Developing Ecosystem Indicators for Responses to Multiple Stressors

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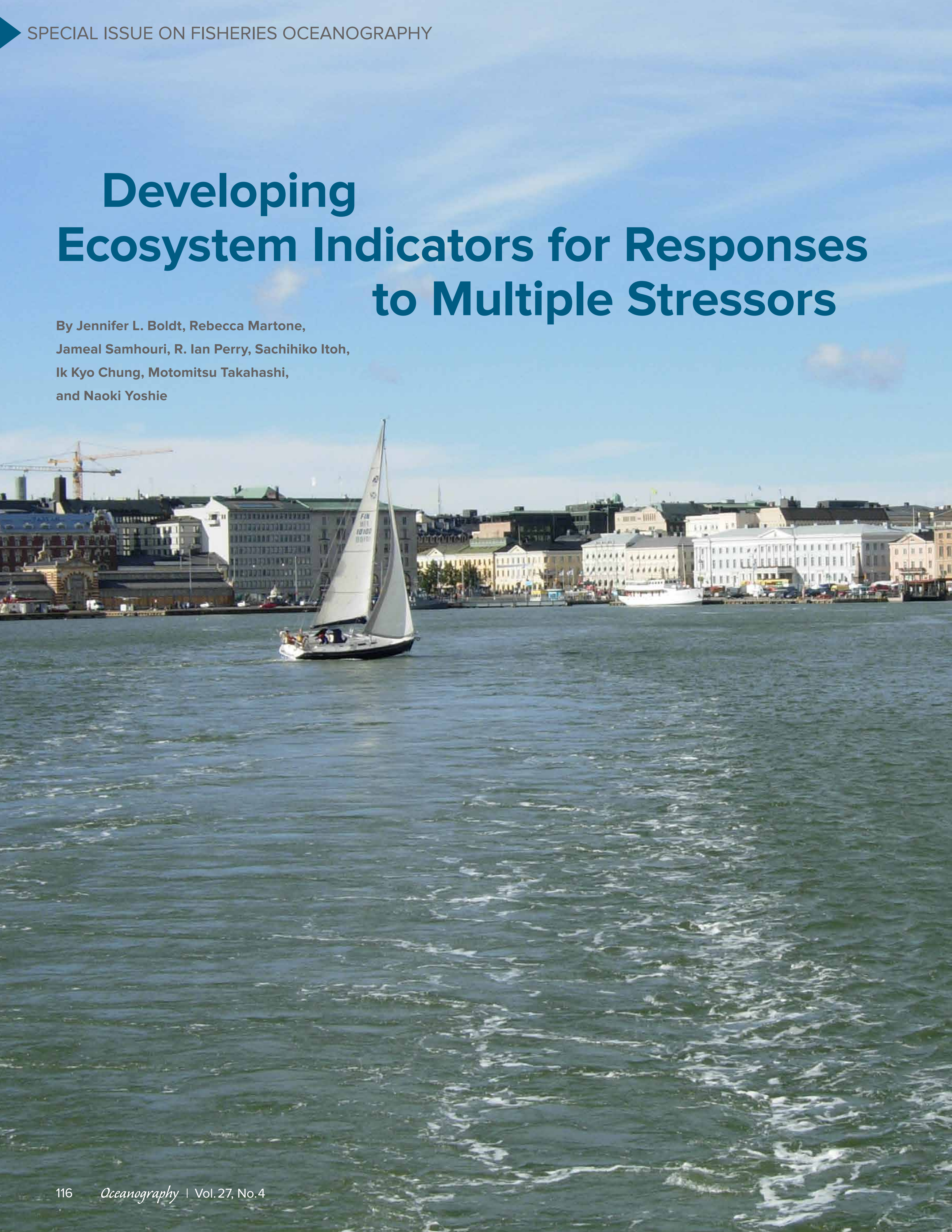

ABSTRACT. Human activities in coastal and marine ecosystems provide a suite of benefits for people, but can also produce a number of stressors that can act additively, synergistically, or antagonistically to change ecosystem structure, function, and dynamics in ways that differ from single stressor responses. Scientific tools that can be used to evaluate the effects of multiple stressors are needed to assist decision making. In this paper, we review indicator selection methods and general approaches to assess indicator responses to multiple stressors and compare example ecosystem assessments. Recommendations are presented for choosing and assessing suites of indicators to characterize responses. Indicators should be chosen based upon defined criteria, conceptual models linking indicators to pressures and drivers, and defined strategic goals and ecological or management objectives. Indicators should be complementary and nonredundant, and they should integrate responses to multiple stressors and reflect the status of the ecosystem. An initial core set of indicators could include those that have been tested for the effects of climate and fishing and then expanded to include other pressures and ecosystem-specific, feature-pressure interactions. Identifying indicators and evaluating multiple stressors on marine ecosystems require a variety of approaches, such as empirical analyses, expert opinion, and model-based simulation. The goal is to identify a meaningful set of indicators that can be used to assist with the management of multiple types of human interactions with marine ecosystems.

INTRODUCTION

Globally, research organizations are focusing on the need to provide science advice to marine management clients on a broad range of issues under changing environmental conditions (e.g., NOAA, 2006; DFO, 2007; ICES, 2013). Scientific support is required for ecosystem-based management of the diverse range of human activities and ocean use sectors. To address this need, various approaches and frameworks, such as integrated ecosystem assessments (IEA) and risk-based assessments, have been developed to assess ecosystems and potential risks to valued ecosystem components (e.g., Levin et al., 2009; DFO, 2012; Dickey-Collas, 2014; Levin et al., 2014; Link and Browman, 2014; Samhouri et al., 2014). A goal of these approaches is to integrate scientific understanding into management measures and into the development of conservation objectives (Levin et al., 2009; DFO, 2012; Borja et al., 2013). In addition, IEAs and other frameworks should facilitate exploration of decision making and policy options that can contribute to weighing trade-offs among various environmental, social, and economic objectives (Dickey-Collas, 2014).

A broad range of human activities across a wide array of coastal and marine systems provides a suite of benefits for people. Much valuable research has focused on understanding the effects of single stressors such as fishing or climate on fisheries resources (e.g., Megrey et al., 2007; King et al., 2011). Human activities, however, can produce a number of stressors (also sometimes referred to as pressures) from both land and sea that can impact the surrounding environment simultaneously (e.g., sedimentation, nutrient input, contaminants, shading, noise; Smeets and Weterings, 1999; Knights et al., 2013). Multiple stressors can act additively, synergistically, or antagonistically to change ecosystem structure, function, and dynamics in unexpected ways that differ from single stressor responses (Adams, 2005; Crain et al., 2008; Darling and Cote, 2008; Halpern et al., 2008; Ban et al., 2010, 2014; Micheli et al., 2013). Cumulative effects can result from the incremental, accumulating, and/or interacting impacts of an activity and its stressors on habitats and species (Hegmann et al., 1999). In order to fully account for the cumulative effects on coastal and marine ecosystems that arise from multiple human activities and their associated stressors, scientists and managers must be able to understand: (1) the stressors caused by activities; (2) the magnitude, frequency, and spatial scale at which the activities occur; (3) the resulting direct and indirect cumulative effects; and (4) the responses of multiple interacting ecosystem components.

Addressing all changes in an ecosystem is complex. Establishing causal relationships between stressors and observed effects in natural systems is difficult due to: (1) biotic and abiotic factors that can modify responses of biota to stressors (McCarty and Munkittrick, 1996), (2) compensatory mechanisms that operate in populations (Power, 1997), (3) time lags between cause and effect (Vallentyne, 1999), (4) multiple pathways by which stressors can disrupt ecosystem functions, and (5) potentially spurious correlations between stressors and observed effects. The complexity of marine ecosystems, their high variability and nonstationarity, and the broad array of activities that may impact aspects of these ecosystems suggest that no single measure is adequate for assessing the effects of multiple stressors. Thus, there is a need to identify suites of ecosystem indicators that can be used to provide an understanding of how coastal and marine ecosystems respond to multiple stressors.

Various tools and approaches have been and are currently being developed to characterize ecosystem responses to multiple stressors and cumulative impacts (e.g., Levin et al., 2009; Ban et al., 2010; Halpern et al., 2012). The focus of this paper is to review indicator selection methods as well as general approaches that have been used to assess indicator responses to multiple stressors. We compare and contrast example ecosystem assessments to identify similarities and differences in the pressures and indicators selected and how responses to multiple stressors were addressed. Finally, we conclude with recommendations for identifying suites of indicators and approaches for assessing indicator responses to multiple stressors.

WHAT ARE INDICATORS?

Indicators are useful tools because it is not possible to measure everything in a complex, dynamic ecosystem. In the scientific literature, indicators are defined in several ways (OECD, 1999, 2003; Jackson et al., 2000; Dale and Beyeler, 2001; Kurtz et al., 2001; Carignan and Villard, 2002). Hayes et al. (2012) succinctly summarized the definitions and identified two key properties of indicators: (1) "components or processes of the ecosystem that can be measured in order to tell us something about the impacts of anthropogenic activities on the health or sustainability of the system", and (2) "reduce the complexity of real-world systems to a small set of key characteristics that are useful for management and communication purposes." Additionally, indicators reflect changes taking place at various levels, from genes to species to regions (Dale and Beyeler, 2001). This is captured in Niemi and McDonald's (2004) definition of indicators as: "measurable characteristics of the structure (e.g., genetic, population, habitat, and landscape pattern), composition (e.g., genes, species, populations, communities, and landscape types), or function (e.g., genetic, demographic/life history, ecosystem, and landscape disturbance processes) of ecological systems." The function of indicators is to quantify, simplify, and communicate (Elliot, 2011) as well as to synthesize information and facilitate interpretation (Doren et al., 2009). Science has developed indicators and suites of indicators to communicate responses to individual stressors such as fishing (e.g., Blanchard et al., 2010; Coll et al., 2010). More recently, various tools and approaches have been and are currently being developed to characterize ecosystem responses to multiple stressors and cumulative impacts (e.g., Levin et al., 2009; Ban et al., 2010; HELCOM, 2010; Borja et al., 2011; Halpern et al., 2012; Korpinen et al., 2012).

IDENTIFYING INDICATORS

Explicit objectives for management should be the basis for developing and selecting indicators within an ecosystem-based approach to marine management (Levin et al., 2009; Perry et al., 2010a). There is a vast quantity of literature identifying ecosystem indicators and a general agreement that utilizing a suite of indicators is the best approach to understanding ecosystem responses to drivers and pressures (Link, 2002, 2005; Fulton et al., 2005; Greenstreet et al., 2012). Which indicators are included in that suite is determined by using a framework and selection criteria (e.g., Rice and Rochet, 2005; Borja and Dauer, 2008). Niemeijer and deGroot (2008; Table 1) summarize common indicator selection criteria used in the literature. Additional criteria include "nondestructive" (Elliot, 2011), data accuracy and precision (Rice and Rochet, 2005; Painting et al., 2013) and indicator independence of sample size (Noss, 1990). Most criteria apply to single indicators; however, one key criterion is that suites of indicators should be integrative, covering key components and gradients in the ecosystem.

Choosing a suitable suite of indicators that is complementary and nonredundant, and that integrates responses to multiple stressors and reflects the status of the ecosystem is a difficult process (Painting et al., 2013). Considerations for selecting a suite of indicators include ensuring they (1) cover key ecoregions and the appropriate boundary settings to achieve adequate spatial and temporal coverage (Doren et al., 2009; Birk et al., 2012), (2) consider different levels of biological organization, from cellular to ecosystem levels (Adams and Greeley, 2000; Elliot, 2011) and key functional groups (Rombouts et al., 2013), and (3) cover the essential ecosystem characteristics or attributes (Harwell et al., 1999; Fulton et al., 2005) and processes (Rapport et al., 1985) with fast and

TABLE 1. Common indicator selection criteria as summarized from the literature by Niemeijer and deGroot (2008) and adapted here.

theoretically sound	time bound	understandable by the public
credible	measurable	compatible at different scales
integrative	repeatable	links to socioeconomic indicators
important	specific	links to management
historical data available	good statistical properties	links to policy targets
reliable	applicable to other areas	apparent significance
anticipatory	applicable to other situations	relevant
predictably responds to changes	applicable to other scales	appropriate spatial and temporal scales
insensitive to interference	cost effective	thresholds to determine action
sensitive to stresses	operationally simple	user driven
space bound	achievable and timely	

slow dynamics (Fulton et al., 2005). To incorporate these and other considerations in the selection of a suite of indicators, frameworks and procedures for selecting indicators are used. Examples of frameworks that can inform the selection of a suite of indicators include an Ecosystem Risk Assessment Framework (DFO, 2012), environmental assessments (US Environmental Protection Agency [USEPA]), hierarchical frameworks (Dale and Beyeler, 2001; Kershner et al., 2011), an eight-step process defined by Rice and Rochet (2005), and causal chain frameworks such as Driver-Pressure-State-Impact-Response (DPSIR; Elliot, 2002).

To assess ecosystem integrity, indicators must account for ecosystem "structure, composition, and natural processes, including function and dynamics of its biotic communities and physical environment" (Borja et al., 2008). Because it is not possible to study all components of a marine or coastal system, a set of species, habitats, or community properties may be selected to serve as sentinel indicators of the overall health or integrity of the ecosystem (Rapport et al., 1985) or that reflect a particular management goal. Identifying appropriate indicators for ecosystem responses to multiple stressors requires an understanding of (1) how ecological components are connected in the ecosystem and the roles they play in energy flow in the system, (2) the hierarchical pathways through which sector activities affect ecosystem components, and (3) how changes manifest in species or habitats (Canter and Atkinson, 2011). These can be measured in a number of different ways but are generally captured using metrics of, respectively, (1) connectivity or importance of the ecosystem component in the food web (e.g., important trophic positions or niches, keystone species that contribute significantly to the biomass or energy flow of a system, or species or habitats that are particularly sensitive or vulnerable to stressors in the system, or are particularly good for monitoring biomarkers of exposure), (2) exposure of the ecosystem component to the stressor, and (3) vulnerability or sensitivity of the ecosystem component to the stressor(s) (Borja et al., 2008; Samhouri and Levin, 2012).

The exposure attribute describes how much activities or stressors interact with the ecosystem component in space and time. Depicting exposure can be done using metrics that capture the level of activities or stressors in ecosystems (e.g., levels of nutrient loads, urbanization, ocean noise) or by using abiotic and biotic markers of exposure, such as physiochemical measurements, DNA damage, or expression of stress proteins in organisms (Adams and Wendel, 2005). Vulnerability or consequence describes the potential for long-term harm to an ecosystem component as a result of interactions with one or more stressors. This represents the capacity of the ecosystem component to resist and/or recover from exposure to stressors. Indicators of vulnerability or consequence can be identified at varying levels of organization, such as individual-organism condition; population-level demographic rates or abundance; species-level distribution, interactions, or diversity; community-level functional diversity; and ecosystem-level states and functions (Rombouts et al., 2013).

The combination of multiple stressors in marine systems can affect their resilience and push them toward thresholds, ultimately leading to regime shifts (Hughes et al., 2013), beyond which ecosystems may fail to recover to their previous states (Duarte et al., 2009). To develop management strategies that identify impending ecological thresholds or tipping points before they occur, researchers are developing early warning indicators by combining methodologies from economics, climatology, and ecological modeling and testing them primarily in model systems that have already crossed a threshold (Scheffer et al., 2009, 2012; Dakos et al., 2012). One of the most robust early warning indicators of impending ecological thresholds is a "critical slowing down" (Drake and Griffen, 2010; Dakos et al., 2012), resulting in longer recovery times from a disturbance due to the loss of resilience (Scheffer et al., 2009). Ecosystems have also been shown to exhibit rising system memory (i.e., correlation; Biggs et al., 2009; Dakos et al., 2010, 2012), increased variability (Carpenter and Brock, 2006; Daskalov et al., 2007), and "flickering" between alternate ecosystem states (Dakos et al., 2012) as they approach thresholds. Recovery from a degraded ecosystem structure and function can take many years, and ecosystems may never recover to a previous state due to shifting baseline environmental conditions (Duarte et al., 2009, 2013; Borja et al., 2010).

APPROACHES TO ASSESSING INDICATOR RESPONSES TO MULTIPLE STRESSORS

A broad group of approaches have been used to assess indicators of multiple stressors, including data based, expert opinion and judgment, combined

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observation and expert judgment, and model based. Some of the strengths and challenges of each approach were identified by examining several examples in the literature. The goal of comparing approaches was to recommend a strategy for assessing indicators of responses to multiple stressors.

Data Based

Data-based approaches for evaluating indicator responses to multiple pressures include, for example, empirical observations (e.g., Peterson et al., 2013), biomarkers of exposure (e.g., Mussali-Galante et al., 2013), bioindicators of effects (e.g., Adams, 2005), meta-analysis (e.g., Crain et al., 2008), and multiple regression interaction terms (Thrush et al., 2008). For example, the Northwest Fisheries Science Center (NWFSC, Seattle, USA) used empirical observations of a suite of indicators to provide qualitative forecasts of coho salmon (*Oncorhynchus kisutch*) and Chinook salmon (*O. tshawytscha*) survival (Peterson et al., 2010, 2013). As noted by the authors, this approach did not work in all years, and there is a need to consider the strength and collinearity of multiple stressors at different life-history stages (Peterson et al., 2013). Burke et al. (2013) used a multivariate approach to forecast salmon returns using 31 indicators of large- and local-scale environmental conditions, growth, feeding, predation, disease, and cohort abundance. This type of analysis has the benefit of identifying the relative importance of indicators and can include covarying indicators.

Rohr et al. (2006) used a laboratory experimental approach to examine the effects of multiple stressors on salamander survival. Their results indicate that amphibian mortality is directly affected by contaminants, not only during exposure but also months after exposure, and can be mediated by animal density. Laboratory experiments such as these are valuable for clearly identifying effects of a small number of stressors; however, it is difficult to replicate multiple stressors experienced by animals in the natural habitat. There may be annual variation in the number, type, or strength of stressors animals encounter, susceptibility to stressors may vary among species, and effects of stressors may depend on community structure (Rohr et al., 2006).

An integrated bioindicators approach has been used to understand mechanisms of ecosystem responses to stressors in field situations. Adams and Greeley (2000) used an integrated bioindicators approach in which indicator responses were measured at different levels of biological organization and at appropriate time scales to link stressors with indicator responses. They noted several advantages of this approach, including: "(1) early warning signals of environmental damage and (2) assessment of the integrated effects of a variety of environmental stressors on the health of organisms, populations, and communities."

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Meta-analyses have been used to explore potential patterns in indicator responses to multiple stressors (e.g., Crain et al., 2008; Darling and Cote, 2008). This approach entails searching published studies for impacts of multiple stressors, and results show that responses can be additive, synergistic, or antagonistic (Ban et al., 2014; Crain et al., 2008; Darling and Cote, 2008). Furthermore, Crain et al. (2008) noted the importance of understanding mechanisms by which single stressors affect indicator responses as a step toward improved understanding of responses to multiple stressors. Meta-analyses are limited by the studies available in the published literature. To date, most studies are on species-level responses conducted in laboratory settings, and there are few replicate studies on many potentially important stressors (Crain et al., 2008; Darling and Cote, 2008).

Some advantages of data-based approaches to evaluating indicator responses to multiple pressures are: (1) causal relationships between pressures and indicator responses can be established, (2) emerging stressors can be tracked in cases where expert input is untested or models are unavailable, (3) indicators can be tailored to the physical and biological nature of the ecosystem, and (4) remotely sensed data are available for many physical environmental variables (Table 2). However, it is sometimes difficult to find data at scales that link multiple pressures to ecosystem indicators, and this may limit analyses to the shortest available time series and/or the smallest common spatial domain (Table 2). Multivariate statistical analyses can address correlation among indicators but may eliminate critical information. It is also difficult to replicate multiple stressors in a laboratory setting and document the number, type, or strength of stressors animals encounter or are susceptible to in the natural environment.

Expert-Judgment Tools

Researchers and managers across the globe have turned to risk assessment frameworks based on expert judgment to prioritize and identify indicators of potential impacts from multiple stressors by integrating across multiple activities and ecological components (e.g., Halpern et al., 2007; Weisberg et al., 2008; Teck et al., 2010; Teixeira et al., 2010; Hobday et al., 2011; DFO, 2012; Samhouri and Levin, 2012). Some risk assessment frameworks have been modified for specific ecosystem components, such as seagrass or marine mammals (Grech et al., 2011; Lawson et al., 2013), or activities (DFO, 2013), while others are generalized to include multiple stressors and multiple ecological components (Suter, 1999; Hayes and Landis, 2004; Hobday et al., 2011; DFO, 2012; Samhouri and Levin, 2012). Based on qualitative and/or quantitative data, indicators of exposure include the spatial and temporal extent of the stressor and the intensity of the stressor in terms of concentration or effort. The consequence scoring can be based on expert judgment of population or habitat responses to stressors, life-history attributes of species, habitat attributes, or community attributes that indicate vulnerability of a particular ecosystem component to stressors (e.g., Figure 1).

A framework for integrated system-level assessments that relied on expert judgment was developed for Australia's marine environment (Ward, 2014). This framework was applied in Australia and the South China Sea marine ecosystems where indicators were populated using a rapid expert elicitation process to provide a synthesis of the pressures on and conditions of components of the ecosystems (Feary et al., 2014; Ward et al., 2014). Knights et al. (2013) used a combination of expert knowledge and published literature to identify linkages among activities, pressures, and ecological characteristics. Rather than a linear DPSIR or PSR (Pressure-State- Response) approach, Knights et al. (2013) developed a network of linkages among multiple activities, pressures, and responses. Network topology metrics, such as linkage density and number of links per ecological characteristic, along with cluster analyses, permitted the grouping of similar impact chains (Knights et al., 2013).

Some advantages of expert-based judgment tools are that they provide some insight in cases where data are unavailable, they are useful for prioritization of ecological components or stressors, the methods are transparent and repeatable, and they can be appropriate for global and regional visualization (Table 2). In the case of network and network analyses, management measures may be more efficient if they address groups of pressures (Knights et al., 2013). However, there is often not enough information for specific response variables, and these approaches generally do not provide a mechanistic understanding of stressor-response interactions (Table 2).

Combined Observation/ Expert Judgment: Mapping and GIS Approaches

Combined data-based and expert-opinion methods have recently been applied along with mapping approaches to address ecosystem responses to multiple stressors. A spatial analysis tool called Cumulative Impacts was developed by the National Center for Ecological Analysis and Synthesis (NCEAS), University of California, Santa Barbara, and Stanford University to map human activities and their ecological impacts (http://www.nceas.ucsb.edu/globalmarine). The scientific community has mainly used the Cumulative Impacts tool to understand

TABLE 2. Some strengths and challenges of general alternative approaches for evaluating ecosystem responses to multiple stressors: (1) data based, (2) expert judgment, (3) combined data based and expert judgment (only additional strengths and challenges of combining the two approaches are listed), and (4) model-based approaches.

	Strengths	Challenges
Data based	Causal relationships established	Difficult to replicate multiple stressors in laboratory setting
	Track emerging stressors where expert input is untested or models are unavailable	Difficult to find data at appropriate scales
	Appropriate indicators tailored to physical and biological nature of ecosystem	Analyses limited to least common denominator (shortest time series, smallest common spatial domain)
	Remotely sensed data available for many physical variables	Multivariate analyses may eliminate critical information
Expert judgment	Provides insight where there are no data	Often not enough information for specific response variables
	Prioritization of ecological components or stressors	Does not provide mechanistic understanding of stressor-response interactions
	Appropriate for global and regional visualization	
	Network approach may be made more efficient by addressing groups of pressures	
Combination data based and expert judgment	Incorporates data into the expert judgment approach	Assumptions (e.g., additivity of responses) on outputs have not been fully explored
Model based	Can generate as much data as needed	Must have a model (data and time intense)
	Can create an ensemble of models using different frameworks	Outputs are only as good as the data that go into the model

broad-scale patterns in stressor interactions and ecosystem health. This approach models and maps the intensity of each stressor in the ocean, maps the location of each habitat type or species in the ocean, and applies a vulnerability weight derived from expert judgment that translates the intensity of a stressor into its predicted impact on the habitat or species; it creates a metric of impact that can be compared across stressors or ecological components (Halpern et al., 2007, 2008; HELCOM, 2010; Teck et al., 2010; Kappel et al., 2012). These individual impact scores for each stressor in each habitat can then be summed to obtain a total cumulative impact score. The summed impact scores or the individual scores for each habitat can be used to identify which habitats are vulnerable to specific stressors or to the cumulative effects of multiple stressors, or to identify those stressors that in combination are widespread and may have major consequences for ecosystems.

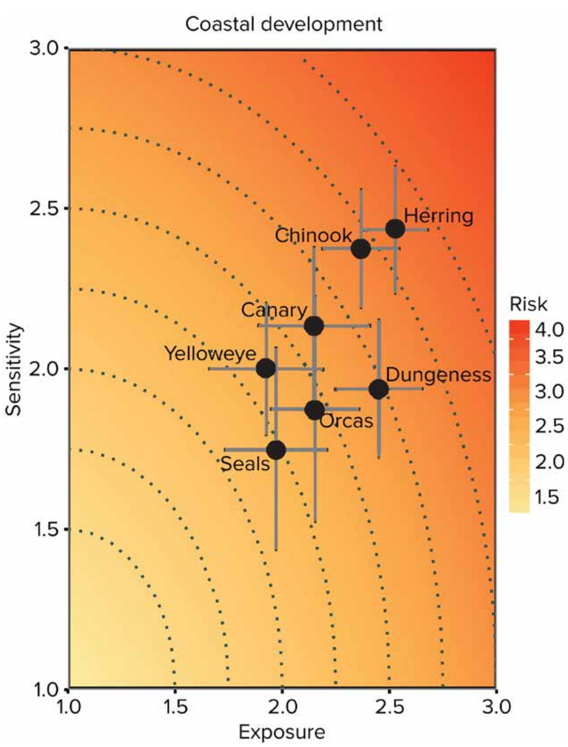

FIGURE 1. Risk to indicator species in Puget Sound, USA, due to coastal development from Samhouri and Levin (2012, their Figure 1). The relative risk is expressed as the Euclidean distance of the species from the origin in the exposure-sensitivity space. *Image Courtesy of Elsevier*

Though there have been recent advances in the quality and quantity of data available for this type of cumulative impact mapping (e.g. Maxwell et al., 2013), opportunities remain to improve these models for identifying indicators (Halpern and Fujita, 2013). For example, groundtruthing the scores using field-collected data on ecosystem condition may improve indicator selection. Finally, most management focuses on the delivery of benefits from nature to people (Millennium Ecosystem Assessment, 2005). Understanding impacts to ecosystem service provision would improve the linkage between cumulative impact mapping and decision making (Halpern and Fujita, 2013). There are some examples of this type of analysis (Altman et al., 2011; Allan et al., 2013), but there is a need for additional research on this topic.

The combination of approaches (data based and expert judgment) addresses one of the challenges of expert-opinion methods by incorporating data on exposure to human activities and stressors, and it is appropriate for global, regional, and local-scale visualization of impacts to the ocean (Table 2). The challenges of this approach are that the models of activities and stressors are built on a suite of assumptions (e.g., additivity of responses to multiple stressors), and the effects of these assumptions on model outputs have not been fully explored. In addition, these approaches still use vulnerability rather than measures of consequence and do not include a mechanistic understanding of the impacts of human activities on ecosystems and ecosystem services, in part due to limitations in empirical research on such relationships (Table 2).

Model Based

A variety of modeling approaches have been developed to assess ecosystem responses to multiple stressors. Effective approaches and analyses have been developed or applied that use qualitative models, a combination of data and models, multivariate analyses, and quantitative models, including ecosystem models. For example, in Australia, CSIRO (Hayes et al., 2012) used qualitative models of feature-pressure interactions to identify ecological indicators. Qualitative models were used because there was not enough quantitative data available. Key ecological features and the drivers and pressures that affect them were mapped. Using the qualitative model, various "pressure scenarios" were examined to assist in the identification of indicators robust to uncertainty about ecosystem structure, and selection criteria were used to refine the indicator list. Notwithstanding the shortage of empirical data, this unique approach resulted in identification of one to four ecological indicators and one to three pressure indicators for some of Southwest Australia's key ecological features (Hayes et al., 2012).

Painting et al. (2013) developed a valuable approach to testing indicators that, with a well-developed model including all potential pressures combined with field-collected data, enabled identification of indicators that met several selection criteria (e.g., sensitive and specific). They examined two pressures, climate and trawling, and found three potential indicators sensitive and specific to climate effects (primary production, phytoplankton productivity, and near-bed oxygen concentrations) and one indicator sensitive to demersal trawling (oxygen penetration depth; Painting et al., 2013).

There are several efforts to understand the multiple factors that affect salmon throughout their complex life history, as previously mentioned (Burke et al., 2013; Peterson et al., 2013). Mantua et al. (2007) used a policy gaming model (MALBEC) for assessing links between ecosystems to integrate spatially explicit impacts of multiple stressors on all life stages of salmon. This type of modeling strategy required data for several ocean and freshwater regions of the North Pacific, such as salmon abundance, oceanographic data, and zooplankton biomass from field or model-derived time series, data that are not always available. Due to a lack of data, Araujo et al. (2013) built a probabilistic network that utilized available data and observations, expert opinion, and model output to examine factors (physical, biological, and hatchery production) affecting the early marine survival of coho salmon in the Strait of Georgia, Canada.

At the global scale, comparative modeling efforts have been utilized to draw generalities about ecosystem responses to multiple stressors. Programs such as Global Ocean Ecosystem Dynamics (GLOBEC; Megrey et al., 2007), Comparative Analysis of Marine Ecosystem Organization (CAMEO; Link et al., 2012), and Indicators for the Seas (IndiSeas; Bundy et al., 2012) have used a combination of data and modeling approaches to compare ecosystems. As part of CAMEO, Fu et al. (2012) used partial least squares (PLS) regression to infer pressure-response interactions for nine ecosystems. They found the advantages of this type of statistical analysis to be that predictor variables (pressures) can be correlated and multiple response variables can be included, unlike in regression analyses. Fu et al. (2012) also observed that PLS regression may be better for predicting indicator responses than, for example, principal components from multivariate analyses. The authors noted that trophodynamic data time series were unavailable for some ecosystems, again highlighting one challenge in large-scale, multinational ecosystem comparisons.

In addition to the previously mentioned multinational programs, Barange et al. (2014) explored the effects of climate change on fish production and the economies of 67 ecosystems/nations. A climate model was used to drive a dynamic size-based food web model; the nutritional and economic consequences to nations were examined using an index of fisheries dependency based on measures of vulnerability (Barange et al., 2014). The authors point out that model results may be sensitive to assumptions that are necessary in the modeling process. Other data-intensive ecosystem models, such as Object-oriented Simulator of Marine ecOSystems Exploitation (OSMOSE) have been used to simulate indicator responses to pressures, such as fishing, climate change, and their interactions (Fu et al., 2013). One advantage of this approach is that model results can be additive, synergistic, or antagonistic (Fu et al., 2013).

Many early warning indicators of ecological thresholds, such as increased variance, critical slowing down, and flickering, have been identified using modeling simulations and long-term data sets (Daskalov et al., 2007; Dakos et al., 2012). Identifying reliable indicators and quantifying thresholds in ecological systems can be challenging due to lack of appropriate data (deYoung et al., 2004; Håkanson and Duarte, 2008; Goberville et al., 2010). Many early warning indicators require long-term, high-resolution data with relatively little noise, which are uncommon in ecological systems (Dakos et al., 2008, 2012; Scheffer et al., 2009). Furthermore, recent studies show that threshold detection via a single early warning indicator is insufficient, that using multiple indicators could strengthen predictions of impending thresholds (Dakos et al., 2012), and that some indicators may be correlated (Contamin and Ellison, 2009; Ditlevsen and Johnsen, 2010). Boettinger and Hastings (2012) suggest it is unlikely there are early warning indicators common across ecosystems and recommend that data-driven exploration within ecosystems be utilized to identify system-specific characteristics of ecological thresholds. Experimental approaches may help to address this issue by capturing the context-dependent nature of thresholds (Thrush et al., 2009; Hewitt and Thrush, 2010), particularly when conducted across environmental and/or disturbance gradients.

Model-based approaches are, perhaps, among the best tools for understanding ecosystem responses to multiple stressors, but they require the greatest data and time investments. A variety of frameworks can be used to create an ensemble of models, and models can generate data as needed; however, the outputs are only as good as the data that go into the models (Table 2). Also, setting up models and supplying them with data may not be feasible due to lack of resources and/or data availability (Table 2).

The various approaches to assessing responses to multiple stressors (data based, expert judgment, combinations of data based and expert judgment, and model based) have several strengths and challenges. As noted above, data-based approaches enable the establishment of causal relationships between pressures and indicator responses. Three of the approaches—data based, combined observation and expert judgment, and model based—share a common challenge in that they all depend on data availability. Expert opinion approaches avoid this problem, but may not provide a mechanistic understanding of stressor-response interactions. Modeling approaches are recommended as the best ways to assess indicator responses to multiple stressors; however, they require significant investment in data and resources, which are often not available. The strengths and challenges of the three approaches also depend on the objectives. For example, is the objective to determine the state of ecosystems or to identify management interventions? Although providing a general understanding of the state of ecosystems and ecosystem responses is a key scientific goal for ecosystem-based management, identifying clear management objectives is a key aspect of choosing appropriate indicators. Thus, linking scientific pursuits directly to specific decision contexts is a next step. In light of the strengths and challenges of the described approaches and the fact that data availability will continue to be lacking for some stressors and ecosystems, we recommend using multiple approaches to identify indicators and evaluate multiple stressors on marine ecosystems.

COMPARISON OF PROGRAMS THAT HAVE IDENTIFIED SUITES OF INDICATORS

Many programs have identified suites of indicators for monitoring and assessing the status and trends in ecosystem composition, structure, and function. Here, we discuss several examples of programs that have taken various approaches to assessing the state and trends of marine ecosystems. The US National Marine Fisheries Service Alaska Fisheries Science Center's (AFSC's) Ecosystem Considerations report (Zador, 2013) uses the DPSIR approach to assess several ecosystems, thereby providing an opportunity to compare suites of indicators arising from the same process and institution across multiple ecosystems. The USEPA and Environment Canada jointly assembled a report on the Salish Sea, and the Puget Sound Partnership assembled a Puget Sound Vital Signs (PSVS) report (Figure 2). These reports cover overlapping ecosystems, providing an opportunity to compare indicators and approaches used by different organizations for an overlapping geographic area. The Helsinki Commission (HELCOM, 2013) assembled a core set of indicators for the Baltic Sea using a PSR approach. The US National Oceanic and Atmospheric Administration (NOAA) used a hierarchical selection process to choose indicators that represent a broad set of ecosystem management goals ranging from sustaining fisheries to maintaining ecological integrity and protected species. Finally, Europe's Marine Strategy Framework Directive (MSFD) identified 11 descriptors of ecosystems in good environmental status. We compared indicators used in these different ecosystem assessments and identified sources of differences.

FIGURE 2. The Puget Sound Vital Signs Wheel or Dashboard is a part of Puget Sound Partnership's Puget Sound Vital Signs report (PSVS). The Dashboard identifies the key ecosystem indicators and pressures, incorporates targets, and will serve as a report card on success in meeting targets. *Image courtesy of the Puget Sound Partnership (2013)*

Example 1: Alaska Ecosystem Considerations

The AFSC successfully manages groundfish fisheries while incorporating ecosystem considerations (Livingston et al., 2011). The AFSC's Ecosystem Considerations report provides an assessment of multiple pressures on ecosystems: fishing, human-induced, and natural pressures such as climate variability (http://access.afsc.noaa.gov/reem/ecoweb/index.php). The report comprises three main sections: (1) Executive Summary (Report Card), (2) Ecosystem Assessment, and (3) Ecosystem Status and Management Indicators for the different ecosystems in Alaska (Zador, 2013). The Executive Summary provides a Report Card on key status and trend indicators in the eastern Bering Sea and the eastern, western, and central Aleutian Islands. The Ecosystem Assessment contains a synthesis of climate and fishing effects on Alaska ecosystems (Arctic; eastern Bering Sea; eastern, western, and central Aleutian Islands; and Gulf of Alaska) using a short list of indicators. Both the Report Card and the Ecosystem Assessment sections use selected indicators from the Ecosystem Status and Management section, which provides information on the status and trends of ecosystem components (e.g., physical environment, habitat, plankton, fish, marine mammals, seabirds, community-level indicators), early detection of direct human effects on the ecosystem, and effectiveness of management actions (Zador, 2013). In the Ecosystem Assessment section, indicators were selected using the DPSIR approach (Elliot, 2002) to address four ecosystem-based management objectives: maintain predator-prey relationships, maintain diversity, maintain habitat, and incorporate/monitor effects of climate change. Drivers and pressures pertaining to these objectives were identified and a list of candidate indicators were selected based on qualities such as, availability, sensitivity, reliability, ease of interpretation, and pertinence. Indicators of three broad categories were included: biology/biodiversity, climate, and fishing. Finally, for the Report Card, an Ecosystem Synthesis Team refined an indicator list focused on broad, community-level indicators to assess current and potential future ecosystem states (biology/biodiversity, climate, fishing) and included human quality-of-life indicators (Zador, 2013).

Example 2: Salish Sea and Puget Sound

Two programs assessed the adjoining coastal waters of British Columbia, Canada, and Washington State, USA. The Puget Sound Partnership assembled a PSVS report (Puget Sound Partnership, 2013; Figure 2), and USEPA and Environment Canada jointly assembled a report on the Salish Sea (Georgia Basin-Puget Sound ecosystem; (http://www2.epa.gov/salish-sea). The PSVS report used DPSIR and integrated ecosystem assessment (IEA) approaches to communicate project progression, use of funds, and status of the Puget Sound ecosystem with a longer, but overlapping, list of indicators compared to the USEPA report. The USEPA report used a DPSIR approach to communicate the state of the Salish Sea (a larger body of water that includes both Puget Sound and the Strait of Georgia) to the public using a short list of indicators.

Example 3: Baltic Sea – HELCOM

The Helsinki Commission identified a core set of indicators to assess the Baltic Sea ecosystem, choosing core indicators using a PSR framework to address strategic goals (favorable biodiversity and undisturbed by hazardous substances and eutrophication) and ecological objectives (e.g., clean water, viable populations of species; HELCOM, 2013). The 20 core indicators for biodiversity, 13 for hazardous substances, and four for eutrophication met predefined HELCOM principles (e.g., monitored, covers the entire area, reflects pressures, quantitative, updated regularly) and measured current status relative to targets outlined in the Baltic Sea Action Plan (http://helcom.fi/baltic-sea-action-plan). Indicators included the main ecosystem components (mammals, birds, fish, and nonindigenous species) and habitats (pelagic, seabed; HELCOM, 2013). A unique feature of the HELCOM (2013) report was the pressure indicator matrix that identified the multiple pressures most likely to affect each biodiversity indicator. The strengths of the pressure-indicator interactions were also included. For example, higher trophic level animals were most likely affected by fishing and contaminants, lower trophic level animals by eutrophication, and benthic habitats and communities by fishing and eutrophication.

> "...the selection of suites of indicators should be based on clear conceptual models linking indicators to pressures and drivers, on management objectives (Perry et al., 2010a), and on established criteria, while ensuring that the final suite consists of indicators that are complementary, nonredundant, and integrative."

Example 4: US California Current Integrated Ecosystem Assessment

On the west coast of the United States, NOAA is developing an IEA for the US California Current (CCIEA; http://www.noaa.gov/iea/regions/california-current-region). The IEA report is intended to deliver integrated, cross-sector science to support ecosystem-based management (Levin et al., 2009). The five ecosystem goals on which the IEA is focused include conserving or managing wild fisheries, protected resources, habitat, vibrant coastal communities, and ecosystem integrity. Indicators for each of these goals, along with a set of natural and anthropogenic drivers and pressures, were selected using a hierarchical indicator selection process based on a series of criteria similar to those listed in Table 1 (Kershner et al., 2011; Andrews et al., 2013). Thus, each ecosystem state indicator reported in the IEA maps to key ecosystem attributes, which in turn are related to one of the five ecosystem goals. Anthropogenic drivers and pressures fall into one of 23 categories as varied as fisheries removals, commercial shipping activity, and pollution. Natural drivers and pressures—due to changes in oceanography and climate—fall into nine different categories, including influences like changes in ocean temperature, decreasing oxygen, and ocean acidification. All of the IEA indicators were developed via data-based, expert-judgment, and model-based approaches. To date, the indicators of drivers and pressures have not been quantitatively linked to ecosystem states in the IEA, though that is the intention in future iterations of this report.

Example 5: European Marine Strategy Framework Directive

In Europe, the MSFD was introduced to protect and restore Europe's regional seas and designed to achieve good environmental status through coordinated and integrated research by 2121 (COM, 2005a,b). To determine environmental status, 11 descriptors were identified, including biological diversity, nonindigenous species, exploited fish and shellfish, food webs, human-induced eutrophication, seafloor integrity, hydrological conditions, contaminants, contaminants in fish and seafood, litter, and energy and noise (European Commission, 2010; Borja et al., 2011; Figure 3). Expert groups developed considerations for application and methodological standards for each descriptor (Cardoso et al., 2010). Attributes, criteria (29), and indicators (56) were selected for each of the descriptors (European Commission, 2010), and recommendations have been proposed for articulating good environmental status (Mee et al., 2008; Borja et al., 2013; Tett et al., 2013; Figure 3). One goal of the MSFD was to have each EU Member State conduct an initial assessment of the current environmental status of its waters and the environmental impact of human activities on them (COM, 2005b; Cardoso et al., 2010). Toward that goal, Borja et al. (2011) implemented the MSFD to assess the environmental status of the Bay of Biscay (Basque Coast) and proposed a method for integrating the descriptors into an overall ecosystem status.

Comparison of Examples

Among the example programs examined, there were both commonalities and differences in the pressures and indicators that were identified and how responses to multiple stressors were addressed. Some differences were due to the overall goals and objectives of the reports. For example, some reports focused on assessing the state of an ecosystem (e.g., Salish Sea USEPA, Bay of Biscay MSFD), while others also assessed progress toward targets (e.g., HELCOM, PSVS) and/or addressed ecosystem-based fishery management goals (e.g., Alaska) or marine management goals (e.g., CCIEA). All programs used a causal-chain conceptual framework, such as DPSIR or PSR to address pre-defined strategic goals and ecological or management objectives. HELCOM (2013) identified the difficulty in differentiating pressure and state indicators; for example, dissolved oxygen can be a state indicator of water quality but also a pressure indicator for sessile or low motility animals. This highlights the need for clearly documented conceptual or pathways-of-effects models and risk assessments.

For all ecosystem reports examined, a list of potential indicators that reflect identified pressures was established and refined by data availability, selection criteria, and, in some cases, expert knowledge. All ecosystem reports included indicators that reflect climate and fishing pressures; however, the other types of pressures included varied among reports (Figure 4). The Alaska Ecosystem Report Card, CCIEA, and the PSVS report had indicators of human quality of life. The HELCOM, CCIEA, and Bay of Biscay MSFD reports included indicators of eutrophication. Five of the reports (CCIEA, HELCOM, PSVS, Salish Sea USEPA, and Bay of Biscay MSFD) had indicators of hazardous substances, whereas the Alaska reports did not include hazardous substance indicators. Differences in the pressures identified in each report are a reflection of the main pressures acting on ecosystems and the spatial delineation of the ecosystems. For example, most reports that included hazardous substance indicators were for semi-enclosed waters (e.g., Baltic Sea) that included nearshore areas (e.g., Puget Sound), whereas the Alaska ecosystems are large oceanic ecosystems that encompass waters 3 nm to 200 nm from shore.

Comparisons between ecosystem

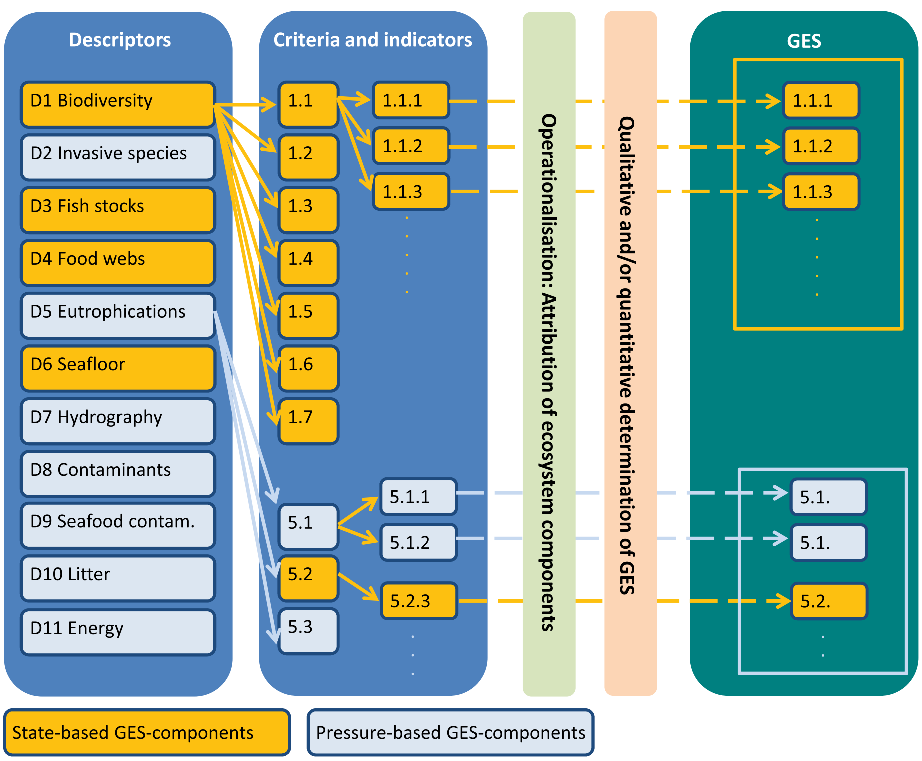

FIGURE 3. Components for the determination of Good Environmental Status (GES) in the Marine Strategy Framework Directive (MSFD). There are 11 descriptors, 29 criteria, and 56 indicators. Due to variability among ecosystems, descriptors, criteria, and indicators used in assessments may vary. *Image courtesy of the MSFD Guideline produced by Knowseas* (http://www.msfd.eu/knowseas/guidelines/3-INDICATORS-Guideline.pdf)

reports also revealed similarities and differences in response indicator selection (Figure 4). A feature of all the examples is that key functional groups with fast and slow dynamics and essential ecosystem characteristics were represented in the suites of indicators. For example, most reports included indicators of marine mammals, representing key functional groups at high trophic levels with slower dynamics. All reports also include estimates of fish biomass or abundance, representing key functional groups at lower trophic levels with faster dynamics. Differences in indicator selection among reports reflected a variety of factors. For example, differences between the indicators presented in the Salish Sea USEPA and PSVS reports, assembled for overlapping waters by different organizations, may reflect the level of detail thought appropriate for communicating to a public (nonscientific) audience, the experts involved, and perhaps data availability common to both US and Canadian waters. The PSVS report included most of the indicators that were in the Salish Sea USEPA report (all except air quality indicators); however, there were differences between the two reports in the types of indicators utilized to represent some components of the ecosystem. For example, both reports included an indicator of Chinook salmon; however, the PSVS report used the number of natural origin adult Chinook salmon returning to spawn, and the Salish Sea report used the number caught, number of returns, and total abundance of Chinook salmon. A unique feature of the PSVS report is that it identified the current status of indicators relative to baseline values as well as predefined targets. Differences in indicators among the Alaskan ecosystems, assessed by the same organization, highlighted the unique characteristics of each ecosystem and the spatial and temporal differences in (1) the main climate and human-induced pressures, (2) species composition and key functional groups/features, (3) data availability and extent of knowledge about the ecosystem, and (4) the particular expertise of team members (Zador, 2013). For example, zooplankton times series were available for the eastern Bering Sea but not for the Aleutian Islands. Instead, planktivorous seabird reproductive success was used as an indicator of zooplankton in the central and western Aleutian Islands, while no indicator was available for the eastern Aleutian Islands.

Each report considered the effects of multiple stressors on ecosystems. The HELCOM project (HELCOM, 2013) clearly outlined multiple pressures that affected each core indicator in a matrix and ranked the expected level of impacts of pressures on each indicator. The CCIEA has created an ecosystem risk assessment framework to assess the risk to marine habitats due to a variety of activities and pressures (Samhouri and Levin, 2012). The Alaska Ecosystem Assessment (upon which the Alaska Ecosystem Report relies) outlined multiple indicators of each pressure in a table (Livingston et al., 2011; Zador, 2013). The PSVS and Salish Sea USEPA reports outlined single or multiple pressures that affect each indicator in the text of the reports. The Bay of Biscay MSFD report proposed an environment status score based on combining indicators that were reflective of multiple pressures (Borja et al., 2011). In addition to pressures, all reports had indicators of most ecosystem services as defined by the Millennium Ecosystem Assessment (2005) and adjusted for marine ecosystems (Liquete et al., 2013): provisioning (food provisioning, water storage and provision, biotic materials and biofuels), regulating and maintenance

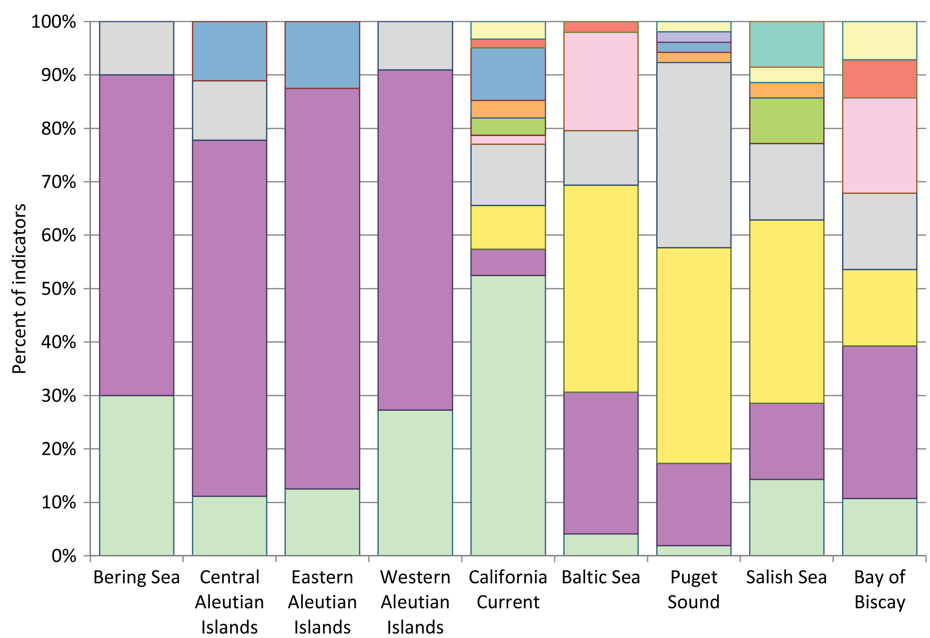

FIGURE 4. Percent of indicators used to reflect general categories of pressures in ecosystem assessment reports: the eastern Bering Sea and the central, eastern, and western Aleutian Islands (Alaska Ecosystem Report Card; Zador, 2013), the California Current Integrated Ecosystem Assessment (Andrews et al., 2013; Hazen et al., 2013; Norman and Holland, 2013; http://www.noaa.gov/iea/CCIEA-Report/index.html), the Baltic Sea (HELCOM, 2013), Puget Sound (Puget Sound Partnership, 2013), the Salish Sea US Environmental Protection Agency and Environment Canada report (http://www2.epa.gov/salish-sea), and the Bay of Biscay (Borja et al., 2011, using the European Marine Strategy Framework Directive).

(water purification, air quality regulation, coastal protection, climate regulation, weather regulation, ocean nourishment, life cycle maintenance, biological regulation), and cultural (symbolic and aesthetic values, recreation and tourism, cognitive effects).

There are other approaches to assessing ecosystems in addition to those described above. For example, several multinational efforts utilize a comparative approach to identify common pressure-indicator links among ecosystems. Multinational programs that have facilitated effective ecosystem comparisons include the Marine Ecosystems of Norway and the United States (MENU; Link et al., 2009), GLOBEC (e.g., Megrey et al., 2007), and CAMEO (e.g., Link et al., 2012). Also, IndiSeas (Bundy et al., 2012; Shin et al., 2012) is a collaborative program that selected a suite of eight indicators to examine the effects of fishing on multiple ecosystems and address defined ecological objectives (Bundy et al., 2012; Shin et al., 2012). A common component of all these multinational projects is the involvement of local experts to provide data and interpret results. The approach of using a common suite of indicators to compare multiple ecosystems is limited by the type, quantity, and quality of data that is common among all ecosystems; however, comparative analyses provide additional insight and improved understanding of pressure effects.

CONCLUSIONS AND RECOMMENDATIONS

Given the variability in the types and intensities of pressures affecting ecosystems, key ecosystem features, ecosystem types, data availability, number and background of experts involved, and approaches (single ecosystem vs. comparison of multiple ecosystems), it is apparent that one definitive list of specific indicators cannot be exclusively used to assess the states of all types of marine ecosystems. This is the case regardless of the conceptual framework and selection criteria by which potential individual indicators are identified. There are at least two general approaches (within a causal chain framework) in the literature by which suites of indicators are assembled: (1) develop indicators that are specific to individual ecosystems or key ecological features (e.g., Hayes et al., 2012), or (2) utilize recommended indicators (of important pressures or responses to those pressures) that, given data availability, can be calculated for multiple ecosystems to address ecological or ecosystem-based objectives (e.g., Jamieson et al., 2010; Bundy et al., 2012). The advantage of the former approach is that relevant pressure-indicator interactions are ecosystem specific, and there is potential for recommending a set of indicators for a range of ecological features. The advantage of the latter approach is that responses to pressures, such as fishing, can be compared across multiple ecosystems, potentially providing further insight into pressure-response interactions common among ecosystems. A third and potentially promising approach (e.g., IndiSeas2) is to use a core set of recommended indicators for all ecosystems and include additional ecosystem-specific, pressure-linked response indicators not reflected in the core set. Additionally, as done in the MSFD, those indicators that are relevant and can be calculated for an ecosystem are selected from a core set of indicators identified by expert groups. These approaches would enable comparisons of common pressure-indicator interactions across ecosystems, and enable a complete characterization of pressures and indicators specific to each ecosystem. Regardless of approach, the selection of suites of indicators should be based on clear conceptual models linking indicators to pressures and drivers, on management objectives (Perry et al., 2010a), and on established criteria, while ensuring that the final suite consists of indicators that are complementary, nonredundant, and integrative.

Suites of core indicators have been tested and recommended for evaluating the effects of fishing and assisting with ecosystem-based fisheries management (Table 3). For example, Fulton et al. (2005) tested the performance of indicators using simulation models and recommended a suite of indicators to examine the effects of fishing on ecosystems. Link (2005) recommended a list of indicators that could be translated into ecosystem-based fishery management decision criteria (Table 3). Jamieson et al. (2010) adapted and added to the indicators recommended by Fulton et al. (2005) and Link (2005), including biophysical indicators of climate change. IndiSeas identified a suite of indicators to examine the effects of fishing (Bundy et al., 2012). In addition, a factor analysis by Greenstreet et al. (2012) indicated a suite of seven or eight indicators was necessary to assess the state of the demersal fish community with respect to the goal of restoring

> “These suites of indicators provide a valuable starting place for examining the effects of climate change and fisheries on ecosystems, and they could be broadened to include other pressure and response indicators for marine management of activities beyond fisheries.”

biodiversity in the North Sea (Table 3). These suites of indicators provide a valuable starting place for examining the effects of climate change and fisheries on ecosystems, and they could be broadened to include other pressure and response indicators for marine management of activities beyond fisheries, such as those used in the MSFD.

Fishing and climate are two important pressures that have been examined (e.g., Perry et al., 2010b); however, there are other environmental, human activity, and sociopolitical-economic pressures that may be important in ecosystems (Table 4). Examples of other activities and associated pressures include nutrient loading, contaminants, oil and gas development, aquaculture, seafood demand, and coastal infrastructure (Table 4). There are many ecosystems with specific management objectives and conceptual frameworks that have identified these types of pressures as important (e.g., Halpern et al., 2008; Knights et al., 2013), and there are programs that have been making progress in assessing multiple pressures, such as HELCOM, the California IEA, and the European MSFD. As regions move toward developing suites of indicators of responses to multiple stressors, it will be valuable to consider the extent to which data are available. Given that data availability will continue to be a challenge, we recommend using a variety of approaches, such as expert opinion, model-based simulation, and empirical analysis to identify indicators and evaluate multiple stressors on marine ecosystems.

Future considerations for assessing the effects of multiple stressors should incorporate uncertainty in indicator development. Sources of uncertainty can include natural variability, observation error, model structural complexity, inadequate communication, unclear objectives, and implementation or outcome uncertainty. Another difficult issue to resolve is the interaction between pressures that are sustained over a long duration and those pressures that are intense, but episodic.

TABLE 3. A compiled suite of indicators recommended for ecosystem-based fisheries management by (1) Fulton et al. (2005), (2) Perry et al. (2010a), (3) Link (2005), (4) Greenstreet et al. (2012), and (5) IndiSeas (Bundy et al., 2012).

Recommended Indicators	Reference	Objective*
Biomass by group or community (e.g., flatfish, pelagic species, piscivores)	1, 2, 3, 5	Maintain resource potential
Total abundance	4	Conserve biodiversity
Abundance of scavengers	3	Maintain structure and function*
Volume of gelatinous zooplankton	3	Maintain structure and function*
Consumption	1	Maintain structure and function*
Species richness (number of species)	1, 2, 3, 4	Conserve biodiversity
Hill's species evenness	4	Conserve biodiversity
Mean von Bertalanffy growth parameter	4	Conserve biodiversity
Mean number of interactions per species	1, 3	Maintain structure and function*
Slope of size spectrum, all species	1, 2, 3	Conserve biodiversity*
Large fish indicator	4	Conserve biodiversity
Proportion of predatory fish	5	Conserve biodiversity
Number of cycles	3	Maintain structure and function*
Maximum or mean length	2, 3, 5	Maintain structure and function
Mean life span	5	Maintain stability and resistance
Mean length at maturity	2, 4	Conserve biodiversity
Mean individual fish weight	4	Conserve biodiversity
Mean age at maturity	4	Conserve biodiversity
Number of groups representing 80% of biomass	1	Maintain structure and function*
Nutrient cycling; estimated denitrification, particularly for shallow-water ecosystems; dissolved inorganic nitrogen, network total production	1	Maintain structure and function*
Production; total primary production	1	Maintain structure and function*
Respiration or total production from network models; otherwise use total production by group, denitrification in shallow-water systems	1	Maintain structure and function*
Biomass ratios (e.g., large:small plankton); length of maximum catch	1, 2	Maintain structure and function*
Mapping biomass indicators	1	Maintain structure and function*
Throughput estimated using network model; alternatively, estimated total production, consumption, respiration	1	Maintain structure and function*
Trophic level or trophic spectrum of catch	1, 2, 3, 5	Maintain structure and function
Biophysical characteristics	2	
Habitat-forming taxa	1, 2, 3	Maintain structure and function*
Fishery removals of all species (e.g., landings, bycatch, discards)	2, 3	Maintain structure and function*
Landings of target species	3	Maintain structure and function*
1/(landings/biomass)	5	Maintain resource potential
Proportion of non-fully exploited stocks	5	Conserve biodiversity
1/coefficient of variation of total biomass	5	Maintain stability and resistance

* Indicates objective was identified either in text of the document or deduced for this paper.

TABLE 4. Some broad-scale activities, stressors, and indicators for consideration in suites of indicators for marine ecosystems such as the North Pacific Ocean.

Environmental Stressors/Indicators	Human Activities and Stressors	Socioeconomic-Political
Temperature	Fishing	Seafood demand
Sea ice	Oil and gas	Coastal population trends
Chlorophyll-*a*	Military activity	Marine employment
Nutrients	Wave/wind/tidal energy development	Marine revenue
River discharge	Shipping	Marine exports/domestic consumption
Toxic contaminants	Coastal engineering	Participation/stakeholder involvement
Large-scale climate index (e.g., Pacific Decadal Oscillation, El Niño-Southern Oscillation)	Aquaculture	Governance
pH	Ecotourism	Happiness
Oxygen	Land-based pollution	Satisfaction with ocean status
		Community vulnerability
		Coastal infrastructure

Also, it will be valuable to explore the possibility of developing reference levels for indicators and suitable methods of communicating results. Presenting indicators of responses to multiple stressors succinctly and unambiguously to policy- and decision makers is a challenge for future ecosystem assessment processes.

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