REVIEW

Practical implementation of cumulative-effects management of marine ecosystems in western North America

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Abstract
Globally, ecosystem structure and function have been degraded by the cumulative effects (CE) of multiple stressors. To maintain ecosystem resilience, there is an urgent need to better account for CE in management decision-making at various scales. Current laws and regulations are supported by a multitude of frameworks and strategies that vary in application and terminology use across management agencies and geopolitical boundaries. We synthesized management frameworks that accounted for CE in marine ecosystems at the regional and national levels across western North America (Canada, United States, Mexico) to identify similarities and shared challenges to successful implementation. We examined examples of solutions to the identified challenges (e.g., interagency and cross-border partnerships to overcome challenges of managing for ecologically relevant spatial scales). Management frameworks in general consisted of 3 phases: scoping and structuring the system; characterizing relationships; and evaluating management options. Challenges in the robust implementation of these phases included lack of interagency coordination, minimal incorporation of diverse perspectives, and data deficiencies. Cases that provided solutions to these challenges encouraged coordination at ecological rather than jurisdictional scales, enhanced involvement of stakeholders and Indigenous groups, and used nontraditional data sources for decision-making. Broader implementation of these approaches, combined with increased interagency and international coordination and collaboration, should facilitate the rapid advancement of more effective CE assessment and ecosystem management in North America and elsewhere.

KEYWORDS
Canada, ecosystem-based management, environmental impact assessment, Mexico, multiple stressors, United States

Implementación Práctica del Manejo de los Efectos Acumulativos de los Ecosistemas Marinos en el Oeste de América del Norte

Resumen: A nivel mundial, la estructura y función de los ecosistemas ha sido degradada por los efectos acumulativos (EA) de varios estresantes. Para mantener la resiliencia de los ecosistemas existe una necesidad urgente por explicar de mejor manera los EA en la toma de decisiones de manejo a varias escalas. Las leyes y normas actuales están respaldadas por una multitud de marcos y estrategias que varían en el uso de aplicaciones y terminología a través de las agencias de manejo y las fronteras geopolíticas. Sintetizamos los marcos de manejo para los EA en los ecosistemas marinos a nivel regional y nacional por todo el oeste de América del Norte (Canadá, Estados Unidos y México) para identificar las similitudes y los obstáculos compartidos para una implementación exitosa. Analizamos ejemplos...
de las soluciones a los obstáculos identificados (p. ej.: asociaciones interagencias o transfronterizas para sobrepasar los retos en el manejo para escalas espaciales ecológicamente relevantes). Los marcos de manejo consistieron generalmente de tres fases: exploración y estructuración del sistema; caracterización de las relaciones; y evaluación de las opciones de manejo. Los obstáculos para la implementación robusta de estas fases incluyeron la falta de coordinación entre agencias, la incorporación mínima de perspectivas diversas y deficiencias en la información. Los casos que proporcionaron soluciones a estos obstáculos fomentaron la coordinación a escalas ecológicas en lugar de sólo hacerlo en escalas jurisdiccionales, mejoraron la participación de los actores y los grupos indígenas y usaron fuentes de datos no tradicionales para la toma de decisiones. Una implementación más generalizada de estos enfoques, combinada con el incremento en la coordinación y colaboración interagencias e internacional, debería facilitar el rápido avance de evaluaciones más efectivas de los EA y del manejo de ecosistemas en América del Norte y en otras partes del mundo.

PALABRAS CLAVE
Canadá, Estados Unidos, estresantes múltiples, evaluación de impacto ambiental, manejo basado en el ecosistema, México

INTRODUCTION
Most of the world’s marine ecosystems are threatened by a combination of local, regional, and global stressors (Lotze et al. 2006; Halpern et al. 2008). Managing the cumulative effects (CE) of multiple stressors is paramount and one of the greatest challenges of the modern era. The difficulty arises in part because ecosystems can respond in a variety of ways to individual stressors and in part because stressors can interact to produce a variety of ecosystem responses. For example, ecosystems can exhibit either linear or nonlinear changes in state or function as stressor intensity or duration increase, and this variation is currently not well understood (Schepfer et al. 2001; Burkett et al. 2005; Hunsicker et al. 2016). Moreover, when local, regional, or global stressors co-occur spatially, their effects on ecosystem state or function may be additive or nonadditive (i.e., the response to the interacting stressors may be greater or less than the sum of individual stressor responses) (Côté et al. 2016). Currently, the nature of multiple stressor interactions cannot be predicted (Côté et al. 2016; Schäfer & Piggott 2018); yet, an approximate understanding of nonlinearities, stressor interactions, and associated uncertainties is needed to assess CE and identify management options.

As the number and intensity of local and global stressors mount due to growing human populations and unequal resource use, especially in the coastal zone, governments around the world are increasingly requiring that decisions about management or development projects assess how human activities will contribute to CE (e.g., Appendix S1; Korpainen & Andersen 2016). These assessments are generally known as cumulative-effects assessments (CEA) and are used to evaluate how multiple human activities within a defined temporal and spatial scale may interact to affect a target species or ecosystem. A CEA is often a core element of ecosystem-based management (EBM)
Cumulative effects assessment (CEA) is a process used in management decisions to consider multiple pressures and impacts across environmental and developmental activities. It was developed in response to difficulties inherent in managing ecosystem responses to novel conditions (Judd et al. 2015). The result has been a proliferation of CEA definitions, management frameworks, methods, and terminologies designed to address differing needs and risk thresholds of those making management decisions. This diversity of approaches has hampered cross-jurisdictional CE management (Hodgson et al. 2019; Foley et al. 2021).

Previous reviews of CEA methods either narrowly focused on key considerations for specific types of developments (e.g., marine renewable energy developments [Willsteed et al. 2017]) or broadly compared variation in methodological approaches across oceans and development projects (e.g., Korpinnen & Andersen 2016; Hodgson and Halpern 2019). We focused on a defined geographic area and ecological zone, which allowed us to delve deeper into regionally implemented methods, challenges, and solutions. We identified challenges and solutions through a review of the CEA literature, conversations with CEA practitioners, and personal experiences with the conservation instruments in the focal regions. Our personal experiences include work with managers and stakeholders in the Salish Sea and California to model multiple stressors on kelp and salmon with driver–pressure–state–impact–response (DPSIR) and pathways-of-effects models and work with the Mexican Comisión Nacional de Áreas Naturales Protegidas (CONANP) on marine protected areas (MPAs) in the Mexican Pacific (J.A.H.); identification of ecologically or biologically significant marine areas in Canada, especially as they pertain to invasive species and high-use marine areas with multiple interacting stressors (T.W.T.); and contributions to the development of frameworks for the implementation of Oceans Act MPAs and MPA networks in Canada, the critical evaluation of impact assessments for coastal and riverine development projects, and the consideration of multiple pressures in pathway-of-effects models (I.M.C.). We sought to provide a step-by-step guide to CEA to achieve a variety of management goals, not to specify an approach that fulfills a specific legal mandate. Thus, our approach is independent of whether the user is motivated by legal requirements and highlights shared approaches, challenges, and solutions that can create a basis for cooperation and mutual understanding that ultimately enhance the usefulness of and buy-in to the assessment.

We focused our assessment on Canada, the United States, and Mexico, which steward a continuous coastline of more than 8000 km. Although each country manages marine resources and development projects in its own territorial waters independently via a mix of national and local agencies, Indigenous governments, and nongovernmental organizations, the extensive connectivity—ecologically and from transboundary commercial activities—and similarity of nearshore species and temperate ecosystems suggest the need for common CEA methods and terminology (Judd et al. 2015; Korpinnen and Anderson 2016; Hodgson et al. 2019). We focused on management and assessment efforts in the nearshore marine zone, which lies at the confluence of stressors originating from the open ocean, human use of the coastal zone, and upland changes. Considering that ocean, coastal, and upland zones are highly interconnected yet often managed by different jurisdictions, the nearshore provides an opportunity to investigate how different groups approach decision-making, interorganizational communication, and ultimately CEA.

We first examined some of the legal mandates underpinning CEA in Canada (British Columbia [BC]), the contiguous United States (Washington, Oregon, and California), and the Gulf of California states of Mexico (Baja California and Baja California Sur) and then compared the main CEA frameworks used in environmental CE decision-making. We identified shared challenges and locally implemented solutions for management that account for CEs on the scales of ocean basins to estuaries, with the goal of improving overall application and effectiveness.

**LEGAL MANDATES**

Although we focused on broad approaches to CEA that transcend individual legal mandates, these mandates provide an important context for the current state of affairs across the focal jurisdictions. Legislation mandating CEA has varied definitions of cumulative effects but generally states that CEs include the direct and indirect effects of human actions on scales relevant for the action, ecosystem, or species being assessed. Across western North America, environmental legislation at national and regional levels is increasingly including CEA mandates, though the target for the assessment is still often separated into fisheries, MPAs, species of special interest (e.g., endangered species and marine mammals), nearshore development, forestry, and so forth (Figure 1). Furthermore, there is increased recognition of the need to include climate-change impacts (e.g., Squillace & Hood 2012; Gissi et al. 2021). Under the Canadian Impact Assessment Act (2019) (previously the Canadian Environmental Assessment Act 2012), the U.S. National Environmental Protection Act (NEPA, 1970), and the Mexican Law of Ecological Equilibrium and Environmental Protection (Ley General de Equilibrio Ecológico y Protección Ambiental [LEGEEPA] 1988), projects requiring federal permits must conduct a CEA. These federal laws are complemented by a variable suite of additional legislation at the state or provincial level (Appendix S1). The CEA process has been criticized in all 3 countries for failing to live up to the intent of the laws to effectively incorporate CE into management decision-making (see “SHARED CHALLENGES”).

**FRAMEWORKS**

A multitude of frameworks guide the process of accounting for CE in management decisions, ranging from EBM frameworks intended to be broadly applied to diverse situations (e.g., Levin et al. 2009) to those developed to address a specific question in a specific ecosystem (e.g., McClanahan & Cinner 2008). Despite differences in intent and terminology, CEs in the 3 countries have 3 phases: first, scoping and structuring the
In some frameworks, including the U.S. integrated ecosystem assessment (Levin et al. 2009), this initial phase explicitly includes extensive stakeholder and Indigenous input. When implemented meaningfully, this collaborative and transparent approach leads to successful EBM plans and outcomes (e.g., Gaichas et al. 2018, but see “SHARED CHALLENGES”). In other frameworks, including most environmental assessment processes (e.g., Canadian Environmental Assessment Act [2012] and U.S. National Environmental Protection Act), this initial phase loosely encourages a collaborative approach but does not specifically require it. Depending on the project, the approval process for projects in Mexico may require input from relevant experts and the general public, such as is the case for establishing new national or state MPAs. In contrast, establishing protected areas in the national marine terrestrial zone, which includes beaches, lagoons, and estuaries, requires no consultation with the public or relevant experts (Koch 2015).

Once the problem, goals, and ecosystem components of interest have been defined, frequently the next step is to create a conceptual causal network that illustrates the connections among stressors and ecosystem components. These networks are known as pathways of effects models in Canada (Government of Canada 2012), drivers–pressures–state–ecosystem services–responses (DPSER) models in the United States (Kellbe et al. 2013), and pressure–state–response networks in Mexico (SEMARNAT 2020). All are closely related to the DPSIR model, which is widely used in Europe (Smeets & Wetttings 1999; Figure 2). The exact steps and the terminology used to describe each step vary, but the structures and components are readily comparable and outline linkages among human

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**FIGURE 1** A subset of lead agencies and federal legislation (in parentheses) that apply to cumulative effects on marine ecosystems in western North America, with the exception of nearshore logging in Canada, which is regulated at the provincial level (asterisks indicate legislation that includes cumulative-effects assessment). Art is by Eleanor Schiffler.
activities, environmental changes that affect ecosystem components of interest, and management responses to mitigate the pressures (Figure 2).

**Phase 2: Characterizing relationships**

Once the structure of the system and scope and goals of the assessment have been clearly defined, the relationships between the components of the system are evaluated, often via a risk assessment. The goal of a risk assessment is to assess the likelihood of undesirable events and their impacts (Holsman et al. 2017). Risk assessments can distinguish between an event with a low likelihood of occurrence but a high impact and an event with a high likelihood of occurrence but a low impact: situations that often require different management strategies. Many frameworks for risk assessment have 3 levels: qualitative rapid assessment (level 1), semiquantitative prioritization (level 2), and quantitative calculations of risk (level 3) (Hobday et al. 2011; Levin et al. 2014; O et al. 2015) (Appendix S2). Levels can be treated hierarchically, such that components identified in phase 1 are whittled down to focus on the species or ecosystem targets at greatest risk that are thus prioritized for management or mitigation actions. Levels can also be targeted specifically depending on resources, data availability, time lines, and project goals, among other considerations. Approaches for phase 2 fall into 3 broad and overlapping categories of risk assessment: multiplicative and Euclidean models, network models, and spatial models (Appendix S2).

Among national agencies, DFO, NOAA, and CONANP, guidelines for conducting risk assessments to support management decision-making in marine systems are provided. On the west coast of Canada, DFO uses the ecological risk assessment framework (ERAF; O et al. 2015). In the United States, NOAA’s risk assessment is based on the integrated ecosystem assessment (IEA) framework, which builds on the ecological risk assessment for the effects of fishing approach (Levin et al. 2008; Hobday et al. 2011; reviewed in Holsman et al. 2017). In Mexico, CONANP, the agency tasked with managing Mexican national parks and MPAs, uses a scorecard approach for rapid assessment of 3 specific themes: water, habitat, and wild resources. For creating environmental impact assessments, risk assessments for a given activity are only required if impacts to the environment from the proposed activity are likely to be significant (LGEEPA 2000; CEAA 2017; CEQ 2007) (Appendix S3).

**Phase 3: Evaluate management options**

Once risk has been assessed and certain pressures or targets have been highlighted as at greatest risk, then management decisions must be made regarding how to reduce or eliminate risk. The approach for evaluating management trade-offs may
depend on the level of analysis used in the previous phase to assess risk.

For assessments based primarily on qualitative rapid assessment (level 1), management scenarios may be evaluated using expert and stakeholder opinion (CONANP 2016), although here a wide diversity of perspectives is especially important to prevent an outcome biased toward one sector (e.g., conservation vs. resource extraction) (Krupnick & Gordon 2015). Options may also be evaluated by simple ranking of risk; management focuses on mitigating local pressures on the highest risk targets or on altering local pressures that drive the highest cumulative risk (Battista et al. 2017). For semiquantitative (level 2) and quantitative (level 3) risk assessments, management scenarios can be tested by running hypothetical scenarios through the constructed models and observing the probable results of various actions (e.g., Reum 2015; Punt et al. 2016; Surma et al. 2018). Regardless of the method used, mitigation efforts often focus on minimizing the likelihood of occurrence rather than the impact in order to decrease overall risk (O et al. 2015).

In EIAs, managers or decision makers at the lead agency must decide among the proposed and alternative actions provided in the assessment, including a no-action option, after soliciting input from the public. Decisions are based on trade-offs of economic, social, and ecological risk and benefit, the possibility of future applications and setting precedents, and other relevant policies and political priorities (LGEEPA 2000; CEQ 2007; CEAA 2017).

Ideally, phase 3 is iterative. Indicators are monitored after a chosen management strategy is implemented, and the success of that strategy is assessed and refined based on specific and measurable outcomes. This approach, known as adaptive management, is the cornerstone of NOAA’s IEA (Harvey et al. 2017) and is being used increasingly in Mexican regional planning (Wachtel 2018). However, adaptive management is not a required aspect of project plans under NEPA, CEAA, or LGEEPA, and monitoring of a given risk is only required if it was deemed significant and monitoring was explicitly stated in the approval decision (LGEEPA 2000; CEQ 2007; CEAA 2018; Emerson & Baldwin 2019). Implementation of adaptive management is also vulnerable to challenges similar to those of CEA, such as inconsistent terminologies and definitions used across projects, undefined scope and scale, lack of engagement with diverse constituents, undefined success and uncertainty, and requirement to work within legal and institutional constraints (Rist et al. 2013; Benson and Stone 2013; Williams 2011).

**SHARED CHALLENGES**

Despite CEA requirements being written into law decades ago, significant challenges still remain for its effective implementation (Baxter et al. 2001; Canter & Ross 2010; Foley et al. 2017; Harvey et al. 2017). Persistent hindrances to effective CEA in western North America include challenges in defining scale, incorporating diverse perspectives, and making decisions in data-poor systems or when access to data is limited (Table 1). Many of these hindrances are exacerbated by the complex patchwork of legislation (Figure 1), which is driven by changing political priorities that affect available resources, project time lines, engagement strategies, and willingness to share data. The power of political will to shape how environmental impacts are characterized and managed is exemplified by previous administrations in Canada, the United States, and Mexico during which environmental protection regulations and legislation and scientific and public input into management decisions were eroded or reduced (Barnett and Wiber 2019; Hejny 2018; Balderas Torres et al. 2020).

**Defining scale**

Spatial and temporal scale can be determined based on many factors, including the estimated scale of a pressure (i.e., the footprint of a development project) and the jurisdictional boundaries of an agency. In a survey of resource managers across multiple countries, Foley et al. (2017) found that nearly 40% of practitioners defined the scale of the analysis based on jurisdictional boundaries. The sectioning of modern jurisdictions into land, rivers, and oceans, based on geopolitical boundaries rather than ecological boundaries (e.g., watersheds or migration pathways), can present a major barrier to effective CEA (Hodgson et al. 2019; Figure 1). Pacific salmon (*Oncorhynchus* spp.), for example, move through many different ecosystems, jurisdictions, and political boundaries over the course of their lives. As stocks continue to decline over much of their range, many are pointing to the CE of a multitude of stressors as the culprits of this decline, though management actions continue to be narrow in scale in most jurisdictions (Price et al. 2008; Lacy et al. 2016; Hodgson et al. 2020). Like spatial scale, the specified temporal scale can also greatly influence the conclusions of a CEA, but notably, Korpinen and Anderson (2016) found that no published CEA contains acknowledgment of or includes historical modifications to the environment. Most often the process of EIA defers to the smallest temporal or spatial scale in the assessment of cumulative risk or it is otherwise limited to the impacts from a given project, rather than examining how multiple projects in an ecologically relevant area or time span may interact (Ma et al. 2012; Mach et al. 2014; Clarke Murray et al. 2018). These narrow definitions of spatial and temporal scale present significant hindrances to effective CEA, which by most definitions should look at the relationship of a pressure with past, present, and future conditions and at scales relevant to the species, habitat, or ecosystem in question.

**Diverse perspectives**

Also important in determining the scale, and thus the scope, of a project is who contributes to that decision-making. Although expert opinion is a critical tool used for assessing risk (phase 2), input from diverse experts, Indigenous groups, and stakeholders is less frequently sought while structuring the system.
Challenges: Some challenges in cumulative-effects management and solutions implemented by national and regional governing bodies across the west coast of North America.

### Table 1

| Challenge                              | Potential solution                                         | Example implementation                                                                 |
|----------------------------------------|------------------------------------------------------------|----------------------------------------------------------------------------------------|
| Ecologically relevant temporal and spatial scales | Interagency and cross-border partnerships and coordination | Large ocean management Areas (Can.)<br>Regional planning bodies (US)<br>Marine plan partnership (BC, Can.)<br>Delta Stewardship Council Interagency Adaptive Management Integration Team (CA, US)<br>Puget Sound Partnership (WA, US)Partnerships with NGOs and nonprofits |
| Incorporating diverse perspectives      | Decision-making power at the local level                   | Comanagement between Indigenous groups and federal or state or provincial government<br>Marine resources committees (WA, US)<br>Fishing cooperatives (BC and BCS, Mexico) |
| Low availability of quantitative data  | Statistical and numerical models using expert and community knowledge | Statistical modeling with expert knowledge (Bayesian belief networks, qualitative network models, fuzzy cognitive maps)<br>Qualitative risk assessments<br>Collaborative modeling (CA, US and BC, Can.) |

Abbreviations: BC, British Columbia; CA, California; Can, Canada; US, United States; WA, Washington.

(Phase 1) (Baxter et al. 2001; Clarke Murray et al. 2018; Clarke Murray et al. 2020; but see Levin et al. 2009). However, most of this first scoping phase consists of value judgements: what is important in a region or ecosystem, what represents appropriate proxies for these valuable components, what is an acceptable level of change, what baselines should be used, and what spatial or temporal scale is appropriate. Interagency communication can facilitate cooperation across jurisdictions so that the scope of the project is ecologically meaningful; expert involvement can improve understanding of the current knowledge of ecosystem processes or vulnerability; and input from Indigenous groups and stakeholders can reveal the priorities of the local communities that may be directly or indirectly affected by management decisions. Seeking different perspectives involves additional resources, but the result of the CEA can be dramatically different depending on which level of communication is included. For example, the selection of valued ecosystem components and definitions of limits of acceptable change anticipated from a natural gas pipeline project in British Columbia differed greatly between the project proponent’s experts and between the people of the 2 First Nations consulted (Joseph et al. 2017).

Incorporating the priorities and perspectives of relevant stakeholders and communities is especially lacking in the process of EIA across western North America. A survey of 10 environmental impact statements from 2010 to 2014 in British Columbia revealed that no projects used a collaborative approach to determine the scope and thus the significance of risk (Clarke Murray et al. 2018). Proposals for 27 water- and energy-related projects seeking approval under NEPA from 2012 to 2017 did not change substantively in response to public comments, suggesting that stakeholder input contributed little to the approval process and that consultation was not meaningful (Ulibarri et al. 2019). In Mexico, projects are not required to consult with local communities or to provide documentation translated into local languages, which can present a huge barrier to engagement in rural communities in which literacy may be low and most people speak an Indigenous language (Palerm & Aceves 2004).

### Data availability

An additional and related challenge often cited as a hindrance to effective CEA is the availability of data relevant to the management or EIA needs (Leslie & McLeod 2007; Harvey et al. 2017; Clarke Murray et al. 2018). The scope of the project determined in phase 1 can be influenced by available data, including the spatial or temporal resolution of data, available baselines, projected significance of impacts, and availability of input from experts. Available data can also determine in phase 2 whether ecosystem responses to stressors should be represented as linear or non-linear and whether stressor interactions should be represented as synergistic, antagonistic, or additive. Because there is often a paucity of data in complex systems, most CEA are based on the assumption that interactions among stressors are linear and additive (Halpern & Fujita 2013; Korpinen and Andersen 2016), even if these types of interactions are in the minority (Côté et al. 2016; Hunsicker et al. 2016). Simulation models show that management actions based on the assumption of additive effects can be more damaging than no action when the underlying multiple stressor interactions are actually antagonistic (e.g., Brown et al. 2013). Lack of data or high uncertainty in those data can also be used to prematurely conclude that impacts are not significant in EIA, which can result in a dramatic underestimation of risk to a system (Clarke Murray et al. 2018).

### Implemented solutions

Although the barriers to management that effectively incorporates CE are significant, some approaches used in western North America and more broadly overcome some of the challenges identified above. These include policies for interagency...
cooperation so management can be defined at ecologically relevant scales, strategies increase participation of and collaboration with stakeholders and Indigenous groups, and tools to make effective decisions are based on diverse data sources (Table 1).

Cross-boundary communication

The amount of area that must be covered to effectively conduct CE management depends on the region, pressures, and goals. At the largest scale, useful for managing wide-ranging fisheries or networks of MPAs, the Canadian and U.S. federal governments have created ecologically defined large regional planning areas. These are known as Large Ocean Management Areas in Canada (Oceans Act 1997) and Regional Planning Bodies in the United States (U.S. Executive Order 13547 [https://obamawhitehouse.archives.gov/the-press-office/executive-order-stewardship-ocean-our-coasts-and-great-lakes]). These planning areas are intended to increase interagency communication in these regions and have resulted in improved implementation of EBM principles in regional risk assessments (Marshall et al. 2017), fisheries management (Gaichas et al. 2018), and marine spatial planning (Clarke Murray et al. 2015). Long-term collaborative partnerships can also be built around these large regions, such as the Marine Plan Partnership (MaPP) in British Columbia. The MaPP is a collaboration between the provincial government and 16 First Nations to create EB plans based on traditional knowledge and current data (MaPP, 2021).

These principles of communication and ecologically defined boundaries also apply to regional scales, such as watersheds, delta or river systems, or semienclosed seas. To address CE at these scales, California and Washington have state-funded groups tasked with coordinating efforts to increase the sustainability of specified water bodies. In California, the San Francisco Bay Delta is part of a large watershed that spans multiple management jurisdictions across which water resources are divided among food production, urban centers, recreation, and protected species or ecosystems. These water resources are very vulnerable to drought and flood cycles that will likely be exacerbated by climate change (Hanak and Lund 2008), making adaptive management for CE critical. To address this increasingly urgent need for interagency cooperation, the Delta Stewardship Council has led the formation of an Interagency Adaptive Management Integration Team (Delta Stewardship Council 2019) that brings together most federal and state agencies in the delta to collaboratively provide feedback on project design and permitting. This approach facilitates communication among agencies that may have differing priorities and allows projects to get rapid and diverse feedback. In Washington, the Puget Sound Partnership is a state agency tasked with bringing together the many management groups, nonprofit organizations, Indigenous groups and other stakeholders, and researchers to facilitate adaptive management and EBM in the region (Puget Sound Partnership 2010).

In some cases, nongovernmental or international organizations can serve similar roles as the aforementioned agencies and enhance communication among relevant parties. In partnership with the California Department of Fish and Wildlife, The Nature Conservancy led the stakeholder collaboration phase of the creation of the Red Abalone Recovery and Management Plan (Jackson 2020). In Mexico, CONANP partnered with the German federal government to use some of Germany’s resources and expertise to create a regional sustainability plan for the Gulf of California marine ecosystems that incorporated a regional perspective on pressures from fisheries and development (Wachtel 2018).

However, for examples of marine management partnerships that span national borders one has to look to other countries and regions. The Baltic Sea in Europe has many parallels to the Salish Sea in North America—both are semienclosed bodies of water shared by multiple countries and fed by watersheds that are patchworks of heavily urbanized areas, agricultural land, and forest. Although management of the Salish Sea remains separate between the United States and Canada, the overarching governing body of the European Union granted a legal and logistical platform for international cooperation in managing CEs affecting the Baltic Sea (Bergström et al. 2019). A joint approach to CEA has allowed for more efficient sharing of tools and data, agreement on shared definitions and approaches, and monitoring and evaluation plans for the future that will be readily comparable across the participating countries (Bergström et al. 2019). In North America, the Great Lakes Fisheries Commission provides an example of international cooperation for management in a freshwater system. Originally conceived to address the impacts of invasive lampreys (Petromyzon marinus), the commission now coordinates fisheries research across all U.S. states, Canadian provinces, and Indigenous nations that border the Great Lakes (Great Lakes Fishery Commission 2020).

Community-based management

Another approach to EBM and CEA is to empower local communities and stakeholders to participate more fully in and guide the management process, which can include planning, implementation, and monitoring. Iterations of this approach include giving increased power and sovereignty to Indigenous groups over the management of their traditional lands and waters, alternative marine management areas based around local communities, and cooperatives that communally manage a shared resource. Small-scale, locally managed fisheries and protected areas have been shown to be resilient to disturbance, more likely to implement principles of adaptive management, and more likely to achieve desired socioecological management outcomes, although this topic warrants continued investigation (Sayce et al. 2013; McCay et al. 2014; Ban & Frid 2018; Fulton et al. 2018). Similar improvements to environmental outcomes when resources are governed collaboratively have been observed in the water quality of rivers (Scott 2015), a range of terrestrial case studies in North America (Newig and Fritsch 2009), and resource management outside North America (Agrawal and Chhatre 2007; Dodson 2014). In an analysis of over 300 published studies of collaborative management cases, Jager et al.
Embracing diverse data sources

Although data scarcity has been used to justify inaction or maintenance of the status quo (Clarke Murray et al. 2018), novel modeling techniques are increasingly allowing meaningfully incorporation of qualitative data, such as expert opinion. Bayesian belief networks (BBNs), qualitative network models (QNM), and fuzzy cognitive maps (FCMs) are all forms of network models that can be constructed using quantitative or qualitative data and have been used to weigh management options in data-poor nearshore marine systems. For example, Stafford et al. (2016) used BBNs to assess the potential effects of conservation and management actions in and out of marine reserves off mainland Ecuador; Reum et al. (2020) used QNMs to evaluate management options for rebuilding the collapsed blue king crab fishery in the northeast Pacific; and Kontogianni et al. (2012) used FCMs to identify CE and weigh management options to enhance Black Sea ecosystem resilience in Ukraine. Many other classes of models based on expert solicitation exist beyond these 3 examples. It remains important to choose the right model to fit a given system, available data, and question and to be aware of the strengths and limitations of each.

The benefits of using alternative modeling approaches can go beyond the goal of making a decision given insufficient quantitative data. Inviting and meaningfully using additional data sources, namely, the expert opinions of Indigenous groups, industry stakeholders, and other users or stewards of an ecosystem, can result in improved management outcomes through increased support of the process and results and potentially different outcomes of the CEA itself. For example, in California the process to designate new MPAs was highly collaborative, involving extensive input from and engagement with many different communities of nearshore marine users (Sayce et al. 2013). The result was increased support from the public and increased transparency of the decision-making process (Weible et al. 2004). In central Canada, a collaborative BBN was developed to determine the downstream impacts of mining;
Western science and Indigenous knowledge were incorporated in a participatory process of “two-eyed seeing,” a Mi’kmaw framework of knowledge coexistence between Western science and Indigenous knowledge (Mantyka-Pringle et al. 2017; Reid et al. 2021). Importantly, the probabilities that aspects of the ecosystem had changed due to upstream mining differed between the model built with quantitative data versus that built with qualitative data. Considering that both forms of data are prone to biases, whether based on defined scope (see “Defining scale”), selected indicators (see “Diverse Perspectives”), or the personal biases of participating experts, models that are built with multiple knowledge sources may lead to results that better account for these biases and resulting uncertainties (Reid et al. 2021), which can be especially important when considering the CE of multiple stressors.

CONCLUSION

Meaningful assessment of the cumulative effects of human activities on ecosystems is imperative for the current and future well-being of society and the natural world. Patchwork legislation and agency-specific strategic frameworks exist in countries and municipalities along the west coast of North America. However, significant barriers to effective CEA remain, including incorporating diverse perspectives across multiple levels of society, government, and jurisdictional and geopolitical boundaries; accessing appropriate data and technical expertise; and implementing adaptive management beyond narrow administrative time lines. Even the high variability in language used to describe CEA frameworks across agencies and jurisdictions can make interagency and cross-border communication and collaboration difficult, highlighting the need for explicitly defined shared terminology (Hodgson et al. 2019). By identifying the similarities in commonly used national CEA frameworks and highlighting regional efforts to enhance communication and diverse community involvement in the CEA process, we hope our synthesis can provide a roadmap for regional groups beginning the CEA process and serve as a basis for communication and common language for national agencies planning to extend CEA beyond jurisdictional and geopolitical boundaries.

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