



Implications of fisheries impacts to seabed biodiversity and ecosystem-based management

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Biodiversity has many key roles in ecosystems, and many elements of biodiversity support fish species and therefore also fisheries. At the same time, cooperation fisheries also often affect seabed biodiversity. Furthermore, fisheries may also change the composition of fish communities, and we illustrate why changes in fish communities can matter to seabed biodiversity. These issues should matter to fisheries management. Biodiversity research by definition addresses heterogeneity and this should influence the nature of questions the science seeks to address and how empirical studies are designed. However, to date biodiversity research has not been fully incorporated into mainstream fisheries science. We aim to facilitate the transition to a more transdisciplinary framework, and move beyond the fisheries-focused management. Human pressure is increasing, and many ecosystems are affected by cumulative impacts from different sources of disturbance. We discuss insights from biodiversity and ecosystem function research, and we advocate for a focus on cumulative impacts from disturbance and resilience. We consider these to be critical elements of the transition into ecosystem-based management. The ecological systems and the services that they generate can be either degraded and support less biodiversity and a smaller range of human values, or they can be resorted. The choice is ours. We advocate for a development of participatory multi-sector management that integrates different institutions to contribute to cultural, social, economic, and biodiversity values for ocean governance.

Keywords: biodiversity, cumulative disturbance, EBM, fishing impacts, regime shift, resilience.

Introduction

Biodiversity has many elements that support fisheries. Concurrently, fisheries often affect biodiversity. The term biodiversity encompasses the extent of genetic, taxonomic, and ecological diversity over different spatial and temporal scales. It includes elements of both structure and function and these elements are often interlinked (Thrush and Lohrer, 2012). Biodiversity research deals with heterogeneity, processes that interact across scales as well as functions that emerge from biogeophysical–chemical interactions. Many of the organisms that live on the seabed are the architects of their own habitats (Thrush and Dayton, 2002, 2010). Organisms that burrow, bioturbate, and bioirrigate the sediment create structure below the sediment–water interface. These activities influence the rate and nature of ecosystem processes such as organic matter degradation

and nutrient recycling (Huettel *et al.*, 2014). Organisms that protrude above the sediment surface or modify the sediment–surface topography often provide refugia and settlement sites for multiple species. Such organisms influence benthic boundary flows affecting the transport and deposition of food, larvae, sediments, organic material, and dissolved chemicals (Thrush and Dayton, 2002, 2010). Thus, these elements of biodiversity support fish species and therefore also fisheries. Many of the species that perform these functions are vulnerable to physical disturbance. The size, density, and patch structure of organisms can affect their functional performance. These species can therefore become functionally extinct if they are impacted by physical disturbance. Many of these ecosystem processes are studied at spatial scales of a few metres or less. However, such small-scale processes do affect large-scale processes and contribute to global chemical cycles (Middleburg *et al.*, 1997).

The current rate of biodiversity loss is substantial and this threatens ecosystem services, many of which influence fisheries. Furthermore, it is influential to future options and many societal values (Vitousek *et al.*, 1997; Millenium Assessment, 2005; Cardinale *et al.*, 2006). Fisheries scientists and managers need to engage in concerns over loss of biodiversity in marine ecosystems, and with this comes the need to expand engagement with a broader range of scientists and perspectives. Concordant with the loss of biodiversity is a growing awareness from society of the multiple uses and benefits derived from marine ecosystems. This has highlighted the need to engage in management actions that support multiple uses and values of the marine environment.

Hitherto, however, the knowledge generated by biodiversity research has not been fully incorporated into the mainstream fisheries science. As biodiversity research by definition addresses heterogeneity, this should influence the nature of questions the science seeks to address and how empirical studies are designed. Fisheries science has been an exemplary in the application of population dynamics and mass-balance foodweb models, but this does not provide all the biophysical knowledge needed to define the impact or recovery, stability or resilience, or ecosystem function and service relationships. Biodiversity has many key roles in marine ecosystems, including affecting ecosystem functions such as productivity. Knowledge of biodiversity has important consequences for our ability to assess risks and make choices about how we use marine ecosystems into the future. As ecosystems undergo change, the society learns to better value ecosystem services. The society also learns to value multiple components of the marine economy. An integration of science into transdisciplinary problem solving is clearly required for marine ecosystems.

Whether the focus is on the provision of nursery habitat, settlement sites for exploited species, nutrient regeneration or fuelling of foodwebs, biodiversity matters to fishers and fisheries managers. However, fisheries management has tended to get locked in by dealing with challenges at the spatial scale of fisheries management units. Furthermore, fisheries management has tended to focus on single-species population dynamic models. However, important ecological insights can be gained from a broader perspective. Recent trophic models and field studies highlight how interactions between benthic food resources and predatory fish can drive differences in the abundance of certain species with increases in seabed disturbance (Hiddink *et al.*, 2008; van Denderen *et al.*, 2013). More recently, there are promising signs of a transition to a more integrated approach in fisheries science (Levin *et al.*, 2009; Fulton *et al.*, 2011; Rice *et al.*, 2012; Foley *et al.*, 2013; Fogarty, 2014).

In this paper, we aim to facilitate the move to a more transdisciplinary framework. We focus on insights from biodiversity and ecosystem function research, particularly in relation to changes in seabed habitats, cumulative impacts from disturbance and resilience. We consider these to be critical elements of the transition to ecosystem-based management (EBM). Yet, these elements are often overlooked in fisheries decision making, were knowledge of stock population dynamics and trophic interactions drive the discourse. We argue that new models of governance are needed to address the multiple uses in marine ecosystems. We illustrate opportunities to move beyond the fisheries-focused management, by enhancing transdisciplinary research.

Disturbance

Disturbance is a natural phenomenon in marine systems. Small-scale disturbance generated by burrowing animals or predators

can be important in defining habitat heterogeneity, while larger and rarer disturbances associated with landslides, diseases, or storms can generate legacy effects on habitats and communities. It has been estimated that ~10% of the earth's seabed is subjected to physical sediment disturbance by natural hydrodynamic processes (Harris, 2014). The successional processes that follow disturbance are the product of interactions within the disturbed area and the supply of recruits (Pearson and Rosenberg, 1978; Thrush and Whitlatch, 2001). The term connectivity is commonly used in landscape, meta-population, and metacommunity ecology. In an ecological context, connectivity includes the combined effects of landscape structure, dispersal, and mortality risks as organisms move among habitat patches (Tischendorf and Fahrig, 2000; Thrush *et al.*, 2013). Ecological resilience can be defined as the capacity of a system to absorb perturbation and reorganize to retain essentially the same function, structure, identity, and feedbacks (see Folke *et al.*, 2004). Thus, limits to dispersal and decreased connectivity are important constraints for resilience of benthic communities and their responses to landscape changes (Thrush and Whitlatch, 2001; Thrush *et al.*, 2008).

In addition to the different sources of natural disturbance, human activities strongly influences the earth's ecosystems on a global scale (Folke *et al.*, 2004; Thrush *et al.*, 2009; Hughes *et al.*, 2013). Many anthropogenic activities, such as habitat modification/destruction, pollution, eutrophication, and overexploitation as well as climate change are changing the disturbance regime.

Cumulative impacts from disturbance, habitat loss, and fragmentation

Many ecosystems are affected by cumulative impacts. Although they may not be individually catastrophic, the combined effect of pressures can result in loss or fragmentation of habitats. Furthermore, it can result in ecosystem changes and shifts in biodiversity, associated with the removal of habitat-specific or functional important species (Thrush *et al.*, 2008). Habitat loss has large, consistently negative effects on biodiversity. The empirical evidence shows that habitat loss has a much larger effect than habitat fragmentation, i.e. breaking apart of habitat (Fahrig, 2003). However, fragmentation implies a larger number of smaller patches, and if each patch of habitat is too small to sustain a local population or even an individual territory, this can also cause negative effects on biodiversity. Increased predation at the edges of patches is another cause of negative effect of habitat fragmentation on biodiversity (Fahrig, 2003). Consequently, habitat destruction and fragmentation have been recognized as major threats to terrestrial biodiversity (Vitousek, 1994). Homogenization of landscapes due to human activity is a recognized broad-scale impact in terrestrial ecosystems (Western, 2001). However, despite many forms of anthropogenic disturbance to the seabed, such effects have not yet received equivalent attention (Thrush *et al.*, 2008). Nevertheless, evidence is amassing that seabed habitats throughout the world's oceans are being impacted by physical destruction or selective removal of habitat-forming species, and seabed habitats are being homogenized. Bottom trawling is a major contributor to this disturbance both on the continental shelf and in deeper ocean habitats (e.g. Thrush and Dayton, 2002; Thrush *et al.*, 2006).

Thus, understanding the degree to which different locations on the seabed landscape are interconnected becomes especially important when we consider the isolation or fragmentation of habitat patches. Recently, the prevailing concept that all marine ecosystems

are open and composed of highly dispersive species has been challenged, indicating that marine ecosystems may be more sensitive to cumulative impacts than previously thought (Thrush *et al.*, 2008, 2013). Habitat loss, fragmentation, and homogenization of natural communities alter the patterns of connectivity, potentially isolating populations and communities and limiting them to sub-optimal habitats (Tilman *et al.*, 1994; Thrush *et al.*, 2008). The problem is that measuring connectivity in the marine environment is difficult. We can tag and track large organisms, but for most marine species this is not possible. We can use hydrodynamic models to disperse particles, but often these models do not capture the complexity of topography–hydrodynamic interactions. Furthermore, the modelled dispersion patterns do not usually have the diversity of behaviours of organisms that form most seabed communities. Thus, when considering the recovery and resilience of communities (rather than single populations), we need to develop surrogates for ecological connectivity. β -Diversity when viewed as the component of species richness that connects estimates of local site richness (α -diversity) to estimates of the regional species pool (γ -diversity) is potentially useful in this regard (Thrush *et al.*, 2009, 2013). The idea is that cumulative impacts will, when approaching a threshold, result in increasing variability in α -diversity leading to an increase in β -diversity and a reduction of ecological connectivity between locations (Hewitt *et al.*, 2010; de Juan *et al.*, 2013). This implies some critical feedback between the disturbance regime, changes in the seabed communities and the recovery of disturbed patches. Such feedbacks are important in defining resilience and the potential for regime shifts. Importantly, loss of resilience through the combined and often synergistic effects of pressures can make ecosystems more vulnerable, and the self-repairing capacity of ecosystems should no longer be taken for granted (Folke *et al.*, 2004).

Until we have good data on seabed habitats and how the communities respond to increases in (cumulative) disturbance, we can only rely on theory and small-scale field experiments. Heuristic models of disturbance and recovery dynamics, including the consequences to the spatial arrangements of habitats across landscapes, is a way forward. Such models may highlight the potential for threshold changes due to disturbance rates that outstrip the colonization and growth of habitat-forming species, feedbacks between habitat structure and recovery and dispersal limitation (Pascual and Guichard, 2005; Thrush *et al.*, 2005; Lundquist *et al.*, 2010). These models have a heuristic value in considering the consequences of different disturbance regimes to seabed biodiversity and can assist in participatory decision-making processes such as marine spatial planning.

Not surprisingly, the magnitude and intensity of disturbance associated with natural phenomena and human activities influences the dynamics of habitat patches and seabed landscapes. These effects often involve feedbacks and non-linear processes meaning that individual disturbances can have time and context dependent effects. Changes in the disturbance regime due to environmental processes, anthropogenic impacts, or a combination of the two can lead to major changes in seabed biodiversity and ecosystem function. Without considering processes and properties of populations, communities, and ecosystems, these changes in seabed ecology are likely to lead to surprises for resource managers, users, and the society.

Why changes in fish communities can matter to seabed biodiversity

Loss of a major functional group, such as apex predators, may cause drastic alterations in ecosystem functioning (see Folke *et al.*, 2004).

Overexploitation and collapse of large apex predators are well recognized in marine systems (e.g. Atlantic cod: Frank *et al.*, 2005; large sharks: Myers *et al.*, 2007), but impacts on biodiversity at the ecosystem level remain poorly understood (but see Ellingsen *et al.*, 2015). The overfishing and collapse of cod populations and the decline in other commercially exploited groundfish in the Northwest Atlantic (e.g. eastern Scotian Shelf and Newfoundland Shelf) has coincided with marked increases in abundances of small planktivorous fish species and predatory crustaceans, pre-dominantly snow crab (*Chionocetes opilio*) and northern shrimp (*Pandalus borealis*), once among the primary prey of the benthic fish community (Worm and Myers, 2003; Frank *et al.*, 2005, 2011; Quijón and Snelgrove, 2005). No other fish species in the system has replaced the role of cod or other commercially exploited groundfish species, and there has been a reduction of an entire functional group. Thus, the fish communities have changed (Frank *et al.*, 2011; Shackell *et al.*, 2012). Ellingsen *et al.* (2015) found that α and β diversity in fish communities increased with decreases in cod occurrence, and the fish communities were less homogeneous and more variable in areas where cod no longer dominated. This means that the functional characteristics of the fish species in the system have changed, and this affects the seabed biodiversity. Quijón and Snelgrove (2005) found that predation by snow crab and other crustaceans significantly influenced infaunal composition and abundance. They found that sedentary polychaetes and bivalves unable to escape predation by emigration or burial benefited from the exclusion of crustacean predators in both laboratory and field experiments. Their results indicate that community changes at broad-scales may be taking place in North Atlantic benthic ecosystems as a result of the historical changes that have occurred in predator composition due to cod overfishing. Thus, this is an example of indirect effects of fishing on the seabed biodiversity, and importantly, we know that seabed biodiversity matters to fishers and fisheries managers.

Disturbance and regime shifts

The surprisingly rapid loss of cod from the Northwest Atlantic is commonly cited as an example of a regime shift. Regime shifts are often described by thresholds, phase shifts, step-trends, criticality, rapid transitions, or tipping points (Folke *et al.*, 2004; Thrush *et al.*, 2009). Regime shifts in ecosystems are increasingly reported as a consequence of anthropogenic stress, associated with resource exploitation, pollution, land-use change, and climate/oceanographic change or cumulative effects. In some cases, regime shifts move the ecosystem into an alternative state that itself is resilient to change (Troell *et al.*, 2005; Eriksson *et al.*, 2010). There is growing evidence that interactions between the intrinsic ecological dynamics and chronic, cumulative, or multiple stressor effects can lead to the loss or reduction of resilience and an increased risk of regime shift (Thrush *et al.*, 2009, 2014). As resilience declines, the ecosystem becomes vulnerable, and progressively smaller external shocks can cause shifts (Folke *et al.*, 2004). Gradual changes in external mechanisms, such as fisheries exploitation or climate change, might have little effect until a threshold is reached. Then, surprisingly large shifts occur in ecosystems that might be difficult to reverse (Scheffer and Carpenter, 2003; Scheffer *et al.*, 2005). The changing status of the eastern Scotian Shelf ecosystem exhibited a transition from a “pre-collapse” state dominated by groundfish to a “collapsed” state dominated by crustacean and small planktivorous fish in the early 1990s that was largely a result of overfishing of cod and other groundfish (Frank *et al.*, 2011). The ecosystem on

the eastern Scotian Shelf has failed to recover to its formerly cod-dominated situation, despite the cessation of fishing since 1993 (Frank *et al.*, 2005). Interestingly, in the Gulf of Alaska and southeast Bering Sea there was a transition in the opposite direction in the late 1970s, from a community dominated by crustaceans and small pelagic fish to a community dominated by groundfish (Litzow *et al.*, 2008). This transition coincided with the 1976–1977 climate regime shift of the Pacific Decadal Oscillation (Litzow *et al.*, 2008). Overfishing of crustaceans may also have contributed to this community transition (Orensanz *et al.*, 1998). Thus, both fisheries and climatic perturbations may have important consequences for the composition in ecosystems, including the seabed community composition and biodiversity.

Managing fisheries solely based on single species models and variations in Maximum Sustainable Yield (MSY) models is likely to lead to surprises in ecosystem dynamics. To date, identifying signs of shifts in ecosystems that forecast abrupt future changes remains a major challenge (Thrush *et al.*, 2009). We need to understand the role of key species influencing habitats, recovery and the flux of energy and matter. Such knowledge will underpin the interpretation of indicators of rapid transitions associated with changes in time series as a threshold is approached (Dakos *et al.*, 2012). Nevertheless, the use of time-series data to forewarn regime shifts requires good monitoring data. However, this is further complicated because when indicators have been applied to ecological time series, different indicators warn the transitions in different systems (compare Litzow *et al.*, 2008, Hewitt and Thrush, 2010, and Lindegren *et al.*, 2012). Other empirical approaches include the analysis of ecosystem interaction networks to identify changes associated with the loss of positive feedbacks (Thrush *et al.*, 2012, 2014).

A true ecosystem approach is needed if we want to predict, and eventually avoid, adverse shifts in marine communities (Scheffer *et al.*, 2005), and this include valued fish stocks and the supporting ecosystem services. Non-linear changes in ecosystems profoundly affect the nature of advice needed from scientists to inform society's choices about the use of marine ecosystems. Current models of exploitation often attempt to 'titrate' stress or resource extraction against system state or stocks. The assumption is that the system will respond reasonably consistently to the stress and once a predetermined limit is approached we can dial back on the stressor and the system will start to recover. However, in a system exhibiting resilience, there may be no apparent relationship between ecosystem state and stress detected until the threshold is crossed. This pre-threshold phase may indicate to resource users that the system can handle more pollution or more extraction. However, once the threshold is crossed, the system will change to a different and potentially less valued state. If this transition is associated with changes in the intrinsic dynamics of the ecosystem then hysteresis will markedly slow the rate of recovery. These changes highlight not only the value of better biophysical knowledge but also a need to consider resilience in the management of marine systems (Levin and Möllmann, 2015). We may need new governance structures capable of allowing for informed choices of multiple uses that seek to maintain or restore resilience and respond rapidly to surprises.

Implications of multiple uses and multiple values for ecosystem-based management

When management is narrowly focused, it is often blind-sided to different economic and environmental drivers. Taking a more inclusive systems view is important, because many losses in ecosystem

resilience are the unintended consequences of narrowly focused optimization (Walker and Salt, 2006). Changes in marine ecosystems are driven by multiple factors; some of which can be directly managed, while other factors will require adaptation of our resource use. We use marine ecosystems in many different ways, from the direct and active extraction of living and non-living resources (e.g. fish and oil), to the provision of ecosystem services (e.g. carbon sequestration and denitrification) and the provisioning of cultural and spiritual benefits (e.g. sense of wonder, wildness, and wise stewardship). As we respond to changes in one use, such as fishing, management actions are both likely to affect multiple values in society and be affected by other drivers of change.

In fisheries science, the uptake of different perspectives and types of knowledge has been a slow process. There has been a sharp focus on the research challenges of modelling the population dynamics of single fish stocks, i.e. dealing with single-issue concerns. These problems have led to the development of Ecosystem-based Fisheries Management (EBFM) for industrialized fisheries (Fogarty, 2014). New management frameworks such as EBFM are essentially social experiments. They seek to achieve multiple and often conflicting goals that require trade-offs or sharing of resources by different resource uses. This is very different from business-as-usual (BAU) management where agencies representing different sectors fight their corner, leading to conflicts between sectors (Salomon *et al.*, 2011). This spills over to the use of science to resolve conflicts, where its role is often denigrated to one sectors science being better than another. Often these conflicts do not relate to scientific rigor, ethics, or relevance, but rather on funding, power, and social priority. New developing fisheries would offer opportunities to set the groundwork for new forms of ecosystem management that do not carry such legacy. However, most places in the oceans that can be fished are already utilized, and therefore most management experiments will need to be implemented within the context of the history of local fisheries management.

Environmental problems that are associated with high scientific uncertainty and risk, as well as a multiplicity of conflicting values are often referred to as "wicked". Conventional command and control approaches typically fail to achieve positive management outcomes when faced with wicked problems (Berkes *et al.*, 2003). This problem stems from the multiple interactions in the systems, so that when a management intervention occurs in part of the system, other unintended consequences pop-up elsewhere. This can occur in the biophysical domain of the ecosystem or in cultural, social, or economic aspects. Fisheries are wicked socio-ecological problems and new approaches to knowledge production, management, and governance must be developed to address these problems (Jentoft and Chuenpagdee, 2009; Khan and Neis, 2010; Moser *et al.*, 2012). Emerging approaches often focus on the development of adaptive governance and ecosystem-based management frameworks (Folke *et al.*, 2005).

Ecosystem-Based Management (EBM) is designed for decision making in ecosystems subjected to multiple uses and multiple values (McLeod and Leslie, 2009; Tallis *et al.*, 2010). EBM is basically different from EBFM because EBM does not place one use or value at the centre of its focus or decision making. This is critical because as we explore the potential for participatory processes in the co-production of knowledge and the resolution of problems, we need to consider how power is vested within a group and how inequity will influence processes and outcomes (Cote and Nightingale, 2012). As a socio-ecological construct, EBM depends on environmental, social, and economic context and thus practice and

process will vary depending on place and the spatial scale of management. Nevertheless, EBM has core principles that seek to restore biodiversity, recognize the multi-functional nature of ecosystems, and maintain adaptive capacity of ecosystems. These principles in turn seek to ensure that the system can sustain multiple uses and societal values. One thing is clear in terms of implementing EBM that non-integrative bureaucratic structures do not help because they hinder the design and implementation of systemic solutions. Ecosystem-Based Management is inherently future focused because it fosters pathways to achieving ecological sustainability.

The Barents Sea-Lofoten area is an example of a large ocean area where an integrated EBM plan has been implemented. The process started in 2002, the plan was adopted by the Norwegian Parliament in 2006, and it was updated in 2011 (Norwegian Ministry of the Environment, 2006, 2011; Olsen *et al.*, 2007; Knol, 2010). Clearly, EBM is an ongoing process to provide a framework for managing all human activities (oil and gas industry, fishing, and shipping) that ensure the continued health, production, and function of the Barents Sea ecosystem (Olsen *et al.*, 2007).

Human activity in the Barents Sea has increased. Concurrently, the Barents Sea ecosystem has changed. Here we give some examples of changes that are potentially relevant in the context of seabed biodiversity. The sea ice extent in the northern parts of the Barents Sea is declining and the sea temperature has increased since the 1970s (Michalsen *et al.*, 2013). The Atlantic cod (*Gadus morhua*) population has increased dramatically both in numbers and distribution over large spatial scales in the Barents Sea (ICES, 2011, 2013). A large-bodied predatory crustacean, the snow crab (*Chionoecetes opilio*), has increased significantly in abundance and distribution since it was first recorded in the Barents Sea in 1996 (Agnalt *et al.*, 2011; Sundet, 2014). Furthermore, the red king crab (*Paralithodes camtschaticus*) was intentionally introduced to this region in the 1960s to create a new commercial fishery (Jørgensen and Nilssen, 2011). Knowledge from the Northwest Atlantic suggests that changes in predator composition may affect biodiversity (e.g. Quijón and Snelgrove 2005; Ellingsen *et al.*, 2015). Moreover, based on the fishing pressure in the Barents Sea and the current knowledge gathered during the last 2–3 decades on the impacts of bottom trawling on benthic communities (e.g. Thrush and Dayton, 2002), this implies that bottom trawling is a potential important driver for seabed biodiversity in large areas in the Barents Sea. In addition, the oil and gas industry is expanding in the Barents Sea (Norwegian Ministry of the Environment, 2006; Norwegian Oil and Gas, 2013). Indeed, the emerging petroleum activity has been the main driver for integrated ocean management in the context of the Barents Sea (Knol, 2010). Thus, the Barents Sea ecosystem clearly faces threats from combined disturbance arising from human activities. However, there is a persisting lack of understanding about the consequences of human impact on the ecosystem (Knol, 2010). Will increasing human pressure result in habitat loss, fragmentation, and homogenization? We consider changes in seabed habitats, connectivity, and resilience to be critical elements with regard to the move to EBM. Importantly, long-time monitoring providing high-quality ecological information is needed to gain insights into changes in ecosystem structure, ecological processes, and the services the ecosystems provide (Lindenmayer and Likens, 2010). Knowledge gained from successful long-term monitoring is crucial to be able to predict and avoid adverse shifts in communities. The development of integrated monitoring frameworks is considered to be a central part of the instrumentation of EBM, and this is an ongoing process in the Barents Sea-Lofoten area (Knol,

2013). Experience and challenges from the Barents Sea in the EBM process, such as monitoring, marine spatial planning, and participatory decision making, can inform the implementation of EBM in other ecosystems.

Where a common ocean governance view cannot be supported by integrating individual management agencies, there are activities that can foster more integrated management. These include the implementation of marine spatial planning (Crowder and Norse, 2008; Stelzenmuller *et al.*, 2008; Douvere and Ehler, 2009; Foley *et al.*, 2010). Multiple values imply multiple ecosystem services and multiple functions (Snelgrove *et al.*, 2014). Developing multi-party engagement in defining values and using ecosystem services as a communication bridge to connect values to ecosystem functions can also help identify common futures and shared ecosystem benefits as well as vulnerabilities and impacts (Granek *et al.*, 2009). Ecological systems and the services that they generate can be either degraded and support less biodiversity and a smaller range of human values, or they can be restored. The choice is ours.

Socio-ecological surprises imply we need a transformation in both the diversity of science needed to inform the choices and more integrated governance (Dickey-Collas, 2014). The socio-ecological context of individual ecosystems and the legacy of impact within them will mean the implementation of place-based management. “There is no *panacea* for ecological monitoring, just as there is no one-size-fits-all solution for EBM” (Knol, 2013). However, given the diversity of ways fisheries are managed and integrated with other socio-ecosystem components around the world there are opportunities to learn from successes. We also need new experiments in ocean governance that are accountable, adaptive, and responsive to surprises.

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