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Contemporary marine science, its utility and influence on regulation and government policy

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Contemporary marine science, its utility and influence on regulation and government policy

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Editorial: Contemporary marine science, its utility and influence on regulation and government policy

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Editorial on the Research Topic

Contemporary marine science, its utility and influence on regulation and government policy

The purpose of this Research Topic is to evaluate the quality of contemporary marine science and to examine relationships between science, regulation and government policy in the marine environment. The quality of marine science matters; not just to advance knowledge on understanding marine ecosystems, but also to guide marine management. Marine environments are increasingly the location of a wide variety of human activities, all of which are subject to design- and risk-related research, and a range of applied science. Our motivation for hosting this Research Topic is a concern as to whether the most appropriate science exists and is being used to underpin regulation and policy in the most effective manner.

The range of papers received reflect the breadth and complexity of the topic. The types of marine development considered by the papers ranged from the generic to the specific with respect to offshore renewables, marine archaeology, fisheries, and ballast-water treatment. In this editorial, to describe these contributions, we use the various phases of an integrated marine programme designed to help achieve marine sustainability (Figure 1). Given known issues and overall purpose (including sustainability), the process begins with identifying general questions, followed by the choice of specific scientific questions to be answered (not just ‘addressed’). These questions then drive the choice of integrated services (scientific studies) to collect appropriate data and information, which together form the basis for an analysis and increased understanding of marine sustainability, and so around the circle again, improving understanding with each cycle.

Taken together, the Cormier et al. papers cover most aspects of this cycle. Cormier et al. focus on “science used for technical measures” that appear in codes of practice, guidelines and regulations (“practice, regulation and understanding”) asking whether the outcomes of technical measures meet the expected outcomes. Cormier et al.) focus on different scales at which ocean impacts might be viewed and how such scales inform legislation and administrative structures. They note that “marine management implies that the spatial

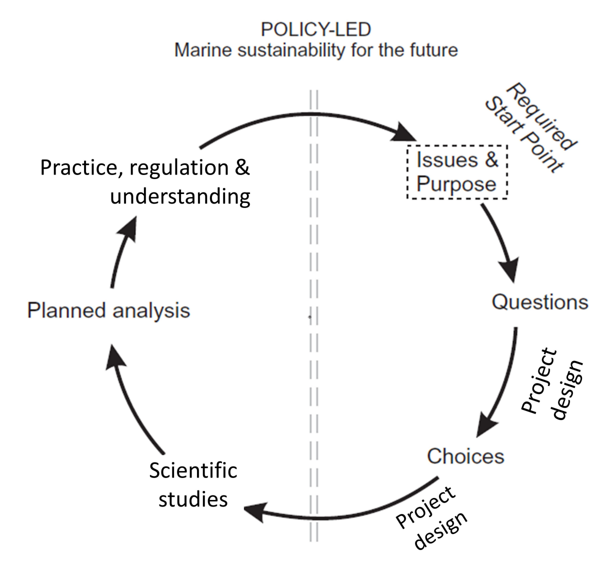

FIGURE 1
The policy cycle represented as key elements of an integrated marine programme designed to help achieve marine sustainability. (Source: modified from Morris et al., 2011, /doi.org/10.1016/j.marpolbul.2011.02.027).

and temporal scales of management are understood and built into prevailing legislation and administrative structures", a position expounded upon by Cormier et al.

The significant time lags involved in the processes of publication, scrutiny, and acceptance by the science community can obfuscate an understanding of these scales, so that regulations are always far behind the established science and further behind the latest evidence (Morris et al., 2011). Time lags can lead to collective inaction and might be part of the reason for the occurrence of Schwenkenbecher et al.'s "status quo" bias. Their work focuses on the philosophical underpinnings, including psychological traits, behind cases where there is limited empirical evidence to help form the specific purpose of work, and they include useful considerations of bias in evidence.

Such an environment where information is lacking is the deep sea, where the advent of likely mining activity is noted by Christiansen et al. The regulatory need to focus on 'baselines' leads the authors to the pertinent question of what defines a "baseline study", and they present suggestions for criteria to help assess the quality of scientific studies. In these deep-sea systems, natural variability is poorly known, so the authors indicate the primary need for spatial mapping and time-series measurements that include sediment cores, to help assess the various timescales of change. The choice of representative control sites and the significance of before-after comparisons depend entirely on such data, applying equally to shallower marine environments.

Similar points are made by Ward et al. regarding the need for greater awareness and integration of archaeology and cultural heritage management with marine sciences—especially the physical sciences. Issues of spatial variation and temporal change also arise here, especially concerning the project design and scientific study aspects of Figure 1, the key issues of Indigenous knowledge, and the importance of developing co-designed and -led projects and cultural management. On overlapping themes, Hewitt et al. describe a series of existing barriers to the effective use of science (in New Zealand), advocating that education of various sorts across society and within the relevant organisations is the prime avenue of improvement.

Although there is a large literature upon submarine pipelines, Griffiths et al. demonstrate that the default transfer by regulators of this literature for use in offshore renewable development (e.g., submarine cables) is inappropriate, posing a variety of attendant risks. Regulators need to commission work to develop relevant guidelines for this burgeoning industry to support understanding of the risks and ensure effective and defensible regulation.

The need for the updating of policy and regulation is a common theme in this Research Topic, with another example being the work of Gozzer-Wuest et al. The authors examined the priorities for fishing policy reform (in Chile) and the need for a national research agenda to improve fisheries management, finding that current laws and policies need updating. A similar conclusion is arrived at by Nie et al. in their assessment of the costs of compliance with different ballast water management policies.

Collectively the papers in this Research Topic showcase the important role that marine science in general and the scientist authors in particular can play in informing and guiding marine policy and practice. The relationship between science, regulation, and community acceptance is an ongoing issue warranting vigilance and ongoing attention. As the papers in this series indicate, ultimately rigour and credibility through evidence-based policy making is not only possible but essential in the quest for sustainable marine development and management.

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Managing Marine Resources Sustainably – The 'Management Response-Footprint Pyramid' Covering Policy, Plans and Technical Measures

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The plethora of human activities and their pressures and impacts in the oceans require managing at local, national, regional and international scales. This requires management responses in a programme of measures to determine (a) the area in which the human activities take place, (b) the area covered by the pressures generated by the activities on the prevailing habitats and species in which pressures are defined as the mechanisms of change, and (c) the area over which any adverse effects (and even benefits) occur on both the natural and human systems. The spatial and temporal scales of these leads to the concepts of activity-, pressures-, effects- and management responses-footprints, defined here. These footprints cover areas from tens of m^2 to millions of km^2, and, in the case of management responses, from a large number of local instruments to a few global instruments thereby giving rise to what is termed the management response-footprint pyramids. This may operate from either bottom-up or top-down directions, whether as the result of local societal demands for clean, healthy, productive and diverse seas or by diktat from national, supranational and global bodies such as the United Nations. These concepts are explained and illustrated using marine examples based on experience from many jurisdictions.

Keywords: DAPSI(W)R(M), UNCLOS, European Directives, technical measures, policy performance, regulatory equivalency

INTRODUCTION

Marine management, as with all environmental management, is implicitly or explicitly based on a cause-consequence-response framework whereby human activities then lead to consequences, as effects on both the natural system and the way society uses the natural system, which then need management actions to alleviate, reduce or remove those consequences. As a manifestation of this approach, Driver-Pressure-State-Impact-Response (DPSIR) frameworks have long existed to integrate the relationship between development and their pressures and impacts to the environment (Wascher, 1962). Over time, DPSIR has also been modified and refined into the most recent, and arguably a more complete, approach such as the DAPSI(W)R(M) (pronounced *dap-see-worm*) framework (Cooper, 2013; Patrício et al., 2016; Elliott et al., 2017). In this, **D**rivers of basic human needs and values (such as the need for food and recreation) need to be fulfilled by **A**ctivities (e.g. fishing, tourism) that create **P**ressures (e.g. seabed abrasion, pollution); in turn, those **P**ressures, as the mechanisms of change lead to **S**tate changes on the natural system (e.g. turbidity increase, oxygen depletion) and **I**mpacts (on human **W**elfare) for the human system (e.g. biodiversity loss, ecosystem services provision depletion). The **R**esponse (using management **M**easures), i.e. a policy response, then implies that society responds to those environmental and societal consequences (Elliott et al., 2017).

A policy response is very dependent on the context of a policy and the goals and objectives established by its governance processes; here we define governance as the combination of policies, politics, administration and legislation. The use of the term "policy response" may express the intent of international and national agreements such as United Nations agreements that are ratified by their member States and legislation enacted by national governments. A policy response may also express very specific procedures to be followed in emergencies such as marine accidents and oil spills as well as the prevention, reduction and control of pollution and other hazards to the marine environment. Currently, the interpretation of response (R) varies somewhat in the literature expressed as environmental policy goals and visions, marine plans objectives or the outcomes of technical measures (Cormier et al., 2017). A policy response as conventions and legislation is not the same as the implementation of a marine plan by a competent authority nor the conditions of licences and permits by a regulator. The term policy responses therefore is an integrated system of policies, plans and measures to address goals and objectives established by national governance structures and implemented through management and regulatory processes (Elliott et al., 2020b).

The management of maritime activities is the integration of environmental and development objectives generated through marine planning processes across sector management of their respective activities which should also integrate protection and conservation strategies (Stephenson et al., 2019). In contrast, marine planning processes is the vertical integration of environmental and socio-economic policies as mandated by the national governance structures (Cormier et al., 2019). However, national public policymaking processes also have to integrate obligations established through regional and global governance processes such as European directives and United Nations conventions. The complexity of the marine environment and its management requires horizontal and vertical integration – horizontal integration is across all of the various activities (e.g. fishing, aquaculture, navigation, etc.) whereas vertical integration goes from local and immediate to global and long-lasting. Vertical integration between policies, plans, and technical measures from the local to the global is key to achieving policy objectives and sustainability goals with the understanding that these are achieved through effective and reliable technical measures dealing with the specific activities and their impacts (Stephenson et al., 2019). Therefore, this system of global, regional and national policies, plans and technical measures implemented through treaties, conventions, agreements legislation and regulatory frameworks were developed within the scope of different organizations that framed the context of their policies (Elliott et al., 2020b). It is contended here that there is a poor understanding of the vertical integration of global, regional, and national policy responses and the links to their implementation through marine plans and technical measures. Therefore, here we explore the need for a clear link between the different levels of policy responses to ensure that marine plans and the technical measures used to manage maritime activities are effective and informed by relevant and fit-for-purpose natural and social sciences (Elliott et al., 2020b). For example, this is where a regulator establishes conditions as part of a project approval process to address the natural and societal effects identified in an Environmental Impact Assessment (EIA).

Previously, we proposed that an activity in the marine environment and its contribution to pressures and effects (both on the natural and human systems, i.e. both the S and the I(W) in the DAPSI(W)R(M) framework) could be organized in terms of their 'footprints', i.e. the area and/or time covered by the activity, pressures and effects (Elliott et al., 2020a). We consider that this structure provides a more practical understanding that management actions within the activity-footprint are most effective at addressing pressures that are the root causes of effects. Activities, pressures and effects have overlapping footprints but that because of the dynamic nature of the marine environment then the pressures-footprints will be larger than the activity-footprint and the effects footprint will be larger and longer-lasting still. Therefore, such a structure also helps understand the spatial and temporal causal scales of activity-pressure-effects and in turn is needed to decide what management responses are required to address the activity, pressures and effects. In turn, those management responses are needed to address hazards from anthropogenic and natural sources which occur in the marine environment and that can become risks to nature, property, human health and livelihoods (Cormier et al., 2019; Elliott et al., 2019). Hence, those hazards and their risks need to be addressed through technical measures that avoid and control their causes or mitigate and compensate their consequences.

Policies regarding resource sustainability and conservation most often are developed to address environmental effects out of concerns for human well-being, such as providing sustainable and safe sea foodstuffs (Elliott et al., 2017). From such policies, the administrations and statutory bodies, i.e. those implementing legislative instruments and agreements, develop marine plans with objectives to reduce the risks of such effects from the pressures generated by activities in the marine environment which are then integrated in regulatory and non-regulatory tools used to manage those activities (Cormier et al., 2017; Gorjanc et al., 2022). Therefore, here we explore the contention that the management response measures also have a spatial extent and/or temporal duration that can be described as a footprint – i.e. the compound term the management response-footprint. The footprints of these management responses (the R(M) in the DAPSI(W)R(M) framework), by necessity, have to reflect the footprints of the activities, pressures and effects (Elliott et al., 2020a). However, the footprint of the responses are also constrained by jurisdictional boundaries and even the areas beyond such jurisdiction but because of the dynamic marine nature, they do not necessarily align with the footprints of the effects and pressures that can be addressed through measures on the activity that then generate them (Verlaan, 2021; Cormier and Minkiewicz, 2022).

Hence, we aim to show firstly, that responses in terms of policies, marine plans, and technical responses do not necessarily have the same footprint. Secondly, the lack of a clear understanding of the hierarchy of management response-footprints that are developed and implemented by different actors are thereby creating a fragmented system of marine management. Based on the insights from the activity-, pressures- and effects-footprint definitions (Elliott et al., 2020a), we define the management response-footprint and demonstrate the importance for understanding the hierarchy of policy, marine plans and technical measures responses from a global, regional, national and local footprint perspective. Finally, we emphasise the summary of these ideas using the concept of two related 'management response-footprint pyramids' as the underlying framework and hierarchy of marine management; a spatial management response-footprint pyramid reflects management responses from very local scales to global scales and a pyramid reflecting the very large number of local management instruments (indeed, for example, one for each activity) feeding through a hierarchy to very few global instruments. By presenting an understanding of the complexities and differences in terms of policymaking approaches and capacities across national jurisdictions, we hope that this response-footprint concept will help to improve our understanding of the hierarchy between policies, marine plans and technical measures in relation to global, regional and national footprints. Here we put more emphasis on the spatial nature of these footprints than their temporal nature given that the spatial coverage is the precursor to long-term marine management.

DEFINITIONS OF FOOTPRINTS

As a starting point and to place this discussion in context, it is necessary to suggest definitions of the various types of footprints; note that those for the activity-, pressures- and effects- are modified as shown in **Table 1**.

UNDERSTANDING THE FOOTPRINTS OF POLICIES, MARINE PLANS AND TECHNICAL MEASURES

Marine policies and management measures can be derived either top-down, from international, regional, or national diktats, or bottom-up by demands from those being managed or from the different groups of stakeholders (Newton and Elliott, 2016). For example, these can range from international agreements such as the United Nations (UN) Sustainable Development Goals (SDGs) (UN, 2016) to the requirement by regulators, local communities and pressure groups for management measures on new marine industrial development such as an offshore wind-farm. The footprints of policies, marine plans, and technical measures are closely linked to the boundaries and mandates of their respective governance processes (**Figure 1**). Here, we propose five management response-footprints reflecting different governance, administration and regulatory processes showing the importance of top-down vertical integration from the global policy response-footprint that is needed to ensure an effective bottom-up vertical integration at the technical response-footprint to ultimately achieve global policy objectives (Cormier et al., 2017; Cormier et al., 2019; Stephenson et al., 2019).

TABLE 1 | Definitions for activity, pressures and effects footprints (adapted from Elliott et al., 2020a).

Activity-footprint	The area and/or time, based on the duration, intensity and frequency of an activity which ideally has been legally sanctioned by a regulator in an authorisation, licence, permit or consent, and which should be so clearly defined and mapped in order to be legally-defendable; it should be both easily observed and monitored and attributable to the proponent of the activity.
Pressures-footprint	The area and time covered by the mechanism(s) of change resulting from a given activity or all the activities in an area once avoidance and mitigation measures have been employed (the endogenic managed pressures). It does not necessarily coincide with the activity-footprint and may usually be larger but could be smaller. It also needs to include the influence and consequences of pressures emanating from outside the management area (the exogenic unmanaged pressures); given that these are caused by wide-scale events (and even global developments) then these are likely to have larger scale (spatial and temporal) consequences.
Effects-footprint	The spatial (extent), temporal (duration), intensity, persistence and frequency characteristics resulting from (a) a single pressure from a marine activity, (b) all the pressures from that activity, (c) all the pressures from all activities in an area, or (d) all pressures from all activities in an area or emanating from outside the management area. They include both the adverse and positive consequences on the natural ecosystem components and on the ecosystem services and societal goods and benefits. They need to include the near-field and far-field effects and near- and far-time effects because of the dynamics and characteristics of marine areas and the uses and users of the area. They may be larger in extent and more persistent than the causing activity-footprint and the resulting pressures-footprints. They also need to encompass the effects of both endogenic and exogenic pressures operating in that area.
Response-footprints	The area and time covered by the governance means of monitoring, assessing and controlling the causes and consequences involved in the use of the marine environment through public policy-making, marine planning and regulatory processes. The policies, marine plans and technical measures produced by these processes indicate the means of determining if legal controls are satisfied, and of providing information and data to national and supra-national bodies. They focus on the area and/or time covered by the marine management actions and measures (e.g. programme of measures), including the distribution and range of a species.

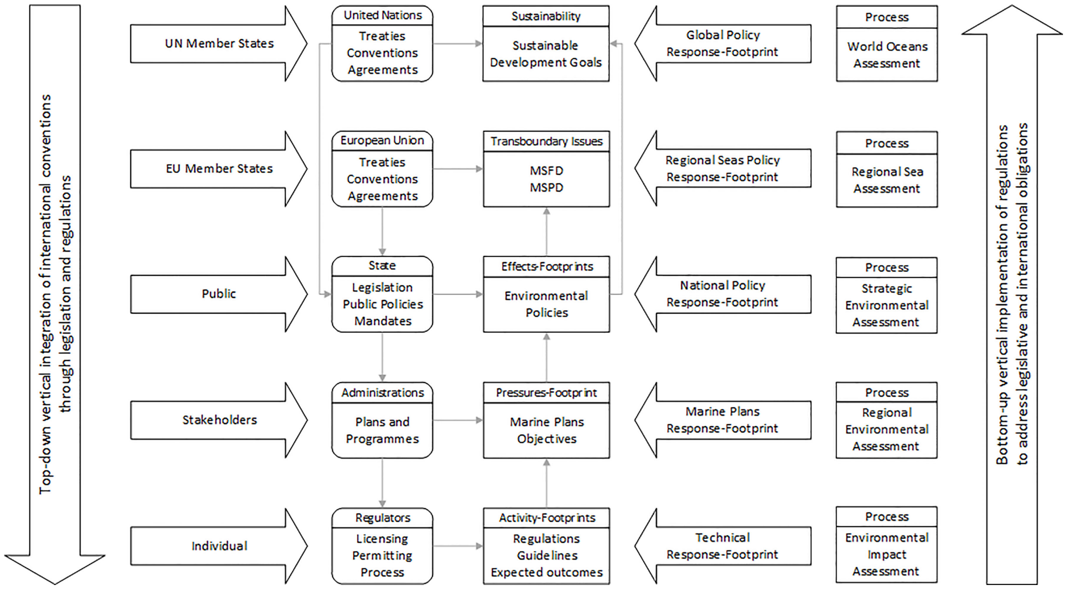

FIGURE 1 | The vertical link between policies, plans and technical response-footprints. Key for examples provided: MSFD, Marine Strategy Framework Directive; MSPD, Maritime Spatial Planning Directive; UN, United Nations; EU, European Union.

- ***Global policy response-footprints*** are the goals and objectives outlined in treaties, conventions and agreements such as those that are ratified and implemented by UN Member States legislation to fulfil in good faith their obligations. Within the spirit of international peace and sovereign equality, territorial integrity and political independence of its members under the UN Charter (UN, 1945), the role of the organizations and agencies of the United Nations is to coordinate and facilitate the negotiations and drafting of treaties, conventions, and agreements as directed by the UN Member States. In the marine environment, the United Nations Convention on the Law of the Sea (UNCLOS) (UN, 1996) defines the spatial boundaries of the sovereignty of Coastal States regarding the physical and biological resources as well as the right of innocent passage of any State Party to UNCLOS that does not have a coast - State Parties are those that have ratified the UNCLOS. UNCLOS also establishes the accountabilities of any State Parties regarding marine activities in the high seas from vessels flying their respective flags. UNCLOS is highlighted here because it ultimately frames the footprints of global policy responses of many other UN environmental instruments such as the Convention on Biological Diversity (CBD) (UN, 1992) as well as the International Convention for the Safety of Life at Sea (SOLAS) (UN, 1974) and the International Convention for the Prevention of Pollution from Ships (MARPOL) (UN, 1973). Although there are many assessments and scientific panels involved throughout these UN organizations and agencies, the World Oceans Assessment II (UN, 2021) is listed here as an example of the type of assessment that informs such global governance processes.
- ***Regional seas policy response-footprints*** are similar to the goals and objectives outlined for the global policy response-footprint. However, their treaties, conventions and agreements are applicable to specific and often designated regional marine areas even though these are still signed by member States as contracting parties. In the marine environment, the European Union (EU) directives are examples of regional seas governance processes that are legally-binding for the EU Member States. In the marine environment, directives such as the Marine Strategy Framework Directive (MSFD) (European Union, 2008; European Union, 2017) and the Maritime Spatial Planning Directive (MSPD) (European Union, 2014) are to be implemented within the context of the Regional Seas Conventions (RSC). Similar to ratification and implementation of UN instruments discussed above for UN Member States, such EU regional responses require the transposition of directives into national regulations for EU Member States and Acts of Parliament in non-EU States framing the regional seas policy response-footprint. The MSFD requires that EU Member States undertake an initial assessment of their marine waters to ultimately identify the programmes of measures which need to be taken in order to achieve or maintain good environmental status (Borja et al., 2013). In a wider European context, these regional instruments also include the Oslo and Paris (OSPAR), Helsinki (HELCOM), Barcelona (UNEP-MAP) and Bucharest Regional Seas Conventions (RSC) for the North-East Atlantic Ocean, the Baltic Sea, the Mediterranean and the Black Seas, respectively. As an indication of the reach of the regional agreements, the UNEP (UN Environment

Programme) Regional Seas Programme encompasses 3 types of Regional Seas Conventions and Action Plans (RSCAPs) (https://www.unep.org/explore-topics/oceans-seas/what-we-do/regional-seas-programme). This includes 18 different regions and other action plans: the UNEP-administered ones established and administered by UNEP include: the Caribbean Region, East Asian Seas, Eastern Africa Region, Mediterranean Region (Barcelona), North-West Pacific Region, Western Africa Region (with Regional Office for Europe administering the Tehran Convention for the Caspian Sea); the Non-UNEP administered ones were established by UNEP but have different secretariat bodies, including the: Black Sea Region (Bucharest), North-East Pacific Region, Red Sea and Gulf of Aden, ROPME Sea Area, South Asian Seas, South-East Pacific Region, Pacific Region, and thirdly the Independent ones not established by UNEP but cooperating with the RSC: Arctic Region, Antarctic Region, Baltic Sea (HELCOM), North-East Atlantic Region (OSPAR). These include both developed and developing countries and those with long and short histories of managing their sea region (for example the Baltic and North-East Atlantic RSC were established in the early 1970s) and therefore they give the more recent ones and those areas with a lesser capacity the chance to learn from the other RSCAPs. Most importantly, the RSC requires signatories to carry out the monitoring, assessment and reporting of the status of their marine environments. While the RSC requirements are not legally binding, the signatories have agreed to their implementation and there is an arbitration process for disputes between country signatories. Again, they require to be implemented through the national regulations and instruments of a signatory country. The regional Quality Status Reports produced by the RSC such as OSPAR and HELCOM give excellent examples of integrated marine assessments.

- ***National policy response-footprints*** are reflected by the legislation and policies that are developed through national policymaking processes within established jurisdictional boundaries of a State. Coastal States that have ratified and implemented UNCLOS may have different jurisdictional configuration that may typically start from the normal baseline up to the 12 nm for territorial seas and may include another 24 nm for the contiguous zone and outwards to the 200 nm (or the mid-line between adjacent states) for the exclusive economic zone (EEZ) that may be extended to the continental shelf. In the high sea, the jurisdiction of any State extends to any vessel or infrastructure flying their flag. For example, the United Kingdom (UK) has different jurisdictions and competent authorities that can take management actions within different boundaries such as the areas from 3, 6 or 200 nautical miles (nm) (Boyes and Elliott, 2015). National policy responses reflect public values and objectives expressed through policymaking processes that can follow very different national governance structures and are limited to the activities that occur within the boundaries of their jurisdictions. National legislation and policies are also needed to fulfil the obligations of global and regional policies. National policy response-footprints tend to be influenced by a wide range of concerns such as water quality, productivity or cumulative effects which can be more or less aligned with the effects-footprint (Elliott et al., 2020a). Within the context of regional seas policy response footprints, any transboundary issues between two jurisdictions would ultimately need some form of agreement to resolve these issues within their respective legislative authorities and policies. At this level, there are many examples of strategic environmental assessments used to assess the wider effects of plans and programmes on the environment (Weiland, 2010; Noble and Nwanekezie, 2017; Rehhausen et al., 2018). Indeed, as shown by the European Strategic Environmental Assessment Directive, SEAs are processes covering a regional area and designed to inform policy decisions in contrast to EIA that are processes to inform regulatory decisions. As indicated below, regulatory decisions include technical measures to address the impacts of the activities both singly and cumulatively.
- ***Marine plans response-footprints*** are the plans and programmes that are developed and implemented by administrations having received a mandate from their governments. These plans may outline strategies for new maritime activities, spatial allocation for many maritime activities including protection and conservation strategies for the marine environment. In consultation with relevant stakeholders, their planning and management processes are conducted within jurisdictional boundaries that frame the marine plans response-footprints established by their legislation within the national policy response-footprint. For example, in England, the Inshore Nature Conservation and Fisheries Authorities (IFCAs) cover fisheries to 6 nm whereas the Marine Management Organisation operates to 200 nm, and Natural England manages conservation to 12 nm whereas the Joint Nature Conservation Committee covers to 200 nm. Maritime spatial planning and integrated coastal and oceans management are also examples of marine plans response-footprints legislation (Scotland, 2012; Canada, 2019). Marine plans response-footprints may but not necessarily overlap in whole or in part with the pressures-footprints (Elliott et al., 2020a). There is a wide variety of regional environmental assessments produced to inform these processes for different purposes and environmental contexts such as ecosystem overviews and assessment reports (DFO, 2005), integrated ecosystem assessments (Diekmann and Möllmann, 2010), or the State of marine ecosystem reports (Bernier et al., 2018; Devlin et al., 2019).
- ***Technical response-footprints*** are the technical measures that are implemented through regulatory and non-regulatory frameworks to manage specific operations of an activity undertaken by an individual or a corporate entity. As part of a regulatory framework, technical measures are implemented as regulations, standards, standardized operating procedures that regulate and control a variety of impacts from individual activities (e.g. physical changes to

habitats, contaminants to mitigate pollution effects, biological disturbance to species life cycle, etc.). Technical measures dealing with continuous, often daily, tasks such as inspections and maintenance, monitoring and reporting, incident response plans and corrective actions are implemented through non-regulatory frameworks such as guidelines, codes of practice, good industry practices, etc. These response-footprint are also tightly linked (and of the same size and duration) to the activity-footprint that has been sanctioned by a regulator and regulatory approval processes including certifications (Elliott et al., 2020a). Such footprints are typically informed by an environmental impact assessment (EIA) and its environmental statement that scope the ecological, cultural, social and economic impacts for which the individual or corporate entity is accountable to address through their regulatory approvals issued by regulators (Elliott et al., 2020b). An EIA is very prescriptive being tied to the impacts of a particular development, at a specified time and place, performed in a given way with certain mitigation and communicated widely. Although an EIA must be carried out as part of the regulatory approval process, the identified impacts are then used by regulators to establish the technical measures as conditions for licensing or permitting how, where and when, for example, a land-based discharge, a sea-based dumping site or a marine oil and gas operation is to be undertaken (Lonsdale et al., 2015; Lonsdale et al., 2017). Each technical measure is designed to produce a specific expected outcomes to avoid, reduce, compensate or offset impacts within a mitigation hierarchy (Arlidge et al., 2018; Duarte and Sánchez, 2020). For example, technical measures would be implemented to fulfil the expected outcomes of the input controls, the spatial and temporal distribution controls and the output controls of the programmes of measures of the MSFD.

VERTICAL COHERENCE OF POLICY INTEGRATION VERSUS EQUIVALENCY OF REGULATORY IMPLEMENTATION

As a management system, policies are conditional on the performance of plans and programs that are in turn conditional on the effectiveness and reliability of the technical measures implemented for specific activities and their impacts (Cormier et al., 2018; Elliott et al., 2020b). Vertical and horizontal policy integration is imperative to implement local to global ecosystem management strategies in the marine environment (Rosendo et al., 2018; Kidd et al., 2020; Winther et al., 2020). Vertical integration encompasses policy and management responses from the global to the local whereas horizontal policy integration operates and integrates across the sectors and activities (fishing, aquaculture, navigation, recreation, etc.) (Boyes and Elliott, 2014). Although maritime spatial planning is considered as a key to policy integration, integration, in practice, depends on the context of the policy objectives involved such as sustainable development, ecosystem-based management or marine protected areas (Zaucha and Gee, 2019). Integration may be applied to decision-making and planning processes, risk assessments and management or stakeholder consultation and participation (Lombard et al., 2019). Thus, vertical and horizontal integration is still necessary but difficult to achieve because of capacities needed for planning processes including the governance structures and decision-making processes in a given national context (Cormier et al., 2019; Stelzenmüller et al., 2021).

Top-down vertical policy integration across the response-footprints implies that global and regional policy responses have to be integrated in the development of national policy responses (**Figure 2**). After the treaties, conventions or agreements have been ratified and signed, it is up to the member States or contracting parties to take the *actions* necessary to implement these as their national

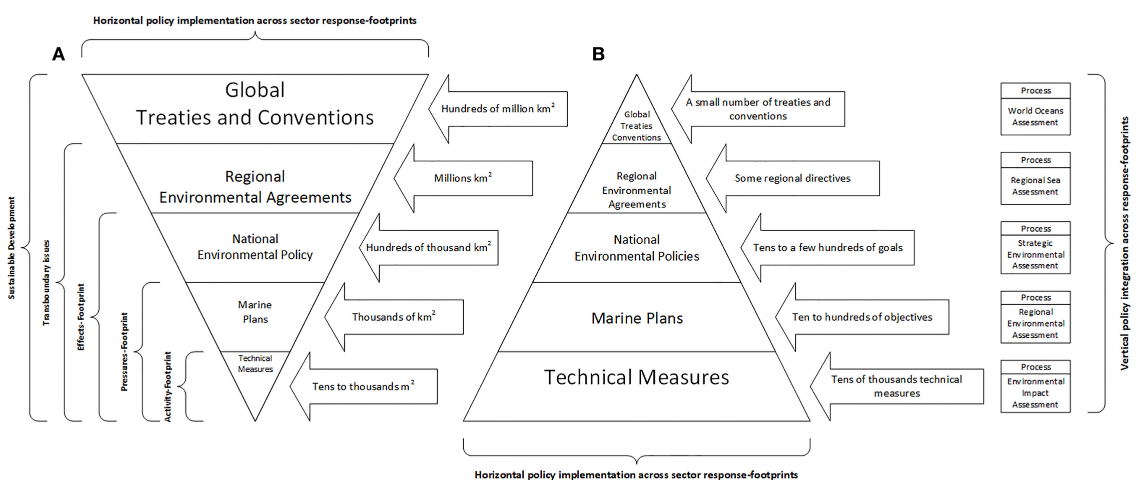

FIGURE 2 | The 'management response-footprint pyramid' – showing both **(A)** the areas covered by management response-footprints and **(B)** the number of policy instruments; horizontal and vertical policy integration is also denoted.

policy responses. It is subsequently the administrations that have to take the necessary *actions* to initiate marine planning processes to develop marine plans to address their respective national policy responses. The ultimate *action* is taken by the regulator to identify the technical measures to address the objectives of marine plans that start the bottom-up policy implementation across response-footprints (Stephenson et al., 2017).

Therefore, the linking and illustration of the size of the management response-footprints takes the form of a pyramid which can be presented in either the inverted or standard form (**Figure 2**) but which respectively indicate that (a) local management initiatives (such as an EIA) may cover a small area or a short timeline whereas regional and then international/global initiatives cover larger areas and timelines, and (b) that there are many statutory instruments or agreements at local levels leading up to a few global agreements. We have termed this 'the management response-footprint pyramid' (Ruini et al., 2015). As discussed above, **Figure 2** also indicates that marine management responses have to be integrated horizontally (across the width of the pyramid) and vertically (up the height of the pyramid). Whether the pyramid is then being used to determine top-down or bottom-up management results in the pyramid being inverted or the usual way around, one would expect that the footprints of these responses are spatially integrated into one another to ensure coherence across the responses by each level of governance (Inverted pyramid **Figure 2A**).

In essence, global and regional treaties, conventions and agreements have a response-footprint that can span millions km^2 of global oceans to achieve the UN SDG 14 "Life below water" (Cormier and Elliott, 2017). These overlap with or encompass in whole or in part hundreds of thousands km^2 of national policy responses-footprints such as the EEZ of a State or the territorial waters. This is further exacerbated by the footprint of marine plans that can span thousands of km^2 such as the maritime spatial plan for the southern North Sea or the Belgian Shelf area (Elliott et al., 2020a). At the smallest scale and much more locally there is the footprint of the technical responses that deal with tens to thousands m^2 such as the conditions established in licences and permits that are informed by an EIA (e.g. for an offshore windfarm or an aggregate extraction area).

The integration challenges of each footprint are also influenced by the individual governance, management and regulatory processes involved in each footprint that are not conducted by the same authorities and within the same time frame. Most if not all maritime States have a plethora of marine management organisations and statutory bodies, often with overlapping mandates and competences (Boyes and Elliott, 2015). Based on the premise that sustainable development and transboundary issues are ultimately addressed through national policy responses within the footprint of their jurisdictions (ICES, 2021; Cormier and Minkiewicz, 2022), any given State can only address global and regional policies within the footprints of their legislative, policymaking, marine planning and regulatory processes. Without the collaboration of multiple coastal States within a regional sea, a State can only address the environmental impacts, pressures and effects that occur within their national policy response-footprints (Elliott et al., 2020a). In cases where pressures- and effects-footprints overlap across Coastal State boundaries, horizontal integration of policies is dependent on the level of policy coherence across national policy response-footprints (Elliott et al., 2020b).

A large number of technical measures implemented through regulatory and non-regulatory frameworks are used to reduce the impacts within a specific activity-footprint (**Figure 2B**). As discussed above, an EIA identifies the impacts to subsequently identify the technical measures needed to minimise the size and duration of their impacts and, ultimately, the pressures and effects they may collectively generate. Given that operating licences, authorisations and permits are based on sector planning permissions, the technical measures relate mainly to the activity itself while the pressures and effects they generate often disperse across jurisdictional boundaries and persist for as long as the activity operates (Trendall et al., 2011; Borgwardt et al., 2019). For example, an individual dredging programme requires a permit that is issued by a national competent authority to the dredging company. The Member State of this competent authority is then required to report this under the relevant RSC giving the contaminant levels in the dredged material, the quantities of sediment moved and the ability to meet quality standards (Alvarez-Guerra et al., 2007). Even though there might be very good coherence in the vertical integration of policies across global, regional and national footprints as well as in the horizontal integration across marine plans, there might not be any equivalencies of the technical measures used across national regulatory and non-regulatory frameworks for the impacts of an activity within the context of transboundary pressures and effects (Cormier et al., 2017). Continuing with this example, the RSC and the UN Member State that ratified the London Convention and Protocol for dumping at sea may not be able to ensure effective control of marine pollution from dumping of wastes and other matter and ultimately address the targets outlined for the United Nations SDG14 (life below water) (UN, 2016).

While the technical response-footprints may only apply to tens to thousands of m^2, there are many technical measures that are implemented through regulatory and non-regulatory frameworks across jurisdictions (**Table 2**). In addition, the performance of marine plans and the success of national policies depend on the implementation of effective and reliable technical measures that provide equivalent levels of protection across their respective activity-footprints (Cormier et al., 2018; Murillas-Maza et al., 2020). Given the challenges of technical equivalencies across jurisdictions, the global and regional governance processes would require effective collaboration to promote equivalency of technical measures to address transboundary issues as dedicated by global and regional policy responses such as the case for MARPOL and SOLAS (Cavallo et al., 2018).

VERTICAL INTEGRATION OF ENVIRONMENTAL ASSESSMENTS ACROSS RESPONSE-FOOTPRINTS

In recent decades, policies in the form of treaties, conventions, agreements, legislation, plans and programmes have evolved into a

TABLE 2 | Examples of regulations, codes of practice, and guidelines as technical response-footprints.

Technical responses	Type of response	Authority
Wastewater Systems Effluent Regulations	Deleterious effect to fish regulations	*Fisheries Act* https://laws-lois.justice.gc.ca/eng/acts/f-14/
Potato processing plant liquid effluent regulations	Deleterious effect to fish regulations	*Fisheries Act* https://laws-lois.justice.gc.ca/eng/acts/f-14/
Disposal at sea regulations	Pollution prevention regulations	*Canadian Environmental Protection Act* https://laws-lois.justice.gc.ca/eng/acts/c-15.31/
Persistence and bioaccumulation regulations	Pollution prevention regulations	*Canadian Environmental Protection Act* https://laws-lois.justice.gc.ca/eng/acts/c-15.31/
Environmental code of practice for metal mines	Complete life cycle of mining	Environment and Climate Change Canada https://www.ec.gc.ca/lcpe-cepa/documents/codes/mm/mm-eng.pdf
Canadian environmental quality guidelines	Quality of aquatic and terrestrial ecosystems	Canadian Council of Ministers for the Environment https://ccme.ca/en/summary-table
New-Brunswick watercourse and wetland alteration regulations and guidelines	Manage the operations of an activity	*New Brunswick Clear Water Act* https://www.canlii.org/en/nb/laws/regu/nb-reg-90-80/latest/nb-reg-90-80.html https://www2.gnb.ca/content/dam/gnb/Departments/env/pdf/Water-Eau/WatercourseWetlandAlterationTechnicalGuidelines.pdf

complex structure of instruments that reflect global, cultural, social, economic, and environmental concerns (as in the horrendogram given in Boyes and Elliott, 2014). Having been developed independently, policies have become issue- and concern-centric that have likely contributed to the so-called *fragmentation* of policy responses to broader environmental issues (Raakjaer et al., 2014; Michanek et al., 2018). Given the need for scientific knowledge and advice, these issue- and concern-centric policies have also framed the science produced to inform those policymaking processes independently for each response-footprint. There are many forms of environmental assessments (**Table 3**) but this leads to what may be called the environment assessment paradox – ‘that there are more and more environmental initiatives requiring assessments but there is less funding for achieving them (or the funding is put onto industry)’ (Borja and Elliott, 2013; Strong and Elliott, 2017; Borja and Elliott, 2021).

The performance of policy responses and their footprints span significant spatial and temporal scales (**Figure 3**). Based on the premise that societal goals and objectives, and indeed the vision for our seas, rely on regulatory frameworks (Elliott et al., 2020b), the performance of global, regional, national marine policy and management responses ultimately rely on the technical measures of the regulatory frameworks to approve and regulate activities within their technical response-footprints. For example, defining and controlling the footprint of an offshore windfarm and considering its pressures and effects based on national legislation are only as good as our ability to (a) carry out the EIA, (b) ensure that there are effective and reliable technical measures to mitigate and/or compensate the impacts on nature and society, and (c) check *a posteriori* that the predictions of impact and the effectiveness of the technical measures were accurate (i.e. the management measures really did address the impacts which occurred). Paired with fragmented scientific advice related to management and operational implementation (DFO, 2014), ineffective and unreliable technical measures used to regulate the root-causes of environmental pressures and effects contribute to the uncertainties of achieving national, regional and global goals and objectives and ultimately, the performance of their policy response-footprints (Cormier et al., 2018). Given that risk is defined as the effect of uncertainty on objectives (ISO, 2018;

TABLE 3 | Examples of Environmental Status Assessments.

Intent of the instrument	Instruments
Catchment quality	• EU Water Framework Directive • US Clean Water Act
Habitat and species conditions	• EU Habitats Directive • Canada National Marine Conservation Act
Marine regional quality	• EU Marine Strategy Framework Directive • US Oceans Act • Canadian Ocean Act
Cumulative impacts and effects assessments	• EU Cumulative Impact Assessment Directive • Canadian Impact Assessment Act • Canadian Fisheries Act
Strategic environmental assessments	• EU Strategic Environmental Assessment Directive • Canadian Impact Assessment Act
Environmental impact assessment	• Environmental impact assessment legislation worldwide
Regulations and codes of practice for industry and marine activities	• Canadian Environmental Protection Act regulations • Fisheries Act regulations

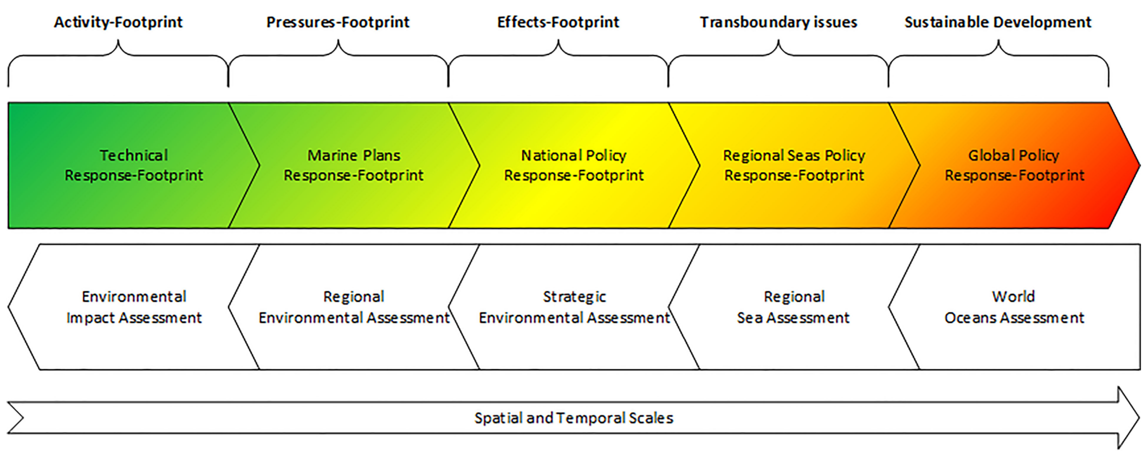

FIGURE 3 | Vertical integration of assessments.

IEC/ISO, 2019), a preventive risk management strategy of the root-causes of risk carries the least uncertainty in achieving objectives while a reactive risk management strategy of the consequences of risk carries the most uncertainty in achieving objectives. Thus, effective and reliable technical measures implemented within the technical response-footprint for an activity carries the least uncertainties in achieving objectives of national, regional and global policy responses (Green to red colour in **Figure 3**). However, relying on global, regional and even national policy responses in reaction to issues and concerns carries the most uncertainties in achieving objectives (Red to yellow transition in **Figure 3**). In risk management, controls are implement to prevent the causes of risk and mitigate the consequences of risk effectively reducing the uncertainty of achieving objectives.

As discussed above and illustrated in **Figure 2** for vertical policy implementation across response-footprints, this also requires top-down vertical integration of the science and knowledge generated across the assessments to ultimately provide the context for an EIA at the technical response-footprint level (**Figure 3**). An assessment conducted for a given policy response should inform the responses that are and can be implemented for the specific footprint. Thus, there is as much need for top-down vertical integration of the scientific knowledge as there is advice generated by the assessments within each response-footprint. For example, there is an increasing number of methods for ocean status assessments (Borja et al., 2016). One would expect that a world oceans assessment conducted for a global policy response could inform the context for regional sea assessments to integrate the relevant global knowledge in such assessments (as in UN, 2021). Not all global pressures and effects can be dealt with at a given regional policy response; for example, climate change adaptation requires global initiatives such as Paris COP but as the coordination of national initiatives with global science. The same can be said of a regional sea assessment where regional pressures and effects cannot necessarily be dealt with one national policy and marine plan responses, as happens in the case of the MSFD in the different regional and sub-regional seas (Borja et al., 2019).

FACTORS AFFECTING THE RESPONSE-FOOTPRINTS

There are many factors that can facilitate as well as impede the integration of the response-footprints (**Figure 4**). For example, global policy response-footprints in the high seas depend on the cooperation of UN Member States that are parties to UN conventions and agreements to find solutions (Blaustein, 2016). In essence those States need to coordinate their activities to protect and preserve the marine environment and its biological diversity as well as address demands from their constituents (UNCLOS and CBD). These goals especially depend on the coherence of marine policies and plans within the boundaries of regional sea and the capacity of coastal States to ratify and transpose such policies into national legislation and/or regional legislation as in the case of EU Directives. Even when UN conventions and agreements are ratified and implemented by UN Member States through their legislation, the performance of national plans and programmes depends on the mandates and the complexities of national competent authorities to lead and facilitate marine planning processes across internal jurisdictions. It also requires that stakeholders have the capacity to deal with the collective pressures generated by multiple marine activities. Ultimately, the success of global and regional policy responses depends on the performance of national policy responses, the integration of marine plans and the equivalency of technical measures implemented across national jurisdictions including the high seas. As discussed for **Figure 3**, it is the equivalency of technical measures implemented across the technical response footprints of multiple jurisdictions that carries the least uncertainty in achieving global goals and objectives such as the UN Sustainable Development Goals shown as an example in **Figure 4**.

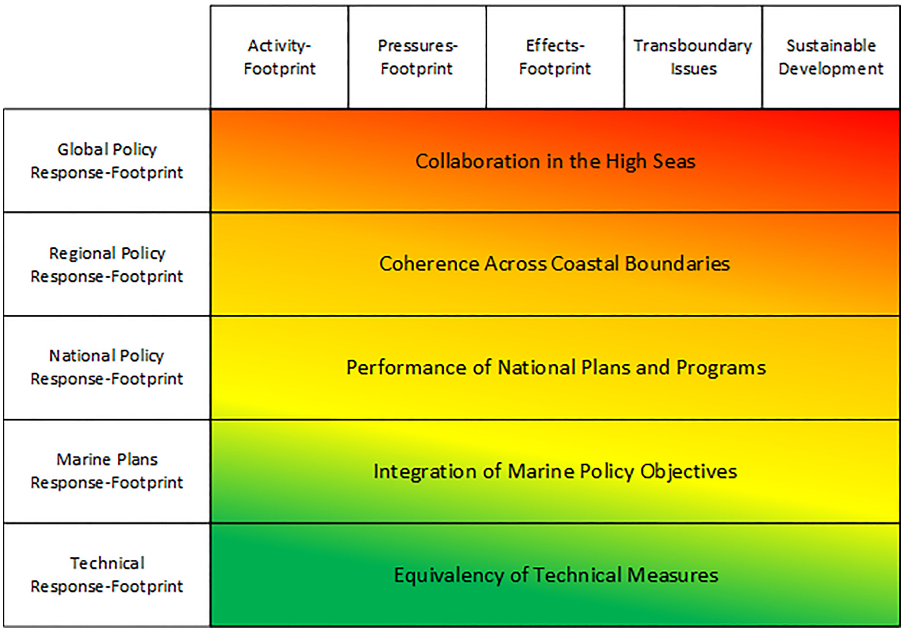

FIGURE 4 | Factors affecting the efficacy of the response-footprints.

Although the response-footprint framework discussed here is primarily focused on a structured governance processes of policymaking, marine planning and regulatory approval, the capacity of the governance and administrative systems of States have to be acknowledged. Following the principles of sovereign equality, territorial integrity and political independence of UN Member States (UN, 1945), we have to recognize that States have inherently different political and policymaking processes, legal and administrative systems that may or may not reflect the structure of the framework discussed here.

As examples, **Table 4** links activity-, pressures- and effects-footprints with the management response-footprints according to the specific technical measures implemented to address marine plans objectives and ultimately national, regional and global policy responses. A technical response-footprint is much more specific compared to broader goals established for regional and global response-footprints. For example, a national law may only control sea dumping of dredged material whereas the global goal may be to protect the whole marine system and, under the principle of subsidiarity, whereby decisions should be taken as close to the population as possible, devolving decisions to the lowest practical political level, leave the precise mechanism of achieving this to the State. At a lower level, this is analogous to the EU setting a framework directive, such as the MSFD, and then leaving the precise implementation to an EU Member State.

DISCUSSION

The residual impacts of each activity generate pressures that are specifically tied to its precise, often daily operations and ultimately contribute to the effects on the natural and societal systems (respectively the State change and Impacts (on human Welfare) under DAPSI(W)R(M)). The activity-footprint may be located on land, in rivers and lakes and still generate pressures in these adjacent estuarine, coastal and marine environments (Borgwardt et al., 2019). For example, agriculture, urban and industrial developments create diffuse and point source emissions which then create pressures and effects far from their source. As such, the amount of pressures to estuaries, coastal zones and the seas are highest from land-based, estuarine, and coastal activities. Some of these pressures ultimately disperse to the marine environment causing effects at multiple ecosystem scales (Borgwardt et al., 2019). It is axiomatic that in developed countries, and many developing countries, any activity that has the potential to adversely affect the environment needs legally-enforced conditions of approvals issued through authorisations, licences, permits or consents. As conditions of approval, technical measures are typically used to regulate human activities and the impacts within their individual footprints wherever they are located. As regulations impose compliance requirements on an individual or a corporate entity, regulations are implemented through the authority of national legislation that also establishes the footprint of their internal jurisdictions (Cormier and Minkiewicz, 2022). In the case of supra-national bodies, such as the EU, sanctions to the Member State for infraction proceedings for non-compliance of EU legislation can be actioned by the European Court of Justice while compliance for individuals or corporate bodies remain with the competencies of that EU Member State and its judicial system (De Santo, 2011).

Marine management responses are a means to integrate the technical responses and national legislation from multiple internal jurisdictions. Hence, the plethora of bodies with a

TABLE 4 | Examples of linkages between the different types of footprints (Elliott et al., 2020a).

Environmental Footprints			Management Response-Footprints				
Activity	**Pressures**	**Effects**	**Technical Measures**	**Marine Plans**	**National Policy**	**Regional Seas Policy**	**Global Policy**
Land-based undertakings and activities such as urban development, agriculture, and forestry	Catchment input of nutrient and organic matter	Eutrophication and anoxia of estuaries and coasts	Catchment regulations and environmental quality guidelines to control the sources nutrients	Catchment planning of activities and assessment of their collective pressures	Territorial and coastal development and environmental protection legislation and policies	EU Water Framework Directive	UN Sustainable Development Goals and Targets for oceans (14) and for land (15)
Estuarine works and infrastructure such as crossing and ports	Barriers to hydrological flows and flushing	Change in migration patterns of species and fragmentation of species populations	Regulations and guidelines for the location and design of works and infrastructure	Coastal and estuarine integrated plans	Territorial and coastal development and environmental protection legislation and policies	EU Marine Strategy Framework Directive	UN Convention on the Law of the Sea Part XII Protection and Preservation of the Marine Environment
Marine transportation and shipping	Input of contaminants	Pollutions effects in the estuarine, coastal and marine environments	Implementation of IMO MARPOL Codes and recommendations into maritime shipping regulations	Maritime spatial plans	Ratification MARPOL and transposition of EU MSPD into legislation	EU Maritime Spatial Planning Directive	MARPOL codes, guidelines and recommended practices
Marine fisheries	Fishing mortality of targeted and non-targeted species and gear impacts to seafloor	Decreased fishery productivity and changes to the integrity of the seafloor	Fisheries regulatory conditions of licence	Integrated fisheries management planning processes	Fisheries sector development and environmental protection legislation and policies	EU Common Fisheries Policies	UN Convention on Biological Diversity and Code of Conduct for Responsible Fisheries

marine management competency, as shown for one country within the UK by Boyes and Elliott (Boyes and Elliott, 2015), require their response-footprints to be formally or at least informally coordinated. Using that example from the UK, the different bodies in England responsible for managing inshore and offshore fisheries (the Inshore Fisheries and Conservation Authorities (IFCAs) and the Marine Management Organisation (MMO) respectively) have to ensure compatibility in their response footprints. Hence, marine management planning processes have to provide a more holistic approach as an overall response-footprint bringing together managers, regulators and stakeholders to determine how best to manage multiple activities within the marine plan response-footprint while implementing and complying with existing regulatory and non-regulatory requirements. As an overall national policy response-footprint (for example to ensure that a State fulfils the SDG14 to protect its marine waters (Cormier and Elliott, 2017), national legislation establishes both the territorial boundaries of sovereignty to natural resources and the authority to regulate activities. In turn, regional and global policy responses ultimately rely on the national ratification or transposition to achieve their policy goals and objectives.

Technical responses indicate how, where and when an activity can take place while reducing, mitigating and controlling impacts to address the objectives to be achieved in marine plans (Murillas-Maza et al., 2020). National policy responses reflect the societal values of the people living within the boundaries of their States providing the reasons why actions regarding development and sustainability are to be taken which are expressed as legislation, policies and priorities. Global and regional policy responses reflect the transboundary issues that Member States or Contracting Parties have identified as priorities to be addressed through international collaboration and coordination (Cavallo et al., 2018). Regardless of the treaties, conventions and agreements established as global or regional policy responses, current governance structures still require the State to legislate any actions on its individuals and corporate bodies. The principle of subsidiarity is important especially at the level of supranational bodies such as the United Nations and European Union, thereby allowing (or requiring) Member States to take action (Koivurova, 2009). Therefore, these organizations play a major role to ensure coherence across the policies and equivalencies of the management strategies of their Member States to achieve common goals and objectives.

As shown here, the management response-footprint pyramids operates both in a bottom-up and a top-down manner and shows the clearly delimited size (and duration) of the response-footprint for an individual development such as an offshore oil extraction platform. The activity-footprint is well-known in both space and time (i.e. the area occupied by an oil platform and the length of time it is being constructed, operated and been decommissioned are easily determined) and hence so would be the management response-footprint for the activity; in contrast, the management response-footprints for the pressures and the effects on the natural and social systems are less easily defined given the dynamic nature of the marine environment, the dispersion properties of materials emanating from the site and the often highly mobile nature of the organisms affected by the development. At larger scales, the management response-footprints for multiple activities, cumulative and strategic effects, including maritime spatial planning become much harder to define and quantify. In addition, these often require consideration of transboundary consequences, given that the pressures and effects emanating from an activity in the waters of one country can extend to the waters of other adjacent countries (European Commission, 2020).

At the highest level, the current configuration of the governance, management and regulation of maritime activities are framed by the key principles of the United Nations Charter (UN, 1945) that recognizes sovereign equality of all its Members and to refrain from threats or force to their territorial integrity and political independence. UNCLOS simply transposes these principles to the sea in terms of territorial seas and contiguous zones (Part II), straits used for international navigation (Part III), archipelagic States (Part IV), exclusive economic zones (Part V), the continental shelf (Part VI), the high seas (Part VII), regime of islands (Part VIII), enclosed or semi-enclosed seas (Part IX), and the right of access of land-locked States to and from the sea and freedom of transit (Part X). Only the Area (Part XI) curtails the sovereignty of the Members regarding the physical resources and beneath the seabed, including polymetallic nodules which fall under the authority of the International Seabed Authority. Although above we provide generic definitions for the types of the response-footprints, the boundaries of the national policy response-footprints in the marine environment and the marine management and regulations of their maritime activities will ultimately reflect their sovereignty to physical and biological resources within their jurisdictional boundaries as defined by UNCLOS as listed above.

Currently, top-down vertical integration loosely integrates the policy responses from global, to regional, to national and their implementation through marine plans and technical responses (Stelzenmüller et al., 2021). Therefore, an evaluation of the level of integration of marine plans can seldom be linked to the performance of national plans and programs as well as issues surrounding the coherence of global and regional policy responses regarding the management of maritime activities. Global, regional and national policymaking processes most often leave the implementation of such policies to future national regulatory programmes (Marsden, 1998). For example, States Parties to UNCLOS dedicated considerable efforts to ratify UNCLOS to establish their boundaries in the marine environment since coming into effect in the 1990s (e.g., Part II to Part X). Although some have ratified the provisions for the protection and preservation of the marine environment within their jurisdictional boundaries (Part XII) (Cormier and Minkiewicz, 2022), State Parties to UNCLOS have yet to extend these provisions to the high seas to address transboundary issues globally (Verlaan, 2021). Because of the fragmentation of policies, it is also difficult to infer that legislation and policies *carry into effect* the goals and objectives they were set to achieve even though considerable scientific knowledge and stakeholder inputs were brought to bear (Pearl, 2014; Cormier et al., 2017; Korkea-aho, 2022).

We acknowledge that our analysis is written largely from the perspective of the developed countries, principally in Western Europe, North America and Australasia, but also including considerations from the developing and less-developed countries (e.g. Dunstan et al., 2021). It is emphasised here that countries have different capabilities and capacities for marine management and those countries may be regarded as separated into capability, data and skills rich and capability, data and skills poor. It is expected that those with lesser histories of marine management can learn from more-experienced countries and regions and implement marine management policies suited to their particular circumstances. We consider that it is notable that many governance measures and legal instruments can be adopted and often verbatim by other countries without 're-inventing-the-wheel'. Indeed, although outwith the current analysis, given that most if not all maritime states have similar governance structures, it is suggested that the 'law of diminishing returns' applies here in that some marine management initiatives, such as the coordination of ministries and legal instruments related to the marine can be achieved for less-developed maritime states. While we have not attempted a discussion of the financial means of implementing the management response-footprints, we acknowledge that the different measures differ in their cost-effectiveness. For example, the costs of the activity-based measures will be placed upon the developer and industry rather than the state, under the polluter-pays principle. Multiple UN conventions and agreements, such as UNCLOS, have provisions for scientific and technical assistance regarding global and regional rules, standards and recommended practices to address marine pollution and environmental concerns. Examples of international scientific and technical assistance are the international standards, codes of practices and guidelines from the International Maritimes Organization for security and marine pollution including a broad range of other concerns such as the World Health Organization for human health (https://www.who.int/), the Codex Alimentarius for food safety (https://www.fao.org/fao-who-codexalimentarius/en/), the World Organization for Animal Health (https://www.oie.int/en/home/), and the work of the International Plant Protection Convention (https://www.ippc.int/en/). These international organizations have a long history of collaboration in the development of technical measures that can be used by any country.

The capability and capacity of a state to enact the management response-footprints described here also relate to the past or current nature of the state. It is of note that many post-colonial countries have administrative and legal systems derived from their past colonial powers in Western Europe and so may already have an appropriate governance framework. It is expected that as, for example, the UNEP Regional Seas Programmes expand to include states with lesser histories of marine management that can learn from other programmes then good practice in marine management can be transposed to more areas. Despite this, it is also realised that those countries with current or recent unstable geopolitical systems will have priorities other than the integrated management of their seas.

We further acknowledge that we have placed more emphasis on the spatial element of management response-footprints and that the temporal element is of equal importance. This has partly been due to the space available in the manuscript but we also take the view that the temporal aspect cannot be addressed in detail until the spatial element is defined but also that the temporal aspect is even more dependent on the capacity and capability of a country than is the spatial element.

CONCLUSIONS

Marine management implies that the spatial and temporal scales of management are understood and built into prevailing legislation and administrative structures. Those temporal and spatial scales are needed to embody the footprints of activities, their pressures and effects on the marine natural and human systems. However, given that the dynamic nature of the seas requires actions not just at the national level but also the regional and supranational and global levels, those management actions and responses all have their own footprints, even if some of these are overlapping. It is emphasised here that the sustainable management of the seas and their resources requires that the different types and magnitudes of footprints to be understood, quantified and integrated into a holistic marine management approach.

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RC, ME, and AB conceived the paper and contributed in an equal manner to the preparation of the paper. All authors contributed to the article and approved the submitted version.

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Evaluating the quality of environmental baselines for deep seabed mining

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Generating environmental baseline knowledge is a prerequisite for evaluating and predicting the effects of future deep seabed mining on the seafloor and in the water column. Without baselines, we lack the information against which to assess impacts and therefore cannot decide whether or not they pose an acceptable risk to the marine environment. At present, the International Seabed Authority (ISA), which is the international regulator for seabed mining, requires contractors engaged in mineral exploration to establish geological and environmental baselines for their respective contract areas. However, there are no criteria for evaluating what a robust baseline entails. This paper seeks to address this gap by not only analyzing the role and importance of baselines for environmental management but also suggesting criteria for evaluating the quality of baselines. Such criteria (which we present in tabular format) should include at least a minimum amount of technical information, based on best available scientific information and process, in standardized format to enable comparison between contractors and regional synthesis. These criteria should also allow baselines to be used for before-after comparisons through the choice of appropriate zones for comparison of impacts, and to prepare and test a suite of monitoring indicators and their metrics. Baseline studies should identify uncertainties, vulnerable species and habitats, and include transparent reporting as well as exchange with independent scientists and other stakeholders. The quality criteria suggested in this paper build on the ISA's existing Mining Code and seek to support the development of a more standardized catalogue of requirements for environmental baselines. This will allow states, mining operators, the ISA, and the public to gain a better understanding of the environmental impacts of seabed mining and available mitigation measures.

KEYWORDS

deep ocean, environmental baseline, governance, International Seabed Authority (ISA), mining impact, scientific knowledge, seabed mining

Introduction

Deep seabed mining (DSM), if it were to become reality, would yield metals, such as copper, manganese, nickel, and cobalt from some of the least understood places on Earth, in the deep ocean. Knowledge about deep ocean ecosystems remains largely rudimentary and uncertainties remain about the role of the deep ocean in carbon capture, climate regulation, food provision, and other "ecosystem services" (Amon et al., 2022). Importantly, deep ocean species, their life histories and functional relationships remain largely unknown. These uncertainties currently make it almost impossible to predict the precise environmental effects of DSM.

Environmental baselines are the foundation for assessing and managing the environmental effects of DSM, as well as for regulating and permitting seabed mining (Johnson and Ferreira, 2015; Clark, 2019). Baselines document the past and present natural conditions at a future mine site including physical conditions and ecology and help to understand the environment in which DSM might take place. Baselines identify key species and their tolerance for stresses, such as pollution, which can then serve as indicators for measuring and managing the impact of DSM.

Any DSM in the Area[1] is controlled by the International Seabed Authority (ISA) and will require a prior environmental impact assessment (EIA). This, in turn, requires knowledge of the current state of the environment. Thus, the purpose of baselines is to identify scientific questions (such as what level of noise pollution is tolerated by key species at a mine site) and the methodology for answering them.

This paper suggests broad criteria for evaluating the quality of an environmental baseline for DSM as summarized in Table 3. This discussion is warranted not least after an ISA mining contractor submitted an EIA without site-specific baseline data in 2021 (NORI, 2021), which was only rectified (NORI, 2022b) after stakeholder feedback (DOSI, 2021; Pew Charitable Trusts, 2021; NORI, 2022a).

The importance of baselines is undisputed (Johnson and Ferreira, 2015; Clark, 2019, page 458). As the ISA's Mining Code itself states, 'baseline data documenting natural conditions prior to test-mining or testing of mining components are *essential* in order to monitor changes resulting from these activities and to predict impacts of commercial mining activities'[2]. Nonetheless, the ISA has been 'operating in a data-deficient environment, particularly as regards resource data and environmental data' for some time[3]. While data submission by contractors may have improved[4], '[t]here remain, however, ongoing questions about whether enough was being done for the baseline studies, across a range of environmental aspects'[5]. A lack of transparency around data submission, and DSM governance more generally, is compounding the problem (Ardron, 2018; Ardron, 2020; Amon et al., 2022).

This paper discusses the role of, and requirements for, baselines from a scientific and environmental management perspective, while also offering thoughts on how baselines can be legally integrated into the ISA regime. However, the paper does not purport to exhaustively discuss governance questions around baselines. Our aim is to articulate what a robust environmental baseline for DSM would need to entail in order to reflect Best Environmental Practice[6]. Moreover, while acknowledging that geological data are also required for DSM, this paper focuses largely on environmental baselines as these remain subject to significant scientific uncertainty. While the paper focuses on polymetallic nodules, for which mining technology is most advanced, much of the discussion is equally relevant to polymetallic sulphides and ferromanganese crusts.

International legal and policy framework for environmental baselines

Overview

All DSM in the Area is regulated and managed by the ISA on behalf of humankind as a whole[7]. The relevant legal framework

1 Area is a legal term to describe 'the seabed and ocean floor and subsoil thereof, beyond the limits of national jurisdiction'. See United Nations Convention on the Law of the Sea, 1982, article 1(1)(1) (UNCLOS).

2 ISA, 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.1, 30 March 2020, para. 14 (emphasis added).

3 ISA, Developing a Regulatory Framework for Mineral Exploitation in the Area: Report to Members of the Authority and All Stakeholders, ISBA/Cons/2015/1, 2015, https://www.isa.org.jm/files/documents/EN//Survey/Report-2015.pdf, p. 41.

4 ISA, Review of the Implementation of the Environmental Management Plan for the Clarion-Clipperton Zone, ISBA/16/C/43, 1 June 2021.

5 ISA, Report of the Chair of the Legal and Technical Commission on the work of the Commission at the second part of its twenty-sixth session, ISBA/26/C/12/Add.1, 25 September 2020, para. 13.

6 The latest ISA draft exploitation regulations define Best Environmental Practice as follows: '"Best Environmental Practices" means the application of the most appropriate combination of environmental control measures and strategies, that will change with time in the light of improved knowledge, understanding or technology, as well as the incorporation of the relevant traditional knowledge of Indigenous Peoples and local communities, taking into account the applicable Standards and Guidelines.' ISA, Facilitator's Revised Draft Regulations on Exploitation of Mineral Resources in the Area: Parts IV and VI and related Annexes. ISBA/27/C/IWG/ENV/CRP.1/Rev.1, June 2022, Schedule.

consists of the United National Convention on the Law of the Sea (UNCLOS) and its 1994 Implementing Agreement[8] as well as the ISA's Mining Code, which is an umbrella term for the ISA's existing and future rules, regulations, procedures, and recommendations.

The ISA and its 167 member States have far-reaching environmental obligations, such as to 'ensure effective protection for the marine environment from harmful effects' of mining activities, including 'prevention of damage to flora and fauna of the marine environment'[9]. The ISA also needs to assess the predicted impacts of DSM,[10] protect vulnerable ecosystems,[11] and determine whether a proposed mining plan provides for environmental protection[12]. Achieving such a high bar requires environmental baselines, including a basic understanding of the environmental dynamics and its history so all data may be put into a context of periodic and episodic change to be able to apply robust and proactive environmental management. This involves at least ten tasks, the first six of which already apply during the exploration phase and are required under the ISA's Mining Code, as summarized in Table 1. Tasks seven to ten should be required for any future mineral exploitation, although some of those tasks will already be carried out during the exploration phase in order to be ready by the time a contractor applies for exploitation.

An environmental baseline should be established during the 15-year exploration stage, prior to any mining, to allow for an EIA. As the LTC notes, a baseline is designed '*to acquire the capability necessary to make accurate environmental impact predictions*'[13], and to demonstrate that the activities planned will not cause serious harm to the marine environment[14].

7 UNCLOS, articles 1(1)(1), 136, 137, 157.

8 Agreement Relating to the Implementation of Part XI of the United Nations Convention on the Law of the Sea, 1994.

9 UNCLOS, article 145.

10 UNCLOS, article 165(2)(d).

11 ISA, Regulations on Prospecting and Exploration for Polymetallic Sulphides in the Area, ISBA/16/A/12/Rev.1, 7 May 2010, regulation 33(4); ISA, Regulations on Prospecting and Exploration for Cobalt-rich Ferromanganese Crusts in the Area, ISBA/18/A/11, 22 October 2012, regulation 33(4); see also ISA, Regulations on Prospecting and Exploration for Polymetallic Nodules in the Area, ISBA/19/C/17, 22 July 2013, regulation 31(4).

12 UNCLOS, article 165(2)(b); ISA, Draft Regulations on Exploitation of Mineral Resources in the Area, ISBA/25/C/WP.1, 22 March 2019, draft regulation 13(4)(e).

Current governance challenges

While the fundamental importance of baselines is clear, their use in decision-making processes is somewhat less understood. There are at least three governance questions.

First, how is the quality of environmental baselines assessed? There are currently no publicly available criteria for assessing the quality and completeness of baselines (Clark, 2019, p. 457; Ginzky et al., 2020, p. 6-7). Such criteria are important both for transparency of environmental decision-making and to ensure all contractors are held to the same standard and address comparable questions. The section on quality criteria for a robust baseline below offers criteria that could be used as a starting point to assess environmental baselines.

Second, who assesses the quality of baselines? Within the ISA, the LTC is the primary body that reviews environmental data, although constraints on the LTC's time and expertise in the fields of environmental management and marine biology are well known[15] (Jaeckel, 2017b, ch 8.3; Ginzky et al., 2020).

Third, when is the quality of baselines assessed? Ultimately, when a contractor submits an application for mineral exploitation, the LTC needs to assess a prior EIA which should be based on site-specific environmental baseline data collected during the exploration period. At that point, the LTC can assess the baseline information and approve or reject a plan of work for exploitation[16], as visualized in Figure 1.

Ideally, the LTC should assess baseline data well before the exploitation application stage, not least to assist the contractor in improving its baseline, where necessary. Contractors have to report on their baseline investigation program annually[17], during 5-yearly reviews of their plans of work for exploration[18], as well as at the end of an exploration contract[19]. Arguably, the 5-yearly reviews as well as the final review should be used to conduct a thorough assessment of baseline data and determine which gaps will need filling. This would support the contractor and improve the Council's ability to 'exercise control over activities in the Area'[20]. The review and baseline studies should re reported transparently (Ardron, 2018; Ardron, 2020; Haeckel et al., 2020; Komaki and Fluharty, 2020; Willaert, 2022), not least to address challenges around non-compliance with annual reporting requirements.[21]

13 ISBA/25/LTC/6/Rev.1, para. 13.

14 ISBA/25/LTC/6/Rev.1, annex I paras. 2, 65

15 ISA, Suggestions for facilitating the work of the International Seabed Authority – Submitted by the Delegation of Germany. ISBA/24/C/18, 27 June 2018.

16 ISBA/25/C/WP.1, draft regulation 13(4)(e).

17 ISBA/19/C/17, regulation 32, annex IV sections 5, 10.

18 ISBA/19/C/17, regulation 28.

TABLE 1 Selected environmental tasks and requirements involved in DSM and selected corresponding legal provisions.

	Environmental tasks for ISA contractors	Selected legal references
Exploration phase		
1	Create environmental baseline	ISBA/19/C/17, regulations 18, 31, annex IV sec. 5 ISBA/25/LTC/6/Rev.1, paras. 8, 11, annex I para 2 1994 Implementing Agreement, annex sec. 1(7)
2	Provide methods to monitor and evaluate environmental impacts	UNCLOS, articles 204, 206 ISBA/19/C/17, regulations 31(6), 32 ISBA/25/LTC/6/Rev.1, paras. 8, 11, annex I para 2
3	Conduct EIA for particular exploration work	UNCLOS, article 206 ISBA/19/C/17, regulation 18(b), 31(6) ISBA/25/LTC/6/Rev.1, para. 8, annex I para 2
4	Provide data for regional management	ISBA/25/LTC/6/Rev.1, paras. 15, 16, annex I para 2
5	Establish procedures to demonstrate no serious harm from exploration work	UNCLOS, article 145 ISBA/19/C/17, regulations 2(2), 31(4), 22, 34(4) ISBA/25/LTC/6/Rev.1, para. 11, annex I para 2
6	Establish preservation and impact reference zones	ISBA/19/C/17, regulation 31(6) ISBA/25/LTC/6/Rev.1, paras. 35, 38(o)
Exploitation phase		
7	Conduct EIA for exploitation work	UNCLOS, article 206 ISBA/25/C/WP.1, draft regulation 3(e), 7(3)(d), 47
8	Monitor impacts before, during, after exploitation	ISBA/27/C/6, para. 40
9	Compare monitoring data with baseline data	/
10	Create an Environmental Management and Monitoring Plan, incl mitigation measures	ISBA/25/C/WP.1, draft regulation 48

When a contractor conducts equipment testing during the exploration stage, or similar activities which require an EIA[22], the LTC has an additional opportunity to review the baseline and assess the quality of the monitoring programfor the testing. From a regulatory perspective, this is a key point at which the contractor must present baseline knowledge 'that would enable an assessment of the potential environmental impact, including, but not restricted to, the impact on biodiversity, of the proposed exploration activities […]'[23]. This presents an important opportunity for the LTC to indicate whether the contractor has conducted sufficient baseline studies and which gaps may need attention.

The EIA process during the exploration stage is problematic (Jaeckel, 2017b, p. 240). The ISA cannot formally accept or reject an EIA during the exploration stage, partly because the EIA occurs *after* the exploration contract has been concluded. Instead, the LTC merely reviews the resulting environmental impact statement (EIS) for 'completeness, accuracy and statistical reliability'[24]. The LTC can then recommend to the Secretary-General whether or not the EIS should be incorporated into the contractor's 5-year program of activities[25]. If the LTC makes a negative recommendation, the contractor needs to amend and resubmit its EIS. It remains unclear whether the LTC's Recommendations will be published in full and whether its recommendations are binding on the Secretary-General. A regulator should assess and approve or reject an EIS to avoid EIAs becoming box-ticking exercises (Jaeckel, 2017b, p. 238) rather than the important environmental management tool they can and should be.

To fully operationalize the EIA and monitoring program, the Mining Code should require contractors to submit to the LTC a comparison of the environmental effects of mining (tests) with the established environmental baseline. This would require full-scale mining tests of sufficient duration to enable a reliable evaluation of impacts to be expected from commercial mining (Singh and Christiansen, 2022, pp. 198, 200). Similarly, for the exploitation phase, it will be important to compare baseline data, pre-mining monitoring data, and monitoring data collected during and after mining to determine the actual impacts caused by mining. While the current draft exploitation regulations do not explicitly include such a requirement, the relevant *Draft guidelines for the preparation of environmental*

19 ISBA/19/C/17, annex IV, sec 11.2.

20 UNCLOS, article 162(2)(l).

21 ISA, Report of the Chair of the Legal and Technical Commission on the work of the Commission at its session in 2017, ISBA/23/C/13, 9 August 2017, para. 15(c).

22 ISBA/19/C/17, annex IV, section 5.2; ISBA/25/LTC/6/Rev.1, para. 33.

23 ISBA/19/C/17, regulation 18(b), see also regulation 32.

24 ISBA/25/LTC/6/Rev.1, para. 41(c).

25 ISBA/25/LTC/6/Rev.1, para. 41(h)-(i).

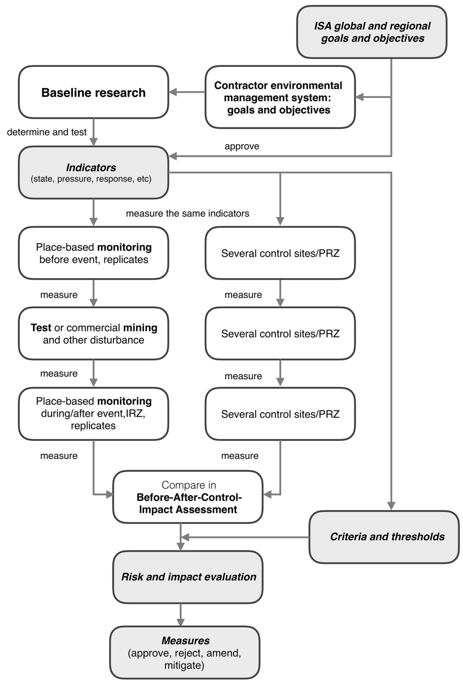

FIGURE 1
Timeline of key environmental management steps for seabed mining.

management and monitoring plans express an expectation that monitoring data will be compared against baseline data.

Legal consequences of inadequate baselines

Failure to establish an adequate environmental baseline can have a range of legal consequences. First, it can ultimately lead to a mining operator having their exploitation application rejected because without a compliant baseline, a plan of work should not meet the required standard of providing for 'effective protection of the Marine Environment'[26].

If little to no baseline data has been shared with the ISA, an application could also be rejected based on draft regulation 7(3)(a) of the draft Exploitation Regulations, which requires an applicant to have submitted baseline and other environmental data from the exploration stage to the ISA. Moreover, in assessing an exploitation application, the applicant's 'previous operating record of responsibility' will likely be taken into account[27]. Hence, any breach of the relevant Exploration Regulations could affect a contractor's chances of obtaining an exploitation contract.

26 ISBA/25/C/WP.1, draft regulation 13(4)(e).

During the exploration stage, there are few consequences for failing to establish adequate baselines. Indeed, the Exploration Regulations lack procedural safeguards to ensure baseline data is submitted (Jaeckel, 2017b, p. 246). While the regulations require contractors to collect baseline data, there are no direct consequences for failing to do so (Ginzky et al., 2020, pp. 6-7). Ultimately, failure to comply with the requirement to establish a baseline, may lead to the contractor losing their preferential or priority option to apply for an exploitation contract in respect of the area in question[28]. Additionally, a lacking baseline is unlikely, though not impossible, to lead to a termination of an exploration contract, which can only occur in rare circumstances[29].

In principle, inadequate baselines could also lead the LTC to voice concerns when reviewing an EIA for test mining or other exploration activities. Though, as noted above, the LTC has no formal power to reject an EIA during exploration phase.

The risk is that as long as there are no agreed quality criteria for an "adequate" or "sufficient" environmental baseline, an inadequate baseline may still satisfy the legal requirements. Thus, the key question is on the basis of what criteria can a baseline be regarded as adequate or robust? The Exploration

27 ISBA/25/C/WP.1, draft regulation 12(4)(c).

28 ISBA/19/C/17, regulation 24(2).

29 ISBA/19/C/17, annex IV sec 21.

Regulations require a baseline which takes into account the LTC's Recommendations for EIA[30]. These Recommendations specify the type of information baselines must include but do not set out criteria for assessing the adequacy of baselines. The section on quality criteria for a robust baseline below seeks to fill this void.

Technical baseline requirements

Existing technical requirements for baselines

The LTC has issued recommendations for building environmental baselines (LTC Recommendations)[31], which while non-binding, carry significant weight within the ISA regime[32]. These already require exploration contractors, inter alia, to characterize all environments likely to be affected by DSM during exploration, mining tests and exploitation[33]. The studies should cover not only all aspects of biodiversity, the physical, chemical, geological, biological and sedimentary properties of the seafloor and the water column[34], but also the background levels of contaminants[35], noise[36], and other anthropogenic pressures[37] prior to any testing or mining, as well as provide an integrated view on ecosystem functioning[38] and genetic connectivity[39]. Temporally, investigations must be continued long enough to describe the natural variability of parameters[40], including prevailing trends, e.g., due to climate change. With the deep-sea environment governed by periodic and episodic long-term cycles, however, the required duration of three years for long-term measurements is unlikely to be sufficient[41]. Spatially, baseline investigations should extend to all parts of the exploration contract area, with a higher intensity of sampling at potential mine or test sites as well as broader reference areas[42]. The latter is required for identifying representative impact reference zones and preservation reference zones, as well as for not impacting larval source areas and connectivity patterns of benthic species and communities in the wider region[43] (Baco et al., 2016). The LTC's Recommendations include over 100 requirements (Bräger et al., 2020) and incorporate sampling methodologies and broad impact indicators for benthic and pelagic habitats, as collated in Table 2[44].

Suggestions for additional technical baseline requirements

While the current LTC Recommendations are a good starting point for establishing environmental baselines, they are incomplete and at times relatively vague, prompting this section to suggests additional requirements for baseline studies in the Area.

Comparability

A baseline offers the comparator against which to assess the effects measured by the monitoring program for a scientific disturbance or testing exercise. Generally, only those parameters measured and recorded during the baseline studies can be compared later, and only when employing comparable methods. Therefore, a baseline investigation requires careful and systematic planning from the start of the exploration period to ensure all parameters that may have to be captured by the monitoring program are already included during baseline investigations. For example, the baseline studies need to identify those species and habitats which are presumed to be particularly vulnerable to mining and might serve as indicators for the environmental state and for the range of mining-related effects.

All contractors exploring the same resource should attain baselines of comparable quality, using the same or directly comparable methods to ensure a level playing field and similar costs (Glover et al., 2016). The LTC recognized such

30 ISBA/19/C/17, regulation 32; ISBA/25/LTC/6/Rev.1.

31 ISBA/25/LTC/6/Rev.1.

32 ISA Standard clauses for exploration contract, ISBA/19/C/17, annex IV section 13.2 (e).

33 ISBA/25/LTC/6/Rev.1, para. 13.

34 ISBA/25/LTC/6/Rev.1, para. 15.

35 ISBA/25/LTC/6/Rev.1, annex I para. 45.

36 ISBA/25/LTC/6/Rev.1, annex I para. 43.

37 ISBA/25/LTC/6/Rev.1, annex I para. 61.

38 ISBA/25/LTC/6/Rev.1, para. 40(b), annex I paras. 29, 41(e), 46, 60.

39 ISBA/25/LTC/6/Rev.1, para. 15(d)(vii), annex I paras. 30, 38.

40 ISBA/25/LTC/6/Rev.1, paras. 15(d)(vi), 15 (f), annex I paras. 21, 29, 46

41 'Temporal variation must be evaluated for at least one test-mining site and the preservation reference zone following the terminology agreed prior to the test-mining activity (ideally, with a minimum of annual sampling over at least three years).' (ISBA/25/LTC/6/Rev.1, annex I para. 46)

42 ISBA/25/LTC/6/Rev.1, para. 38(o), annex I para. 67.

43 ISBA/17/LTC/7, paras. 25, 51, 52.

44 With scientific methods rapidly developing, Table 2 is intended to only provide examples, concentrating on fundamental issues that are unlikely to change soon.

TABLE 2 Selected environmental exploration requirements from ISBA/25/LTC/6/Rev.1 sorted by impact, methodology and habitat. All references are to ISBA/25/LTC/6/Rev.1.

Impact	Benthic habitat	Pelagic habitat
Turbidity	annex I para. 53(b)	para. 15(a)(ii); annex I para. 53(b)
Heavy metals	para. 15(c)(ii); annex I para. 45	para. 15(c)(ii); annex I para. 45
Smothering	para. 40(b(-(d); annex I para. 38	
Interrupted connectivity	para. 15(d)(vii); annex I para. 38	
Discharge plume		annex I para. 42(c)
(Lack of) bioturbation	annex I para. 52	
Methodology	Benthic habitat	Pelagic habitat
Spatial sample distribution	annex I para. 10; annex I para. 38	para. 15(a)(ii); annex I para. 12; annex I para. 21; annex I para. 53(b)

Temporal sample distribution	para. 15(d)(vi);
	annex I para. 41(g);
	annex I para. 46
		annex I para. 21;
		annex I para. 42(c);
		annex I para. 53(b)

Species-specificity & ecology	para. 15(d)(vii);
	para. 38(o);
	para. 40(b)-(d);
	annex I para. 38;
	annex I para. 41(a);
	annex I para. 47
		para. 15(d)(iv);
		para. 15(d)(v);
		para. 15(d)(vii);
		annex I para. 42(c);
		annex I para. 47

Statistical robustness	annex I para. 39	annex I paras. 39, 44
Experimentation	annex I para. 14; annex I para. 45	

standardization of methods as 'extremely important'[45]. Comparable baselines in terms of quality and methodology will allow pooling of contractor data to create regional baselines. This also helps to determine whether faunal uniqueness so far observed at every geographic scale (e.g. Washburn et al., 2021) is real and requires precautionary protection. Regional baselines are required for assessing cumulative effects of multiple projects in regions, providing the basis for a regional management plan[46].

Indicators

Baselines need to identify suitable indicators to be monitored, which could be set at a regional level through regional environmental management plans (REMPs). Suitable indicators need to be informed by the knowledge about the respective environment and by environmental goals and objectives for a particular region. As the latter do not exist in sufficient regional detail for ISA contractors, it is currently not foreseeable that all contractors will work towards the same indicators and proceed with similar monitoring methods and strategies. Furthermore, the indicators identified from the baseline studies, as well as eventual thresholds for deviations from normal, can only be verified with information from small-to-medium-scale test mining yet to take place.

Monitoring of the selected indicators would have to start a long time before any (test-) mining to establish the indicators' representativity and sensitivity to the pressure from test mining. Ideally, the deep-sea ecosystem would need to be investigated and well understood before allowing impacts. Equally undefined is whether the impact monitoring should take place only inside (possibly numerous) Impact Reference Zones (IRZs) and Preservation Reference Zones (PRZs), or whether a wider area needs to be surveyed, e.g., radially and with distance from the mine site, to follow mining-related plumes throughout their impact area on the seafloor and the water column.

Duration

A baseline should study the environment for a defined period of time to facilitate understanding of the long-term effects of DSM. Current knowledge is rather limited, but indicates that processes take place over long to very long (if not geological) time-scales (Jones et al., 2017; McQuaid et al., 2020). Deep ocean ecosystems are subject to large-scale natural

45 ISBA/25/LTC/6/Rev.6, annex I para. 37.

46 ISBA/17/LTC/7, paras. 34, 37, 40, 51; ISBA/25/LTC/6/Rev.1, annex I para. 51

variability (e.g. El Niño Southern Oscillation, a global oceanic-atmospheric phenomenon)[47] and long-term trends (e.g., due to climate change). These need to be identified in the data, including from sediment cores, and incorporated in assumptions on the future development of the relevant aspects of oceanography and ecosystems including, for example, increased vulnerability to other stressors and impacts on ecosystem services (Levin et al., 2020). This requires the monitoring and assessment of time-series of sufficient duration[48] and spatial extent over all habitat types[49] to detect the long cycles and ephemeral events such as eddy formation (Aleynik et al., 2017).

Sample sizes

Baselines must include statistically meaningful sample sizes for biological community metrics such as biomass, abundance, community diversity, structure and composition, and a list of species present in the contract area (Glover et al., 2018). In particular, for benthos sampling, this requires multiple replications in impacted and control areas, covering all size classes of organisms (Glover et al., 2016; O´Hara et al., 2016; Schiaparelli et al., 2016; Jones et al., 2020). Samples taken in a particular environment cannot be used to extrapolate predictions for mining impacts in different environments. This is particularly important given the high percentage of species that are only found at a single site (Washburn et al., 2021, Fig. 13).

Detection of any environmental change with some confidence, e.g., in the abundance of some (or all) species, will depend on the statistical power which again depends on the number of (high-quality) samples collected under standardized conditions[50]. Where the contractor seeks to prove the absence of a significant difference, such as between species densities before and after mining, an even larger sample size may be required. The sample size required to attain sufficient statistical power can be assessed beforehand with a power analysis (Ardron et al., 2019). For benthic fauna at the sea floor, these requirements should be achievable, but in the highly dynamic water column their application for pelagic fauna may prove to be rather challenging though still necessary.

Quality criteria for a robust baseline

In this section, we offer criteria that are designed to help assess the quality of an environmental baseline for DSM. Environmental baseline investigations have to reflect the whole of the *in situ* environmental situation in a contract area. The baseline captures the environmental state at a certain time, taking into account natural variability and change, in order to determine additional change (impact), as a response to mining-related activities (pressure). The ecosystem approach to management best reflects this comprehensive interaction and provides for transparent and stakeholder-inclusive processes including an assessment of the cost of environmental degradation (Elliott et al., 2017; European Commission, 2020).

In the following, we build on the requirements currently included in the LTC Recommendations[51] under nine topic headings (same numbering as in Table 3) and reorganize them to match elements of good environmental management suggested in the scientific literature. Under each topic heading, we indicate the key question which an assessment of the quality of the baseline should address. Within each topic, we articulate criteria that could be used to assess the quality of environmental baseline programs and the data they produce. The criteria are examples only and are not exhaustive. In fact, Table 3 includes additional criteria, which we deem important but are not discussed here for lack of space. These are topics (8) best environmental practice and (9) collaboration and regional integration. The criteria are designed to offer a starting point for assessing the quality of baselines. Table 3 lists the proposed criteria and example actions and indicators and, where applicable, provides links to the literature that elaborates on the relevant criteria as well as to selected ISA documents or other international instruments that encapsulate the relevant criteria. The last column of the Table indicates whether the proposed criteria are sufficiently covered by existing requirements in the ISA's Mining Code, in a preliminary effort to indicate potential gaps in the current Mining Code.

Substantial LTC requirements

Does the baseline comply with existing LTC requirements and provide essential ecosystem information for evaluating change?

As discussed above, current LTC Recommendations provide an extensive list of elements of the ecosystem to be investigated. However, the Recommendations lack specificity with respect to the temporal and spatial scales to be covered for establishing the natural spatial heterogeneity and temporal variability at the seafloor and in the water column. These gaps are addressed in the additional technical considerations outlined above. A robust baseline should follow the current LTC requirements as well as the additional considerations outline above.

47 cf. ISBA/25/LTC/6/Rev.1, annex I para. 39.

48 ISBA/25/LTC/6/Rev.1, annex I paras. 21, 53(b).

49 ISBA/25/LTC/6/Rev.1, annex I paras. 10, 38.

50 cf. ISBA/25/LTC/6/Rev.1, annex I para. 39.

51 ISBA/25/LTC/6/Rev.1.

TABLE 3 Broad governance criteria and indicators for determining good baseline investigation programs and knowledge.

Topics	Criteria	Example actions and indicators	Example scientific references	ISA documents and other legal instruments	Current requirement for baseline
1. Baseline consistent with substantial LTC requirements	Basics to understand the composition, structure and functioning of ecosystems	Coarse scale topographic and resource multi-beam mapping of the whole contract area; Fine scale description (seafloor mapping) and investigations in all areas of interest; Time series measurements, including sediment cores, long enough to identify effect of large scale (e.g. El Nino/La Nina) and long-term trends (i.e. climate change) on ecosystems and predict future development (all aspects of oceanography and ecosystems, incl. ecosystem services); Statistically adequate sampling; Highest possible taxonomic resolution.	(Snelgrove et al., 2014; Swaddling et al., 2016; Volz et al., 2018; Haeckel et al., 2020; Levin et al., 2020)	ISBA/25/LTC/6/Rev.1, paras. 15(d)(i), (vi), annex I para. 25	Many individual requirements, but no comprehensive system
		Life history, feeding types, food web relations, reproduction, respiration, mobility, feed-back loops between biota; Hydrodynamics incl. natural sinking flux of materials and biogeochemistry; Bioturbation, stable isotopes and sediment community oxygen consumption for nodules communities, food webs, stable isotopes, fatty acids and methane and hydrogen sulphide metabolism in sulphides communities, and food webs, stable isotopes and fatty acids in ferromanganese crust communities.	(Snelgrove et al., 2014; Christiansen, 2016; Christiansen et al., 2020; Clark et al., 2020; Haeckel et al., 2020)	ISBA/25/LTC/6/Rev.1 paras. 15(f), (g), (h); ISBA/21/LTC/15, para. 9 (g) annex I, II	Major deficiencies, especially with respect to ecology and foodwebs
		Species, community, and population connectivity.	(Taboada et al., 2018; Dunn et al., 2019; Popova et al., 2019; Yearsley et al., 2020)	ISBA/25/LTC/6/Rev.1, paras. 14, 15(d)(vii), 38, annex I paras. 30, 47; ISBA/21/LTC/15, annex I-III para. 10(h)	Required but not linked to action
		Relate regional productivity, depth, current speed, topographic features, nodule abundance with benthic and benthopelagic community composition (and dynamics).	(Amon et al., 2016; Leitner et al., 2017; Simon-Lledó et al., 2019; Simon-Lledó et al., 2020; Leitner et al., 2021)	ISBA/25/LTC/6/Rev.1, annex I para. 38	No regional long-term synthesis required
		Relate oceanographic patterns and processes with pelagic community composition and development.	(Drazen et al., 2021; Perelman et al., 2021)	ISBA/25/LTC/6/Rev.1, paras. 15(a), 15(d)(iv), annex I paras. 8, 10, 12, 16, 42(c), 53(b).	No regional long-term synthesis required
	Statistically validated results	For biological communities: Benthos: predefined high accuracy sampling schemes (depends on heterogeneity of habitat and abundances of individuals) to provide statistically meaningful sample numbers, sample sizes, and multiple replications in impacted and control areas, covering multiple size classes of organisms; Benthopelagos and Pelagos: sampling grid to represent a) main biotic and abiotic features of the mine site, b) at least three locations representing maximum, medium and minimum particle concentrations from operational and discharge plumes, and c) one or more reference stations. Replications needed; Deepsea fish and scavengers; Megafauna; Birds.	(Christiansen, 2016; O´Hara et al., 2016; Schiaparelli et al., 2016; Ardron et al., 2019; Christiansen et al., 2020; Jones et al., 2020)	ISBA/25/LTC/6/Rev.1, annex I paras. 30-52 ISBA/25/LTC/6/Rev.1, paras. 13, 15 ISBA/25/LTC/6/Rev.1, paras. 15(a), 15(d)(iv), annex I paras. 8, 10, 12, 16, 42(c), 53(b)	Required (criterium for EIS review)

(Continued)

TABLE 3 Continued

Topics	Criteria	Example actions and indicators	Example scientific references	ISA documents and other legal instruments	Current requirement for baseline
	Provides baseline knowledge of biological community metrics	Biomass, abundance, species richness, diversity, water column and seafloor topography, structure and composition; Species, community, and population connectivity; Species and habitats to be protected.	(Swaddling et al., 2016; Ardron et al., 2019; Ardron, 2020)	ISBA/21/LTC/15, annex I- III para. 9(g); ISBA/25/LTC/6/Rev.1, paras. 14, 15(d)(vii), annex I paras. 30, 47; ISBA/21/LTC/15, annex I-III, para. 10(h)	No standard metrics required
	Characterizes the environments likely to be impacted by exploration, or testing	Baseline assesses the dispersal potential for particles and dissolved substances; Full impact areas defined, including buffer zones; Suitable indicator organisms or processes identified for monitoring mining effects	(Aleynik et al., 2017; Gillard et al., 2019; Baeye et al., 2021; Muñoz-Royo et al., 2021)	ISBA/25/LTC/6/Rev.1, paras. 14, 15(a)(ii), (iv), (v), annex paras. 20-21 (ISA, 2017, p. 20)	No definition of impact and impacted area
		Establishes background levels of heavy metals in sediments, water column and interface.	(Mestre et al., 2017)	ISBA/25/LTC/6/Rev.1, paras. 15(b), 15(c)(ii), 40(f), annex I paras. 14, 28, 45	Required
		Establishes background levels of ambient noise, light (here bioluminescence), litter, and other pressures.		ISBA/25/LTC/6/Rev.1, paras. 13, 14, annex para. 51 (only noise)	Only noise
	Documents the avoidance of serious harm	Demonstrate that there is no serious environmental harm from any activities being conducted on the seabed, in mid-water, and in the upper water column.	(Levin et al., 2016)	ISBA/25/LTC/6/Rev.1, annex I, paras. 2, 65	Required but no criteria for checking
2. Baseline established with highest standards	Baseline programme makes a clear statement of objectives	Project-specific objectives, including environmental objectives, identified and embedded into goals and objectives of respective region and globally; Objectives, incl. for PRZ and IRZ, inform sampling design; High level of precaution to account for deficiencies in knowledge.	(Sullivan et al., 2006; Clark et al., 2016b; O´Hara et al., 2016; Swaddling et al., 2016; Van Dover et al., 2016; Jones et al., 2020)	ISBA/21/LTC/15 para. 6 (plan of work, PoW), annex I-III para 9(a) (monitoring); ISBA/17/LTC/7, para. 41(a) as part of ISO 14001	Only general objectives in PoW
	Baseline programme operates with a conceptual model	Model provides e.g. for a framework for characterizing environment, making predictions, and testing hypotheses.	(Clark et al., 2016b; O´Hara et al., 2016)	ISBA/25/LTC/6/Rev.1/, annex I para. 65 speaks of a plan of the contractor reviewed by LTC	Not required
	Desktop review of scientific literature provides basis for field studies	Characterizes all environments likely to be impacted by mining-related activities; Includes all topics named in LTC Recommendations (physical oceanography, geology, chemistry/geochemistry, sediment properties, bioturbation and sedimentation, and biological communities); For biological communities: reports on the likely vulnerability, temporal and spatial scale of certain species, communities, ecosystem processes to mining-related impacts.	(Durden et al., 2017; Clark, 2019)	ISBA/25/LTC/6/Rev.1, paras. 13-18	Desktop study not required as a starting point
	Systematic design of field surveys	Systematic investigation and monitoring concept from the start, including: Specification of aims, design and plan for spatial and temporal execution of multidisciplinary research; All ecosystem elements possibly affected by mining are covered; Data collection to cover the entire water column and the seafloor in appropriate spread and repetition over sufficient periods to measure natural variability; Use of high-resolution seafloor maps to plan biological sampling,	(Glover et al., 2016; O´Hara et al., 2016; Schiaparelli et al., 2016; Swaddling et al., 2016; Durden et al., 2017; Glover et al., 2018; Jones et al., 2019)	ISBA/25/LTC/6/Rev.1, paras. 13, 14, 15(d)(vi), 15(f), annex I paras. 8, 10, 12, 18, 21, 29-32, 37, 38, 46, 53(b)	Many individual requirements, but no comprehensive systematic design

(Continued)

TABLE 3 Continued

Topics	Criteria	Example actions and indicators	Example scientific references	ISA documents and other legal instruments	Current requirement for baseline
		Standardised sample processing and analysis.			
	Best scientific methods	Independent and replicate samples for any combination of space, time, habitat, and gradient to estimate variability between samples/groups of samples. Higher variability requires more replicates; Gear-type and deployment standardized between surveys, and documented in detailed protocol; The sampling methods are suitable for measuring the reported parameter, including sufficient sample size, and as specified in LTC Recommendations.	(Underwood, 1994; Underwood and Chapman, 2003; Underwood and Chapman, 2005; Clark et al., 2016b; Stocks et al., 2016)	ISBA/25/LTC/6/Rev.1, para. 15(d), annex I paras. 8 (speaks of a 'stratified random sampling programme'), 30, 37, 45. ISBA/25/LTC/6/Rev.1, annex I paras. 39-41	Required but no criteria for checking
	Data presented are complete, and representative	Complete means no sampling data are excluded from analysis, plausibility check, and no null data. Metadata are provided with the data; Representative means that the variability of environment and biota is captured and higher heterogeneity is sampled with an increased spatial resolution.	(Stocks et al., 2016, p. 370)	ISBA/25/LTC/6/Rev.1, paras. 15(d)(ii), 20-21	Required but no criteria for checking
	Conclusions are supported by statistical rigour and sound logic for analysis and interpretation	Variability and confidence limits are essential information to validate all analyses; Power and effect size of statistical analysis should be disclosed; Adequacy and uncertainties of interpretation described.	(Underwood and Chapman, 2003; Underwood and Chapman, 2005; Jones et al., 2020)	ISBA/25/LTC/6/Rev.1, paras. 14, 15(d)(ii), annex I paras. 39-40; ISBA/21/LTC/15, annex I-III para. 10(d)	Required but no criteria for checking
	Reporting delivers documentation of methods, results, and conclusions;	Uncertainties and knowledge gaps in baseline specified; Where relevant, differences in scientific understanding acknowledged; Research plan identifies how to address gaps.	(Gerber and Grogan, 2020)	ISBA/21/LTC/15, Annex I-III para. 10(e) (gaps with regard to plan of work)	Required but only formal gap analysis
	Results published and peer reviewed	Mandatory publication of results as reports or peer-reviewed literature.		ISBA/21/LTC/15, Annex I-III, para. 17(a) (list of publications), environmental data included in ISA database	Not required
3.Baseline informs Reference Zones	Baseline provides data for the identification of one or more PRZ and IRZ (in the likely path of mining effects in the test or mine site)	Baseline enables spatial planning of contract area, incl. PRZ and IRZ location; All reference zones are within the contract area; Baseline documents that PRZ are truly representative of the mined area (IRZ) and justifies that it will not be affected by mining over the progressing mining period; Baseline determines and justifies the size, number and locations(s) of PRZ and size of buffer zones; Baseline determines necessary number and size of IRZ along the gradient of environmental effects of each contractor's activities, such as from testing or mining, on the seafloor and the water column.	(Underwood, 1994; Underwood and Chapman, 2003; Underwood and Chapman, 2005; Billett et al., 2019; Haeckel et al., 2020; Jones et al., 2020) (Underwood, 1994; Underwood and Chapman, 2003; Underwood and Chapman, 2005; Billett et al., 2019; Haeckel et al., 2020; Jones et al., 2020)	ISBA/25/LTC/6/Rev.1, para. 38(o); ISBA/17/LTC/7, para. 41 (c-e) (ISA, 2017; ISA, 2018)	No contractor guidance as to positioning, number, size, permanence etc.
4. Baseline informs a comprehensive monitoring programme	State-of-the-art monitoring methodology developed based on baseline results	Monitoring programme is in line with ISA LTC Regulations, Recommendations and standards and guidelines; Methods and parameters are verified, standardised and regularly updated; Sampling strategies are adequate:	(Zampoukas et al., 2013; Haeckel et al., 2020; Jones et al., 2020)	ISBA/19/C/17, reg. 32; ISBA/25/LTC/6/Rev.1, paras. 2, 8, 11, 35-37, annex I paras. 65, 65; ISBA/21/LTC/15, annex I-III paras. 9(d), 10(e), 10(g), 10(h) (ISA, 2017, p. 35)	No clear criteria for monitoring programme, no link to indicator development

(Continued)

TABLE 3 Continued

Topics	Criteria	Example actions and indicators	Example scientific references	ISA documents and other legal instruments	Current requirement for baseline
		To measure a comprehensive set of parameters which reliably detect effects from activities; In fit-for-purpose locations, periodicity, intensity and methods; To link with project-scale and regional assessments; To detect impacts in time and space and provide statistically defensible data; To establish the whole impact area from a test or mining event, including beyond the contract area; To apply the precautionary principle; Contractors must consider variance and statistical power in IRZ and PRZ monitoring.			
	Baseline provides the basis for identifying and justifying indicators	The chosen environmental indicators are based on longer term measurements and are: Anticipatory to provide an early warning of deterioration; Biologically important, applicable and indicative over space and time; Scientifically sound and measurable and quantifiable over space and time; Sensitive to different levels of harm, including serious harm from pressures and responsive to management; Socially relevant and interpretable by stakeholders. Reference points are determined for the chosen indicators from which to measure: Pressure, environmental state and change; Progress towards environmental targets; The effectiveness of measures.	(Rice and Rochet, 2005; Elliott, 2011; Zampoukas et al., 2013)	ISBA/25/LTC/6/Rev.1, annex I para. 41(e) speaks of the integration of metabolites information into the analysis of taxonomic and function gene diversity provides additional quantitative indicators of ecosystem functions (and services)'.	No requirement to determine and justify indicators
	Baseline indicates preliminary precautionary thresholds to avoid harm;	Baseline research justifies appropriateness of indicators and preliminary thresholds (desktop review or experimental); Baseline shows iterative improvement of appropriate threshold type (ecological tipping points, management) and level (normal/precautionary/limit).	(Groffman et al., 2006; Levin et al., 2016; Tunnicliffe et al., 2020)	Not required during exploration, but should have been developed until prior exploitation EIA[52]	Not required
5. Baseline Informs environmental impact assessment[54]	Baseline includes results of mining tests	Test design adequate for conclusions on type, scale, duration of impacts; Full impact area known and sampled along environmental gradient; Cumulative impacts, e.g. from additive or synergistic sources considered.	(Clark, 2019; Jones et al., 2019)	ISBA/25/LTC/6/Rev.1, para. 13	Tests not required
	Data enable environmental risk assessment	Sensitivities of biota and communities identified; Vulnerability to mining-related hazards identified.	(O et al., 2015; Hauton et al., 2017; Mestre et al., 2017; Kaikkonen et al., 2018; Cormier and Londsdale, 2020)		Not required

(Continued)

TABLE 3 Continued

Topics	Criteria	Example actions and indicators	Example scientific references	ISA documents and other legal instruments	Current requirement for baseline
	Data enable reliable environmental impact predictions	Environmental and technical information and data Enable a reliable prediction of changes and harm to be expected under commercial mining conditions; Estimate recovery times; Ascertain that the plan of work does not induce serious harm.	(Durden et al., 2018; Jones et al., 2020)	ISBA/25/LTC/6/Rev.1, para. 13, annex I, paras 2, 29, 30, 64; ISBA/21/LTC/15, annex I-III para. 10(f)	No comprehensive requirements
	Numerical models	Models are validated by field studies; Models to support prediction of environmental impact, e.g. predictive habitat mapping, plume modelling, toxic effects; Models to assess extinction risks under various management strategies, including various options for the design of protected areas; Modelling undertaken, collaboratively where possible, and linked closely to field studies.	(O´Hara et al., 2016, p. 389; Jones et al., 2020)	ISBA/25/LTC/6/Rev.1, annex I paras. 21, 56, 61	Required
6. Baseline demonstrates good governance	Baseline reflects a precautionary approach	Sources of uncertainty are identified, reduced, acknowledged, quantified, managed, and communicated; Where there are uncertainties, precautionary buffers are to be applied; Conservative estimates to err on the side of caution.	(Underwood and Chapman, 2003; Van Dover et al., 2016, p. 423; Durden et al., 2018; Clark, 2019; Clark et al., 2020; Hyman et al., 2022, p. 607)	Seabed Disputes Chamber, 2011, paras. 125-135; ISBA/25/LTC/6/Rev.1, only annex III (EIS template); ISBA/21/LTC/15, annex I-III, para. 10(e) (only formally with respect to PoW goals); ISBA/17/LTC/7, para. 13 (b)	Not mentioned
	Baseline results are communicated transparently	Results published and/or reviewed by experts in a transparent process: External experts consulted; Effective stakeholder interface is operational; Adjacent coastal States and contractors have been informed of the ongoing activities.	(Van Dover et al., 2016, p. 423; Jones et al., 2020)		Not required for exploration
	Respects measures of other bodies	Identifies and maps existing and planned spatial measures such as marine protected areas, EBSAs and closures for vulnerable marine ecosystems.	(Van Dover et al., 2018; Johnson, 2019; Jones et al., 2020)	ISBA/17/LTC/7, paras. 12, 36(b) (only tasks ISA as a whole in Clarion Clipperton Zone)	Not required from contractors
7. Contractor Environmental management system (EMS)	EMS is established and operational	Each contractor should have an environmental managem ent unit, responsible for the tasks below; Each organization to establish its environmental objectives and targets in line with e.g. ISO14001.	(Swaddling et al., 2016; Durden et al., 2017; Komaki and Fluharty, 2020)	ISBA/27/C/7, para. 14; ISBA/21/LTC/15 para. 6; ISBA/17/LTC/7, para. 41(a) creation of site-specific EMP; ISO 14001:2015 on environmental management[55]; the European Union Eco-Management and Audit Scheme[56]	Only project management required during exploration
	EMS provides for advanced data and information management	Data archival and retrieval system to enable exchange with researchers, regulator, other contractors and reporting.	(Stocks et al., 2016)	ISBA/25/LTC/6/Rev.1, paras. 15(d) (viii), 19-24, annex I paras. 26, 32 (a), (e) 33, 34, 36, 41(a), 54, 55	Required, but no standard procedures
	EMS provides for quality control procedures	Follow best available methodology and the use of an international quality system and certified operations and laboratories; Standard procedures, chain of custody for site identification and sample tracking, taxonomic standardization, sample preservation and archival, and suitable analytical detection limits; Types of data to be collected, the frequency of collection and the analytical techniques should follow the use of an international quality system and certified operation and laboratories.	(Stocks et al., 2016, p. 375)	ISBA/25/LTC/6/Rev.1, para. 19, annex I paras. 31, 32, 47, 54, 55-56.	Required, but no standard procedures determined

(Continued)

TABLE 3 Continued

Topics	Criteria	Example actions and indicators	Example scientific references	ISA documents and other legal instruments	Current requirement for baseline
	EMS ensures best environmental and research practices and techniques are employed	Apply state-of-the-art research standards, Best Environmental Practice, Good Industry Practice, and Best Available Techniques; Avoid research practices which disturb or compromise sites, populations, or processes and aim to collaborate; Provide for independent auditing.	(Devey et al., 2007; OSPAR Commission, 2008; Clark et al., 2016a; Clark et al., 2016b; Swaddling et al., 2016; Gerber and Grogan, 2020)	ISBA/25/LTC/6/Rev.1, annex I para. 54; ISBA/17/LTC/7, para. 38(a); (see also general obligation in ISBA/25/C/WP.1, draft reg 46(2)(b))	Framework or reference for application needed
	EMS provides for transparent public reporting	Assessed and interpreted results are reported; Environmental management activities are made public, e.g. on website; Contractor publishes the environmental sections of its annual and periodic review reports to the ISA.	(Komaki and Fluharty, 2020)	ISBA/25/LTC/6/Rev.1, paras. 24-27, annex I para. 54; ISBA/21/LTC/15, annex I-III paras. 17(a), (b); ISBA/17/LTC/7, paras. 13(f), 41(b)	Transparent reporting required, but not enforced
	EMS provides for adaptive management cycle	Mechanisms available for e.g. designating and relocating preservation/no mining areas and developing Best Environmental Practice	(Durden et al., 2017; Craik, 2020; Gerber and Grogan, 2020; Jones et al., 2020)		Not required
8. Baseline informs Best Environmental Practice	Data suitable to ensure that mine plan and mining practices protect the environment	Baseline provides for spatial planning of the contract area; Baseline establishes relevant data for the selection and design of test and mine sites; Baseline informs Best Environmental Practice.	(Gerber and Grogan, 2020; Haeckel et al., 2020)	ISBA/25/LTC/6/Rev.1, para. 38(l) (no requirement to take account of biotic environment when locating test/mine site)	Not required
	Mine sites do not comprise habitats or species in need of protection	Identifies vulnerable species and habitats in potential mining areas, including in impact areas due to plume dispersal; Identifies whether potential mine sites include unique, rare or threatened habitat or species.	(Ardron et al., 2014; Watling and Auster, 2017; Wagner et al., 2020; Watling and Auster, 2021; Gollner et al., 2021)	(CBD, 2009; FAO, 2009; CBD, 2012)	Not required
9. Baseline collaboration and regional integration	Collaboration with independent research or other contractors	Partnership with scientific community or relevant scientific body; Partnership with other contractors; Respect the PRZ of other contractors; Cooperate to ensure permanent protection of PRZs from mining impacts.	(Van Dover et al., 2016, p. 412; Haeckel et al., 2020)	ISBA/25/LTC/6/Rev.1/, annex I paras. 54, 57-63; ISBA/21/LTC/15, para. 14; ISBA/17/LTC/7, paras. 40, 41(e).	Encouraged
	Data integration	Methodologies are standardised to enable regional assessments, incl. inter-contractor comparisons and compilation of experiences, i.e. a greater database to predict the effects of large-scale disturbance such as from commercial mining of minerals; cumulative impact assessment in related REMP; development of best practices through ISA.	(Jones et al., 2017; Billett et al., 2019; Haeckel et al., 2020)	ISBA/25/LTC/6/Rev.1, para. 15(d); annex I paras. 55-56; ISBA/17/LTC/7, paras. 49-52 (ISA tasks)	Well covered but optional

The Table collates suggestions from the literature as well as existing ISA requirements and recommendations.

Highest standards

Do the reported baseline investigations demonstrate best available scientific information and process?

Applying 'Best Available Techniques'[57] and 'Best Environmental Practices'[58] in protecting the marine environment are general obligations of the ISA, sponsoring States and contractors[59] and Best Environmental Practices are also considered part of the sponsoring state's due diligence[60]. The draft exploitation regulations also introduce a different terminology, requiring environmental decision-making to integrate 'Best Available

57 Defined in ISBA/25/C/WP.1, schedule 1 as: 'the latest stage of development, and state-of- the-art processes, of facilities or of methods of operation that indicate the practical suitability of a particular measure for the prevention, reduction and control of pollution and the protection of the Marine Environment from the harmful effects of Exploitation activities, taking into account the guidance set out in the applicable Guidelines'.

Scientific Evidence'[61], 'including all risk assessments and management undertaken in connection with environmental assessments, and the management and response measures taken under or in accordance with Best Environmental Practices'[62].

As the combination of these terms suggests, it is important for baselines to focus not only on the outcome (the baseline data), but also on the scientific processes to achieve the outcome, including clearly stated research objectives, good experimental design, robust methods, and peer review of results. The limits of the conclusions, uncertainties and knowledge as well as underlying values should be disclosed and discussed, and ideally confirmed or dismissed by independent review to allow for transparent environmental decision-making (Sullivan et al., 2006). Unless such limitations are clearly stated, the predictive value of ecological modelling may be much overstated and provide a false basis for assessing the environmental impacts of mining (Bowden et al., 2021). Therefore, independent expert review and the standardization of procedures and criteria for decision-making on environmental matters are important (Sullivan et al., 2006; Lallier and Maes, 2016). At present, environmental baseline data are not subject to a mandatory review by experts or the public. The LTC may consult external experts when reviewing an EIA, which should include baseline data, and may 'encourage' the sponsoring state to seek views from stakeholders on the EIA[63]. Stakeholder review and independent scientific review should become mandatory under the future exploitation regulations[64], although it is too early to say.

Best practice for deep-sea biological study methods, including cruise and sampling design as well as data management, are compiled by Clark et al. (2016a); Swaddling et al. (2016), and Glover et al. (2016). All research programs should focus on producing best scientific knowledge, and to proceed from a desktop review of the state of research to formulating hypotheses and operational objectives for the research program. This requires defining the survey design, extent of the survey area, ecosystems to be investigated, necessary sampling gear, and of course the required time and human effort (Swaddling et al., 2016).

Reference zones

Do the baseline studies appropriately inform and justify the selection of impact and preservation reference zones?

Baseline investigations have to inform the selection of impact and preservation reference zones in terms of their location, size, number and representativeness. These PRZ and IRZ form part of the before-after-control-impact (BACI) design (Green, 1979; Underwood, 1994; Stewart-Oaten & Bence, 2001; Urban et al., 2021, and articles therein), which, if applied correctly, enables an assessment of environmental impacts in defined places. As Figure 2 illustrates, baselines are the basis for the BACI design as they determine the natural situation prior to the start of activities. Representative impact and control/ reference sites are used for long-term monitoring of the impacts. The ISA's Mining Code already implies a BACI design by requiring (1) baselines, (2) Preservation Reference Zones, (PRZ), (3) Impact Reference Zones (IRZ), and (4) monitoring. In order to operationalize these requirements and fulfill the BACI design, robust baselines are thus essential.

Similarly, in order for BACI to be successful and function as a comparison with impacted sites, the PRZs have to be comparable to the IRZ in all respects except for the impact of the activities. This comparability requires them to be located at similar depth and within the same biogeographic zone, include an equivalent size and density distribution of nodules featuring the same habitat composition and housing self-sustaining populations of the entire species assemblage (McQuaid et al., 2020; Stratmann et al., 2021; Washburn et al., 2021). In other words, the baseline investigations have to demonstrate that the PRZs are truly representative of the IRZ. The IRZ should be located along the gradient of environmental disturbance expected to occur from testing or mining (Billett et al., 2019; Jones et al., 2020) to determine the footprint of biologically relevant mining effects beyond the immediate mine site, i.e. the sediment load, concentration and toxicity of the plumes near the seabed and in the water column, but also vibrations, noise, light and other artificial disturbances. Based on dispersal studies, the baseline data will also have to justify that the PRZ will not be affected by mining and indicate the buffer zones required.

59 See e.g. ISBA/25/C/WP.1, draft regulation 44; see also ISBA/25/LTC/6/Rev.1, annex I para. 54.

60 *Responsibilities and Obligations of States Sponsoring Persons and Entities with Respect to Activities in the Area*, Seabed Disputes Chamber of the International Tribunal for the Law of the Sea (case No 17), 17 February 2011, para. 136.

61 Defined in ISBA/25/C/WP.1, schedule 1 as: 'the best scientific information and data accessible and attainable that, in the particular circumstances, is of good quality and is objective, within reasonable technical and economic constraints, and is based on internationally recognized scientific practices, standards, technologies and methodologies'.

62 ISBA/25/C/WP.1, draft regulation 44.

63 ISBA/25/LTC/6/Rev.1, para. 41(c), (d).

64 ISA, Facilitator draft - Draft regulations on exploitation of mineral resources in the Area: Parts IV and VI and related Annexes, ISBA/27/C/IWG/ENV/CRP.1, 8 February 2022, draft regulation 46bis(4).

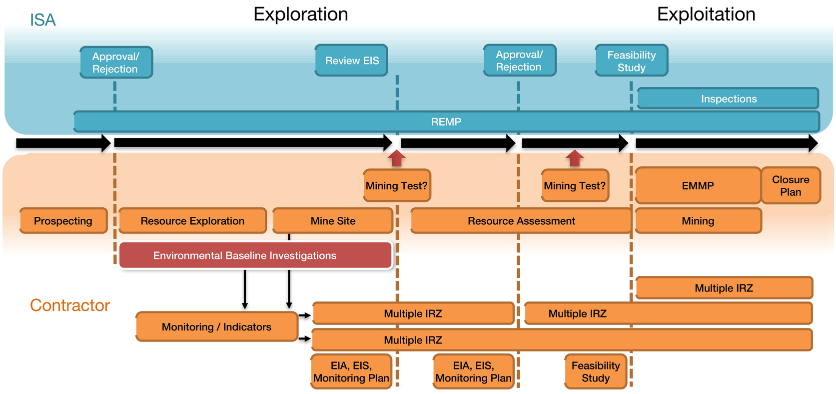

FIGURE 2
The key role of baseline research for environmental management by the ISA.

Monitoring program

Do the baseline studies deliver the necessary knowledge to inform a comprehensive monitoring scheme, including the determination and testing of indicators, their metrics and also thresholds?

Contractors have to monitor the environmental effects of their exploration activities[65]. The monitoring program will have to measure a selected set of most significant indicators[66] as derived from the baseline investigation to allow for temporal and spatial comparisons and help to communicate the results (Elliott, 2011). Figure 2 highlights that baseline investigations during the exploration phase are (1) the crucial first step to determine monitoring sites and suitable indicators, (2) start an iterative process to develop measurable thresholds for harm and serious harm, and (3) inform impact assessment.

An effective monitoring and management strategy will to a large extent also depend on the understanding of the environmental drivers of the variability and patterns in ecosystem processes and functions, biota diversity, dominance and relative abundance of certain taxa as well as community patterns across the microscale to regional gradients in topography, bathymetric and oceanographic variables (Snelgrove et al., 2014). On the seafloor, topographic features, from shallow depressions and slight depth gradients to abyssal hills and seamounts, have a strong influence on the type of substrate, its organic content and hence also community composition. The density of nodule cover of the sedimentary plains is an important habitat factor for the benthic fauna (Amon et al., 2016; Simon-Lledó et al., 2019) as well as for the abundance and community composition of top-level benthopelagic predators and scavengers (Leitner et al., 2017; Simon-Lledó et al., 2020). At regional scale, in the Clarion-Clipperton Zone a decreasing north-south and east-west pelagic productivity in conjunction with increasing water depth results in changed community compositions and lower abundances (Simon-Lledó et al., 2020). Other variables include bottom current speed and direction. When planning a monitoring program, care has to be taken to consider multiple factors for ensuring that the locations of impact and control sites are comparable.

To assist the development of a monitoring program, a qualitative, so-called Level 1 risk assessment could be carried out at the beginning of the exploration phase to identify the most critical issues to be investigated (Clark, 2019, p. 461 and literature cited). To enable quantitative risk and impact assessments and the development of measures, the formulation of indicators and other monitoring parameters has to be SMART[67], and they should be (Elliott, 2011):

a. anticipatory to provide an early warning of deterioration;
b. biologically important, applicable and indicative over space and time;
c. scientifically sound and measurable and quantifiable over space and time;
d. sensitive to pressures and responsive to management; and
e. socially relevant and interpretable by stakeholders.

65 ISBA/19/C/17, regulation 32; ISBA/25/LTC/6/Rev.1.

66 An indicator is a parameter, or a combination of parameters, chosen to represent (indicate) a certain situation or aspect and to simplify a complex reality, see (Zampoukas et al., 2013).

67 SMART stands for specific, measurable, achievable, realistic, and time-bound; see e.g. (Rice et al., 2005).

However, some elements that are important for the monitoring program have yet to be decided, such as the definition of 'serious harm' (Levin et al., 2016). It is agreed that seabed mining must provide for effective environmental protection and not cause serious harm[68] but it remains unclear how the ISA will determine whether this threshold is reached or crossed. For example, it has to be considered how rare species could be represented, and what the role of rare species is when determining the level of harm (Chapman et al., 2018). Comparable to fishing (Jennings and Dulvy, 2005), for each indicator an unexploited reference point as well as a precautionary limit (harm) and a limit (risk of serious harm beyond which operations are to be stopped) will have to be determined. In other words, the baseline investigations have to determine which indicators should be sampled for the monitoring program (Rice and Rochet, 2005) to be able to compare them later on.

EIA

Are the baseline studies suitable to inform a prior EIA of commercial mining with a degree confidence?

Any mining-related changes in the environment, as monitored in the area impacted by mining during and after the operations take place, will have to be assessed against what would be the "normal" environmental state. This is the major task of environmental baseline investigations, which have to provide reliable information on the natural spatial and temporal patterns of ecological community development and interaction. Due to the long-term patterns of climate oscillations and trends, these baseline parameters may have to be investigated over a long time period, i.e. throughout the exploration period. In addition, the baseline studies must provide the contractor and the regulator with a first set of possible indicators and thresholds which will enable the measurement and monitoring of the effects of mining. Most importantly, together with well-designed testing, monitoring and impact assessment of the tests, the baseline should enable the contractor to make accurate predictions on the degree of environmental harm to be expected.

Based on real data, modelling can further extend the temporal and spatial understanding of environmental interaction. For example, McQuaid et al. (2020) used data on topography, particle flux to the seafloor and nodule abundance as a proxy to classify and predict the regional distribution of habitats and associated biological communities across the Clarion-Clipperton Zone. For the same region, Wedding et al. (2013) offered modelling to support the establishment of APEIs. Similarly, Uhlenkott et al. (2020) modelled the predictive distribution of meiofauna communities on the scale of a contract area. The classification, if ground-truthed and refined for each contract area, will help to establish representative preservation reference zones as control sites when monitoring the effects of mining activities. However, Bowden et al. (2021) caution that the predictive value of habitat models is usually more limited than extrapolation from photographic seabed identification, because uncertainties of modeled variables cannot be quantified. This includes, for example, the certainty of taxonomic identification, the lack of some ecologically important variables that influence distributions, the lack of confirmed absence data for most taxa, and modeling at a rather coarse taxonomic resolution. For this reason, the LTC also requires contractors to confirm photographic species identification by collection and permanent storage of specimens[69]. Overconfidence in modelling, and underestimation of uncertainties is likely to result in poor decision-making (Regan et al., 2005; Bowden et al., 2021).

Environmental baseline studies, where possible undertaken collaboratively with researchers and other contractors, will feed conceptual and numerical models which can assist in the prediction of at least some aspects of environmental harm from commercial mining operations, although upscaling (i.e., extrapolation beyond the period or area measured) is a contentious issue and needs detailed verification. Overall, the environmental baseline studies should provide as much certainty (sensu Clark, 2019, p. 460) as possible about the environmental effects of permitting a mining operation to take place, so that decision-making can be best informed.

Good governance

Do the baseline studies reflect good governance principles, such as transparency and participation?

As the international seabed is the common heritage of humankind, contractors should be required to implement good governance in their baseline investigations. This is somewhat separate from the call on the ISA as a whole to follow good governance practices, an ambition for which the ISA has been found lacking (Ardron, 2018; Ardron, 2020; Woody and Halper, 2022). Good governance is generally understood to include at a minimum transparency, open communication, and stakeholder involvement in decision-making processes (Ardron, 2020). In the context of DSM, precautionary management of the seabed may be added to the list, as a direct obligation of contractors (and the ISA and states)[70]. Implementing the precautionary principle includes acknowledging uncertainties, risks, and knowledge gaps and seeking to close them (Jaeckel, 2017a). In fact, baseline investigations offer a

68 UNCLOS, article 145; ISBA/19/C/17, regulation 33.

69 ISBA/25/LTC/6/Rev.1, annex I para. 41(a).

prime opportunity to address and reduce uncertainties in line with the precautionary principle.

As outline above, baseline data should be subject to expert review. To that end, contractors' annual reviews should be published on the ISA website, which will also support transparent reporting and compliance. Similarly, EIAs by contractors should provide for early and meaningful stakeholder participation, including an opportunity to review the baseline data that informs the EIA. Lastly, an open and transparent dialogue between contractors and the LTC will help guide contractors during their baseline studies and have positive side effects on the governance structure of the ISA as a whole.

Contractor environmental management system

Does the contractor manage its baseline investigations, analysis and reporting in a dedicated environmental management unit or system?

An environmental management system (EMS) as a company-internal quality control system has been recommended by the International Marine Minerals Society and supported by the World Bank (Durden et al., 2017). The value of EMS is explicitly recognized in the environmental management plan for the Clarion-Clipperton Zone, though only as a non-binding management objective[71]. That plan requires contractors to follow Standard ISO 14001,[72] which includes establishing a process for public consultation and making all efforts to limit and control the environmental effects of the contractor's DSM activities, as well as seeking continuous performance improvements.

Essential elements of a contractor-led EMS include the establishment of environmental objectives and targets in line with ISA and states' global and regional goals and strategies (Tunnicliffe et al., 2020), an effective data and information management system (Stocks et al., 2016), including quality control (Stocks et al., 2016), development and implementation of best environmental practice and best available techniques (Clark et al., 2016b; Swaddling et al., 2016; Gerber and Grogan, 2020), as well as ensuring fit-for-purpose reporting and adaptive management of activities (Durden et al., 2017; Komaki and Fluharty, 2020). Moreover, baseline data should be published in (open access) peer-reviewed literature to ensure its quality has been independently verified, and the data is stored in an open, accessible, and safe manner, can contribute to capacity building, and is accessible for scientists and contractors conducting regional or connectivity studies.

Conclusion

While improvements have been made, there continues to be a lack of environmental baselines for deep ocean sites that are being eyed for mineral mining (Amon et al., 2022). In fact, the LTC has repeatedly called on contractors 'to include in the annual reports a review of how the baseline data are building up to a level sufficient to support a robust environmental impact assessment'[73]. This paper is designed to help with such a review by demonstrating the key role of baselines as the foundation for assessing and managing the environmental impacts of DSM. We set out criteria for evaluating the quality of a baseline in Table 3 and the corresponding text, which could be further refined by the LTC. These criteria are compiled from peer-reviewed literature and build upon existing ISA requirements. The criteria are designed to ensure contractors, sponsoring States, the ISA, and the public can evaluate the quality of baselines ahead of any formal environmental impact assessments, which will increase confidence in the decision-making process for both current and future generations. This is arguably especially important in light of the minerals on the international seabed being the common heritage of humankind[74].

At present, there are no publicly available criteria for the ISA's Legal and Technical Commission to evaluate the quality of a contractor's environmental baseline. The criteria set out in this paper seek to address that gap and could inform the ISA's future exploitation regulations:

1. Specifically, the future exploitation regulations should require the BACI approach and the ISA should specify in detail the data and information required from baseline studies and the procedures to be complied with, including those for a reliable and replicable application the BACI approach for assessing mining impacts.

70 ISBA/19/C/17, regulation 31(2); *Responsibilities and Obligations of States Sponsoring Persons and Entities with Respect to Activities in the Area*, Seabed Disputes Chamber of the International Tribunal for the Law of the Sea (case No 17), 17 February 2011.

71 ISBA/17/LTC/7, paras. 40-41.

72 ISO 14001 is the international standard for environmental management systems (EMS), accompanied by ISO 14004 Environmental Management Systems – General Guidelines on principles, systems and support techniques. Available from the website of the International Organization for Standardization at: http://www.iso14000-iso14001-environmental-management.com/.

73 ISA, Report of the Chair of the Legal and Technical Commission on the work of the Commission at the second part of its twenty-sixth session, ISBA/26/C/12/Add.1, 25 September 2020, para. 14; See also ISA, Report of the Chair of the Legal and Technical Commission on the work of the Commission at the second part of its twenty-fifth session ISBA/25/C/19/Add.1, 11 July 2019.

2. Moreover, a compulsory test mining stage would greatly increase the information available to contractors, the ISA, states, and the public in order to assess and discuss the impacts of DSM.
3. Lastly, a standardized methodology, statistical robustness and (public) transparency should be critical parameters to help characterize a high-quality baseline.

The criteria set out in this paper will support precautionary management of DSM by contributing to a meaningful assessment of the risks and impacts of DSM. The criteria will also help to achieve the aim articulated by the Seabed Disputes Chamber, namely the 'uniform application of the highest standards of protection of the marine environment'[75].

75 *Responsibilities and Obligations of States Sponsoring Persons and Entities with Respect to Activities in the Area*, Seabed Disputes Chamber of the International Tribunal for the Law of the Sea (case No 17), 17 February 2011, para. 159.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material. Further inquiries can be directed to the corresponding author.

Author contributions

All authors conceived the manuscript and all authors provided input on all sections and participated in the editing and final preparation of the manuscript. SC conceived and created the list of criteria for evaluating baselines (see Table 3).

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Conflict of interest

For information, SB was employed at the ISA Secretariat as Scientific Affairs Officer from 2013 to 2018, which we do not consider posing a conflict of interest.

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Barriers to coastal planning and policy use of environmental research in Aotearoa-New Zealand

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Identifying barriers to the effective use of science in coastal management of Aotearoa-New Zealand is easy, due to the present lack of complicated governance and management structures, coupled with an emphasis on funding science that includes pathways to implementation. This opinion piece discusses four areas that still hinder effective use of science, all of which are likely to be problematic for other countries. We initially focus on why the science may not be used related to: misunderstandings (linguistic and conceptual differences including indigenous world views); timing of information delivery; uncertainty surrounding the information (knowledge limitations and funding); and top-down constraints (legal systems, politics and institutional objectives). We use Aotearoa-New Zealand examples to demonstrate the barriers operating within each area and discuss three potential solutions. Importantly our analysis indicates that researchers alone cannot transcend these barriers; rather, we need to work as part of an ecosystem, requiring commitment from all society, extending beyond the usual suspects (management agencies). We believe that ecological and systems education from junior school levels through to universities have an important role to play in setting the context to overcome current barriers.

KEYWORDS

management, planning, science provision, education, science-policy liaison, co-development

1 Introduction

Around the world there is recognition that, for coastal planning and management to achieve good environmental outcomes, there is a need for effective use of relevant science (Nursey-Bray et al., 2014; Dale et al., 2019). Unfortunately mechanisms to achieve this are largely lacking (Karcher et al., 2022). Two important factors should enhance the ability of Aotearoa-New Zealand to manage its coastal waters: no internationally shared responsibilities; and a fairly flat management hierarchy (national or sub-national within a national framework). However, policy and planning within both national and sub-national government agencies frequently appears to work in a vacuum, relatively uninformed by current, and sometimes even past, research (Gluckman, 2013; Urlich, 2020a). For example in Aotearoa-New Zealand, bottom-trawling and excessive terrigenous sediment inputs to coastal waters continue despite decades of national research demonstrating adverse effects (e.g., Shears and Babcock, 2002; Thrush et al., 2004; Pratt et al., 2014; Urlich, 2020a). Moreover, although government reports have lately summarized and described cumulative effects of multiple stressors on marine ecosystems (e.g., Ministry for the Environment and Statistics NZ, 2019), activities continue to cause ongoing adverse effects to marine biodiversity and ecosystem processes, apparently, to outsiders, with the permission of central (national) and regional (sub-national) agencies.

International literature has focused on researchers needing to improve their science communication styles, create effective knowledge exchange and increase the accessibility of information (Cvitanovic et al., 2016; Fernández, 2016; Greenhalgh et al., 2022). Frameworks have been developed to assist with this e.g., CRELE (credibility, relevance, legitimacy) and ACTA (applicability, comprehensiveness, timing, accessibility) to guide information presented at the interface of science and policy (Greenhalgh et al., 2022). Our experience as marine ecologists working in the field of disturbance and recovery highlights that this may be a simplistic view. For example, those who could ensure that the problems and solutions identified by researchers are used in plans, policies and decision-making frequently say that scientists focus on unnecessary detail and sensitivities, rather than producing lay summaries with clear understandings of risks and benefits of different options. However, this 'unnecessary' detail frequently provides the information needed to accurately contextualize and detail the risks and benefits. Similarly, planning legislation often uses the existing, and often degraded, ecological baselines from which to assess effects of activities and to measure change, failing to recognize how these baselines have shifted (e.g., Urlich and Handley, 2020). This constrains the scope in decision-making, planning and policy to facilitate restoration of degraded habitats. Scientists have sometimes not helped this situation, with overly cautious advice in the absence of complete information (Hendy, 2016).

Beyond the obvious differences in language and underlying concepts between marine researchers and those who could use the information, we feel that there are also many other issues. In this opinion piece, we begin by discussing misunderstandings caused by different use of languages and concepts. We also discuss: the difficulties of getting information to the right people at the right time; the effect of uncertainty surrounding the information (knowledge limitations and funding); and top-down constraints (laws, politics and agency objectives). We use Aotearoa-New Zealand examples to highlight these issues and discuss potential solutions. Our focus is not just on policy but also on planning and decision-making.

2 Issues

2.1 Misunderstandings

2.1.1 Scientific concepts

Translating the complexity of social-ecological systems, and their associated uncertainties, into accessible language for both science and non-science (i.e. policy, planners) audiences is critical (Le Heron et al., 2016; Gluckman, 2017). Over-simplification of complex ecological systems may result in failing to consider key environmental drivers or anthropogenic stressors, and incomplete understanding of systems dynamics and resilience (Scheffer et al., 2001, Lundquist et al., 2016a). In the Introduction we highlighted a problem associated with shifting baselines, but there are other essential science concepts that are often not considered by policy makers. For example, marine spatial planning has been influential in conveying the need to explicitly consider the mismatches between planning, decision-making and management with the ecology and environment. However, the realities of temporal variability in ecologies and their dynamics are important issues that are not well conveyed to the non-expert in such plans. Increasingly we are observing tipping points and thresholds in degradation of marine species and systems (Conversi et al., 2015). These abrupt, and often unexpected, changes mean that the operational practice of monitoring, predicting the need for action based on dose-response type relationships and having at least some time for institutions to make decisions around management frequently no longer work. Instead, we not only need to explain that a threshold may be approaching (despite no signs of any effect), but also that timely action is required (Hewitt and Thrush, 2019). Furthermore, when a threshold is passed and ecological states are degraded, then we need to predict whether recovery is possible once management actions to aid recovery are implemented and explain the likely time scales of any lags in recovery. In general, threshold responses and slow recovery appear much easier for Aotearoa-New Zealand indigenous communities (iwi (tribal), hapū (subtribal)) and the wider public to understand, and for them to support timely management actions, than for most businesses and government management agencies (McCarthy et al., 2014). This has been demonstrated in various ways and places around Aotearoa-New Zealand. For example, recent iwi and public pressure to close scallop fisheries, and over 50,000 signed a petition to ban bottom-trawling on seamounts in 2020, https://www.rnz.co.nz/news/political/430888/bottom-trawling-petition-delivered-to-parliament.

Variability in coastal ecosystems is also viewed differently among planners, managers and ecologists. In Aotearoa-New Zealand, temporal variability in coastal ecological and environmental responses can be particularly high because the southern decadal oscillation and El Niño/-La Niña weather patterns have a strong effect on physical, chemical and biological parameters (Hewitt et al., 2021). The problem here is not convincing people that climate variability occurs, as in Aotearoa-New Zealand the El Niño or La Niña statistics are frequently reported on during the year, rather it is convincing them that this does not preclude understanding what is going on, and that small effects within this climate variability can still drive large changes.

2.1.2 Indigenous world views

Many countries need to work with Indigenous people when managing the environment (e.g., Soumi in Finland and Norway, First Nations in Canada, Aborigines in Australia, Mapuche in Chile etc.). Aotearoa-New Zealand is increasingly seeking to address Indigenous world views in its environmental management, with Māori concepts, such as *kaitiakitanga* (guardianship or stewardship for future generations) and *whakapapa* (ancestral connections with the environment) being incorporated in environmental management (Dick et al., 2012). Rivers and mountains have been given status as legal persons in an attempt to recognize in law the ancestral relationships of Indigenous peoples with these ecosystems, and to change the power relationship between Indigenous people and government agencies (Macpherson and Clavijo Ospina, 2018). Community activism for legal rights for rivers and ecosystems has occurred in countries as diverse as Mexico, the United States (US), Columbia and Bangladesh. *Ki uta ki tai* is a holistic concept that represents the connectivity within and between ecosystems for example from the mountaintops to the sea, and the concept showcases that, from a holistic Māori viewpoint, management should recognize the connections between land and sea, and that humans are embedded in the ecosystems (Tipa et al., 2016; Hepburn et al., 2019).

While we can attempt to translate these concepts into other cultural contexts, we often lose the depth of the relationships between Indigenous peoples and nature. Indigenous worldviews often more readily recognize environmental degradation, but existing systems often lack structures to incorporate indigenous knowledge into decision-making, and entrenched power dynamics mean that the role of indigenous peoples, their knowledge and their worldviews are often not recognized as equal to scientific evidence (Ens et al., 2015).

2.1.3 Sectoral and discipline linguistic differences

Languages also differ between different groups interested in environmental management, with different bottom lines (economic, societal, cultural, environmental) based on their key values. Terminology can appear similar, but when used in the context of a particular industry, meanings can differ. For example, the terms "baselines", "business as usual", and "sustainability" all have different interpretations across industry, government and environmental sectors. Similarly, many terms can become politically charged within a particular group due to perceived biases against the values of that group, and quickly fall out of favour. For example, marine protected area (MPA) and marine spatial planning (MSP) are terms that include a wide spectrum of approaches, but contentious debate is often based on a single approach. MPAs may be spoken of as if they only consist of fisheries no-take, although in Aotearoa-New Zealand, and many other countries, there are a range of protection levels (Douvere, 2008; Day et al., 2012; Grorud-Colvert et al., 2021). In some countries (including European Union countries), marine reserves (a subset of MPAs) prohibit any resource extraction (e.g., OSPAR, 2016, and sections of the Great Barrier Reef Fernandes et al., 2005). MSP may be relegated to simply being spatial allocations of various extractive uses (businesses) or always resulting in the production of an MPA (fishers), whereas it can be an extensive exercise with multiple stakeholders and create a variety of management options (Lundquist et al., 2005; Sayce et al., 2013; Davies et al., 2018a). A recent marine spatial planning initiative in Aotearoa-New Zealand (Sea Change Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan) at one stage drafted over 180 recommendations spanning multiple management categories (e.g., Marine protection, Protected Species, Aquaculture, Habitat restoration, Biosecurity, Ahu Moana, Fisheries management, Governance).

Ecosystem-based management (EBM) is another term that has evolved over recent decades from a simple approach considering the environment to a complex concept that also covers people, intergenerational use and knowledge uptake etc (McLeod and Leslie, 2005; Long et al., 2015; Hewitt et al., 2018). Again, the term is interpreted differently by different people. For example, in many areas around the world, the fishing sector has introduced the concept of EBFM, which is typically defined as fisheries management that takes into account environmental and ecological impacts on an ecosystem, and the interconnectedness and interdependence of various components of the ecosystem, but does not take into account the needs of other users.

Knowledge gathering approaches and analyses also vary across disciplines such as biophysical sciences, indigenous and local knowledge, legal, social, and economic data (Allison et al., 2019). This also affects use of the term "best" available information, which can be found in a number of New Zealand policies and statutes (Davies et al., 2018b), with what is "best" for one situation not being the most relevant in another (Rudd et al., 2018).

Finally, probably the greatest variability in expectations between groups is generated by use of the terms "degraded", "healthy" and "desired states". In Aotearoa-New Zealand, policy is leaning towards defining environmental health states, measured by nationally consistent methods, and encouraging locally derived targets or bottom lines based on local values. Even this is not easy as, amongst ecologists, health can be variously associated with ecological functioning, multi-functionality, network connectivity, or animal or plant community-based health indices.

2.2 Mismatches between timing of information need and its availability

At present in Aotearoa-New Zealand, scientists need to time delivery of information to match policy and planning needs. Some of these needs are cyclic, with the timing dependent on the relevant government agency. For example, the New Zealand Coastal Policy Statement (NZCPS, see section 4.1) is mandated to safeguard the integrity, form, functioning and resilience of the coastal environment and sustain its ecosystems, including marine and intertidal areas, estuaries, dunes and land and is reviewed at the discretion of the Minister of Conservation. Since this policy statement first came into force in 1994, it has been reviewed twice (with no amendments made) and replaced once (in 2010). Other central government agencies have less clear work structures, driven by funding and political imperatives. For example, Fisheries New Zealand conducts single species fish stock assessments for the most important species every 3 to 7 years, but some stocks may be assessed much less frequently (Cryer et al., 2016; Gerrard, 2021), if at all (Ministry for the Environment and Statistics NZ, 2022). Stock assessment funding is allocated by fisheries working groups, with stocks receiving assessments driven by working group priorities, and tied to economic value. The Ministry for the Environment conducts reviews of, and produces new, national policies with no set time periods for review. Regional Councils (the local government agencies) are charged not only with implementing the NZCPS, but also creating a coastal plan for their region and reviewing this at least every 10 years. Less than half have implemented the 2010 NZCPS and all 16 regional management agencies work to their own timetables (Urlich et al., 2022).

Timing of policy windows was also recognized by Karcher et al. (2022) as a key factor in uptake of science into environmental management internationally. They also rightly recognized a "time for action" where knowledge is presented at the time when people are willing to change and create improved environmental outcomes. All of this means that researchers have to be nimble in adjusting their research schedules to have knowledge ready for use and contacts that will forewarn them about when it will be needed.

2.3 Knowledge uncertainty

2.3.1 Data limitations

A major challenge for all those seeking to manage the marine environment is knowledge sufficiency. Around the world there is strong variability in what is known about coastal marine species and habitats ranging from well-studied areas of the Western European countries and North America, through to less well-studied areas around South-east Asia, South America and the Pacific (Costello et al., 2010; Lundquist et al., 2016b). In Aotearoa-New Zealand there has been a sustained under-investment in nationally coordinated marine environmental monitoring (Parliamentary Commissioner for the Environment, 2020). Regional Councils and central government have responsibilities for providing state of the environment data to the Ministry for the Environment and Statistics New Zealand. This is used for national reporting but the list of variables monitored in common is not comprehensive, varies spatially, and often is insufficient to inform long-term change, or in some cases to confirm that changes have occurred. For coastal regions, even basic oceanographic information such as seawater temperature is not available to inform how systems are changing over time. The lack of consistent data collection challenges our ability to determine when to alter plans, policies or decision-making criteria (Parliamentary Commissioner for the Environment, 2019).

While measures of ecosystem health and environmental baselines provide important context for management decisions, understanding shifts in ecosystem function is critical. Worldwide there is a lack of knowledge around the functional responses of coastal ecosystems to cumulative stressors. Policy- and decision-makers, as well as researchers, resource managers, businesses or interested communities, struggle with this lack of knowledge and the uncertainty it creates. Lack of information is often used to stall creation of policy and decision making, or even used by two opposing sides to demonstrate what sort of decision should be made. Local communities often want information about effects on their local species or places, and may mistrust generalities derived from elsewhere. While functional shifts require an understanding of context, ecological principles are evolving that provide perspective on the detail and facilitate action in the face of uncertainty.

"Adaptive management" is an approach frequently suggested to deal with lack of data. In Aotearoa-New Zealand, for activities that come under the Resource Management Act, this term is taken to mean that a limited form of the activity is allowed if there is good baseline information about the receiving environment, monitoring of effects can be undertaken and thresholds can be set for stopping the activity before effects become irreversible (Supreme Court, 2014). Where responses to activities are approximately linear (that is remedial action can take place before effects become overly damaging, and effects can be remedied before becoming irreversible), this is an appropriate way to gain more data without delaying decisions. Unfortunately, if strongly non-linear responses, thresholds or tipping points occur, and there is general lack of knowledge of appropriate thresholds, this method is inappropriate and the precautionary principle should operate.

In our experience, the precautionary principle suffers from a lack of translatability resulting in uncertainties for both science and non-science. For example, precautionary for who or what, and precautionary in the face of what is usually not well specified. This linguistic uncertainty allows cautious environmental management to be challenged on the basis that information is incomplete, such as the overharvesting of desirable fish species (High Court, 2021).

2.3.2 Research funding

Economically, Aotearoa-New Zealand is a small country, with a population of 5.1 m (per capita GDP is 21st in the OECD), but has the world's fifth largest exclusive economic zone and the 9th largest coastline (~15,000 km) in the world. Aotearoa-New Zealand's national investment in research and development is considered low at 1.4% of GDP in 2018 when compared with an OECD average of 2.4%. In 2018 total expenditure on environmental research was NZ$362 million (Parliamentary Commissioner for the Environment, 2020). How much of this investment is being spent on marine ecosystems is not transparent, due to how funding categories for environmental research are reported, but it is likely to be considerably smaller than funding toward terrestrial and freshwater ecosystems, a reflection of their 'economic value' to society. Most of the government's research funding works on an exceeding low trust model. Allocation of funds to fundamentally understand our natural environment (e.g., Marsden Fund) versus strategic grants to support environmental management (e.g., Endeavour Fund) all suffer from a lack of relevant scientific assessment processes and represent a very small part of the governments research investment. However, the National Science Challenges (formed in 2014, https://www.mbie.govt.nz/science-and-technology/science-and-innovation/funding-information-and-opportunities/investment-funds/national-science-challenges/) were an experiment in more collaborative and mission-led research, on pressing issues identified by the public. One of these Challenges was given the objective, by the Government, of "enhancing the use of New Zealand marine resources within environmental and biological constraints". Workshops with marine researchers determined that an appropriate approach to this objective would be to undertake the underpinning research to support the use of ecosystem-based management. This was accepted by the government funding agency and in 2014 Sustainable Seas (a partnership of research institutes and universities) gained funding for 10 years (in two 5-yr phases) with a vision of "Aotearoa New Zealand has healthy marine ecosystems that provide value for every New Zealander". Sustainable Seas funds research projects using a mainly negotiated process supporting bringing together the best teams, following a research agenda initiated by a leadership team and accepted by a governance group, stakeholder panel and Māori advisory group (Kāhui).

In the context of connecting scientific knowledge to management action the real question is whether the funding structure is optimized to grow the knowledge base and inform environmental management in a manner timely for achieving good environmental outcomes (see section 3.2).

2.4 Top-down constraints

2.4.1 Legal and political constraints

Researchers frequently may not fully appreciate the extent to which policy, plans and decisions are constrained by the law and political considerations. In Aotearoa-New Zealand there are many pieces of legislation affecting the coastal environment (see Figure 1), for example: the Māori Fisheries Act 2004; the Conservation Act 1987; the Marine Mammals Protection Act 1978; the Marine Reserves Act 1971; and the Marine and Coastal Area Act 2011. The Fisheries Act (1996) applies to all fishing activity within freshwaters, the Territorial Sea and the EEZ, with its purpose 'to provide for the utilization of fisheries resources while ensuring sustainability', whereas regional councils are legislatively tasked to manage activities including aquaculture and the environmental effects of fishing on biodiversity out to 12 nautical miles, but not fisheries allocation or access issues.

The Resource Management Act (RMA) is the major legal instrument for much of Aotearoa-New Zealand's coastal management (to 12 nautical miles offshore). Decision-making under the RMA is guided by national policy statements; in the coastal environment the New Zealand Coastal Policy Statement (NZCPS) provides decision makers with specifics on how the RMA is to be applied. A large body of case law has further defined how the RMA is interpreted. Since the passage of the

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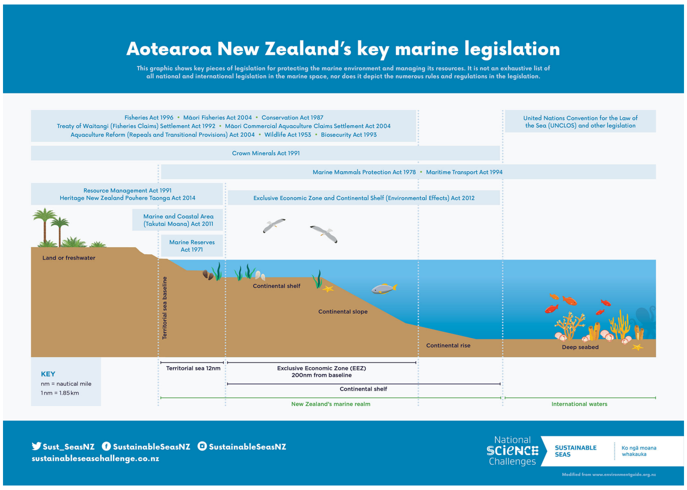

FIGURE 1
Summary of Aotearoa-New Zealand major legislation related to the marine area (from 19679-Sustainable-Seas-Marine-Legislation-Graphic-Nov20-FINAL.PNG (1920×1358) (sustainableseaschallenge.co.nz)). Grey shaded horizontal bars show the areas (terrestrial, coastal, territorial sea, exclusive economic zone and international waters) that the pieces of legislation refer to. Horizontal lines at the bottom define the national and international realms.

RMA in 1991 it has remained controversial, with complaints ranging from lack of protection of the environment, lack of clarity for decision makers leading to legal challenges, and for making development of resource use slow and expensive (Brown et al., 2016; Randerson et al., 2020). Both major political parties have seen the need for reform and the government of the day is presently considering replacement legislation.

These laws, regulations, policies and plans constrain the policies, plans and decisions made by central and regional government agencies. Inevitably, they contain phrases that allow for differing interpretations. For example, use of the word "should" rather than "must" creates options of whether to take an action or not, as does following the word "must" by "take into account" or "consider". Further many words are left undefined, for example, "cumulative effects", "precautionary", "adverse effects" and even "maintenance of biodiversity". For example, the RMA states that cumulative effects should be taken into account.

Local government agencies are overseen by locally elected representatives whose politics determine the balance between economic, social or environmental imperatives. The balance achieved in the decision or policy is not always transparently communicated and biases can be created (or in the case of existing uses) maintained. This balance is not always supported by public surveys or submissions and communities can surprise agencies in their desire to see environmental improvements (Spash, 2006). For example, in Aotearoa-New Zealand, the upgrade of the principal wastewater treatment plant in Auckland was undertaken between 1998 and 2005, at a cost of $450 million. A survey and public workshops around costs (reflected in rate increases) and options for treatment and disposal resulted in support for high quality tertiary treatment. Similarly, petitions to Parliament calling for controls on single-use plastic bags had attracted over 103,000 signatures prior to 2018. This resulted in the Ministry for the Environment seeking feedback on a proposal to implement a mandatory phase out through a submission process. Total submissions received were 9,354 submissions with the majority supporting the proposal (Ministry for the Environment, 2018).

Further constraints for decision-making result from Aotearoa-New Zealand's reliance on case law. Local decisions are frequently challenged in the Environment Court, where judges will often set precedents for future. Interestingly, this is one area where researchers can have an influence and information is actively sought (Urlich et al., 2022).

However, this court-based process does mean that new policies and plans based on environmental research (even when supported by local politicians and agencies) can be slowed. Industries with investment based on previous compliance may become litigious if their operations are then to be constrained (e.g., by replanting controls on erodible slopes to reduce excess fine sediment discharge into freshwaters and (finally) estuaries). Litigation that scrutinizes the science and cross-examines the scientist and their models is important but the burden of proof often falls on the regulator (or iwi, hapū, local community) to convincingly demonstrate the need for change. Given the political implications, scientists within, or contracted by, regulatory agencies can be understandably cautious in their advice, unless the research clearly demonstrates causal attributions. This is difficult to do where there are multiple stressors from different, often diffuse, sources, such as a range of waste nutrients from intensive agriculture discharged into freshwaters. Policy makers are understandably nervous about scientific uncertainty if changes are to be made to regulations. Consequently the regulatory system is reluctant to shift from the status quo, and the expense of investment in science to determine causality becomes prohibitive.

Recently some regional councils have attempted to move the status quo and manage the effects of bottom trawling and shellfish dredging on biodiversity of the seafloor. This issue came to the Environment Court and eventually was determined by the Court of Appeal finding that a regional council may control fisheries, provided it does not do so to manage those resources for Fisheries Act purposes. This means it may control relevant activities for biodiversity purposes. However, implementation of the court decision is proving problematic as most regional councils have yet not introduced measures to regulate the environmental effects of fishing, and some are awaiting the result of legal challenges (Urlich, 2020b; Urlich et al., 2022). Making ongoing budgetary provision for funding the survey and monitoring of marine biodiversity is also problematic.

Increasingly in Aotearoa-New Zealand resource plans and policies need to reflect the interests of Māori as Treaty of Waitangi (1840) partners alongside the Crown and its representatives. Māori practice and knowledge of kaitiakitanga (Kahui and Richards, 2014) are essentially holistic and strongly based on Māori tribal (iwi and hapū) knowledge (Mātauranga). Mātauranga is founded on place-based dependencies and the interactions and relationships with the environment. Policies, plans and decisions increasingly need to demonstrate the use of Mātauranga in their development and, preferably, embed principles of co-governance. This opens up new opportunities to link traditional knowledge and different world views into the development of environmental policy and actions.

2.4.2 Institutional objectives and silos

Environmental domain (land, freshwater and sea) and geographic scale-specific management structures are common around the world, and Aotearoa-New Zealand is no different (Alexander and Haward, 2019; Flannery et al., 2019, Macpherson et al., 2021).

There are three major central government agencies with responsibility for the environment. All biosecurity issues are dealt with by the Ministry for Primary Industries, who also have oversight of Fisheries New Zealand. The Ministry for the Environment (MfE) has a direct role in reflecting the relationship between the Crown and Māori under the Treaty of Waitangi and in monitoring the outcomes of environmental decision-making. Under the RMA, MfE works with other government agencies to develop national policy statements and national environmental standards. The Department of Conservation (DOC) is charged with promoting conservation of natural and historic heritage with specific roles in conserving protected indigenous marine species (identifying and assessing the adverse effects of fishing on marine mammals and seabirds) and threatened non-protected species. It also has specific responsibilities for coastal management (under the RMA), including preparation of the New Zealand Coastal Policy Statement; facilitating approval of all regional coastal plans by the Minister; deciding on consents for Restricted Coastal Activities; planning and consent responsibilities for the offshore islands; and calling-in consent applications of national significance in the coastal marine area.

Regional Councils (and in a few cases local unitary authorities) manage other activities in terrestrial areas, freshwater and the Territorial Sea. Their interests include water provision, water treatment, parks, land development zoning, ports, airports, etc. Council boundaries are generally aligned with catchments, but can divide up marine systems.

There are two other agencies that also contribute to coastal management. The Environmental Protection Authority (EPA), established in 2011, has oversight of international obligations under the UN Framework Convention on Climate Change, the Kyoto Protocol, the Vienna Convention, and the Montreal Protocol. In the marine environment, it has specific management functions in the EEZ, but for the territorial sea it only evaluates nationally significant proposals. The Parliamentary Commission for the Environment is an independent agency, headed by a commissioner appointed by the Governor General (as advised by the House of Representatives). The Commissioner's role is to select, review and provide advice on environmental issues and the system of agencies and processes that manage the environment. Recent reports include "Managing our estuaries" August 2020 and "A review of the funding and prioritization of environmental research in New Zealand" December 2020.

The different objectives and statutory requirements of different government agencies can result in policy settings and research priorities in one agency conflicting with another's objectives or result in significant areas falling through the gaps (e.g., estuaries management as highlighted by the PCE "Managing our estuaries" August 2020). Information sharing between central government agencies is only mandated in very few instances, e.g., the Fisheries Act specifically brings DOC into the assessment of fishing impacts on seabirds and mammals. Information sharing is beginning to be more common between agency scientists, for example, the recently created Marine Science Advisory Group formed between MfE, DOC and MPI to classify seafloor habitats. Co-governance initiatives with Māori, due to their emphasis on holistic understanding, are likely to aid in decreasing institutional silos at multiple levels.

Even within organizations, barriers can form. Central and regional government agencies generally have teams (policy and scientists) and management plans grouped around terrestrial, freshwater and marine areas. These artificial boundaries disrupt management of cumulative stressors in coastal ecosystems that often result from sediments and nutrients that are transported from land through freshwater streams. For example, recent regulatory plan changes to regional council catchment management plans in Canterbury omitted to control the effects of catchment pollution on an estuary, which was required by the existing coastal plan as well as the NZCPS (Urlich and Hodder-Swain, 2022).

The different objectives and statutory requirements of different government agencies also affect researchers, in the types of knowledge needed and through the level of certainty and type of risk assessments required. For example, additive feed fish farms require permission from regional councils to discharge feed, and for most councils must produce a comprehensive assessment of environment effects, including on the water column, seabed, seabirds, marine mammals, and sharks. An unpermitted activity on land also generally requires a robust risk assessment with strong processes that include assessing risks to the marine environment. In contrast, permitted land-based activities, such as farming, and forestry on low slopes, require no risk assessment, even when the activity affects the marine environment. Similarly, in the marine environment, information requirements are relatively minor for assessment of the environmental effects of bottom-trawling in neighbouring and wider areas, and even habitats of particular significance to fisheries remain generally unidentified after 25 years of the Fisheries Act (Gerrard, 2021).

3 Solutions

3.1 Science-policy liaisons

In Aotearoa-New Zealand, many marine management agencies employ in-house scientists to commission research they feel is needed to meet their objectives, to fill knowledge gaps at appropriate times (section 2.2) and overcome resourcing limitations. Importantly these in-house scientists ensure research findings have accurate and robust lay-summaries (section 2.1), that data limitations are understood (section 2.3.1), and that the findings are moved through the agency once the research has been completed (see Figure 2). This liaison or brokerage role has the potential to increase science use, allowing researchers the freedom to focus on ensuring that the underlying research has been done rigorously and is therefore available to guide actions and support a range of solutions.

However, successful use of research in policy formation and planning relies highly on evidence-based policy and planning development models. To date, in Aotearoa-New Zealand, there are no written requirements for agencies to use scientific data when creating policies, and no process for science-policy liaisons to affect legislation, politics or agency goals (Figure 2). There is an expectation that policy is evidence-based and recently (2021) the Department of the Prime Minister and Cabinet (DPMC) has bought out a policy quality framework - "The Policy Quality Framework - Quality Standards for written policy and other advice (dpmc.govt.nz)" that gives some guidance on the 4 major points that **should** be covered: context, analysis, advice and action.

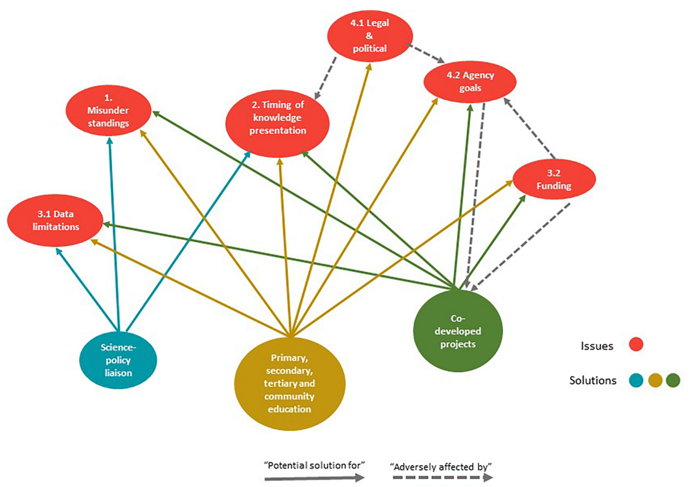

FIGURE 2
Summary of issues and presently used and suggested solutions for Aotearoa-New Zealand. Issues are labelled as they occur in the text. Solutions for the issues are tracked to each issue and colour coded by the solution. Issues are numbered as they are presented in the text, although without the preceding section 2 number. Solutions and issues that are adversely affected by issues are connected by a dashed line.

For analysis it states (final point out of 4) "is well informed (i.e. by up-to-date data, evidence, knowledge, experience, and research from New Zealand and overseas)". However, statements at the start are relatively weak, e.g., "These standards will help you assess and improve the quality of your agency's written policy and other advice, and whether it is fit for purpose——sometimes not all standards will be applicable".

Success also relies highly on effective individuals (Greenhalgh et al., 2022; Karcher et al., 2022). When liaison is working, trust is built, with existing relationships allowing for sharing and transparency, and clarification when concepts or outputs are not understood. Iterative conversations at the science-policy interface can improve understanding of the key opportunities for all parties, with those opportunities more likely to be realized when information flow permeates the agency and other interested parties. While some recommend hierarchical flow of information summaries (Dicks et al., 2014), the information flow process will be unsuccessful if its underlying information is incorrectly interpreted. In addition, uptake into policy can be poor if agency scientists are low in the management hierarchy or are not effective communicators (Greenhalgh et al., 2022). Without a strong science-policy liaison, outdated concepts can be perpetuated, science content reduced, and policy-sized chunks reinterpreted and snipped at each stage up the food chain resulting in incorrect policy advice, plans or decision making.

There are high rates of people turnover in central and regional government agencies in Aotearoa-New Zealand, e.g., 20%-26% for the 2015-2019 period at the Ministry for the Environment. People turnover disrupts relationships with researchers and with others, as time is required to re-establish these relationships and the trust that underpins acceptance of science outputs (Greenhalgh et al., 2022). People turnover and the accompanying loss of agency knowledge can result in problems navigating procurement policies to ensure appropriate research providers are engaged. Agency memory also affects maintenance of datasets and knowledge, wasting scarce resources on reinventing the wheel and not including relevant data in decision making.

High people turnover does offer an opportunity to researchers, as people who have worked in many agencies can build knowledge and connections across them. The National Science Challenge Sustainable Seas has taken advantage of such people, embedding them into projects to guide policy interactions. Conversely, training of new staff by those remaining can re-enforce the status quo, as agencies' cultures can be resistant and slow to reform (institutional inertia).

3.2 Co-development and transdisciplinary projects

Aotearoa-New Zealand is transitioning its government-funded environmental research from inter- and multi-disciplinary (e.g., between biological physical researchers and human geography researchers) to trans-disciplinary research (integrates knowledge across academic disciplines with non-academic stakeholders to address societal challenges). Transdisciplinary research engages stakeholders in significant ways throughout the research process, preferably by co-developing projects. In the view of the government funding agency (Ministry of Business, Innovation and Employment (MBIE)), co-development should deliver a partnership between different knowledge systems, thus helping to achieve management action or policy change. MBIE states a preference for projects that encompass a wide range of stakeholders from central and local government agencies, to businesses and local communities. Projects are expected to also partner with Māori entities, responding to their needs at a variety of levels, utilizing their knowledge and providing any capacity building needed. Thus, co-developed projects should more successfully address the issues solved by successful science-policy liaisons, including gaining funding (2.3.2) and providing a process for affecting agency goals (2.4.2). However, it stops short of providing any process whereby legislation and politics can be influenced (Figure 2).

Co-development offers considerable benefits for researchers, ranging from stakeholders understanding other perspectives, through policy development, to education about science and other knowledge systems. Sometimes the projects can offer a "safe" space for policy makers, planners and environmental decision makers to explore new thoughts. Unfortunately, in our experience working as researchers in such projects, there are several emerging barriers.

- High transaction costs, in terms of researcher time and organizational resources to write grant applications with no guarantee of funding, is inefficient, ineffective and a significant barrier to early career researchers. Currently, much of the funding for this type of research is competitive. Conversely, the Sustainable Seas National Science Challenge has worked with a negotiated process where topics, outcomes and funding are set at a high level and negotiated with a research team (see section 2.3.2). However, high transactions costs still occur, such as those from ongoing co-development processes which can be intensive due to frequent turnover in partners from agencies.
- Environmentally focused proposals not only need to demonstrate that the research is needed, but also need to guarantee delivery of results within 3 to 10-years, with at least some use of the results by businesses, decision-makers, planners or policy within that time period.
- Co-development partners are often time-poor people in operational roles with a limited ability to create change in their own organization. Many central and regional government agencies are not mandated to act on research findings. As we move to partnerships and co-production of solutions involving many different

partners, we need to move into spaces of shared responsibility and actions.

- New initiatives to co-develop projects with Māori partners e.g., MBIE's Vision Mātauranga funding program) are building engagement and involvement in proposal development and in research itself, but these often tax the time capacity of individual Māori as many roles in iwi, hāpu and Māori trusts are often only one person deep.
- Transdisciplinary research is a very human and organic process, yet in Aotearoa-New Zealand funding requires predictions in the proposal as to the timing of steps and milestones (more than one per year are expected) and then reporting on these from 3 monthly to annually. If transdisciplinary research is the way forward, central funding agencies need to create new structures that can accept that progress, like the environment we are trying to manage, is not linear and predictive, but requires flexibility to accommodate engagement and knowledge sharing with stakeholders. Reporting requirements could also be simplified, following the proliferation of reporting, accountability, and technical advisory groups and boards, that while required to some degree, often take up significant portions of research funding, and research time to manage.

3.3 Using education to build understanding across society

We urgently need to develop the ability of legislators, planners and policy- and decision-makers to understand complexity and stop trying to find a "silver bullet" or a "one metric" solution. We also need to shift emphasis from short-term economic imperatives to long-term environmental outcomes that support healthy ecosystems and also increase transparency in decision-making (Tadaki et al., 2021). The chances of good environmental outcomes for the next generation will increase with training to navigate different knowledge systems and undertake the joined-up thinking needed to transform relationships between people and nature. We need to foster development from school children through to universities, and on to whole-of-career learning. Ecology should be a foundation paper for degrees in business, planning etc. In a world of mis-information, alternative facts and complex problems, a critical skill for all is to know when to trust and how to judge the value of knowledge. Part of this may also need a shift in media focus from short catastrophic, or adversarial, stories to deeper narratives. We feel that education to build understanding across society is essential for solving issues related to using knowledge to support our environment (Figure 2).

Enthusiasm for science and an awareness of our interactions with the environment is beginning to be built into the Aotearoa-New Zealand school curriculum, generally on an area-by-area basis. For example, a curriculum around Ecosystem-based management is being developed for secondary school students through interactions between researchers and Marlborough Girls College. A group of schools (from primary to secondary) around the Manukau Harbour are interacting with researchers and scientists from Auckland Council, to understand the health of the harbour, what could be done to manage it better and how science can help. Ecologists working in marine science in many of Aotearoa-New Zealand's universities are co-supervising students working across the biophysical science, social science and economic disciplines in an effort to embed complexity and transdisciplinary understandings into students. Some of these students are already graduating and moving into various roles in government agencies.

An obvious next step is to add law into the educational mix to ensure that non-lawyers, lawyers and courts understand the implications of the language and concepts that science and policy use and vice versa. In the Sustainable Seas National Science Challenge, marine ecologists, policy makers and decision makers from multiple organizations are also working with environmental lawyers.

Many Māori concepts are beginning to resonate in the general public, with the increasing teaching of Te Reo (Māori language) in schools and institutions and the embedding of Mātauranga in all government agencies and partnership with science. These concepts serve Aotearoa-New Zealand well in articulating the importance of our connections with nature and bringing long-term benefits to the forefront of decision making.

Whether or not such educational initiatives will be a successful solution is yet unknown. We feel optimistic that this new knowledge and perspective is slowly diffusing out from successful science-policy liaisons and co-developed projects (particularly those including Māori partners). Directly targeting education initiatives is, however, required to speed up this process. Certainly, without the ability to create better cross-discipline, science-informed and nature-focused people across all of Aotearoa-New Zealand, science researchers will be continually doomed to having only marginal impact with our research, often after avoidable degradation, or restoration failures, have occurred.

4 Conclusions

Finding the place and role for environmental science in Aotearoa-New Zealand and the world is a non-trivial task. We must prize rigorous and relevant science and scholarship but embed this in society to effect the necessary fundamental change. Science's declaration of the existence and nature of the

Anthropocene demonstrated that everything is connected, and that critical problems are multi-dimensional and multi-scalar.

Our experience shows many barriers on the path to increasing the use of science in policy, plans and decision-making. Science researchers can help with barriers relate to different language and concepts, and how to deal with complexity, uncertainty and lack of knowledge, assuming that planners, policy-makers and decision-makers are allowed to listen and foster innovative solutions. However, other barriers, for example, legal and political constraints and conflicting agency objectives are not within the ability of science researchers to directly overcome. We agree that researchers certainly need to do what they can in "taking the horse to water". However, we believe successful use of science to achieve good environmental outcomes requires commitment across society. This requires new approaches and capacity building that can transform the ineffective or incremental ways of approaching crises in biodiversity, sustainability and climate. It also requires a conversation about values and norms towards nature and transparency about how decisions are reached and for what benefit(s). We, therefore, suggest that education from junior levels through to universities has a crucial role to play.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author/s.

Author contributions

All authors contributed to the discussion around the barriers and solutions and each wrote at least one section. JH assembled the individual sections. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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'Out of sight, out of mind' - towards a greater acknowledgment of submerged prehistoric resources in Australian science-policy as part of a common heritage

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There is growing awareness of the need for greater acknowledgement of underwater prehistoric cultural resources as part of management and regulation of the seabed around many maritime countries, especially those with large indigenous populations and history such as Australia. Prehistoric cultural places and landscapes inundated by Post-glacial sea-level rise on Australia's continental shelf remain largely out-of-sight and out-of-mind, hence awareness and hence legal protection of this resource is lacking. There is a clear need for greater integration of archaeology and cultural heritage management within the marine sciences as well as a greater awareness of this resource as part of a common heritage more generally. This paper explores some of the dichotomies between Western and Indigenous cultures in valuing and managing the seabed. We argue that in developing science-policy, an attempt at least needs to be made to bridge both the gap between the nature and culture perspectives, and the jurisdictional divide between land and sea. Part of the answer lies in a convergence of Indigenous knowledge with Western science approaches, focused around our understanding of physical processes impacting past and present coastal landscapes and on the seabed itself. We explore several case studies from northern and Western Australia that are trying to do this, and which are helping to provide a greater appreciation of the inundated landscapes of the inner shelf as part of a common heritage.

KEYWORDS
cultural heritage management, submerged cultural resources, marine science, geoarchaeology, Indigenous, Western Australia

Introduction

Indigenous occupation in Australia dates from 65,000 years b.p. (Clarkson et al., 2017) but the most significant part of this – over 55,000 years and more than 2 million km^2 of the continental landmass – is now underwater, drowned by sea-level rise over the last 20,000 years. Prehistoric cultural places and landscapes inundated by Post-glacial sea-level rise on Australia's continental shelf have to date been largely out-of-sight and out-of-mind. This article reflects on submerged prehistoric cultural heritage resources as part of a 'common heritage', and as part of sustainable marine management. Distinction is made here between submerged landscapes as part of the global commons[1] from those that are part of the cultural heritage of Traditional Owners. The former holds that the sea and seabed within the Exclusive Economic Zone (EEZ) are "common to all men", with individual nation states sharing in its management and the benefits of its exploitation (Guntrip, 2003; see also Smyth and Isherwood, 2016). The latter relates to the Indigenous understanding of the sea as an inseparable extension of the land (e.g., Yunupingu and Muller, 2009; James, 2019) and hence subject to the same aspects of custodianship, exclusive resources and customary law. Hence "Sea Country" and "Saltwater Country" refers to any environment within broader traditional estates that are associated with the sea or saltwater—including coastal areas, estuaries, beaches, marine areas and islands and their living and non-living natural resources (Rist et al., 2019).

We argue that in developing ocean science and ocean science-policy for Australia, greater attempt needs to be made to bridge the gap between Western science and Indigenous knowledge and also the jurisdictional divide between land and sea (see also Yunupingu and Muller, 2009), not least because sea level has changed over the 65,000-year period of human occupation. This necessarily includes an emphasis on the past and present physical (seabed) landscape but also the more challenging realm of perception of seascapes in cultural heritage management (Kikiloi et al.,2017; Wickham-Jones, 2019). Currently, an upfront integration between cultural heritage and marine sciences is lacking (Trakadas et al., 2019). We explore a number of case studies from northern and Western Australia that attempt to combine these ideas, and which are helping to provide a greater appreciation of the submerged landscapes of the inner shelf – and natural and cultural elements of these, as something of 'common concern of humankind' (aka Forrest, 2007).

Our approach is largely an interrogation of the literature around a broader topic of marine science, cultural heritage and the seabed, as there are very few studies that deal directly with the question of science policy on submerged prehistoric landscapes. As emphasis of this, a Scopus search using the keywords of marine, prehistoric, cultural, science, policy yielded zero results. A Scopus search using the keywords indigenous, submerged, landscape, policy yielded two results (Ward et al., 2018; McCarthy et al., 2022), whilst indigenous, submerged, landscape, science produced only one result (Flatman and Evans, 2014). Marine, indigenous, cultural, science, policy yielded 16 results, most of which were related to inclusion (e.g., Kikiloi et al., 2017; Johri et al., 2021; Worm et al., 2021) and none of which directly referred to the seabed or submerged landscapes. Whilst ocean-science policy is arguably directed towards regulators and developers, it is driven by values and interests as much as by evidence and research. Raising awareness of novel topics, such as submerged landscapes, is key (Zuercher et al., 2022) and the general lack of awareness of submerged prehistoric cultural resources means that this discussion is relevant to all who have a value and interest in the marine environment.

Past and present sea country

Over the 65,000 years of Aboriginal occupation of Australia, sea levels have fluctuated, rising from a peak low of -120 m at around 21,000 years ago relative to present levels and resulting in inundation of vast areas the continental shelf. Indigenous people witnessed, adapted to and "remember" many phases of falling and rising sea-level and associated geomorphological change along the coastline, particularly across northern Australia's low gradient continental shelf. Change and adaptation – and not just to climate or sea level, is a constant feature, rooted in history and time and connected to country and everything relating to it (Nursey-Bray et al., 2019). The Gunggandji people of North Queensland, for example, "*have lived through a 10-metre rise in sea level, great changes in rainfall, the arrival of new plant and animal species and the great upheavals caused by volcanic activity as river courses changed and new land forms emerged*" (Gunggandji Land and Sea Country PBC Aboriginal Corporation, 2013). Many Indigenous people still relate to land that was inundated by sea-level rise and before current coastal ecosystems began to establish when sea level stabilized about 5000 years ago (Smyth, 2002), with marine sacred sites recorded up to 80 km off the Northern Territory coast (Peterson and Rigsby, 1998; see also Kearney & Bradley, 2009). Visual narratives and oral histories involving mythological creatures that affect coastal and landscape change provide another form of agency to relate to and make sense of the evolving landscape, with oral histories dated on the basis of correlation with sea-level curves to at least 12,600 years ago (Nunn and Reid, 2016; Nunn, 2018; see also Wickham-Jones, 2019).

Compared to Western understandings of the coastal and offshore zone, Indigenous ways of knowing and managing Sea

[1] Under UNCLOS the seabed and ocean floor within each nations' EEZ is viewed as the 'common heritage' of mankind (United Nations 1982).

Country are more geosophical (earth-centered) and emphasize the interconnectedness of people and nature, land and sea, and of physical (tangible) and metaphysical (intangible) elements within these (Kwaymullina and Kwaymullina, 2010; Korf, 2019; Tilot et al., 2021).[2] These physical elements extend beyond specific economic resources (flora, fauna, geology) to detailed knowledge of oceanography (e.g., tides and currents) with the implicit emphasis on understanding of process and change (see also Lee, 2016; Stevens and Paul Brake, 2021). The latter pairs place and memory, including through songlines or 'Dreamings'[3] and language, so that knowledge is grounded in landscape and landscape evolution. Ancestral journeys often commence out at sea then move closer to land, creating seascapes - islands, reefs, rocks, sand banks, cays, patches of seagrass - and travel on to create emergent landscapes. Extant connections exist from named places in the sea (reefs, rocks, etc.), including named zones of the sea defined by water depth (Chase and Sutton, 1981) and named bodies of water associated with ancestral dreaming tracks (Myers et al., 1996; Peterson and Rigsby, 1998). The Mayala people of the West Kimberley, for example, know the complex tides and tidal currents (*loo*) and travel on the *noomoorr*, which resembles a saltwater highway (Mayala Inninalang Aboriginal Corporation, 2019). Similarly, the Yanyuwa language or 'Tiger shark language' originates from a 40,000-year-old relationship with the tiger shark and the ocean (Kearney and Bradley, 2009)[4].

Indigenous understandings of Sea Country also counter conventional Western notions of the shoreline as a boundary marking the divide between land and sea, often with separate jurisdictional arrangements. Western convention, including the UN *Convention on the Law of the Sea* (UNCLOS), uses defined baselines, such as the high-water mark as the upper boundary of territorial waters and the EEZ. As such, this has to be recalibrated at regular intervals to allow for sea-level change and anthropogenic structures that may extend the agreed land area of a state (Zacharias and Ardron, 2020)[5]. Notably, some of those jurisdictions were originally related to the defence of a state, for example, the area controlled by cannon-fire from the land. This is in large contrast to the way Indigenous peoples define their lands. For many such peoples across Australia (e.g., Kearney and Bradley, 2009; James, 2019; Mayala Inninalang Aboriginal Corporation, 2019) and also the Pacific Islands (Tilot et al., 2021), the sea is not only a physical and temporal space, but also a mental map of ancestral journeys and ritual renewals with a view to nurturing and passing on place-based knowledge and its biological, cultural, and linguistic endowment to future generations (see also Vierros et al., 2020). This is truly a sustainable view of ocean use by society and perhaps broader than that envisaged by the UN Sustainable Development Goals. Although arguably a recent distinction (Wickham-Jones, 2010), the Western separation of land and sea as conceptual and physical entities is, Henderson (2019) argues, ultimately responsible for the underappreciated role of the importance of the sea in human history. This is particularly at odds for an island nation such as Australia whose history and ecology were shaped by the sea, and whose 200-nautical-mile exclusive economic zone (EEZ) is greater than the land mass of the nation itself (Figure 1; Symonds et al., 2009).

As studies across various maritime nations with large Indigenous histories, and especially those in the Southern Hemisphere, are revealing, these shelf areas offer new insights into past coastal and ecological dynamics and, by inference, new understandings of past human occupation and dispersal, as well as potentially of seafaring and maritime trade (Henderson, 2019; Ward et al., 2022a and references therein). However, as in other parts of the world (e.g., Quig, 2004; Wickham-Jones, 2010), the lack of research and sometimes even the lack of awareness of the cultural and ecological value of submerged landscapes is a serious hindrance to good management. In the Kimberley, for example, the coastal area between the shoreline (defined as Mean High Water Mark) and 2 km inland was found to be disproportionally valued over areas 20 km and even 200 km landward or seaward (Kobryn et al., 2018). In addition, of the thirty critical research needs identified for the Kimberley marine environment in Western Australia, submerged cultural heritage was not identified by any of the Healthy Country[6] managers, natural resource managers or scientists (Cviyamovoc et al., 2021). However, as Kobryn et al. (2018) identify, places that are not mapped should not interpreted as the absence of values, but simply places that require greater research effort, which we argue includes submerged cultural landscapes. All marine protected areas, including Sea Country IPAs (Figure 1), recognized by the International Union for the Conservation of Nature (IUCN) are obliged to protect the associated cultural values of those areas, which includes the seabed. Hence to achieve a better understanding of the ocean and its common heritage, we need to merge various types of evidence and give

[2] See also https://nntc.com.au/news_latest/the-state-of-intangible-cultural-heritage-in-australia/

[3] Songlines or dreaming tracks are maps of the land that show the connectedness between places and Creation events, and a central part of Australian Indigenous culture (see also Malcolm and Willis 2016).

[4] See also https://www.bbc.com/travel/article/20180429-australias-ancient-language-shaped-by-sharks

[5] For the current jurisdictional zones of Australia's marine environment, see https://soe.environment.gov.au/science/soe/2011-report/6-marine/1-introduction/1-1-the-jurisdictions

[6] Healthy Country Planning (HCP) is an adaptation of the Conservation Standards used and adopted by Aboriginal land management teams across Australia (see Carr et al. 2017).

FIGURE 1
Overview of Australian protected areas and Indigenous Protected Areas (IPA) (modified from Collaborative Australian Protected Area Database, Australian Government Department of the Environment and Energy, 2014). Australian state and territories (capitalized), regions (large text), names of Indigenous groups (italics), locations (bold) mentioned in text are also included. For a more comprehensive map and list of current and proposed IPAs, see Gould et al. (2021), their Figure 2.

greater credibility to cultural knowledge systems such as that passed down in oral-histories.

The (un)known cultural heritage resource

Establishing a baseline

The National Marine Science Plan 2015-2025[7] states that to improve the management of Australia's marine estate, marine science needs to improve the collection of data relevant to resource allocation, particularly for Indigenous use and rights and other social and economic attributes. A resource or system cannot be managed unless it is measured or mapped (Borja and Elliott, 2021), or as Indigenous elder Edvard Hviding (2005) explains, "*those who cannot name the good things of sea and land, cannot find them, and therefore cannot eat or otherwise benefit from them, nor will they know how to look after them well*". This ultimately leads to a need for systematic assessment of ecological and cultural heritage resources – both known and unknown in coastal and marine settings (e.g., Gee et al. 2017), such as was done for Groote Eylandt (Davies et al., 2020) and is being done for the Recherche Archipelago (Guilfoyle et al., 2019) (see locations on Box 1). The community-led study in the Recherche Archipelago is exploring the transformation of the coastal plain from the late Pleistocene, including traditional creation stories of the islands, to the more recent historical use of the archipelago (Box 1: Recherche Archipelago). Indigenous perspectives and traditional knowledge can be integrated with western approaches to document this drowned landscape as a new form of ecosystem-based science and shared solutions for its future management.

For Western Australia, this baseline understanding is very uneven, as identified in a statewide review of coastal waters for potential marine conservation (CALM, 1994). The latter report recommended areas of protection but also highlighted the lack of scientific research available to justify that decision. A more recent report undertaken for the southern coast similarly found it difficult to identify areas of higher conservation value – whether ecological, geological or cultural, due to the lack of information (Sutton and Day, 2021; see also Smith, 2021), and made no mention of coastal or submerged prehistoric cultural heritage. A robust analysis of the cultural goods and benefits, both current and past, for the area would also help create an inventory of its value. There are now many indicators of such cultural goods and benefits (Atkins et al., 2015), although consideration also needs to be given as to whether identifying the

[7] National Marine Science Committee (2015) https://www.marinescience.net.au/nationalmarinescienceplan/

Box 1 | Ancient corridors, continuous connections, Recherche archipelago.

Coastal and offshore landscapes are cultural places that are protected by cultural customs as well as heritage legislation. The Recherche Archipelago is situated along the southern coast of Western Australia, and is bordered on either side by Commonwealth Marine Parks (Figure 1). A proposed south coast marine park, incorporating the Recherche Archipelago, is being proposed that will be jointly managed between Department of Biodiversity Conservations and Attractions (DBCA) and the area's Traditional Owners - Wudjari. Accordingly, Esperance Tjaltjraak Native Title Aboriginal Corporation (ETNTAC), on behalf of the Wudjari Traditional Owners, have embarked on a community-led, multi-disciplinary programme to study, monitor and protect Sea Country across the entire Recherche Archipelago. The programme involves collaboration with the Federal statutory body Parks Australia to implement shared Healthy Country Plans and Australian Marine Park Management Plan priorities. The health of the marine life, the island habitats, and the cultural places of this seascape is of paramount importance to the Elders and the wider community. A member of the Circle of Elders was paraphrased as saying, "We know that the only way to live well and flourish on Boodja (land and Sea Country) is to know it well." So, making Sea Country (Boodja) healthy is also making it well understood. The remoteness of the region has up till now resulted in limited coordinated investment in research in this area, hence the renewed focus on addressing baseline data gaps on cultural and natural values that will support effective management of the ancient coastlines.

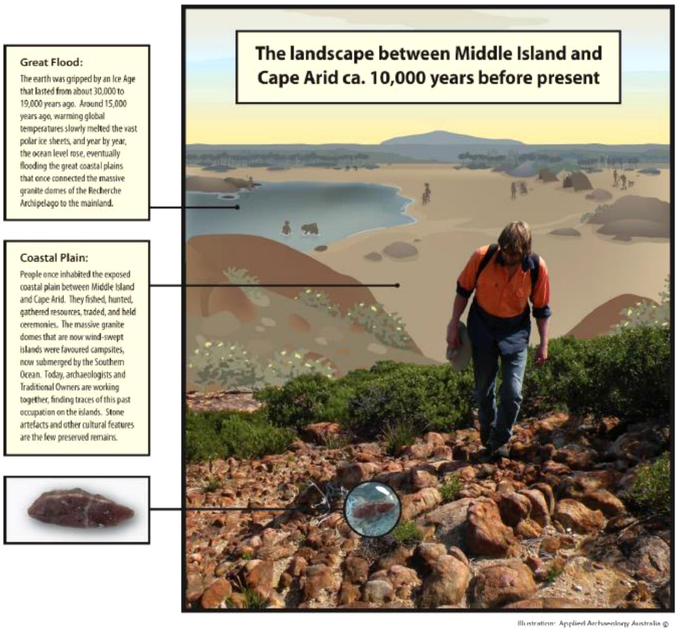

Figure 2

(Left) Proposed model of landscape around Middle Island, one of the islands of the Rercherche Archipelago (sourced from Guilfoyle et al. 2020). (below) Healthy Sea Country ranger projects are running in tandem with cultural values mapping and models of the ancient coastal plain.

The Ancient Corridors project (Guilfoyle 2019) will integrate palaeoenvironmental, archaeological and ecological data with cultural knowledge to explore human-environment dynamics over the last 15,000 years. This cultural knowledge includes cultural stories and Songlines that extend from the mainland and across the Archipelago, and demonstrate ongoing connections to the sea and coast. At the peak of the last Ice Age, approximately 21,000 years ago, the coastline would have extended 80 – 100 km further offshore. Archaeological evidence for the use of the now submerged plain is in the form of stone artefacts, middens, man-made structures and other cultural features located on the islands of the Archipelago and also on the mainland from at least 13,000 years ago (Guilfoyle 2019).

With post-glacial sea-level rise, the vast coastal plain flooded to create the 105 islands of Recherche Archipelago that stretch 230 km from east to west and to 50 km offshore (Jackson 2008). The Tjaltjraak Rangers are working with specialists to explore the potential for sites of cultural significance and natural biodiversity through high-resolution coastal and seabed mapping, including of drowned reefs, palaeo-channels and submerged shorelines. The Ancient Corridors programme recognises that cultural systems in the past were interwoven with the landscape and its ecosystems, just as today the cultural landscape is an inherent part of the natural landscape. Hence effective management of the modern coast necessarily involves an understanding of how the processes of climate and sea-level change, and human occupation have affected and influenced the biodiversity and ecology over time. The Ancient Corridors project is just one of a range of collaborative research projects are underway in tandem with an adaptive management worksprogram led by the Tjaltjraak Rangers.

location of a cultural places(s) and quantifying those goods and benefits would increase or decrease the likelihood of desecration.

The terrestrial archaeological record holds many examples of the past use of marine resources in the form of midden sites, coastal fish traps, shell artefacts, rock art depicting marine motifs, and other parts of the material record where an association with the marine environment can be made (e.g., McNiven, 2003; Ward et al., 2018; see also Feary, 2015). Similar site types are likely preserved on the shelf, even though the past landscape context may differ (Ward et al., 2022a and references therein). Part of the scientific or global commons perspective for investigating similar sites on the continental shelf is how they might reflect change in marine, coastal and terrestrial ecosystems and landscapes and traditional resource exploitation and management of these. For many traditional owners, the existence and acceptance of such sites is not a discovery but rather validation of the continuing existence of ancestral sprits in the present and ongoing custodial responsibilities to Sea Country (McNiven, 2016). Hence what Western science offers the Indigenous community is in the opportunity to add to an existing body of traditional knowledge to better understand and manage Sea Country (see also Box 3). It also arguably provides relevance, credibility and legitimacy for Western purposes of a cultural landscape that warrants management, protection and potentially even implied ownership. At present the invisibility of submerged prehistoric cultural heritage means that it is what Larcombe and Morrison-Saunders (2017) might describe as 'out of sight – out of mind'.

Mapping submerged cultural landscapes

Increasing resolution in seabed mapping data and their manipulation (O'Leary et al., 2020; Lebrec et al., 2022) shows that the shelf is not featureless and, in some parts, has well-preserved remnants of former coastal landscapes and hence potential prehistoric cultural places. The North West Shelf (NWS) of Australia (Figure 1) is an extensive shallow marine region up to 220 km wide with extensive oil and gas reserves (Longley et al., 2002) and a range of unique coastal, reef and offshore environmental features from periods of lower sea level that have significant economic, ecological, cultural, social and geoheritage values (Wilson, 2013; Brooke et al., 2017; Lebrec et al., 2022). These remnant geomorphic features have had a significant influence on the pattern of biodiversity and species endemism over extensive areas of shelf (Nichol and Brooke, 2011; Wilson, 2013), as well as shaping the landscape and coastal resources that humans formerly accessed, occupied and utilized, as early as 50,000 years ago (Veth et al., 2017). Further south in the Esperance region, remnant low relief (< 2 m) linear calcarenite deposits representing drowned shorelines form important habitats for sessile organisms (Ryan et al., 2014). Similar to the cemented shoreline deposits of James Price Point in the Kimberley (see Box 2 - Case Study 2: James Price Point), these have high cultural potential.

Pleistocene sea-level fluctuations have also left a clear genetic signature in phylogeographic patterns of iconic species such as the dugong, *Dugong dugon* (Blair et al., 2014), common pig-eye shark, *Carcharhinus amboinensis* (Tillett et al., 2012) and some freshwater fishes (Shelley et al., 2020) across northern Australian waters (see also Ludt and Rocha, 2015). These distributions in turn relate to former seagrass meadows, turbid coastal waters and freshwater streams respectively, and by inference the cultural environments that people once occupied. Hence identifying these sedimentary and geomorphic contexts is important towards identifying and resolving past natural and cultural landscapes. However, the marine sedimentary record is discontinuous and there are large knowledge gaps. Amongst the palaeoecological unknowns for the Barrow Island region, for example, is the shelf location of the early sedimentary record of mangroves, even though the zooarchaeological records indicate foraging of fauna from these environments from as early as 15,000 years ago (Ditchfield et al., 2018) and a near absence of them today.

Whilst high-resolution mapping is useful for deeper settings, in shallower waters local knowledge can be as important to revealing (or hiding and protecting) cultural heritage. For example, a blog post from the Deep History of Sea Country (DHSC) project team members indicates it was local knowledge rather than systematic survey (c.f. Benjamin et al., 2020; Wiseman et al., 2021) that directed scientists to the submerged stone features in the Cape Bruguieres channel in the Dampier Archipelago (CRARM, 2020) (Figure 1). Claims that the Cape Bruguieres site represents the first *in situ* submerged archaeological site in Australia (Benjamin et al., 2020) have unfortunately not stood up to scientific scrutiny, with the site almost certainly representing a secondary (i.e., reworked) and ponded artefact scatter, i.e., artefacts accumulated in ponded water above lowest tide level (Ward et al., 2022b). This re-analysis emphasizes the importance of understanding the evolution of the physical seascape and of past and present physical processes to interpreting site formation (Ward et al., 2014; Ward et al., 2015; Larcombe et al. 2018) and not emphasizing the significance of a site for merely being under water (Lemke 2020). Arguments that this discovery has helped highlight the lack of awareness of submerged cultural heritage in Australia are less valid when the credibility of the science and the understandings are questioned, and further erodes science as an arbiter of good policy in cultural resource management. At worst, such poorly justified interpretations are in danger of changing the traditional narrative around such sites.

There are good arguments for greater integration with, and even prioritization of, Indigenous cultural values over Western scientific approaches as part of cultural heritage assessment and sustainable management (Tutchener et al., 2020; Tutcherner et al., 2021), a key aspect of which is the emphasis given to landscapes rather than to the artefact or site. Landscapes that are 'rare' and therefore significant, contain remnant (i.e., pre-colonial) or unusual landforms or other geographic or environmental characteristics. All archaeological material in such landscapes is considered rare and to have a high level of significance (Tutchener et al., 2021) but the presence of tangible cultural material is not necessarily a criterion for significance and the presence of oral histories needs to be regarded as adequate evidence of that significance (see also McNiven, 2003). These criteria overlap with those used for geoheritage significance, with archaeology and cultural heritage linked by sedimentary units that comprise these landforms (Brocx and Semeniuk, 2007; Brocx, 2008; Ward et al., 2014) both in terrestrial and marine contexts.

Looking below the seabed

Due to climate, sea level and environmental changes, former natural and cultural landscapes of the shelf are not always preserved at the seabed surface but are often buried beneath it

Box 2 | Case study 2: James price point, Western kimberley coast.

Around James Price Point (Figure 3), northern WA, high-resolution mapping undertaken as part of the pre-development survey for a Liquid Natural Gas (LNG) facility revealed well-preserved drowned shoreline features, likely formed in the early-mid Holocene. At least two series of north - south trending palaeoshoreline features exist with relief of up to 5 m of more above the surrounding seabed, and are associated with a former lagoon and fossil intertidal flats. These palaeogeographic features have significant geoheritage value and systematic investigation is likely to contribute to our understanding of early maritime adaptation and resource use in this region. Important here is the recognition that landforms and stratigraphic features can represent sites of cultural significance, or natural sites of significance independently of the presence or absence of cultural material.

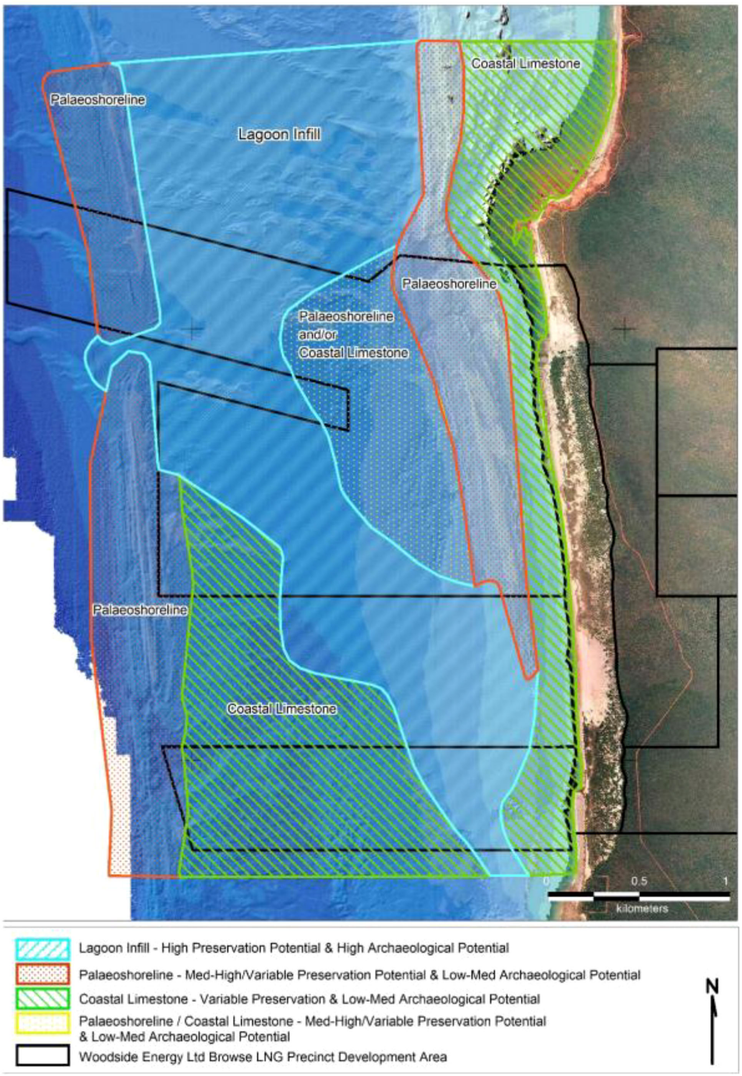

Figure 3
The IMAP developed for the inner shelf off James Price Point, north of Broome (Ward et al. 2016).

The marine component of the Archaeological Impact Study (AIS) was not initially part of any brief but was subsequently requested by the Goolarabooloo Jabirr Jabirr Native Title Claimant Group to be included as an extension of the onshore desk-based assessment. Raw survey data and sediment core samples were not made available, hence cultural heritage was assessed from the nature of the depositional environments as likely sites of occupation and/or concentrations of archaeological artefacts. This was based on documented geological, bathymetric and sedimentary data, the past and present sedimentary processes, as well as existing archaeological information (including fish-traps, midden sites, and stone artefact scatters) on the current coast and adjacent hinterland. As subsequent studies in the James Price Point area identify (Clifford and Semeniuk 2019), the sedimentary bodies and stratigraphic units form a template with which to locate and interpret archaeological sites in the context of coastal occupation, coastal stability and sea-level change.

The result of this was an Indicative Map of Archaeological Potential (IMAP, Figure 3) that identifies specific areas of the coastal and marine zone interpreted as having relatively low, medium or high potential for the presence of archaeological remains in primary and secondary (reworked) depositional contexts (Figure 3; see also Ward and Larcombe, 2008; Cohen et al. 2014). Those areas designated as low potential, and with no visible or known archaeology may still yield archaeological remains. Similarly, areas marked as having the potential for containing artefacts in primary context may also contain artefacts in secondary context, including those eroded from the modern cliff-face (Ward et al. 2016). These then become part of the complex coastal history, linking onshore and offshore, and part of the geoheritage story (Clifford and Semeniuk 2019). The IMAP can then be further refined as archaeological, sedimentological and geomorphological information becomes available and as Indigenous perspectives are incorporated into the assessments.

(e.g., Ward et al., 2015; see also Box 3). Hence, in order to understand these landscapes, we have to look beyond the seabed surface to the underlying stratigraphy. Despite six decades of fieldwork on the NWS (Kirkendale and Richards, 2019), this buried landscape is mostly unknown. Biodiversity and habitat surveys by government, industry and academic groups have to date focused almost entirely on the shallow seabed (Lyne et al., 2006; Kirkendale and Richards, 2019), and often overlook physical sedimentary controls on these (Larcombe and Morrison-Saunders, 2017). Whilst scientific knowledge on marine physical processes does exist, it needs to be understood as a critical element in resolving past and present ecological dynamics and is also pivotal to many studies exploring human-environmental dynamics and sea-level change (see also Cawthra et al., 2020). Even today, there are questions around future sea level rise and how it may impact Traditional customs and use of coastal ecosystems (Zander et al., 2013; Sloane et al., 2019)[8] and also cultural heritage (Carmichael et al., 2018). Both involve identifying and understanding both the physical processes impacting modern coasts and also Indigenous cultural heritage and values.

The sedimentary archive is key to increasing our understanding. Unfortunately, national archives of marine sediment cores out to the 120 m bathymetric contour, which broadly represents the last glacial lowstand (exposed seabed) are sparse (Figure 4), were usually acquired for purposes other than submerged palaeolandscape or cultural heritage research and hence are often of limited use. The value of targeted marine surveys, including high resolution seabed mapping and sub-bottom seismic profiling ground-truthed by core sampling, has been demonstrated worldwide (e.g., Vos et al., 2015; Brown et al., 2018; Marean et al., 2020; Bailey and Cawthra, 2021; Dupont et al., 2022). Hence, there is clear scope to progress similar targeted work, including through Indigenous collaboration and industry partnerships, on the NWS and other parts of the Australian continental shelf.

A good example of this is the Westport development in Cockburn Sound (see Box 3: Westport development), which benefits from previous marine survey and vibrocoring by Geoscience Australia that identified and dated the identified landsurface 3 m below the seabed surface and which was flooded by sea level rise around 9300 years ago. This flooding event is recorded in Nyungar oral history, providing a connection to a lived landscape but also to the present islands and associated geomorphology. Thus what lies below the water and below the seabed is as much a component of the cultural present as it is the cultural past and, as Diver (2017) notes, contributes generations of knowledge of the land and tradition in this country. Marine geophysical survey data can be combined with analysis of marine core material to re-envisage this lived landscape and its ecology, and to provide a physiographic context for the geoprospection of possible archeological sites that might otherwise remain unknown and unprotected. Research evidence that combines Western science and Indigenous knowledge can thus be used to define criteria for assessment and as the rationale for policy intervention.

Australian legislation

Marine management has to encompass many spatiotemporal realms and regimes, with each maritime state being responsible for the coastal baseline (often the high-water mark) out to the extent of the territorial waters (often 12 nautical miles), then the seabed out to the extent of the 200 nm EEZ and, in some cases to the further limit of the continental shelf (UNCLOS 1982). Hence that management has to encompass local/state, regional, national and international legislation (e.g.,Delgado et al., 2022; see also Boyes and Elliott, 2014), with democratic marine governance also factoring in community-based management and international environmental agreements (Techera, 2012). The attribution under UNCLOS of the EEZ seabed as 'common heritage' (UNCLOS 1982) was the first to incorporate economy and societal needs, and concepts of conservation (UNCLOS Articles 116-120). UNCLOS also provides that modern states also have a duty to protect 'objects of an archaeological or historic nature' out to 200 nm (UNCLOS Article 303). Arguably this does not equate to submerged cultural *landscapes* which, in acknowledging the mobility of hunter-gatherer societies, can be argued to be as much a part of common heritage as any object, site or structure (Ward et al., 2018; see also Bird et al., 2019). As Quig (2004) outlines, for any native title claim it is uncertain whether Indigenous people would have to demonstrate that they physically used and occupied the submerged lands in question through the erection of permanent structures (e.g., fish traps), for activities such as the gathering of marine economic resources (e.g., fish, shellfish), or by simply by engaging in fishing, navigation and spiritual activities. Irrespective of this, UNCLOS does not recognize Indigenous rights and, to be compliant with international law, a state may have to dilute or even negate Indigenous rights over offshore areas for economic interests (Kaye, 2001; see also Quig, 2004; Zacharias and Ardron, 2020).

Marine (and estuarine) ecosystems are the sites of many human influences such as tourism, commercial shipping, fisheries, oil and gas exploration and production, offshore wind farms as well as many traditional activities both contemporary as well as in the past (Borja and Elliott, 2021; Figure 7). Marine ecosystems may be considered from their extrinsic (e.g., economic) or intrinsic (e.g., scientific, historical, spiritual, cultural) value with protection provided through the implementation and maintenance of laws and legislation (Boyes and Elliott, 2014; Cormier et al., 2022). For example, activities may be permitted in areas after being legally sanctioned following a planning application and an

[8] See also https://ourworld.unu.edu/en/sea-level-rise-in-kowanyama

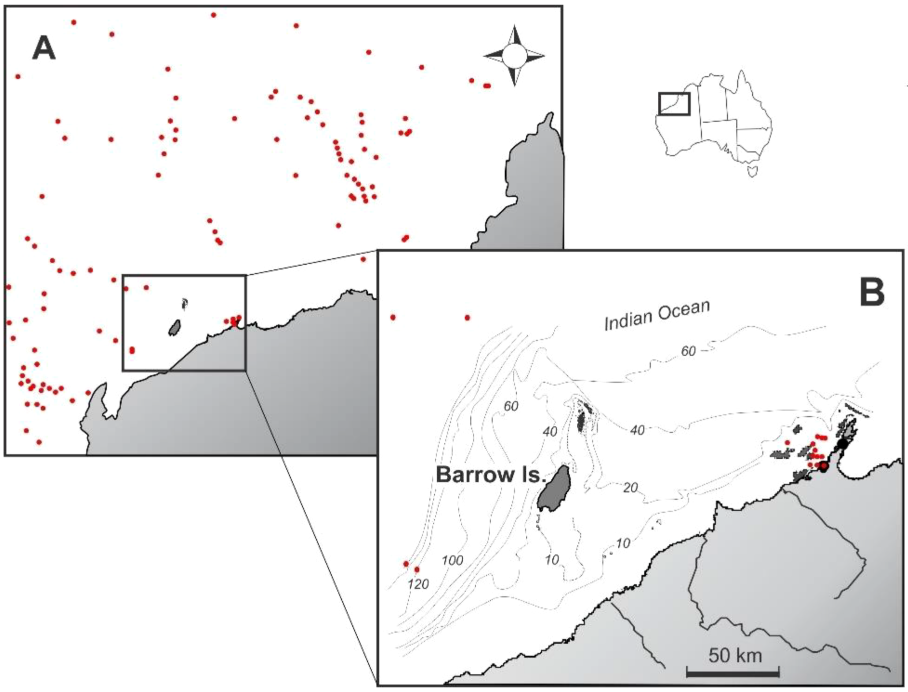

FIGURE 4
Available marine core data from Geoscience Australia for **(A)** the North West Shelf and **(B)** around Barrow Island, where archaeological records go back 50,000 years (sourced from http://dbforms.ga.gov.au/pls/www/npm.mars.search).

Environmental Impact Assessment (EIA), or they may be allowed and legally defended in areas where they have been 'traditionally practiced'. As an example of this, under European legislation, bathing waters may be protected given a common history of practice rather than a legally enforced boundary dictated according to a set of criteria such as the number of people bathing at any one time. Similarly, some Indigenous or 'customary' practices, particularly fishing (Evans, 2004; Hiriart-Bertrand et al., 2020), although not legally sanctioned, may also be recognized within national and interstate frameworks and hence have similar protection. However, the characterization of Indigenous marine interests in Australia's Marine Science Plan 2015-2025 as solely "Indigenous fishermen" (Figure 7) relegates Traditional Owners of Sea Country to a user group rather than a people with a comprehensive cultural, social, spiritual and knowledge-based relationship with Sea Country. Indigenous marine interests also include scientific, conservation and sovereign matters and failure to acknowledge all these values within marine policy development can lead to tensions (e.g. Hiriart-Bertrand et al., 2020). Smyth and Isherwood (2016), Rist et al., (2019) and Leary et al. (2021) provide comprehensive explanations on Indigenous Australian's legal rights in marine areas.

Furthermore, there has always been the debate regarding the provenance of records relating to the use of an area by any group, including Indigenous people. In the Western legal system, documented sources of evidence in languages using written (including observational records by non-indigenous people of Indigenous traditions and customs) or pictogram communication may be regarded more highly and less open to challenge than spoken/oral, story-based information. In contrast, Indigenous legal systems value the spoken word (Gray, 1998), and any formal acknowledgement of a traditional law or custom is ultimately an objective one (Smyth, 2002). Recently, however, there was a landmark agreement in the Land Court of Queensland for First Nations people to be allowed to present their evidence again a mining application on their island in the Torres Strait, with the presiding judge stating, "*written evidence from a First Nations witness is a poor substitute for oral evidence given on country and in the company of those with cultural authority*." The First Nations groups argued the mining project would contribute to climate change and sea level rise, which will have a negative impact on their human rights to practice cultural activities (Maddison, 2022).

The Western Australian *Aboriginal Cultural Heritage Act* 2021 (WA Act 2021) also recognizes Aboriginal cultural heritage as 'the tangible and intangible elements that are important to the Aboriginal people of the State, [and are recognized] through social, spiritual, historical, scientific or aesthetic perspectives'

Box 3 | Case study 3: Westport development, Underwater cultural heritage

In 2020, the Western Australian Government announced a new development project for a new port in Cockburn Sound (Figure 5), to be called Westport. Derbal Nara means Estuary of the Salmon and is the Nyungar name for Cockburn Sound (see also https://derbalnara.org.au/). Gabee-wodin or wardan (sea) is of great spiritual significance to the coastal Nyungar, who have used the resources of the coastal plain for food, shelter, ceremonies and trade for tens of thousands of years as recognised under Native Title. As part of this, a Westport Noongar Advisory Group has been established to provide ongoing specific input, knowledge and the endorsement of the Aboriginal cultural content incorporated into the Westport development.The Nyungar people explain how Derbal Nara formed through a fight between the Waugal (rainbow serpent) and the Spirit Crocodile, with the sea waters rushing in as they rolled and tumbled. The Waugal won the fight, biting the tail of the Spirit Crocodile and placing it at the mouth of the Swan River to prevent salt water coming up the river. The tail became a limestone sand bar, which is still present today.

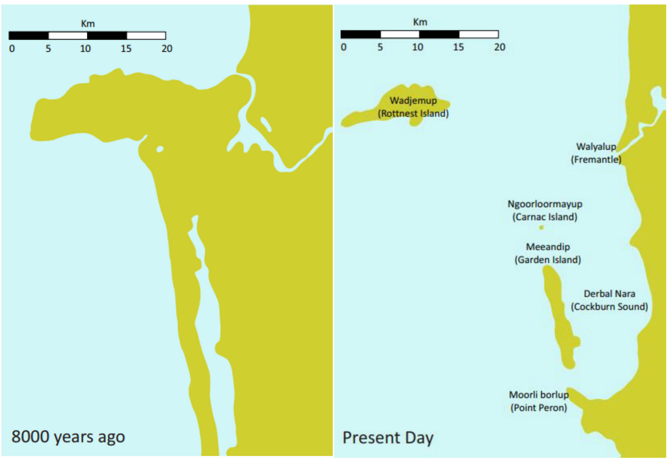

Figure 5
Schematic outline of the landscape around *Derbal Nara* before rising sea levels formed the current line of remnant islands and submerged reef that extends from Point Peron to Rottnest Island (sourced from https://derbalnara.org.au/).

From a Western science perspective, the fight between the Waugal and the Spirit Crocodile is interpreted as relating to post-glacial sea level rise and flooding of the shelf. Geomorphic, stratigraphic and sedimentological data obtained by Geoscience Australia indicate that flooding occurred around 9300 years ago, with the clay soil of the former terrestrial land surface now preserved beneath a layer of marine mud in the central basin (Figure 6).

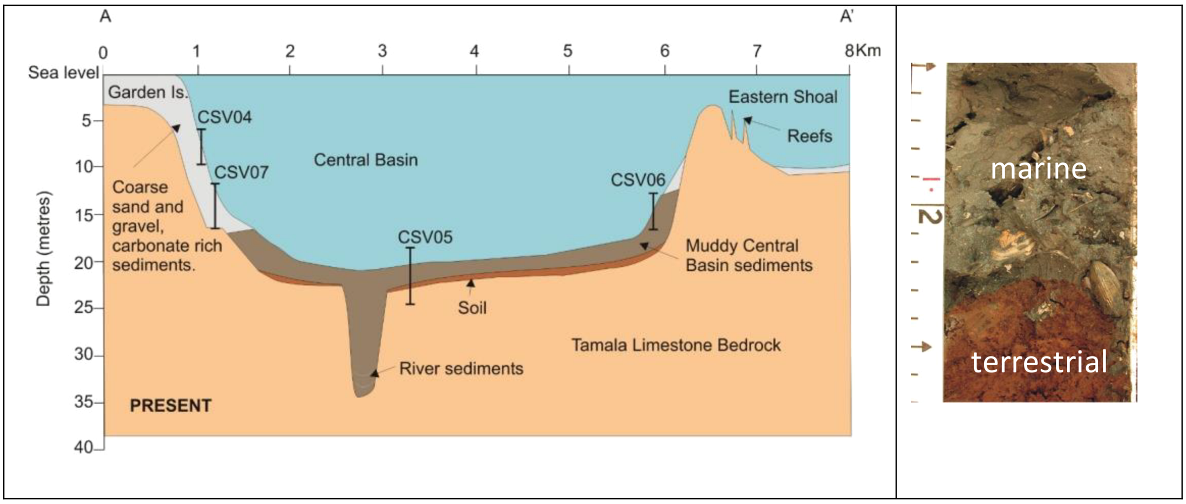

Figure 6
Stratigraphic cross section of Cockburn Sound, incorporating data obtained from vibrocores (sourced from and reproduced with permission from Skene 2005; their Figures 21 and 17E respectively). Scalebar on right is in cm increments.

In late 2021, the State Government allocated $13.5 million to a three-year research programme to manage and support the Cockburn Sound marine environment. The 30 different funded projects include programmes relating to key ecological and biological processes, and to social values research and protection strategies. The latter ideally include research aimed at integrating Western scientific and local Indigenous knowledge to reveal the submerged landscape and past and present ecology, helping to inform future EIAs and the future management of Cockburn Sound. At the same time, such research can help people from the Stolen Generation (Tatz 1999) rebuild intergenerational identity as well as providing a means of validating cultural sites and landscapes as needed under Western governance.

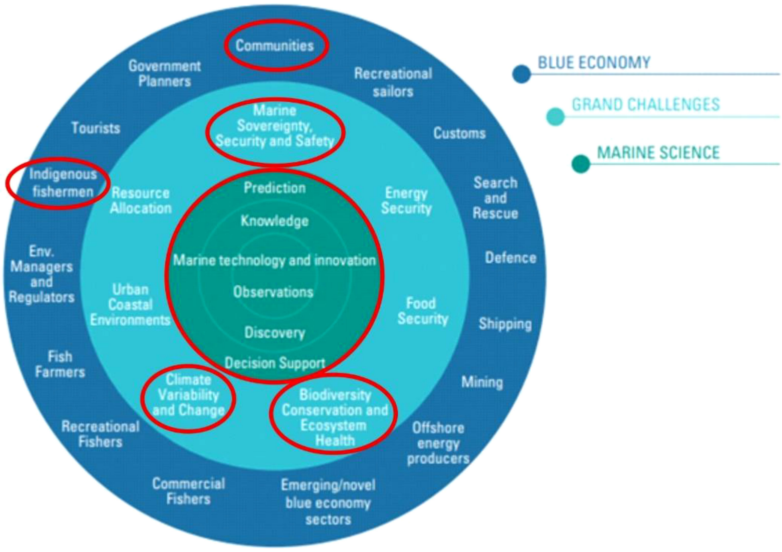

FIGURE 7
Schematic diagram outlining the range of stakeholder interests and science needs for Australia's marine estate, with Indigenous interests in submerged cultural resources possibly (although not explicitly) coming under the banner of communities and biodiversity, conservation and ecosystem health. Circles indicate main elements that relate to Indigenous communities (modified from the National Marine Science Plan 2015-2025).

(Section 12(a)) and specifically includes an Aboriginal place or cultural landscape (Section 12(b)). This Act is very recent, hence has yet to be applied to any underwater cultural site or landscape. It could, for example, be applied to any mythological site within the area of Cockburn Sound in Western Australia that relates to Indigenous narratives (recorded by Armstrong in 1836 and Moore in 1884) describing the separation of the islands from the mainland as influenced by the Rainbow Serpent (Waugal). In 2004, the Aboriginal Cultural Material Committee (Resolution 2004/082) reassessed this mythological site (Department of Indigenous Affairs (DIA) Site 3776) and deemed it 'Not a Site' under the Aboriginal Heritage Act (1972). Under the new WA Act 2021, Aboriginal people could register the area in an Aboriginal Cultural Heritage Management Plan (ACHMP) or apply to make it a Protected Area. Potential impacts or 'activities' in the area would then be graded under four tiers (levels), with the latter built into a ACHMP between a proponent and local Aboriginal cultural heritage services (LACHS). A similar Special Area Management Plan (or Ocean SAMP) exists for Rhode Island in New England, and combines stakeholders, including native Narragansett interests, with the best available science to develop a regulatory framework for the management and protection of Rhode Island ocean heritage (Fugate 2012; Olsen et al., 2014).[9] However, what many consider a shortfall in the current draft of the ACHMP is the ability of the Minister of Aboriginal Affairs to override these agreements for the 'wider public interest' (e.g., economic gain). Whilst there is uncertainty with regard to what and where submerged cultural resources exist, current statutory and regulatory regimes will continue to govern the use and management of coastal and marine zones (Quig, 2004). In the meantime, there is the continued need to engage with traditional owners and improve our understanding of these shared natural and cultural landscapes.

The way ahead - Ecosystem-based and community-led management

Successful and sustainable marine management needs to cover all the natural and social aspects of the seas. These can be described as the 10-tenets, nine of which relate to the socio-economic system and include human behavioral aspects of all parts of society (Elliott, 2013; Barnard and Elliott, 2015). A culturally-inclusive tenet was added particularly to accommodate countries with large indigenous populations such as Australia, New Zealand and Canada. All of the tenets rely on having a broad range of natural and social sciences and a fit-for-purpose understanding of the way policy and science interlink. This includes the need to obtain and use knowledge, data and understanding from all areas, both conventional

[9] See also http://seagrant.gso.uri.edu/oceansamp/

'Western-type' science as well as Indigenous knowledge and other stakeholder inputs.

Whilst based on terrestrial forest management with the Xáxli'p community in British Columbia, Diver (2017) nevertheless provides an excellent example of the mutual benefits of integrating Indigenous knowledge in science-policy that can be easily translated to the marine environment. Amongst the things that Diver (2017) lists in terms of shaping environmental science-policy are:

* acknowledging differences in cultures and worldviews but at the same time, generating strategic knowledge linkages between the two,
* training community members in scientific methods and technologies alongside cultural training,
* documenting and quantify specific components of Indigenous knowledge, and
* encouraging greater creativity in developing sustainable land (or marine) management policies

There are good examples in Australia where ecosystem-based management has been aligned with Indigenous rights and Indigenous expertise (Weiss et al., 2013; Davies et al., 2020; Goolmeer et al., 2021; Macpherson et al., 2021), and this simply needs to be extended to include historic and prehistoric marine cultural heritage, such as in the community-led archaeological research program in the Recherche Archipelago (see Box 1 – Recherche Archipelago). As Guilfoyle et al.,(2019) notes, the strength of this program is that the researchers, traditional owners and volunteers involved all bring different perspectives while sharing the same goal: to learn how best to understand, manage and protect these shared natural and cultural landscapes.

In contrast to the USA (Olsen et al., 2014), Canada[10] (Quig, 2004; Jones et al., 2021; see also Garrison and Hale, 2020) and the United Kingdom (Wickham-Jones, 2010), prehistoric cultural heritage in Australia has yet to be acknowledged as a critical resource in any part of the coastal or marine planning process, or in any nationally-coordinated seabed mapping, marine benthic studies or other related research. If the Australian commitment towards holistic marine management is to be achieved, then some revision is needed of environmental legislation within the marine environment. This includes adding cultural heritage assessments (potential or known) to any coastal and marine development work as a form of compliance or regulatory and industry monitoring, such as EIAs, best practice guidelines, or equivalent Ocean SAMPs. There are many exemplar studies of EIA's that accommodate Indigenous perspectives (e.g., O'Faircheallaigh, 2007; McKay and Johnson, 2017; Muir, 2018)[11] that can be applied to marine prehistoric cultural heritage and to marine spatial planning (e.g., Gee et al. 2017; Diggon et al. 2022). [3]The plethora[4] of marine governance illustrated by Boyes and Elliott (2014) also shows that there is a place for marine archaeology in European marine management and that it implicitly or explicitly is included in existing legislative instruments whether Acts or Regulations. However, such features are required to be identified and assessed before being protected. For the most part, this kind of assessment has fallen under the banner of self-monitoring, e.g., Ports Authorities and traditional owner groups (Guilfoyle et al., 2019), or investigative monitoring by researchers, increasingly with Indigenous involvement, showing the importance of citizen science (see also Borja and Elliott, 2021).

Similarly, as demonstrated for the Salish Sea in North America, political and administrative boundaries are often artificial and can lead to segmenting of ecosystems, with the alternative and preferred approach involving the use of ecological planning units such as catchments or estuaries, and direct stewardship by traditional owners (Jones et al., 2021). Whilst the catchment approach has yet to be considered, custodial rights of some Native Title groups along Australia's coasts are being extended from the land to adjacent waters in the form of joint management agreements with regulatory Marine Parks bodies. These at least provide scope to explore any ecological or cultural continuum between onshore and offshore areas. Successful Traditional Use of Marine Resources Agreements (TUMRAs) exist for parts of the Queensland coast, with the largest recently set up with the Darumbal people on the southern Great Barrier Reef. Similar Indigenous Land Use Agreements (ILUA) exist for the Wagyl Kaip and Southern Noongar traditional owners, with new agreements being set up along other parts of southern Western Australia (Guilfoyle et al., 2019). For most parts, non-exclusive sea rights of Native Title holders largely limit them to being stakeholders rather than resource custodians in conventional (i.e., Western) commercial or ecosystem management (Kaye, 2001).

Progress has also been made towards integrating cultural management with other policies such as Australia's Ocean Policy 1998, as well as supporting Indigenous people to develop their own management goals through Healthy Country Planning (as part of Conservation Action Planning)[12], the Ocean Discovery and Restoration Program[13] and associated government grant schemes[14], as well as the Indigenous Protected Areas (IPAs) designation (Smyth et al., 2016; Rist et al., 2019; Gould et al., 2021). The latter is part of a positive shift towards the more-

[10] https://coastalfirstnations.ca/our-sea/collaborative-governance-and-reconciliation-with-first-nations/a-first-nations-marine-planning-and-management-reconciliation-table/

[11] https://www.canada.ca/en/impact-assessment-agency/services/policy-guidance/practitioners-guide-impact-assessment-act/overview-indigenous-engagement-partnership-plan.html

proactive Indigenous-led planning, research, governance and management (e.g., Mayala Inninalang Aboriginal Corporation, 2019), as opposed to more-reactive Indigenous-engagement initiatives led by government and non-government agencies, mining/exploration companies, researchers amongst others (see also Smyth et al., 2016). Most marine-based programs and grant schemes fall within the latter, with the development of strategic alliances and partnerships between Traditional custodians and marine science and management agencies, with the shared realization that the integration of traditional knowledge and Western science provides a better way forward (e.g., Lincoln and Hedge, 2019; Shamsi et al., 2020; Diggon et al. 2022; Murley et al., 2022). Such aims are exemplified in the Australian Marine Parks Indigenous Engagement Program[15] and the Australian Institute of Marine Science (AIMS) Indigenous Partnerships Plan[16] (Evans-Illidge et al., 2020; Bock et al., 2021). To fully embrace Indigenous perspectives, these schemes need to be inclusive of all Sea Country, including submerged cultural heritage resources, and not separate from traditional terrestrial estates (see also Henderson, 2019).

There is also an increasing number of global initiatives to which many countries are signatories that aim to create more sustainable oceans for the coming decades (Borja et al., 2022). For example, the United Nations (UN) Decade on Ecosystem Restoration[17], the UN Decade of Ocean Science for Sustainable Development 2021–2030[18] (hereafter Ocean Decade), and the Global Sustainable Development Goal #14, Life Below Water[19] are all aimed at sustainable use of ocean resources. The Ocean Decade in particular promotes "the science we need for the ocean we want", with ocean science broadly encompassing social sciences and human dimensions. The Ocean Decade Heritage Network (ODHN)[20] was later established within the Ocean Decade to more explicitly integrate cultural heritage. The involvement of Indigenous groups in management need to feed into both achieving the Sustainable Development Goals (notably SDG#14) and those of the Ocean Decade. As population and demands upon the coastal and marine environment increase, marine management becomes increasingly complex (e.g., Elliott et al., 2020a; Elliot et al 2020b; Cormier et al., 2022) with cultural values sometimes a secondary consideration to more direct economic benefits and nature conservation (Atkins et al., 2015; Lee, 2016). Yet, as Henderson (2019) argues, activities in the marine zone can actually be linked to and given context by cultural heritage and, moreover, they can provide economic, social and cultural benefits and contribute to coastal and ocean sustainability (see also Lepofsky and Caldwell, 2013; Khakzad et al., 2015; Henderston et al., 2021; Yet et al., 2022). Furthermore, indicators have been derived for these aspects and monitoring and management can and should be directed towards the achievement of those indicators (Atkins et al., 2015).

As the studies above indicate, holistic- and process-based approaches provide a better way forward for investigating and managing underwater environments and their associated cultural heritage, with Indigenous knowledge engaging with the physical science 'on shared and equal terms' (Stevens and Paul-Burke, 2021). Both a top-down regional-scale approach and a bottom-up, site specific approach are needed (e.g., Gregory, 2015), and may incorporate high-resolution imagery to seamlessly link the seabed with adjacent coastal areas, sub-bottom profiling and sediment coring to investigate past sedimentary contexts, together with a range of oceanographic modeling exercises to identify and interrogate modern physical processes. From a global commons perspective, there should be much more sharing of this kind of data from offshore commercial development. This information can be used to relate past resource use and ecosystem features – and potentially cultural resources - to current geomorphological features and within the context of climate changes and its moving baselines (e.g., Vos et al., 2015). Furthermore, all of this can be integrated with an Indigenous understanding of landscape and the biophysical changes for maritime spatial planning. The ultimate aim is to "achieve the long-term conservation of values of nature, culture and associated ecosystem services" (Lee, 2016) for all interested stakeholders and the wider community.

It is important to learn lessons from similar situations worldwide. In New Zealand, this convergence (pūtahitanga) of physical knowledge (mātai) with Indigenous knowledge (mātauranga Māori) and including oral history, provides a set of tools for understanding past and present ocean currents, waves, tides, climate and so on to directly inform biophysical oceanography, ecology (Stevens et al., 2021) and by inference cultural heritage. In other words, the pūtahitanga allows for a better understanding and application of what science is required, how its results should be applied, and what the wider impacts be. Te pūtahitanga can be equally applied in an Australian context,

[12] https://www.natureaustralia.org.au/about-us/who-we-are/our-science/conservation-planning/

[13] https://parksaustralia.gov.au/marine/management/partnerships/ocean-discovery-and-restoration/

[14] https://www.awe.gov.au/agriculture-land/land/indigenous-protected-areas/sea-country-grant-opportunity

[15] https://parksaustralia.gov.au/marine/management/programs/indigenous-engagement/

[16] https://www.aims.gov.au/indigenous-partnerships

[17] https://www.decadeonrestoration.org/

[18] https://en.unesco.org/ocean-decade

[19] https://www.globalgoals.org/goals/14-life-below-water/

[20] https://www.oceandecadeheritage.org/

incorporating knowledge (*kaartdijin* in Noongar), language and songlines of Sea Country to better identify the Ocean Decade's "science we want for the ocean we need". Hence just as the sea connected communities in the past, it should serve to connect scientific approaches, management approaches, historical narratives, and human activities in the maritime space today (Henderson, 2019).

Conclusions

The UN Ocean Decade 2021-2030 has the aim of developing the "science we want for the ocean we need" (Borja et al., 2022), which we argue for Australia can be better achieved by being inclusive of underwater prehistoric heritage. Numerous studies demonstrate that cultural knowledge and practices can be integrated with science and policy to create successful management strategies appropriate for both natural and cultural resources (see also Kikili et al., 2017). However, at present, there remains a mismatch between what is known from an ecological and commercial exploration perspective with what is known (or even lost) in terms of the 55,000 years of more of marine prehistoric heritage on the Australian continental shelf. It is difficult, therefore, to delimit areas for protection and marine spatial planning where there is no written documentation or mapped cultural landscapes.

There also remains a mismatch between the protection of the seabed from a Common law perspective (for Australia relating to English Common Law) with that from Traditional lore, and local, passed-down (oral) knowledge. However, for both approaches the notion of commonness is viewed in terms of trusteeship and management participation rather than ownership. Whether on land or under water, traditional patterns of use and occupation constitute the source of Aboriginal title and as such, mandate both our understanding and respect (Quig, 2004). Accordingly, traditional knowledge related to marine ecosystems and seabed resources should be integrated with more conventional (Western) data and information in marine spatial planning and management (Tilot et al., 2021; Diggon et al. 2022). The studies presented aimed to demonstrate that fostering two-way knowledge of the submerged landscape and associated cultural resources allows for a more adaptive and holistic approach to marine governance.

Cultural landscapes are an inherent part of the natural landscape, and both inform each other. As new marine funding schemes are announced and more coastal and offshore areas are given over to Indigenous governance, these demonstrate the growing recognition of the ecological and cultural place of traditional marine management systems. However, a huge missing piece of the puzzle is an awareness and understanding of how these ecological and cultural places have evolved during the period of human occupation. Much of this understanding lies submerged or buried beneath a blanket of modern sediments, and is only through unearthing this that we can begin to really reveal our common heritage.

Data availability statement

The original contributions presented in the study are included in the article, further inquiries can be directed to the corresponding authors.

Author contributions

IW was the driving force behind the paper and all authors contributed to providing information and drafting and editing the text. The views expressed are those of the authors and not necessarily those of organization that they may represent or advise. All authors contributed to the article and approved the submitted version.

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Conflict of interest

Author ME is the unpaid Director of International Estuarine & Coastal Specialists Ltd. Author DG is employed by Esperance Tjaltjraak Native Title Aboriginal Corporation.

The remaining author declares that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Framing the science for technical measures used in regulatory frameworks to effectively implement government policy

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Regulatory and non-regulatory frameworks are used extensively to establish standards and guidelines for the technical measures implemented to manage freshwater and marine activities to achieve environmental policy objectives. Scientific and technical knowledge about the effectiveness of such measures is needed to ensure the success of these objectives, and yet there is general lack of scientific information on the effectiveness of technical measures. Used as conditions of approval for a variety of industry sectors, regulations and environmental quality guidelines establish the outcomes that are expected for the technical measures used in the daily activities of a given worksite. This paper suggests that the science to determine the effectiveness of technical measures should be framed from the requirements established in regulations and environmental quality guidelines. Such studies should also use methods, indicators and metrics that are often part of those requirements. This paper also puts forth that a more focused scientific effort is needed to determine the effectiveness of technical measures given the thousands of technical measures used to manage a wide range of activities.

KEYWORDS

technical measures, regulations, expected outcomes, effectiveness science, environmental quality guidelines

Introduction

Technical measures are controls, procedures, barriers, safeguards, and specifications that are implemented to address environmental policy objectives as well as health and safety concerns (Silva and Acheampong, 2015). The success of environmental legislation and policies depends greatly on the effectiveness of the technical measures implemented by development projects and industrial activities through regulations and guidelines (Cormier et al., 2022). Issued as authorizations, licenses, or permits, regulations and guidelines are used to establish the conditions of approval to undertake such projects or manage the daily activities of industry to comply with legislation. These conditions typically establish the outcomes that are expected for the technical measures that are implemented for these projects and activities. Ultimately, individuals and corporate entities have the responsibility to implement technical measures that are tailored to the specific activities of their worksite to comply with their conditions of approval (Smyth et al., 2015; Burdon et al., 2018).

Much of the environmental monitoring in natural resource management has been directed toward assessing the compliance of proponent activities against the conditions of permits, licenses, and authorizations to determine if these are appropriate (Van den Bosch and Matthews, 2017; Himberg et al., 2018). While compliance to law and regulations is clearly important, we propose that compliance is not likely to achieve management objectives if the outcomes of technical measure that are implemented in a worksite do not correspond with the outcomes that are expected in regulations and guidelines (Rytwinski et al., 2015; Theis et al., 2019). While this statement may seem self-evident, many of the technical measures currently in use have not been scientific evaluated for their effectiveness while others may still be using outdated information that has not been subject to review (Reichenberger et al., 2007; Gwimbi and Nhamo, 2016; Evans et al., 2021). There are likely several reasons for the paucity of information on the effectiveness of technical measures, we suspect that one important reason is that there is little guidance on how to frame scientific assessments of effectiveness for technical measures (May et al., 2017; Cormier et al., 2018; Getty and Morrison-Saunders, 2020).

Before exploring ways to frame the science for the effectiveness of technical measures, it is important to consider a working definition of effectiveness (Cormier et al., 2017). Effectiveness is used interchangeably to mean different things in policy, decision-making or environmental management (Giebels et al., 2016; Bigard et al., 2017). Effectiveness is sometimes used to express the performance of environmental conservation programs (Katsanevakis et al., 2020). In other situations, effectiveness may also be expressed in terms of the measures used to reduce the environmental impacts of an activity or the pressures from multiple activities (Borgwardt et al., 2019; Duarte and Sánchez, 2020; Elliott et al., 2020). Effectiveness of technical measures implemented to prevent and mitigate environmental impacts from the activities within a worksite is very different from the effectiveness of marine plans to reduce the pressures generated by multiple activities to address environmental effects (Stelzenmüller et al., 2021). In order to frame the science for technical measure effectiveness; it would be important to describe the role of technical measures in contrast to environmental policies and management plans.

In this paper, we aim to open discussions on how to frame science for evaluating the effectiveness of technical measures within the context of regulatory frameworks. We define components of such frameworks in terms of policies, plans and programs and describe the use of technical measures within the administration of regulatory programs. We do this in order to improve clarity on the role of technical measures and what we mean by the effectiveness of technical measures. We draw on a selection of Canadian codes of practice, regulations and environmental quality guidelines to demonstrate that these provide the expected outcomes required to frame scientific studies of technical measures effectiveness. We also discuss the importance of indicators, metrics, and methods established in such instruments to measure the outcomes of technical measures to ensure that the evidence generated is relevant to regulatory decision-making.

Understanding policies, regulatory and non-regulatory frameworks and technical measures

Figure 1 is used to understand the importance of the effectiveness for technical measures used in the implementation of regulatory and non-regulatory frameworks in contrast to the development of environmental management strategies (e.g. Fish and Fish Habitat Protection Policy Statement, August 2019 and European Marine Strategy Framework Directive (MSFD) (EU, 2008; EU, 2017; DFO, 2019a)). The questions asked by a manager having been given the mandate to develop such strategies (Figure 1: left pointing arrow) are not the same as for the regulator tasked with identifying the conditions of approval for development projects and industry activities (Figure 1: Right pointing arrow) (Cormier et al., 2022). A regulator has to review the technical measures being proposed for a given development project to determine if these can effectively meet the requirements of the regulations and environmental quality guidelines. In such a regulatory implementation, the focus shifts from scientific, technical, and management assumptions of what is needed for a management strategy to the assumptions that the expected outcomes established in regulations and guidelines can adequately protect and conserve the environment. The regulator works from the premise that the expected outcomes are *de facto* tolerance levels given the type activity being proposed for the worksite.

In risk management (IEC/ISO, 2019), minimum tolerance levels for acceptable risks are used when risk cannot be eliminated and that technical measures can only reduce the risks to a level "As Low As Reasonably Practicable" (ALARP)

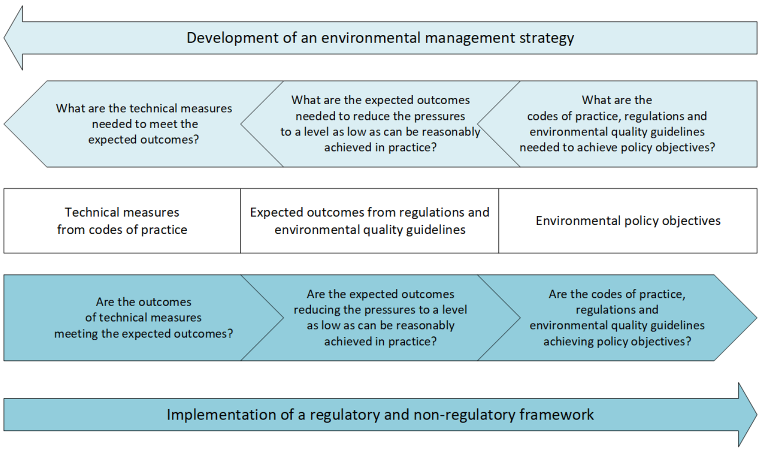

FIGURE 1
The difference between the questions for developing an environmental management strategy versus the implementation of regulatory and non-regulatory frameworks.

(Baybutt, 2014). Pressures are considered as the mechanisms and rates of change to the aquatic environment that occur in an area once avoidance and mitigation measures have been employed (Cormier et al., 2022) such as the disturbance of species due to human presence, mortality or injury to wild species, physical disturbance to seabed and input of substances, litter or energy (e.g. MSFD Table 2a. Anthropogenic pressures on the marine environment (EU, 2017)). Here, an expected outcome established in a regulation or an environmental quality guideline could be considered as the tolerance levels for the effectiveness of the technical measures used to reduce the pressures by operational activities within a worksite. Not discussed here is the scientific advisory processes used to establish such tolerance levels in the development regulations and environmental quality guidelines where new scientific knowledge would be needed to trigger a review of the regulations and guidelines.

The following examples are used to illustrate the differences between environmental policies (Table 1), regulations and environmental quality guidelines (Table 2), and technical measures (Table 3). In this paper, the science to determine the effectiveness of technical measures is framed around the question "Are the outcomes of technical measures meeting the expected outcomes?" (Figure 1).

Environmental policy objectives

Environmental policy objectives are typically found in international conventions and agreements as well as national legislation and policies (Cormier et al., 2022). Such policies provide the rationale for the actions that are needed and the objectives that are to be achieved. However, they do not specify how those objectives are to be achieved. The development of such policies are informed by the scientific advisory and peer review processes and assessments at various scales to reach a consensus as to the evidence used to formulate the advice (UN, 2021; DFO, 2022a; ICES, 2022; OSPAR, 2022). There is a long history of such advisory processes used to ensure the independence of the science used and the advice provided (CSTA, 1999; Rose and Parsons, 2015; Gluckman, 2016).

Table 1 summarizes three examples of such policies for discussion purposes. Although their rationale and objectives are similar, they differ mainly in terms of the spatial scale and the effects that are of concern (e.g. biological diversity, pollution, etc.).

Expected outcomes of regulations and environmental quality guidelines

Under the authority of legislation, regulations are used in the application and enforcement of that legislation (Canada, 2019; Canada, 2021). For example, regulations may include prohibitions for specific activities and standards for the release of substances as well as methods for monitoring those standards. Regulations are typically used by one competent authority as conditions for authorizations, licenses or permits. In contrast, environmental quality guidelines provide policy direction that may be adopted across multiple jurisdictions and industries (CCME, 2022). Similar to as in the case for regulations, they

TABLE 1 Examples of environmental policy rationales and objectives (EU, 2008; UN, 2015; EU, 2017; DFO, 2019a).

Policy	Why action is needed	Summarized Objectives
United Nations Sustainable Development Goals 14 Life below water (UN, 2015)	The ocean drives global systems that make the Earth habitable for humankind. Our rainwater, drinking water, weather, climate, coastlines, much of our food, and even the oxygen in the air we breathe, are all ultimately provided and regulated by the sea.	Conserve and sustainably use oceans, seas and marine resources for sustainable development in terms of the targets for marine pollution, ocean acidification, harvesting and overfishing, conserving coastal and marine areas, fisheries subsidies, and marine resources including capacity for scientific research and technologies, access for small scale artisanal fisheries and implementation of UNCLOS
Marine Strategy Framework Directive (EU, 2008; EU, 2017)	The marine environment is a precious heritage that must be protected, preserved and, where practicable, restored with the ultimate aim of maintaining biodiversity and providing diverse and dynamic oceans and seas which are clean, healthy and productive.	Achieve or maintain good environmental status in the marine environment in terms of biological diversity, non-indigenous species, commercially exploited fish and shellfish, marine food webs, eutrophication, sea-floor integrity, hydrographical conditions, pollution effects, fish and seafood, marine litter, as well as energy and noise
Canadian Fish and Fish Habitat Protection Policy Statement, August 2019 (DFO, 2019a)	Fish have long had economic, environmental, cultural and spiritual value to Canadians. Indigenous peoples have been fishing for many generations in Canada's oceans, along the coasts, in lakes, and in rivers. Commercial and recreational fisheries generate billions of dollars every year for the Canadian economy. Importantly, the productivity of a fishery is inextricably linked to the health of the habitat in which fish reside. Fish need suitable places to live, feed, and reproduce. They also need unobstructed corridors to migrate between these places.	Conserve and protect fish and fish habitat from habitat degradation, habitat modification, aquatic invasive species, overexploitation of fish, pollution, and climate change

may also establish standards for environmental quality. Indicators and metrics outlined in regulations and guidelines can be used to gauge the effectiveness of the technical measures implemented to manage operational activities within a worksite or the collective pressures generated by multiple activities within a management area. These indicators and metrics are not necessarily the same as the ones used to assess environmental impacts and effects.

The development of regulations and environmental quality guidelines are also informed by scientific advisory and peer review processes similar to the ones discussed above. However, the type of advice for such regulations and guidelines is about how much disturbance or change can be tolerated considering scientific uncertainties and the potential for impacts and effects (DFO, 2014). In principle, the development of regulations and environmental quality guidelines aims to achieve a balance between regulations that are too stringent to implement and regulations that are insufficient to protect people and the environment (Gouldson et al., 2009; UNECE, 2012).

Table 2 provides examples of regulations and guidelines that establish expected outcomes for very different development projects and industry activities. Expected outcomes are much more specific in terms of tolerance levels that are established for very specific causes of environmental impacts. As mentioned earlier, it is up to the individuals or corporate entities to engineer and implement technical measures that can meet the requirements of the regulations and environmental quality guidelines.

TABLE 2 Examples of regulations and environmental quality guidelines (Canada, 2019; Canada, 2021; CCME, 2022).

Regulations and environmental quality guidelines	Expected outcome
Canadian Environmental Protection Act, 1999 (S.C. 1999, c. 33) (*Canada, 2019*)	Part 7: Controlling Pollution and Managing Wastes. Division 1 – Nutrients Division 2 – Protection of the Marine Environment from Land-based Sources of Pollution Division 3 – Disposal at Sea
Fisheries Act *Potato Processing Plant Liquid Effluent Regulations (C.R.C., c. 829)* (*Canada, 2019*)	Schedule I: Authorized Deposits of Deleterious Substances levels for biochemical oxygen demanding matter and total suspended particulate matter
Fisheries Act *Metal and Diamond Mining Effluent Regulations (SOR/2002-222)* (*Canada, 2019*)	Schedule 4: Maximum authorized concentrations of prescribed deleterious substances
Canadian environmental quality guidelines for the protection of aquatic life in freshwater and marine systems (CCME, 2022)	Establishing guidelines for a variety of substances, total particulate matter, temperature, pH, nutrients, etc.

TABLE 3 Examples of technical measures from codes of practice (USDA, 2001; ECCC, 2009; AB, 2011; NB, 2012; DFO, 2019b).

Codes of Practice	Controls, procedures, barriers, safeguards, and specifications
Fish and fish habitat protection standards and codes of practice (DFO, 2019b)	Measures to protect fish and fish habitatBeaver dams removalCulvert maintenanceEnd of pipe fish protection screens for small water intakes in freshwater Routine maintenance dredgingTemporary cofferdams and diversion channelsTemporary stream crossings
Environmental Code of Practice for Metal Mines (ECCC, 2009)	Mine life cycle activitiesEnvironmental concerns through the mine life cycleRecommended environmental management practices
New Brunswick Watercourse and Wetland Alteration Guidelines (NB, 2012)	Site and water managementSurface erosion and sediment controlsTiming of instream workMigratory and sensitive periods for aquatic speciesGuidelines for the type of watercourse and wetland alterations
Erosion and sediment control manual for transportation (AB, 2011)	Selection of BMP for erosion and sediment controlPermanent erosion and sediment control plan
Stream corridor restoration: Principles, Processes, and Practice (USDA, 2001)	Restoration techniques and criteria

Industry codes of practice for technical measures

Here, the term code of practice is used generically as best industry practices, industry standards, standard operating procedures, quality management programs, etc. Codes of practice provide practical guidance as to how the operational activities are to be controlled within a worksite to comply with regulations and environmental quality guidelines. The keyword here is "practice". Codes of practice are to put into practice the technical measures needed to meet the expected outcomes of regulations and environmental quality guidelines (Cormier et al., 2022).

The development of codes of practice also requires the input of scientists, engineers, and regulators considering the environmental implications of failing to meet the expected outcomes and the practical implementation of the technical measures in the daily operational activities of a worksite. The technical measures outlined in a code of practice provide guidance for the engineering needed to tailor these measures to the worksite of a development project or industry activity. In an environmental context, every worksite is located in very different environmental situations. Although the effectiveness of a technical measure to meet an expected outcome seems straightforward, these measures may not be reliable in every environmental situation where additional measures may be needed to meet the conditions of approval.

Table 3 provides examples of different codes of practices. These contain technical measures to address very specific activities. Some are for very small undertakings such as removing a beaver dam while others involve large industry activities such as construction, operation, and decommissioning in mining.

Framing the science for the effectiveness of technical measures

Up to this point, we discussed the roles of regulations and environmental quality guidelines in setting requirements and the role of codes of practices that outline the type of technical measures needed to meet these requirements. In the following, we examine the practical application of these ideas and concepts to demonstrate how the expected outcomes established in regulations and environmental quality guidelines are used to frame a study that would be needed to determine effectiveness. The examples presented start with the more prescribed requirements of a regulation in contrast to an environmental quality guideline and restoration techniques.

Potato processing plant liquid effluent regulations

Under the authority of the Canadian *Fisheries Act* (Canada, 2019), deleterious substances are managed by limiting the daily amounts to be deposited through regulations. As a policy objective, the Fish and Fish Habitat Protection and Pollution Prevention provisions of the *Act* prohibits the deposit of deleterious substances to fish unless the deposit has been authorized by regulation. These regulations establish the conditions that individuals and corporate entities must comply with having the responsibility to engineer their processes in such a way that their effluents do not exceed the authorized daily deposits. In this example, we used the potato processing regulation for liquid effluent established in 2009 (Table 4). For discussion purposes, the expected outcomes of this regulation are considered here as tolerance levels to avoid the degradation or alteration of the quality of fresh and marine waters for such operational activity.

A potato processing plant has to meet authorized deposits for biochemical oxygen demanding matter and total suspended matter. Biochemical oxygen demanding matter and total suspended matter would be the indicators of effectiveness for the expected outcomes of the technical measures implemented to control the processes of the plant. In this example, the regulation also prescribes the standard analytical methods (e.g. APHA) that would be needed for such a study. The technical measures would be considered effective when their outcomes meet the expected outcomes of the regulation consistently over time. Given that the regulation prescribes the standard analytical methods, the results of any other scientifically valid indicator and metric would not be admissible to determine the effectiveness of the technical measures in meeting the requirements of the regulation.

Water quality guidelines for total particulate matter

Since 1964, the Canadian Council of Ministers for the Environment (CCME, 2022) has established a broad range of environmental quality guidelines for use in the various jurisdictions of the country. The Canadian Water Quality Guidelines for the Protection of Aquatic Life covers a broad range of water quality issues that can be used in freshwater and marine environments. Compared to a regulation, an environmental quality guideline does not have the force of law; but, can still be used to identify the expected outcomes needed to study the effectiveness of technical measures. In this example, we use the CCME guideline for total particulate matter (Table 5). Updated in 2002, this guideline provides tolerance levels for suspended sediment, turbidity, bedload sediments, and streambed substrate for freshwater, estuarine and marine environments. The levels established in the guidelines are calculated against natural background levels.

Adapted from multiple sediment and erosion control technical measures (AB, 2011; NB, 2012; DFO, 2022b), the concentration of sediments or the increase in Nephelometric turbidity units (NTU) of the watercourse would be the indicators of effectiveness for the expected outcomes of the sediment and erosion controls implemented within a worksite. The study would also have to establish the background levels for the same indicators and would need to track the number of times and duration that those levels were exceeded. Although this particular guideline may not prescribe standard analytical methods as discussed for the potato effluent regulation, the indicators and metrics used for such study would, nevertheless, have to match those of the guideline. The sediment and erosion controls would be considered effective when their outcomes are below the expected outcomes established in the guideline.

Stream corridor restoration

Revised in 2001, the stream corridor restoration manual provides a wide range of restoration techniques for instream practices, streambank treatment, water management, channel reconstruction and other stream corridor measures (USDA, 2001). For example, a development project near any

TABLE 4 Fisheries Act (R.S.C., 1985, c. F-14): Potato Processing Plant Liquid Effluent Regulations (C.R.C., c. 829)* (Canada, 2019).

Technical measures	Measured Out-comes	Expected outcome
Authorized Deposit of Deleterious Substances5 Subject to these Regulations, the owner of a plant of a class set out in Column I of Schedule I may deposit a deleterious substance prescribed by section 4 if(a) the actual daily deposit of each deleterious substance, determined in accordance with subsection 11(1), does not exceed the authorized daily deposit of that substance for that class of plant as set out in Column III of that Schedule;(b) the average daily deposit of each deleterious substance during a month, determined in accordance with subsection 11(2), does not exceed the authorized average daily deposit of that substance for that class of plant as set out in Column IV of that Schedule; and(c) the pH of each composite sample of effluent, determined in accordance with subsection 9(3), is between 6.0 and 9.0.	InterpretationBiochemical oxygen demanding matter means the substance contained in the effluent from a plant that results from the operation of a plant and that will exert a biochemical oxygen demand;Total suspended matter means the non-filterable residue that results from the operation of a plant, that is contained in the effluent from that plant.	Schedule IPotato Chips Plant:Authorized actual daily depositBiochemical Oxygen Demanding Matter: 1.5 kg/tonne of raw potatoes processedTotal Suspended Matter: 2.1 kg/tonne of raw potatoes processesAuthorized average daily depositBiochemical Oxygen Demanding Matter: 0.5 kg/tonne of raw potatoes processedTotal Suspended Matter: 0.7 kg/tonne of raw potatoes processesOther Potato Products Plants: Canned potato products, dehydrated potato products, frozen potato products and potato starchAuthorized actual daily deposit Biochemical Oxygen Demanding Matter: 2.7 kg/tonne of raw potatoes processedTotal Suspended Matter: 2.4 kg/tonne of raw potatoes processesAuthorized average daily depositBiochemical Oxygen Demanding Matter: 0.9 kg/tonne of raw potatoes processedTotal Suspended Matter: 0.8 kg/tonne of raw potatoes processesSchedule IIAnalytical Test Methods For Determining Presence and Concentrations of Deleterious Substances in EffluentsBiochemical Oxygen Demanding Matter (BOD): APHA Section 507Total Suspended Matter: AHPA Section 208DAHPA: Standard Methods for the Examination of Water and Waste Water, 14th Edition (1975), published jointly by the American Public Health Association, American Water Works Association and the Water Pollution Control Federation

*The information presented here is to be used within the context of this paper discussion only. Please refer to the actual regulations for its application.

TABLE 5 Canadian Water Quality Guidelines for the Protection of Aquatic Life - Total Particulate Matter (CCME, 2002).

Technical measure	Measured Outcome	Expected outcome
Install sediment and erosion controls prior to beginning the work and maintain controls until all banks and exposed soils have been stabilized	Changes in sediment concentration above background levels of the water course during the activities within the worksite	**Suspended Sediments for clear flow** Maximum increase of 25 mg·L^{-1} from background levels for any short-term exposure (e.g., 24-h period). Maximum average increase of 5 mg L^{-1} from background levels for longer term exposures (e.g. input lasting between 24 hours and 30 days. **Suspended sediments for high flow** Maximum increase of 25 mg·L^{-1} from background levels at any time when background levels are between 25 and 250 mg·L^{-1}. Should not increase more than 10% of background levels when background is >250 mg·L^{-1}. **Nephelometric turbidity unites (NTU) for clear flow** Maximum increase of 8 NTUs from background levels for a short-term exposure (e.g., 24 hours period). Maximum average increase of 2 NTUs from background levels for a longer term exposure (e.g., 30 day period). **Nephelometric turbidity unites (NTU) for high flow** Maximum increase of 8 NTUs from background levels at any one time when background levels are between 8 and 80 NTUs. Should not increase more than 10% of background levels when background is >80 NTUs.

watercourse most often require temporary changes of a watercourse and its banks during the construction phase of the project. Once the construction is completed, the temporary changes need to be restored to return the watercourse to a state and function essential to support aquatic life.

This restoration manual is used as our final example because effectiveness in this situation is not simply related to the expected outcome of one or more indicators. Adapted from technical measures outlined in multiple watercourse alteration guidelines (AB, 2011; NB, 2012; DFO, 2022b), the recommended techniques and criteria established in this manual would be used to evaluate the effectiveness of the stream geomorphology restoration in terms of the techniques and criteria used. Although monitoring would be required to determine the success of the restoration in terms of the return of aquatic life in the longer term, the restoration would be considered to be effective through the application of the recommended techniques and practices.

Discussion

The expected outcomes of technical measures are established by regulations and environmental quality guidelines. As such, those expected outcomes should ultimately frame the science needed to determine the effectiveness of technical measures. As shown for the potato effluent regulation, the total particulate matter guideline, and the stream restoration techniques, the expected outcomes may be expressed as one or more indicators or as techniques and criteria. Regulations can also prescribe the indicators, the metrics and the analytical methods to be used for such a study. Management would not be able to use other scientifically valid indicators, metrics and methods in a regulatory decisions. The latter could not be used as evidence of non-compliance with regulatory requirements when such a study did not use the prescribed analytical methods in regulations.

Expected outcomes established in regulations and environmental quality guidelines are tolerance levels considering the policy objectives that are being sought. The total particulate matter guideline (e.g. CCME) is a good example because it provides tolerance levels for the magnitude of change and duration in the increase in sediments and turbidity above background levels in relation to the exposure of the aquatic organisms that were considered when these were established. As long as the outcome of the implemented sediment and erosion controls remain below the tolerance levels for sediment and turbidity, the increase and duration of the changes in sediment and turbidity is considered tolerable given the need to protect aquatic life. This would imply that the sediment and erosion controls of a worksite are effectively reducing the quantity of sediment laden water reaching a watercourse to levels as lows as can be reasonably expected in practice. However, the science to establish such tolerance level would have been based on the sublethal and lethal effects of habitat impairments caused by suspended sediments and habitat sedimentation within the context of a policy for the protection of fish and fish habitats (CCME, 2002; DFO, 2019a).

Once a regulation and an environmental quality guideline are in effect, their expected outcomes are used systematically as conditions of approval for thousands of development projects and industry activities from that moment onwards (Cormier et al., 2022). The same can be said of the technical measures outlined in codes of practice. As long as the technical measures meet the expected outcomes, they are considered effective. For the three examples provided (Tables 4, 5, 6), they have been used for decades with updates in the last ten years or so. Changes to expected outcomes and technical measures require scientific studies that are dedicated to effectiveness in order to provide the justification for updating regulations and environmental

TABLE 6 Stream Corridor Restoration: Principles, Processes, and Practices (USDA, 2001).

Technical measure	Measured Outcome	Expected outcome
Restore stream geomorphology (i.e., restore the bed and banks, gradient and contour of the waterbody) to its initial state	Changes to the geomorphology and habitat structure of a watercourse	**Appendix A: Techniques** **Instream Practices**: Boulder Clusters, Weirs or Sills, Fish Passages, Log/Brush/Rock Shelters, Lunker Structures, Migration Barriers, Tree Cover, Deflectors, Control Measures **Streambank Treatment**: Bank Shaping and Planting, Branch Packing, Brush Mattresses, Coconut Fiber Roll, Dormant Post Plantings, Vegetated Gabions, Joint Plantings, Live Cribwalls, Live Stakes, Live Fascines, Log, Rootwad, and Boulder Revetments, Riprap, Stone Toe Protection, Tree Revetments, Vegetated Geogrids **Water Management**: Sediment Basins, Water Level Control **Channel Reconstruction**: Maintenance of Hydraulic Connections, Stream Meander Restoration **Stream Corridor Measures**: Livestock Exclusion or Management, Riparian Forest Buffers, Flushing for Habitat Restoration

quality guidelines and for improving technical measures outlined in codes of practice.

Conclusions

Technical measures are used to manage thousands of activities and their pressures in both freshwater and marine environments. Technical knowledge is needed to understand the effectiveness of technical measures in meeting the requirements of regulations and environmental quality guidelines. This need does not preclude the importance of the scientific knowledge used to establish the expected outcomes of these regulations and guidelines. The science to determine the effectiveness of technical measures is very different from ongoing scientific research on impacts and effects. Effective technical measures are needed to deliver programs for the protection and conservation of aquatic life and their habitats in both freshwater and marine environments. These programs have to provide a comprehensive suite of regulations, environmental quality guidelines and codes of practice to provide guidance for those that have to engineer and tailor technical measures to their activities and worksites to effectively reduce their pressures.

In this paper, we demonstrate the importance of using regulations and guidelines to frame the science needed to determine the effectiveness of technical measures using a few examples. We recognize that there would also be a need for scientific research to inform management decisions to establish the tolerance levels used in regulations and guidelines and also to revise the levels that are already in place. We consider that this paper is a small step in moving from the current science-policy interface providing scientific knowledge for policy to a needed science-management interface of structured scientific and technical frameworks to establish tolerance levels and to determine the effectiveness of technical measures.

Author contributions

RC, TT and MM contributed to the conception of the work, the analysis of causality and the writing of the paper. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Identifying priority areas for improvement in Chilean fisheries

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Chile is amidst an unprecedented legal and institutional change since the restoration of democracy at the end of the 80's, which is expected to affect fisheries governance. A global lead in marine resource landings, Chile implemented significant fisheries management reforms in the past decade. Yet, Chilean fisheries still face sustainability challenges. In this paper we reflect on the results of a survey carried out in 2019-2020 with key informants aimed to identify fisheries policy reform priorities in country. Addressing Illegal, unreported, and unregulated fishing; Developing a priority national research agenda to improve fisheries management in Chile; Addressing the lack of legitimacy of the fisheries law; Developing a new national fisheries policy; and Update the Artisanal Fisheries Registry were identified as priority topics by respondents.

KEYWORDS

Chilean fisheries, fisheries management, policy, governance, sustainability

1 Introduction

In a declaration made on August 2022, the Chilean President Gabriel Boric Font gave "extreme urgency" to the bill, that was initially presented on January 19, 2016, that declares the nullity of the current Fisheries Law, which was delegitimized due to the court's conviction, after 9 years since, a former senator and a former Member of Parliament for receiving bribes from a major fishing company (El Mostrador, 2022a) during the development of the law. The government also announced that a Bill for a New Fisheries Law will be presented between April and May 2023. This process should start in September 22 with debates in a pre-legislative work (El Mostrador, 2022b). All these processes are very relevant for ocean conservation and fisheries sustainability as Chile is one of the leading producers and exporters of marine resources in the world (FAO, 2022). In 2019, fisheries sector landings amounted to 2 million tons, 49 percent of which were caught by the artisanal sector (SERNAPESCA, 2019). Anchoveta, Jack mackerel, Araucanian herring, Chub mackerel, Jumbo flying squid, and Chilean hake are among the most important resources in terms of catch volume (SUBPESCA, 2020a).

In recent years, Chile has implemented significant fisheries regulation and management reforms, including updating its fisheries legislation with the implementation of a polycentric governance model (Gelcich, 2014) that incorporates co-management mechanisms covering the most important fisheries through Management Committees[1] (MC) and Technical Scientific Committees and the subsequent modernization law (N° 21.132, 2019) of the governmental fisheries control agency SERNAPESCA (National Fisheries and Aquaculture Service). By 2019 Chile took a step forward in terms of transparency by uploading satellite vessel information to the Global Fishing Watch platform[2], and has considerably increased the surveillance efforts at the landing sites. These and other developments allowed some industrial fisheries, such as Southern hake and Jack mackerel, to obtain MSC certification[3]. The 2020 accountability paper of the Undersecretariat for Fisheries and Aquaculture (SUBPESCA as per its acronym in Spanish) reported improvements in the stock health of Anchoveta fisheries (from Valparaíso to Los Lagos), Jack mackerel (from Arica and Parinacota to Los Lagos), Chilean seabass (from Arica and Parinacota to -47° SL) and Chilean seabass (47°-57° SL) (SUBPESCA, 2021). Despite this, the SUBPESCA, 2021 report shows that, out of the 27 national fisheries managed with biological reference points, 1 was underexploited, 12 were fully exploited, 8 were overexploited, and 6 were depleted or collapsed. These findings indicate that, Chilean fisheries still face important sustainability challenges.

This document presents an analysis of the results of a 2020 survey to key stakeholders to identify policy improvement priorities related to the fisheries sector. The aim of the survey was to understand the stakeholders' points of view, to contribute to the improvement processes in public policy, governance, and fisheries management by identifying priorities and solutions. After what can be considered a political impasse in Chilean politics since the so called "social explosion" in October 2019 and the follow-on constitutional process coupled with the COVID pandemic, we consider the results of this 2020 study are still relevant for the institutional process going forward as the policy focus harks back to legal reform affecting fisheries governance.

2 Methodology

This study builds upon a consultation process carried out with key fisheries stakeholders of Chile, specifically focused on gathering their opinion on public policy improvement priorities to advance sustainability in Chilean fisheries. The study also aimed to identify the main issues along with the main possible improvement measures for the different prioritized topic. Data gathering was carried out in three stages.

During the first stage, a literature review was conducted in 2019 to identify recurrent topics pointed to in the literature as policy improvement areas for fisheries management in Chile. A total of 16 topics were identified in peer-reviewed journals, gray literature and journalistic reports written by fisheries specialists. The topics identified were (1) Illegal, unreported and unregulated (IUU) fishing; (2) Integration of an economic and social perspective in fisheries research and management; (3) Update of the Artisanal fisheries registry (RPA); (4) Development of a priority national research agenda to improve fisheries management in Chile; (5) Legitimacy of the current fisheries law; (6) Participative assessment of the fisheries management systems effectiveness; (7) Development of a new national fisheries policy; (8) Use of fishing gear which damages coastal habitats within the first five marine miles; (9) Consumption of seafood from Chile in order to guarantee food security in the country; (10) Incentives to the participation of professionals in the scientific technical committees (CCT), (11) Economic resources for the operation of management committees; (12) Transfer of tradable fishing licenses among sectors; (13) Creation of the Ministry of Fisheries and Aquaculture; (14) Representativeness of women in the consultation and management decision making bodies; (15) Indigenous people's rights on fishing resources; and (16) Structure of the Fisheries Promotion Institute's governing board (IFOP) (Table 1).

In a subsequent stage, an online survey[4] (Supplementary Material 1) was carried out comprising of open-ended and closed-ended questions divided into three sections. The first section sought to collect general information. The second section was composed by closed-ended questions and allowed each informant to determine the relevance of each of the 16 previously identified topics using the Likert scale (from "not relevant" to "very relevant") and also included a space to propose additional topics. Finally, the third section required each respondent to choose the most urgent topic, as well as the second and third most urgent topics. Respondents were also asked to explain the issues derived from each topic and to propose specific actions to solve them.

The online survey was distributed to the largest possible number of fishery experts and stakeholders through a snowball sampling strategy. A total of 152 responses were collected from key informants from the artisanal sector, academia, government officials, industrial sector professionals, Small and Medium Enterprises (SMEs) and non-governmental organizations (NGO)s. The responses were collected between February 4 and April 7, 2020.

The third stage of the study focused on analyzing the content of the surveys. The respondents were classified by sector and by years of experience (Figure 1), as well as by gender (Figure 2). The relevance given by respondents to each topic was quantified. The relevance level of each topic was considered as the weighted average of the answers given by each respondent. Each respondent had the opportunity to rate each topic by assigning a score. The topics considered "not relevant" were given a "0" score, topics "slightly relevant" scored "1", "regular" topics scored "2", "relevant" topics scored "3" and topics considered "very relevant" were given the score of "4". The topics with a weighted average between 4.00 and 3.01 were considered "very relevant", those scoring between 3.00 and 2.01 were considered "relevant", scores between 2.00 and 1.01

1 http://www.subpesca.cl/portal/616/w3-propertyvalue-38010.html

2 https://globalfishingwatch.org/

3 https://fisheries.msc.org/

4 https://es.surveymonkey.com/r/5SYV9VW

TABLE 1 Topics considered in the consultation process.

Topic	Source
Illegal, unreported and unregulated (IUU) fishing	(FAO, 2016), (Ramírez, 2018), (IFOP, 2018a), (Biblioteca del Congreso Nacional de Chile, 2018), (IFOP, 2016), (Ríos, 2015), (Ruiz Muller et al., 2020), (Portal del Campo, 2017), (Equipo el Día, 2019), (Empresa Océano, 2017), (Diario Concepción, 2018), (Carrere, 2018), (AQUA, 2018)
Integration of an economic and social perspective in fisheries research and management	(Aceituno et al., 2017), (Tapia et al., 2016), (González Poblete et al., 2013)
Update of the Artisanal fisheries registry (RPA)	(FAO, 2016), (Bezamat, 2017), (Villanueva García Benítez and Flores-Nava, 2019), (AQUA, 2019), (AQUA, 2017), (Chile Atiende, 2019), (Tapia et al., 2016)
Development of a priority national research agenda to improve fisheries management in Chile	(FAO, 2016), (Ríos, 2015), (Observatorio Regional de planificación para el Desarrollo de América Latina y el Caribe, 2018), (Tapia et al., 2016)
Legitimacy of the current fisheries law	(Reyes et al., 2017), (Senado de la República de Chile, 2019), (Andrade Bone, 2016), (Ibañez, 2018), (Partarrieu, 2015)
Participative assessment of the fisheries management systems effectiveness	(FAO, 2016), (Monteza, 2020), (Ríos, 2015), (Reyes et al., 2017), (Tapia et al., 2016)
Development of a new national fisheries policy	(FAO, 2016), (Aceituno et al., 2017), (Ríos, 2015), (Reyes et al., 2017)
Use of fishing gear which damages coastal habitats within the first five marine miles	(Cuba, 2019), (Queirolo et al., 2019)
Consumption of seafood from Chile in order to guarantee food security in the country	(FAO, 2016), (FAO, 2017), (Aceituno et al., 2017), (Villanueva García Benítez and Flores-Nava, 2019), (IFOP, 2018b), (Tapia et al., 2016)
Incentives to the participation of professionals in the scientific technical committees (CCT)	(FAO, 2016), (Aceituno et al., 2017), (Reyes et al., 2017)
Economic resources for the operation of management committees	(FAO, 2016), (Aceituno et al., 2017), (Reyes et al., 2017)
Transfer of tradable fishing licenses among sectors	(Bezamat, 2017), (Ríos, 2015), (Ríos and Gelcich, 2017)
Creation of the Ministry of Fisheries and Aquaculture	(Aceituno et al., 2017), (Portal Frutícola, 2020), (Colegio de Ingenieros Agrónomos de Chile, 2019)
Representativeness of women in the consultation and management decision making bodies	(Servicio Civil, 2017), (SUBPESCA, 2019), (Villanueva García Benítez and Flores-Nava, 2019), (Soy Chile, 2019)
Indigenous people's rights on fishing resources	(FAO, 2016), (Aceituno et al., 2017), (Villarroel, 2017), (Bezamat, 2017), (Villanueva García Benítez and Flores-Nava, 2019), (Ruiz Muller et al., 2020), (Hiriart-Bertrand et al., 2020)
Structure of the Fisheries Promotion Institute's governing board (IFOP)	(Aceituno et al., 2017), (Ríos, 2015)

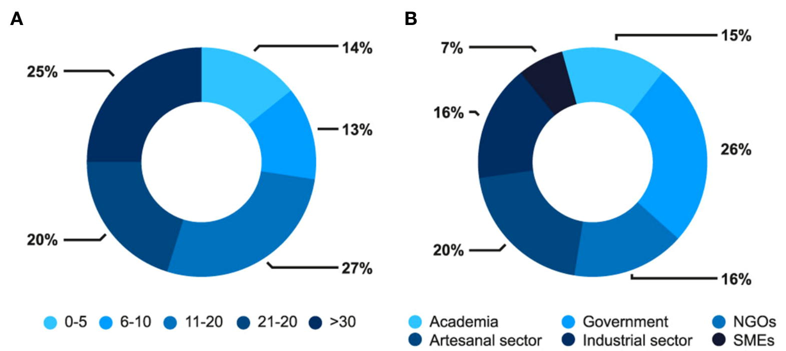

FIGURE 1
(A) Respondents by years of experience in the fisheries industry and **(B)** Respondents by sector.

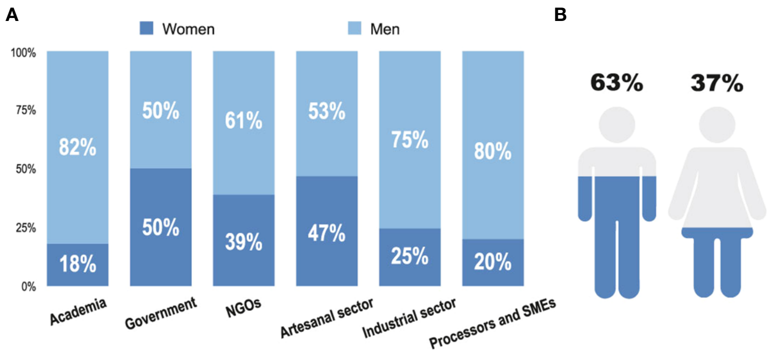

FIGURE 2
(A) Respondents by gender according to sector and **(B)** total number of people surveyed.

were considered "regular" and those between 1.00 and 0.01 were considered "slightly relevant".

Further to the above, the level of urgency to address to the topics was quantified, assigning a score of three to the most urgent topics, and a score of two and one to the second and third most urgent topics respectively. In this way, the study defined the priority topics that, according to the respondents, are the most urgent to be addressed. Finally, using the information provided by the respondents on the issues and possible solutions, the study described the five most urgent topics to be addressed in order to improve fisheries management in Chile.

3 Results

"Illegal, unreported and unregulated (IUU) fishing" was by far the most urgent issue to be addressed, scoring 120 points (Table 2; Figure 3). This was followed in second and third places by "developing a priority national research agenda to improve fisheries management in Chile" and "legitimacy of the current fisheries law," scoring 71 and 69 points, respectively. Ranked in fourth and fifth places, with 63 and 62 points, respectively, were "developing a new national fisheries policy" and "updating the Artisanal Fisheries Registry (RPA by its acronym in Spanish)". Further to the 16 improvement areas initially

TABLE 2 Relevance and urgency score of the issues proposed in the survey.

Issue		Relevance	Urgency
T1	Illegal, unreported, and unregulated (IUU) fishing	Very relevant (3.60)	120
T2	Development of a priority national research agenda to improve fisheries management in Chile	Very relevant (3.32)	71
T3	Legitimacy of the current fisheries law	Very relevant (3.32)	69
T4	Development of a new national fisheries policy	Very relevant (3.33)	63
T5	Update of the Artisanal Fisheries Registry (RPA)	Very relevant (3.27)	62
T6	Creation of the Ministry of Fisheries and Aquaculture	Relevant (2.94)	56
T7	Integration of economic and social perspectives in fisheries research and management	Very relevant (3.41)	51
T8	Participative assessment of the fisheries management systems effectiveness	Very relevant (3.23)	48
T9	Consumption of seafood from Chile in order to guarantee food security in the country	Very relevant (3.22)	37
T10	Representation of women in the consultation and management decision-making bodies	Relevant (2.75)	30
T11	Use of fishing gear that damages coastal habitats within the first five marine miles	Very relevant (3.13)	23
T12	Economic resources for the operation of management committees	Relevant (2.92)	23
T13	Indigenous peoples' rights on fishing resources	Relevant (2.52)	23
T14	Transfer of tradable fishing licenses among sectors	Relevant (2.83)	21
T15	Incentives for the participation of professionals in the scientific technical committees (CCT)	Relevant (2.88)	12
T16	Structure of the Fisheries Promotion Institute's governing board (IFOP)	Relevant (2.34)	7

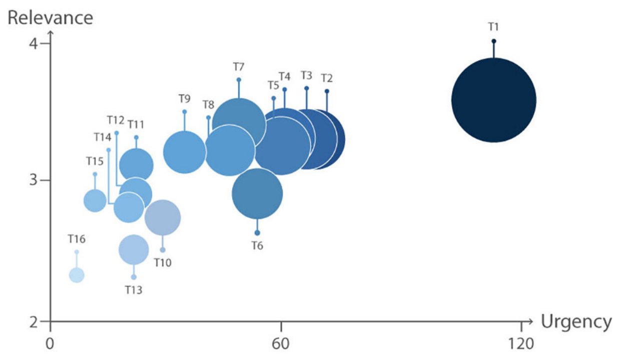

FIGURE 3
Topics by relevance and urgency level. Relevance: "Very relevant": [4.00;3.00> and "relevant": [3.00:2.00>. Urgency is measured as the score assigned by respondents surveyed. Circle diameter = relevance x urgency. The topics are provided in Table 2.

identified in the first stage (literature review) and scored for relevance and urgency by all 152 respondents, two new improvement areas were identified by 27 respondents, mainly from the NGO sector. Specifically, the development of an ecosystem approach and climate change adaptation within the fisheries[5]. Yet, the analysis below focuses only on the five top priority improvements areas for policy reform. For details on the distribution of priority improvement areas per respondent's sector see Figure 4.

3.1 Illegal, unreported, and unregulated fishing

Most respondents chose IUU fishing as the most urgent issue to be addressed in the Chilean fisheries sector. In fact, it was ranked the most urgent issue by government officials, the industrial sector and academia. In such regard, a law was enacted in January 2019 to modernize and strengthen SERNAPESCA.

Despite this law being relatively new, respondents were able to identify certain structural problems and possible solutions to tackle problems associated with IUU fishing. Respondents pointed to structural gaps such as the lack of human resources (HR), technology, and budget available for SERNAPESCA to perform monitoring, control, and surveillance (MCS) operations. Furthermore, some respondents stated that a main problem that undermines the credibility of the control system itself is the lack of trust in the fishery industry data collection. According to respondents, in addition to enhancing data-collection systems and investing more in MCS resources (e.g., infrastructure, staff), the creation an official baseline on IUU fishing in Chile was proposed in order to quantify illegal fishing, identify hot spots, and rebuild catches for every fishery. This exercise would reduce uncertainty of stock assessment models and the associated management measures. Additionally, these measures would allow stakeholders to effectively integrate IUU fishing in management plans (MP), setting specific goals and deadlines to solve this problem and to implement strategies to tackle structural challenges and specific critical issues.

In addition to improving MCS systems, respondents highlighted the need to develop strategies that would discourage IUU fishing and generate a better understanding of its impact. They also advocated the creation of incentives to favor compliance with the regulations in force. To this end, more than only enforcing more inspection activities, respondents believe that information and education campaigns, as well as compliance incentives, needs to be implemented.

Respondents also highlighted the need to raise consumer awareness and encourage effective market participation to become more demanding and request products from accredited legal fisheries.

3.2 Development of a priority national research agenda to improve fisheries management in Chile

This second issue identified by respondents was mainly prioritized by government officials, academia and SMEs, who consider there is a need to set strategic priorities to be included in medium- and long-term planning, which in turn should inform the research needs of government institutions involved in the fishing sector. According to respondents, existing research focuses on the short-term and exclusively on the fishing activities, neglecting the social and economic perspective. There was also the perception that, in some cases, there is a disconnection between research and the decision-making processes by management. This is particularly relevant for fisheries with Management Plans that have to deal with delayed fishery statistics, making it difficult to take decisions with the updated available information.

Respondents believe that there are constraints concerning Human Resources, infrastructure and funding for research. Nevertheless, the absence of a strategic vision stands out as a key opportunity for improvement. The lack of prospective studies on species that are currently not being and that could potentially contribute to diversifying the sector's production, was exploited mentioned as an important gap resulting from this lack of strategic planning. A better understanding of environmental variability impacting fisheries are also gaps to be included in a future agenda.

To provide the national research agenda with a strategic vision, some respondents suggest creating a coordinating body responsible for identifying and systematizing research challenges, create and monitor plans and build strategic alliances with different agencies. This proposal is presented as a continuous and multidisciplinary process with a medium and long-term vision, aiming at the integration of principles of ecosystem management. It also incorporates the human component and seeks to strike a balance between ecosystem health and the well-being of fishing communities. In that regard, respondents believe that planning should be a participative process that involves research and government institutions, the industrial and artisanal sectors and relevant supply chain stakeholders. For the process to prove effective, some consider starting by identifying current limitations of resources available to the fisheries administration and in terms of fisheries knowledge. Another

5 The relevance and urgency of these two improvement areas cannot be compared with the relevance-urgency scoring results of the other topics due to the differences in the sampling size. The two newly identified improvement areas were scored only by the 27 respondents who identified them as a priority, in contrast to the 152 respondents who scored the initially identified 16 improvement areas.

FIGURE 4
The top four improvement priorities in fisheries policies identified per respondent's sector. The artisanal sector has five improvement priorities because the last two priorities were tied in terms of scoring points. The topics are provided in Table 2.

factor to be considered, is the adequate integration of local ecological knowledge and the empirical information provided in real time by the producing sector for research purposes.

3.3 Legitimacy of the fisheries law

According to artisanal sector stakeholders, NGOs, industry and government representatives, this issue required urgent action. An amendment to the fisheries legislation was approved in 2013, but despite the progress made in fisheries management (Ríos, 2015), the fisheries law was largely deemed illegitimate due to the corruption cases associated with its origin (Reyes et al., 2017).

Three different narratives were identified among the answers collected in 2020. Some respondents considered that the law was legitimate and appropriate from a technical standpoint. Others believed that the law has some positive aspects, but there is a need to change aspects linked to the assignment of rights. Finally, other respondents consider that, despite the positive aspects, the law should be repealed because it is illegitimate. Regardless the contrasting narratives, there was consensus regarding the urgent need to address this issue. Some even warned against the risks derived from delaying finding a timely solution, both for institutions and for users, given that the lack of legitimacy increases the risk of non-compliance.

Despite the polarizing climate, the 2020 survey shed light on shared concerns, not only regarding a call for action, but also regarding the proposed solutions. Respondents indicated that the solution was to create a democratic and participative space to identify strengths and weaknesses of the law, and to establish the most appropriate corrective actions in the development of a new legislative framework. This issue was highlighted by respondents as a challenge due to the lack of trust amongst sectors, specially between the artisanal and the industrial sectors, at the time the survey was carried out.

3.4 Development of a new national fisheries policy

This fourth issue was considered urgent by respondents from academia, government, the artisanal sector and NGOs. Respondents referred to the lack of an explicit fisheries policy and pointed out that the 2007 is out of date.

Respondents suggested that the current vision and goals of the fisheries policy is too narrow as it focuses mainly on the maximum exploitation of resources, neglecting other areas, such as conserving the ecosystem, adapting to climate change, creating greater added value, and maintaining the jobs and the economic well-being of the fishing communities. A new fisheries policy should therefore have clear goals on areas beyond resource exploitation (e.g., job creation, food security, export revenues) and actions to meet those.

Respondents refer to the need for the new fisheries policy proposal to build a common vision for the development of the sector in the long term by performing a holistic analysis of the Chilean fisheries sector. This analysis should result from the work of multidisciplinary groups and consultation processes among stakeholders. In addition, to avoid legitimacy problems, this process requires a high level of independence of the authorities.

3.5 Updating the Artisanal fisheries registry

This issue was considered urgent by the artisanal and industrial sector, government officials and NGOs. There seemed to be ample consensus around the fact that Chilean fisheries are facing great challenges because the RPA is outdated. This out-of-date registry has failed to limit fishing efforts creates perverse incentives by enabling IUU fishing.

The original purpose of the RPA was to adapt fleets size to biologically sustainable exploitation. Nevertheless, respondents state that this has not been accomplished, because the registration process is not dynamic enough: fishers who have passed away or who have retired remain on the list, while active fishers cannot access the registry. As a result, many young fishers end up fishing illegally or buying a place in the registry from retired fishers.

Stakeholders suggested the need to apply existing expiration rules for inactive fishing permits in the short term. In the medium and long terms, respondents recommend migrating to a more dynamic system that articulates RPA data with landing certificates and other databases. This new system would allow rapid identification of fishers who discontinue their fishing activity, enable a better control of artisanal fishing efforts and provide the dynamism that the activity requires in the assignment of artisanal fishing permits.

4 Actionable recommendations

This section summarizes the five main actionable policy recommendations in ranking order to improve fisheries governance and to address sustainability challenges in Chilean fisheries:

1. Improve budget allocation for MCS activities by SERNAPESCA and related government agencies. Prepare an official diagnosis on the impacts of IUU fishing in Chile to estimate its value in terms of volume and economic impact. This issue should also be included in the Management Plans of different fisheries to design a joint strategy, set goals and deadlines that will help reach specific milestones in the eradication of IUU fishing. Activities should also include information and education initiatives as well as the implementation of incentives for harvesters.
2. Develop a national research agenda to identify and systematize research gaps and needs through a continuous, multidisciplinary process with a medium- and long-term vision. This new research agenda should not only focus on the exploitation of target species, but it should also be aligned with ecosystem management principles and incorporate socioeconomic research seeking to maintain the balance of environmental health and human well-being.
3. Create a participative space where stakeholders can discuss the strengths and weaknesses of the legislation and agree on any corrective actions. Today's polarized scenario among sectors represents a barrier, so tackling the lack of legitimacy of the fisheries law is an urgent priority measure.
4. Update the fisheries policy to establish a common vision between sectors and stakeholders and promote the long-term development of the sector. This should be a binding political process.
5. Update the RPA in order to have a dynamic tool to grant fishing rights and adequate the effort to each fishery. The update of the RPA should involve connection with other relevant registries that control artisanal fisheries (e.g., sanctions, landings, logbooks).

5 Discussion

The 2020 survey to key informants involved in the fishery served to identify some of the priority policy areas that stakeholders from different sectors within the sector considered as priority. Yet, it also served to highlight the areas where more awareness is needed, such as social, gender and equity issues. The profiles of the informants, which for example did not include specific groups (e.g., indigenous peoples) as respondents, have probably impacted the focus on some relevant issues, such as the tenure rights of indigenous communities. The effect of this methodological gap has been probably exacerbated by the discrimination that Indigenous peoples suffer in Chile, not only when it comes to fisheries related issues (Hiriart-Bertrand et al., 2020), but in other areas of life (see e.g., Merino et al., 2009; de Cea et al., 2016).

Since the development of the survey in 2019-2020, a number of papers have reinforced the relevance of IUU fishing in Chile (Oyanedel, 2019), analyzing the specific motivations for non-compliance with specific rules (such as the quota) (Oyanedel et al., 2020), the essential role of traders in shaping illegality in market trends (Oyanedel et al., 2021) and the required multidimensional interventions that are required to resolve the wicked challenges of illegal trade in the Chilean artisanal hake fishery (Oyanedel et al., 2022). Building upon analysis of 20 Chilean fisheries, Donlan et al. (2020) estimated that illegal landings account for as much as 70% of the national landings and propose a methodology that, building upon enforcement officers' knowledge, enables effective institutionalization of illegal fishing estimates. All this recent evidence (see as well RPS Submitter et al., 2022) points that the issue of illegal fishing remains largely unresolved, and that the development of a new legal framework can contribute to address, at least partially. New policy developments in this field will largely benefit from the extensive advances in knowledge on the topic in Chile resulting from the increased interest by a number of researchers.

Since the completion of the online survey in 2020, some advances have been made in regards the update of the Artisanal Fisheries Registry. The Chilean government opened the RPA to crew members in some specific fisheries as jumbo flying squid, stone crab among other fisheries (SUBPESCA, 2020b). By June 2022 the Chilean government initiated an agenda with 20 actions to support the artisanal fishing sector; amongst them the issuance of a bill aimed at improving the system of deadlines to regularize the registration in the RPA in the event of the death of the vessel owner. Additionally, the government rolled out a system for replacing vacancies generated by fishers that have passed away or have retired in fisheries without risk of overexploitation or whose status allows it (Ministerio de Economía, Fomento y Turismo, 2022). Effectively addressing the shortages of an outdated RPA is a crucial issue for the governance of small-scale fisheries as it has direct ramifications for tenure rights and IUU fishing, but also to the entire co-management system because, as pointed out by Tapia-Jopia (2022), the lack of a complete RPA prevents equitable access to representation within the Management Committees. Álvarez Burgos (2020) points as well to the ramifications that an incomplete RPA have in terms of gender equity, as women working in the sector who are not directly involved in the fishing itself cannot register in the RPA, leaving them as informal workers, without social protection and with less tools to participate in productive development funds. Some improvements have been incorporated in August 2021 in order to promote gender equity in the fisheries and aquaculture sector (Biblioteca del Congreso Nacional de Chile, 2021). Amongst other issues, the policy ensures that women's representation in any participatory platform of the fishers' governance system (such as the Management Committees and the Technical Scientific Committees) is at a minimum of one third of the total elected members (with a maximum of two thirds). It is still soon to evaluate the implementation of these recent policy improvements.

Upon recent years, challenges have also been raised regarding communications flows, mainly downward and upward, between fishers and Management committees' representatives. Regarding downward communications, Reyes et al., 2017 recommends disposing an operating protocol for Management Committees that includes a communication and dissemination plan, since it is essential to inform the management measures included in the management plans to local users. Nevertheless, little is still known about upwards communications, namely the way fishers communicate their concerns to Management Committees representatives. In this sense, progress needs to be done in the design of fisheries management models that consider local information and traditional knowledge. This issue becomes more relevant when we analyze data-poor populations. According to Reyes et al. (2017), participation could be improved if Management Committees establishes periods for groups of people and/or institutions to present their concerns, suggestions, studies, background regarding to managements plans.

The current government has taken steps to address one of the main policy improvements identified in the 2020 survey, the lack of legitimacy of the fisheries law. The development of a new legal framework should focus on addressing the root causes that triggered the lack of legitimacy the previous legal framework (tenure rights for the artisanal sector), strike a balance with the positive elements of the old fisheries legal framework (as pointed out by the 2020 survey informants, mainly linked to co-management and the polycentric approach) and include the few but important policy improvements that were introduced since 2020. Achieving this balance will benefit from extensive consultation and stakeholder participation, as was pointed out by survey respondents.

Author contributions

RG-W and RV: conceptualization, methodology, data curation, data analysis, formal analysis, writing – original draft, writing – review and editing, visualization. GO, GA, LH-B and RL-C: writing – review and editing. EA-P: conceptualization, methodology, writing – original draft, writing – review and editing. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1073397/full#supplementary-material

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The offshore renewables industry may be better served by new bespoke design guidelines than by automatic adoption of recommended practices developed for oil and gas infrastructure: A recommendation illustrated by subsea cable design

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Introduction: There is an emerging need for the offshore renewable industry to have their own bespoke design guidelines because the associated projects and offshore facilities differ in fundamental ways to oil and gas facilities. Offshore renewable energy (ORE) facilities have already surpassed the numbers of installed facilities in the oil and gas industry by an order of magnitude and demand is forecast to continue growing exponentially. In addition ORE facilities often have different response characteristics and limit states or failure modes as well as profoundly different risk and consequence profiles given they are generally uncrewed and do not contain explosive hydrocarbon fluids which might be released into the environment. Therefore, the purpose of this paper is to advocate for licensing bodies and regulators (such as the various national PEL 114 committees) to challenge the process of automatic adoption of oil and gas design processes, while pushing for offshore renewables to be treated differently, when appropriate, with more relevant and applicable guidance.

Methods: To support this argument we present new bespoke design guidance developed for subsea cables based on specific modes of cable behaviour, which often differ from pipelines. We also show worked examples from recent project experience. The results from on-bottom stability analyses of a set of cables are compared between conventional oil and gas guidance following DNV-RP-F109 versus the stability using cable-optimised approaches.

Results: The outcomes from the 'conventional' oil and gas results are not simply biased compared to cable-optimised design methods, with a trend of being either conservative or unconservative. Instead, the results of the two methods are very poorly correlated. This shows that the oil and gas approach isn't simply biased when applied to cables, but is instead unreliable because it doesn't capture the underlying failure conditions. These analytical comparisons are supported by field observation - the ocean doesn't lie, and makes short work of any anthropogenic structures which are designed with inadequate appreciation of the real world conditions.

Discussion: To support the rapid growth of ORE, we should therefore actively pursue opportunities to rewrite the design rules and standards, so that they better support the specific requirements of ORE infrastructure, rather than legacy oil and gas structures. With more appropriate design practices, we can accelerate the roll out of ORE to meet net zero, and mitigate the climate crisis.

KEYWORDS

design guideline, recommended practice, offshore renewable energy, subsea cables, on bottom stability

1 Introduction

To alleviate future climate change humanity must reduce the reliance on fossil fuels for energy production by adopting renewable energy sources, primarily wind and solar (IPCC, 2021; UNFCCC, 2021).

To achieve this aim, the offshore renewables industry must grow exponentially. Current government targets of installed offshore wind capacity are approaching the value of 380 GW by 2030 that was proposed in the 2021 UN Energy Compact by the International Renewable Energy Agency (IRENA) and Global Wind Energy Council (GWEC) (GWEC, 2022). To meet this aim, tens of thousands of new structures and tens of thousands of kilometers of subsea power cable must be installed in the next decade. That growth rate needs to continue to 2050, when humanity must achieve net zero carbon emissions if global warming is to be limited to 1.5°C – which is a goal that most nations have now committed to.

The offshore industry is heavily regulated, partly due to its origin in oil and gas development, with the associated human and environmental risk. The design of offshore infrastructure is therefore tightly controlled through standard documents (or recommended practices). The adherence to standards has many benefits, but these documents evolve slowly, with revisions typically only approved twice per decade. In contrast, the climate crisis requires urgent rapid action.

1.1 Paper structure

The purpose of this paper is to highlight that bespoke new design guidelines may be more appropriate for the emergent but rapidly-growing offshore renewables industry, rather than adopting legacy practices from the offshore oil and gas industry. The paper is structured as followed:

- Section 1 sets out the industry context, and introduces the engineering challenge of cable stability design. We discuss how cable stability could be tackled by borrowing approaches from oil and gas pipeline stability design, but we highlight the flaws in this approach.
- Section 2 discusses the background to standards, recommended practices and engineering reliability. We show how the underlying mechanisms of failure and limit states differ between cables and pipelines.
- Section 3 and Section 4 introduce bespoke approaches for cable stability design, for rocky and sandy seabeds respectively. These methods have a different basis to the conventional approaches inherited from oil and gas experience. Practical case studies are used to illustrate their performance.
- Section 5 closes the paper with conclusions.

1.2 Industry context: offshore oil and gas vs. offshore renewables

Following the establishment and growth of the offshore oil and gas industry in the mid 1960's, major research centered on the North Sea has been undertaken through until the 1990's to develop and refine the models of behavior for subsea oil and gas pipelines used to transport and export production to shore for further processing. These design methods have been codified into recommended design practices, with the family of guidance published by DNV having become globally ubiquitous, despite their lack of substantial evolution or refinement over the last two decades. The widespread adoption of these design methods has resulted in remarkably few catastrophic oil or gas pipeline failures over this time.

In 1991, the world's first offshore wind farm (OWF) was established off the coast of Vindeby, Denmark. Since that time, the offshore wind industry has grown rapidly and now contributes a significant fraction of the total electrical power supply in some locations. During 2021, the total global installed capacity is reported to have reached 57 GW (GWEC, 2022) – hence despite exponential growth, the offshore wind industry lags roughly 3 decades behind the oil and gas industry in evolutionary terms.

For comparison purposes, it is reported that there presently exist around 184 offshore oil rigs in the North Sea (Statista, 2023). In contrast, it is reported that there are presently approximately 4000 offshore wind turbines in the same area (Crown Estate, 2022). This means that the offshore wind industry has already built over an order of magnitude more ocean-founded structures than the oil and gas industry that is twice its age. These offshore wind structures are almost universally uncrewed, whereas the majority of the oil and gas structures are crewed. The future prospects for the oil and gas industry are for very few new platforms to be installed, whereas in stark contrast the 2030 targets for installed offshore wind capacity are 65 GW for the European Union (EU) bordering the North Sea (through the Esbjerg Declaration, 2022) and 50 GW for the United Kingdom (UK) (HMG, 2022). These targets represent increases of around 49 GW (EU) and 39 GW (UK) and correspond to a combined increase of around 5,000 to 8,000 turbines in the next 8 years depending on how quickly these turbines increase in unit power. This represents exponential growth on the present offshore wind installed capacity, meaning that 'business as usual' design and engineering practices should be subject to review and challenge for their suitability going forwards.

1.3 Prevalence and industry drivers

The vast majority of offshore wind developments have, to date, been located in shallow coastal areas on soft sediments including sand and clay – resulting in the widespread adoption of trenching and burial of the inter-array and export power cables to negate the risks of instability due to hydrodynamic loading and third-party mechanical damage. It has therefore been expedient for the marine renewables industry to adopt the subsea pipeline design practices from the oil and gas industry for application to array and export cables, and use them to model the on-bottom stability and allowable spanning of the cables. Despite the high reliability of subsea pipelines in the oil and gas industry, the integrity of offshore renewable energy cables has been found to be much less reliable - over 80% of insurance claims by the offshore wind industry have been attributed to cable failures (Boehme and Robson, 2012; Jee, 2016). This is despite the integrity of cables being critical to the financial performance of these projects. The suitability and applicability of the existing body of oil and gas design guidance for application to cables is therefore worthy of review.

Over the last few years, the offshore renewables industry has begun expanding into new areas, including:

1. Shallow coastal windfarms located in areas prone to the rapid onset of severe cyclonic storm conditions during cable lay operations – leading to challenging on-bottom stability conditions during the installation phase prior to cable burial.
2. Shallow coastal windfarms in areas prone to severe metocean conditions (such as off the Atlantic coast of Europe) where persistent breaking waves sweep the seabed clear of sediment, resulting in the cables needing to remain exposed on the seabed during their operational lifetime.
3. Floating wind farms (for example off the west coast of Norway), where seabed conditions typically comprise exposed bedrock.
4. The further development of wave and tidal stream energy where the presence of strong and persistent tidal currents and/or waves also leads to seabeds featuring exposed bedrock and power export cables subject to implausibly high hydrodynamic loads.
5. The construction of major subsea High Voltage Direct Current (HVDC) interconnector cables to join different electricity networks, for example between Norway and Germany, to enable balancing of hydro and wind power production with variation in consumption demand in each network.

The on-bottom stability design of subsea pipes is important to ensure safety and reliability but can be challenging to achieve, particularly for renewable energy projects which are preferentially located in high energy metocean environments. Often, these conditions lead to the seabed being stripped of all loose sediment, leaving the cables to rest on exposed bedrock, boulders or cobbles where roughness features can be similar in size to the cables. As novel offshore renewable energy projects such as tidal stream energy, floating wind and wave energy devices increasingly evolve from concept demonstration to commercial-scale developments, new approaches are needed to capture the relevant physics for small diameter cables on rocky seabeds to reduce the costs and risks of power transmission and increase operational reliability. Similarly, where shallow water depths and unpredictable severe storms can occur during the cable installation phase, novel design approaches that capture more of the true tripartite interaction between cables, seabeds and fluid forcing have the potential to unlock significant improvements in reliability and reductions in costs for the marine renewable energy industry. In reality, the power cables are agnostic to whatever is attached at each end from the perspective of seabed/ fluid interaction.

1.4 On-bottom stability: an exemplar of knowledge transfer between oil and gas and renewables

Subsea pipeline on-bottom stability is adopted herein as a convenient and relevant design aspect to study the evolution and refinement of the design approaches by the oil and gas industry, followed by the widespread adoption of these same approaches in the offshore renewables industry.

On-bottom stability design aims to ensure that pipelines do not move excessively on the seabed under loading actions from waves and currents. Design guidance for subsea pipeline on-bottom stability has evolved over approximately 5 decades from the publication of:

1. DNV '76 Rules for Submarine Pipeline Systems (DNV, 1976), where the design approach adopted absolute stability as a force-balance between stabilizing friction and destabilising hydrodynamic forces.
2. DNV '81 Rules for Submarine Pipeline Systems (DNV, 1981) where significant refinement in the hydrodynamic force model was introduced following extensive industry research;
3. DNV-RP-E305 (DNV, 1988) where enhanced models of lateral resistance and dynamic stability methods were introduced, together with calibrated methods for capturing typical results from many dynamic simulations.
4. DNV-RP-F109 (DNV, 2008; DNV, 2021a). First issued in 2008 then reissued in 2011, 2017, 2019 and 2021 each revision has introduced minor incremental edits and adjustments to the above design approaches. For simplicity hereon this recommended practice is referred to as 'F109'.

At the Offshore Marine and Arctic Engineering (OMAE) 2008 conference in Estoril Portugal, Zeitoun et al. (2008) summarised the 'state of the art' in key aspects of pipeline on-bottom stability design processes, including the above historical perspectives. Zeitoun et al. (2008) discuss the advantages and shortfalls of the different design approaches in order to aid the reader's understanding.

Since that time, a decade of research and further methodology refinement has extended the boundaries of the industry's knowledge and understanding of the behaviour of subsea pipes, including geotechnics, hydrodynamics, oceanography and structural response modelling. Particular progress has been made in:

1. The response of pipelines to sediment transport and scour.
2. Understanding the behaviour of small diameter pipelines and cables within wave and current boundary layers.
3. The behaviour of cables on rocky seabeds in high energy marine environments.

Despite this extensive body of research findings, negligible change or enhancement to either of the prevailing design approaches in widespread use around the world: F109 (DNV, 2021a); and the American Gas Association (AGA) pipe stability software tool developed by Pipeline Research Council International (PRCI). AGA (2002) has been made to incorporate these improvements.

Since the publication of the Zeitoun et al. (2008) overview of the then-state-of-the-art in pipeline on-bottom stability design, a number of major research efforts have been undertaken, some of which are still works-in-progress. There has also been the design, construction and initial operation of a number of significant subsea pipelines offshore Australia and elsewhere. The learnings from undertaking the design for these projects has filtered into the public domain *via* a number of academic and industry conferences and publications. Together these include:

1. The University of Western Australia (UWA) O-tube project as a cornerstone of the STABLEpipe Joint Industry Project (JIP), including the Australian Research Council-supported On-Bottom Stability of Large Diameter Submarine Pipelines Linkage Project LP0989936 (Cheng et al., 2009) and Hydrodynamic Forces on Small Diameter Pipelines Linkage Project LP150100249 (Cheng et al., 2015).
2. The DNVGL-led StabUmCa and PILS JIPs (Vedeld et al., 2018), which had claimed to focus on cable stability.
3. Wood Plc-led ongoing methodology development and research including a number of sponsored UWA CEED projects (for example Shen et al., 2013), as well as the Cability JIP led by the Paris office, which also aimed to specifically focus on cable stability.

An updated summary of the research contributions made over the last decade in this field has been provided by Griffiths et al. (2018a). These works point to a broad body of expertise and industry understanding gained from the use of existing recommended practices in design, such as the commonly-used F109 (DNV, 2021a), and the less-well-used but still-relevant AGA/PRCI design methods (AGA, 2002). Each of these practices have a 'family' of antecedent incarnations which vary imperceptibly from one to the next, with the overarching design architecture having remained largely unchanged for decades. Where pipe (or cable) on-bottom stability is not excessively onerous and where conventional metocean, geotechnical and pipe properties are relevant, these families of design approaches are characteristically employed within the offshore industry and considered to be broadly conservative and utilitarian within their limiting bounds of validity.

Each of these families of design approaches adopts one of three distinct methods:

1.4.1 Absolute stability method (F109 Section 4.5)

The absolute stability design method evaluates the stability of the pipe by considering its submerged weight and diameter, the environmental forces acting on the pipe, and the resistance acting on the pipe from the seabed soil as a balance of loads divided by resistances, adjusted by a safety factor in accordance with:

$$\text{Utilisation} = \gamma_{sc}\,(F_y^* + \mu\,.\,F_z^*)/(\mu\,.\,W_s + F_R\,) \leq 1.0 \qquad 1)$$

where γ_{sc} is the required safety factor based on safety class and geographic location, F_y^* and F_z^* are the horizontal and vertical forces associated with the single largest design wave plus current, after factoring to allow for embedment or trench shielding, μ is the Coulomb friction factor, w_s is the pipeline or cable submerged weight per unit length and F_R is the passive soil resistance for sand and clay soils.

The approach is described as a ‘single design wave’ method which looks to determine the largest anticipated wave (H^*) and its associated period (T^*). In conjunction with the relevant design near-bed current the near-bed velocity U^* is found in order to calculate the maximum hydrodynamic forces experienced by the pipe, which it is required to resist without movement based on the available lateral resistance from the soil, which is calculated from active and passive friction accounting for any embedment.

The above limit state criteria only makes logical sense when the pipe can be treated as being prismatically uniform along its longitudinal axis, leading the stability problem to degenerate to a two-dimensional behavioural model. In practice no pipes are ever prismatically uniform, however that is profoundly so for cables placed onto rocky seabeds where the vast majority of the cable is suspended above the seabed with only occasional localized points of contact occurring. In the case of the MeyGen cables Griffiths et al. (2018b) found that less than 1% of the cable length was in contact with the seabed. Under these conditions, the limit state proposed by DNV in Eq (1 above) only makes logical sense as a length-averaged condition, recognising that a natural consequence of this longitudinal averaging is that some intermediate pipe movement may occur as a result, for example at each spanning section between touchdown points.

1.4.2 Calibrated stability methods

The AGA Level 2 and DNV Generalised Lateral Stability methods calculate the required submerged weight for the given environmental conditions against a set of calibration coefficients that have been determined from the performance of large numbers of dynamic stability analyses using ‘sand’ and ‘clay’ seabed types. The coefficients have been calibrated to result in no more than the target level of lateral displacement (for DNV, equivalent of 10 D lateral movement, or 0.5 D lateral movement dependent on the criteria selected, where D is the external diameter of the pipe or cable). The validity of analysis performed to this design method is dependent on the validity of the underlying assumptions implicit within the proscriptive method – pipe surface coating, presence of marine growth and soil properties are either absent or profoundly simplified.

1.4.3 Dynamic stability

Seabed stability analysis may be carried out in accordance with AGA Level 3 or F109 dynamic stability method. Time-domain solvers have been developed by DNV through the PILS JIP (Vedeld et al., 2018) and by industry (Zeitoun et al., 2009; Youssef et al., 2011; Abdolmaleki and Gregory, 2018). These predict the 1D (lateral), 2D (lateral and vertical) and 3D (lateral, vertical and longitudinal) solutions for pipe displacement as a function of time resulting from a simulated near-bed velocity storm time-series. The methods incorporate corrections to hydrodynamic forces and lateral resistance for pipes partially embedded in ‘sand’ or ‘clay’ seabed types as described in Sections 6.4 and 7 of F109. The objective of a dynamic lateral stability analysis is to calculate the lateral displacement of a pipe subjected to hydrodynamic loads from a given combination of waves and current during a design sea state.

Displacements are extracted for a number of random seeds from the analysis, with reported displacement being equal to the mean value plus one standard deviation, as specified in F109. No user guidance is offered by DNV for 3 dimensional simulations on whether the mean plus one standard deviation on displacement should consider the mean and standard deviation of the displacement along the model pipe, as well as the mean and standard deviation between the 7+ simulations – this issue has been explored by Robertson et al. (2015).

In terms of work specifically focussed on small diameter pipes, relatively little has been published from the DNV-led StabUmCa and PILS JIPs. Vedeld et al. (2018) provides some insight into both of these research programs, which were intended to provide new design guidance to reduce unnecessary conservatism for smaller diameter pipes, however the resulting research outcomes are limited to consolidation of a small quantum of the existing and decades-old published body of knowledge of pipe on-bottom stability design. No new experimental or other research has been produced through these costly programs, which to-date have not been reflected in the incorporation of new and updated design guidance in F109 – albeit the most recent (2021) revision to the recommended practice claims without substantiation that the guidance is relevant to umbilicals and cables.

1.5 Stability and spanning of small diameter cables and umbilicals: fundamental differences compared to pipelines

Subsea power cables and umbilicals differ to typical oil and gas pipelines in a number of important aspects as follows:

1. A cable is smaller than a pipeline. This means that for a given flow condition, the cable is located more deeply into the miasma of the near-bed boundary layer, resulting in the ratio of wave loading often increasing relative to steady current loading. Being much smaller than typical oil and gas pipelines means the effects of wave boundary layers are far more pronounced and should be accounted for in design.
2. This smaller diameter also means that often cables and umbilicals experience design wave conditions which exceed the tested range of Keulegan-Carpenter number values which inform the underlying hydrodynamic model embedded in F109, hence leading to an uncertainty in the validity of the limiting hydrodynamic force coefficients.
3. In general, the average specific gravity (SG) of a cable is much higher than for a hydrocarbon pipeline (typically 50-100%). Despite this being typical, because the submerged weight varies with D^2 and the hydrodynamic forces vary with D, it is possible to show (using conventional design methods) that a solid gold bar will be deemed unstable at a certain small diameters, as illustrated in Figure 1.

4. On rocky seabeds the pipe or cable is not able to become embedded. So it rests on the seabed with frequent meso-scale spans between points of contact as shown in Figure 2. The vast majority of the cable is therefore in span with only point contacts supporting the cable. This behaviour is relevant to offshore renewables because their cables are often situated in high energy metocean conditions (tidal or wind-driven seas) and their compliance considering their lower stiffness (axial and bending) is an advantage to be considered and enjoyed. A conventional steel-pipe on flat-rock model overlooks the above.
5. The structural response of a pipeline is dominated by the steel element, with the internal and external coatings having minimal influence. In contrast, a cable is a composite structure with many different material layers, including steel strips or wire in a woven form, rather than solid tubing. As a result, the structural properties of cables differ significantly from pipelines, with cables having lower bending and axial stiffness and much higher hysteretic structural damping. This damping is due to the friction properties between the internal layers and elements, which control the axial stick-slip sliding between cable elements (conductors, armour wires) when the cable bends. This has significant influence on the relative risks of Vortex Induced Vibration (VIV) induced fatigue failure, since high levels of internal damping are known to suppress the susceptibility to in-line VIV as well as reduce the amplitude of vibration and therefore fatigue damage for cross-flow VIV (DNV, 2021d).

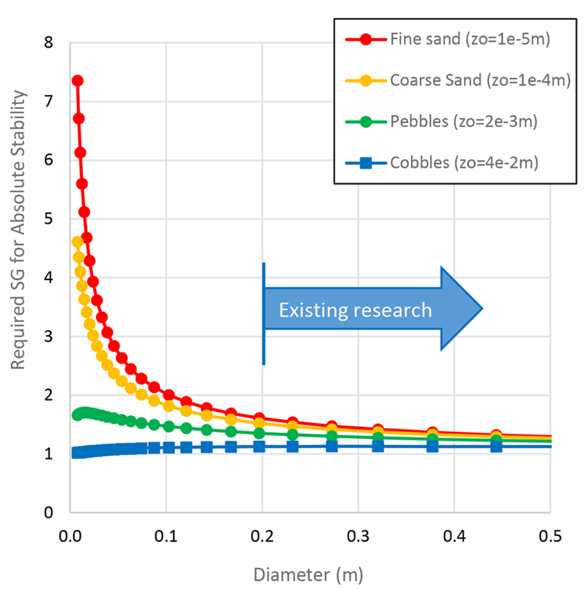

FIGURE 1
Inverse relationship between required SG and D versus seabed roughness under steady current conditions following F109 (DNV, 2021a) Equation 3.1. The plot shows the general trend that the required SG for stability tends to infinity as diameter gets small, with the existing design guidance for on-bottom stability based on hydrodynamic research using pipes greater than 0.2 m in diameter.

FIGURE 2
Example images of power cables on rocky seabeds (Images courtesy Simec Atlantis, Griffiths et al., 2018b) showing the subsea power export cables from the MeyGen tidal stream energy project. Key features to be observed are the size/scale of the rocky boulders/outceops compared to the diameter of the cable and the resulting wedging of the cable into crevices at the points of contact between cable and seabed, between which the cable is in span above the seabed.

6. Where it is typical for a subsea pipeline to add a 50 mm increase in diameter as an allowance for marine growth, the same allowance on a subsea power cable results in a profound increase in the challenges of demonstrating stability using conventional methods. On a 1000 mm diameter pipeline, 50 mm adds 5% to the hydrodynamic forces – compared to 50% on a 100 mm diameter cable. Subsea cables are therefore very much more sensitive to the presence of marine growth, and yet there is very little published research relating to the hydrodynamics of marine growth on horizontal near-bed pipes or cables. It should be noted that in widespread surveys of on-bottom subsea cables, there is no basis to support such a large allowance for marine growth.
7. In general, the minimum allowable curvature of a cable or umbilical is around 2 m, which expressed in terms of the ratio of bend radius to diameter is orders of magnitude smaller than a rigid steel-walled hydrocarbon pipeline. This has implications for cables to vertically conform to the seabed profile far more than a steel pipeline, especially in conjunction with their typically higher SG.
8. In terms of lateral response, a subsea cable has far lower bending stiffness than a typical rigid steel pipeline, hence the lateral response of the cable transitions to being governed by the axial tension and axial stiffness far sooner than for a rigid pipeline, where the bending stiffness dominates for longer. This effect has been investigated and useful insights are available from the

work of Robertson et al. (2015) and is very relevant to both stability and spanning. The study by Robertson set out to investigate the influence of pipeline bending and axial stiffness on the predicted displacement over time of a 3D dynamic on-bottom stability model. The results of this study showed the somewhat unexpected outcome that the stability of a flexible pipe, umbilical or subsea cable is lower than that of a similar rigid steel pipeline of the same diameter and submerged weight – with the dominant parameters influencing the response being the axial stiffness and the crest width of waves hitting the pipeline synchronously. Other interesting findings from this study were that the waves producing the greatest displacement to a pipeline in a random seastate are those with large nearbed velocity with the widest crest width along the axis of the pipeline, rather than the single largest wave in the seastate, which tended to have a very short crest width along the axis of the pipeline. The conclusions are that on-bottom stability is intrinsically three-dimensional and that it is inadequate to treat a cable as just a 'small pipeline' with respect to on-bottom stability.

9. Finally, oil and gas pipelines also have significant loading actions from the effect of internal pressure, which can contribute to buckling, leading to lateral and axial movement of the pipeline. During this movement, which can be deliberately engineered, the pipeline stresses must remain within limit states. Cables are not subject to internal pressure, and so these types of behaviours and the corresponding limit states are not applicable, and therefore nor are the corresponding design procedures, which are a key focus on pipeline design guidance.

In each of the above scenarios, the conventional published design methods (typically DNV) yield results which are extremely onerous for cable on-bottom stability and allowable spanning.

For completeness, the requirements of a number of alternative design standards and recommended practices have been reviewed for their guidance on what designers should do to address cable on-bottom stability. In summary:

- DNV-ST-O359 (DNV, 2021b) Subsea power cables for wind power plants defines on-bottom stability as the ability of a subsea power cable to remain in position under lateral displacement forces due to the action of hydrodynamic loads. This design standard requires the on-bottom stability to be addressed as part of detailed design if applicable (Clause 2.3.2) as well as protection against movement during installation between laying and subsequent protection (Clause 2.3.4). No guidance is offered on how the designer should achieve this requirement, and no reference is provided to F109.
- DNV-RP-O360 (DNV, 2021c) Subsea power cables in shallow water contains the same definition of stability above but further clarifies in Clause 3.3.6 that currents may affect the stability of cables lying unprotected on the seabed. The recommended practice also states that where the cables are unburied, the on-bottom stability of appurtenances including tubular products (e.g. ductile iron shells), mattresses and bags, as well as rock placement. No guidance is offered to the designer on how this stability is to be achieved, other than for the stability of rock berms.
- ISO 13628-5 (ISO, 2021) Petroleum and natural gas industries – Design and operation of subsea production systems Part 5: Subsea umbilicals advises that as part of "load effects analysis" the displacement due to on-bottom stability from functional and environmental loads may be required. The standard states that "*DNV RP-F109 is an example of a standard suitable for assessing the lateral stability of umbilicals exposed to current and wave loading.*"

The context is therefore noted that whilst subsea cables are required to have adequate on-bottom stability by a number of leading design standards, none of the standard industry design codes for seabed cables or umbilicals mandate the use of the F109, and for subsea power cables no guidance is offered on how to design the cables to be stable.

2 Safety philosophy and reliability

2.1 Philosophy

The potential failure of cables and umbilicals due to on-bottom instability has minimal environmental, health or safety impact. There are consequences such as a loss of power transmission or in the case of an umbilical the triggering of an automatic well shut-in. However, the failure of a hydrocarbon pipeline has far more dramatic and significant consequences, which can have major human and environmental impact, as illustrated in Figure 3.

The uncertainty analysis undertaken by DNV that underpins the factor of safety presented in Section 4.5.3 of F109 is intended to address the risks of hydrocarbon pipeline failure leading to loss of containment, rather than umbilicals or cables. F109 recommends for umbilicals and cables that the factor of safety be agreed on a project-by-project basis. The major consequences of umbilical and cable failure are therefore anticipated to be financial, including repair of the damage, any remedial stabilization, and the consequential loss of production and associated non-supply commercial costs.

The over-arching context therefore leads to a fundamental question –

Should humanity set out to build tens of thousands of new uncrewed unexplosive relatively simple and standardized offshore structures using practices which have largely been developed many decades ago to suit a few hundred bespoke-designed highly complex crewed but potentially highly explosive and environmentally-catastrophic structures?

Set in that context, the sensible answer appears to be *"Probably not!"*.

FIGURE 3
Health, safety and environmental consequences of hydrocarbon pipeline failure (ABC7, 2021). The video shows a sea-surface fire resulting from a hydrocarbon gas release from a subsea pipeline in close proximity to a platform, representing a potentially catastrophic safety risk to any personnel on the platform and potentially significant environmental consequences for the marine environment due to pollution.

2.2 Industry background on reliability modelling of on-bottom stability

The approach of adopting a reliability-based design methodology has been investigated and incorporated into the subsea pipeline industry through the SUPERB project (Sotberg et al., 1996) which was undertaken in the late 1990s. The then-new reliability-based design approach has been embedded across the spectrum of subsea pipeline design aspects and incorporated into all of the guidance documents, superseding the previous approach, which was based on the application of deterministic parameter values and a codified margin to allow for 'safety'.

However, in practice the application of reliability-based design to subsea pipeline on-bottom stability has been subject to much less widespread scrutiny or challenge. A summary of the identified peer-reviewed published literature on stability reliability design is presented in Table 1 (except for the DNV report regarding factors of safety included in F109 which remains unpublished). Of these works, only the stability design rationale articulated by Tornes et al. (2009) provides a direct link between the on-bottom stability response of a subsea pipeline, and limit states which constitute outcomes involving a loss of containment of the hydrocarbon contents of the pipeline. These are expressed through the DNV concepts of Fatigue Limit State (FLS), Ultimate Limit State (ULS) and Accidental Limit State (ALS) but which translate into rupture of the pipe wall resulting from fracture of the steel due to fatigue, excessive bending or local buckling of the pipe.

2.3 What are the 'real' limit states for subsea cables and umbilicals?

The new British Standard for on-bottom stability of subsea cables on rocky seabeds (BSI, 2023) gives the on-bottom stability limit states as:

1. Excessive bending or tension in the cable resulting in mechanical failure which exceeds the manufacturer's allowable envelope for operational or installation conditions, as relevant.
2. Fatigue failure of an element of the cable (e.g. armour wire, insulation or conductor core) due to excessive cyclic

TABLE 1 Summary of the present literature regarding reliability design approaches to subsea pipeline on-bottom stability.

Authors	Date	Title	Limit State Criteria
Brown, 1999	1999	A risk-reliability based approach to pipeline on-bottom stability design	Mixed including 2D quasi-stability limit
Wu and Riha, 2000	2000	Reliability analysis of on-bottom pipeline stability.	Absolute stability
Ewans, 2003	2003	A Response-Based Method for Developing Joint Metocean Criteria for On-Bottom Pipeline Stability	Allowable lateral displacement
Daghigh et al., 2008	2008	Applying the reliability analysis concept in on-bottom stability design of submarine pipelines	DNV-RP-E305 generalised method (but not certain)
Tornes et al., 2009	2009	A stability design rationale	ULS, FLS, ALS based on full 3D dynamic simulation
Gibson, 2011	2011	Metocean design criteria for pipeline on-bottom stability	Absolute stability
Elsayed et al., 2012	2012	Reliability of subsea pipelines against lateral instability.	Allowable 3D lateral displacement (with deterministic Von Mises stress checks)
Yang and Wang, 2013	2013	Dynamic stability analysis of pipeline based on reliability using surrogate model	Allowable lateral displacement
Youssef et al., 2013	2013	Application of statistical analysis techniques to pipeline on-bottom stability analysis. engineering 135.3 (2013).	Allowable 3D lateral displacement
Bai et al., 2015	2015	Reliability-based design of subsea light weight pipeline against lateral stability	Allowable lateral displacement
Li et al., 2017	2017	Quantitative risk assessment of submarine pipeline instability	Absolute stability

bending and/or tensile strains imposed on any point of the cable.

3. Excessive damage to outer layers of the cable incurred by relative movement against the seabed surface which may compromise the strength required for retrieval or in service integrity, expose components, and potentially changes cable behaviour in the affected section leading to excessive movement and subsequent mechanical or electrical failure.
4. Excessive local contact force.
5. Excessive impact force or repetitive impact damage.

It is therefore considered that the Net Present Value (NPV) of possible failure and repair costs should form the basis of the reliability and integrity philosophy. This approach must also account for the limit states and uncertainties intrinsic in the COREstab and STABLEpipe methods, which require careful consideration of the relevant real behaviour of cables on rocky and sandy seabeds, respectively. It is proposed that projects should adopt the above limit states as those which are used to determine the acceptance limits on the design to achieve the required levels of reliability driven by the NPV assessment of cable or umbilical failure.

An enormous variety of array and export cable layouts have been constructed, resembling the collective outcomes of many rainy-days of playing '*Pipopipette*' (also known as 'dots and boxes' or 'paddocks', Édouard, 1895) as shown in Figure 4 – however the detailed arrangement of each development is assumed to follow logical and optimised methods as described (for example) by Pillai et al. (2014) and Fischetti and Pisinger (2018). The outcome is that unequal volumes of power are anticipated to flow between each individual array cable connection, with the consequence that the individual risk and consequence of failure for each cable is not uniform. This represents a fundamental difference with hydrocarbon pipelines, where any loss of containment anywhere in the system guarantees front-page infamy for the operator concerned (see Figure 3).

2.4 10^{-6} and all that: What do failure probabilities *really* mean?

As a cautionary note, it is reminded that the essential requirement of a reliability-based design approach is not the exhaustive analysis of enormous numbers of numerical simulations of uncertain parameters against some form of potential failure limit state (as appears to be the case in many of the papers in Table 1).

Instead, as stated by Sotberg et al. (1996) "the performance of offshore pipelines is subjected to uncertainties in the physical quantities and models governing the structural behaviour. Application of reliability methods guided by engineering judgement and experience is thus a rational way to include the effect of these uncertainties in the final design assessment."

This quote has two critical elements relevant to this discussion. Firstly, it acknowledges that uncertainties exist in *models* as well as the input parameters; this uncertainty cannot be quantified by running a single model repeatedly with different inputs. Secondly, it recognises the application of judgement and experience, which is key to recognizing when conventional models may be inappropriate.

The design of cables on rocky seabeds introduces both of these critical elements, because it involves stepping outside of the bounds of collective knowledge of the offshore oil and gas industry and exploring accumulated knowledge from a broader context to find more appropriate models of behaviour. Such experience includes the published lessons learnt and observations of cable failures and damage such as the example shown in Figure 5 at the European Marine Energy Centre (EMEC) renewable energy site in the UK,

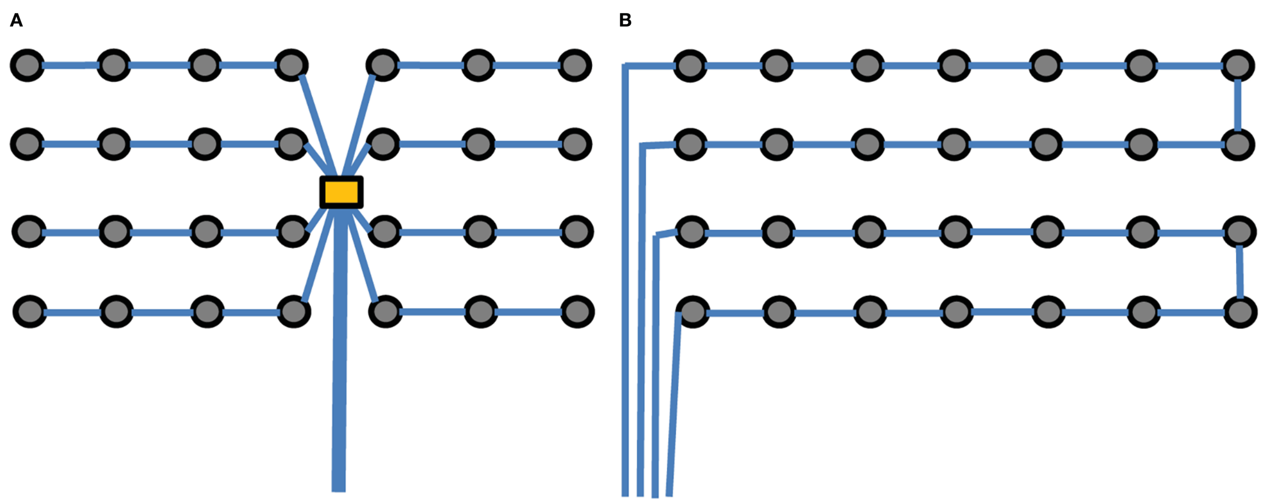

FIGURE 4
Example OWF cable layouts including **(A)** 'spider' and **(B)** 'loop' designs. The different cable layouts demonstrate very different levels of risk to any individual cable segment, with both arrangements having lower consequences of failure (in terms of lost power) for segments further from the export cables, whilst the export cables each have the highest levels of failure consequence. The 'spider' arrangement has higher risk than the 'loop' arrangement due to increased redundancy.

which is from the review published by The Crown Estate (2015). Other cable incidents have been collated by Conférence Internationale des Grands Réseaux Électriques (CIGRE).

Similarly, the late great Prof. Andrew Palmer (2012) expressed significant reservations about the 10^{-6} target failure probabilities and whether these were grounded in reality or were 'nominal'. DNV responded to these challenges (Agrell and Collberg, 2014) to explain that *"although these numbers provide a strong tool for evaluating the 'robustness' of a design, it is not straightforward to see how they relate to the probability that a given pipeline will fail. Moreover, as the definition of nominal probability in design codes is not very clear, it might mislead the end user to interpret this as an actual failure frequency. Such concerns are by no means confined to risk-based design of pipelines, but are also a continuing debate within risk analysis in general".*

Hence, in order for the on-bottom stability design of a given cable or umbilical to achieve the low level of failure risk which is warranted based on the (anticipated) high consequential cost of failure, the design approach needs to carefully navigate between 'nominal' and real probabilities of exceeding the (very real) limit states which govern the failure modes of subsea cables and umbilicals. To do this we must avoid blind direct adoption of 'nominal' failure targets from subsea hydrocarbon contexts which may be (or more likely are not) relevant.

3 State-of-the-art subsea cable stability design methods on rock: COREstab

3.1 Industry examples and context for rocky seabeds

A number of marine renewable energy projects around the world have been either proposed or actually developed where they have been located in areas where shallow water depths and strong currents and/or large and persistent wave action means the seabed has been swept clear of sediment. For example, a high number of installed wave and tidal facilities have required cable stabilisation measures (Sharkey, 2013) such as armour casings and concrete mattresses (at the EMEC site, off the Scottish coast), rock dumping (at the Wavehub site off the Cornish coast) and horizontal directional drilling (the Marine Current Turbines SeaGen project in Strangford Lough, Northern Ireland). Examples of cables installed over rocky seabeds can also be found at the tidal energy sites in the Bay of Fundy that feature medium to coarse gravel and cobbles (Stark et al., 2013), including potentially mobile gravel dunes. Rocky seabeds also occur on the Australian continental shelf along hydrocarbon pipeline and cable routes, including relatively flat limestone pavements and calcarenite caprock, as described by Sims et al. (2004) and Duncan and Gavrilov (2012).

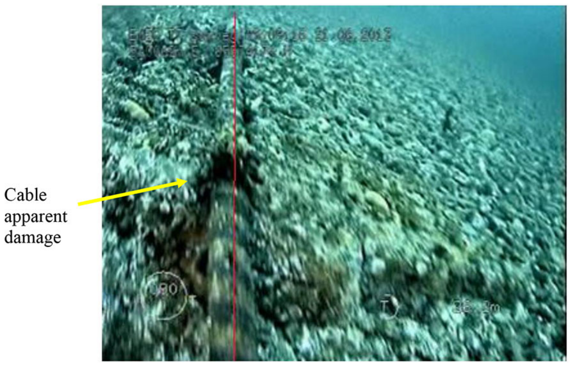

FIGURE 5
Cable damage example: reported case of abrasion due to strumming which may occur due to cable lateral sliding under wave and/or current loading (The Crown Estate, 2015).

The marginal commercial viability of renewable energy projects means that they are still trying to reduce costs while competing with other projects in less demanding locations. The prevailing design methodology (The Crown Estate, 2015) used to evaluate the on-bottom stability of pipelines and cables on rocky seabed for the marine renewable energy industry is F109 (DNV, 2021a), which was originally written for the offshore oil and gas industry for hydrocarbon-containing pipelines. This recommended practice features three different approaches to stability design, which compare the actions on the pipe/cable, including pipe weight, hydrodynamic loading and geotechnical seabed restraint. Conventional cable stabilisation designs and methods are simply too costly for these projects to be viable. Hence there is a strong appetite by these projects to identity where existing design methods can be radically re-engineered to capture additional relevant physics and better understand the real behaviour of cables under these conditions.

It has been recognised that on rocky seabeds the local profile of the seabed surface, at length scales comparable to the cable diameter, can have a very significant influence on the behaviour of subsea pipes and umbilicals, as demonstrated by the MeyGen cables shown in Figure 2. Where the rugosity of the seabed includes length scales of a similar order of size as the cable diameter, both the lateral resistance and hydrodynamic forces are dramatically altered, as documented by Griffiths et al. (2018b); Griffiths et al., (2018c). In order to correctly predict the behaviour of seabed cables, it is necessary to be able to model the meso-scale roughness elements which are often too small to be resolved by conventional MBES survey methods.

3.2 COREstab method development

The COREstab (Cables On Rock Enhanced stability) approach has been developed to address these considerations and has been described in Griffiths et al. (2018b); Griffiths et al. (2018c) and Griffiths (2022). The meso-scale approach consists of four steps:

1. Analyse the video records of the seabed survey to extract and statistically characterize the roughness elements present.
2. Use the measured statistical properties of the meso-scale seabed elements, randomly generate a synthetic blanket of

roughness elements as shown in Figure 6 to simulate the seabed profile, matching the size shape and orientation of the observed seabed roughness features with the synthetic roughness stochastic features also plotted in Figure 7 for comparison with the measured values. As this process represents a random representation of the seabed profile, following the guidance in F109 for Dynamic stability analysis, at least 7 random simulations are analysed.

3. Drape this synthetic blanket over the MBES macro-scale seabed bathymetry profile to produce a composite seabed at a scale which is small compared to the cable diameter.
4. Lay the cable down onto the composite seabed profiles. By laterally sliding the umbilical by a nominal distance of 10D each way, the lateral resistance of the cable can be calculated from the micro-scale interface friction coefficient and the methods documented by Griffiths et al. (2018c). This distance is chosen for two reasons. Firstly, because it is consistent with the maximum lateral displacement adopted in conventional dynamic pipeline stability analysis (e.g. DNV 2021), and secondly because it is sufficient distance relative to the seabed roughness wavelength for representative average values of the lateral resistance to be found. Based on the local gappiness and seabed profile, the hydrodynamic forces on the cable can also be calculated following the methods described in Griffiths et al. (2018b). The on-bottom stability factor of safety can then be found by applying the F109 Absolute stability calculation method, accounting for the increase in lateral resistance and reduction in hydrodynamic forces. Note that the adoption of 10D here is arbitrary in order to get a reasonable indication of the natural fixation points along and across the cable route.

3.3 New industry guidance

The COREstab design approach is presently being drafted into a new British Standards Institute guideline for the on-bottom stability of cables on rocky seabeds. This guideline is presently available for public review.

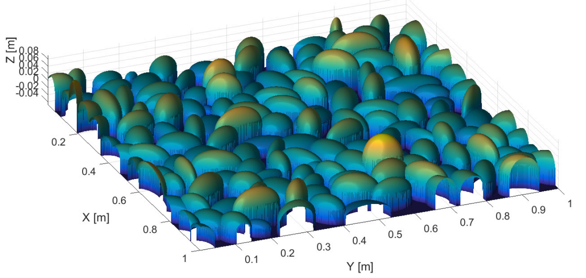

FIGURE 6
Example Tetris-packed synthetic rock blanket produced to represent the meso-scale roughness of the rocky seabed located within the MeyGen tidal stream energy project (Griffiths et al., 2018a). Colours represent shading/rendered illumination.

3.4 Project worked example 1: cables on rocky seabeds: meygen tidal stream energy

The new COREstab models and approaches to predicting the on-bottom stability of seabed cables have been used to back-analyse the stability of the subsea cables that MeyGen installed for Phase 1A of the Pentland Firth Inner Sound tidal stream energy project as published by Griffiths et al. (2018b).

These cables are located on rocky seabeds in an area where severe metocean conditions occur. The MeyGen Phase 1A project represents the first stage of the UK's commercial-scale tidal stream energy project. MeyGen has been awarded a Crown Estate lease for the option to develop a tidal stream project of up to 398 MW in the Inner Sound between Scotland's northernmost coast and the island of Stroma within the Pentland Firth. The initial phase of the project consists of four 1.5 MW horizontal axis turbines each with a dedicated power export cable supplied by JDR Cable Systems and routed approximately 2 km south to the Scottish mainland. The subsea cable installation and commissioning was undertaken in September 2015. Since the turbines were installed in 2017, total power production has now surpassed 37 GWh (Simec Atlantis, 2020).

The cables were analysed during the design phase using conventional F109 stability analysis methods and shown to be unstable, however the project decided on the balance of risks to install the cables without secondary stabilization. Since installation, repeated ROV field observation of these cables shows them to be stable on the seabed with little or no movement occurring over almost all of the cable routes, despite conventional engineering methods predicting significant dynamic movement.

The back-analysis by Griffiths et al. (2018b) was undertaken retrospectively, after several years' operation of the cables. This analysis shows that the loads and lateral resistance are modelled in an over-conservative way by conventional pipeline engineering techniques and was able to explain why the cables were actually stable, despite predictions to the contrary using F109. The COREstab design method involves developing a much more relevant model of the seabed features that are similar in size to the diameter of the cable. It was found that by capturing the meso-scale seabed roughness which resulted in over 99% of the cables being suspended above the seabed in a profusion of small spans such that:

1. Vertical hydrodynamic lift forces were reduced by over 90%.
2. Horizontal hydrodynamic forces were reduced by around 30%.
3. Due to the enhanced lateral resistance of cable interaction with meso-scale seabed roughness the lateral resistance to movement was increased on average by over 70%.

Overall, our analysis highlights that current on-bottom stability design methods can be unnecessarily conservative on rocky seabeds. The dramatic contrast is between the predictions by F109 that the required SG for stability was around 14 – between the density of solid lead (SG = 11.3) and solid gold (SG = 19.3). In contrast Griffiths et al. (2018d) showed that using the COREstab method the

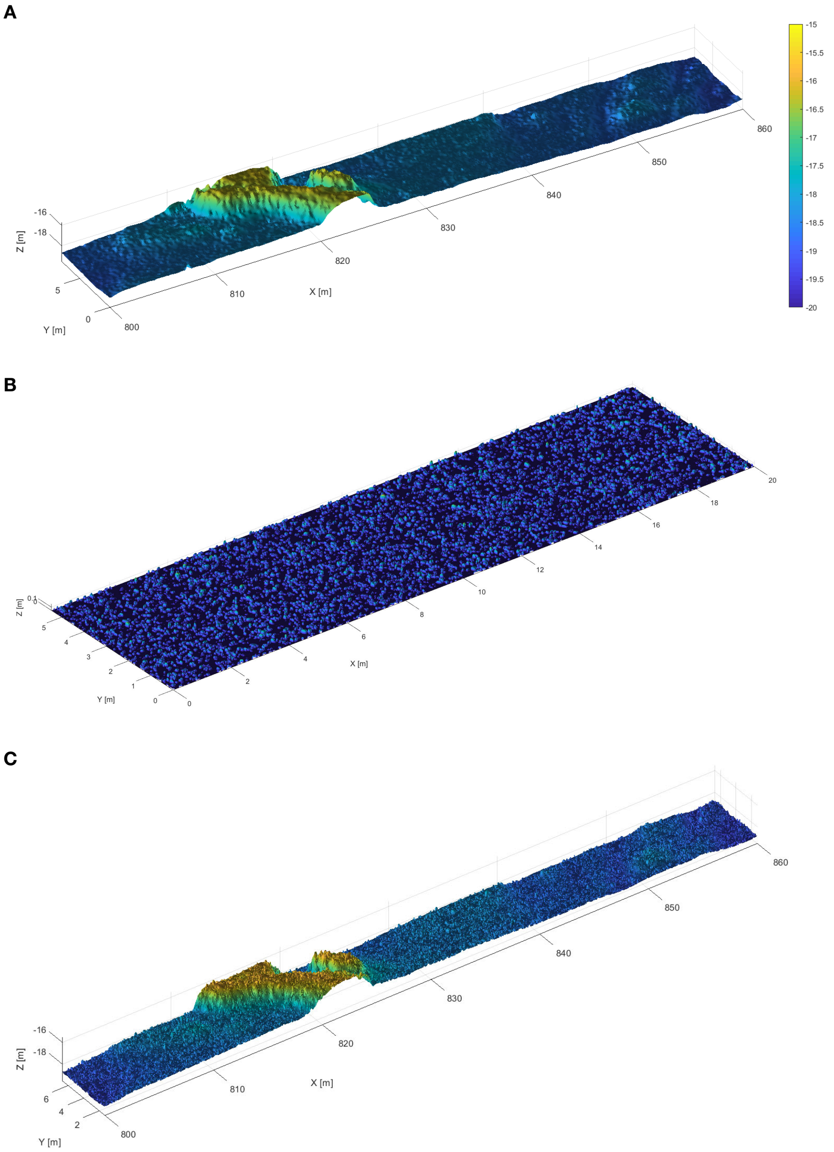

FIGURE 7
Illustration of **(A)** macro scale, **(B)** meso scale and **(C)** composite seabed sections.

cables (with actual SG = 3.34) were stable with a “factor of safety” of between 3.1 and 5.1. Whilst the COREstab method was only retrospectively applied to the MeyGen cables, it is understood that the project avoided over £1M in costs by deciding not to install secondary stabilization.

4 State-of-the-art subsea cable stability design methods on sand/silt: STABLEpipe

4.1 Stable pipelines on an unstable seabed

It was shown many decades ago by the late Prof. Palmer that sandy seabeds become mobile well before the on-bottom stability limit for subsea pipelines is reached (Palmer, 1996), leading to scour and sedimentation which profoundly alters the seabed profile and condition of the pipe. While this is acknowledged in F109 Section 8.5, F109 does not provide any useful design method guidance but instead refers to Griffiths et al. (2018d). This reference describes the extensive research program which has been completed through the STABLEpipe JIP, using the UWA recirculating O-tube (Cheng et al., 2014) as a transformational research tool in the development of a new design guideline which has now been co-developed with DNV using the design methods described by Draper et al. (2018a); Draper et al. (2018b). The STABLEpipe guideline has been used on a number of projects and remains the most thorough DNV-endorsed description on how to design pipelines on erodible seabeds

The fundamental change from conventional design is to recognize that there exists a tripartite interaction between the umbilical (or pipe), soil and the fluid loading which means each element influences the other, as illustrated in Figure 8.

4.2 STABLEpipe method development

In early 2008, Woodside initiated a research program with the University of Western Australia with an aim to establish an O-tube flume facility as shown in Figure 9 that is capable of modeling the tripartite pipe-soil-fluid interaction at approximately 1:1 scale for cables (Figure 8). This design of flume allows a model pipeline to be subjected to near-seabed flow conditions, such that wave-induced liquefaction and local scour may evolve naturally, concurrent with hydrodynamic loading of the pipeline and the mobilization of soil resistance. The O-tube project was also supported by a grant from the Australian Research Council (ARC) under the ARC Linkage Projects Program (2009) and the resulting facility is described in more detail by An et al. (2011).

It was expected that the insights from successful O-tube tests would allow the understanding of pipe-soil-fluid interaction to be updated and refined. When distilled into revised analysis procedures, these advances might produce CAPEX savings on new projects in the order of tens of millions of dollars per project by reducing the extent of secondary stabilization and/or the degree of primary stabilisation. Another motivation for Woodside and UWA initiating this project was to enable the ongoing stability of the existing 40-in North Rankine trunkline to be proven, so as to support the life extension of that facility, as reported by Jas et al. (2012).

Also in early 2008, JP Kenny (now Wood Plc.) initiated Phase 1 of the STABLEpipe JIP, looking at value definition. The project name comes from "STAbility of on-Bottom pipeLines under Extreme conditions Joint Industry Project". Phase 1 of the STABLEpipe JIP had the primary goal of improving industry understanding and engineering design practices in relation to offshore pipeline stabilisation in challenging environments.

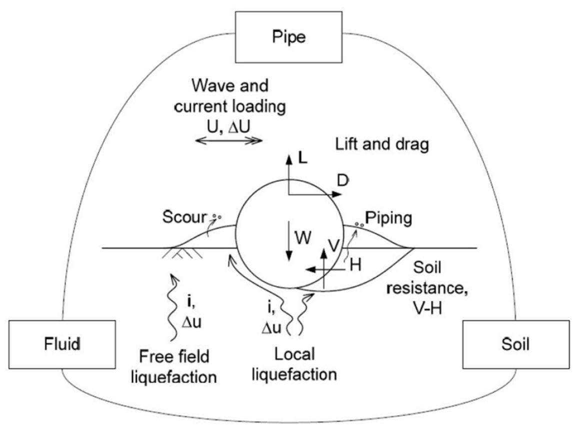

FIGURE 8
Tripartite interaction between umbilical, soil and fluid (Griffiths et al., 2018d). The diagram illustrates that the response of a pipe or cable on a sandy erodible seabed cannot be correctly understood without capturing each element of the interactions between pipe and fluid (hydrodynamic forces as well as flow amplifications around the pipe), pipe and soil (lateral and vertical resistance, as well as contributing to liquefaction) and soil and fluid (scour, erosion, sedimentation and liquefaction).

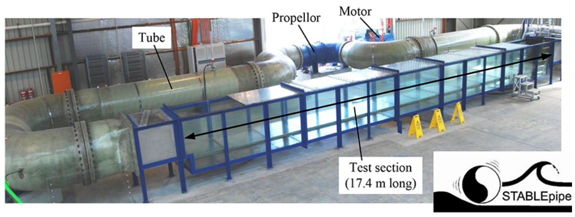

FIGURE 9
UWA O-tube flume and the STABLEpipe JIP logo (Griffiths et al., 2018d). The main test section is 17 m long with the fluid zone being 1 m wide by 1 m high, and the soil zone being around 0.4 m deep. The flow in the test section is rectilinear representing both wave and current motions over the bottom meter of the ocean. The facility is remarkable in that any sediment lost downstream out of the test section is transported around through the pump and returns to nourish the upstream mobile seabed. Steady currents up to 3 m/s and waves of 2.5 m/s velocity with a period of 15 s are feasible.

Among the outputs of Phase 1 were studies that identified the potential benefit from further definition of each aspect of on-bottom stability design. Based on these outputs, the participants and sponsors agreed to undertake a range of research programmes to tackle these critical knowledge gaps, including large scale testing (as had been initiated by Woodside and UWA), engineering studies and field monitoring if future funding permitted. At the end of Phase 1 of the JIP, Woodside proposed to lead Phase 2 of the JIP. By including the existing Woodside-UWA O-tube project in Phase 2 of the STABLEpipe JIP, with Chevron as a co-sponsor, additional scopes of work were possible, to the mutual benefit of all participants.

Extra leverage was created through parallel research funded by the ARC, the LRF and Shell, who supported academics and PhD students at UWA over the same period, working on related activities, with the outcomes feeding into STABLEpipe.

The aim of the STABLEpipe JIP was to assist the development of practical and locally-applicable stabilisation solutions that will provide operators with methodologies and cost-saving approaches to economically develop prospects in the NWS – of which there were many being pursued at that time. A key goal of the JIP participants was to produce a readily usable and clearly articulated design guideline: this was achieved, with the guideline being co-developed and published by DNVGL (2017). In this respect STABLEpipe was a mechanism to bring together operators, engineering organisations, industry experts and research professionals to deliver cost effective high integrity stabilisation solutions.

The outcomes of the STABLEpipe JIP together with the design methods have been described in the literature as follows:

1. A review of the broad industry research effort over the last decade (of which STABLEpipe JIP research is just one part) to improve our ability to model the on-bottom stability and behavior of subsea pipelines, as summarized by Griffiths et al. (2018d).
2. An understanding of the fundamental influence of the evolution of storms on the stability outcomes, by Draper et al. (2015).

3. An illustration of practical methods for modelling changes to submarine pipeline embedment and stability due to pipeline scour (Draper et al., 2018a).
4. An investigation of the influence of fine-grained soils and variable metocean conditions described by Draper et al. (2018b).
5. An investigation of the influence of shallow mobile sediment layers on the evolution of scour as described by Draper et al. (2014).

The predictions of the STABLEpipe method have been validated by back analysis against field observations of existing subsea pipelines, for which significant post-lay morphodynamic processes were observed to occur through routine integrity surveys over their lifetime:

1. The cable/pipe remaining at approximately the same elevation with respect to the far-field seabed, but experiencing significant local sedimentation as described by Leckie et al. (2016).
2. The cable/pipe experiencing significant lowering compared to the far-field seabed with a significant proportion of the pipe/cable remaining in span above the scoured trench and only small localised sections of pipe/cable touching the seabed as described by Leckie et al. (2015).

Each of these scenarios represent an improved outcome with respect to the on-bottom stability compared to the as-installed condition, as discussed by Leckie et al. (2018).

The key design and analysis steps are set out in detail in Draper et al. (2018a) and summarized in Figure 10 with the key elements being to:

1. Predict the initial embedment of the cable and establish the likely distribution of initial spans present as pre-existing spans which may form the initiation points for scour progression.
2. Model the evolution of metocean conditions as illustrated in Figure 11.
3. Predict the evolution of seabed morphodynamics around the cable, through the process of spans lengthening and deepening due to sediment transport, leading to either pipe sagging at the mid-span until touchdown occurs onto the bottom of the scour hole, or the shoulders of the span collapse again leading to increased pipe embedment compared to the far field.
4. Check the stability of the pipe through each timestep in this simulation.

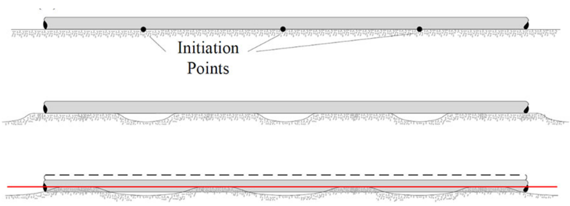

FIGURE 10
Scour initiation, span growth and umbilical sinking for 'close' initiation points (Draper et al., 2018a). The plot shows the evolution of seabed morphodynamics from the initiation of scour at points along the pipe followed by longitudinal and vertical deepening of the scour span through to bearing collapse of the span shoulders leading to enhanced far-field embedment of the pipe.

4.3 STABLEpipe relevance to cables

The STABLEpipe method was originally developed to aid in the stability design of subsea hydrocarbon pipelines – especially the large and relatively light gas export trunklines used to export gas from offshore production facilities to shore. However a number of aspects of seabed cables mean that the STABLEpipe design method is particularly effective and relevant, including:

1. Cables associated with OWF projects are frequently placed on shallow sand banks and in areas where the seabed is mobile.
2. These locations frequently feature ripples and megaripples which provide highly reliable initiation points for onset of scour and the orphodynamics processes which are modelled by the STABLEpipe method.
3. The volume of seabed soil requiring to be scoured for a cable is extremely small – with the horizontal scour rate equation featuring a 1/D term which accelerates the scour processes. When this is combined with the much smaller L_{cr} typical of cables, the STABLEpipe method works profoundly well to capture benefits to the on-bottom stability of cables.

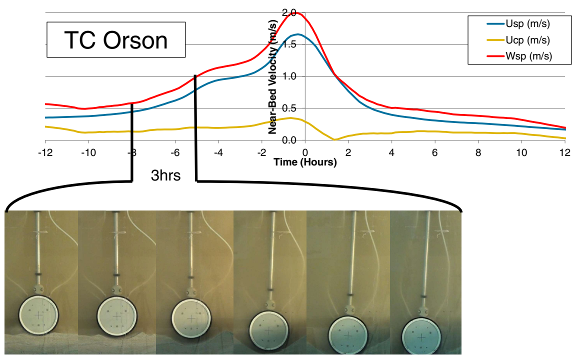

FIGURE 11
Storm evolution over time, showing active morphodynamics well before storm peak (Griffiths et al., 2018d). The major epoch of active scour and lowering of the pipe in this example storm occurs between 8 and 5 hours prior to the peak of the cyclone, resulting in the pipe lowering so very substantially into the seabed compared to its initial as-laid embedment. The conventional F109 design approach of evaluating the stability at the peak of the storm using the as-laid embedment is therefore profoundly irrelevant.

Given the severity of the metocean conditions found across many shallow-water OWF and other marine renewable energy project sites and that sections of the surficial seabed soil may be sandy, the likelihood is that enhanced on-bottom stability will be achieved with the application of design methods incorporating sediment transport and scour. While these methods extend beyond F109 Section 8.5, they have been applied to multiple projects, with Woodside providing feedback to shareholders on the savings they achieved on just the first project it was applied to (Woodside Energy, 2012).

A key question which arises in most laboratory testing is how model tests can be adequately scaled to prototype conditions to correctly account for the fact that many properties (e.g. hydrodynamic forces and sediment transport and scour rates) physically scale with contradictory relationships (Le Mehaute, 1976; Hughes, 1993). The interesting observation is that the majority of testing undertaken in UWA's Large O-tube for the STABLEpipe JIP used a model pipe which was 200 mm in diameter, as shown in Figure 12. This results in a model:prototype scale of approximately one (1:1) for many subsea power cables used in the offshore wind industry. It is therefore clear that the results of this testing are of direct relevance, without any scaling, to predicting the behavior of subsea cables on sandy erodible seabeds.

4.4 Project worked example 2: cables on soft sandy/silty seabeds

The STABLEpipe design method was applied to the on-bottom stability design of the array and export cables for an OWF located in Asian waters. The water depth varied from zero at the shore crossing to around 30 m in the field, with the stability analysis addressing the temporary condition where the cables were laid on the seabed prior to being trenched for lifetime protection and stabilization. The project site is in an area prone to experiencing a number of tropical revolving storms (Cyclones/Typhoons/Hurricanes) each year. In terms of project drivers, improvements in the predicted stability of the cables had the potential to increase the allowable time (and risk of storms occurring) between the cable lay and trenching operations.

The results of the on-bottom stability analysis considered the potential for beneficial increases in cable embedment during the build-up phases of possible storms, with the results of these assessments being compared against the predictions using just the conventional un-modified F109 design approaches. The results of this comparison in stability design methods is presented in Figure 13, showing a scatter-plot of the relative stability ranking of each cable segment. This plot clearly shows no correlation between the predictions of cable-focused stability design methods and the results of using F109. This lack of correlation flags very significant concern regarding the validity of using an un-modified generalized subsea pipeline design guideline on subsea cables – most especially given its almost ubiquitous utilization in the offshore renewables industry.

5 Conclusions

The proposition has been put forward that the offshore renewables industry should take great care in deciding whether or not to adopt existing oil and gas industry design recommended practices and guidelines, or whether to develop bespoke design guidance. New bespoke guidance can begin with fresh assumptions regarding (i) the consequences of failure, (ii) the failure modes, (iii) the target reliability and (iv) the engineering system behaviour relevant to ORE infrastructure.

This proposition has been explored by studying the applicability of existing industry guidance for the on-bottom stability of subsea cables. The study has considered the case of cables on rocky seabeds, and the contrasting case of cables on mobile sandy seabeds. These case studies have demonstrated profound differences between the design outcomes using conventional F109 design methods – evolved through oil and gas experience – compared to the more relevant and applicable response predictions when using the COREstab design method for cables on rock and STABLEpipe for cables on sand.

Considering the results of these case studies and the underlying physical differences between subsea cables and conventional oil and gas pipelines, it is concluded that for cable on-bottom stability design bespoke design approaches for offshore renewables are clearly warranted. This conclusion is supported by the successful experience applying these design methods to over 9.1 GW of new offshore wind projects globally.

As the offshore wind industry and other ORE sectors continue to mature and evolve, we encourage the industry to remain open and proactive in seeking design guidance that is tailored to ORE, in order to best support the rapid energy transition to net zero, to mitigate the climate crisis.

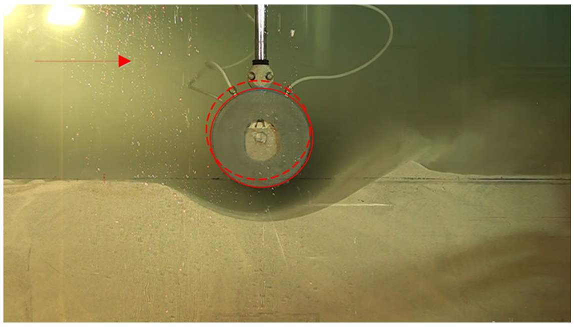

FIGURE 12
"Scale" model testing in UWA Large O-tube of gas trunklines using 200mm OD pipe is actually 1:1 scale testing for subsea cables. As per the STABLEpipe design method it is easy to show that the volume of soil needing to be mobilized to result in more than 50% lowering of a cable into the seabed can be achieved (and was frequently observed in UWA Large O-tube tests to occur) in about 10 minutes. It is therefore very much easier for a cable to be come self stable than (for example) a large-diameter gas trunkline.

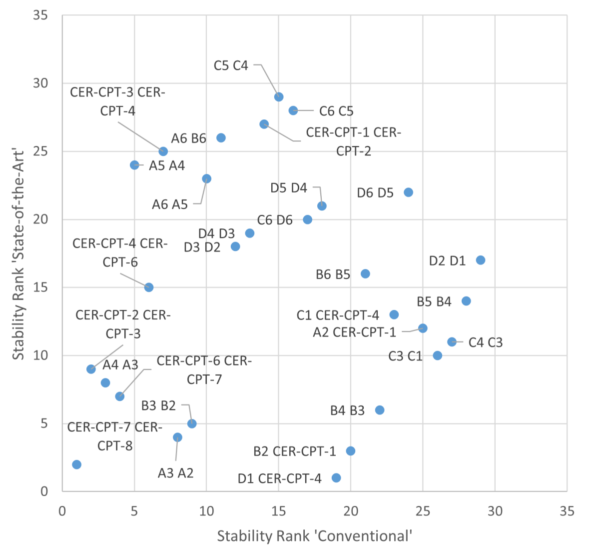

FIGURE 13
Ranking of cable routes by relative stability: conventional F109 rank versus 'state-of-the-art' methods, which demonstrates no discernable correlation whatsoever. The point of this plot was to assist the project team in determining the priority for cable trenching sequence to ensure as low as possible a risk of instability. It was found that the ranking of stability using conventional versus cable-optimized STABLEpipe design approaches resulted in such a low correlation that the reliability of the cables if assessed using the conventional approach might be severely compromised.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material. Further inquiries can be directed to the corresponding author.

Author contributions

TG drafted the majority of this work, which draws together ideas and concepts from a broad body of research work including prior publications by the co-authors. The remaining authors contributed significantly to the underlying research and industry case studies as well as the editorial evolution and refinement of the paper. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Glossary of terms

The following definitions are adopted in this paper:

COREStab: Cables on Rock Enhanced Stability (COREstab) is an innovative stability design method that recognises and quantifies the interaction between the cable and rocky seabed features. This creates much improved stability outcomes compared to conventional methods.

Macro scale: The macro-scale seabed survey features are those which are significantly larger (>10D) than the umbilical diameter. These features have horizontal and vertical lengths which result in them being captured within the Multi-Beam Echo-Sounder (MBES) seabed survey results. Note that the adoption of 10D here is based on the experimental findings of Griffiths et al. (2018c) and is unrelated to the 10D lateral displacement limit proposed in F109.

Marine growth: The communities of epibenthic (live on the surface) sessile (stay in one spot fixed to the surface) biota (plants and animals) which are predicted during design or observed during operation to settle (move there and live) on subsea cables or pipelines.

Meso scale: The meso-scale seabed features are those which are comparable in diameter to the umbilical (0.1D<L<10D). These features have horizontal and vertical lengths which result in them being too small to be captured within the MBES seabed survey results but are clearly visible in photographs or video survey results. These features are also of greatest importance in determining the umbilical on-bottom stability. Methods for characterising these features are described in Griffiths et al. (2018b).

Micro scale: The micro-scale seabed features are those which are much smaller than the diameter of the umbilical (<0.1D). These features have horizontal and vertical lengths which result in them being too small to be sized from the diver video surveys of the seabed. These features together with the exterior surface of the umbilical/ballast units are of greatest importance in determining the interface friction factor between the umbilical and the seabed.

STABLEpipe: Developed as an industry backed JIP at UWA, referenced in F109 and published as Griffiths et al. (2018d). The STABLEpipe methods are used for the design of pipelines and cables on sandy and silty seabeds. This incorporates sediment transport and scour models.

Subsea cable: This report primarily addresses the stability of cables being primarily multi-core helically-wound electrical conductors encased in layers of elastomeric sheaths and galvanised steel wire armour. However both functionally and in terms of on-bottom stability considerations, umbilicals and cables can be considered similar. That is, they have relatively small diameter, are heavily armoured, high SG and high flexibility (compared to a rigid hydrocarbon pipeline). Within the context of this report, the two terms (umbilical and cable) may be considered to be interchangeable, whilst pipeline is reserved for hydrocarbon service and pipes refers to all of the above elongate cylindrical products.

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Cost-benefit analysis of ballast water treatment for three major port clusters in China: evaluation of different scenario strategies

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Introduction: The expansion of maritime trade has led to the introduction of invasive species into aquatic ecosystems through ballast water discharge. China, being a major player in global trade and manufacturing, has experienced negative impacts on its coastal ecosystems and marine biodiversity.

Methods: This study examines the cost–benefit trade-offs of ballast water management policies for major port clusters in China and other global ports. This paper evaluates compliance costs for individual vessels and fleets under different policy scenarios and ballast water treatment system (BWTS) installation strategies.

Results: The onboard BWTS installation strategy appears to be more cost-effective under the existing International Maritime Organization (IMO) policy. However, with stricter global discharge requirements or a substantial increase in BWTS capital and operating costs, strategies based on port location could prove more beneficial due to potential economies of scale. Notably, ships with high ballast water discharge volumes, like bulk carriers, are potentially better equipped to cope with future policy shifts. In the face of stricter regulations in China, projected annual compliance cost increases range from $456 million (cost data based on China) to $1.205 billion (cost data based on US).

Discussion: Policymakers are advised to adopt a comprehensive view of ballast water management policies, taking into account the trade-offs between compliance costs and environmental risks. Other essential factors, such as advancements in BWTS technology, fuel consumption, emissions, and maintenance costs, also demand careful consideration in policy development.

KEYWORDS

marine transportation, biological invasion, ballast water, scenario analysis, cost-benefit analysis

1 Introduction

Ballast water is used to maintain the stability of ships during navigation, but it also provides a carrier for the transfer of nonindigenous species (NIS). As global trade increases, the problem of NIS introduction has become more pressing for the shipping industry (Drake et al., 2007; Ruiz et al., 2011). Aquatic species in ballast water can be transported from one port to another, outcompeting native species and causing ecological disruption and economic losses (McGeoch et al., 2010; Wan et al., 2016).

In China, the management of ballast water has become an increasingly pressing issue due to the rapid growth of its maritime industry. With its vast coastline and busy ports, China is particularly vulnerable to the introduction of invasive species through ballast water. In 2017, the first record of *scyphomedusa* in aquaculture ponds in China's southern Yellow Sea was reported, which may have negative impacts on the local ecosystem and industries such as aquaculture and tourism (Dong et al., 2019). The local species community structure in the South China Sea was altered by *Perna viridis*, *Pterois volitans*, *Penaeus monodon*, *Caulerpa racemose* and green crab (*Carcinus maenas*), which had negative impacts on the local ecosystem and economy (Compton et al., 2010; Wang et al., 2021). *Pseudocochlodinium profundisulcus*, a type of algae, has been consistently reported to cause algal bloom pollution events in the ballast water of vessels traveling between ports in China and North America since its initial discovery in China in 2006 (Shang et al., 2022). The microbial community in ballast water is diverse and complex, with a large number of bacteria, cyanobacteria, and actinomycetes that may carry toxins or introduce antibiotic resistance genes (Gerhard and Gunsch, 2019). During screening, 83.3% of ballast water samples from ships at China's Jiangyin Port were found to contain antiretroviral drugs, as well as antibiotic-resistant bacteria and multidrug-resistant bacteria (Lv et al., 2021). The total economic losses caused by invasive alien species to China in 2000 were estimated to be 144.5 billion US dollars (Xu et al., 2006).

Meanwhile, NIS from the East Asian coast also causes water pollution and damages aquatic ecosystems in other seas around the world. An invasive population of Chinese mitten crabs (*Eriocheir sinensis*) has recently formed in the San Francisco Bay system on the west coast of North America, causing millions of dollars in economic and ecological losses by damaging fishery resources and the aquarium industry (Dittel and Epifanio, 2009). Golden mussel (*Limnoperna fortunei*) is an invasive species mainly from China and Korea that has caused significant damage to the ecosystem and infrastructure of the Prata Basin in Argentina, Brazil, and other South American countries (Abelando et al., 2020; de Paula et al., 2020). One study estimated that the global economic cost of invasive species caused by ballast water discharge was approximately $162.7 billion in 2017 (Diagne et al., 2021).

To address these challenges, the International Maritime Organization (IMO) established the International Convention for the Control and Management of Ships' Ballast Water and Sediments (IMO regulations) in 2004 (IMO, 2004). This convention requires all ships to implement measures to manage their ballast water to minimize the transfer of harmful organisms and pathogens. IMO regulations establish the D-2 standard for ballast water performance, which establishes water quality standards for ballast water discharged after treatment by approved ballast water treatment systems (BWTS). Regarding ballast water treatment systems, physical and mechanical treatment technologies are considered primary treatment technologies. Common physical and mechanical treatment technologies include filtration, cyclone separation, heating, ultrasonic treatment and ultraviolet treatment. Chemical treatment technologies (such as chlorine gas in chlorination technology) may corrode tanks and ballast water tanks. In addition, toxic chemicals and volatile disinfection by-products generated during the production of biocides and during the treatment process pose a danger to crew members, human health and the environment (Benson et al., 2017; Ziegler et al., 2018). Reviewing historical literature reveals that different treatment technologies have significant differences in microbial inactivation efficiency because biological genome sequences, cell membrane structures, morphology and size, and evolutionary stages also affect the inactivation efficiency of treatment methods in addition to external factors such as pH value, temperature, turbidity (Sayinli et al., 2022). Therefore, different combinations of treatment systems must be implemented for various organisms present in ballast tanks, which can help improve microbial inactivation efficiency and better treatment efficiency of ballast water (Bradie et al., 2021; Lakshmi et al., 2021).

BWTS aims to reduce the biological concentration in ballast water to a very small fraction and is expected to significantly reduce the risk of potential invasive species spread. According to a study, as of November 2017, more than 80% of ocean-going ships traveling between the United States and Australia did not install BWTS, and the main method of reducing biological concentration in ballast water during navigation is still ballast water exchange (BWE) with the mid-oceanic waters (Gerhard et al., 2019). Compared with BWE, the installation, operation and maintenance of BWTS that meet the D-2 discharge standard will inevitably bring costs to shipowners and operators (Werschkun et al., 2012). However, in the long run, the potential environmental damage caused by invasive species may be much higher. With the continuous updating of ballast water management and the strengthening of marine safety and pollution prevention, policy implementation and supervision at all levels of government are gradually increasing demand for BWTS, forcing more and more ships to choose more efficient and effective systems to comply with D-2 standards.

The efficacy of IMO regulations in preventing the spread of invasive species has been questioned due to the need for regular review and updating, given the emergence of new species, lack of global awareness of BWM issues, and insufficient national institutional regulations (Čampara et al., 2019; Wright, 2021). To address these concerns, the state of California in the United States has implemented its own stricter regulations for ballast water management. The California State Lands Commission has established a ballast water treatment technology verification and evaluation program, requiring ships to meet a higher standard of treatment performance than the IMO regulations (CA State Lands Commission, 2021). Although this approach aims to reduce the risks associated with ballast water discharge, it also poses significant challenges for the global shipping industry, particularly for vessels operating between California and other regions (Čampara et al., 2019).

From the perspective of treatment efficiency and analysis, the main difference between IMO regulations and stricter California regulations is different ballast water discharge standards. If a few regions transition from IMO regulations to stricter regulations such as those of California, the shipping industry will face new challenges in meeting compliance requirements. This will involve increased costs associated with the installation and maintenance of BWTS, as well as the risk of noncompliance and potential legal penalties. Nevertheless, Strict discharge standards limit the biological concentration in ballast water discharged from ship ballast tanks. This greatly reduces the survival rate of carried species, reduces the damage of invasive species to ecology and economy, and makes it an important area for continued research and improvement. China's ports have played a critical role in the country's economic growth, serving as major gateways for international trade and commerce (MOT, The Ministry of Transport, 2020). Due to China's complex and diverse coasts and massive shipping volume, the country faces a significant risk of species invasion. Fortunately, China became a contracting party to the IMO regulations, which took effect for China on January 22, 2019 (Wan et al., 2021). Zhang et al., 2017 evaluated the total volume of ballast water discharge in China's major port clusters from 2008 to 2014. The results showed that the top three foreign ballast water discharge volumes were the Yangtze River Delta (31.7% to 39.0% of all ports in the country), followed by the Bohai Rim (27.6% to 36.5%) and the Pearl River Delta (24.7% to 28.8%). The southeast and southwest coasts were the regions with the smallest amount of ballast water received, accounting for 4.8% to 6.7% and 1.8% to 2.4%, respectively. Considering the high weight of the ballast water discharge volume of the former, this paper takes the three major port clusters as the research object.

Based on this background, this study examines the potential for China to adopt regulatory standards stricter than those of IMO regulations, considering both economic and environmental perspectives. Through an analysis of the costs and benefits of each approach, the study aims to determine the most effective strategy for managing ballast water in China while balancing compliance costs with the risks of invasive species introduction. Ultimately, the goal of this study is to provide guidance for policymakers and stakeholders seeking to enhance ballast water management in China and worldwide. The innovation of this study is to simulate the transition from IMO regulations to stricter regulations in specific regions (such as China) and evaluate the economic impact and feasibility analysis of strengthening ballast water management on individual ship or global fleet.

2 Method

2.1 Ballast water

The automatic identification system (AIS) is a ship reporting system that uses transponders installed on ships and at ports, canals, and other waterways. Through dual positioning and communication services provided by land-based equipment and satellite systems, AIS records information such as real-time ship position and displacement, ship number, ship type, port of departure, and port of arrival. In this study, the study used the 2018 global shipping data recorded by AIS, and after data preprocessing, screening, and other operations, the study obtained a dataset of 42,108 independent ships and 2,341,480 voyage records, with the unique code Maritime Mobile Service Identify (MMSI) used as the unique identification of the ship.

The National Ballast Information Clearinghouse (NBIC) based in the USA is a vital resource for researchers and policymakers seeking to understand and manage the environmental impact of ballast water discharge. Since 2004, the NBIC has been collecting ballast water exchange records for every ship entering US ports (NBIC Online Database, 2012). In this study, the study utilized ballast water discharge data from 2017-2018 to perform regression analysis and estimate the volume of ballast water discharge. While the NBIC's records are limited to vessels entering US ports, the study acknowledge that the discharge patterns observed in this region may not necessarily reflect those worldwide, including in China as a major exporting country. It is important to consider the potential differences in ballast water discharge patterns between countries. However, due to the lack of available data from other parts of the world, the study made the working assumption that all ships worldwide follow similar discharge patterns as captured in the NBIC dataset. This assumption allows us to utilize empirical data combined with regression analysis to gain in-depth insights into the global distribution and ecological impacts of ballast water discharge while acknowledging the potential uncertainties associated with applying US discharge patterns to China or other regions.

Faced with regulatory pressure on ballast water management policies, the compliance costs of ships include not only the capital costs of purchasing, installing and maintaining BWTS but also the operating costs of ensuring daily operations, which depend on the total ballast water discharge volume. Therefore, before calculating the compliance costs of fleets or individual ships, it is necessary to be able to obtain the actual ballast water discharge volume of global ships. Based on a working hypothesis (that all ships around the world follow similar discharge patterns captured in the NBIC dataset), this study uses empirical data and gravity models (Seebens et al., 2013; Wan et al., 2021) to predict the ballast water discharge volume of each ship, using ship type and deadweight tonnage as variables. The dataset revealed that ships do not always discharge ballast water every time they arrive at port, which is related to the nonzero discharge probability β. Furthermore, the ballast water discharge probability varies significantly between different ship types. For instance, the probability of discharging ballast water every time a bulk carrier calls is 57%, while the probability of discharging ballast water every time a container ship calls is only 9% (Table 1). The calculation formula is:

$$\text{Discharge}_{volume} = \beta * \text{discharge}_{volume} \tag{1}$$

where $\text{Discharge}_{volume}$ represents the average ballast water discharge volume, and $\text{discharge}_{volume}$ represents the average

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TABLE 1 Prediction of ballast water discharge and nonzero discharge ratio.

Type	a	b	β
Bulk carrier	-0.5445	0.8783	0.57
general cargo	-0.5968	0.80865	0.33
ro-ro ship	2.39503	0.05436	0.09
chemical carrier	0.716941	0.56166	0.41
passenger ship	-1.3222	0.98214	0.23
container ship	-0.006	0.632	0.09
oil tanker	-0.697	0.861	0.41
offshore working ships	2.18543	0.06062	0.19

discharge volume excluding zero discharge. According to the research of Seebens et al. (2013), if the ship discharges ballast water, the corresponding discharge is proportional to the size of the ship and varies with the ship type. Since the dimension of ballast water discharge is different from ship deadweight ton, regression fitting is performed after logarithmic processing. The calculation formula is:

$$log_{10}\ \text{discharge}_{volume} = \text{a} + \text{b} * log_{10}\ \text{DWT} \quad (2)$$

Among them, a and b represent the regression coefficients, and the parameter values are shown in Table 1. DWT: deadweight tonnage of the ship.

2.2 Policy scenario

Scenario analysis has a good effect on the choice of facing uncertainty (Morgan, 2017). In light of the uncertainties surrounding ballast water management and policy implementation, the study proposes three policy scenarios that take into account different ballast water management plans and policy implementation areas (Table 2). Ballast water management policies are categorized into IMO regulations and stricter regulations, while policy implementation areas are divided into three major port clusters in China (the Yangtze River Delta, Pearl River Delta, and Bohai Rim) and all other ports around the world. The strategies for treating ballast water discharged by ships include direct disinfection using ship-based BWTS and indirect disinfection using port-based BWTS. The barge BWTS is utilized to represent port-based BWTS in light of its economic feasibility and low occupancy benefits, as previously demonstrated by Wang et al. (2020).

Policy Scenario 1 represents the IMO regulations currently adopted by most countries, requiring the installation of IMO standard BWTS worldwide. In Policy Scenario 2, the study assumes that the three major port clusters in China serve as pilot areas for implementing stricter regulations, while other regions of the world continue to follow IMO regulations. In Policy Scenario 3, other regions of the world gradually transition to stricter regulations, following the lead of the three major port clusters, and eventually BWTS with stricter standards will be adopted globally.

Implementing stricter regulations, as represented in Policy Scenario 2 and Policy Scenario 3, is likely to present challenges and opportunities for the shipping industry and policymakers. For example, the cost of installing and maintaining BWTS may increase, reducing the competitiveness of the shipping industry in some regions. However, high-standard BWTS can eliminate or eradicate more organisms in ballast water before discharging it into a new environment, thereby preserving marine biodiversity and enhancing ecosystem resilience (Hess-Erga et al., 2019). Therefore, reducing the spread of NIS and potential risks to ecosystems could yield benefits in terms of biosecurity and ecosystem health.

2.3 BWTS cost data

Given the volatility of the BWTS market and the uncertainty of technology, the cost data used in this analysis are subject to change.

TABLE 2 Policy Scenario Description.

Policy Scenario	Compliance strategy	Description
1. Consistent IMO regulation	strategy1.1	IMO-BWTS on all vessels
	strategy 1.2	IMO-BWTS at all ports
2. Inconsistent regulation: The CN adopts stricter standards, while other regions of the world adopt IMO standards	strategy 2.1	Stricter-BWTS on Vessel-may-CN[1] IMO-BWTS on Vessel-never-CN
	strategy 2.2	Stricter-BWTS at CN Ports IMO-BWTS at non-CN Ports
	strategy 2.3	Stricter-BWTS at CN Ports IMO-BWTS on all vessels
	strategy 2.4	Stricter-BWTS on Vessel-may-CN IMO-BWTS at non-CN Ports
3. Consistent stricter regulation	strategy 3.1	Stricter-BWTS on all vessels
	strategy 3.2	Stricter-BWTS at all ports

[1]CN: China's three major port clusters (Yangtze River Delta, Pearl River Delta and Bohai Rim).

To address this issue, the study conducted a sensitivity analysis by incorporating a range of cost estimates in this analysis, including the lowest cost and 1.5 times the highest cost of BWTS. Through this approach, the study was able to assess the robustness of cost-effectiveness strategy in the face of uncertainty and variability.

To account for the potential impact of cost estimates on our analysis, the study captures BWTS cost variations associated with different geographical regions and regulatory regimes. The choice of data sources may impact the optimal strategy under different scenarios, and the study conducted sensitivity analysis to assess the robustness of the findings.

2.3.1 US-based cost data

To estimate the cost of BWTS produced in the United States that meets the IMO regulation, this paper relies on cost data (Table 3) from the research conducted by King et al. in 2009. The annual purchase, installation, and operating costs of the BWTS are based on the BWTS purchase and installation capital, with a BWTS lifespan of 30 years, a discount rate of 6%, and an annual inflation rate of 2.5%. In the case of BWTS that meets stricter standards, the cost data used in this study are obtained from the Delta Stewardship Council (Glosten, 2018), which includes the cost of barges and tugs (Table 4).

2.3.2 China-based cost data

Regarding the cost of BWTS produced in China that meets the IMO standard, the study consulted with a senior engineer from the China Classification Society and learned that the current cost price of BWTS ranges from 1 million RMB to 5 million RMB (Communications, Senior Engineer, China Classification Society, July 12, 2021). Additionally, by analyzing the BWTS bidding documents on the National Bidding and Purchasing Information Platform in 2021 and referring to the Chinese national industrial power consumption standard (China Tendering and Bidding Public Service Platform, 2021; NEA, National Energy Administration, 2021), the study obtained the cost data of BWTS in China (Table 5).

When using BWTS cost data produced in China, the annual purchase and installation costs of BWTS are calculated based on the purchase capital and installation costs, assuming a service life of 20 years. Operating expenses are equal to the product of the ballast water treatment capacity and the unit electricity charges for ballast water treatment (Communications, Senior Engineer, China Classification Society, July 12, 2021). Additionally, the cost estimate data for stricter compliant BWTS comes from Chinese BWTS manufacturers. According to their predictions, the treatment efficiency of the stricter BWTS is expected to be approximately 10 times higher than the IMO standard. The treatment effect of stricter BWTS is predicted to be approximately 10 times higher than that of the IMO standard (Communications, Senior Engineer, China Classification Society, July 12, 2021).

2.4 Compliance cost model

Based on three scenario strategies, the corresponding cost models are established based on the two frameworks of world fleet and single ship.

2.4.1 Compliance cost model of world fleet

If BWTS is installed based on ships, the compliance cost of the world fleet at this time is:

$$C_{fleet} = \sum^{N}(C + O) + V * T \tag{3}$$

where N represents the number of ships in the fleet, C represents the BWTS annual capital and installation cost of a ship, O represents the BWTS annual operating cost of a ship, V represents the total amount of ballast water discharged by the global fleet, and T represents the treatment cost of unit ballast water.

If the BWTS is installed on a port barge, the fleet compliance cost also includes the purchase, installation and operation of the barge itself. Calculate the number of barges required for each port, taking into account the difference in the annual ballast water handling capacity of each port:

$$n_{barge} = V_{port}/365day/24hour/Capacity \tag{4}$$

Where n_{barge} represents the minimum number of barges required by a port, V_{port} represents the total ballast water treatment capacity of a port per year, and *Capacity* represents the processing capacity of BWTS. The estimated processing capacity of equipment in the United States is 2000 MT/h, and the processing capacity produced in China is 300 metric tons per hour to 1500 metric tons per hour (MT/h) (Senior Engineer of China Classification Society, July 12, 2021, communication). Under this setting, based on the barge each time the BWTS handles ballast water, it will require a corresponding number of tugboats to provide transport capacity:

TABLE 3 Annual cost profile of BWTS based on US data.

US data	Total capital and installation cost ($)	Annual capital and installation cost ($)	Annual operating cost ($)	Unit treatment cost ($/MT)
Lower bound	65.8 (460) [1]	3.5776 (25.0108)	0.9 (32.6)	0.02 (0.27)
Average	90.1 (700)	4.8989 (38.0599)	1.35 (50.2)	0.135 (0.48)
Upper bound	114.4 (990)	6.2202 (53.8276)	1.8 (67.8)	0.25 (0.68)

[1]The cost under the IMO standard (the cost under the stricter standard), the same below.
King et al., 2009; Glosten, 2018.

TABLE 4 Annual cost profile for a barge.

A barge	Total capital and installation cost ($)	Annual capital and outfitting cost ($)	Annual operating cost ($)	Tug ($/treatment)
Lower bound	630	34.254	23.1	1.18
Average	1010	54.915	23.1	1.18
Upper bound	1550	84.2755	23.1	1.18

Glosten, 2018.

$$C_{fleet} = \sum_{i=1}^{M}(C + O + C_{barge} + O_{barge}) + V * T + \sum_{i=1}^{M} n_{barge} * T_{tug} \quad (5)$$

where M represents the number of ports, C_{barge} and O_{barge} represent the annual capital cost and operating cost of a barge, respectively, and T_tug represents the towing cost of a barge when participating in ballast water treatment.

2.4.2 Compliance cost model for a single ship

If BWTS is installed on a ship basis, the compliance cost for a single ship at this time is:

$$C_{vessel} = \frac{V_{vessel}}{V_{all}} * \sum_{i=1}^{N}(C + O) + V_{vessel} * T \quad (6)$$

where V_{vessel} represents the annual ballast water discharge of a single ship, and V_{all} represents the annual ballast water discharge of all ships.

If the BWTS is installed on port barges, similar to the compliance cost calculation of the global fleet, the compliance cost of a single ship also includes the purchase, installation, operation and towing costs of the barge itself. Considering the difference in the maximum deadweight tonnage of ships, the calculation formula for the minimum number of barges required for a single ship is:

$$n_{vessel} = V_{vessel}/365day/24hour/Capacity \quad (7)$$

where n_{vessel} represents the number of barges required for a single vessel.

Therefore, the formula for calculating the compliance cost per vessel based on port barges is:

$$C_{vessel} = \frac{V_{vessel}}{V_{all}} * \sum_{i=1}^{N}(C + O + C_{barge} + O_{barge}) + V_{vessel} * T + T_{tug} * n_{vessel} \quad (8)$$

2.5 Model assumptions

To simplify the model and calculations, this study makes the following assumptions:

Assumption 1: The ballast water discharge model fitted using regression analysis based on NBIC data is applicable to all ships in service in 2018.

Assumption 2: Ships and ports in the global shipping network have the necessary capacity or suitable conditions to install BWTS that meet regulatory requirements.

Assumption 3: Since there are currently no barges specifically designed and researched for installing BWTS in China, the study assumes that the cost data for installing BWTS on barges in China are the same as those published in the United States.

Assumption 4: The study does not differentiate between BWTS based on their disinfection ability or method; instead, the study categorizes them only according to whether they meet IMO standards or stricter standards.

2.6 The export price of China's BWTS

The Ballast Water Management Convention, which mandates that all ships must be equipped with BWTS to meet the D-2 standard by 2024, was implemented in 2017. D-2 refers to the

TABLE 5 Annual cost profile of BWTS based on China data.

China data	Total capital and installation cost ($)	Annual capital and installation cost ($)	Annual operating cost ($)	Unit treatment cost ($/MT)
Lower bound	15.41 (30.82)	0.7705 (1.541)	0.036 (0.072)	0.02 (0.04)
Average	46.23 (92.46)	2.3115 (4.623)	0.036 (0.072)	0.135 (0.27)
Upper bound	77.05 (154.1)	3.8525 (7.705)	0.036 (0.072)	0.25 (0.5)

King et al., 2009; China Tendering and Bidding Public Service Platform, 2021; Senior Engineer of China Classification Society, 2021.

discharge standard for ballast water management, which sets the maximum allowable concentration of indicative microorganisms in ballast water discharged into the environment. As a result, this study only focuses on predicting the BWTS price from 2020 to 2024. The study collected the export sales and quantity of ship BWTS from 2014 to 2020 from the China Customs Import and Export Statistics website using the commodity number (84212191) (General Administration of Customs, 2021).

To understand the changing trend of BWTS cost data, the study used a combination of the gravity model and polynomial regression to fit the export price per unit of BWTS. The study then used this model to predict the expected export price from 2020 to 2024.

3 Result

3.1 Overview of ballast water discharge

Policy scenarios 1 and 3 entail globally consistent regulations (IMO or stricter), while policy scenario 2 involves different regulations being applied in pilot areas compared to control areas. Therefore, the study take into account the differentiated ballast water management approach where ships are divided into two categories based on their history of visiting the policy pilot area. The statistical analysis results in Table 6 present the number of ships and the total amount of ballast water discharged by ships in different regions. Based on the data, the study estimated that the global shipping fleet discharged approximately 1.26 billion tons of ballast water in 2018. These findings provide an overview of the ballast water discharges of ships and help to understand the potential impacts of ballast water on the marine environment.

3.2 Cost-effectiveness analysis of the world fleet

3.2.1 Assessment based on US data

The compliance cost of the world fleet under each policy scenario and optimal strategies are presented in Table 7, assuming that the BWTS produced in the United States is used globally. In Policy Scenario 1, where the world adopts consistent IMO regulations, Strategy 1.1 has the lowest compliance cost among all strategies, with an average cost of only $2.802 billion. This suggests that the current strategy of installing BWTS based on ships is cost-effective in promoting IMO regulations and protecting marine life and ecological safety.

In Policy Scenario 2, where stricter regulation is adopted in China's pilot regions while IMO regulation is adopted in other regions, Strategy 2.4 is the most cost-effective among all compliance cost strategies. This strategy involves all ships installing IMO standard BWTS and port barges in China's pilot areas installing stricter BWTS. This policy adjustment only affects the pilot areas of China and reduces the pressure on shipowners to retrofit BWTS. Moreover, stricter BWTS installed on port barges can help reduce the probability of marine invasive species invading Chinese waters and the potential losses they may cause.

In Policy Scenario 3, the compliance cost of Strategy 3.2 is significantly lower than that of Strategy 3.1, indicating that shore-based BWTS is more cost-effective as the ballast water discharge standards become stricter. Despite the higher capital cost, economies of scale make shore-based BWTS a more cost-effective option for the world's fleet. Additionally, shore-based barges equipped with BWTS can process contaminated ballast water 24/7, achieving high equipment utilization, while ship-based BWTS are only needed during the period of port calls. Under optimal strategy 3.2, the maximum cost of installing Stricter-BWTS at all ports worldwide is $14.839 billion.

The robustness of the study's results was verified by adjusting the upper and lower limits of capital costs for IMO-BWTS. The optimal strategies for each policy scenario remain unchanged, with Strategies 1.1, 2.4, and 3.2 continuing to be the most cost-effective options across all scenarios (Table 7).

3.2.2 Assessment based on China data

Table 8 presents the annual fleet cost of each compliance strategy when using the BWTS cost data produced in China, with the optimal strategy under each scenario. The cost of BWTS production in China is much lower than that in the United States, resulting in a lower overall compliance cost of the ship fleet based on Chinese BWTS cost data. The cost difference is particularly significant in policy scenario 3. The optimal compliance cost based on US data is almost 6 times that based on China data. However, this will also be accompanied by a significant reduction in environmental risk, as stricter BWTS produced in the United States can achieve disinfection performance that may be more than 100 times that of the IMO standard (Glosten, 2018).

Compared with the United States, the significantly reduced cost of BWTS manufactured in China has also changed the best compliance strategy. The relatively low cost of BWTS is more friendly to ship-based strategies, because this strategy uses more BWTS than port-based strategies and does not depend on barge disinfection of ballast water. Consequently, the compliance cost

TABLE 6 Overview of ships and estimated ballast water discharges in 2018.

Ship category		Number of ships	Number of discharges	Discharge volume (million tons)
Never to the pilot area		14624	834163	576.7
Ever to the pilot area	Discharge in the pilot area	27484	94793	196.7
	Discharge to nonpilot area		241783	489.9
Total		42108	1,170,739	1263.3

TABLE 7 Annual compliance cost of the world fleet based on US data (US$100 million).

US data	Strategy	Lower bound	Average	Higher bound	Lowest bound with the lowest IMO-BWTS capital cost	Highest bound with the highest IMO-BWTS capital cost
1. Consistent IMO regulations	**Strategy1.1**	**19.11**	**28.02**	**36.93**	**14.86**	**50.03**
	Strategy1.2	138.85	140.46	142.14	138.84	142.16
2. Inconsistent regulations: The CN adopts stricter standards	Strategy2.1	98.53	150.03	206.02	95.25	214.57
	Strategy2.2 (1)	185.03	232.09	283.11	185.02	283.13
	Strategy2.2 (2)	212.33	258.93	309.54	212.33	309.56
	Strategy2.3	149.51	155.86	163.26	149.50	163.28
	Strategy2.4	**30.91**	**40.07**	**49.21**	**25.89**	**62.31**
3. Consistent stricter regulations	Strategy3.1	246.00	377.71	520.74	246.00	520.74
	Strategy3.2	**142.39**	**145.41**	**148.39**	**142.39**	**148.39**

the optimal strategy scenarios are shown in bold.

based on port barge-installed BWTS is higher than that based on ship-installed BWTS, and the optimal strategy in Scenario 2 becomes Strategy 2.1, while the optimal strategy in Scenario 3 becomes Strategy 3.1 (Table 8).

The results based on China's cost data also exhibit excellent robustness, even in the face of price changes. Therefore, they can serve as a reliable guide for decision-making.

3.3 Single ship compliance cost strategy

To account for the temporal variability of the cost data used in the analysis, the study have calculated not only the mean compliance costs but also the upper and lower limits to represent the extreme values under cost fluctuations. This provides decision-makers with a range of potential costs and allows them to assess the robustness of their compliance strategies under different cost scenarios. These sensitivity analyses are critical for ensuring the effectiveness and sustainability of BWTS compliance policies in the long term.

3.3.1 Evaluation based on US data

The analysis of compliance costs reveals that the average unit cost of ballast water treatment per vessel increases as regulations become stricter, with Strategy 1.1 having the lowest cost and Strategy 3.2 having the highest cost (Table 9). However, there is significant variability in the unit cost of ballast water treatment per vessel, ranging from as low as 0.24 USD to almost 13,000 USD per ton of ballast water treated. This variation is due to the inclusion of smaller vessels, such as passenger and coastal vessels, which discharge smaller volumes of ballast water annually (with the smallest volume being only 4.90 MT).

To capture the variability in ballast water discharge volumes and probabilities among different vessel types, the study calculated

TABLE 8 Annual compliance cost of the world fleet based on China data ($100 million).

China data	Strategy	Lower bound	Average	Higher bound	Lowest bound with the lowest IMO-BWTS capital cost	Highest bound with the highest IMO-BWTS capital cost
1. Consistent IMO regulations	**Strategy1.1**	**3.95**	**11.89**	**19.84**	**2.87**	**27.96**
	Strategy1.2	139.27	141.46	144.96	139.27	145.02
2. Inconsistent regulations: The CN adopts stricter standards	**Strategy2.1**	**5.46**	**16.45**	**27.43**	**4.75**	**32.73**
	Strategy2.2 (1)	102.13	109.50	117.97	102.13	118.01
	Strategy2.2 (2)	130.39	137.19	145.10	130.39	145.14
	Strategy2.3	144.57	157.01	166.45	154.57	166.50
	Strategy2.4	15.31	23.61	35.11	14.23	40.22
3. Consistent stricter regulations	**Strategy3.1**	**7.90**	**23.79**	**39.67**	**7.90**	**39.67**
	Strategy3.2	139.99	143.66	148.69	139.99	148.69

the optimal strategy scenarios are shown in bold.

TABLE 9 The unit compliance cost of a single ship under the optimal cost strategy.

Source	Optimal strategy	Unit compliance cost($/MT)		
		Min	Max	Average
US data	Strategy1.1	0.24	12750.11	49.6
	Strategy2.4	0.24	12750.11	53.42
	Strategy3.2	1.17	1204.83	72.81
China data	Strategy1.1	0.21	4716.45	18.47
	Strategy2.1	0.21	4716.45	20.12
	Strategy3.1	0.42	9432.9	36.93

the upper and lower limits of the unit cost of ballast water treatment per vessel (Supplementary TableS1). The results indicate that among ships of the same size, the annual discharged amount of ballast water per ship is the lowest for passenger ships, while the annual compliance cost per ship is the highest. This suggests that while passenger ships may have a smaller environmental impact in terms of ballast water discharge, they may also face higher regulatory costs compared to other types of ships. On the other hand, bulk carriers have the largest annual ballast water discharge volume per vessel, and the economies of scale allow for a lower compliance cost, making it more advantageous for potentially stricter policies. These findings provide valuable information for decision-making regarding the implementation of BWTS regulations and the management of ballast water in different types of vessels.

Notably, Strategy 3.2 has a lower maximum compliance cost than Strategy 1.1, with a cost of only 1204.83 USD/MT (Table 9). This is because the port-based strategy allows shipowners to share the regulatory pressure on ballast water discharges, and the compliance cost of a single vessel primarily depends on its ballast water discharge volume. In contrast, Strategy 1.1 requires installing BWTS on each vessel, and the capital, operation, and ballast water treatment costs of the BWTS are borne by the vessel itself. For vessels with low discharge volume, the capital cost of BWTS per MT is extremely high, resulting in a high compliance cost for each vessel. However, Strategy 3.2 installs BWTS on barges in ports, and the costs of BWTS are calculated based on the needs of each port and then shared among all vessels in operation. In this way, the compliance cost borne by each vessel is proportional to its annual ballast water discharge volume.

To address the issue of high compliance costs for vessels with low discharge volumes, the study removed vessels with low discharge volumes in a proportion of 1%-30% and recalculated the compliance costs for each policy scenario by separately counting vessels above each discharge threshold (Table 10; Figure 1A). The results showed that as the discharge threshold increased, the compliance cost per vessel gradually decreased for all three strategies, indicating that installing BWTS is more cost-effective for vessels with larger ballast water discharge volumes. For vessels with a discharge threshold below 500 MT, the cost-effectiveness was low, and there was an obvious inflection point in the compliance cost per vessel for both Strategy 1.1 and Strategy 2.4. Therefore, from the perspective of shipowners' interests, such vessels may not be suitable for installing BWTS (as used in this study), and alternative methods, such as the use of pure water, can be considered (Wang and Corbett, 2020).

3.3.2 Evaluation based on China data

The study findings suggest that the stricter installation standards of BWTS on ships, as implemented in Strategy 2.1 and Strategy 3.1, have a significant economic advantage over port-based strategies. This strategy may be effective and applicable in terms of economic and technological aspects for specific ports, as evidenced by the study conducted by King and Hagan, 2013. This is due to the lower capital and operational costs of the BWTS produced in China compared to those in the United States and the high cost of the full set of barges announced by California (Glosten, 2018).

Table 9 shows that the difference in the unit compliance cost per vessel between the strategies based on US data, Strategy 1.1 and Strategy 2.4, and the strategies based on Chinese data, Strategy 1.1 and Strategy 2.1, is minimal. This indicates that it is economically feasible for shipowners to implement stricter regulations in China's three leading regions.

Figure 1B demonstrates that strategies based on ship-based installation of BWTS are more advantageous for high-emitting vessels, as they benefit from economies of scale, resulting in low unit compliance costs per deadweight ton. The inflection point at the same location again indicates that vessels with an annual ballast water discharge of less than 500 MT are not suitable for strategies based on ships.

TABLE 10 Annual ballast water low discharge thresholds for different proportions.

Discharge thresholds	4000	3000	2500	1800	1000	500	180	80
Number of ships removed	12586	10135	8609	6466	4133	2204	1013	429
Ratio	30%	25%	20%	15%	10%	5%	2.5%	1%

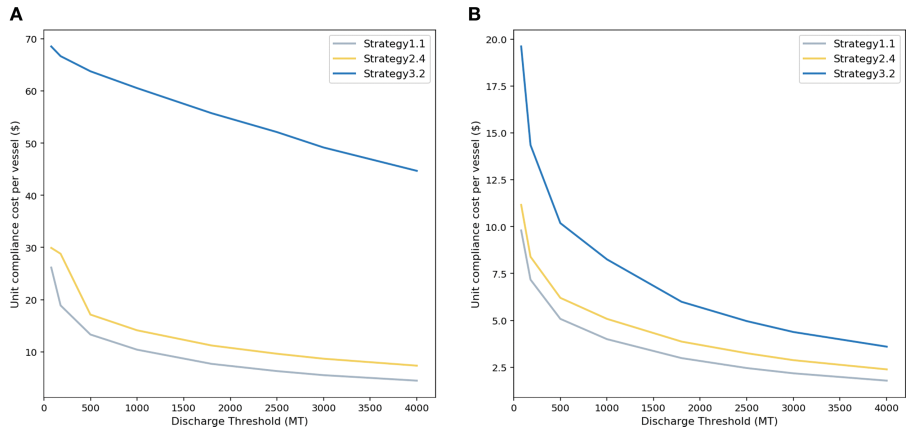

FIGURE 1
Changes in compliance costs of optimal strategies under different discharge thresholds. **(A)** Based on US data, **(B)** Based on China data.

3.4 Forecast of BWTS price change

While the BWTS cost used in this study may differ from the BWTS price exported by Chinese Customs, future changes in the latter can reflect the changes in BWTS cost data to some extent (Figure 2). The results indicate that the fourth-order polynomial regression curve fits better, with a smaller residual of 0.00058 for the least squares fitting, compared to the corresponding residual of 0.001 for the third-order fitting. The predicted results suggest that the future export price of BWTS produced in China will show a slow growth trend, indicating the stability of BWTS cost data. Therefore, the evaluation results of this study are robust and still provide good reference value in the coming years.

4 Discussion

Compliance costs are contingent upon the digital standards and data sources needed. Calculations based on China data indicate that implementing ship-based compliance technologies is more cost-effective than centralized compliance using barges in the three major port clusters. Thus, regardless of which standard is used, it is recommended to install ship borne BWTS in Chinese ports. Similarly, calculations based on US data indicate that ship-based compliance technologies are cheaper than centralized compliance using barges if consistent IMO standards are globally adopted. However, if stricter standards are implemented regionally or globally, or if the cost of BWTS increases significantly,

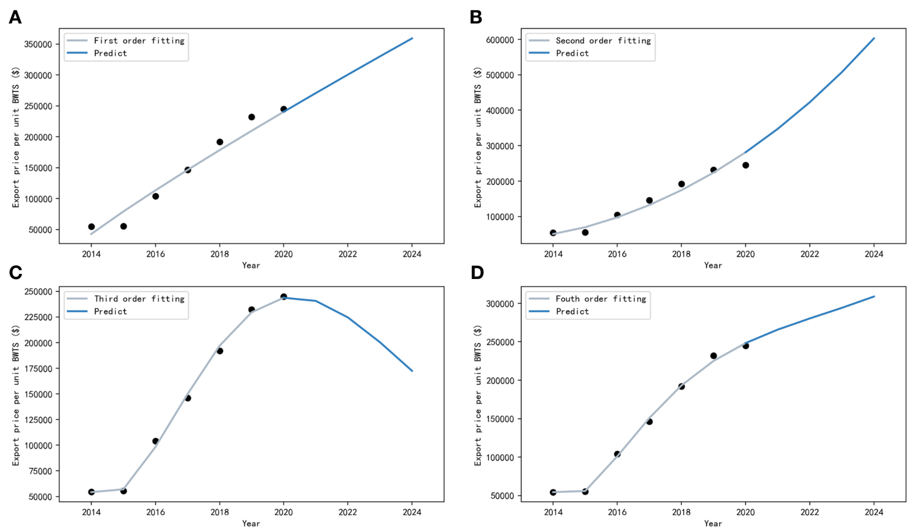

FIGURE 2
Estimated changes in export costs per unit BWTS. **(A–D)** represent the regression curves at different orders and the predicted price of unit BWTS.

compliance costs for barge-based systems may be lower than retrofitting the entire global fleet, as supported by Wang and Corbett, (2020). As China transitions from IMO regulations to more stringent ones, the annual increase in compliance costs varies from $456 million (based on Chinese data) to $1.205 billion (based on US data).

As for individual ships, the cost of unit ballast water discharge is closely related to the digital standards required by regulations. In simple terms, the higher the anti-pollution level, the higher the cost of ballast water treatment that shipowners or operators need to bear. If certain regions (such as China's three major port clusters) adopt stricter standards, the average cost of ballast water treatment per ton only increases by $1.65 (an increase of 8.9%, based on Chinese data) or $3.82 (an increase of 7.7%, based on US data) relative to the relatively weak IMO standard adopted globally. From the perspective of shipowners' interests, if the management of specific regions considers the pressure of anti-pollution safety and adopts stricter ballast water management, the increase in operating costs per ship is relatively small. This indicates that the transition from global standard IMO regulations to stricter anti-pollution policies is feasible, and this finding also applies to regions other than Chinese ports.

The result shows that regardless of the policy scenario or the way BWTS is installed, the compliance costs per ship will gradually decrease as the annual ballast water discharge volume increases. Different types of ships have significant differences in the average annual ballast water discharge volume. Ships with larger discharge volumes (such as bulk carriers) have higher cost-effectiveness in compliance costs and are more likely to ease economic pressure due to economies of scale when facing new management policies. Ships with low discharge volumes (such as passenger ships) have higher operating costs and may not be cost-effective to use BWTS to treat ballast water. Here, it is recommended that ships with an average annual ballast water discharge volume of less than 500MT rely on current BWTS equipment for disinfection treatment and can choose other ballast materials such as pure water to replace. Effective management of specific ship types is important, although this paper did not consider the differences in disinfection effects of different types of BWTS on different ship types. It is recommended that different ship types can adjust their own choices according to actual financial, legal and operational conditions when choosing BWTS, and shipowners and operators can choose the most suitable BWTS for their ships based on indicative factors such as tonnage and age (Satir, 2014).

The stability of BWTS cost data is a key factor in determining the best compliance strategy for the shipping industry. According to our forecast, the export price of BWTS in China may continue to rise in the future, which may affect the cost-effectiveness of compliance strategies. Considering future uncertainties, incorporating the latest information on shipping trends, technological advances, and potential changes in regulations or trade patterns into cost-benefit analysis can more accurately understand current and future challenges in ballast water management. According to an estimate, China's ballast water discharge volume is slowly increasing (Zhang et al., 2017). This paper recommends that decision-makers need to establish a ballast water management information platform to ensure mandatory reporting of ballast water discharge and unified detection and monitoring methods for ballast water. Cost-benefit analysis should be used to formulate ballast water management regulations and monitor harmful aquatic organisms and pathogens in ballast water (Rey et al., 2018). It is also necessary to encourage the development of more effective, efficient and environmentally friendly new technologies for BWTS to ensure the safety of ships and crew members and minimize compliance costs.

5 Conclusion

The findings of this study have important implications for Chinese policymakers and other countries with similar ecological conditions. Adopting stricter regulations can significantly reduce the risk of biological invasions and associated economic losses, but it is also crucial to balance the environmental benefits and economic costs of implementing BWTS policies. The study emphasizes the importance of evaluating and updating the cost-benefit analysis regularly to provide more comprehensive policy guidance.

However, the study still has several limitations: 1. Lack of explicit data recording or sources: The interpretation of results is subject to limitations and potential biases due to the absence of clear data records or sources. 2. Limited consideration of environmental risks: The primary focus of this study is on compliance costs related to ballast water management policies, but it may not fully address all potential environmental risks associated with invasive species nor adequately quantify the mitigation effects of different ballast water treatment strategies on coastal ecosystems. 3.Insufficient assessment of influencing factors: While the study mentions the importance of evaluating potential factors such as fuel consumption, emissions, and maintenance costs, including BWTS technology improvements, it does not delve into how these factors impact the overall cost-effectiveness and feasibility of different ballast water management policies.

Future studies should pay attention to the changes in BWTS cost data, especially from different sources, to improve the accuracy and reliability of the evaluation results. Furthermore, the study highlights the potential long-term economic and environmental benefits of adopting more efficient and sustainable BWTS technologies, such as reduced fuel consumption, emissions, and maintenance costs. As such, there is a need for continued research into the impact of such factors on the cost-effectiveness of compliance strategies.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Material. Further inquiries can be directed to the corresponding authors.

Author contributions

AN: Writing—Original draft preparation, Methodology, Visualization, Formal analysis, Investigation, Validation. ZW: Conceptualization, Resources, Data Curation, Writing—Original draft preparation, Writing—Review & Editing, Supervision, Funding acquisition. ZS: Methodology, Visualization, Writing—Original draft preparation, Formal analysis, Investigation, Validation. ZJW: Writing—Original draft preparation, Methodology, Review & Editing, Supervision. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2023.1174550/full#supplementary-material

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Epistemology of ignorance: the contribution of philosophy to the science-policy interface of marine biosecurity

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Marine ecosystems are under increasing pressure from human activity, yet successful management relies on knowledge. The evidence-based policy (EBP) approach has been promoted on the grounds that it provides greater transparency and consistency by relying on 'high quality' information. However, EBP also creates epistemic responsibilities. Decision-making where limited or no empirical evidence exists, such as is often the case in marine systems, creates epistemic obligations for new information acquisition. We argue that philosophical approaches can inform the science-policy interface. Using marine biosecurity examples, we specifically examine the epistemic challenges in the acquisition and acceptance of evidence to inform policy, discussing epistemic due care and biases in consideration of evidence.

KEYWORDS
invasion ecology, evidence, bias, obligation, uncertainty

Introduction

The use of evidence in policy and decision making is increasingly promoted as highly desirable, especially for environmental issues. This has resulted in the adoption of evidence-based policy (EBP) ostensibly to provide greater transparency and consistency in decision making by relying on evidence that can be externally verified and validated (Wesselink et al., 2014). Yet the adoption of EBP creates epistemic challenges and responsibilities (i.e., with regard to the acquisition and reliability of knowledge) requiring decision-makers to use relevant scientific research findings, often on topics in which they have little or no expertise. Additionally, decision making in the face of uncertainty, specifically where limited or no prior empirical evidence exists, requires transparent approaches to determine how information is acquired, considered and accepted (or not) (Meßerschmidt, 2020).

Public policymaking has variously been viewed as a collective process of mediation and codification of social ideals or in a contrary view as an authoritative means to enact the will of government on the people (Wesselink et al., 2014; see also Wears and Hunte, 2014; Pak et al., 2021). Increasingly, there is a desire to shift from ideological and intuitive processes to systems that provide greater transparency and consistency by relying on 'evidence' through an EBP approach (Wesselink et al., 2014; Sánchez-Bayo et al., 2017).

The centrality of evidence in EBP generates epistemic responsibilities and challenges that are fundamentally philosophical by nature: What counts as evidence? Who is responsible for providing evidence? Where should the burden of proof lie? How far do our epistemic obligations extend? Additionally, there can be personal and systemic incentives (accidental and intentional biases) to encourage and maintain ignorance (defined as the absence or lack of knowledge or understanding). While different ways of effectively navigating the science-policy 'space' have frequently been debated (e.g., Ban et al., 2013; Sánchez-Bayo et al., 2017), the discipline of philosophy can help to illuminate the epistemological challenges arising therein.

Philosophy as a discipline is experiencing a surge in research activity regarding the social and ethical dimensions of knowledge and ignorance. One focus has been the novice-expert problem: how non-experts identify, access and interpret reliable sources of information (e.g., Goldman, 2001; Anderson, 2011; Guerrero, 2017). Using formal modelling techniques, philosophers have examined how knowledge spreads (or fails to spread) from scientists to decision-makers, and how propagandists may influence this process (Weatherall et al., 2020). This extends work in history of science showing how perceptions of the scientific record can be distorted by amplifying scientific findings that favor specific conclusions, thereby creating a false sense of legitimate controversy, confusing decision-makers and the public, and delaying action (Oreskes and Conway, 2010). Other work focuses on epistemic failings of our social structures, e.g., rejection of established scientific findings along partisan lines (Levy, 2019), or the incentive to rush into print and the resulting exacerbated risk of replicability problems (Heesen, 2018). Further, substantial philosophical debate exists on the assignation of responsibility for ignorance – when should we have known what we did not to know and to what extent we are required to investigate the impacts of our actions and omissions (Miller, 2017).

Here we consider these epistemic challenges in the acquisition, consideration and acceptance of scientific evidence to inform marine environmental policy, including standards of epistemic due care and biases in consideration of evidence, to demonstrate the contribution philosophy can make at the science-policy interface. Specifically, we consider the case of marine biosecurity (i.e., the management of human mediated biological introductions) that requires immediate action, but is also heavily impacted from limited scientific information. In doing so, we leave aside some of the wider challenges of interpreting and accepting scientific evidence, where the problem may be one of ignorance of, or, more neutrally put, a lack of appreciation for, methods of scientists, including the issue of statistical significance and replicability.

Standards of epistemic due care and epistemic obligations

The old saying that 'ignorance is bliss' rings hollow when it comes to irreversible changes to our social and natural environment that may have undesirable if not catastrophic consequences. Policy decisions are always made under some level of uncertainty – our knowledge concerning any issue is never (and can never be) complete. This raises the question what reasonable standards of epistemic due care consist in. What responsibilities do policy-makers have to seek sufficient evidence to make an informed decision and to what extent can decision-makers reasonably be expected to investigate the ramifications of proposed policies and regulation? Naturally, appropriate standards of epistemic due care will always be context-specific.

In the case of marine biosecurity incursions, a biosecurity response may involve trade or port closures to reduce the likelihood of spread and impact while balancing such a response against potentially significant impacts to industries causing wider economic repercussions for society. The rapid response to the Black Striped Mussel, *Mytilopsis sallei* (Récluz, 1849), incursion in Darwin, Northern Territory, Australia in 1999 was based on the then-available evidence. The incursion was determined to pose a sufficient risk to enact a quarantine closure of three commercial and recreational marinas in order to enact an eradication response (Bax, 1999; Willan et al., 2000), despite significant economic impact to charter and tourist vessel operators. The eradication was successfully conducted over a 15-day period. In determining marine biosecurity action, policy-makers and decision-makers will – often implicitly and perhaps even unconsciously – make decisions about how much evidence is enough, what kind of evidence is needed, whether to investigate the issue further to collect more evidence and, if so, which direction such investigations should take. Evidence gathering takes time and it is often necessary to act before much evidence becomes available. The highly complex, diverse and dynamic character of the systems in which conservation initiatives operate means that the people making decisions will unavoidably be ignorant of the full range of facts and potential outcomes of their decisions. The detection of a novel marine species requires rapid action – it is frequently detected only after the population has reached sufficient density to be observed, reported and identified.

In the case of the Black Striped Mussel incursion the then-available evidence, even though incomplete, was deemed to be of sufficient quantity and quality to warrant the above-described action response. The success of that response, the eradication of *Mytilopsis sallei* (Récluz, 1849) from the area, appears to suggest that appropriate standards of epistemic due care where met. However, where decisions concerning the adequacy of existing evidence are made without recourse to general principles or overarching standards, policy decisions are not based on solid foundations.

We believe that research in applied epistemology, focusing on the epistemological aspects of EBP can provide such foundations. In saying so, we want to emphasize that we are not advocating for a mere, straight-up 'application' of concepts developed in philosophical epistemology to the questions faced by EBP. Rather, we see a need for sustained engagement between philosophers, scientists and EBP practitioners. Such a systematic and comprehensive approach would enable us to develop general principles and guidelines for epistemically sound policy-making in the marine biosecurity space and beyond. One aim of such an approach would be to integrate existing epistemic principles into an overarching set of criteria for assessing the adequacy of one's evidence. In the following, we discuss two such principles that are already being employed, albeit not necessarily in a systematic or even explicit way.

The first one is a type of epistemic proportionality principle: it would appear that the greater the potential detrimental impact of our actions (including inaction), the more demanding are our obligations to improve our epistemic position vis-à-vis the issue at hand, e.g. the characteristics and potential impact of *Mytilopsis sallei*. An action (or omission) that could lead to the eradication of an entire species plausibly requires more thorough investigation than an action that may merely affect the local population. In other words, we have obligations to gather evidence which we know bears on policy responses, in proportion to the significance of the problem. In the case of marine biosecurity responses this may impose requirement for baseline knowledge to inform rapid response (Chapman and Carlton, 1991; Chapman and Carlton, 1994; Ojaveer et al., 2015; Campbell et al., 2018).

The second one is an epistemic precautionary principle. The precautionary approach developed for the UN Convention on Biological Diversity dictates that "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation". In the context of conservation, often a type of *epistemic* precautionary principle is adopted; in the absence of knowledge or certainty concerning potential detrimental impacts on species and biodiversity we should choose to err on the side of caution to prevent conservation impacts. While is difficult to establish what standard of epistemic due care (and which level of epistemic obligations) are appropriate in which context, it might be perfectly appropriate to have specific standards of epistemic due care for specific issues, for instance where the risk of irreversible or unacceptable impacts are high due to inaction such as biosecurity concerns for the survival of a rare, threatened or endangered species or a specific standard for public health threats such as COVID-19. In practice, however, we often see the epistemic precautionary principle reversed: no action is taken where uncertainty is high or where there is no explicit evidence of impact, possibly resulting from explicit tradeoffs between value systems (e.g., Campbell et al., 2009; Sánchez-Bayo et al., 2017; Meßerschmidt, 2020) or from individual or systemic biases.

A philosophically informed approach to unifying standards of epistemic due care in EBP would also reflect further insights from epistemology, such as the notion of blameworthy ignorance and of the collective nature of much of our knowledge.

In the face of unknown unknowns, it is particularly difficult to determine the extent of our epistemic obligations. In retrospect, we regularly evaluate cases of harm caused (or facilitated) by ignorance by asking whether a particular agent or agency could and should have known the consequences of certain actions and measures. From an ethical perspective then, lack of knowledge is no excuse if agents are culpably ignorant – if their ignorance arises in a negligent or even reckless way, e.g., where they violated accepted epistemic standards in their field of operation. These can be explicit, codified standards, but decision-makers may find themselves at a loss where these standards are inadequate, unsystematic, or completely lacking.

A final observation of how philosophical research can inform our understanding of epistemic standards of due care in EBP can be drawn from research on collective forms of knowledge. Decisions about policy responses tend to get made by (often very diverse) groups of people rather than by individuals. In order for such groups to make informed decision, knowledge has to be distributed in the group in the right way. Often, group members will need to know what others know. This is what philosophers call second-order knowledge: they know (or have beliefs about) what other people know (or believe). Ignorance of facts can obtain at all levels and in many of the above cases there will be an easy remedy; in others it will be very difficult. It is much more difficult to induce higher-order knowledge in larger and dispersed groups (Schwenkenbecher, 2022). Consequently, individual agents' epistemic obligations do not just concern *their own* knowledge, but that of others, too. Or, to put it more clearly: *one person's* epistemic obligations may concern a group's *shared or higher-order* knowledge or beliefs (ibid.).

Biases in acquisition and consideration of evidence

There are a number of internal and external biases that can impact on people's capacity and willingness to collect and appropriately evaluate evidence in the process of policymaking. Our focus here is on the *philosophical* dimensions of such psychological biases.

One famous example is the so-called status-quo bias, which philosophers examine to determine if such biases are failures of rationality (Douglas, 2009; Pauly and Zeller, 2015). Bostrom and Ord (2006) understand status quo bias to be "… an inappropriate (irrational) preference for an option because it preserves the status quo", while Nebel (2015) defines it more neutrally as "a disposition, or tendency, to prefer some state of affairs because it is the status quo" that need not be irrational.

In the previous section we noted a tendency in practice towards inaction when uncertainty is high or explicit evidence of impact is lacking. We suggest that this tendency may be understood as a form of status quo bias. Using this perspective, we can apply existing philosophical analysis of status quo bias and proposed remedies. For example, Bostrom and Ord (2006) suggest a "reversal test" to mitigate status quo bias. In the context of marine biosecurity, this would involve combining the question "should we allocate money to investigate the risks associated with this (potential) invasive species?" with the question "suppose we had already allocated money to investigate such risks; would now be a good time to stop doing so?" If the answer to both questions is "no", it suggests that status quo bias is at work, and the tentative decision not to allocate money should be seriously reconsidered, if not reversed. Or likewise, if the question at hand is "should we allocate money to attempt to eradicate this invasive species?", we should also consider the hypothetical question "if there were a standing effort to eradicate this species, would the current situation be one in which we would be happy to end this effort?" Again, if the answer to both questions is "no", status quo bias appears to be at work, and the case for allocating funds for eradication may be stronger than it is given credit for.

Moving beyond status quo bias, human and economic resource tradeoffs at the operational level may effectively lead to systemic biases against investigating and collecting evidence. One example is the official global fisheries data suggesting catches are increasing or stable, however reconstructed data accounting for a negative bias in reporting suggest fisheries stocks are significantly declining (Pauly and Zeller, 2015). While these may seem mundane, at the bottom of such tradeoffs are always value-based cost-benefit analyses – however informally conducted (Davidson and Hewitt, 2014). The requirement for further investigations to obtain additional knowledge or determine the potential impact of policies may be considered too expensive and unjustified given certain assumptions about the value of the expected outcome. Philosophers can help expose and evaluate the use of such non-epistemic values in science (Douglas, 2009; Elliott, 2017).

Loss aversion and temporal discounting can also be expected to influence what data is collected and what weight it is given. Loss aversion predicts we will weight evidence of loss more heavily than evidence of foregone gains (Kemel and Paraschiv, 2018). Temporal discounting – our tendency to weigh near term effects much more heavily than those that are delayed – has played a significant role in deferring actions associated with resource management challenges to the future including decisions on habitat and species loss and other environmental problems. These can be further exacerbated by Treasury applied discount rates (Ananthapavan et al., 2021). Salience bias and the availability heuristic may play a role in limiting further evidence gathering. In particular, these biases will militate against epistemic actions that might reduce our ignorance, because what we do not know is usually not salient to us.

Discussion

Many challenges to implementing evidence-based policy are not only conceptual but philosophical in nature. These cannot be truly understood, let alone resolved, by using the tools of the natural and the social sciences alone; nor are there simple fixes from philosophy. Rather, an ongoing, trans-disciplinary, collaborative effort to improve our collective understanding of the nature of these problems and the values expressed in opting for certain choices and not for others is much needed.

Author contributions

AS, CH and RH conceptualized, led and wrote the manuscript. AS and CH organized the workshop where all authors participated in ideation. All authors contributed to the article and approved the submitted version.

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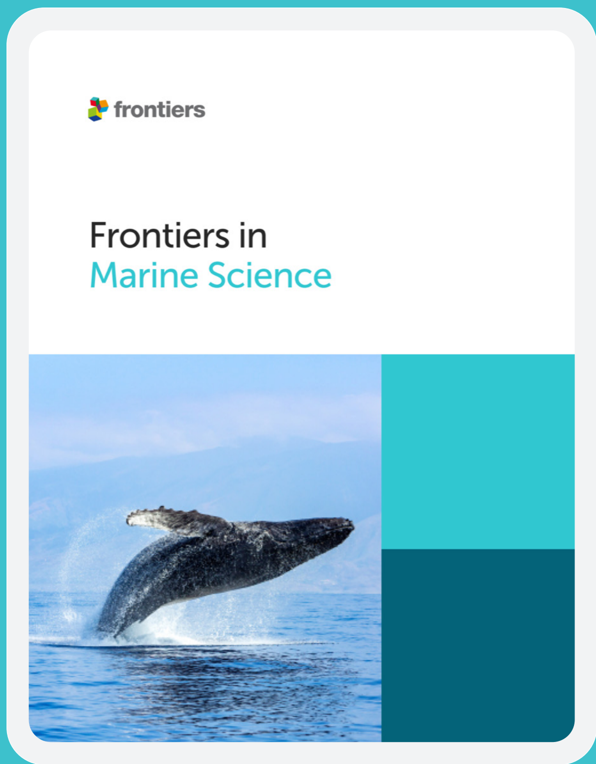

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