterations of model building, testing and revision with the expectation of increases in data and knowledge from an integrated monitoring program that will eventually support greater precision and detail in quantitatively specified operating models. Beyond these operational aspects, the successful implementation of this approach will require extensive and ongoing consultation with managers, regulators and stakeholders.

5 References

Anthony, K.R., Dambacher, J.M., Walshe, T. and Beeden, R., 2013. A framework for understanding cumulative impacts, supporting environmental decisions and informing resilience based management of the Great Barrier Reef World Heritage Area: Final Report to the Great Barrier Reef Marine Park Authority and Department of the Environment.

Armstrong, C.W., Foley, N.S., Tinch, R. and van den Hove, S. 2012. Services from the deep: Steps towards valuation of deep sea goods and services. Ecosystem Services, 2, pp.2-13.

Bagheri, A. and Saadati, M.J.I.J.S.T., 2015. Exploring the effectiveness of chain referral methods in sampling hidden populations. Indian Journal of Science and Technology, 8(30), pp.1-8.

Blanchard, C., Harrould-Kolieb, E., Jones, E. and Taylor, M.L., 2023. The current status of deep-sea mining governance at the International Seabed Authority. Marine Policy, 147, p.105396.

Clarke Murray, C., Mach, M.E. and Martone, R. 2014. Cumulative effects in marine ecosystems: scientific perspectives on its challenges and solutions. World Wildlife Fund Canada.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210: 23–253.

Coll, M. and Libralato, S. 2012. Contributions of food web modelling to the ecosystem approach to marine resource management in the Mediterranean Sea. Fish and Fisheries, 13(1), pp.60-88.

Colvin, R.M., Witt, G.B. and Lacey, J., 2016. Approaches to identifying stakeholders in environmental management: Insights from practitioners to go beyond the ‘usual suspects’. Land use policy, 52, pp.266-276.

Culhane, F.E., Frid, C.L.J., Royo Gelabert, E., White, L., Robinson, L.A., 2018. Linking marine ecosystems with the services they supply: what are the relevant service providing units? Ecol Appl 28, 1740–1751.

Czúcz, B., Arany, I., Potschin-Young, M., Bereczki, K., Kertész, M., Kiss, M., Aszalós, R. and Haines-Young, R. 2018. Where concepts meet the real world: A systematic review of ecosystem service indicators and their classification using CICES. Ecosystem Services, 29, pp.145-157.

Dambacher, J.M., Li, H.W. and Rossignol, P.A. 2002. Relevance of community structure in assessing indeterminacy of ecological predictions. Ecology, 83(5), pp. 1372-1385.

Dambacher, J.M., Luh, H.K., Li, H.W. and Rossignol, P.A. 2003. Qualitative stability and ambiguity in model ecosystems. The American Naturalist, 161(6), pp. 876-888.

Dambacher, J.M. and Ramos-Jiliberto, R. 2007. Understanding and predicting effects of modified interactions through a qualitative analysis of community structure. The Quarterly review of biology, 82(3), pp. 227-250.

Dambacher, J.M., Rothlisberg, P.C. and Loneragan, N.R. 2015. Qualitative mathematical models to support ecosystem-based management of Australia's Northern Prawn Fishery. Ecological Applications, 25(1), pp.278-298.

de la Mare, W.K. 1996. Some recent developments in the management of marine living resources. In Frontiers of Population Ecology, pp. 599–616. Ed. by RB Floyd, AW Shepherd, PJ De Barro. CSIRO Publishing, Melbourne, Australia.

de la Mare, W.K. 2005. Marine ecosystem-based management as a hierarchical control system. Marine policy, 29(1), pp.57-68.

DePiper, G.S., Gaichas, S.K., Lucey, S.M., Pinto da Silva, P., Anderson, M.R., Breeze, H., Bundy, A., Clay, P.M., Fay, G., Gamble, R.J. and Gregory, R.S. 2017. Operationalizing integrated ecosystem assessments within a multidisciplinary team: lessons learned from a worked example. ICES Journal of Marine Science, 74(8), pp.2076-2086.

Dunstan, PK., Dambacher, JM., Thornborough, K., Marshall, N &Stuart-Smith, R. 2019. Technical Report describing Guidelines for analysis of cumulative impacts and risks to the Great Barrier Reef (Part 1). Report to the National Environmental Science Program, Marine Biodiversity Hub.

European Commission. 2013. Mapping and assessment of ecosystems and their services: an analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020: discussion paper – final, April 2013. Publications Office of the European Union, LU.

Forrester, J., Cook, B., Bracken, L., Cinderby, S. and Donaldson, A. 2015. Combining participatory mapping with Q-methodology to map stakeholder perceptions of complex environmental problems. Applied Geography, 56, pp.199-208.

Fulton, E.A., Smith, A.D., Smith, D.C. and Johnson, P. 2014. An integrated approach is needed for ecosystem based fisheries management: insights from ecosystem-level management strategy evaluation. PloS one, 9(1), p.e84242.

Goodman, L.A. 1961. Snowball sampling. The annals of mathematical statistics, pp.148-170.

Goodman, L.A. 2011. Comment: On respondent-driven sampling and snowball sampling in hard-to-reach populations and snowball sampling not in hard-to-reach populations. Sociological methodology, 41(1), pp.347-353.

Groffman PA, Driscoll CT, Likens GE, Fahey TJ, Holmes RT, Eagar C and Aber JD. 2004. Nor gloom of night: a new conceptual model for the Hubbard Brook ecosystem study. BioScience 54: 139–148.

Gross J.E. 2003. Developing conceptual models for monitoring programs. United States National Parks Service, Inventory and Monitoring Program, Ft Collins, Colorado; downloaded 8 April 2023, http://npshistory.com/publications/interdisciplinary/im/conceptual-model-overview.pdf.

Halpern, B.S. and Fujita, R. 2013. Assumptions, challenges, and future directions in cumulative impact analysis. Ecosphere, 4(10), pp.1-11.

Hayes, K.R., Dambacher, J.M., Hosack, G.R., Bax, N.J., Dunstan, P.K., Fulton, E.A., Thompson, P.A., Hartog, J.R., Hobday, A.J., Bradford, R. and Foster, S.D. 2015. Identifying indicators and essential variables for marine ecosystems. Ecological Indicators, 57, pp. 409-419.

Hedge, P., Molloy, F., Sweatman, H., Hayes, K.R., Dambacher, J.M., Chandler, J., Bax, N., Gooch, M., Anthony, K. and Elliot, B. 2017. An integrated monitoring framework for the great barrier reef world heritage area. Marine Policy, 77, pp.90-96.

Hiddink JG, Valanko S, Delargy AJ, van Denderen PD. 2023. Setting thresholds for good ecosystem state in marine seabed systems and beyond. ICES Journal of Marine Science. 2023 Mar 6:fsad035.

Hitchin, B., Smith, S., Kröger, K., Jones, D.O., Jaeckel, A., Mestre, N., Ardron, J., Escobar, E., Van Der Grient, J. and Amaro, T. 2022. Thresholds in Deep-seabed Mining: A Primer for Their Development. https://doi.org/10.20944/preprints202207.0270.v1.

Hosack, G.R., Hayes, K.R. and Dambacher, J.M. 2008. Assessing model structure uncertainty through an analysis of system feedback and Bayesian networks. Ecological Applications, 18(4), pp.1070-1082.

Hosack, G.R., Hayes, K.R. and Barry, S.C. 2017. Prior elicitation for Bayesian generalised linear models with application to risk control option assessment. Reliability Engineering & System Safety, 167, pp. 351-361.

Hosack GR, Ickowicz A, Hayes KR, Dambacher JM, Barry SA and Henderson BL. 2018. Receptor impact modelling. Submethodology M08 from the Bioregional Assessment Technical Programme. Department of the Environment and Energy, Bureau of Meteorology, CSIRO and

Geoscience Australia, Australia. http://data.bioregionalassessments.gov.au/submethodology/M08.

International Council for the Exploration of the Sea. 2005. Guidance on the Application of the Ecosystem Approach to Management of Human Activities in the European Marine Environment, ICES Cooperative Research Report, No. 273. 22 pp.

International Law Commission. 2001. Draft Articles on Prevention of Transboundary Harm from Hazardous Activities with Commentaries: Report of the International Law Commission on the Work of Its Fifty-Third Session.

International Seabed Authority. 2000. Regulations on the Prospecting and Exploration for Polymetallic Nodules in the Area, ISBA/16/A/12 /Rev.1 (approved on 13 July 2000 and amended on 25 July 2013).

International Seabed Authority. 2019 Draft regulations on exploitation of mineral resources in the Area. ISBA/25/C/WP.1.

International Seabed Authority. 2020. Recommendations for the guidance of contractors for the assessment of the possible environmental impacts arising from exploration for marine minerals in the Area. ISBA/25/LTC/6/Rev.2: 40.

International Seabed Authority. 2022a. Normative environmental thresholds for deep-seabed mining. Submitted by the delegation of Germany. ISBA/27/C/30.

International Seabed Authority. 2022b. Draft standard and guideline for the environmental impact assessment process: draft prepared by the Legal and Technical Commission. ISBA/27/C/4.

International Seabed Authority. 2022c. Draft guidelines for the preparation of environmental impact statements: prepared by the Legal and Technical Commission. ISBA/27/C/5.

Jaeckel, A. 2015. An environmental management strategy for the international seabed authority? The legal basis. The International Journal of Marine and Coastal Law, 30(1), pp.93-119.

Jones, D.O., Kaiser, S., Sweetman, A.K., Smith, C.R., Menot, L., Vink, A., Trueblood, D., Greinert, J., Billett, D.S., Arbizu, P.M. and Radziejewska, T. 2017. Biological responses to disturbance from simulated deep-sea polymetallic nodule mining. PLoS One, 12(2), p.e0171750.

Kirkfeldt, T.S. 2019. An ocean of concepts: Why choosing between ecosystem-based management, ecosystem-based approach and ecosystem approach makes a difference. Marine Policy, 106, p.103541.

Le, J.T., Levin, L.A. and Carson, R.T., 2017. Incorporating ecosystem services into environmental management of deep-seabed mining. Deep Sea Research Part II: Topical Studies in Oceanography, 137, pp.486-503.

Leduc, D., Clark, M.R., Rowden, A.A., Hyman, J., Dambacher, J.M., Dunstan, P.K., Connolly, R., Fulton, E.A., Hosack, G.R., O'Hara, T. and Parr, J.M., 2024. Moving towards an operational framework for defining serious harm for management of seabed mining. *Ocean & Coastal Management*, *255*, p.107252.

Levin, P.S., Fogarty, M.J., Murawski, S.A. and Fluharty, D., 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. PLoS biology, 7(1), p.e1000014.

Levin, L.A., Mengerink, K., Gjerde, K.M., Rowden, A.A., Van Dover, C.L., Clark, M.R., Ramirez-Llodra, E., Currie, B., Smith, C.R., Sato, K.N. and Gallo, N. 2016. Defining “serious harm” to the marine environment in the context of deep-seabed mining. Marine Policy, 74, pp.245-259.

Levins, R. 1966. The strategy of model building in population biology. American Scientist, 54(4), pp. 421-431.

Levins, R. 1998. Qualitative mathematics for understanding, prediction, and intervention in complex ecosystems. Ecosystem health, pp. 178-204.

Link, J.S. and Browman, H.I. 2017. Operationalizing and implementing ecosystem-based management. ICES Journal of Marine Science, 74(1), pp.379-381.

Long, R.D., Charles, A. and Stephenson, R.L. 2015. Key principles of marine ecosystem-based management. Marine Policy, 57, pp.53-60.

Long, R.D., Charles, A. and Stephenson, R.L. 2017. Key principles of ecosystem-based management: the fishermen's perspective. Fish and Fisheries, 18(2), pp.244-253.

Maddox, D., Poiani, K. and Unnasch, R. 1999. Evaluating management success: Using ecological models to ask the right monitoring questions. Ecological Stewardship: A common reference for ecosystem management, 3, pp.563-584.

Mengerink K. 2018. Defining “Serious Harm” and “Harmful Effects” for Deep Seabed Mining in the Area. Chapter 16 In: Ocean Law Debates: The 50-Year Legacy and Emerging Issues for the Years Ahead, edited by Harry N. Scheiber, *et al.*, BRILL.

Murray CC, Wong J, Singh GG, Mach M, Lerner J, Ranieri B, Peterson St-Laurent G, Guimaraes A, Chan KMA. 2018. The insignificance of thresholds in environmental impact assessment: An illustrative case study in Canada. Environmental Management 61, 1062-1071.

Opdam, P.F.M., Broekmeyer, M.E.A. and Kistenkas, F.H. 2009. Identifying uncertainties in judging the significance of human impacts on Natura 2000 sites. Environmental Science & Policy, 12(7), pp.912-921.

Röckmann, C., van Leeuwen, J., Goldsborough, D., Kraan, M. and Piet, G. 2015. The interaction triangle as a tool for understanding stakeholder interactions in marine ecosystem based management. Marine Policy, 52, pp.155-162.

Smith, A.D M. 1994. Management strategy evaluation: the light on the hill. In: Hancock, D A ed, editor/s. Population Dynamics for Fisheries Management, Australian Society for Fish Biology Workshop Proceedings; August 24-25, 1993; Perth, WA. Perth, WA: Australian Society for Fish Biology; 1994. 249-253.

Smith, DC, Fulton EA, Bax N, Boxshall A, Dunstan P, Hayes K, Jordan A, Ward T. 2021. Implementing Integrated Ecosystem Assessments in Australia. A report from the Integrated Ecosystem Assessments Working Group prepared for Australia’s National Marine Science Committee.

Stelzenmüller, V., Fock, H.O., Gimpel, A., Rambo, H., Diekmann, R., Probst, W.N., Callies, U., Bockelmann, F., Neumann, H. and Kröncke, I. 2015. Quantitative environmental risk assessments in the context of marine spatial management: current approaches and some perspectives. ICES Journal of Marine Science, 72(3), pp.1022-1042.

Tetlock, P.E. and Gardner, D., 2016. Superforecasting: The art and science of prediction. Random House.

Thornborough, K.J., Juniper, S.K., Smith, S. and Wong, L.W., 2019. Towards an ecosystem approach to environmental impact assessment for deep-sea mining. Environmental Issues of Deep-Sea Mining: Impacts, Consequences and Policy Perspectives, pp.63-94.

Trenkel, V.M., Rochet, M.J. and Rice, J.C., 2015. A framework for evaluating management plans comprehensively. Fish and Fisheries, 16(2), pp.310-328.

Vierros, M. 2008. The ecosystem approach of the convention on biological diversity. *In* The ecosystem approach to fisheries (pp. 39-46). Wallingford UK: CABI.

Walters, C.J. and Holling, C.S., 1990. Large-scale management experiments and learning by doing. Ecology, 71(6), pp.2060-2068.

As Australia's national science agency, CSIRO is solving the greatest challenges through innovative science and technology.

Contact us

1300 363 400
+61 3 9545 2176
csiro.au/contact
csiro.au

For further information

CSIRO Environment
Dr Piers Dunstan
+61 3 6232 5382
piers.dunstan@csiro.au
research.csiro.au/dsm