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Ecosystem-Based Management, Ecosystem Services and Aquatic Biodiversity

Theory, Tools and Applications

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Part I
Introduction

Using the Concepts and Tools of Social Ecological Systems and Ecosystem Services to Advance the Practice of Ecosystem-Based Management

Timothy G. O'Higgins, Theodore H. DeWitt, and Manuel Lago

Abstract Environmental problems are very often wicked problems: they are persistent, they have no clear end, and involve moral choices resulting in winners and losers. Just as the ecological and biological elements of these problems are dynamic and complex, so the social and political elements are also constantly changing and do not follow linear patterns. Ecosystem-Based Management (EBM) is an approach developed to work on wicked problems that recognizes social-ecological systems and the need to incorporate systems thinking into natural resource management. In this chapter we describe the scope and scale of this book and briefly discuss its four sections:

- foundational concepts
- tools for the practice
- national and international governance contexts
- case studies.

We then go on to identify some of the main lessons learned, challenges and the main needs required to further advance the applications of EBM. We conclude with an exhortation for readers to learn from our experience, to use and adapt the tools and techniques we present here and a call for continued international collaboration.

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1 The Problem

Environmental problems are ubiquitous. At the local scale, human activities gradually supplant natural environments with local environmental consequences, and summing all these localised activities has resulted in global-scale crises in biodiversity and climate. These problems are vast in scale, can seem overwhelming, and have major consequences for human well-being. Destruction of aquatic ecosystems is of particular concern. Marine ecosystems, which make up the largest part of the globe, are coming under increasing pressure, and freshwater ecosystems are experiencing biodiversity loss more rapidly than their terrestrial counterparts (Dudgeon et al. 2007; Halpern et al. 2008; EEA 2010). It is easy to point fingers at human population growth, economic systems, industrial sectors, governments, and society as a whole for their role in the gradual but accelerating erosion of ecosystems, biodiversity, and the benefits they provide. Recognising the problems caused by degradation and destruction of the environment and identifying the social causes of environmental damage are essential in raising public awareness, but this is only the starting point for environmental management.

Environmental problems are very often wicked problems: they are persistent, they have no clear end, and involve moral choices resulting in winners and losers. Just as the ecological and biological elements of these problems are dynamic and complex, so the social and political elements are also constantly changing and do not follow linear patterns. For example, the phenomenon of eutrophication involves interactions between natural processes such as weathering, nitrogen fixation, decomposition and mineralisation, as well as anthropogenic factors such as wastewater discharge, fertilisation and animal feeds, all operating at different rates and over a range of spatial scales from catchment to ocean basin. While the processes and activities contributing to eutrophication may occur in one location, their effects may be felt in another, resulting in winners and losers from diverse sections of society including, for example, farmers and city dwellers who depend on different aspects of the environment (rivers, lakes, coastal zones) for a range of different uses from drinking water extraction to recreation, all of which are underpinned by biodiversity. To complicate matters further, these users may be in different jurisdictions regulated by different economic and political systems, and with different sets of values and economic needs.

While the early environmental movement typically focussed on humans and their impacts on the environment (e.g., Ehrlich 1968; Ehrlich and Holdren 1971; Hardin 1974), solutions tended to be simplistic. For example, Hardin (1968) identified the problem of the "Tragedy of the Commons", recognising the lack of incentive for conservation actions in the management of common pool resources and identifying assignation of ownership rights as a potential regulatory solution. More recent empirical studies on the factors leading to successful natural resource management regimes have led to the concept of Social-Ecological Systems, recognising that any given resource management problem is comprised of subsystems including resources, users, institutions and rules (Östrom 2009) and enabling the identification of characteristics of successful resource management regimes (Östrom 1990). While research into social-ecological systems has provided a very promising direction for improved environmental management and while the potential for this new, more holistic approach to environmental management, generally termed Ecosystem-Based Management (EBM), has been widely recognised, to date there has been limited progress in incorporating such practice into large scale policy. For example, the mid-term review of the European Biodiversity Strategy (EC 2015) indicates that the strategy is failing, largely due to the lack of inter- or transdisciplinary knowledge and suitable assessments to inform policy choices on ecosystem restoration options. Despite high-level international commitments to reverse declines in biodiversity under the Convention on Biodiversity, declines in environmental quality, biodiversity and ecosystem services are a global phenomenon.

The goal of this book is to examine the current state of the art of holistic and collaborative techniques to address wicked environmental problems through the application of Ecosystem Based Management, and describe how these techniques can be effectively implemented at multiple spatial scales.

2 Emerging Solutions

Ecosystem-Based Management (EBM) is an approach developed to work on wicked problems that recognises social-ecological systems and the need to incorporate systems thinking into natural resource management. EBM takes the perspective that human social systems are contained within and completely dependent on the broader ecological system, and works backwards from the problem to identify the causes and actors. EBM also recognises and incorporates the ecological complexity associated with environmental problems and the interdependencies of organisms (including humans) and ecological processes, as well as the potential for multiple interacting causes of specific problems. As a working definition, we consider that EBM "describes the comprehensive integrated management of human activities based on the best available scientific knowledge to achieve sustainable use of ecosystem goods and services and maintenance of ecosystem integrity" (Le Tissier 2020). There are many other definitions of EBM with a variety of different emphases; Delacámara et al. (2020) synthesize the high level characteristics that define EBM as a distinct management approach. A common factor in much of EBM is the inclusion of different stakeholders to understand the needs and behaviours of different groups, to identify trade-offs and develop consensus. In this book, we provide a snapshot of the current state of the art of EBM, including the main concepts, conceptual frameworks, tools, legal frameworks and specific examples of application to a range of different study sites, mainly from Europe and North America. This book marked a unique transatlantic collaboration, bringing together cutting-edge science from a range of EU research institutions under the umbrella of the AQUACROSS project and from the US EPA ecosystem research community. Our research was driven by a common need for ecosystem-based approaches to similar environmental problems. We recognised the value of bringing together our approaches and experiences. The authors of each chapter have identified key lessons and outcomes of their studies as they pertain to the development and implementation of EBM as well as key needs for the advancement and application of EBM.

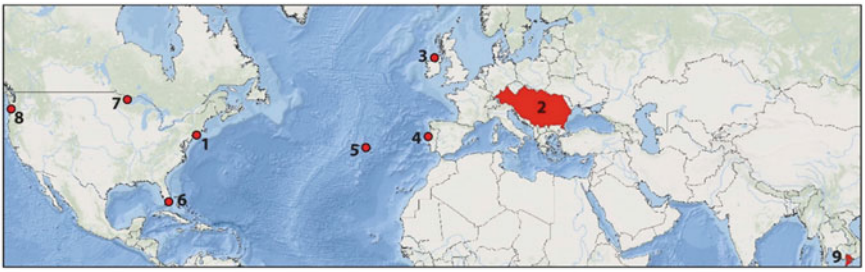
3 Structure

The book is divided into four sections. Section 1 deals with foundational concepts such as the definition of EBM itself, the concept of ecosystem services, and the development of conceptual frameworks that connect human activities and ecosystem components. At the practical level, any effective management programme requires a clearly defined process and a series of discrete and replicable steps. When managing complex systems whose behaviour is not fully understood, the requirement for adaptive management, or "learning by doing" by which a cyclical management process can adapt to the changing conditions of the system (Holling 1978) has long been recognised. In this volume, based on experiences with implementation of EBM in the North Sea, Piet et al. (2020) set out their vision of an effective EBM process, which can provide a template for other areas. As with EBM definitions, a variety of standard processes have been put forward. DeWitt et al. (2020) describe a process of structured decision making designed specifically to enable the consideration of ecosystem services in EBM. While the choice of formal management systems may be dependent on the institutional and regulatory culture in a specific context, the management processes described by Piet et al. (2020) and DeWitt et al. (2020) share a number of common factors that set them apart from traditional management approaches and illustrate the emerging consensus on EBM best practices. Both studies follow the well-established Plan-Do-Check-Act cycle, and although the implementation, monitoring and evaluation steps are generic, the types of activities used during the Planning Phase are what characterise EBM and involve an holistic and inclusive perspective that is often stakeholder-led. These activities include a variety of models and techniques to connect different elements of social- ecological systems.

A particular challenge of systems approaches is the need to integrate knowledge from ecological and social sciences. Ecosystem service concepts and tools provide the analytical basis to connect the materials and processes of the ecological system with the needs and wants of the social system. Flood et al. (2020) introduce ecosystem services, providing an overview of the development of ecosystem services concepts and the promises and pitfalls of their application. Culhane et al. (2020) focus on the supply of all types of ecosystem services, with examples of how these concepts have been practically applied in a variety of different contexts. DeWitt et al. (2020), Russell et al. (2020) and Bousquin and Mazzotta (2020) each explore the relationship between the production of services by ecosystems and the subset of services that are directly used, enjoyed or appreciated by people, known as Final Ecosystem Goods and Services (FEGS). Understanding both the supply and demand side of ecosystem services is central to EBM; this is reflected in the "Butterfly" conceptual framework (Gómez et al. 2016; Delacámara et al. 2020; Elliott and O'Higgins 2020) employed throughout much of this volume. One commonly used framework, the Driver-Pressure-State-Impact-Response, has been widely used and adapted for the analysis of social-ecological systems. Elliott and O'Higgins (2020) describe the evolution of this framework as well as its latest adaptation, which has been applied in many of the studies presented here (e.g. Culhane et al. 2020; O'Higgins et al. 2020; Lillebø et al. 2020; McDonald et al. 2020).

Section 2 describes tools for ecological modelling, stakeholder engagement and analysis of ecosystem services that can support different steps along the EBM process. Tools to help implement EBM come in many forms. Linkage frameworks (Robinson and Culhane 2020) retain the complexity of human interactions while helping to elucidate the connections between human activities and environmental impacts, and can be extended to explore human effects and risks to ecosystem service supply. These linkage frameworks can provide a useful tool for exploring and communicating the complexity of social-ecological systems, particularly in situations where quantitative data are scarce. Ecological models can provide more quantitative approaches connecting ecosystem condition to the production of ecosystem services and their benefits. Developing and applying such models can be a complex and challenging process. Fulford et al. (2020) provide insights into the process of model development and many other chapters provide examples of model applications that vary in their data requirements, modelling approaches and complexity (e.g., Funk et al. 2020; O'Higgins et al. 2020; Lillebø et al. 2020). Decision support systems can be used to integrate multiple models, and include ENVISION, explored by McKane et al. (2020), ARIES and InVEST (Lewis et al. 2020; Funk et al. 2020; Lillebø et al. 2020). Many of the chapters in the tools section, which focus on demand side aspects of the EBM process, have a particular focus on transparency and replicability (e.g., Sharpe et al. 2020; Russell et al. 2020). This focus is of particular importance for mainstreaming of EBM, where subjective human values, as well as objective, measurable ecological parameters, are involved in public decision making.

Decision contexts for environmental problems are location specific; understanding the legal basis and background of environmental law both enables and sets constraints on the scope of management. Section 3 deals with the International, US and European basis for the governance of social-ecological systems. Enright and Boetler (2020), O'Hagan (2020) and Harwell (2020) explore the legal and institutional context for EBM internationally, within Europe and in the US respectively, while Le Tissier (2020) explores how the legal origins of related management approaches and tools, such as integrated coastal zone management and marine spatial planning, have shaped understanding of EBM. The international conventions and treaties that have given rise to the prominence of the EBM concept play out in different ways between European and US jurisdictions. Understanding these institutional contexts is also vital in the selection of appropriate tools that can support policies and practices at national and local scales.



1. Long Island Sound, New York, U.S. (Myer and Johnson, 2020)
2. The Danube Catchment (Funk et al., 2020)
3. Lough Erne, Northern Ireland (O'Higgins et al., 2020)
4. The Vouga Estuary, Portugal (Lillebø et al., 2020)
5. Falial Picos Channel, Azores (McDonald et al., 2020)
6. The Everglades, Florida, U.S. (Gibble et al., 2020)
7. Duluth, Minnesota, U.S. (Williams and Hoffman, 2020)
8. Tilamook , Oregon, U.S. (Sharpe et al., 2020)
9. Mekong Delta, Vietnam (Johnston et al., 2020)

Fig. 1 The geographic diversity of place-based studies addressed in this volume

Section 4 brings together a selection of cutting-edge case studies of EBM from diverse geographic and environmental settings (Fig. 1), from the Great Lakes of North America to the Mekong Delta in Vietnam, from marine and estuarine systems to freshwater rivers, lakes and wetlands of the Danube catchment (the most international river basin in the world).

These case studies explore the approaches and challenges of implementing EBM and they range from simple, but effective, common-sense, mapping and communication of ecosystem services (Myer and Johnston 2020) to applications of highly sophisticated ecological models. For example, a more sophisticated model might use artificial intelligence combined with Geographic Information Systems to generate practical management solutions (Funk et al. 2020) or complex statistical techniques to integrate ecosystem services into long-term climate adaptation (Johnston et al. 2020). Some case studies illustrate how approaches to EBM have evolved over time, adapting and bringing together several strands of existing activity (e.g., Gibble et al. 2020; Williams and Hoffman 2020; Funk et al. 2020), others describe initial steps toward integration of ecosystem services into resource management (Lillebø et al. 2020). In data-rich cases, different methodologies have been tested on the same system (e.g., Funk et al. 2020), whereas in data-poor cases, simple stakeholder opinions on system function have been used to develop models that can inform ecosystem management (O'Higgins et al. 2020).

4 Lessons Learned

Overall, the various tools, techniques and case studies demonstrate a variety of parallel approaches to developing ways of managing complex, adaptive social-ecological systems. In combination, these studies have revealed a number of important key lessons learned.

- *Mixing models and multidisciplinary approaches*

EBM requires inputs from multiple disciplines each with their own tools and techniques. Methods for combining models of different sub-systems within the social-ecological system are vital to EBM, but the types of models best employed depends on the availability of data and expertise. One size doesn't fit all; however, the toolbox of methods highlighted in this book are usable across a wide range of decision contexts and problem complexities, ecological and social settings, and spatial and temporal scales. Suitable tools range from the highly detailed, complex and computationally intensive models to simple box and arrow models based on expert opinion. The best model or approach is context dependent.

- *Include diverse stakeholders in all stages of problem formulation and assessment of solutions*

Many different groups are affected by environmental decisions and successful EBM can elicit and incorporate information from different stakeholders to develop socially acceptable solutions. The appropriate types of information are required to communicate with appropriate stakeholders. While environmental information may be most important to conservationists, economic values can be persuasive tools in the development of management options.

- *Recognition of ecosystem services*

Understanding the links between human welfare and ecosystem integrity is a vital component of EBM, and explicit consideration of ecosystem services, whether focussed specifically on those providing direct human benefits (and their valuation) or qualitative assessments including the full suite of all ecosystem services, offers a means of incorporating these considerations into management. Valuation can be extremely useful but is not the only aim of ecosystem services assessment.

- *Problem specific solutions*

Each social-ecological system is unique, and the type of management best suited to each system depends on the characteristics of the system itself. Deploying a subset of tools and techniques best suited to a particular situation can enable problem specific management solutions, but requires consideration of the amount and type of data, as well as the social and ecological context.

5 Challenges and Needs to Advance EBM

EBM is an evolving field and in need of new knowledge and methods to further its success. At the beginning of each chapter, authors have identified up to three "needs for advancing EBM".

Several of the governance chapters point to the need for enhanced clarity as to what the process of EBM entails, standardisation of EBM approaches, the provision of guidance and a clarification of EBM concepts (Enright and Boetler 2020; O'Hagan 2020; Le Tissier 2020). This need for clarity is also reflected in many of the chapters describing tools, as well as in case studies (e.g., Harwell et al. 2020; Sharpe et al. 2020; Lewis et al. 2020), which advocate for structured and documented methods for incorporating specific aspects of EBM into practice.

Several of the European case studies identified the need for harmonisation and coordination of policy objectives across major European Directives (O'Hagan 2020; Lillebø et al. 2020) to develop more integrated approaches to food security and environmental conservation. Integration of EBM with administrative and legal frameworks can help embed EBM into mainstream environmental management. On the one hand, there is a role for scientists in defining EBM, to this end, standardized methodologies such as those described here (e.g. Piet et al. 2020) may help; on the other hand, there is an onus on scientists to understand and adapt to the legal frameworks set down. How can decision makers be expected to use EBM when it is poorly defined in law? The chapters in Section 3 illustrate that laws and environmental directives are often not well integrated and this may result from the legacy of single sectoral approaches. Harmonization of the new more holistic scientific approach with the practice and policy of natural resource management needs to be achieved if EBM is to become widely adopted.

This need for standardisation and harmonisation, to a certain extent, leads to a tension with the need for a stakeholder-driven process. Lillebø et al. (2020), McDonald et al. (2020), Williams and Hoffman (2020) and O'Higgins et al. (2020) all stress the requirement for co-design of the EBM process in collaboration with stakeholders and the development of problem-specific solutions. In the situation where every social-ecological system is different, comprised of unique ecological, social and political contexts and characteristics, the flexibility of stakeholder driven processes enables the development of unique solutions to unique (wicked) problems. At the same time, this requirement for flexibility inhibits our ability to proscribe one-size-fits-all standardised methodology. This situation requires a clear set of logical steps, which can be conducted in a flexible fashion. To this end, we hope that the tools and cases presented here can provide some examples that can be organised and re-arranged to meet practical management needs and conditions.

In order to embed EBM in practical natural resource management, there is a clear need for improved communication and capacity building, and this is identified by several authors herein (Russell et al. 2020; Myer and Johnston 2020; Williams and Hoffman 2020). Harwell et al. (2020) directly address the issues of strategic communication with respect to individual projects. The issue of capacity will require learning-by-doing, and we hope that our experiences in developing tools and applying methodologies will provide useful examples and ideas for future applications.

6 Conclusions

While the studies in this book illustrate that there is an emerging consensus amongst experts in best practices for EBM, there is room for improvement in the development of methods to implement EBM. Though considerable progress has been made in development of techniques and tools for developing stakeholder consensus, for incorporating multiple values into management, and for modelling and predicting flows of ecosystem services, embedding these practices into standardised practical methodologies that meet conservation and legislative standards remains a major challenge. This will require redoubling of effort and can no doubt be enhanced by similar collaborations in the future.

We have found that our research has been considerably enriched by identifying similar EBM initiatives being conducted under different social and institutional settings. We were pursuing parallel paths and recognised that by bringing together the parts and their applications, we could identify emerging best practices that are beginning to define the science of EBM. While the complexity of addressing social-ecological systems and the challenges of multi-disciplinary research may seem daunting, we encourage readers to embrace the idea of adaptive management, of learning-by-doing. Many of the techniques and tools identified here are low cost or open source and can be applied to an array of environmental challenges. Each of the tools and case studies presented has particular strengths. Readers considering the implementation of EBM practices are encouraged to identify areas where these tools can be combined in new ways to complement each other. We encourage the reader to follow the links to the various tools, to play with and familiarise themselves with their capabilities and to contact the authors who are (mostly!) approachable. We hope that they can, as we have, develop a new understanding of the power and promise of holistic Ecosystem-Based Management.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Part II
Foundational Concepts

Advancing Aquatic Ecosystem-Based Management with Full Consideration of the Social-Ecological System

Gerjan Piet, Gonzalo Delacámara, Marloes Kraan, Christine Röckmann, and Manuel Lago

Abstract In this study we present an integrated Ecosystem-Based Management (EBM) approach that attempts to reconcile several concepts including integrated ecosystem assessment (IEA), marine spatial planning, resilience thinking, and complex adaptive systems. The approach builds on the IEA process but enhances it by explicitly considering the full social-ecological system (SES) and the creation of a generic framework for assessment of ecosystem status and management strategy evaluation.

Lessons Learned

- This approach reconciles many existing concepts that describe the ecological system, the social (or socio-economic) system and EBM into a unifying approach
- It consists of concrete steps which identify issues for the practitioner to consider, gives examples that provide the basis for a common framework,
- It provides guidance on how to make the framework (more) operational and is applicable to any aquatic ecosystem

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- It allows the incorporation and synthesis of interdisciplinary information on SES into practical and useful linkage frameworks for EBM plan development and implementation

Needs to Advance EBM

- Where consideration of the full SES can be overwhelmingly complex leading to inaction, we propose to work with a subsection of the SES (called subSES). This subSES can then be the starting point for building the knowledge base for EBM decision making. We provide practical guidance how to construct such a knowledge base for both the ecological and the social system.
- The subSES, in conjunction with the available knowledge base, then drives the development of the knowledge base and determines the type of risk assessment(s) that can be applied. Science should then inform the process to translate the high-level societal goals into operational objectives, identify the main barriers that prevent achievement of these objectives, and guide the relevant authorities that develop an EBM plan.
- A novel component of this approach is that the EBM plan distinguishes between management measures (interacting with the ecological system) and policy instruments (interacting with social processes) that together harness the knowledge base of the subSES.

1 Introduction

Conventional management of aquatic resources, based on a specific policy or directed to a sub-sector or flagship species, has often failed to deliver on the policy objectives (Long et al. 2015). As an alternative, Ecosystem-Based Management (EBM), which has a more holistic understanding of the ecosystem and its linkages, is widely accepted as the key concept to guide contemporary decision-making (Börgstrom et al. 2015; Cormier et al. 2017). The conventional management perspectives have assumed, either implicitly or explicitly, that effective policy making is hindered by lacking or inadequate knowledge of ecological processes, functions, and services (Ruckelshaus et al. 2009). Such an ecological focus has failed to produce the full picture of best available knowledge for effective decision-making (Christie 2011). Cormier et al. (2017) distinguish decision-making, which is essentially a specific choice among alternatives, from policy-making, which is a process of identifying a problem and setting societal goals and objectives. In practice, the policy is implemented through a management plan expected to 'carry into effect' the policy objective. An effective policy cycle requires the incorporation of social and institutional processes, such as the involvement of various institutional actors (Röckmann et al. 2015) and an understanding of the governance context (Bissix and Rees 2001). The need for a more holistic approach that incorporates ecological and socio-economic factors (Bianchi 2008), however, can result in in-action due to overwhelming complexity (DeFries and Nagendra 2017).

As a paradigm, EBM addresses uncertainty and complexity, it is an interdisciplinary visioning of multiple objectives, and as such, EBM can be categorised as a 'wicked problem' (Berkes 2012). 'Wicked problems' have no definitive formulation, no clear stopping rule and no objectively right or wrong solutions and no final resolution (Rittel and Webber 1973); (Ludwig 2001). DeFries and Nagendra (2017) suggest the following solutions for addressing wicked problems: multi-sectoral decision-making; institutions that enable management to span across administrative boundaries; adaptive management; markets that incorporate natural capital; and collaborative processes to engage diverse stakeholders and address inequalities. Integrating environmental and socio-economic processes (including institutional, ethical and cultural) requires a single conceptual framework (Christie 2011). The concept of social-ecological systems (SES) can provide such a framework (De Lange et al. 2010). This SES can be split into smaller social-ecological sub-systems (subSES) to address the issue of complexity.

Relevant information and processes can be considered in individual compartments that form the SES. Applying the concept of SES in aquatic EBM implies that any distinction between social and natural systems is artificial and arbitrary by definition, because they are connected (Berkes 2012).

Socio-Ecological Systems can be considered complex adaptive systems (CAS) (De Lange et al. 2010) as the structure, functions, and dynamics of CAS emerge from the interaction and connectedness of the system's constituent parts and with other systems (Hagstrom and Levin 2017). By acknowledging that SES are CAS, management can overcome the drawbacks of conventional approaches. Instead of searching for optimal solutions, linear dynamics, or marginal changes under complete information, a shift towards a more dynamic management approach can be made, where non-linear changes, uncertainty, and surprise are intrinsic characteristics of the system. In addition to the known unknowns, e.g., lack of historic data of species, CAS come with new uncertainties that cannot be tackled through standard sensitivity analysis (Polasky et al. 2011).

Sustainability is among the ultimate objectives of EBM (Long et al. 2015). Ostrom (2009) identified four core subsystems of SES sustainability: (1) resource systems; consisting of (2) resource units; (3) governance systems; and (4) users. When modelling a SES, it may be worth distinguishing these subsystems. In the face of ongoing changes and their uncertain consequences as well as exposure to uncertain shocks, the key to sustainability is enhancing the resilience of a SES (Folke et al. 2005; Nelson et al. 2006; Biggs et al. 2015). Enhancing resilience in terms of persistence, adaptability and transformability (Folke et al. 2010) means preserving the SES's adaptive capacity in order to remain within a certain range of conditions that meet the sustainability goals. Resilience thinking promotes governance frameworks that are able to reconcile the conflicting interests and visions of different stakeholders in a transparent and accountable way so as to foster cooperation among them and enhance stakeholders' ability to commit to legitimate and transparent policy objectives (Dietz et al. 2003). In addition, these governance frameworks should also pave the way to achieve collectively agreed goals through robust institutions, with stakeholders who are able to regularly adjust to changes in the ecological and the social-economic systems (Nelson et al. 2006).

Linkage frameworks such as DPSIR, Driver-Pressure-State-Impact-Response (OECD 1994; EEA 1995; Elliott 2002), are commonly used in the context of environmental management to describe how human activities impact the state of the ecosystem (Halpern et al. 2008; Knights et al. 2013) and hence the supply of ecosystem services to human well-being (Elliott et al. 2017). Linkage frameworks rely on accurate descriptions of linkages (e.g., stressor-receptor or pressure-state relationships) and can be informed by qualitative, quantitative, or expert judgement-based assessments, or any combination of these (Knights et al. 2014). Such linkage frameworks have been applied in an EBM context to guide the selection of management measures and their evaluation using risk-based approaches (Knights et al. 2015; Piet et al. 2015; Borgwardt et al. 2019; Teixeira et al. 2019; Culhane et al. 2019).

Few, if any, examples of EBM planning initiatives informed by advanced science have been implemented across multiple sectors (Katsanevakis et al. 2011; Cormier et al. 2017). In order to enhance the use of salient science (Röckmann et al. 2015) into the policy-making process, Cormier et al. (2017) proposes four steps: strategic goal setting; tactical objectives; management measures; and adaptive management. Integrated ecosystem assessments (IEAs) have also been proposed as a tool to operationalise EBM as they provide a framework for organizing science in order to inform decisions in marine EBM at multiple scales and across sectors (Levin et al. 2009a; Walther and Möllmann 2014; Tallis et al. 2010; Harvey et al. 2017). It is therefore not surprising that the IEA process described by (Levin et al. 2009b) is closely aligned to the EBM processes described by Börgstrom et al. (2015) or Ansong et al. (2017). The (further) development and operationalization of an IEA/EBM approach is however hindered by the lack of a systematic, critical appraisal.

In this study, we merge the existing IEA and EBM processes into a single generic approach that can be applied in all aquatic systems. This approach is then enhanced by explicitly considering the full social-ecological system (SES), which incorporates both environmental and socio-economic processes (Christie 2011). In order to advance this process, we provide the systematic criteria that allow critical appraisal of its progress based on the aspects of EBM, identified in the review by Long et al. (2015) together with additional practical guidance that emerges from concepts such as resilience thinking and CAS. Further, we provide operational guidance on the development of relevant subSES to inform this approach.

2 Advancing EBM

EBM should be able to guide decision-makers by identifying trade-offs between societal goals (Walther and Möllmann 2014). These trade-offs may involve the choice between the conservation goals of specific ecological components (Aanesen et al. 2014), between ecological and socio-economic objectives (ICES 2017), between different ecosystem services (Turkelboom 2017; Dick 2017), or the conflicting interests of specific stakeholders. While decision-making is often primarily aimed at effectively achieving specific conservation goals it ultimately involves socio-economic considerations including: (1) the sharing of costs and benefits among stakeholders; (2) the balance between short- and longer-term benefits; (3) the need to forgo current rents in exchange for future security; and (4) local opportunity costs and regional and global benefits.

EBM should be considered an incremental process as opposed to a single, giant leap away from traditional management (DeFries and Nagendra 2017; Borgstrom et al. 2015). As EBM revolves around a cyclical process, advancements can be made with every iteration of the adaptive management cycle. In this study, we adopt the IEA process and enhance earlier frameworks by incorporating the SES concept

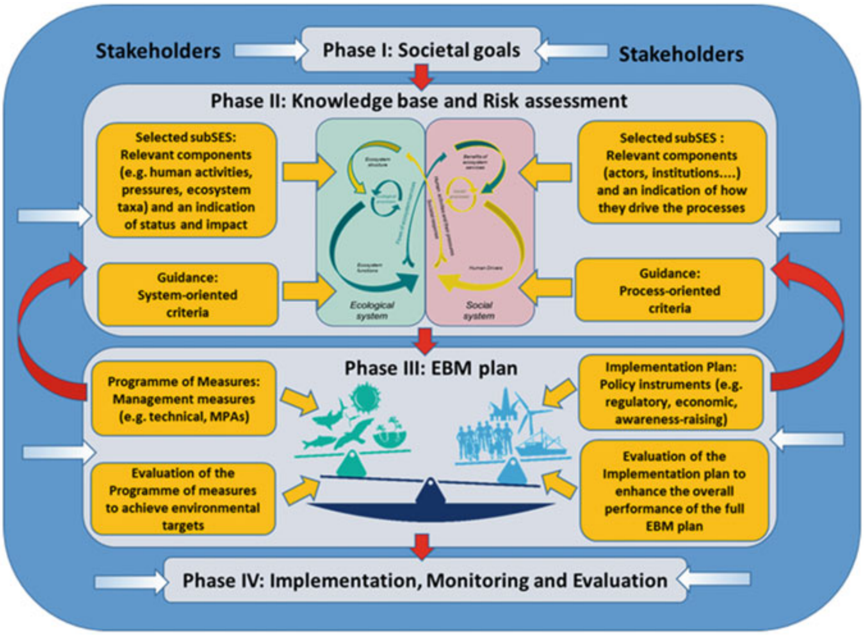

Fig. 1 One cycle of the adaptive, cyclical ecosystem-based management approach built around a balanced representation of the selected social-ecological sub-system (subSES). The figure depicts the phases occurring in the science domain (identified in Fig. 2) but identifies where in each phase cross-domain interaction occurs (i.e., stakeholder involvement) with the wider society. In each phase, the main contributions of this study to advance EBM are indicated

(Fig. 1) to shape the knowledge base and by providing practical guidance that allows an appraisal of progress. We assess progress in developing the knowledge base by looking at the key principles of EBM according to Long et al. (2015) and their alignment to relevant concepts such as IEA (Levin et al. 2009b; Samhouri et al. 2014), wicked problems (DeFries and Nagendra 2017), EBM phases (Borgstrom et al. 2015), policy-making processes (Cormier et al. 2017), ecosystem-based marine spatial planning (Ansong et al. 2017), and resilience thinking (Folke et al. 2010). We build a common framework around these concepts and organise them into four operational phases (identification of societal goals; developing the knowledge base and risk assessments; EBM plan development; implementation, monitoring, and evaluation), which differ in the scientific expertise required.

2.1 Phase I: Identification of Societal Goals

Phase I involves the scoping of societal goals, policy objectives, and perceived threats form the starting point of the IEA (Levin et al. 2009b, 2014) and EBM (Ansong et al. 2017; Cormier et al. 2017). It includes the analysis of policy synergies, conflicts, and an understanding of opportunities and challenges for developing EBM alternatives as well as stakeholder participation to help identify and prioritise among them. A possible issue is that policy objectives, mostly applying to global or regional scales, often refer to conditions of the ecological system only while at the local level, the objectives often aim to restore the sustainability of the whole SES. This may require reconciling objectives at different scales.

2.2 Phase II: Setting up the Knowledge Base and Conducting a Risk Assessment

Phase II builds on the inventory of societal goals in the previous phase and identifies relevant social-ecological sub-systems (subSES), similar to the 'focal SES' (Ostrom 2009). Such subSES should consist of one or more linkages in the linkage framework tied to one or more societal goals and are the basis to elaborate those subSES into what is to become the EBM knowledge base (Fig. 2), and which may contain both qualitative and/or quantitative information. The linkage framework is constructed using consistent typologies of the activities, their pressures and the ecosystem components affected by them (see e.g. Knights et al. 2015; Borgwardt et al. 2019) and with categories that resonate with stakeholders. If needed hierarchical typologies can be applied. For example the activity 'fishing' can be divided into specific types of fishing (e.g. demersal or pelagic) or an ecosystem component can be divided into functional groups or even species. The application of the linkage framework should ascertain that those subSES cover all the elements that matter to

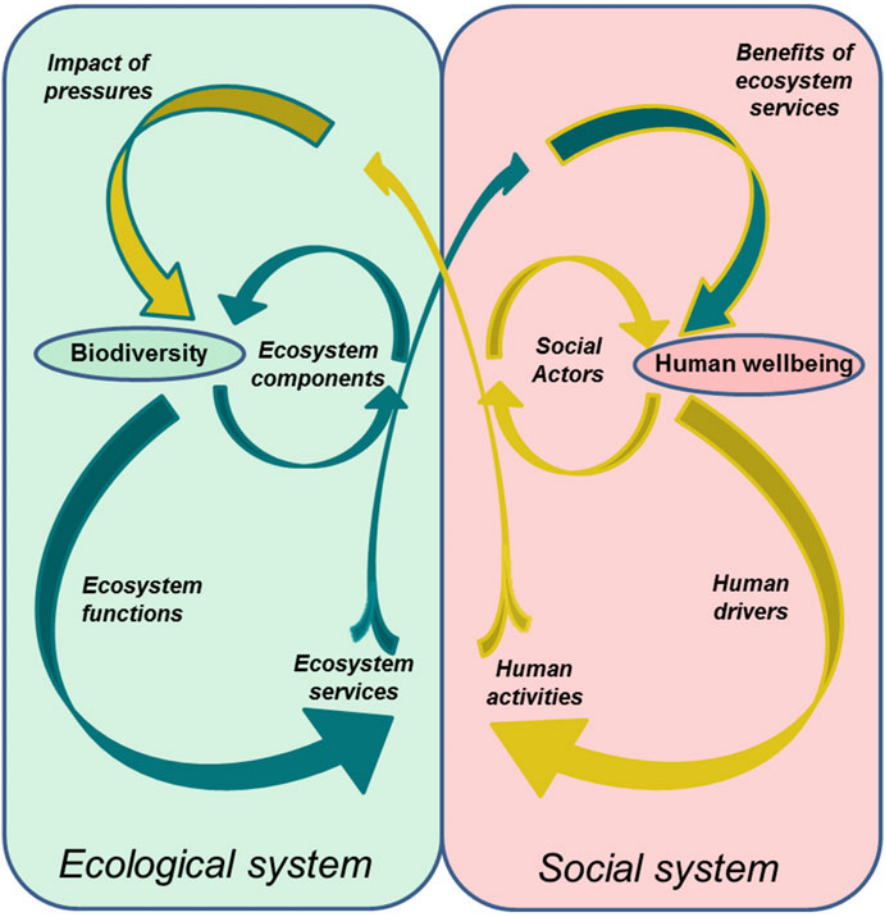

Fig. 2 The social-ecological system (adopted and modified from (Gómez et al. 2016)) consisting of an ecological system-based supply-side where the flow of ecosystem services into the social system contributes to human well-being and a social-system-based demand-side where the human activities and their pressures impact the ecosystem components and their functioning

the problem identified in Phase I, as well as any additional issues that have emerged through stakeholder consultation. Note, however, that with the introduction of more detailed categories, the complexity of the linkage framework (i.e. number of linkages) and thus information demand increases.

Different types of data, generally covering different subSES, are to be used in EBM phases II and III. Information on SES complexity is a requirement for diagnosing why some SES are sustainable while others are not (Ostrom 2009). Therefore, the inherent complexity of the SES and subSES should be harnessed, rather than eliminated. In practice, this implies that different subSES can be analysed at the appropriate level of detail without being hindered by the complexities and requirements of the full SES. At the same time the full SES, in which this subset operates, helps with interpretation of the outputs and their communication to stakeholders.

Table 1 provides a suite of system-oriented criteria to determine the suitability of the (sub)SES knowledge base to describe the ecological system and guide EBM. A similar list of process-oriented criteria for the social system is given in Table 2. As IEA/EBM is supposed to be an adaptive process, the aim is to gradually improve the knowledge base of the ecological system or the institutional set-up of the social system against those criteria in each of the subsequent iterations of the EBM cycle.

With the structure of the knowledge base established, we can consider the three IEA steps: (1) development of ecosystem indicators; (2) identification of reference levels; and (3) conducting risk analyses (Levin et al. 2014). For risk analysis, we suggest to consider the Environmental Risk Assessment (ERA) approach, which distinguishes the different levels of risk analyses and classes of system complexity (Holsman et al. 2017). Level 1 of the ERA consists of a qualitative evaluation that is often based on expert opinion. Level 2 consists of semi-quantitative ERAs based on estimates of exposure and severity, and Level 3 consists of fully quantitative assessments based on a mechanistic understanding of the system (see Holsman et al. (2017) and Stelzenmüller et al. (2015) for additional examples). The complexity of the system can be described using three nested classes. Class 1 encapsulates the direct impact of a single pressure on a given social or ecological subject (e.g., bottom trawl fisheries catching cod). Class 2 measures the direct and indirect effects of a single pressure on multiple interacting subjects (e.g., fishing impacts on ecosystems as in (Hobday et al. 2011)) or the effects of multiple pressures on a single subject. Class 3 refers to the direct and indirect effects of multiple interacting pressures on multiple interacting subjects (e.g., bottom trawl fishing, dredging and contaminants affecting the seabed habitats and the fish foraging there).

It is important to distinguish between the overall, less detailed ERA of the whole SES and the detailed ERAs conducted on the subSES. Within the overall ERA of the whole SES, the qualitative or semi-quantitative ERAs of the full SES (i.e., Class 3) can be used to identify the linkages that introduce the greatest risk to the ecosystem (aligned to the threats in Phase I). These Level 1 or 2 ERAs are primarily aimed at guiding decision makers on which sector-specific management measures to focus (Cormier et al. 2017).

The levels of risk analyses and classes of system complexity described above determine the type of ERA required. Formal description of the SES, through a linkage framework, can provide further characterisation of the classes. This linkage framework can then also be used to assess the quality of the ERA for each level of the analytical tools. Ultimately, the aim is to advance from a qualitative ERA towards an increasingly quantitative ERA that is more elaborate and realistic in terms of the 'reciprocal and cumulative interactions among multiple (interacting) pressures and multiple interacting subjects' (Holsman et al. 2017) that make up the full SES. Distinguishing the subSES and the full SES, each with complementary ERAs can reduce the level of complexity in order to ensure salient information into the policy cycle (Folke et al. 2005; Biggs et al. 2015).

Table 1 Ecological EBM criteria based on the EBM principles in Long et al. (2015) for assessing the knowledge base of the ecological part of the social-ecological system (SES) and hence the core sustainability SES subsystems: resource systems and resource units (Ostrom 2009). Guidance is provided to assess to what extent the knowledge base has advanced in order to support EBM. Links to other criteria in this table or Table 2 are in *italics*

EBM criteria	Generic assessment guidance and considerations
Ecological integrity and biodiversity	This can be achieved by defining and conserving a diversity of species traits or functional groups that support the integrity of the ecosystem, or check the three aspects: variety; balance; and disparity. Are these explicitly considered in the knowledge base? The ecological structural components determine the functioning of the ecological system. Hence the link to the '*Consider ecosystem connections*' criterion: a knowledge base that covers more relevant components or detail is better.
Consider ecosystem connections	This is determined by the ecological part of the SES (e.g., by mapping critical connections) and is linked to the '*Ecological integrity and biodiversity*' criterion as more components and/or detail increase this aspect (e.g., in terms of taxa considered in the food web) this can be improved with an indication of the importance of a connection (e.g., pressure-state relationships, predator-prey relationships). Knowledge on the ecological functioning of the ecological structural components determines the provisioning of ecosystem services which contribute to human well-being and as such can be incorporated into (economic) markets.
Account for dynamic nature of ecosystems	Variation in the ecological part of the SES (e.g., due to perturbations) should be considered. Longer time-series are better. Question the assumption of perfect foresight. Include exogenous scenarios of socio-economic drivers or environmental change (e.g., climate scenarios). Strengthen feedbacks that maintain desired regimes, break or disturb feedbacks that maintain undesired regimes; look for non-linearity in the system as these are often the cause for the dynamic nature.
Acknowledge uncertainty	This requires transparency on the quality of the knowledge base which could be reflected, for example, through the assessment of uncertainties, reporting of crucial (model) assumptions and confidence intervals in the output. Uncertainty is inherent to complex adaptive systems (such as the marine) and their management. In addition to the known unknowns (e.g., lack of historic data of species), complex systems come with new uncertainties that cannot be tackled through standard sensitivity analysis (Polasky et al. 2011). The social EBM criteria '*Adaptive management*' and '*stakeholder participation*' (see Table 2) become increasingly important if uncertainty in the current knowledge base is high.
Appropriate spatial and temporal scales	What are the appropriate spatial and temporal scales of the (eco) system? For example, resolution of spatial grid and temporal units (e.g., years, months). Which scales to consider? Not just spatial/temporal but also in different domains (e.g., ecological, jurisdictional, administrative or political). Use a systems framework to address relevant scales and how they interact. Assessment should occur at the ecosystem scale. If other scales are relevant and do not match with the ecosystem scale this needs to be identified.

(continued)

Table 1 (continued)

EBM criteria	Generic assessment guidance and considerations
Distinct boundaries	Acknowledge boundaries and thus the fluxes and influences from outside of the boundaries of the ecosystem. Consider both jurisdictional boundaries as well as ecosystem boundaries (see *Appropriate Spatial and Temporal Scales*). Are transboundary issues considered? For example: Terrestrial run-off into rivers and lakes or inflow of rivers into the coastal/marine ecosystem. The definition of boundaries should allow the adaptation of institutions in a good social-ecological fit (see *recognise coupled SES*).
Recognise coupled SES	Are all relevant flows considered between the social and the ecological system that make up the SES? How many linkages, or how much of the activities, pressures, ecosystem components and the ecosystem services they provide is covered in the subSES (used in the management strategy evaluation) compared to the full SES. Is it understood how these link to the actors that drive the relevant social processes (see Table 2)?
Consider cumulative impacts	Apply an integrated perspective, including all relevant activities and their pressures acting on the ecosystem (see *recognise coupled SES*). Consider whether synergistic or antagonistic cumulative effects apply (Crain et al. 2008).

2.3 Phase III: Planning of EBM

This phase should result in a comprehensive EBM plan that can mitigate the threats and achieve the policy objectives identified in Phase I. In building a comprehensive EBM plan, the Phase II knowledge base is used to first guide the design of the EBM plan, and then evaluate the plan's potential performance before implementation in Phase IV.

Most examples of management plans have focused on ecological outcomes (e.g., Rademeyer et al. 2007; Ansong et al. 2017; Samhouri et al. 2014). Since EBM is concerned with the management of SES, the EBM plan that is developed and tested in this phase should cover both the ecological and social components of the system. To that end, the EBM plan consists of two interconnected, structured, yet differentiated sets of decisions, management measures, and policy instruments, each primarily addressing a specific aspect of the SES (see Fig. 3):

- Management measures are integrated into a Programme of Measures, a combined set of actions aimed at achieving environmental objectives and thus to enhance and protect the ecological system. Potential management measures can be classified according to the three ISO 31,000 risk management categories: prevention; mitigation; and recovery controls (Cormier et al. 2013, 2018) that can be aligned to the EBM management measure typology outlined in Piet et al. (2015), depending on where in the linkage framework the measure intervenes.
    - Prevention controls manage the causes of the risk and are aimed at the human activity and/or the pressure. Examples are input control (e.g., scrapping schemes to reduce the capacity of the fishing fleet), output controls that

Table 2 Social EBM criteria for assessing the knowledge base of the social part of the social-ecological system (SES) and hence the core sustainability SES subsystems: governance systems and users (Ostrom 2009). These criteria are based on the key EBM principles identified by Long et al. (2015) and link to the relevant governance actors. Guidance is provided to assess to what extent the institutional set-up and its governance processes can support EBM. Links to other criteria in this table or Table 1 are in *italics*

EBM criteria	Actors	Generic assessment guidance and considerations
Use of scientific knowledge	Science	This includes the scientific use of all types of knowledge including local knowledge, traditional knowledge or citizen science. Has the knowledge been produced according to the scientific standards? Is the methodology appropriate? Are procedures transparent? Peer-reviewed? Is there consensus on the quality of the available (scientific) knowledge? This requires both the interaction between scientists and decision-makers to foster salience in scientific input as well as the interaction between scientists and other actors to foster credibility in knowledge production (see Röckmann et al. 2015). Ultimately the outcome of the process should be perceived as evidence-based.
Inter-disciplinarity		Was the appropriate expertise in terms of relevant disciplines applied when producing the knowledge? Can stakeholder knowledge be integrated? The aim is to progress from multi- to inter- to transdisciplinary science. See *Stakeholder Involvement*.
Stakeholder involvement		Science could benefit from knowledge available with other stakeholders, notably the business sector. Stakeholders can play a role in providing knowledge (see '*Use of Scientific Knowledge*') collecting data (monitoring; cooperative research). The feedback of stakeholders on making choices under uncertainty is also important. Reed (2008) identifies eight best practices that improve the quality and effectiveness of stakeholder participation.
Integrated management	Management	In this context integrated can be interpreted as cross-sectoral, inter-disciplinary and/or holistic (i.e., encompassing the whole SES). Which of these (or other) perspectives are incorporated into the management process? Compliance of the SES aspect *Human activities and their pressures* is a requirement. The Decision-making across administrative boundaries is tightly linked to the *Distinct boundaries* criterion where jurisdictional boundaries may be different from ecosystem boundaries. Building resilience requires a governance capable of balancing heterogeneity, redundancy, modularity and connectivity at *Appropriate temporal and spatial scales* (Elmhirst et al. 2009; Levin et al. 2013)
Adaptive management		The management should be adaptive as it needs to deal with the inherent uncertainty of EBM. Learning-by-doing is needed when outcomes of decisions are uncertain because of complex system dynamics. This is linked to ecological EBM criteria: *Acknowledge uncertainty*, *Account for dynamic nature of ecosystems* and

(continued)

Table 2 (continued)

EBM criteria	Actors	Generic assessment guidance and considerations
		Appropriate monitoring. Rather than choosing optimal paths and decision rules in a deterministic framework, facing current risks and considerable uncertainties requires governance frameworks able to adapt to the multiple circumstances that may prevail in the foreseeable future.
Apply the precautionary approach		Does the institutional set-up allow the application of the precautionary approach? This requires compliance to the SES aspect *Changes and Uncertainty*.
Stakeholder involvement		Managers depend on the input from science but could benefit from knowledge available with other stakeholders, notably the business sector. Also, the feedback of stakeholders on making choices (or co-decision making) under uncertainty is important. As compliance of the SES aspect is a requirement; stakeholder involvement in policy implementation can be instrumental.
Appropriate monitoring		A requirement of *Adaptive management* is adequate monitoring. The quality of the monitoring is reflected in the proportion of the relevant components of the SES for which sufficient data is collected at *appropriate spatio-temporal scale* and the level of *uncertainty* to allow *scientific knowledge to guide informed decision-making*. Monitoring programs can be developed in collaboration with the other stakeholders, i.e. multi-sector actor resulting in cooperative research. The monitoring data should be transformed into salient and legitimate scientific knowledge to guide informed decision-making. The degree to which that actually occurs needs to increase in order to advance EBM.
Decisions reflect societal choice	Policy making	Specifying clear goals increases efficiency and efficacy of the MSP process and allows the identification of potential trade-offs of proposed management strategies. Specify what trade-offs should be considered, e.g. amongst stakeholders, or between short- and longer-term benefits. '*Stakeholder involvement*' is required specifically in order to make decision-making inclusive and reflect societal choice (i.e., legitimacy).
Stakeholder involvement		For rationale see Röckmann et al. (2015). Check if the 'typology of eight levels of participation' (Reed 2008) is applied. The degree of stakeholder interaction should be appropriate for the specific context. Also, the feedback of stakeholders on making choices (or co-decision making) under uncertainty is important. Is there information on compliance? Stakeholder involvement in decision making can be instrumental to ascertain compliance.
Sustainability		All three pillars of sustainability (i.e., ecological, economic and social) should be considered in the trade-offs informing the decision-making process.
Stakeholder involvement	All resource users	The participation and involvement of all the resource users is the backbone of a successful EBM process. This may be

(continued)

Table 2 (continued)

EBM criteria	Actors	Generic assessment guidance and considerations
		through a top-down process initiated by the government or a bottom-up process where the users self-organize. In case of a top-down process stakeholder participation should reflect and be based on all sectors which are affected by the plan, local community actors and environmental non-governmental organizations (NGOs) in addition to the stakeholder groups specifically mentioned (i.e., science, policy makers and managers). This to ascertain all relevant societal claims, values and relevant aspects and impacts can be considered in the process and involved at each stage and that implementation and monitoring of strategies are effectively done. The likelihood of self-organization of the users to achieve a sustainable SES was found to depend on several aspects of the SES covered by the other EBM criteria, see (Ostrom 2009).

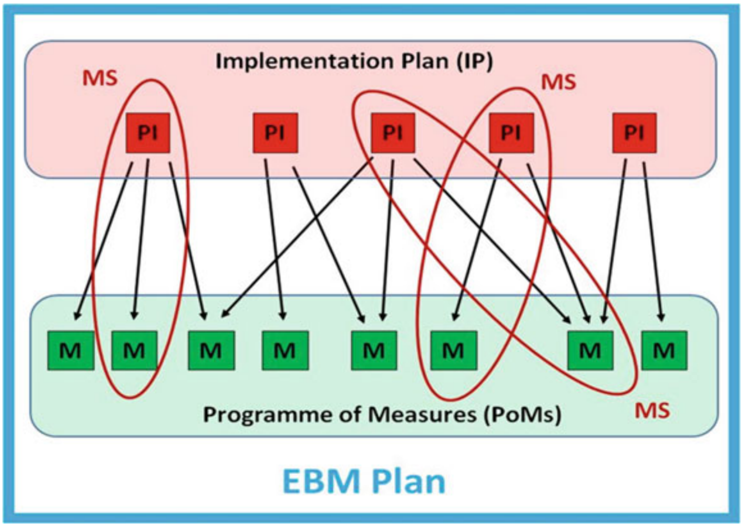

Fig. 3 Diagram explaining the elements that make up an ecosystem-based management plan. PI = Policy Instruments, M = Management Measure, MS = Management Strategy

prevent the pressure from entering the system (e.g., catch controls in fisheries) or spatial- and/or temporal distribution controls (e.g., marine protected areas or real-time closures).

- Mitigation and recovery controls are implemented to reduce the likelihood and magnitude of each consequence as a result of the risk event occurring. We distinguish mitigation controls that target the pressure once it is present in the system (e.g., beach cleaning after oil spills) and recovery controls targeting the

ecosystem component (state) that is impacted (e.g., habitat restoration or stocking programs). For both mitigation controls there may be a lag in the response depending on the persistence of the pressure or the resilience of the species. For recovery controls, the lag is only determined by the resilience. The choice of the type of management measure thus determines the time horizon when results can be expected.
- Policy instruments are integrated into an Implementation Plan. This consists of all the arrangements or reforms that are required in the governing system (as part of the social system) for the implementation of the Programme of Measures and the overall performance of the full EBM plan. The following types of policy instruments can be distinguished (see Lago et al. 2015; Frelih-Larsen et al. 2016):

- Legislative instruments, including various (inter)national conservation laws or regulations
- Regulatory instruments, including the setting of targets or standards aimed at maintaining a certain level of environmental quality, prohibits (i.e., bans) or allows (i.e., permits) an individual or business to perform certain acts, or to have a certain portion (or amount) of a product
- Economic instruments such as pricing mechanisms (e.g., tariffs, taxes and charges, trading of permits), payments, or liability schemes
- Instruments involving information, awareness-raising, and public engagement such as training and qualifications (e.g., obtaining certificates or proof of qualification) related to environmental protection, public information programs, stakeholder and public participation, or innovation groups that aim to build capacity and knowledge about a particular environmental, economic, or practical issue
- 'Nudges' are possible alternatives to the instruments mentioned above, whereby the behaviour of actors (e.g., industry, civilians) is influenced (or nudged) towards the preferred choice via positive reinforcement and indirect suggestions such as changing the default option in a form (Valatin et al. 2016)
- Monitoring and research aimed at improving the SES knowledge base. This may include the collecting of additional data or developing an understanding on specific gaps that may correspond to specific nodes or linkages in the linkage framework that hampered this EBM cycle but may be useful in the next EBM cycle.

The design step of the EBM plan commences with the selection of appropriate candidate management measures. This is guided by the ERA, which identifies the linkages contributing most to environmental impact risk indicating the potential management measures following the approach developed by Piet et al. (2015). Those candidate management measures should cover all the major threats and are considered most likely to reduce the environmental impact risk. A pre-screening exercise using the '10-tenets of adaptive management' (Barnard and Elliott 2015) may be conducted to ascertain *a priori* that all possible issues are considered that may prevent the success of management measures or policy instruments. Failure to comply to any one of the tenets should be reason to re-consider a particular management measures or policy instrument. By applying these tenets as screening criteria for the list of potential issues, those issues that do not tie directly to a tenet, should be removed from further testing in the evaluation step.

In this evaluation step, the future performance of an alternative EBM plan is compared to that of the existing (e.g. baseline, business-as-usual) management plan. Both the alternative and baseline management plans may result in different outcomes depending on exogenous drivers (e.g., socio-economic or climate scenarios). For the evaluation of the alternative EBM plan(s) against baseline, we propose three outcome-oriented criteria:

- *Effectiveness: Does the plan achieve the pre-determined target?* This is what usually constitutes effective evaluation of a management strategy (see Punt et al. 2016 for best practices) and involves the simulation of specific indicator trajectories with their error distributions relative to policy targets (e.g. fishing mortality indicator relative to the target of Maximum Sustainable Yield). Effectiveness of an individual measure, or of a programme of measures, along with its implementation plan, is defined by the contribution it makes to bridge the information gap between understanding baseline conditions and target conditions.
- *Efficiency: Is the plan conducive to enhance human wellbeing?* This refers to the capacity of citizens and social institutions to take advantage of existing opportunities (as determined by technology, resource endowments and actual availability, physical and human capital, etc.) to improve human wellbeing in a sustainable way. It is a concept that applies to the users of a particular service (i.e., those who may have the opportunity to utilise the service without making anyone else worse off), the stakeholders in a particular decision context (i.e., who may have the option to cooperate in the preservation of a resource and share the benefits amongst them), or the government (i.e., who may have the possibility of improving the environment without worsening opportunities in terms of economic activities). This criterion is ultimately an assessment of sustainable development, where each generation should aim at improving its wellbeing within the available opportunities as long as this does not compromise the options available to future generations. The benefits and costs are defined as any positive or negative impacts on human wellbeing, irrespective of whether the affected individuals are aware of them, or whether they can be valued through market prices or any other ad-hoc valuation exercise. When comparing benefits and costs, the issue is that costs are often monetised and are relatively certain, whereas benefits may be difficult to monetise and are definitely more uncertain (though mostly just as real).
- *Equity: Are the benefits being shared in a socially just way?* The distribution of benefits and costs across stakeholders must be perceived as fair. Besides the contribution of the EBM plan, to social equity, the legitimacy of the EBM plan requires the perception that its consequences are fairly distributed among the affected parties both in the present as well as the future (i.e., intergenerational equity).

2.4 Phase IV: Implementation, Monitoring and Evaluation

This is the phase where EBM becomes operational based on the planning in the previous phases. According to Cormier et al. (2017), it is the competent authorities of specific sectors that are accountable to implement the measures that are designed to manage their specific operations. The role of science is to: (1) inform those sector-specific authorities on the detail of the measures before implementation; as well as (2) design the monitoring programs; and (3) conduct and inform the subsequent evaluation of the performance of those measures after implementation. The performance is evaluated testing indicators against a benchmark as a measure of achieving an objective. When the benchmark is not met, the goals and objectives needs to be re-examined and/or the management regime re-assessed (Behn 2003; Poister et al. 2010) in phases I and III of the next EBM cycle.

While a monitoring program is primarily intended to assess the status of the SES and the performance of the EBM plan, it can also feed relevant information into the knowledge base and hence need to be aligned with system-oriented criteria of Table 1. The implementation of any future alternative EBM plan is determined by the governance context and the institutional processes captured in the Phase II knowledge base with the process-oriented criteria in Table 2. The performance of the EBM plan can then be used to guide the planning (Phase III) in the next EBM cycle, which builds on the previous EBM plan by adopting those management measures or policy instruments that performed well, and modifying or replacing those that failed. Incorporating the feedbacks from Phase IV and integrating it into the next EBM cycle is crucial to a successful EBM process.

3 Discussion and Conclusions

The EBM approach presented here is an attempt to combine or reconcile many existing concepts that describe the ecological system, the social (or socio-economic) system and EBM into a unifying approach with guidance on how to make it (more) operational in any aquatic ecosystem based on a diverse existing literature incorporating concepts from IEA, Marine Spatial Planning, ERA, resilience and CAS (Long et al. 2015; Cormier et al. 2017; Levin et al. 2009b; Samhouri et al. 2014; Levin et al. 2009b; Samhouri et al. 2014; Ansong et al. 2017; Folke et al. 2010; Hagstrom and Levin 2017). We have attempted to translate these concepts into concrete steps, identify issues for the practitioner to consider, give examples that provide the basis for a common framework, and suggest practical guidance for the incorporation and synthesis of interdisciplinary information on SES into practical and useful linkage frameworks for EBM plan development and implementation. One example of a practical application of this framework can be found in (Piet et al. 2019).

Integrated ERAs have been included in environmental impact assessments for many decades but it was not until the work of Halpern et al. (2008) that the cumulative effects of multiple stressors received much attention. Since then, many other such integrated assessments have taken place in marine waters (e.g., Coll et al. 2012; Knights et al. 2013; Korpinen et al. 2012) and are now also covering inland and transitional waters (e.g., Borgwardt et al. 2019). These integrated assessments all apply different methodologies which may differ on their ability to inform EBM and associated monitoring requirements (Borja et al. 2016). Even within a specific integrated assessment there are methodological choices to be made depending on the chosen subSES and/or the application of the ERA (Piet et al. 2017).

The EBM process consisting of four phases (I–IV) has been built around the SES concept that brings together the natural and social scientific disciplines involved in EBM. The contribution of this study to advance EBM occurs primarily in Phase II and Phase III. Advancements in Phase II consists of: (1) approaches to reduce a complex suite of SES to one or more focused subSES to avoid inaction from overwhelming complexity, a common problem when resolving wicked problems such as EBM; and (2) practical guidance based on key EBM principles (Long et al. 2015) involving both the ecological system as well as the social system. Advancements in Phase III consists of the organizing the structure and typology of an EBM plan that is explicitly linked to both the ecological system as well as the social system. Even though the findings of this study are primarily relevant for the science domain, it explicitly acknowledges the interaction with wider society. This approach structurally incorporates consultation with other stakeholder groups in order to enhance the credibility in knowledge production and ascertain salient scientific input in the domain of policy-makers, decision-makers, and managers (Röckmann et al. 2015).

Despite the many issues that are still unresolved, this study provides the theoretical and conceptual basis to apply some of the methodological studies in this volume (Borgwardt et al. 2019; Teixeira et al. 2019) in order to advance the implementation of EBM toward and achievement of policy objectives in support of the societal goals for our aquatic systems (see Piet et al. 2019).

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Ecosystem-Based Management: Moving from Concept to Practice

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Abstract Ecosystem-Based Management (EBM), intended to restore, enhance and/or protect the resilience of ecosystems, is gaining momentum. It is often argued, though, that some of the difficulties to provide practical guidance to conduct EBM stems from the lack of a clear definition. EBM emphasises on factoring in complex linkages in social-ecological systems; dealing with adequate scales (both time and space wise); promoting adaptive management of complex and dynamic systems; and adopting integrated assessment and management frameworks. This chapter shows, on one side, challenges to build consensus on a definition that is both conceptually and theoretically sound as well as practicable; on the other, the enabling factors that make EBM actually happen.

Lessons Learned

- Ecosystem-Based Management (EBM) moves away from a limited (partial) consideration of natural systems and society as separate entities.
- Society will face increasing levels of uncertainty—EBM is a more flexible approach that allows for the identification of some of those uncertainties, so

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that they can be considered in decision-making, hence allowing society to adapt to error.

- EBM is a recognition of the need to "juggling all balls" at once. Since reality is complex, responses will also have to be so.
- EBM is an integrated approach in many different ways: acknowledging for the recognition of linkages between society and ecosystems, between different types of ecosystems, throughout time and space, looking at things from multiple standpoints.
- Enhanced governance is critical to make EBM happen. This is not just about transparency, accountability and meaningful social participation. It is also about using the right incentives, cooperating, coordinating decisions between sectors, improving our knowledge and information base, uptaking innovations, etc.

Needs to Advance EBM

- We need to better understand the complexity of the social, behavioural side of social-ecological systems, to match our understanding of the ecological side.

1 Defining Ecosystem-Based Management: Minding Mice at a Crossroads or Not Quite?

Ecosystem-Based Management (EBM) has gained increased popularity in recent years; yet, it has been argued that lack of consensus on its definition is precluding progress in terms of design and practical implementation of such approaches. EBM emphasises on considering ecosystem connections; dealing with appropriate spatial and temporal scales; fostering adaptive management of complex social-ecological systems; managing those systems in an integrated way; accounting for the dynamic nature of ecosystems and society, etc. There is no single agreed-upon definition for EBM, which sometimes is even referred to as the ecosystem approach (EA) itself. What seems common to all definitions, though, is the acknowledgement of the complexity and interspecies relationships within ecological systems, although broader governance elements are also of great importance in most definitions. This chapter does not develop a linguistic, nominal investigation of different definitions of EBM but rather aims at adding conceptual clarity while, at the same time, focusing on enabling factors for their effective uptake in decision- and policy-making for biodiversity and aquatic ecosystem services conservation.

We often demand clear-cut definitions: obvious, without any need of proof, unambiguous, unequivocal. However, demanding rigidly defined areas of doubt and uncertainty is a challenging exercise. Grasping at straws to some extent. Social-ecological systems such as aquatic ecosystems are highly adaptive and complex. How could one expect their management to be defined in an indisputable way if characterised by adaptability and complexity (Preiser et al. 2018)? Nonetheless, this is far from being a futile exercise since it also provides useful insights for the practical implementation of these management approaches (see Robinson and Culhane 2020 for further details on complexity, EBM and a framework to assess linkages).

1.1 *The Many-Sided Definition of Ecosystem-Based Management*

According to the outcomes of the AQUACROSS research project,[1] Ecosystem-Based Management (EBM) can be understood (Gómez et al. 2016) as

> any management or policy option intended to restore, enhance and/or protect the resilience of the ecosystem.

As such, EBM would stand for any course of action intended to improve the ability of an ecosystem to remain within critical thresholds, to respond to change, and/or to transform, so that a new equilibrium or a new evolutionary path follow (ibid.).

When applied to aquatic ecosystems, EBM can be said to set the foundations for effective and widely applicable management concepts and practices. Yet, one may think that EBM actually lacks a definition (and a universal 'grammar'), which may hinder implementation (Long et al. 2015, 2017; Willaert et al. 2019); that EBM requires extensive data and sophisticated modelling (Addison et al. 2019); that EBM is linked to naïve attempts to describe complex and adaptive systems, squeezing the universe between our fingers; and, on more practical grounds, that there are neither enough resources to deliver EBM approaches (Curtice et al. 2012) nor a clear mandate and institutional setup for EBM in prevailing legislation (Nilsson and Bohman 2015; Link et al. 2019). Harwell (2020) presents a non-exhaustive overview of the broad suite of U.S. federal environmental laws and regulations linked to ecosystem services and a survey of the legal scholar literature on the topic in the U.S. environmental law. The combined outcome of all these premises, whether utterly true or false, is definitely limiting the ability of EBM to deliver, let alone in aquatic ecosystems as shown in Langhans et al. (2018). As a matter of fact, EBM is often defined using combinations of underlying principles (Long et al. 2017). EBM aims at ensuring that choices do not negatively affect ecosystem functions, so that the delivery of aquatic ecosystem services and benefits stemming from them can be

[1]AQUACROSS was a Horizon 2020 project aimed to support EU efforts to protect aquatic biodiversity and ensure the provision of aquatic ecosystem services. As such, AQUACROSS sought to advance knowledge and application of ecosystem-based management (EBM) for aquatic ecosystems to support the timely achievement of the EU 2020 Biodiversity Strategy targets. AQUACROSS has received funding from the European Union's Horizon 2020 Programme for Research, Technological Development and Demonstration under Grant Agreement no. 642317.

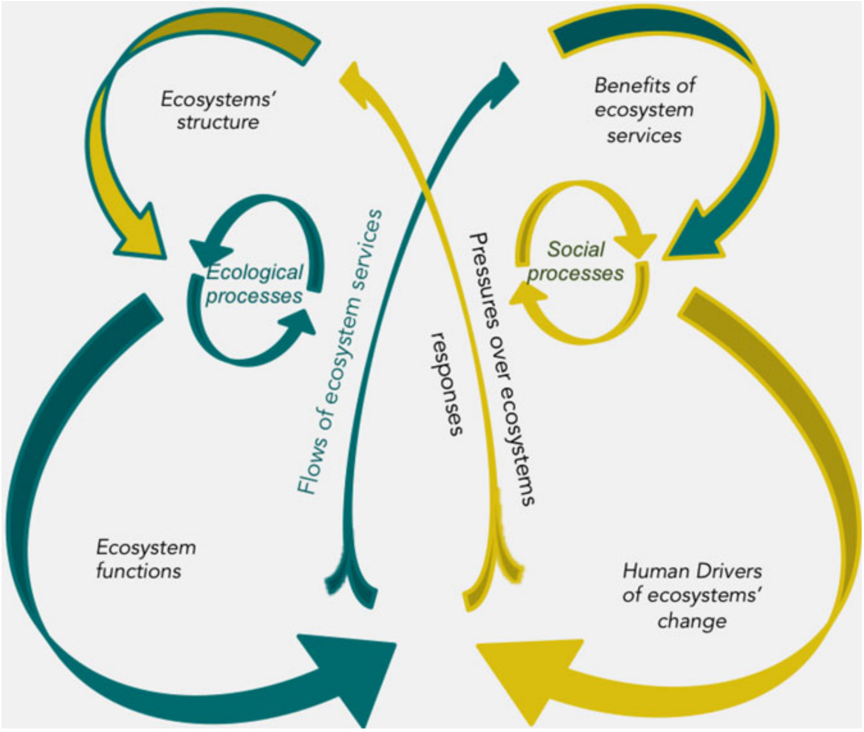

Fig. 1 Social-ecological systems as interlinked, complex, adaptive systems. Source: Gómez et al. (2016)

sustained in the long term. This makes EBM highly relevant to maintain and restore connections between social and ecological systems (Keesstra et al. 2018).

Social-ecological systems (SES) are therefore complex and adaptive systems that should be analysed in a holistic, integrated way (see Ostrom 2009). On the one hand, through the notion of SES, one can analyse the detrimental consequences over ecosystems that result from the satisfaction of multiple demands of services provided by nature and to society. As in Fig. 1 (see below), this could be seen as the demand side of the "butterfly" used to represent a SES. It shows how the demand and use of naturally provided services is an outcome of social processes, including markets and governing institutions, and determined by multiple factors (such as population and economic growth, climate change, technological progress, etc.). These demands of services result in pressures over ecosystems and further changes in their structure. On the other hand, one can analyse the potential of ecosystems to continue delivering ecosystem services on which human life, the social system and the ecological system itself depend upon, and how this all affects human wellbeing. This is the supply side, which if analysed can shed light on the functioning of ecosystems and how changes, induced by human actions, are linked to human wellbeing and sustainability.

EBM is also a way to address uncertainty and variability in ecological systems (Link et al. 2012) that, by definition are dynamic (DeFries and Nagendra 2017), in an attempt to embrace change, to learn from the past and to design and implement adaptive policies throughout the management process (Schultz et al. 2015). In other words, in the absence of sound governance frameworks and inclusive models (covering a wide range of social actors), EBM seems deemed flawed, as in the case of relevant sectoral policies such as the EU Common Agricultural Policy (Pe'er et al. 2019) or the Common Fishery Policy (Lado 2016). On the contrary, if properly designed and implemented, EBM rather shows the flexibility to identify uncertainties and incorporate those into decisions (through exploration of alternatives), and to adapt to error or uncertainty through an iterative review of progress toward the goal for which an EBM-based plan was developed.

By explicitly factoring in the full range of ecological and social interactions and processes necessary to sustain ecosystem structure, functions and services, EBM seems to offer great expectations for aquatic ecosystems management (Tallis et al. 2010; Langhans et al. 2018, 2019). One of the potential advantages of EBM is that, in principle, it should go well beyond conventional scientific and policy practice.

All dimensions of this multi-layered definition of EBM foster integration across ecological and social systems (Collins et al. 2011), explicitly address the need for sustainable patterns of resource use (Grehan et al. 2009), acknowledge interspecies relationships (Long et al. 2015), and call for the consideration of ecological thresholds and other environmental limits (Samhouri et al. 2017; Möllmann et al. 2015). Basing management decisions on the ecosystem entails that planning needs to be adapted to the dynamics of the whole ecosystem to at least preserve, if not to enhance, their potential to delivering services and benefits society depends upon.

Albeit difficult, describing is always simpler than analysing; defining may also be easier than understanding. Stefan Zweig, in *Chess Story* (Zweig 2011) said:

> We are happy when people/things conform and unhappy when they don't. People and events don't disappoint us, our models of reality do. It is my model of reality that determines my happiness or disappointments.

It is critical to note that all definitions of EBM have some common features but mainly differ in their view of the connections between ecosystems and society, which has clear implications in terms of modelling (see Fulford et al. 2020 for a detailed discussion on mathematical modelling for EBM and ecosystem services).

1.2 Disambiguation of the Concept

There may be good reasons to believe that EBM does not add too much value if compared with the so-called ecosystem approach (EA), such as introduced by the Secretariat of the Convention on Biological Diversity (CBD 2008) (see Enright and Boteler 2020 for a deep review of the implicit origins of the approach and its subsequent adoption in marine International Law). As a matter of fact, a number of authors still use the latter expression rather than the former (Bennett et al. 2015; McIntyre 2019). According to CBD (ibid.)

> the ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. Thus, the application of the ecosystem approach will help to reach a balance of the three objectives of the Convention: conservation; sustainable use; and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources.

The EA can be considered as a management approach that takes the ecosystem itself as the relevant scale (time and space wise, but also on organisational grounds) but this can only be inferred from some of its 12 principles. For instance, principle 5 states:

> conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach.

Principle 6 in turn adds:

> ecosystems must be managed within the limits of their functioning.

Finally, principle 7 stresses upon the fact that

> the ecosystem approach should be undertaken at the appropriate spatial and temporal scales.

Beyond linguistic flourishes, the EA could certainly be considered an ecosystem approach to management, hence quite close to the notion of EBM, although other authors add some nuances (Kirkfeldt 2019). Do fine distinctions between these terms essentially matter? Not quite, within this context... One may actually argue that the EA does not preclude any other conservation approaches applied to aquatic ecosystems; hence, it may be compatible and consistent with such approaches, such as EBM (if considered a beast of a different coat), integrated water resources management (IWRM), integrated coastal zone management (ICZM), etc. In fact, these approaches may support the implementation of the EA in different biomes (Gómez et al. 2016).

As above, EBM emphasises on considering ecosystem connections (something that is actually coherent with the EA), dealing with adequate spatial and temporal scales, promoting adaptive management, deepening integration, accounting for the dynamic nature of ecosystems, etc. As in Sect. 1.1, there is no single agreed-upon, canonical definition for EBM, which sometimes is even referred to as the EA itself (Kirkfeldt, op. cit.).

Unlike the Common Classification of Ecosystem Services (CICES) (Haines-Young and Potschin 2018), for example, which was created ad hoc to homogenise a typology, not more than an accounting system, just as many other typologies, indicator sets, mapping exercises, databases, etc., this book is meant to shed light on available conceptual and management approaches, rather than to create a new 'taxonomy' of EBM definitions.

Further to the notions of EA and EBM, not more than a decade ago practitioners such as the International Union for the Conservation of Nature (Cohen-Shacham et al. 2016) and the World Bank (MacKinnon et al. 2008; Browder et al. 2019; Erin Gray et al. 2019) did coin the term Nature Based Solutions (NBS) (Davies and Lafortezza 2019), which has pervaded global discussions on water resources management since the release of the World Water Development Report 2018 (UNESCO 2018).

Some of those NBS, for instance, aim at restoring and conserving aquatic ecosystems via natural means. Those NBS can take the form of green infrastructures intended to maintain and enhance landscape, soil cover, and groundwater sources in order to improve their natural traits, the ecological services and other abiotic components they deliver, and to favour climate change adaptation and reduced vulnerability to extreme events and water quality degradation. The distinctive character of NBS would then have to do with their single purpose (i.e. conserving aquatic ecosystems), but also with a specific set of means (i.e. natural ones).

This example clearly underlines some distinctive features of NBS. On the one hand, not every solution that is effective on environmental grounds (e.g., increasing water recharge rates in groundwater bodies) is a NBS. On the other hand, NBS are interventions over ecosystems: they use natural processes rather than replacing nature, hence emulating ecological functions. It is also clear that, as in the case of the example above, the intended environmental outcome (i.e. recharging an aquifer) is not the end but the means that makes these NBS relevant for EBM of those aquatic ecosystems. Last but not least, it should also be mentioned that NBS are not simply means to reset a spoiled ecosystem to its pristine state, which is often not a feasible endeavour (if possible at all), but rather to adapt prevailing developments so as to enhance or to recover ecosystem functions that were either fragile or have been lost altogether.

As in EC (2015), NBS are

> living solutions inspired by, continuously supported by and using nature, which are designed to address various societal challenges in a resource-efficient and adaptable manner and to provide simultaneously economic, social, and environmental benefits.

Maes and Jacobs (2017), in turn, add further conceptual clarity when defining NBS as

> any transition to a use of ecosystem services with decreased input of non-renewable natural capital and increased investment in renewable natural processes.

Nesshöver et al. (2017) stress, though, that NBS need to be developed and discussed in relation to existing concepts to clarify their added value.

Along these lines, Integrated Water Resources Management (IWRM) can arguably be presented as an empirical concept more than a theoretical one (Smith and Clausen 2015). It was also built up from the actual experience of practitioners. The concept has now long been used (since the first global water conference, the United Nations –UN– Water Conference, in Mar del Plata, Argentina, on March 14th, 1977), even if practice is still limited to some areas of the world such as the European Union or Australia, to name a few. However, it was not until the UN Conference on Environment and Development (Rio, Brazil, 1992) that the concept was refined and became the object of wide discussions as to its practical implications. GWP (2000) states:

> IWRM is a process which promotes the co-ordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems.

IWRM could therefore be seen as a wider notion, which could benefit from the design and implementation of EBM approaches.

Quite the same applies to Integrated Coastal Zone Management (ICZM). EC (2012), concerning the implementation of ICZM echoed the previous recognition of EC (2000) that coastal management lacked vision and was based on a very limited understanding of processes and dynamics in coastal ecosystems, as a result of the gap between scientific research and data collection, and end-users (see also Lewis et al. 2020 for a detailed review on the complexity and disparity of results from applying the existing coastal and estuarine ecosystem goods and services models and tools).

To cover the aquatic ecosystems continuum, Marine Spatial Planning (MSP, see Pauli 2010), has often been mentioned as an ecosystem-based approach. Through analysing and allocating parts of marine ecosystems to specific uses in order to meet ecological, economic, and social objectives, MSP integrates across economic sectors and among agencies, delivers adaptive approaches, and is focused on longer time frames. Le Tissier (2020), provides an insightful review on the relationship and contradictions regarding the juxtaposition of EBM and ICM and MSP.

IWRM and ICZM, let alone MSP, are all processes that entail a new fashion of governance, one that goes well beyond conventional and restrictive views based on integrity, stakeholder engagement, transparency and accountability or mere institutional reforms, and progresses towards mastering complexity, thus considering the redesign of incentives, horizontal and vertical coordination of sectoral policies, promoting innovation at different levels, etc. (such as in the OECD Water Governance Programme, for instance: OECD 2011).

Further to these conceptual approaches, a number of policy initiatives have sparked the development of a wide range of assessment frameworks and decision support systems for EBM (Harvey et al. 2017; Link and Browman 2017; Alexander and Haward 2019; Lago et al. 2019). Yet, a major challenge remains as to the design of an operational framework that links, in an efficient way, the assessment of biodiversity and ecological processes and their meaningful consideration in decision-making processes (and not just in policy making), despite recent progress (Langhans et al. 2018; Gómez et al. 2017; Do Yun et al. 2017). EBM faces both conceptual and functional limitations, in particular with regards to the lack of explicit acquaintance of the ecosystem services concept (Jordan et al. 2012), critical to connect ecological analyses with social wellbeing; a certain bias towards ecological dimensions rather than a balanced approach embedding social-ecological processes (Berkes 2012), which would enhance a more holistic understanding of relevant dynamics and feedback loops. Trade-offs, uncertainties, and non-linearities inherent in the management of aquatic ecosystems are somehow downplayed, when not overlooked at all (Curtin and Prellezo 2010; Culhane et al. 2020).

2 Distinctive Features of Ecosystem-Based Management of Aquatic Ecosystems

Despite a certain amount of uncertainty and lack of conceptual clarity, there has been some evolution in the policy use of EBM (Gelcich et al. 2018). There is an actual policy need for a consolidated, practical definition of the term that addresses its different dimensions: understanding complex and adaptive social-ecological systems, resilience thinking (Curtin and Parker 2014), accounting for ecosystem functions and services, favouring multi-functional responses, etc.

EBM of aquatic ecosystems should therefore be assessed, designed and implemented so as to solve specific problems, and these solutions are often likely to maximise contributions to the resilience of social-ecological systems as a whole. Understanding the enabling conditions (see Sect. 3.3 of this chapter) that foster cooperative behaviour, or the factors that hinder it, is of chief importance for the design and implementation of EBM approaches. Progress in evolutionary theory (Di Marco et al. 2019; Wasser 2013) and game theory (Arfanuzzaman and Syed 2018; Punt et al. 2014) have been considered as particularly auspicious.

Identifying biodiversity policy challenges and assessing different alternatives to address them is far from being just a purely scientific or political endeavour. This daunting task requires both a conducive policy process and harnessing scientific knowledge to face stakeholders throughout with the outcomes of their own choices and to support them across the whole policy cycle. There is no such thing as good science or good policy: the former should address societal challenges; the latter, in turn, should be based on evidence.

Due to the complexities involved in aquatic social-ecological systems, it is clear there is neither a design of a one-size-fits-all EBM approach nor just one EBM implementation path. Additionally, it is critical to understand that more science may not necessarily close the existing knowledge gaps. Rather, each individual situation may need to be considered in its institutional and political setting and requires consideration of site-specific trade-offs.

Under an EBM approach science is neither only intended to inform nor to make technically feasible and sound decisions. It is rather a means to build a credible knowledge base through dialogue between stakeholders and scientists. Sharpe et al. (2020) provide more information on the benefits of Structure Decision Making–SDM and Decision Support Tools–DST in engaging with stakeholders and supporting decision makers; Lewis et al. (2020) also shed light on a community-based decision support framework and tool for quantifying trade-offs in ecosystem goods and services.

It is often argued that EBM approaches are characterised by their contribution to ecological integrity, biodiversity, resilience and (in some versions, as in Granek et al. 2010 for coastal ecosystems) to ecosystem services delivery; by their use of scientific knowledge and appropriate spatial scales (Qiu et al. 2018), their acknowledgement of social-ecological connections, stakeholder engagement and accountability (Nunan et al. 2018); by transdisciplinarity and integrated management (Pires et al.

2018; Carmen et al. 2018) and by their adaptiveness (Long et al. 2015). However, unlike common wisdom, EBM does not exclusively show those features and several approaches that are not based on the ecosystem may well do so.

What could be said to be specific of EBM for aquatic ecosystems?

- *Holistic*. EBM gives consideration to ecological and social factors, which demands interdisciplinary knowledge and prominence to water governance and the relationship among aquatic species as well as with their abiotic environment. EBM protects the integrity of aquatic ecosystems as a means to preserve a complementary array of ecosystem services as well as to preserve aquatic biodiversity in its own right. EBM thus acknowledges social-ecological interactions (necessarily including terrestrial ecosystems too) and seeks inclusive policy-making processes that favour transparency, accountability and provide a better framework so as to make different stakeholders understand why they make the decisions they make.
- *Multi-functional*. Unlike more conventional approaches to biodiversity and ecosystem conservation that focus on single benefits, EBM is characterised by multiple functions and benefits, thus being able to strike the balance, at once, between different policy domains. To put it in a different way, EBM aims at maximising the joint value of all aquatic ecosystem services and abiotic components, rather than focusing only on the delivery of single services, such as drinking water, water for irrigation, natural assimilation capacity of pollution, etc.).
- *Based on evidence*. Just like other management approaches, EBM is based on scientific knowledge. This is but solemnising what is obvious, though. What makes EBM stand out is the kind of scientific knowledge that is mobilised, as well as the way in which this is factored into decision-making processes. The role of science in EBM is two-fold: on one side, it is intended to be relevant for policy —and decision-making—; on the other, it is essential for the credibility of social knowledge and for the legitimacy of policy decisions it intends to inform and improve. Needless to say, in addition to scientific knowledge, traditional knowledge on ecosystem management (Joa et al. 2018) also provides critical insights.
- *Mindful of the spatial scale (and ecological organisational levels)*. Management based on the aquatic ecosystem is much more challenging and sophisticated than managing single water bodies (or even watersheds). This might imply devolution to local communities, but may also require action at higher levels through, for example, transboundary or even global cooperation. EBM decisions are to take place at the appropriate level and scale, taking into account aquatic ecosystem boundaries but also complex interlinkages and adaptive processes. Actually, EBM keeps a close relationship with the notion of meta-ecosystem (Loreau et al. 2003). This

(continued)

concept provides a powerful theoretical tool to ascertain the emergent properties that arise from spatial coupling of local ecosystems, such as global source-sink dynamics, diversity-productivity patterns, stabilisation of ecosystem processes, and indirect interactions at landscape or regional scales.

- *Across the water continuum*. Not only complex interactions within the social-ecological system are considered, but also the relationships among aquatic realms (i.e. freshwater ecosystems, coastal ecosystems and transitional waters, marine ecosystems), including unknown and unpredictable effects.
- *Calling for policy coordination*. By definition, EBM calls for (horizontal — across sectoral policies— and vertical —in multi-level governance systems, across different levels of government within the same sectoral policy —) policy coordination. As EBM requires cooperative agreements and collective action to share the range of aquatic ecosystem services obtained across different stakeholders and policy domains, and by seeking to balance ecological and social concerns, as above, these approaches open new opportunities of concurrently pursuing different policy objectives.
- *Adaptive and dynamic*. EBM is a way of adaptive management. Aquatic ecosystem processes and functions are complex and variable. Through acknowledging that there are no optimal solutions and that the future is uncertain, EBM seeks to build or strengthen adaptation capacities by restoring critical aquatic ecosystems and supporting social abilities to respond to a range of possible future scenarios. Low-hanging fruits of management interventions in the short term should be weighed against longer-term benefits of alternative actions. Uncertain events may alter long-term goals or show new, alternate lines of action. As a result of that, not only those aims but also management approaches to reach them should be regularly revisited, hence making monitoring critical so that indications of potential opportunities or difficulties are spotted sufficiently in advance.

3 Enabling Factors for the Effective Uptake of Ecosystem-Based Management

Beyond contributions to conceptual clarity, what seems more important is what makes EBM feasible, i.e. the enabling conditions for this kind of management approaches to happen in practice. Other chapters in this book (quote) will address some of those practical dimensions more specifically. In this chapter we discuss these factors from a conceptual point of view.

3.1 Moving Away from Conventional Praxis in Ecosystem Management

It is very often the case that management toward a single goal often underestimates consequences or dependencies on other parts of ecosystems. For instance, traditional approaches to biodiversity conservation tend to focus on single pressures and specific impacts, flagship umbrella species (Kalinkat et al. 2017), hotspots (Marchese 2015), lack of integration of traditional and scientific knowledge (Sutherland et al. 2014). This is not to say that there is not utility in species-centred conservation approaches. Surrogate species remain popular as they provide useful and necessary shortcuts for some conservation programmes (Caro 2010). As a matter of fact, it is not only a question of the object of conservation programmes but rather about management approaches themselves. Those programmes achieve measurable outcomes (Herzon et al. 2018); however, positive results in the conservation of emblematic species may come at the expense of degrading resilience and increasing vulnerability.

From a management and policy perspective, conventional approaches are based on sectoral (and frequently conflicting) policies. On the contrary, EBM offers a promise of making the multiple co-benefits linked to the enhancement of the ecosystem overall visible. EBM opens new possibilities to meet different policy objectives simultaneously and quite often through cooperative approaches and policy synergies. The so-called nexus perspective is a good example of this (see Venghaus and Hake 2018, for an overview of nexus thinking in EU policies, or Zhang et al. 2018, for a specific application to hydropower generation).

Traditional ecosystem management approaches also tend to maximise the delivery of some ecosystem services (with a bias towards productive ones, such as drinking water, water for irrigation. . .) whilst impairing the ecosystem's capacity to deliver other services, including those linked to self-regulation and support. Traditional management has overreached in altering ecosystems for a single purpose. Emerging EBM, in contrast, finds opportunities in benefits stemming from the restoration of natural features. See, for instance, Sklar et al. (2019), where coastal risk reduction (i.e. storm protection through wave attenuation) and resilience (i.e. flood storage compensation schemes) come in the form of protection, enhancement and restoration of natural features such as mangroves, coral reefs, seagrass beds, sand dunes, inter-tidal and sub-tidal wetlands, mudflats, floodplains, salt marshes, etc.

An additional feature of conventional approaches is the neglect (to a very different extent) of the inherent uncertainties of social-ecological systems and the adoption of mostly deterministic approaches to future challenges when modelling the consequences of future scenarios. EBM necessarily separates from optimality (see Heal et al. 2001 for a seminal discussion on challenges posed by the transition from optimality to sustainability) and, through acknowledging irreducible uncertainties, emphasises on the relevance of building adaptability. The idea of building adaptation capacities, by the way, is far from being just one of restoring critical ecosystems (or ecosystem functions and processes), but also of strengthening social abilities (i.e. investing in social capital) to respond in a robust way to a range of possible futures.

In summary, EBM underlines the trade-offs between focusing on one single species, pressure, impact, sector, or ecosystem service, and adopting a more complex approach. In other words, EBM offers the possibility to discern between planning for the long-term or get caught in a series of lock-ins: technological, analytical, and institutional (Lukasiewicz et al. 2016), which clearly hinder adaptability.

3.2 *Adaptive Governance of Aquatic Ecosystems*

Schultz et al. (2015) defined governance as "the structures and processes by which people in societies make decisions and share power, creating the conditions for ordered rule and collective actions, or institutions of social coordination" (p. 7369). It is precisely the recognition of the above-mentioned lock-ins that supports the idea that ecosystems have to be governed. This implies, among other things, overcoming (impact) remedial, reactive, mostly unplanned, ad hoc approaches to ecosystem management in favour of pre-emptive, proactive, planned, collective, and coordinated decisions. To put it in a different way, EBM provides an opportunity for adaptive governance leading to higher social-ecological resilience (as Boyd et al. 2015 suggested).

There is a binding constraint for collective action, though: EBM is a leap in the dark or, at best, a chimera in the absence of a strong and enabling institutional setup (Börgström et al. 2015). As DeFries and Nagendra (2017) put it, ecosystem management is very much a "wicked problem", with no straightforward solution. Focusing on ecosystems rather than on single species or functions or processes requires defining specific spatial, temporal and organisational scales of ecosystem services provision. This, of course, has implications in turn for the appropriate scale and structure of ecosystem management institutions.

"Good governance" has gained momentum in the discussion about EBM (Long et al. 2015; Soma et al. 2015; Schultz et al. 2015; Gunderson et al. 2016; Bodin 2017; Bundy et al. 2017; Link et al. 2019). As Grindle (2010) pointed out,

> good governance is a good idea. We would all be better off, and citizens of many developing countries would be much better off, if public life were conducted within institutions that were fair, judicious, transparent, accountable, participatory, responsive, well-managed, and efficient [. . .] Who, after all can reasonably defend bad governance? [. . .].

Good governance, though, carries a similar virus to EBM: the appeal of the idea somehow outpaces its capacity to deliver. It is sure-fire that EBM demands transparency, inclusiveness, a good knowledge base, the appropriate spatial scale, policy coordination, etc. Yet, aren't these defining characteristics of good public policy overall?

An effective uptake of EBM requires adapting prevailing institutions and policy-making processes and jumping the fence to address challenges of a political nature.

Firstly, it is important to stress upon the idea that EBM is a means to an end. This entails that the objectives of EBM need to be defined and this requires the identification of what set of ecosystem services may be sustainably delivered as well as their relative importance. Beyond what Irvine et al. (2016) call shared values, it is a fact that there is a number of asymmetries (information, preferences, etc.) that explain why different individuals value ecosystem services in a different way (see Harwell et al. 2020 to deepen on the role of Strategic Communication in EBM for overcoming information asymmetries). Key to achieving mutually beneficial situations for biodiversity and ecosystem services is identifying those differences and trade-offs between users (Cavender-Bares et al. 2015, and also DeWittt et al. 2020 for a discussion on beneficiary-centric orientation supporting EBM). Indeed, what is special about EBM is that it gives prevalence to this collective choice, which makes everyone accountable for her or his own decisions.

Secondly, balancing trade-offs between ecosystem services and users requires finding the best way to meet pre-defined environmental objectives. This can be done through a number of alternatives. Choices between alternative courses of action are a collective endeavour, which also entails trade-offs: between short-term opportunity costs and long-term benefits, mitigated pressures and weaker provision of productive services or enhanced long-term water security, tackling critical events or managing risks and resilience, etc.

Foremost, seizing opportunities stemming from EBM demands a meaningful change of our mind-set. Whilst traditional measures are designed to respond to a particular challenge, EBM is linked, as above, to multiple ancillary benefits and may simultaneously contribute to various policy objectives, well beyond their intended outcome. The essence of EBM is precisely to be able to reap off the benefits of synergies across different ecological and policy realms. Nonetheless, current assessment methodologies such as optimisation models or cost-effectiveness analysis tend to be limited to account for multiple benefits, hence biasing decisions against EBM solutions, but also for uncertainty (Borgström et al. 2015), consideration of local trade-offs and disservices (Traoré et al. 2018), integration of ecosystem services in land and landscape planning and management (Turkelboom et al. 2018; Grêt-Regamey et al. 2017; Arkema et al. 2015), non-linearities (Grêt-Regamey et al. 2014), and non-convexities (Hyytiäinen et al. 2015).

4 Conclusion

Most of us do not internalize that a social-ecological system is a complex entity, dynamic and adaptive both on ecological and social grounds. Virtually all readers would be willing to say they are perfectly aware of that assertion; some could even explain what it connotes beyond its primary or more literal meaning. Yet, most of us would overlook that evidence when making choices affecting ecosystems. Similar to our brain which has one side (left) that helps us think logically and organize thoughts to build sentences, and another one (right) that helps us experience emotions and interpret nonverbal cues, our way of approaching the management of social-ecological systems responds in many ways to that dualism. As Kahneman (2011) would put it: we think fast and slow.

On the one hand, we feel driven to make decisions related to survival (i.e. withdrawing water from ecosystems for irrigation agriculture or drinking water supply). On the other, another part of us leads us to connection, complexity, and relationships. The key to progress in aquatic biodiversity and ecosystem conservation is to contribute, within the context of EBM, to both dimensions operating altogether. For the ecosystem to be genuinely healthy, all (specialized) elements of the social-ecological system must be integrated. That is, each element of the system has to perform its individual function while operating as part of the whole. Integration is neither more (but not less) than that: to bond different elements, already combined de facto (although sometimes they may have been disconnected, the reason why some interventions are aimed at restoring connectivity), so that the whole system works properly. Integration, inherent in EBM approaches, coordinates and balances those individual parts.

In recent decades, scientists from different disciplines have managed to develop assessment frameworks and even technologies that allow us to better understand the causal relationships between biological diversity and ecosystem services, between different types of ecosystem services, or between ecological processes and functions and social well-being (precisely through the notion of ecosystem services). However, none of them seems to guarantee that trends in biodiversity loss in aquatic ecosystems are reversed or that they are managed in a more integrated way. We have more information about aquatic ecosystems, somehow we know them better; yet, this does not always ensure that we understand how these complex and adaptive social-ecological systems do work.

Following the analogy regarding our brain, as above, one of the most relevant findings in scientific literature is the plasticity and ductile nature of our brain. The same applies to social-ecological systems to some extent: that is, they are systems that always change and not only at certain stages of their evolution. What modifies those systems? In essence, a series of experiences does. Depending on decisions and pressures on aquatic ecosystems, different parts of the ecosystem are activated and the way they do so to respond (as is also the case in social systems) determines the health of these socio-ecological systems and their level of biodiversity. Different connections then become more sophisticated and the system is reconfigured. There is some good news in that evidence: no social-ecological system, provided certain thresholds are not irreversibly exceeded, is a slave to each of these decisions and responses (both human and ecological). Thus, EBM approaches can intervene so that ecosystems become healthier and we result happier. The genetic diversity of aquatic ecosystems at stake plays an important role in how to respond to multiple pressures. However, as decisive or more than that genetic component is everything that happens in these social-ecological systems. In other words, beyond the basic 'architecture' of ecosystems, it is our choices that truly determine whether social-ecological systems evolve in a resilient and integrated way or not.

The integration inherent to the EBM concept precisely consists in facilitating the above-mentioned reconfigurations of social-ecological systems, in favouring the connections between the different parts of them. When these links are cooperative, the elements that integrate are created and strengthened.

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From DPSIR the DAPSI(W) R(M) Emerges… a Butterfly – '*protecting the natural stuff and delivering the human stuff*'

Michael Elliott and Timothy G. O'Higgins

Abstract The complexity of interactions and feedbacks between human activities and ecosystems can make the analysis of such social-ecological systems intractable. In order to provide a common means to understand and analyse the links between social and ecological process within these systems, a range of analytical frameworks have been developed and adopted. Following decades of practical experience in implementation, the Driver Pressure State Impact Response (DPSIR) conceptual framework has been adapted and re-developed to become the D(A)PSI(W)R(M). This paper describes in detail the D(A)PSI(W)R(M) and its development from the original DPSIR conceptual frame. Despite its diverse application and demonstrated utility, a number of inherent shortcomings are identified. In particular the DPSIR model family tend to be best suited to individual environmental pressures and human activities and their resulting environmental problems, having a limited focus on the supply and demand of benefits from nature. We present a derived framework, the "Butterfly", a more holistic approach designed to expand the concept. The "Butterfly" model, moves away from the centralised accounting framework approach while more-fully incorporating the complexity of social and ecological systems, and the supply and demand of ecosystem services, which are central to human-environment interactions.

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1 Introduction

There is only one big idea in environmental management, especially that for aquatic ecosystem, that is '*how to maintain and protect the natural ecological structure and functioning and the resultant ecosystem services while delivering the societal goods and benefits*' (Elliott 2011). This is exemplified in Ecosystem-Based Management (EBM) which seeks to "integrate the connections between land air water and all living things including human beings and their institutions" (Mee et al. 2015) and to manage complex adaptive systems towards particular goals and targets. In essence, this is a risk assessment and risk management approach which requires monitoring to determine whether management actions have worked (Cormier et al. 2019). Hence in order to assess whether specific policy decisions are effective management efforts need to agree measurable targets on pre-defined indicators. Yet the complexity of interactions and feedbacks between human activities and ecosystems can make the analysis of such systems intractable and so we need an underlying accepted framework to link together the causes and consequences of change and their management. Systematic methods of accounting for socio-ecological interactions are required to assess the effectiveness of management in such complex adaptive systems. Here we follow environmental accounting as the process of systematically organising and presenting information relating to interactions between the economy and the environment in a standardised way in order to support policy making (UNSD 2012).

While there are many legal instruments which are required to manage the environment, and so fulfil the 'big idea' above (Boyes and Elliott 2014), in Europe there are two major framework environmental policies relating to aquatic environments: the Water Framework Directive (WFD) (European Commission 2000) and the Marine Strategy Framework Directive (MSFD) which both promote an ecosystem approach to freshwater, estuarine, coastal and marine management (see O'Hagan 2020, and Borja et al. 2010 and references therein for detailed accounts). These policies seek to apply common standards, equally, to the conservation of aquatic environments across the European Union Member States (European Commission 2008). This results in a particular need for standardised methodologies because each Member State has its own unique geographic and environmental conditions, as well as traditions of monitoring the environment based on indicators. For example, Teixeira et al. (2016) generated a catalogue of 611 indicators of biodiversity proposed or in use for monitoring European marine environments. Under these conditions, accounting frameworks enable the intercomparison of different Member State efforts and activities. This should enable assessment on a continental scale of whether environmental policies are reaching their objectives and whether EU member states are complying with these and other Directives. Experiences in the implementation of these directives have been instrumental in developing the state-of the art science addressing socio-ecological systems in Europe and has led to many valuable insights into Ecosystem-Based Management.

One environmental accounting framework, which has developed in parallel with the application of European aquatic environmental law is the Driver Pressure State

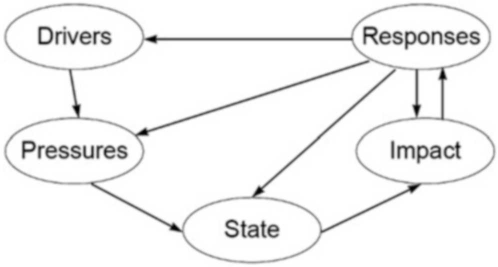

Fig. 1 Original depiction of the DPSIR framework. Source: European Environment Agency (1999)

Impact Response (DPSIR) framework (EEA 1999) which developed from the earlier PSI framework adopted by the OECD in the early 1990s. In brief, human activities or *Drivers* place *Pressures* on the environment resulting in changes to the environmental *State* and *Impacts* which result in a management *Response*. This causal chain in Fig. 1 uses the original diagram of the European Environment Agency (EEA 1999). DPSIR is a conceptual framework for analysing socio-ecological systems which encapsulates the interactions between anthropogenic and natural components of socio-ecological systems (Turner and Schaafsma 2015) where each element in the cause-consequence-management chain can be described by an indicator (see below). As a conceptual framework, the DPSIR is used to compartmentalise and thereby simplify and analyse socio-ecological systems.

With origins in the field of environmental risk assessment and accounting (Rapport and Friend 1979), the DPSIR was first formally published in 1999 (EEA 1999) prior to adopting the WFD (European Commission 2000) and has widely been used and adapted over the intervening 20 years (Patrício et al. 2016). The Web of Science indicates DPSIR has been the subject of 577 peer-reviewed papers, principally in the fields of environmental science ($n = 406$), water resources (88) and ecology (66). As an overarching conceptual frame, the DPSIR has been employed to analyse a broad variety of environmental problems in diverse environments and geographic settings from the study of water scarcity in Oman (Al Kalbani et al. 2015) to biodiversity loss (Maxim et al. 2009). Yet the DPSIR has been most commonly used in Europe (Patrício et al. 2016) and the most highly cited/influential DPSIR papers relate to the European aquatic environmental conservation directives, the WFD (Borja et al. 2006) and the MSFD (Atkins et al. 2011).

Early iterations of DPSIR (EEA 1999), did not rigorously define the various information categories, rather it described their interrelationships:

> . . .social and economic developments [Driving forces] exert Pressure on the environment and, as a consequence, the State of the environment changes, such as the provision of adequate conditions for health, resources availability and biodiversity. Finally, this leads to Impacts on human health, ecosystems and materials that may elicit a societal Response that feeds back on the Driving forces, or on the state or impacts directly, through adaptation or curative action. (EEA 1999)

As a result, through practical application, the DPSIR has been continually refined and redefined in the context of European Marine environments. DPSIR has been applied to the Mediterranean and Baltic Seas (Lundberg 2005; Karageorgis et al.

2005; Skoulikidis 2009). Studies in the Black Sea (Langmead et al. 2009; Knudsen et al. 2010) employed a modified DPSIR (mDPSIR) approach, while, Cooper (2013) made a variety of changes to develop DPSWR (where W is Welfare) which has been used in a variety of geographic contexts from the Black Sea (O'Higgins et al. 2014 to the North East Atlantic (O'Higgins and Gilbert 2014) as well as to explore socio-ecological scale mismatches in marine sectors (O'Higgins et al. 2019) and to explore intertemporal trade-offs in activity and environmental quality (O'Higgins et al. 2014). This evolution of DPSIR is fully detailed by Patrício et al. 2016) which indicated anomalies in the way the concept has been used; this culminated in the development of DAPSI(W)R(M) which incorporated 20 years of the evolution of the concept and attempted to resolve the confusion in the use of the forerunners (as well as being the more pronounceable "*dap- see–worm*") (described in detail in Elliott et al. 2017, but used in their previous papers, e.g. for the Baltic Sea by Scharin et al. 2016 and the Arctic by Lovecraft and Meek 2019).

In this volume, the DPSIR and its variants are critical to many of the case studies. For example, these frameworks form the basis of the linkage framework techniques (Culhane et al. 2020); and were used as an organisational framework for case studies in the Azores (McDonald et al. 2020), the Danube (Funk et al. 2020) and in Lough Erne (O'Higgins et al. 2020). In this paper we first set out the major features of the DAPSI(W)R(M) framework (Elliott et al. 2017). We then consider the broader characteristics of ecosystem-based management (Langahns et al. 2019; Delacamara et al. 2020) and introduce a derivation of the framework which we call "the butterfly". The latter is designed to expand the concept, moving away from the centralised accounting framework approach while more-fully incorporating the complexity of social and ecological systems. We present the butterfly not as a replacement to the DPSIR-family (which has been so successfully applied to the existing marine policy) as a socio-ecological accounting framework, but as a potential tool to enable more fully integrated approaches to the development and application of Ecosystem-Based Management.

2 The DAPSI(W)R(M) Framework

2.1 Drivers (D)

Previous DPSIR frameworks (e.g. Cooper 2013) had an in-built confusion as the Drivers were often synonymous with activities (what is done in the environment) or sectors (groups of activities such as fisheries) (Patrício et al. 2016). There may also be different interpretations of this between natural and social scientists—for example the former may regard Drivers as activities but social scientists regard them as 'basic human needs'. In order to resolve the confusion, and because the later elements in the framework refer to and separate Activities and Pressures, Elliott et al. (2017) emphasised that the Drivers should refer to Basic Human Needs and thus the work of Maslow (1943) who proposed a five-tier hierarchical structure of such Drivers

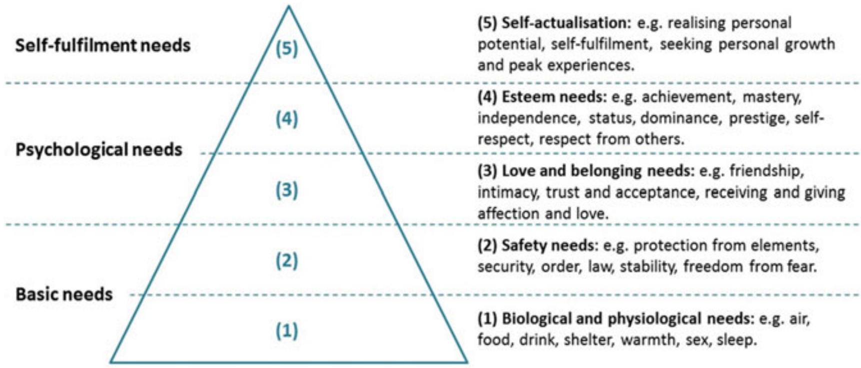

Fig. 2 Maslow's hierarchy of needs and human welfare (adapted from Maslow 1943)

(Fig. 2). At the lowest level these refer to an individual's biological and physiological needs which relate to survival (e.g. oxygen, food, drinking water) and safety and security (e.g. protection from external stressors). At its most simple, we can determine what are our basic human needs and then how do we get those from the natural and human environment.

The intermediate levels of basic human needs then cover psychological and emotional attributes which determine the societal structure and family relationships including love and belonging (e.g. friendship, intimacy, trust and acceptance) and esteem (e.g. prestige, achievement, self-respect). Hence, the lower four levels in Maslow's triangle cover the 'deficiency needs' i.e. what are required to motivate and satisfy people and for which desires and activities increase to satisfy these needs. The fifth and uppermost levels relate to meeting one's emotional fulfilment, such as the personal potential, emotional growth and satisfied experience—these may be regarded as 'growth needs'. Hence, there is the need for both an individual and society to satisfy the lower 'deficit needs' before achieving the higher 'growth needs' (Maslow 1943, 1970a, b). This includes three further intermediate categories of 'cognitive needs' (e.g. knowledge and understanding, curiosity, exploration, need for meaning and predictability—the scientific requirements), 'aesthetic needs' (e.g. appreciation and search for beauty, balance, form, etc. influencing our cultural appreciation) and 'transcendence needs' (e.g. altruistic behaviour helping others to achieve self-actualisation) (Maslow 1970a, b). Many of these needs are directly dependent on ecosystems, the 'basic needs' and safety needs require provisioning ecosystem services, while others for psychological and self-fulfilment needs depend on supply of ecosystem services and societal goods and benefits (Turner and Schaafsma 2015).

All of these basic human needs in turn dictate the way the marine space is used, the competition between individuals, tribes, societies and nations in the way in which the competition now occurs at larger scales from the regional to the global. Satisfying these needs and this competition (the basis of the economy) then leads to Activities inherent, for example, in international marine trade which is needed to deliver increasingly required/desired consumer products and services. All of this shows that the Drivers can ensure both economic benefits as well as high individual (physiological and psychological) and societal well-being and welfare. However, this unchecked growth in Drivers can lead to levels of Activities and Pressures which exceed the carrying and assimilative capacities of a particular system or even of the global system (Rockstrom et al. 2009) resulting in depletion of resources and damage to the natural system. This exceedance then requires measures which aim to minimise conflicts and partition that marine space in order to deliver the human and protect the natural aspects (Elliott et al. 2018).

The aggregate of these human wants and needs (or the urge to maximize utility) are the forces that drive economic development—the Blue Economy or Blue Growth. As set out above, they involve a range of social interactions between humans (e.g. respect from others, intimacy), as well as socio-ecological interactions between humans and the environment (e.g. food, shelter and warmth).

2.2 *Activities (A)*

As indicated above, there is the need to separate Drivers from Activities and Activities from Pressures. In essence, Activities are what we do in the seas to obtain those basic human basic needs, and the Pressures are the resulting mechanisms of change from those Activities (Elliott et al. 2017). Marine activities can be separated into 15 key marine sectors which then represent many individual, but often interlinked, activities which occur in most if not all seas (Table 1 gives 7 of these sectors). Elliott et al. (2017) lists all the Activities and Pressures and Burdon et al.

Table 1 Examples of sectors and activities in a marine environment (adapted from Elliott et al. 2017)

Sector	Examples of activities
Extraction of living resources	Benthic trawling (e.g. scallop dredging), netting, pelagic trawls, potting/creeling; bait digging, seaweed and harvesting, shellfish gathering
Transport and shipping	Mooring/beaching/launching, shipping, ferry operation, waste disposal.
Navigational dredging	Dredging, removal of substratum, disposal of dredge spoil.
Coastal infrastructure	Construction of artificial reefs and barrages, beach nourishment, laying of communication cables; constriction of transport infrastructure, dock and port facilities, land claim; construction of urban dwellings.
Land-based industry	Treatment and discharge of industrial; discharging particulate waste disposal, desalination effluent, sewage and thermal waste discharge
Agriculture	Animal waste disposal, crop fertilisation, farming, coastal forestry,
Tourism/recreation	Angling, boating/yachting, diving, public beach use, tourist resort and water sports operation, whale watching.
Research and education	Marine research; field education and training, research vessel cruises.

(2018) shows the complexity by giving a detailed breakdown of these relating to one main sector—oil and gas decommissioning. Hence the separation of sectors with nested activities may simplify the scheme. For example, the commercial fishing sector includes many types of fishing activity (trawling, potting, long-lines, etc.) and each produces Pressures which may or may not differ across Activities. This therefore creates operational marine management which is sufficiently specific for targeted problem-solving directed at specific activities (Cormier et al. 2019). Such a framework has to include historical marine activities (e.g. fisheries, oil and gas extraction), and newer offshore technologies (e.g. marine biotechnology, nodule mining) which together reflect the expanding global marine economy with its increasing human pressures (Stojanovic and Farmer 2013). Together this constitutes what may be termed the Blue Economy defined as 'smart, sustainable and inclusive economic and employment growth from oceans, seas and coasts' (e.g. marine energy extraction, aquaculture, maritime, coastal and cruise tourism, marine mineral resources, blue biotechnology) (European Commission 2012).

As indicated above, as the original DPSIR framework confused Drivers and Pressures, there was the need to separate them within the DAPSI(W)R(M) framework by adding Activities. As an example, the basic Driver is to obtain food and beam trawling is a fishery Activity which then leads to the Pressure of seabed abrasion caused by towed gear. It is then assumed that such abrasion would cause seabed habitat damage (e.g. a State change in the functional traits of the benthic community) which eventually leads to Impacts (on human Welfare) by reducing the fishable resource (see below). In turn, a Response (using management Measures), such as gear modifications or fishing-period limitation is needed to limit the Activities and hence minimise the Pressures.

The confusion between terms was noted in the otherwise important and seminal study by Halpern et al. (2008) giving the global analysis of 'human impact' on many marine ecosystems. This study listed 17 anthropogenic 'drivers' but, using the definitions here, these comprised seven Activities (including various forms of fishing, shipping and commercial activity) and 10 Pressures (including organic and inorganic pollutants, benthic structures, invasive species, sea temperature and ocean acidification) with none of the categories relating to Drivers as defined above. The global analysis of Halpern et al. (2008) appears to map Activities (and possibly Pressures) but terms them 'impacts'—it is emphasised here that this should not be assumed as mitigation measures may stop Activities creating impacts. This also reinforces the point that it may be easier to map Activities, which are often recorded on maps, photographs and databases, than Pressures. The latter requires the need to detect the spatial and temporal effects-footprints of each Activity which are much more difficult to determine (Elliott et al. 2018). Many of the assessment schemes used worldwide focus on Activities instead of Pressures (Borja et al. 2016), presumably because of the difficulty in determining these effects-footprints. Here the linkage Framework methodologies (Culhane et al. 2020) can be valuable in understanding and untangling activities and their resultant pressures.

2.3 Pressures (P)

Each Activity leads to one or more Pressures and each Pressure can result from one or more Activity thereby creating an interlinked matrix of Activities and Pressures (see Culhane et al. 2020 for approaches to analysing such linkages, and Burdon et al. 2018 for a precise example). A Pressure is defined as the mechanism of change, first to the natural system (State changes) and then to the social system (Impacts (on human Welfare)) (see below). Elliott (2011) then separated the pressures affecting a given sea area into Exogenic Unmanaged Pressures (ExUP) and Endogenic Managed Pressures (EnMP) (Fig. 3). In this way, exerting pressures on the ecosystem will reduce ecosystem services and ultimately affect societal goods and benefits.

The cause of Exogenic Unmanaged Pressures, as the name suggests, emanates outside the area, for example sea-level rise as the result of global climate change, and so management inside the area is only treating the consequences (such as building higher sea defences) (Elliott et al. 2016). Management actions to address these pressures therefore has to be at the large scale and even global level (such as the Paris COP meetings). In contrast, Endogenic Managed Pressures, by definition, occur within the management area and so both their causes and consequences need managing and can be managed. For example, increased infrastructure such as a new bridge or power plant in an area will cause pressures whereby the reduction of their consequences need to be incorporated into a management plan. Hence the

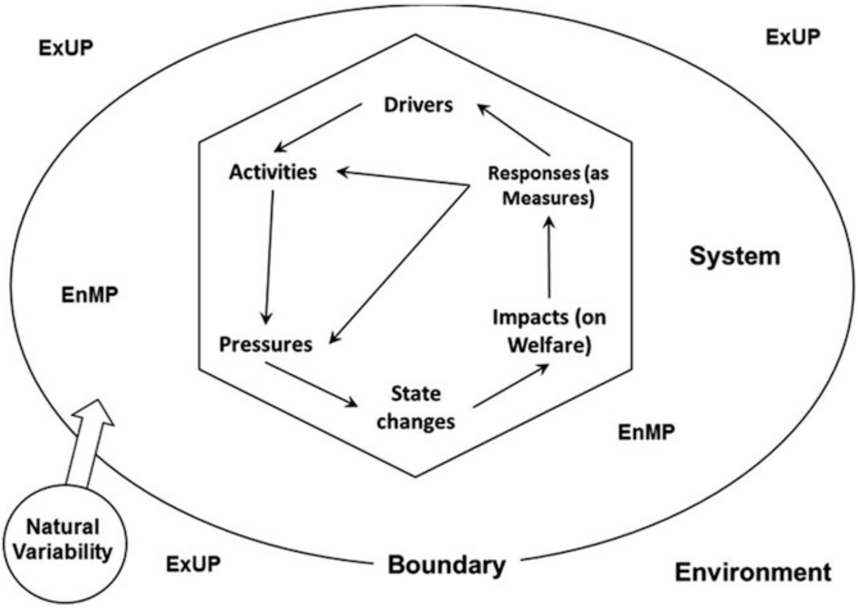

Fig. 3 The DAPSI(W)R(M) problem structuring framework (from Elliott et al. 2017). Key: ExUP = Exogenic Unmanaged Pressures; EnMP = Endogenic Managed Pressures (see text for explanation)

effects-footprints of all the endogenic pressures, both singly and cumulatively, need to be determined and managed in both space and time (Elliott et al. 2018).

2.4 State or State Changes (S)

In the original DPSIR framework, State, State change and Impact were often used interchangeably (Patrício et al. 2016) and often the natural scientists used State as the nature of the natural system and Impact as the change to the natural and social systems. In contrast, social scientists appear to have made the differentiation that State referred to the natural system and Impact referred to the social system. Because of this, the DAPSI(W)R(M) framework has used the term 'State change' (rather than 'State' as we are only interested in anthropogenic changes, i.e. a signal against a background of inherent variability ('noise')) to relate to the natural system (the ecology) due to single or multiple Pressures. This includes both the physico-chemical variables (i.e. sediment type, dissolved oxygen, organic matter, etc.) and biological health at all levels of organisation—the cellular system, individuals, populations, communities and ecosystems. Such changes can be referred to as structural characteristics (the features in each level at one time, for example the number of species in a community) or functioning variables (rate processes, such as productivity or ecosystem carbon flow) (Strong et al. 2015; de Jonge et al. 2003, 2012).

This then leads into the recent discussions regarding ecosystem services and societal goods and benefits which are derived from a healthy functioning natural system (Atkins et al. 2011; Turner and Schaafsma 2015). If the natural system has an appropriate structure and functioning then it is creating a healthy environment, for example if the waves, tides, substratum, etc. are appropriate then they will support the prey which sustains fishes (an ecosystem service). In turn, the natural State should provide the intermediate and final ecosystem services (as defined by Fisher et al. 2009, Turner and Schaafsma 2015, and Elliott et al. 2017) and human activities and pressures could then influence (in a positive or negative way) this natural state (i.e. State change) as well as the underlying marine ecosystem components and processes (Fig. 4, left hand side). The amount and fluxes of physical, chemical and biological materials may be regarded, in economic terms, as marine ecosystem stocks and flows which can be measured and which can have management targets and management measures to achieve those targets (Pinto et al. 2014; Atkins et al. 2015). In essence, the natural system can produce the ecosystem services but it then requires society to input 'human complementary assets' or 'human capital' (such as time, money, energy and skills) to extract societal goods and benefits ('*the sea can produce fish but we need to learn how to catch and cook them*'!) (Elliott et al. 2017).

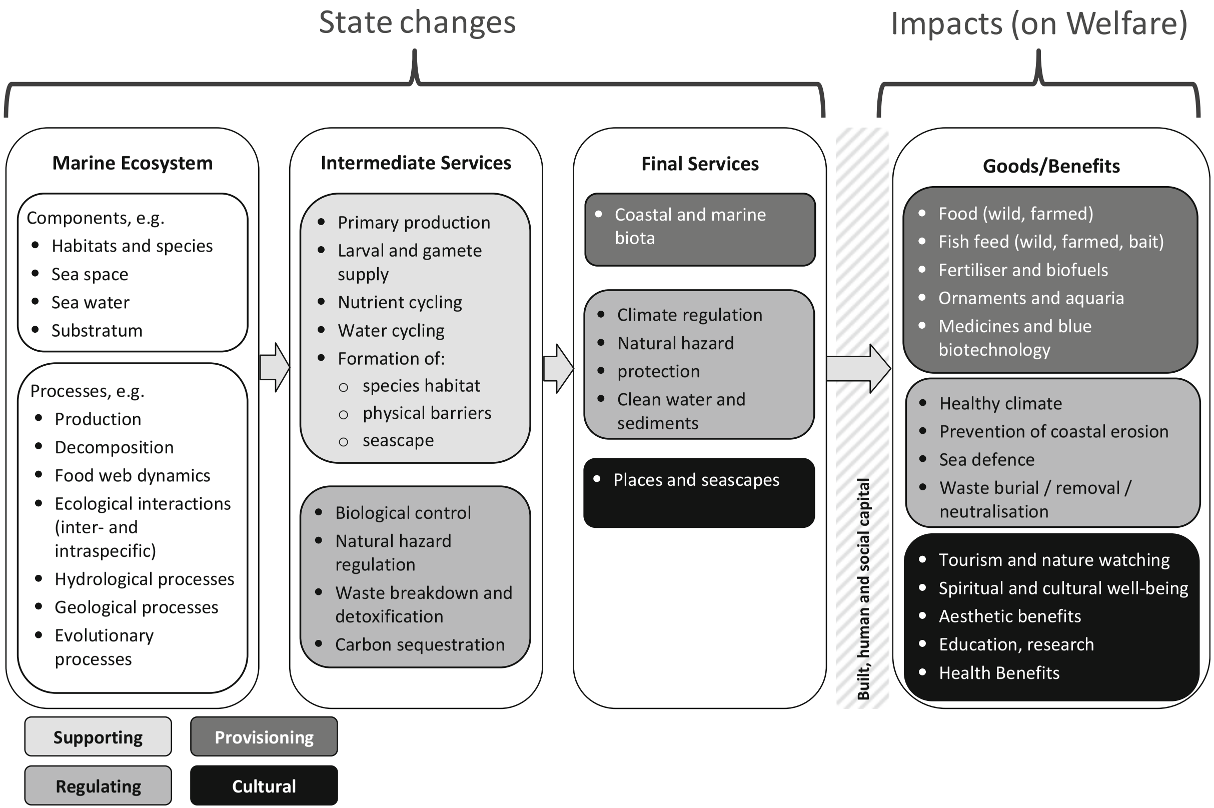

Fig. 4 State changes to the natural system reflected by changes in the marine ecosystem, intermediate and final ecosystem services (left hand side), and Impacts (on human Welfare) reflected by changes to the provision of Societal goods and benefits (right hand side) (modified and expanded from Turner et al. 2015)

2.5 Impact (I) (on Human Welfare)

Once the natural physico-chemical and ecological state has been changed by human activities and pressures then this could eventually influence the marine goods and benefits obtained by society. In DAPSI(W)R(M), this then is reflected as Impacts (on human Welfare). While Cooper (2013) changed the term 'Impact' to 'Welfare' this created a further confusion as it is the Impact *on* our Welfare that is of concern rather than Welfare *per se* (hence the use of parentheses). Hence, those Impacts (on Welfare) reflect the negative or even positive changes in the system providing societal goods and benefits (as defined by Turner et al. 2015, see below). Accordingly, it is necessary to derive and use quantitative indicators to detect and explain such changes in societal welfare (Fig. 5, right hand side). As indicated, societal goods and benefits are obtained by applying human complementary capital (social, human and human-made or built capital) to the natural environment (intermediate and final ecosystem services) (Atkins et al. 2011; Elliott et al. 2017). Therefore, any adverse Impacts on human Welfare are manifest as an inability of the marine system to provide those societal goods and benefits. For example, a healthy natural system is required to create natural places and seascapes which then influence our cultural appreciation of the sea. Society will then spend time and money to enjoy those benefits and, linking back to Maslow's work, this relies on us being sentient beings to appreciate the benefits—'*we need to expend energy to appreciate a blue whale even if we have never seen one*'!

The term Welfare in this element of DAPSI(W)R(M) also includes human well-being and happiness, two important Drivers in the upper part of Maslow's triangle.

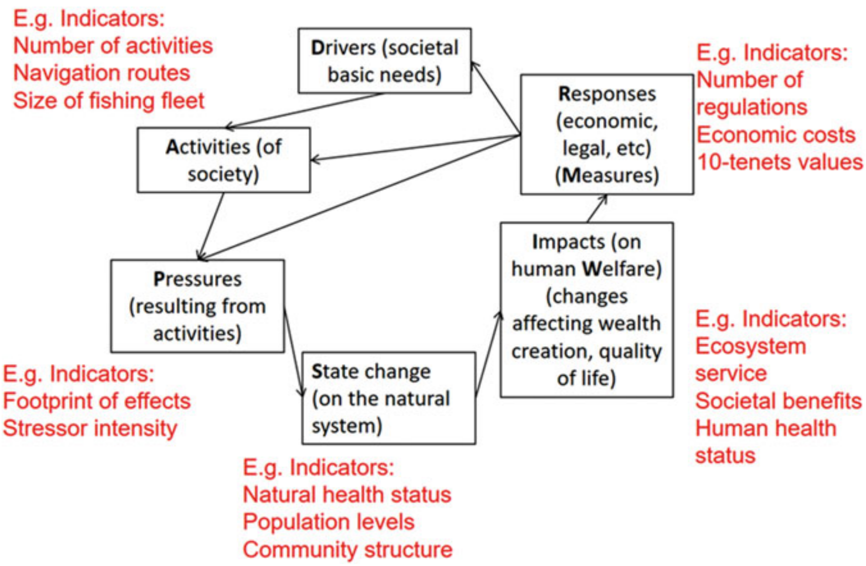

Fig. 5 Examples of indicators for each element of the DAPSI(W)R(M) framework

In addition, given the benefits that we extract from the sea and coastline, this part of the framework as the result of human Activities and Pressures reflects any adverse changes which affect Blue Growth and hence the Blue Economy as defined above. In addition, by adding complementary capital helps to identify the wider human and societal consequences, such as loss or gain in employment. In turn, such employment has a feedback loop to the Activities within the DAPSI(W)R(M) framework. As such, there is the need for operational indicators of the Impacts (on human Welfare) to show the impact of damaging Activities and Pressures assets valued by society (Turner and Schaafsma 2015). Some authors emphasise that changes in Welfare resulting from environmental state changes are the (environmental) costs which need to be traded against the benefits created by the Drivers in Cost-Benefit Analysis (Cooper 2013).

2.6 Response (R) (as Management Measures)

Marine management is dependent on a governance background in which governance relates to the marine policies, politics, administration and legislation (Boyes and Elliott 2014, 2015) as well as other management mechanisms such as economic instruments and technological developments. The management Responses which are required to overcome adverse effects on the natural and social systems (State Change and Impacts on human Welfare respectively) as the result of the Drivers, Activities and Pressures, then should include management Measures. The latter term is used as it appears widely in European Union Directives such as the MSFD and WFD (Borja et al. 2010).

From a regulatory perspective, Responses may be directed at any other element in the DAPSI(W)R(M) cycle but usually we measure the State change and Impacts (on human Welfare) but we use management. Measures to control the Drivers, Activities and Impacts (on human Welfare). A direct response to increasing Drivers might include curtailing economic or population growth. Responses may act on Activities through regulating the levels of activity (e.g. a ban on fishing) or on Pressures by reducing the levels of pressure resulting from a given level of activity (e.g. by modifying the type of fishing gear or employing mitigation and/or compensation). Restoration measures (replanting of saltmarsh, or stocking of fish) are ecoengineering Responses that acts directly on the State of the environment (Elliott et al. 2016). Compensation for environmental damage (for example following an oil spill) is of three types—to compensate the users (e.g. Responses directed at Impacts (on Welfare) of fishermen), the habitat (by habitat creation), or the resource (such as restocking and replanting)—the last two are management responses directed at rectifying State change.

There are many elements involved in defining successful and sustainable adaptive responses. Barnard and Elliott (2015) suggest 10-tenets for successful Measures—that our actions should be *ecologically sustainable, economically viable, technologically feasible, socially desirable/tolerable, administratively achievable, legally* *permissible, politically expedient, ethically defensible (morally correct), culturally inclusive* and *effectively communicated*. While directives and regulations may mandate specific measures and responses to a particular problem, the levels of Response to an environmental problem also depend on social and political perceptions of that problem. For example, increasing awareness of plastics in the marine environment has led to widespread public support, just as the protection of iconic species, the so-called charismatic megafauna, may resonate more easily with the public. However, detecting the problem is only the start of devising management measures (Borja and Elliott 2019).

All of the above relates to the natural and human-derived hazards in the environment and the way in which these become risks when they affect something which we value (Elliott et al. 2014), hence the DAPSI(W)R(M) framework becomes an integral part of a risk assessment and risk management framework (Cormier et al. 2019). However, it has to include adaptive management as in many cases the societal and management response to a particular event is unpredictable. For example the tsunami resulting from the exogenous unmanaged pressure of the 2011 earthquake off Japan and the resulting Fukishima nuclear disaster, resulted in sudden change in attitudes toward nuclear energy in Germany ultimately resulting in the sudden change of German nuclear energy policy (Strunz 2014).

As mentioned above, the elements in DPASI(W)R(M) framework are integral to the management of the seas. It is axiomatic that management cannot be achieved without measurement and that quantitative indicators are needed to determine the amount of each element and to determine whether the management has had the desired effect. Although Teixeira et al. (2016) shows the plethora of marine indicators in existence, Fig. 5 shows the types of indicators adopted for each of the elements.

Learning lessons from the evolution of DAPSI(W)R(M)—the evolving conceptual basis for EBM.

The description of the DAPSI(W)R(M) above illustrates how the simple DPSIR conceptual frame can be expanded, developed and applied to the MSFD. The evolution of DPSIR illustrates how information and concepts from different disciplines have informed the overall approach to analysis, as well as identifying and providing solutions to the problems of disciplinary silos in multidisciplinary research. The DPSIR and its successors (mDPSIR, DPSWR, and now DAPSI(W)R(M)) have for many European scientists and environmental managers been the basis of attempts to integrate our understanding of the ecological and social systems to develop an Ecosystem-Based Approach as mandated by the Directives. This has been a process of "learning by doing" (i.e. adaptive management *per se*) involving iterative improvement and refinement of the conceptual framework such that it now meets the regulatory needs of the Directives, and hence is embedded within the implementation process (European Commission 2017). Following three decades of interdisciplinary research, the framework has now been tailored to meet the requirements of the MSFD, integrating social and ecological information, linking cause-consequence-management, and providing an overarching frame for application and a standardisation of the Directive across European Member States.

The origins of DPSIR as a social environmental accounting framework and its application to a centralised European marine management policy have helped to elucidate, and overcome many of the conceptual difficulties with the practice of socio-ecological system research in the context of Europe-wide marine management. As a Framework Directive, the MSFD seeks to develop an Ecosystem Approach to management while also bringing together existing, more sectoral European marine environmental legislation including the Habitats & Species Directive (European Economic Community 1992) the Water Framework Directive (European Commission 2000) the legislation under the Common Fisheries policy amongst others. Hence while the implementation of the MSFD has developed rapidly since introduced in 2008 (e.g. Boyes et al. 2016), evidence for real improvements to the quality of European marine ecosystems is limited. There is increasing recognition that the sectoral policies for the environment and natural resource management within Europe are failing when it comes to the delivery of biodiversity objectives (Pe'er et al. 2019; O'Higgins 2017; Rouillard et al. 2018) and hence the requirement to develop more holistic approaches to European EBM (Lago et al. 2019). The question then arises, if one were to develop a conceptual framework for EBM *a priori*, what lessons can be learned from the DAPSI(W)R(M).

3 Standing on the Shoulders of Giants

Integration of social and ecological information relevant to stakeholders and managers is an essential component of any efforts which aim to remediate environmental impacts while reaching multiple policy goals. Such goals include those defined under EU Directives and strategies such as the WFD, the MSFD and the EU Biodiversity Strategy to 2020, and globally in the UNCED Sustainable Development Goals and various strategies are incorporated into different conceptual models to support EBM (Ogden 2005; Kelble et al. 2013). The first conceptual models, had a more environmental focus following the OECD pressure-state-impact (PSI). The Response dimension was added subsequently to incorporate policy responses, as in the PSR model (Gentile et al. 2001). In these first linear models, the entire social system was included in the 'Pressure' dimension and the ecological system under 'State'. These models gave little insight into the social processes that result in multiple pressures nor did they describe the entire management cycle. Furthermore, their conceptualisation of ecological systems was limited to the specific structural parameters described by the State term, rather than considering a comprehensive analysis of ecological processes and functions. The evolution of these model into DPSIR and its successors through to the DAPSI(W)R(M) resulted in a number of improvements.

Advances resulted from adding the basic human needs (anthropogenic 'Drivers') as the ultimate causes of ecosystem use and ecosystem change, incorporate a better understanding of the *raison d'etre* and functioning of the social system. Adding the

'Impact on (human Welfare)' dimension helped to provide deeper understanding of the consequences of human pressures on ecosystems (Sekowski et al. 2012). Finally, by incorporating the Response (using management Measures) these models have become important tools in the assessment of terrestrial, freshwater and marine ecosystems (e.g. Atkins et al. 2011; Tscherning et al. 2012; Kelble et al. 2013; Scharin et al. 2016).

This family of conceptual models has supported particular progress in the understanding and mainstreaming of impact pathways through which human activities affect the natural environment, both positively and negatively. However, many previous applications of the DPSIR have focussed on the single, pressures in a particular ecosystem. They are in danger of neglecting the simultaneously effects of multiple interacting pressures (Judd et al. 2015) and only rarely address the complexity associated with the assessment of multiple nested DPSIR causal chains running simultaneously (e.g. Atkins et al. 2011; Culhane et al. 2020). Burdon et al. (2018) shows the multiple links within these chains and the level of detail of the activities and pressures required by managers, in this case for oil and gas decommissioning. Hence there is the need for cumulative effects assessments which can accommodate the multiple activities, pressures, state changes and impacts on human welfare (Lonsdale et al. 2020).

All human activities occur within and are (directly or indirectly) entirely dependent on ecosystems (Boumans et al. 2002). The DPSIR models include information on the importance of nature for human welfare, i.e. integrating the linkage between ecosystem functions and ecosystem services and societal goods and benefits. Ecosystem services and societal goods and benefits are implicit in Maslow's (1943) hierarchy of needs. Within DAPSI(W)R(M), the link between ecosystem state change and human welfare is explicitly and negatively represented (generally by the costs, as economic externalities, in the cost-benefit analysis) (Fig. 3). While the importance of ecosystem services in assessing the human impacts of state changes has been recognised during the evolution of these models (see Fig. 4), it is not fully integrated within the model itself. Similarly, many of the indicators mentioned in Fig. 5 can be given monetary or other means of valuation.

As an extension of this, the 'Bow-tie' risk assessment and risk-management framework (Cormier et al. 2019) (Fig. 6) can then be merged with the DAPSI(W)R(M) framework. This shows that the central environmental concern (the red circle, a State change and/or Impact (on human Welfare)) has causes (the left-hand blue rectangles—the Drivers, Activities and Pressures) which in turn lead to consequences (the right-hand red boxes, also State changes and/or Impacts (on human Welfare)). The prevention, mitigation, compensation and adaptation controls then represent the Responses (using management Measures) (inserted between the causes and the consequences). In this sense, the model implicitly represents the centrality of the ecosystem in the production of human welfare and hence this aspect needs to be expanded to fully reflect EBM.

As a result, approaches based on the DPSIR need to incorporate feedback loops and cumulative forward and backward processes, hence favouring responses that are reactive and remedial rather than proactive and pre-emptive. Because of this, they

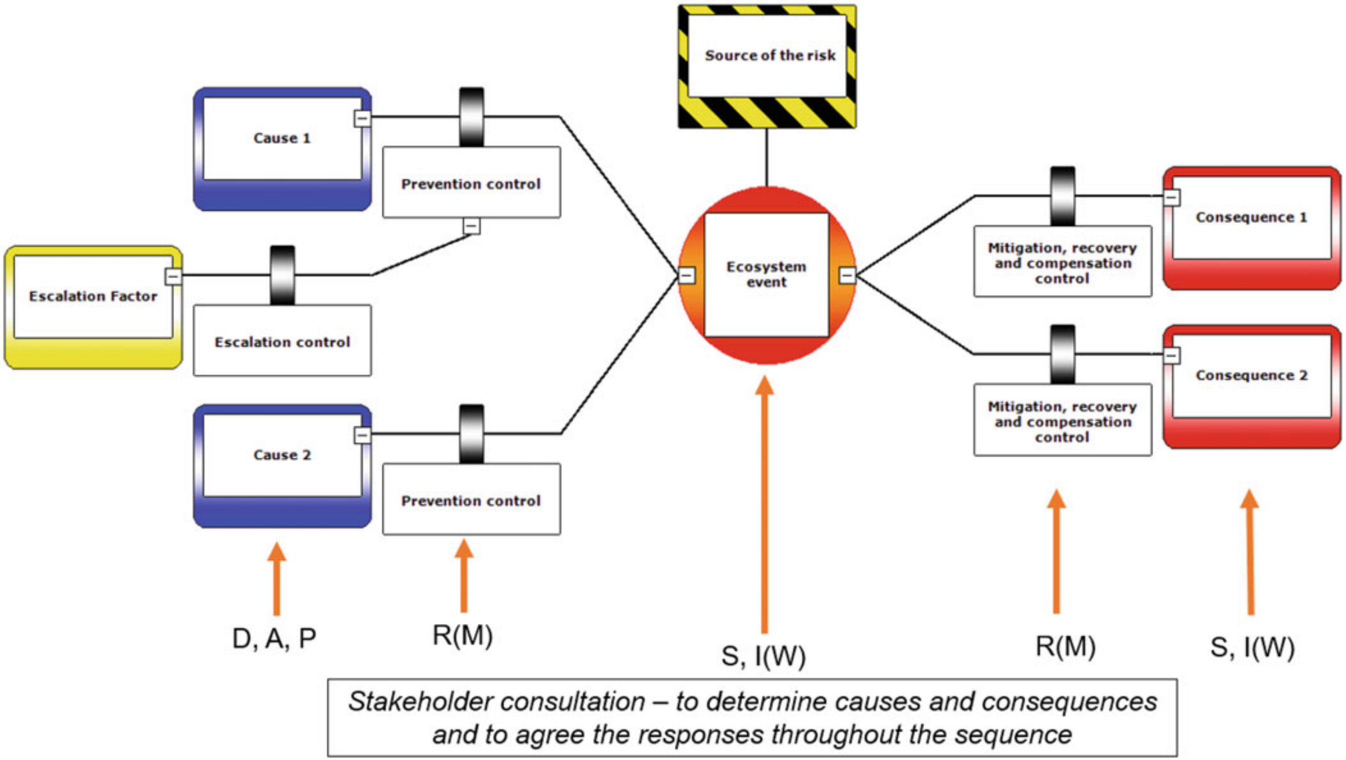

Fig. 6 A Bow-tie diagram (from Cormier et al. 2019, see text for explanation)

may be better suited to assess responses that reduce or modify pressures, regardless of how the socio-economic system and stakeholders adapt their decisions and behaviour and the drivers themselves of ecosystem change. Despite their adaptation to incorporate more fully social-ecological interactions, the causal chains of DAPSI (W)R(M) still reflect their origin in the field of environmental risk assessment. A geographic area and its management will thus need overlapping and interlinked DAPSI(W)R(M) cycles which also may need to be linked to similar cycles outside the immediate management area (see Elliott et al. 2017).

With the advent of the Ecosystem Approach in the 1990s, starting from the global Convention for Biological Diversity (see Enright and Boetler 2020 for a history of the term), the analysis of ecosystem services and their importance for human welfare and societal goods and benefits (Constanza et al. 1997; MEA 2003) shifted the perspective from "*what have we done to nature?*" towards "*what does nature do for us?*". This recognises the centrality of nature and ecosystems services in human well-being, a defining feature of EBM (Tallis et al. 2010). The ecosystem services approach led to a more comprehensive framework including economic perspectives and it called for more effective social action. This has led to new perspectives based upon the potential of ecosystems to provide society with the valuable goods and services they demand and to new conceptual frameworks to integrate these new concepts based on the previous DPSIR (Turner 2000; Cheong 2008; Weinstein 2009) leading to the DAPSI(W)R(M) described above. Ecosystem services and the resulting societal goods and benefits provided the missing analytical block to proceed from the biophysical to the human dimensions of science. Ecosystem services and the resulting societal goods and benefits are the main and most welfare-relevant outcomes from the interaction of social and ecological systems. Therefore, linking ecological and social systems to human welfare (and the goods and benefits it demands) through the notion of ecosystem services is essential to understand and assess the multiple trade-offs involved in individual and collective decisions in a clear and consistent manner and to the development of an EBM conceptual framework.

Ecosystem services and societal goods and benefits are the key emerging outcomes of the interaction between ecological and socio-economic systems (Biggs et al. 2012). They are ‘produced’ and delivered by ecosystems but are also continually shaped by their interaction with socio-economic systems and, in the case of societal goods and benefits, require an input of human capital in order to be realised. They may also favour detrimental, transformative or restorative processes (Biggs et al. 2015). Human actions and institutions shape ecosystem structures, processes and services in landscapes or seascapes by management and uses/users, which in turn shape human behaviour and institutional settings.

The integration of both traditions, impact pathway analysis and the ecosystem services/societal goods and benefits approach, has fostered the emergence of a growing number of alternative socio-ecological system analytical frameworks (Binder et al. 2013). Nevertheless, their success and their capacity for the smooth integration of knowledge may have been impaired by the mismatch resulting from mixing pieces created from slightly different conceptual directions and for different purposes. To some extent, both approaches share the drawbacks of common practice and may only offer a partial view of the complex links between the relevant and important parts of social and ecological systems.

In attempting to define a conceptual framework specifically for the analysis and design of EBM, there is a clear need to reflect the central nature of ecosystem services, the resulting delivery of societal goods and benefits and the costs and benefits of enhancing natural assets or ecosystems to improve resilience and adaptability. Hence there is the need conceptually to represent how ecosystems function in connection with the socio-economic system, delivering ecosystem services and contributing to social goods, benefits and welfare.

4 The Butterfly

Taking the valuable lessons learned through the application of the DPSIR/DAPSI(W)R(M) framework, we seek to develop a transferable framework with the aim of developing, a-priori, a more holistic methodological approach to implementing Ecosystem-Based Management. This is illustrated in the 'Butterfly' diagram (Fig. 7) (see Gómez et al. 2016 for the full explanation).

Two sets of relationships between humans and nature are implicit in the cyclic DAPSI(W)R(M). Drivers are dependent on the supply of ecosystem services and societal goods and benefits (see Maslow's hierarchy of Basic Human Needs above). In turn, these Drivers (through human Activities) cause Pressures and changes in environmental State altering the supply of ecosystems services. In turn, those State changes influence the Impacts (on human Welfare). These links can readily be conceptualised as supply (from nature) and demand (by humans) for ecosystem services and societal goods and benefits.

The supply side (Fig. 7, shown in blue) involves the links between ecosystems and human Welfare. In DAPSI(W)R(M), this is conceptualised as the (cost-benefit) feedback between Impacts on human Welfare and Drivers. The second set of relationships, 'the demand side' (Fig. 7, shown in yellow), refers to how social systems shape and change ecosystems (the links between Drivers and Activities to Pressures and then State changes). These are connected to each other through complex adaptive processes taking place in ecological and social systems.

The supply-side relationship goes from the ecological to the social system. It represents the potential of ecosystems to supply and effectively deliver ecosystem services to the social system, from which it gets goods and benefits. It includes the capacity of the social system to transform those services delivered into benefits for society through human built, financial and social capital (Fig. 3). This is all contingent on the ecosystem structure and on those processes/functioning taking place in the biophysical system from which ecosystem services lead to the most socially relevant outcomes.

The demand-side relationship goes from the social to the ecological system. It represents and explains the demand and the effective use of ecosystem services and

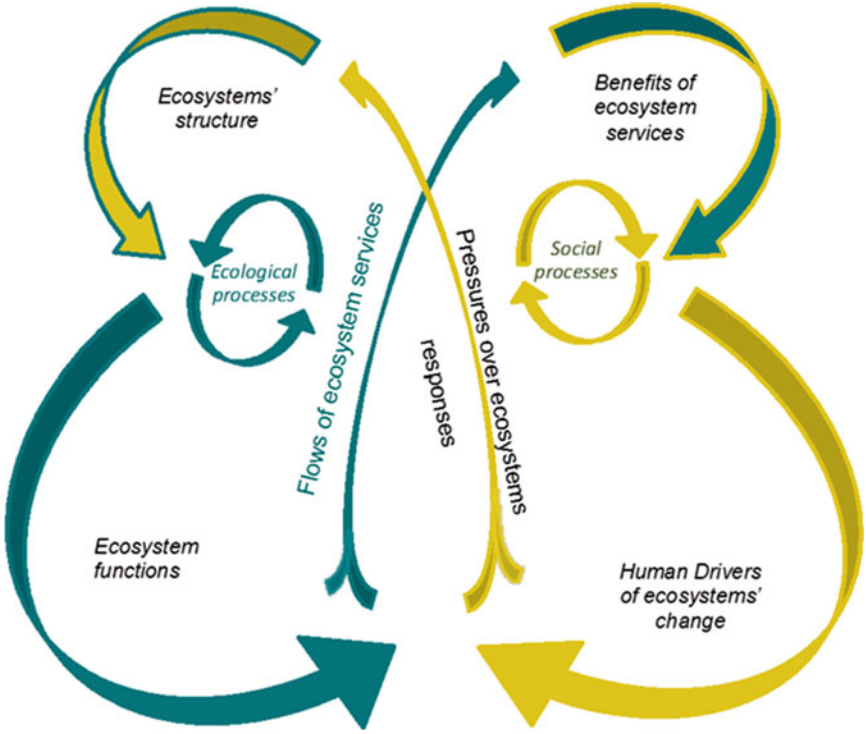

Fig. 7 The butterfly diagram, a new model for the assessment and design of Ecosystem-Based Management. Supply side is shown in blue, demand side is shown in yellow

the impacts on ecosystems. The demand for ecosystem services (and in turn goods and benefits) depends on income, tastes, technology, institutions, and other social and economic factors. Beyond pressures on ecosystems, this demand-side relationship also considers social and individual decisions towards protecting and restoring ecosystems in order to preserve their benefits depending on the governance institutions in place. Hence the need for the ten-tenets of sustainable ecosystem management which can incorporate all the facets of that management, both the natural and the societal. These more fully represent the variety of social processes by which society adapts to manage specific environmental and social conditions (Ostrom 1990) including new types of social innovation (Schor and Thomson 2014), going beyond Responses (and management Measures). This may be best suited to governance or economic instruments, best available technologies, cultural demands, etc. as summarised by the 10-tenets (Barnard and Elliott 2015).

5 Conclusions

Here we present the DAPSI(W)R(M) framework, the latest iteration of the well-known and widely-applied DPSIR framework, as an environmental risk management tool. We draw on advances over the past 3 decades in the analysis of socio-ecological systems, and now include elements such as ecosystem services and societal goods and benefits as well as more a holistic conception of Drivers. However, we emphasise that the causal chain mechanism (of cause-consequence-management) needs to be ideally suited to the cyclical application of adaptive management required under the framework of European environmental directives. The derived system has to fully incorporate the wealth of social and ecological process that result in the dynamics of supply and demand for ecosystem services and the resulting societally goods and benefits, the most socially relevant outputs from ecosystems. The Butterfly enhances the inherited conceptual framework of DAPSI (W)R(M). The Butterfly conceptual framework has been systematically tested through a suite of case studies in aquatic biodiversity management from freshwater rivers (Domisch et al. 2019) and lakes to estuarine and marine systems, some examples of which are presented in this volume (Piet et al. 2020, O'Higgins et al. 2020; Lillebø et al. 2020; Funk et al. 2020).

The ability to meet the predominant aim of satisfying Ecosystem-Based Management, covering both natural and social systems, requires the analysis of socio-ecological systems enabled by the cause-consequence-response chain, and the relationship to European (and other) environmental policies. The widely acknowledged biodiversity and climate crises requires holistic management approaches such as EBM which can then be translated into the supply and demand for ecosystem services and societal goods and benefits at the centre of the Butterfly conceptual framework.

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The Promise and Pitfalls of Ecosystem Services Classification and Valuation

Stephen Flood, Timothy G. O'Higgins, and Manuel Lago

Abstract We are currently facing the triple interlocked threats of climate change, unsustainable land use change, and the sixth mass species extinction. This chapter firstly outlines these interlinked threats making the case for urgent action. It then documents the efforts to assign economic and non-economic value to our biodiversity (Ecosystem Services) through environmental and ecological economics, highlighting the fundamental philosophical principles underlining both approaches. This sets up a discussion on the development and potential of Ecosystem Services (ESS) as a discipline in its own right, the challenges of application, and the awareness of, and priority assigned to, ESS by policymakers and the private sector. The chapter closes by outlining specific methodological challenges and recommendations for substantially increasing the level of attention and action needed to protect and enhance our invaluable ecosystems in our age of potential ecological collapse.

Elements of this chapter are based on the PhD thesis of Flood, S. (2012), which is available online at: http://mural.maynoothuniversity.ie/4760/

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Lessons Learned

- We have documented the development of ESS, from its foundations in environmental and ecological economics to its evolution into FEGS and CICES classifications and valuation frameworks.
- In doing so we have highlighted the challenges of valuation (including issues of scale, biodiversity awareness or literacy, and polycentric governance), and provided lessons and recommendations going forward for the successful application of ESS.
- Combining a number of different tools and methods can help strengthen assessment
- A persistent gap persists between ESS applications and their ability to provide easily usable information for decision-makers
- ESS can help shine a light on what we will lose if we fail to protect valuable (and indeed invaluable) ecosystems and earth's flora and fauna in general.

Needs to Advance EBM

- Keep it simple – Decision-makers are interested in simple, easy-to-use decision support tools that are understandable and can be easily incorporated into science-policy processes (Ruckelshaus et al. 2015; IPBES 2019; Dunford et al. 2018). Scientists should note that even basic tools are ample for parameterizing and interpreting data at the early stages of applying Biodiversity and Ecosystem Services (BES) information (Ruckelshaus et al. 2015);
- It's not always about the money – Attributing economic values to biophysical ecosystem service estimates is an important conceptual advance. This ability to follow biophysical estimates though to economic value has allowed decision makers to begin having conversations they did not previously engage with, and lead to new policy outcomes (Dunford et al. 2018; Ruckelshaus et al. 2015; Barton et al. 2018);
- Relate BES change to livelihoods and other wellbeing metrics.

1 Introduction

"We are the first generation that has a clear picture of the value of nature and its integral link with human well-being. We are also the last generation that has the opportunity to prevent the collapse of our planet's biodiversity in the face of habitat destruction and climate change." (WWF 2018, p. 10).

The scientific evidence indicates that the Earth's climate is changing (IPCC 2014) and, without taking appropriate and early action, climate change will have severe impacts on many of the planet's species and habitats (Scheffers et al. 2016). The 2006 Stern Review emphasises that the benefits of strong early action on climate change is likely to outweigh the costs, and values the cost of inaction at 5% of global GDP each year indefinitely (Stern 2006). It is important to note the value of global biodiversity is not fully captured in this percentage and the fact that crossing critical tipping points in our ecosystems, that would lead to extensive and run-away species and habitat loss, is also not captured. Yet research into the value of ecosystem services reveals that eco-services contribute more than twice as much to human well-being as global GDP (Costanza et al. 2014) and greater investment into the restoration and protection of the ecosystems and habitats that make those services possible can increase resilience to climate change.

1.1 Climate and Biodiversity Crises and the Need for Change

The secretariat of the Convention on Biodiversity considered the interlinkages between climate change adaptation and mitigation and biodiversity in a technical report published in 2009. The report established that biodiversity and climate change are interconnected because climate change effects biodiversity and because changes in biodiversity affects our ability reduce our atmospheric greenhouse gas levels (e.g. our natural carbon sinks) and to adapt to and mitigate against the impacts of climate change (CBD Secretariat 2009). It also highlighted the potential of ecosystem-based adaptation to create co-benefits for climate action and biodiversity conservation.

Furthermore, a recent special report by the Intergovernmental Panel on Climate Change (IPCC) indicates significant impacts to biodiversity and other sectors are set to occur even if we keep climate change to 1.5 °C over preindustrial levels, which are below business-as-usual global average temperature increases by mid-century (IPCC 2018). Biodiversity is at the forefront of climate change impacts globally. Headline results from the 2018 Living Planet Report, published by the World Wildlife Fund (WWF 2018), reveal that Earth is losing biodiversity at a rate seen only during mass extinctions. The report finds that global losses in populations of vertebrate species—mammals, fish, birds, amphibians and reptiles—have averaged 60% between 1970 and 2014. Overexploitation of species, agriculture, land conversion, and climate change are the main drivers of biodiversity decline, with climate change becoming a growing threat (Ibid.).

The 2018 Conference of the Parties to the Convention on Biodiversity and the 2019 report from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) also echo the findings of the WWF Living Planet report in highlighting the critical role of biodiversity and ecosystems functions and services for human well-being with the IPBES reporting that the health of ecosystems, on which we all and all other species depend, is deteriorating at a rapid rate, and only through ‘transformative change’ can nature be conserved, restored and used sustainably (CBD Secretariat 2018; IPBES 2019). Transformative change should be understood as a fundamental, system-wide reorganisation across technological, economic and social factors, including paradigms, goals and values (IPBES 2019). It was also recognised that climate change is a major and growing driver of biodiversity loss, and that biodiversity and ecosystem functions and services, significantly contribute to climate change adaptation, mitigation and disaster risk reduction.

The IPCC’s (2019) Special Report on Climate Change and Land states with high confidence that increasing impacts on land, ecosystems and biodiversity are projected under all greenhouse gas emission scenarios with cascading risks occurring across systems and sectors (IPCC 2019). It also states with high confidence that near-term actions to promote sustainable land management will help reduce land and food-related vulnerabilities, provide both short-term positive economic returns and longer-term benefits for climate change adaptation and mitigation, biodiversity and enhance ecosystem functions and services.

1.2 Ecological Damage as an Externality

Traditional neo-classical economic approaches neglect to account for market failures of ecological damage due to absence of markets for many environmental public goods.

The field of environmental economics was established to address these types of market failures or 'externalities' and aims to internalise market externalities through considering and capturing social and environmental costs relating to economic activities (Flood 2012; Tietenberg and Lewis 2007).

There are two requirements for decision-making when it comes to quantifying environmental damages.

The first is to determine one's fundamental philosophical position and, contingent on one's worldview, the second is the need to know the extent to which people are willing-to-pay to prevent damages or the willingness-to-accept compensation for damages suffered (Spash 1997).

The philosophical position assumed by environmental economists is that the net utility from the consequences of an action determines whether the action is right or wrong.

Cost-benefit analysis and its tools, such as the contingent valuation method, assume that individuals are able and willing to consider trade-offs in relation to public goods, i.e. that individuals follow a utilitarian philosophy (Ibid.).

The contingent valuation method involves directly questioning people, in a survey or interview, how much they would be willing to pay for specific environmental services.

It is called "contingent" valuation, because people are asked to state their willingness to pay, contingent on a specific hypothetical scenario and description of the environmental service.

This utilitarian standpoint is the approach from which the majority of socioeconomic impacts associated with ecological damage are approached in the literature (Flood 2012).

This tendency towards the single metric of monetary valuation and the reluctance of the mainstream to consider other numéraires finds its roots in the epistemology of the Enlightenment or Age of Reason (Flood 2012). Enlightenment thinking originates with 17th and eighteenth century European thinkers such as Voltaire, Rousseau, Kant and Hegel, with foundations built upon the theories of Descartes (Van Asselt and Rotmans 2002). In unpacking the field of economic evalution of ecosystem services, the question of substitutability is one that burns at the very core of the

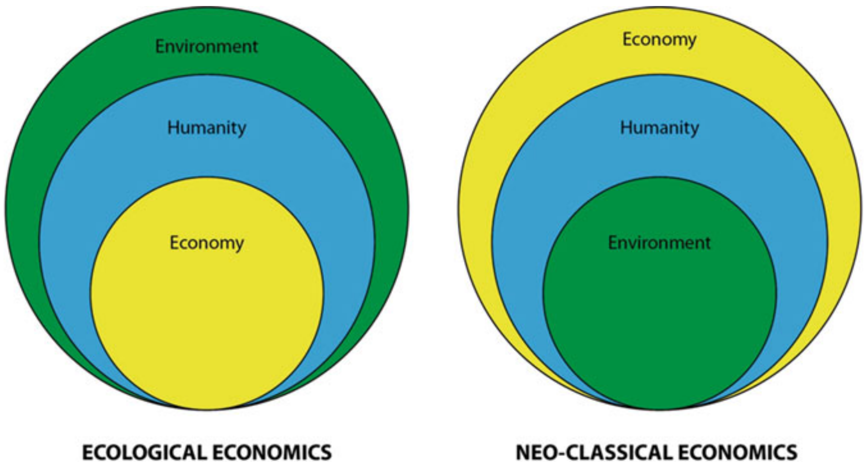

Fig. 1 Foundations of ecological economics and environmental economics

debate and is intertwined with questions of ethics (Flood 2012). The implicit utilitarian viewpoint of environmental economics, and in particular cost-benefit analysis, precludes the preservationist perspective which focuses on non-human intrinsic values associated with environmental systems (Spash 1997). Most environmental policy is couched in terms of calculating the usefulness to humans of preserving specific environmental goods and services provided by environmental systems. This contrasts with the foundations of ecological economics.

Ecological economics is holistic in its approach and much less anthropocentric than environmental economics. It also tends towards rights-based thinking. Figure 1 displays the fundamental differences between ecological economics and traditional neo-classical economics approaches, in terms of their view of the environment, economy and humanity (Flood 2012). Neo-classical economics tends to view the environment and humanity as embedded within the economy. Ecological economics takes a more holistic approach and considers the economy as a part of humanity living within its environment. Making decisions on a utilitarian basis is considered the most sensible approach by the majority of economists (Spash 1997).

However, we must be careful not obviate the fact that it is the analysis of public policy choices (housing, transport, etc) that also marks the way analytical approaches are designed for environmental protection. If we seek economic efficiency in public policy choices, to ascertain if the investment is worth it or money would be better employed somewhere else, the rationalist would suggest the logic of employing the same approach to inform policy choices for the environment in the absence of alternative decision-making techniques or approaches. Moreover, the challenge is that if we take decisions about the environment outside conventional decision making approaches, we may well end up marginalising environmental decisions and, as a society, be unable to assess where money is best employed. In other words, the risk of the ecological economics approach is that environmental decision making may suffer from isolation from other public policy areas, as it is very difficult to consider or mix monetary budget lines with other metrics. It can be argued that looking at environmental choices in isolation will not lead to real-world solutions, and rather they should be considered in the context of a wider public policy debate.

1.3 Traction: ESS as a Discipline in Its Own Right

Ecosystem Services (ESS) are benefits humans recognise as obtained from an ecosystem and that support, directly or indirectly, their survival and quality of life (Millennium Ecosystem Assessment 2005). This recognition of the idea of ‘natural capital’ was first coined in the book Small is Beautiful by E.F Schumacher in the 1970s (Schumacher 1973). The term ‘environmental services’ was introduced in a 1970 report: The Study of Critical Problems (MIT Press 1970). The services listed in the report included flood control, climate regulation, insect pollination, and fisheries. This concept ESS has continued to develop and expand to include both conservation and socio-economic objectives. Ecosystem goods and ecosystem services were combined by Robert Costanza and his colleagues in the Millennium Ecosystem Assessment (MEA 2005). The Assessment conceptualised the interactions between biodiversity, ecosystem services, human well-being, and drivers of change (Fig. 2).

Changes in drivers that indirectly affect biodiversity, such as population, technology, and lifestyle (upper right corner of Figure), can lead to changes in drivers directly affecting biodiversity, such as the catch of fish or the application of fertilizers (lower right corner). These result in changes to ecosystems and the services they provide (lower left corner), thereby affecting human well-being. These interactions can take place at more than one scale and can cross scales. For example, an international demand for timber may lead to a regional loss of forest cover, which increases flood magnitude along a local stretch of a river. Similarly, the interactions can take place across different time scales. Different strategies and interventions can be applied at many points in this framework to enhance human well-being and conserve ecosystems.

2 State of the Art ESS Concepts Complexity and Simplicity

The growing focus on the application of ESS to real world problems has led to a continuous refinement of Ecosystem Services concepts which has reflected the multidisciplinary nature of the research area, at the interface between ecology and society. While the classification system of the Millennium Ecosystem Assessment (MEA), which includes the supporting services, incorporates the full complexity of interactions between ecosystems and human beings, in practice inclusion of

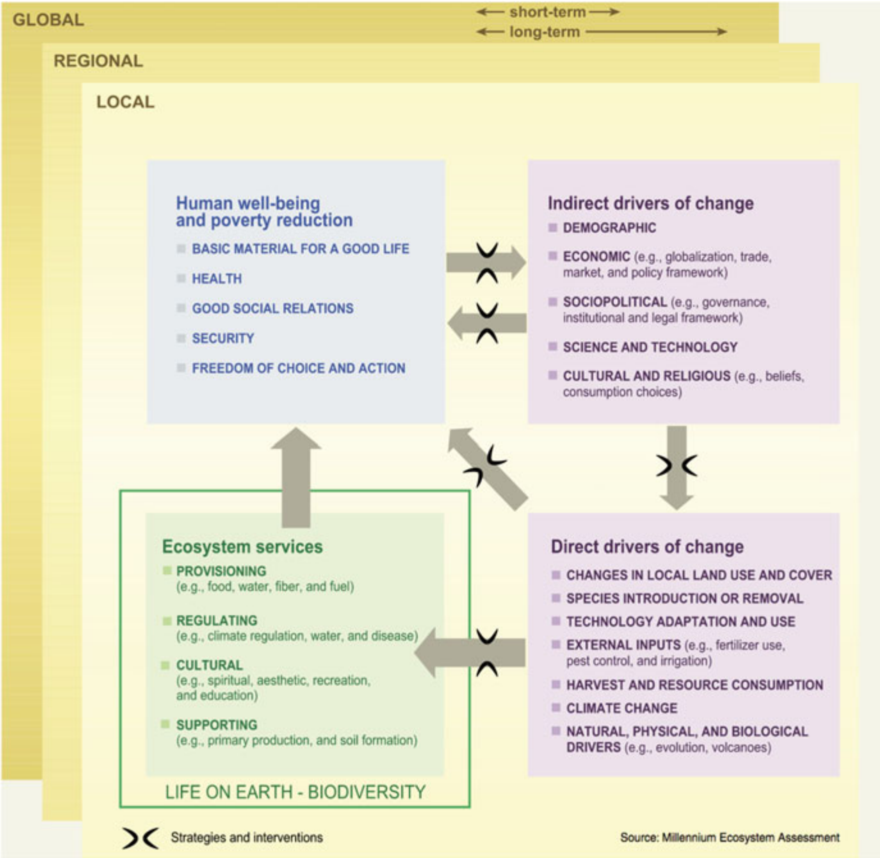

Fig. 2 Millennium ecosystem assessment conceptual framework of interactions between biodiversity, ecosystem services, human well-being, and drivers of change

supporting services in economic analysis can results in double counting. For example, a salmon may result in benefits to human beings, either through commercial harvest (as a provisioning service) or through recreational harvest (as a cultural service). The resulting economic benefits are, the market value (for commercial fisheries) or recreational enjoyment value of the caught salmon (for the recreationally caught fish), less the cost incurred in catching it. If the supporting services such as, habitat provision by freshwater, insect production as a food source for the fish, were also to be valued economically, the *human* benefits from nature would be double counted. By analogy in valuing a car, we consider the value of the final product but do not separately account for the value of the car manufacturing plant (built capital) and the raw materials (inputs). This practical challenge has been met by development of the Final Ecosystem Goods and Services (FEGS) concept. *Final ecosystem services* are "components of nature, directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf 2007). It should also be noted

Table 1 CICES classification system, with examples of class type

Section	Division	Group	Class	Code	Class type
Provisioning (biotic)	Biomass	Reared aquatic animals for nutrition, materials or energy	Animals reared by in-situ aquaculture for nutritional purposes	1.1.4.3	Salmon
Regulating and maintenance (biotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of wastere or toxic substances of anthropogenic origin by living processes	Bio-remediation by micro-organisms, algae, plants and animals	2.1.1.1	In coastal marine systems
Cultural (abiotic)	Direct, in-situ and outdoor interactions with natural physical systems that requires presence in the environmental setting	Intellectual and representative interactions with abiotic components of the natural environment	Natural, abiotic characteristics of nature that enable spiritual symbolic and other interactions	6.1.2.1	Enjoyment of a land or sea scape

that in making just environmental decisions, the consideration of beneficiaries (who benefits) as well as benefits (what type of benefits) is all vital. Applications of the FEGS approaches are described in more detail in this volume by DeWitt et al. (2020) and Yee et al. (2020) and the FEGS concept is essential in the application for the economic valuation of Ecosystem Services and these concepts are commonly employed by ecological economists.

However, the FEGS are only a subset of all the ecosystem services which contribute to human well-being. Internationally, in order to standardise efforts in environmental accounting, considerable efforts have been expended in developing standardised classification systems. Within Europe, the Common International Classification of Ecosystem Services (CICES) is widely adopted, while at the national level with the U.S. the National Ecosystem Services Classification System (NESCS) is emerging (USEPA 2015). These systems attempt to standardise and codify the analysis of ecosystem services, with the aim of informing efforts such as the United Nations System of Environmental-Economic Accounting. The CICES classification system is analogous to the Linnaean biological classification system (phylum, class, genus species) and is comprised of four discrete categories (Table 1).

The CICES classification is the most widely used in Europe, and while developed for the purposes of environmental accounting, it has also been used in efforts to map the supply of ecosystem services (Maes et al. 2015). As with other classification systems, the supporting services of the millennium ecosystem assessment have been dropped and are now incorporated as regulating and maintenance services. CICES also recognises the benefits from non-living components of the environment, as abiotic services. The inclusion of some services which may be seen as intermediate rather than final services is one critique of this classification system, and may reflect a disciplinary bias toward ecology in the development of the system.

The focus of ES research, whether encompassing demand-side, human used of FEGS or covering the whole suite of supporting services and final services often depends on the disciplinary background of the researcher. Economists tend to focus only on FEGS and beneficiaries while ecological researchers with an interest in ecosystem services tend to focus on the full range of services, reflecting their interests in the functioning of ecosystems as a whole and the objective of the analysis, whether for simple accounting or to justify actions to maximize specific services.

2.1 Challenge of Valuation

While all these systems are designed for the purposes of accounting, the challenges of constructing of an agreed international standard are considerable, and will no doubt continue for many years, in many cases the process of (both monetary and non-monetary) valuation itself presents major challenges. For non-market goods and services there are two major categories of valuation methodologies. Revealed preference methodologies are an indirect methods of estimating the monetary value of an ecosystem service based on how much people spend to access or travel to a site (Silvertown 2015).

Revealed preferences are based on indirect calculations, deriving monetary values from the effects of behavioural change associated with the service (or the lack of it) in real markets (Spangenberg and Settele 2010). They are made up of non-use values (existence values) such as knowing about the existence of a deer population in a region; non-consumptive use values (watching them) and consumptive use values (hunting them) (Ibid.). The two main assessment methods for revealed preference are hedonic pricing and travel cost estimates.

Travel cost is mainly applicable to leisure and holiday activities where travelling is voluntary. In these cases, as the *homo economicus* is always maximising their utility, and will only be travelling to a particular location if the time spent there provides more utility than saving the cost and abstaining from the visit. (Ibid.). The travel cost is therefore stand-in for the value of what has been enjoyed at the destination. The method gives higher amenity value to a visitor who travels by car than someone who travels on foot or by bicycle even though the former involves the least effort and is the most environmentally damaging (Silvertown 2015).

Knoche and Lupi (2007) calculate the value of the white-tailed deer by assessing the demand for deer hunting via the hunters' travel costs. As a result, the value of 10,000 deer more per county is the result of additional travel expenditures of US $3.94 per hunting trip for firearm hunters, and of US$1.75 per trip for archery hunters.

Hedonic pricing valuations use relationships between land property prices and property characteristics to value changes in the characteristics (Swinton et al. 2007,

p. 248).They start with the assumption that services/disservices like improved or diminished environmental quality change the willingness to pay for a good associated with them, and this is reflected in the market price (with an implicit assumption of full knowledge and perfect markets), in particular in the housing market. The price change is then a measure of the value of the ecosystem services enjoyed, like a price increase due to the establishment of a nature reserve in the neighbourhood. However, empirical work comparing the changes in individual well-being caused by pollution to housing prices have shown that they do not necessarily reflect the local environmental quality changes (Rehdanz and Maddison 2008; Spangenberg and Settele 2010).

2.2 Stated Preference- Methodological Advances and Subtleties

Stated preference approaches simulate a market for ecosystem services through the generation of surveys on hypothetical (policy-driven) changes in the provision of ecosystem services. (TEEB 2012, p. 20). Stated preference methods can be used to estimate non-use and use values of ecosystems. The main types of stated preference techniques are:

1. Contingent valuation method (CV): Uses questionnaires to ask people how much they would be willing to pay to increase or enhance the provision of an ecosystem service, or alternatively, how much they would be willing to accept for its loss or degradation;
2. Choice modeling (CM): Attempts to model the decision process of an individual in a given context (Hanley et al. 1998; Philip and MacMillan 2005). Individuals are faced with two or more alternatives with shared attributes of the services to be valued, but with different levels of attribute (one of the attributes being the money people would have to pay for the service); and
3. Group valuation: Combines stated preference techniques with elements of deliberative processes from political science (Spash 2001; Wilson and Howarth 2002), and are being increasingly used as a way to capture value types that may escape individual based surveys, such as value pluralism, incommensurability, non-human values, or social justice (Spash 2008).

There is a vibrant economic literature on the refinement of stated preference techniques, and these techniques are constantly being defined and developed.

One major drawback of non-market valuation studies, is that (unlike market values) these figures require significant time, effort and expertise to establish, and while real non-market values will vary based on any given unique situation, it is practically unfeasible for reasons of expense and expertise to carry out such studies in every case. As a result, the use of benefits transfer (the practice of taking values from existing studies and applying them at another site) is commonplace. However this practice results in uncertainty which can undermine their usefulness in decision making contexts. The potential for big data to contribute to the volume of information on human recreational use patterns is beginning to improve the capacity to tailor economic valuation studies to specific locations (Adamowicz et al. 2011).

2.3 Splitters and Lumpers: Real World Complexity, Bundling and Its Implications

In reality ecosystems services generally do not exist in isolation but emerge as a bundle, jointly produced from a range of ecosystem processes and components (Fisher et al. 2009). For example, a healthy river ecosystem provides clean drinking water and salmon and waste remediation services as well as opportunities for active and passive recreational use and the interactions between the ecosystem processes which contribute to these bundled services operate together as a system. Developing quantitative understandings of the ecosystem processes themselves and understanding how these systems respond to human activities may highly complex ecological modelling approaches (see Fulford et al. 2020), yet the joint bundled of benefits to humans is something relatively easily understood intuitively and without deep ecological knowledge. Recognition and communication of the characteristics of joint ecosystem services supply may in some cases be more useful than, providing an exhaustive list of services provided by a given system or developing a mechanistic understanding of how the different ecosystem components interact to produce services.

Carefully developed frameworks and classifications such as the Millennium Ecosystem Assessment (MEA), Final Ecosystem Goods and Services (FEGS), and the Common International Classification of Ecosystem Services (CICES) untangle the complexities of capturing and valuing ESS. However, scholars and practitioners grapple with these frameworks and classifications when applying them on the ground (Harrison et al. 2018).

While Ecosystem Services are intuitively understood, even the youngest school child can understand that “the cow gives us milk” or the “sea gives us fish” or that “a flower is pretty”. As illustrated in Table 1, the formal language used to describe ecosystem services is very precise, but not particularly accessible. While anybody can understand the concept of “a beautiful view” this, same concept is perhaps not optimally expressed as “*Intellectual and representative interactions with abiotic components of the natural environment*”. These difficulties with the communication of Ecosystem services have resulted in a suite of different terms being used, for example “natures benefits”, “natural capital”, “natures’ services”.

2.4 Scale and Polycentric Governance

While philosophical battles have been won and lost over the ethics, as well as practical and theoretical considerations of valuing nature, these debates over valuation have tended to obscure the broader applicability of ecosystem service concepts to the field of environmental management. Flows of ecosystem services from location to location establish transactional relationships between different jurisdictions. Your country may benefit from the ecosystem services produced in my country. The most obvious current example is that of the Amazon rainforest. The Amazon provides climate regulation services (as well as a wealth of biodiversity) for the entire earth, but is largely under the jurisdiction of Brazil, therefore Brazilian management practices have the potential to increase or reduce the supply of ecosystem services to all of us. This situation sets up a power dynamic between Brazil and the rest of the global community. Systematic considerations of ecosystem service flows and the geographic characteristics of supply and demand can enable the development of institutional arrangements accounting for such natural flows.

3 The Power of the Word "Biodiversity" to Communicate with the Public

In light of these issues we propose that an important component of increasing the uptake and application of ESS or BES in everyday decision-making is through mainstreaming into policy as well as the public consciousness. Here, we will concentrate on educating and informing the public consciousness. A good place to start is by looking at the awareness of different populations to the concept of biodiversity, as a foundation for understanding of ES.

A 2015 Eurobarometer captures and presents the attitudes of Europeans towards biodiversity (European Commission 2015). More than half of the 27,718 respondents agree that the European Union (EU) should better inform its citizens about the importance of biodiversity (61%), that the EU should ensure that biodiversity concerns are taken into account when planning new infrastructure investments (55%), and that it should better implement existing nature and biodiversity conservation rules (55%) (Ibid.). Furthermore, of the 60% of respondents who have heard of the term 'biodiversity', only half of them have an understanding of what it means. Those living in western and southern areas of the EU are more likely to have heard of the term 'biodiversity' and know what it means. Two thirds (66%) of Europeans do not feel informed about the loss of biodiversity, with 22% saying that they do not feel informed at all (Ibid.).

The Union for Ethical Bio Trade carried out a six-year survey (from 2009 to 2015) to determine the levels of knowledge of the term biodiversity from 47,000 consumers in 16 countries across the globe (UEBT 2015). The key findings revealed

Table 2 Biodiversity awareness around the world (Adapted from UEBT 2015) numbers indicate percentages

Region/ Country	Have heard of biodiversity	Correct definition of biodiversity	With partial definition of biodiversity
Europe			
UK	68	26	17
Netherlands	59	27	16
France	91	34	25
Germany	38	18	9
Switzerland	83	37	18
Americas			
USA	58	22	18
Mexico	90	46	20
Colombia	93	44	18
Ecuador	82	14	30
Brazil	92	44	19
Peru	52	7	37
Asia			
China	94	64	22
India	40	1	25
South Korea	73	47	16
Japan	62	29	21
Vietnam	95	36	6

that one out of three respondents could provide a current definition of biodiversity. Table 2 provides an overview of the results.

4 Recommendations

It has been suggested that combining a number of different tools and methods can help strengthen assessment (Harrison et al. 2018; Barton et al. 2018). Recent reviews point to a persistent gap in the promise of ESS to provide easily usable information for decision-makers (Ruckelshaus et al. 2015). Based on significant field-based experience of application of ESS frameworks and classifications 6 emerging lessons have been identified (Ibid.):

1. Include Biodiversity and Ecosystem Services (BES) information as part of an iterative Science-Policy Process;
2. Keep it simple – no matter how much interdisciplinary scientists think they are over-simplifying biophysical or socio-economic processes, decision-makers typically ask for simpler, easy-to-use and understandable decision support tools that can be readily incorporated into science-policy processes (Ruckelshaus et al.

2015). Even simple tools are complicated enough for parameterizing and interpreting at early stages of applying BES information;
3. It's not always about the money – having the ability to follow biophysical ecosystem service estimates through to economic values has proven to be an important conceptual advance that has opened many decision makers to discussions they previously did not consider. However, conceptually, considering values of biodiversity for its own sake, in addition to ecosystem services, is completely consistent with an ecosystem services approach;
4. Relate BES change to livelihoods and other wellbeing metrics; and

Furthermore, combining tools and methods can yield significant benefits such as (Dunford et al. 2018):

Individual tools are unlikely to address all the needs of a given context, but a range of approaches can be used to assess different aspects of ES, such as different types of green infrastructure, different groups of services, different geographic scales or time-scales, and different types of value (e.g. biophysical, socio-cultural and monetary).

This chapter has made the case for urgent action to protect our ecological systems (biodiversity) from catastrophic decline. ESS can help shine a light on what we will lose if we fail protect valuable (and indeed invaluable) ecosystems and earth's flora and fauna in general. We have documented the development of ESS, from its foundations in environmental and ecological economics to its evolution into FEGS and CICES classifications and valuation frameworks. In doing so we have highlighted the challenges of valuation (including issues of scale, biodiversity awareness or literacy, and polycentric governance), and provided lessons and recommendations going forward for the successful application of ESS.

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Approaches for Estimating the Supply of Ecosystem Services: Concepts for Ecosystem-Based Management in Coastal and Marine Environments

Fiona E. Culhane, Leonie A. Robinson, and Ana I. Lillebø

Abstract Ecosystem services have emerged as a critical concept for Ecosystem-Based Management (EBM) in aquatic environments, namely in coastal and marine environments. However, despite conceptual advances over the last two decades, major challenges remain in the operationalisation of ecosystem service concepts and practical application. This chapter describes a selection of EBM assessment approaches applied to coastal and marine environments, where the ecosystem services approach is key. These approaches range from qualitative to quantitative, all being transdisciplinary. In the first, (ODEMM project, supported by linkage frameworks) trade-offs in EBM management options can be considered, in terms of their potential to reduce ecological risk, maintain sustainable supply of ecosystem services, and the governance complexity associated with implementing them. In the second, (AQUACROSS project, combining causality links relations and spatial multicriteria analysis) trade-offs are supported by maps with governance boundaries, spatially explicit valuation of ecosystem services and ecological risk. In the third, (MCES project, supported by the GIS-based modelling tool InVest) trade-offs are supported by a developed spatially explicit proxy for the habitats' vulnerability to deliver ecosystem services. Finally, we describe a policy-based operational assessment tool that allows users to assess the current and future capacity of regional seas to supply ecosystem services, based on their current and future ecosystem status reporting. We go on to describe some lessons learnt from our experience in applying these approaches.

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Lessons Learned

- For marine ecosystems, data availability is often a barrier to operationalising the ecosystem service concept. Ideally, spatial data would be available and, in many cases, it is becoming so. However, assessments are needed now. We show that there is existing information that can be applied to ecosystem service assessments for marine environments, and this should not be a barrier to carrying out assessments now.
- Consider all the ways that nature contributes to human wellbeing. There are criticisms of some approaches to ecosystem service classifications and assessments, because the services cannot be clearly linked to market values and economic assessments. This narrows our perspective on the breadth of ecosystem services. We show that by carrying out a supply side assessment, all the ways that nature contributes to wellbeing can be considered. This can be seen as complementary to demand side assessments and as an end-point in itself. Economic valuation does not need to be the only end point.

Needs to Advance EBM

- Scientists and policy makers need to be open to draw on different approaches including expert judgement and stakeholder knowledge, policy information, as well as, detailed habitat mapping or spatially explicit modelling techniques. These can then be used, together or in isolation, to show how ecosystem integrity can affect human well-being, fulfilling the critical need for balanced (economic, ecological, social) management actions to be taken.

1 Introduction

The concept of nature benefits for humans, is not new, however, it was first in 1983, almost forty years ago, that Ehrlich and Mooney used the term '*Ecosystem Services*' in an International Scientific Indexing (ISI) journal (see Flood et al. (2020) for a discussion of earlier development of the concept). Under the title '*Extinction, Substitution, and Ecosystem Services*' and using different biomes, authors showcased the need for a "*conservative approach to the maintenance of services through minimizing anthropogenic extinctions*" (Ehrlich and Mooney 1983). Regarding the marine biome, the biggest biome in the world, and the accompanying ecosystems services, authors highlighted the impact of fisheries over fish stocks and the role of economy as a driver for extinction and for substitution of target species. Their recommendations followed the need for a "*careful preservation of ecosystems and thus of the populations and species that function within them*". Since then the concept of '*Ecosystem Services*' has evolved in order to become effectively operationalised, but major challenges still remain, namely, how to decide who will win and who will lose, as trade-offs are inherent to the decision making process. This is of paramount importance in the context of global socio-ecological challenges and sustainable development strategies for coastal and marine environments. Such challenges include human indirect drivers like sea level rise, extreme weather events (e.g. floods and storm surges) and invasive species, and human direct drivers (e.g. economic activities). Global strategies tackling these challenges include those based around Blue Growth, Biodiversity and Climate Change. Ecosystem-Based Management (EBM) acknowledges that human well-being and ecological status are linked and integrates multiple drivers and pressures into adaptive management plans (UNEP 2011). In this context, ecosystem services have emerged as a critical concept to operationalise EBM.

1.1 Ecosystem Services Concept

Ecosystem services have been defined in different ways over the years, with the term services often being used interchangeably to mean "*the benefits people obtain from ecosystems*" (MA 2005), to the ecosystem structures, processes or functions that generate the services. More recently, the ecosystem service cascade model has been widely adopted to clearly delineate where a service sits in relation to what generates it (in the ecosystem), and what benefits people get from it (in the socio-economic system) (Potschin and Haines-Young 2011). Following the rationale of the ecosystem services 'cascade' model and Culhane et al. (2019a), we define ecosystem services here as:

> *Ecosystem services represent the flow of ecosystem capital that is realised because of a human active or passive demand (modified from EEA (2015)). They are thus the final outputs from ecosystems that are directly consumed, used (actively or passively) or enjoyed by people*. (Fisher et al. 2009; Haines-Young and Potschin 2013; Maes et al. 2013)

Examples of coastal and marine ecosystem services include nutrition from fish and shellfish, flood and coastal protection from saltmarsh habitats, and artistic inspiration from seascapes and marine animals (Culhane et al. 2019a, and see Fig. 1 for further examples). In order to recognise all the services that ecosystems and marine environments supply, typologies of services have been developed. These categorise services and make their assessment operational. Early international initiatives to develop typologies classified services under four broad categories: provisioning (such as food from fish); regulation and maintenance (such as waste regulation); supporting (such as primary production); and cultural services (such as marine species to observe or to research) (MA 2005; TEEB 2010). The concept of Final Goods and Ecosystem Services (FEGS) also developed (see DeWitt et al. (2020) for a summary of this). These are a subset of ecosystem services, generally not including the supporting services, that can be directly linked to a beneficiary, thus avoiding double-counting when assigning a monetary or market value. Following these, ecosystem service typologies have been further developed and/or adapted for the marine environment (e.g. Böhnke-Henrichs et al. 2013), see Fig. 1 for some examples from this typology, which retains supporting services). CICES, the Common International Classification of Ecosystem Services (Haines-Young and Potschin

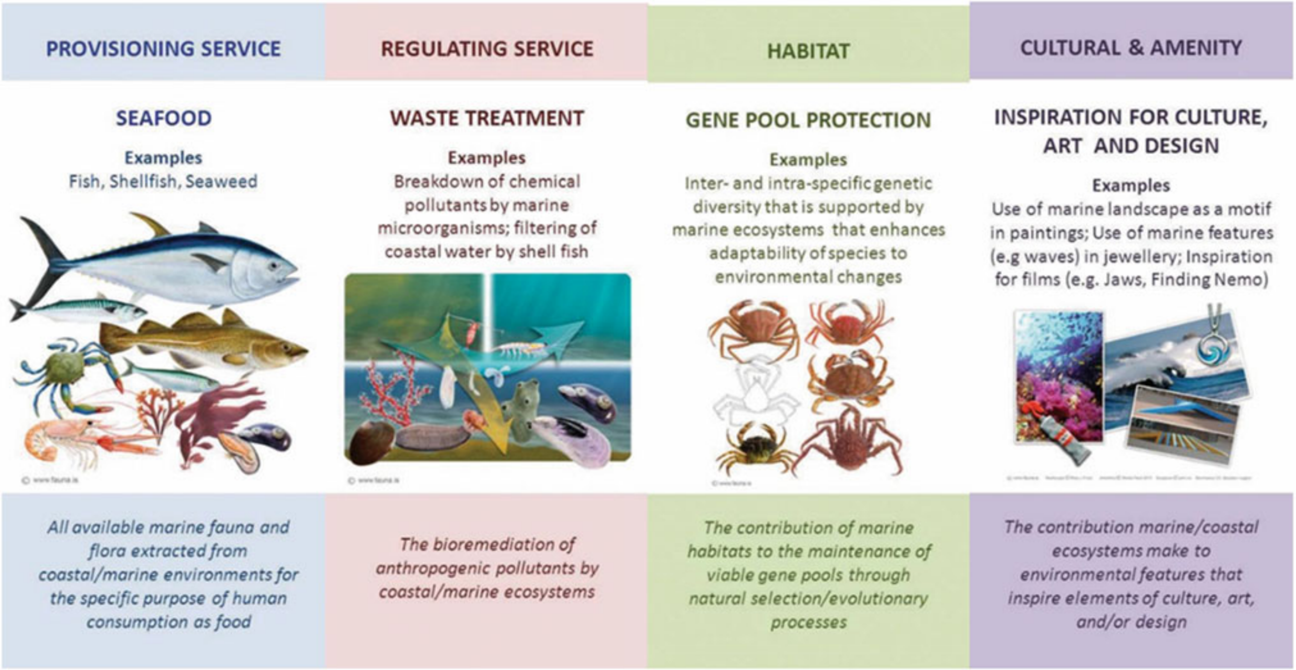

Fig. 1 Examples of marine ecosystem services from the typology of Böhnke-Henrichs et al. 2013, displayed as cards developed in the ODEMM project (Robinson et al. 2014). Habitat services link to those in the TEEB classification and are included under the Supporting Services of other classifications, such as the Millennium Ecosystem Assessment. The ODEMM ecosystem services cards are available to download for use from https://www.odemm.com/content/cost-and-benefits-analyses

2013) is the reference typology for the European Union's assessments linked to the Biodiversity Strategy (EC 2011). It is a hierarchical typology that recognises three categories of ecosystem services: provisioning, regulation and maintenance, and cultural services at the broadest level and further subdivides these categories into more specific services at lower levels (see Flood et al. (2020) for further discussion of ecosystem service classification systems). CICES includes only 'final' services, though one criticism of the typology is that some of the included services are, in fact, supporting (or intermediate). CICES was developed primarily for terrestrial environments but has been adapted for the marine environment (Culhane et al. 2019a), and should be applicable across biomes allowing a Europe wide assessment.

1.2 Policy Background

Around twenty years ago, Costanza and colleagues estimated the value of the world's Ecosystem Services and natural capital, showing that coastal (tidal marsh and mangroves) and marine (open ocean, shelf, estuaries, seagrasses) services accounted for circa 68% of the global value for ecosystem services (Costanza et al. 1997). Just after, in the year 2000, the Millennium Ecosystem Assessment was called for by the United Nations Secretary-General Kofi Annan, as part of the '*The Role of the United Nations in the 21st Century*'. Regarding ecosystems, the objective was "*to assess the consequences of ecosystem change for human well-being and to establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being*" (MA 2005).

Then ten years ago, the global initiative '*The Economics of Ecosystems and Biodiversity (TEEB)*' focused on '*making nature's values visible*' in order to mainstream the values of biodiversity and Ecosystem Services into decision-making at all levels (TEEB 2010). Within TEEB special attention is given to 'blue growth' and human dependence on healthy ocean ecosystems and on coastal and marine biodiversity. About the same time, the European Union (EU) Biodiversity Strategy for 2020 aimed to halt the loss of biodiversity and Ecosystem Services, reflecting the commitments taken in 2010 within the International Convention on Biological Diversity. As part of the strategy, working groups for mapping and assessment of ecosystem services, including a marine pilot, provided supporting guidance for EU member states (Maes et al. 2016). Just after, in 2012, an independent intergovernmental body of the United Nations '*The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)*' was established "*to strengthen knowledge foundations for better policy through science, for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development*". This platform stands for nature and biodiversity, in the same way that the '*Intergovernmental Panel on Climate Change (IPCC)*' provides the latest science-based assessments related to climate change, including possible response options.

In the last five years, different but complementary initiatives have taken place acknowledging the need for global actions towards unprecedented changes in nature, climate and human population growth. In 2015, the United Nations, together with governments, businesses and civil society, agreed on the Sustainable Development Agenda for 2030 that integrates seventeen interlinked Sustainable Development Goals (SDG's). Although, SDGs should not be addressed individually, SDG 14 is devoted to '*Conserve and sustainably use the oceans, seas and marine resources for sustainable development*'. In the same year the 'Paris Agreement on climate change' entered into force, being essential for the achievement of the SDGs. The former initiatives, TEEB, IPBES, as well as the EU Strategy for biodiversity beyond 2020 are also framed in the scope of the SDGs.

In this context, Ecosystem-Based Management (EBM), a holistic approach that aims to balance the multiple interrelated dimensions of ecological integrity and human well-being, appears a useful framework to operationalise the concept of ecosystem services (Gómez et al. 2016, 2017). Likewise, one can argue that achieving EBM might be more attainable where the ecosystem services approach is included, since it enables stakeholders and decision makers to see a tangible way in which the integrity of ecosystems directly (actively or passively) affects the well-being of humans. To this end this chapter aims at showcasing selected EBM assessment approaches applied to coastal and marine environments that incorporate understanding and assessment of ecosystem services and to draw lessons learnt from our experience.

2 Operationalising Ecosystem Services in EBM

2.1 Ecosystem Services and Trade-Offs in EBM Management Options

Different decision support tools can support trade-off options regarding the provisioning of ecosystem services. These can be supported by linkage frameworks (e.g., see Robinson and Culhane 2020), by causality links relations (e.g., AquaLinksTool, Nogueira 2018), by spatially-explicit GIS-based modelling tools (e.g., Willaert et al. 2019) or by a combination of the above mentioned decision support tools (e.g., see Lillebø et al. 2020). Three selected examples are now presented.

2.1.1 ODEMM Project: https://www.odemm.com

In the ODEMM project a typology of marine ecosystem services was developed (Böhnke-Henrichs et al. 2013) and an assessment undertaken whereby stakeholders compared management options based on three major criteria: ecological risk, ecosystem service supply and governance complexity (Robinson et al. 2014: Chap. 7).

As illustrated in Fig. 2, management options could be applied to reduce impacts through a number of different pathways, and the reduction in risk to ecological components was considered in terms of any resultant change in the supply of ecosystem services (a framework that aligns with the DPSIR, and later DAPSI(W)R(M) concepts, see Elliott and O'Higgins (2020)). This was considered against the complexity of governance required to implement the compared management options, and the effect of ecological risk reduction on potential achievement of ecological goals as set out by the Marine Strategy Framework Directive.

It was not always the case that management options delivered benefits in terms of all three criteria considered in the same way (Fig. 3). In terms of the examples explored, stakeholders generally found that the management option that delivered the best reduction in risk to achievement of the MSFD good environmental status objectives, was also least complex in terms of governance required to instigate it, but was not the most promising in terms of the benefits to sustainable supply of services. This helps to illustrate the trade-offs experienced in EBM. Participants involved in the ODEMM trade off exercises found the approach to be very useful in terms of helping to visualise "how the sea works", providing a "practical approach to link management options with potential changes in the provision of ecosystem services" (ODEMM Deliverable 19).

2.1.2 AQUACROSS Project: https://aquacross.eu

The EBM approach (see Piet et al. 2020) was applied to the Vouga river coastal watershed, along a continuum of freshwater to marine habitats under the classification of the Natura 2000 network (Lillebø et al. 2018). This case study also aimed to showcase causality links in a linkage chain relating activities, pressures and habitats/highly mobile biotic groups and ecosystem services (Teixeira et al. 2018, 2019; Culhane et al. 2019b) and to assess the vulnerability of ecosystem components regarding the provisioning of ecosystem services (Lillebø et al. 2018).

In this social-ecological system and as part of the EBM framework, stakeholders were invited to value ecosystem services through a spatial multi-criteria analysis that took place at two different spatial scales (Lillebø et al. 2019; Martínez-López et al. 2019). From the resulting prioritization maps representing stakeholder's perceptions and beliefs regarding ecosystem services, valuations were generated and supported the discussion of the areas for potential interventions (ecosystems restoration) and associated trade-offs. As part of the EBM trans-disciplinary approach, stakeholders were also invited to express their concerns regarding the foreseen management options. Simultaneously, causality links and risk assessment were undertaken though a tool that establishes a linkage chain relating activities, pressures and habitats/highly mobile biotic groups and ecosystem services, to assess the vulnerability of ecosystem components regarding the provisioning of ecosystem services (AquaLinksTool, Nogueira (2018)) as shown in Fig. 4. The full linkage matrices dataset is freely available for download https://zenodo.org/record/1101159#.XbKsfS3MyqA.

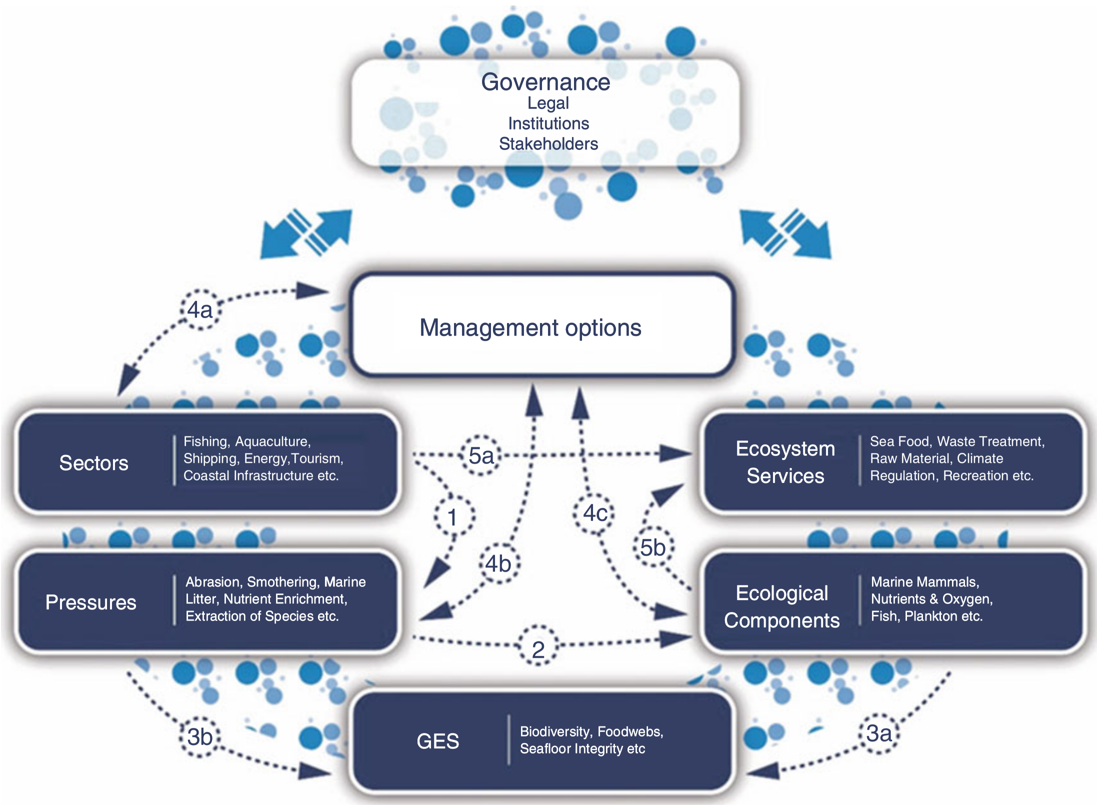

Fig. 2 The ODEMM Cost and Benefits analyses consider how the appraisal of management options can take into consideration both associated costs and benefits (where benefits are described by the supply of ecosystem services). Here, the supply of ecosystem services can be altered when management options are instigated (4 a–c), either though controls on sectors that supply key ecosystem services (5a) (e.g. instigating catch control on fisheries can alter supply of Seafood directly), or though the alteration in state of ecological components that contribute to the supply of ESs (5b). [State of ecological components may change due to management options acting directly on ecological components (4c) or acting on pressures (4b) and/or sectors (4a) that impact ecological components (2)]. Costs associated with management options may arise at both the level of the sector and/or at institutes involved in the governance of those management options. (From Robinson et al. 2014)

Fig. 3 A final outcome table following completion of all three exercises, where numbers represent the order in which the Management Options (MO A, B, C) work best, in terms of outcomes for the different criteria considered

The vulnerable habitats selected through the AquaLinksTool clearly matched stakeholders' concerns, as well as their ecosystem services prioritization maps (Lillebø et al. 2019; Martínez-López et al. 2019). The combined approach contributed to the effectiveness (hitting the environmental target), the efficiency (making the most for human wellbeing), and to equity and fairness (sharing the benefits) of the proposed EBM responses. Lillebø et al. (2020) detail the co-development process of an EBM plan foreseen to mitigate unintended impacts on biodiversity in Vouga estuary and to its end support the development of the Vouga estuary management plan.

2.1.3 MCES Project

In the MCES project, a vulnerability index of the potential of marine and coastal habitats to deliver ecosystem services was developed. This is an example of an approach to implement ecosystem service assessments in EBM using spatially explicit modelling tools, i.e., by generating maps, GIS-based models enable decision makers to assess quantified trade-offs associated with alternative management options and to identify areas where these can take place. Relevant examples of open source models are ARIES—Artificial Intelligence for Ecosystem Services, already applied for machine learning for ecosystem services (Willcock et al. 2018); MARXAN with Zones enabling to '*develop multiple-use zoning plans for natural resource management*' (Watts et al. 2009; Jumin et al. 2018) and InVest—Integrated Valuation of Ecosystem Services and Trade-offs, already used for calculating vulnerability of marine habitats to deliver ecosystem services (Willaert et al. 2019; Cabral et al. 2015). For detailed consideration of EBM modelling tools see Fulford et al. (2020) and Lewis et al. (2020).

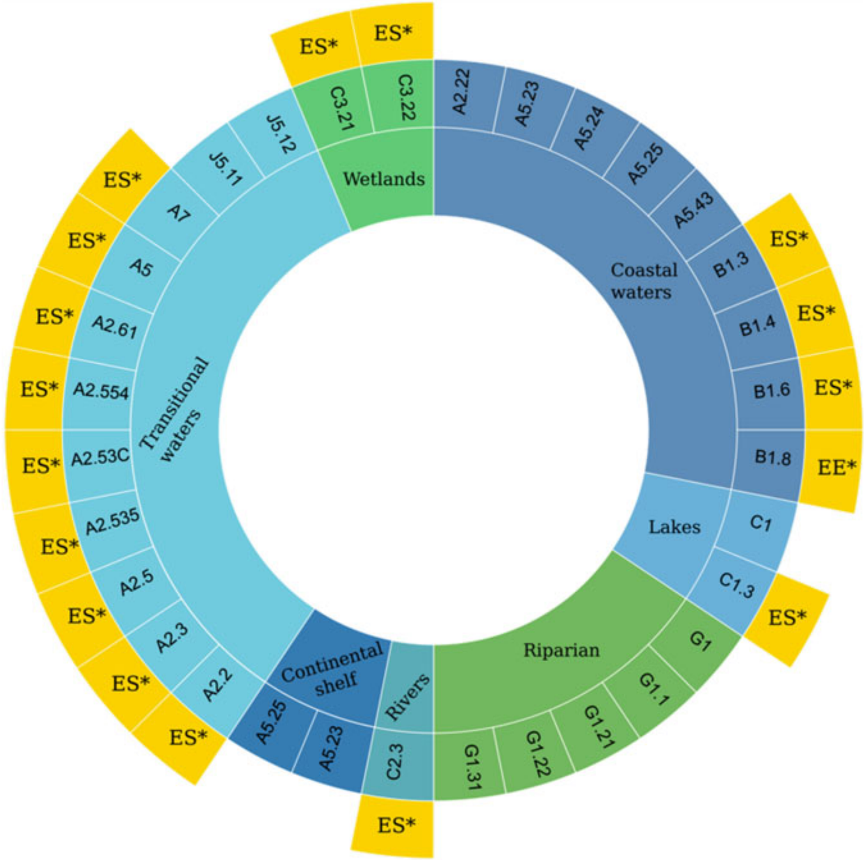

Fig. 4 Vulnerability (ES∗) of the Vouga river coastal watershed habitats under classification of Natura 2000 network to provide ecosystem services defined with AquaLinksTool. (Image source: Adapted from Lillebø et al. (2018), plotted with Mauri et al. (2017)). EUNIS habitats codes: A5.23 Infralittoral fine sand; A5.25 Circalittoral fine sand; A2.22 Barren or amphipod-dominated mobile sand shores; A5.23 Infralittoral fine sand; A5.24 Infralittoral muddy sand; A5.25 Circalittoral fine sand; A5.43 Infralittoral mixed sediments; B1.3 Shifting coastal dunes; B1.4 Coastal stable dune grassland (grey dunes); B1.6 Coastal dune scrub; B1.8 Moist and wet dune slacks; A2.2 Littoral sand and muddy sand; A2.3 Littoral mud; A2.5 Coastal saltmarshes and saline reedbeds; A2.535 *Juncus maritimus* mid-upper saltmarshes; A2.53C Marine saline beds of Phragmites australis; A2.554 Flat-leaved Spartina swards; A2.61 Seagrass beds on littoral sediments; A5 Sublittoral sediment; A7 Pelagic water column; J5.11 Saline and brackish industrial lagoons and canals; J5.12 Saltworks; C1 Surface standing waters; C1.3 Permanent eutrophic lakes ponds and pools; C3.21 Common reed (Phragmites) beds; C3.22 Common clubrush (Scirpus) beds; C2.3 Permanent non-tidal smooth flowing watercourses; G1 Broadleaved deciduous woodland; G1.1 Riparian and gallery woodland (Alnus Betula Populus or Salix); G1.21 Riverine Fraxinus—Alnus woodland; G1.22 Mixed Quercus—Ulmus—Fraxinus woodland of great rivers; G1.31 Mediterranean riparian Populus forests)

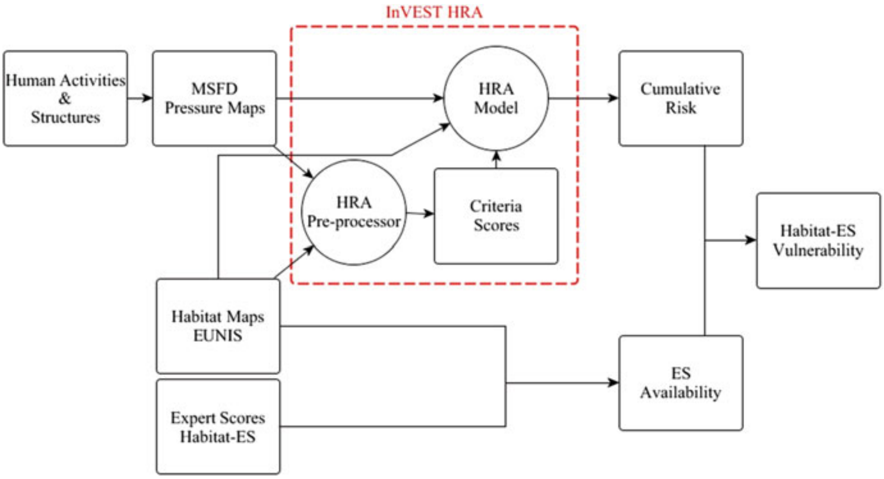

Fig. 5 Schematic representation of the workflow to assess the vulnerability of marine and coastal habitats' potential to deliver ecosystem services. (Image source: Willaert et al. 2019. Note: MSFD—Marine Strategy Framework Directive; HRA—InVEST Habitat Risk Assessment tool; ES—Ecosystem Services; EUNIS—habitat classification system)

The MCES project example showcases a vulnerability-based approach to capture Ecosystem Services in EBM. To this end, the considered approach combined the InVEST habitat risk assessment tool with the identified ecosystem services to create a proxy for the habitats' vulnerability to deliver ecosystem services in support of coastal and marine EBM. Figure 5 illustrates the framework combining the supply and demand for coastal and marine ecosystem services followed by Willaert et al. (2019). The case study was the western-Atlantic coast of Portugal that included the Nazaré Canyon (> 3000 m depth within the study region), Óbidos Lagoon (transitional waters), São Martinho do Porto bay (marine inlet), and Berlengas Archipelago (UNESCO world biosphere reserve).

As concluded by Willaert et al. (2019) "*The mapping of benthic habitats has opened new avenues, contributing to improve not only marine spatial plans, but also the EBM approach by facilitating the combination of spatial explicit GIS tools with supply and demand of marine ES, human activities and their related impacts, as well as with other natural impacts (e.g. climate change) to forecast scenarios (including marine ES trade-offs) and to open the floor to discussion (namely in stakeholders participatory processes) and to sustainable decision making processes in a 'Blue Growth' context by maximizing the net benefits provided by marine environments over time*".

The diversity of habitats in the selected area, the proposed approach to capture vulnerability, and the generated maps (EBM base line and EBM management options prospective scenarios) showcase maritime spatial planning and 'Blue Growth' in the context of the SDGs for 2030.

3 A Policy-Based Regional Seas Assessment of the Capacity to Supply Ecosystem Services

In this section we describe an assessment approach (MECSA: Marine ecosystem capacity for service supply assessment) developed with the European Environment Agency that utilises policy-reported ecosystem status information (from the Marine Strategy Framework Directive and other relevant reporting) to assess current and future capacity to supply ecosystem services (Culhane et al. 2020; Culhane et al 2019a). The key steps and the types of output that can be generated are summarised.

3.1 Using Ecosystem State Information in Ecosystem Service Assessments

The state of the ecosystem can tell us something about the capacity of the ecosystem to supply services, and this is the underlying assumption of this assessment approach. Across multiple services it can be assumed that good ecosystem state underpins good capacity for service supply, since the ecosystem service supply relies on the integrity of the ecosystem (Burkhard et al. 2012; Culhane et al. 2019b). However, at an individual service level, this is not necessarily the case, and we cannot assume that good ecosystem state will necessarily equate to good ecosystem services supply. Thus, the second underlying assumption of this assessment is that we understand something about the ecosystem state-service supply relationship, and we can use this to interpret state information in relation to what it means for the supply of services (Box 1).

Box 1 Understanding the State-Service Relationship

Different services will have different types of relationship with the state of the parts of the ecosystem that supplies them.

Examples:

- Flood protection capacity is better in the presence of a greater area of saltmarsh habitat (King and Lester 1995).
- Heavily-grazed seagrass beds, even though in worse state than ungrazed areas, still support up to three times more coastal protection through their root system that unvegetated areas (Christianen et al. 2013).
- Macroalgae habitat quality is more important than habitat area for service supply e.g. for maintaining fish habitat (van Lier et al. 2018).

It is important to understand this relationship in order to use state information to truly represent the capacity of the ecosystem to supply services. Different services rely on different aspects of the ecosystem. These need to be identified.

3.2 Steps to Implementing the Method

This approach entails three main steps, all set in the context of understanding the state-service relationship (Box 1).

- Firstly, the parts of the ecosystem that supply the service being assessed are identified. We described an approach to identify service providing units (SPUs) (see linkage framework chapter and Culhane et al. 2018). The relative contribution of these SPUs can be determined, and only the critical ones of these taken forward for a full assessment.
- The second step involves determining the relative contribution of SPUs to the service supply and identifying which are critical.
- In the third step, the specific state service relationship for the critical SPUs is described and the most appropriate indicators of these identified. State information is then used to fulfil these indicators, though in practice, this is often determined by what is available rather than what is most appropriate. This is interpreted in relation to ecosystem service supply capacity, using understanding of the state-service relationship.

3.3 Example Case Study: North East Atlantic

The three steps of the method are illustrated through an example of a cultural ecosystem service—recreation and leisure from whale watching in the North East Atlantic.

As well as whale watching occurring from the shore and other non-commercial routes, it is a growing enterprise in many regions (e.g. Elejabeitia et al. 2012; IUCN-ACCOBAMS 2016), often representing important economic benefits to rural communities (Ryan et al. 2018). In the North East Atlantic, commercial whale watching tours operate around the shores of Ireland, Scotland, England, Portugal and Northern Spain. In the first step of identifying the SPUs, we define these for this service as the whales that are watched by people, and the habitats that support those whale populations. Since whales are highly mobile and may make use of many habitats, we focus here on the whales themselves, as their populations are likely to reflect the condition of their habitats. We identified the relevant SPUs by checking whale tour operator websites and the species advertised as being seen on tours (Table 1).

To identify the critical SPUs for step two, we consider whether some species are more important than others. We find that, while common species are more reliably seen, rare species may constitute a special and equally important experience. Thus, in this case, we take all species forward as being critical. We define the state-service relationship as being that a greater whale population will result in a greater likelihood of seeing whales. Thus, in this case, good ecosystem state of whale populations

Table 1 Summary of results for the current state and direction of change of the metrics relevant to assess the 'whale watching' service in the North East Atlantic

Indicator	Classification	% Whale species assigned	Service supply capacity
State	Pass	56	Good
	Fail	17	
	Insufficient information	28	
Direction of change	Increasing	22	Unknown
	Stable	0	
	Decreasing	0	
	Insufficient information	78	

would correspond with a good capacity for the supply of this service. We can then look to policy assessments to find information on the status of each of these whale populations in the North East Atlantic.

To interpret this state information in relationship to the service supply capacity, we go back to our state-service relationship, where we had determined that more whales would lead to greater likelihood of spotting whales, and that each whale species is as important as any other. Therefore, we assess what is happening overall to whale populations in the North East Atlantic. From the policy status assessments, we see that most (56%) are in a good state, passing their policy objectives, but there is insufficient information about the direction of change of their populations (Fig. 6). Thus, the overall capacity for the ecosystem to supply this service is good but we do not know how this is changing.

3.4 Conclusions

This assessment approach uses available ecosystem state information to assess the capacity of the ecosystem to supply services. In doing so, a number of assumptions need to be made that relate to understanding the state-service relationship, and to the suitability of the state information available. In this example, we could assess the current capacity of the service but we could not assess the direction of change. While policy information allows a source of status assessments that can be used for this purpose, there are many unknowns and uncertainties. Nevertheless, this approach allows for assessments at broad regional scales, using existing information where it exists and does not rely on the availability of spatial or other data that may not be available.

It is also important to note that, although this service was found to have a good capacity overall, not every species is meeting its ecological objectives. There is still a

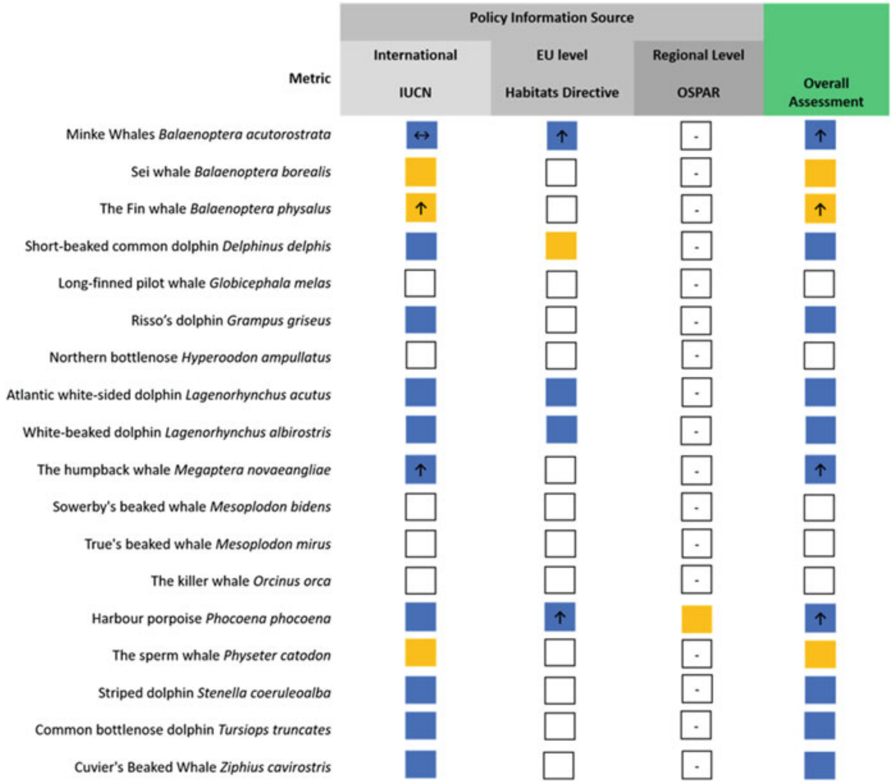

Fig. 6 List of commonly, occasionally or rarely spotted cetacean species advertised by whale watch tour operators (nine tour operators consulted from Ireland, Scotland, England, Portugal and Northern Spain) in the North East Atlantic region. The state of each whale species metric reported under each policy for the North East Atlantic is given. Legend: blue box = policy objectives achieved (here meaning that the whale population is in a good state); yellow box = fail to meet policy objectives; white box = unknown state/no assessment could be made; upward/downward arrow = Trend towards/away from achieving policy objectives; arrow on both sides = no trend; no arrow = unknown trend, 'Minus' symbol means the metric is not relevant for a particular policy)

need for conservation measures to protect species, and an ecosystem service assessment does not replace a biodiversity assessment.

4 Lessons Learnt and Next Steps

We draw a number of conclusions and lessons from our experience in trying to make the ecosystem services concept operational in Ecosystem-Based Management (adapted from Culhane et al 2019a).

4.1 Lessons Learnt

1. Make use of what is currently available
   Approaches supported by spatially explicit data (AQUACROSS, MCES) described above, can make use of detailed spatial data on habitats, ecosystem service supply and demand. However, for marine environments, spatial data is often scarce, in particular at the large spatial scales that may be relevant for policy assessments. Alternatively, the other approaches that we demonstrated (MECSA, ODEMM) show that it is still possible to carry out assessments of marine ecosystem services using what is available. Although these assessments may not be ideal in terms of the information underpinning them, they can form a baseline and indicate where there are potential problems in sustainability, while future assessments can make use of better information availability.
2. Small scale studies are needed to complement high level regional sea studies
   We demonstrated assessments that are high level, at broad ecological (regional sea) scales (ODEMM, MECSA), as well as more local (AQUACROSS, MCES). However, both these types of studies are underpinned by knowledge found from studies on specific habitats, species and ecosystem services, and how people use them. We continue to need these studies to improve understanding and confidence in ecosystem service assessments.
3. Ecosystem service assessments are not equivalent to ecosystem assessments
   Good ecosystem state does not always equal good capacity to supply services. Good service capacity may be satisfied by a broad habitat or group of taxa supplying it, for example in the whale watching example given above. But within this broad taxa group, several species may be failing their ecological assessments and be vulnerable. Ecosystem service assessments do not replace assessments of biodiversity and ecosystem condition—it is essential that these are seen as complementary.
4. Consider all of the contributions of nature to human wellbeing
   There are a number of ecosystem service typologies that can be used today. Although each have their critics, typologies like CICES (v5.1), TEEB and that used in the IPBES (International Panel on biodiversity and Ecosystem Services, (IPBES 2019), which include services that cannot be given a monetary or market value (see discussion on FEGS, DeWitt et al. (2020)), have a broad and encompassing set of ecosystem services that recognises all of the ways that nature contributes to our wellbeing. It is important to recognise services even where they cannot be given a market value, because ultimately, they are contributing to the long-term sustainability of society. It is possible to measure changes to these contributions by using a supply-side approach demonstrated here (e.g. MECSA, MCES), rather than a demand side approach.
5. Establish who are the beneficiaries of ecosystem services
   In order to recognise all the contributions of nature to human wellbeing (see 4 above) and to understand how the marine ecosystem contributes to our wellbeing, we need to explicitly identify all of the beneficiaries of coastal and

marine ecosystem services and how these benefits are perceived. This would help to establish how services are 'final' (sensu FEGS, see DeWitt et al. (2020)) and to establish how best we might measure specific contributions to different parts of society. This may not always be through measuring an economic value, and different, complementary approaches may need to be used side by side to fully capture how nature contributes to human wellbeing.

6. Need for adaptive response to ecosystem service assessment
   Ecosystem services are embedded in the current social and ecological context. As this changes, ecosystem service assessments also need to change to keep up. These assessments can inform management about the need to change and adapt in response to changing ecosystem service supply or demand. Furthermore, as mentioned above, a critical point in the decision-making process is to decide who will win and who will lose over time and space. Therefore, adaptive responses are crucial.
7. Link human activities and pressures to ecosystem service supply
   Increasing activities and pressures in the marine environment, alongside global climate change, requires urgent assessment of how these activities and pressures are affecting ecosystem service supply and what this will mean for long-term sustainability. The ODEMM approach described above demonstrates how trade-offs can be explored for different management options, whilst the MCES vulnerability index allows prioritisation of the most vulnerable or most important habitats needed for service supply. These can help to consider where human activities will impact service supply, and where management can act to be most effective, efficient and equitable.

4.2 Next Steps

The demand side of ecosystem services is socio-economically driven, while the supply side is dependent on ecosystems capacity to provide the required ecosystem services underpinning maritime activities. Europe's Biodiversity Strategy, aims to conserve and restore the supply side of ecosystem services, by halting biodiversity loss and deterioration of services by 2020 (EC 2011). On the other hand, Climate change strategies in the EU are aiming to significantly cut greenhouse gas emissions reaching net zero by 2050 (EC 2018). This will be achieved, at least partly, by increasing the share of renewable energy, including offshore energy. The Blue Growth strategy brings the supply and demand sides together for marine and coastal environments. It aims at supporting an effective implementation of maritime, marine and coastal-related policies, and at "*realising the potential of our seas and oceans for jobs and growth*" following the principles of conservation and of sustainable development (EC 2012). The strategy also foresees approaches to restore marine and coastal habitats, biodiversity and ecosystem services, being in line with the United Nations SDGs for 2030.

Therefore, the operationalization of any of these strategies requires balanced trade-offs between economic, social and environmental aspects supported by coastal and marine ecosystem services (Lillebø et al. 2017). To this end, Ecosystem-Based Management incorporating adaptive management is likely to have a critical role (e.g. Lillebø et al. 2020).

Balancing the demands we put on our ecosystems, with what they can sustainably supply, is the challenge of EBM. Ecosystem service assessments are one tool that can help us to do this, by allowing recognition of all the ways that the ecosystem contributes to human wellbeing and all the ways that human activities can impact the ecosystem and the supply of services. While the original ethos of ecosystem services was based in conservation (Ehrlich and Mooney 1983), it is clear that ecosystem service assessments alone do not replace nature conservation, in particular when individual services of interest are assessed, as opposed to multiple ecosystem services (MA 2005; Culhane et al. 2018; Teixeira et al. 2019).

Using the ecosystem service approach to benefit EBM is still challenging. The challenges lie in our knowledge and understanding of how the ecosystem works, in our understanding of human behaviour and the demands of society and in the resources that we have available, such as data or the means to collect data. Nevertheless, we have presented a number of ways showing that ecosystem service assessments can be integrated into EBM decisions. These assessments draw on different approaches in data-limited situations, including expert judgement and stakeholder knowledge, policy information, as well as, detailed habitat mapping or spatially explicit modelling techniques. These are all ways that can be used to achieve a key tenet of EBM, namely that it allows decision makers and stakeholders to see how ecosystem integrity can affect human well-being, thus allowing more balanced actions to be taken.

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The Final Ecosystem Goods & Services (FEGS) Approach: A Beneficiary-Centric Method to Support Ecosystem-Based Management

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Abstract Ecosystem-Based Management (EBM) and other social-ecological environmental management frameworks recognize that most environmental problems are ultimately social problems, requiring the reconciliation of human needs with the limits of ecosystem productivity and resilience. Using a social-ecological perspective in management practice can be greatly facilitated by identifying the attributes of ecosystems that are directly used, enjoyed, or appreciated by people connected to the environmental issue at hand. These are the *final ecosystem goods and services* (FEGS), which are specific to ecosystem types and how people use or appreciate ecological attributes of those ecosystems. This article: (1) reviews the conceptual basis of a FEGS approach for linking people's well-being to ecosystems; (2) describes how FEGS are identified, organized, and measured using classification systems, and metrics and indicators; and (3) presents examples of how the FEGS approach can be integrated into EBM and other decision-making frameworks.

Lessons Learned

- FEGS are the subset of ecosystem services that are directly used, enjoyed, or appreciated by people. Individual FEGS are identified as the biophysical attributes found within a given ecosystem that are used, enjoyed, or appreciated for a specific purpose.
- FEGS facilitate identifying, quantifying, and assigning value to biophysical attributes of ecosystems that are of greatest relevance to people who care about or depend on those ecosystems.
- FEGS are useful for communicating with stakeholders and policy-makers about how people obtain specific benefits from specific biophysical attributes of an ecosystem.
- Tools have been developed to identify FEGS within all types of ecosystems found on earth, for working with stakeholders to prioritize which FEGS are of greatest concern within a given decision context, and to identify mathematical models useful for estimating FEGS production.

Needs to Advance EBM

- Greater awareness within the EBM community of practice, including developing case-study applications, of the usefulness of FEGS and the availability of tools useful for identifying, prioritizing, and quantifying them.
- A standardized list of metrics or indicators for each FEGS, based on the attributes of ecosystem types that each beneficiary class uses, enjoys, or appreciates. Site-specific metrics or indicators could then be developed from those generic attributes.
- Integration of the FEGS tools (e.g., NESCS Plus, FEGS Scoping Tool, Rapid Benefits Indicators, EcoService Models Library) to facilitate identification of priority FEGS, relevant metrics and indicators for FEGS endpoints and benefits, and models for estimating responses of those FEGS to environmental change or stressors.

1 Introduction

Humankind depends wholly on nature for its well-being. An array of ecosystem goods and services, often available at no apparent cost, sustains our health, economy, and society. Those aspects of nature that people benefit from include productive soil for farmers, clean and safe water for swimmers, and inspirational landscapes for artists. The key feature of the examples in this list are that they link a good or service that is provided by nature to a specific type of beneficiary, or user group. Those ecosystem products and processes that are directly used, enjoyed, or appreciated by people are identified as *Final Ecosystem Goods and Services* (FEGS) (Boyd and Banzhaf 2007; Boyd et al. 2016). Those FEGS are a subset of all ecosystem goods and services (e.g., MEA 2005; Haines-Young and Potschin 2018) distinguished as the *final* “endpoints” in nature’s production networks that people directly use (Fig. 1). The production of FEGS is dependent on “supporting” and “regulating” ecological functions; these intermediate processes (Potschin-Young et al. 2017) are critically important to human well-being, for without them, FEGS would not exist. This is the essence of the FEGS approach: making explicit the biophysical attributes of ecosystems from which specific *beneficiaries* obtain a specific benefit. Beneficiaries are “the interests of an individual (i.e., person, group, and/or firm) that drive active or passive consumption and/or appreciation of ecosystem services resulting in an impact (positive or negative) on their welfare” (Nahlik et al. 2012). For example, beneficiaries are recreational anglers who fish for wild fish for food or pleasure, industrial processors who use water for cooling or product manufacturing, or artists who use attributes of nature for inspiration to produce art (Landers and Nahlik 2013). Beginning from this beneficiary perspective, we identify and quantify the biophysical attributes of ecosystems that people use or appreciate in order to achieve a wide

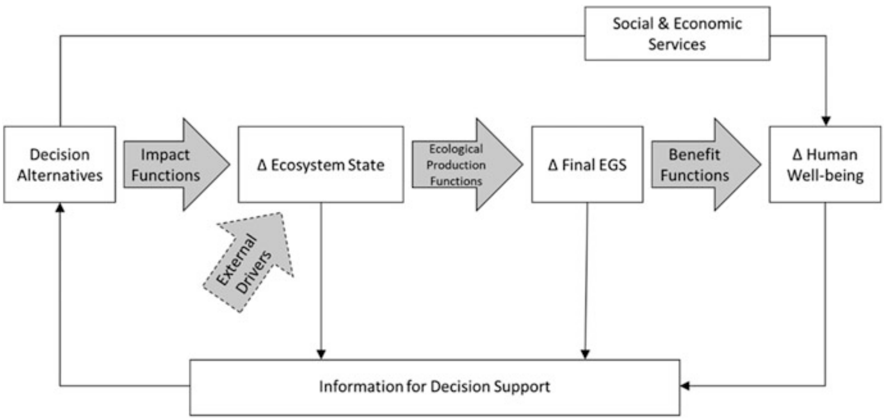

Fig. 1 Conceptual model for how changes to the state of an ecosystem (e.g., its biophysical attributes) and consequent changes in the production of final ecosystem goods and services (FEGS) influence environmental decision-making via impacts to human well-being. For more on this conceptual model, see Harwell and Molleda (2018)

range of benefits. In this chapter, we describe steps being taken to incorporate FEGS into decision-making and policies using: a classification system that identifies the beneficiaries and ecosystem attributes for each FEGS; methods to measure or estimate (through modeling) FEGS stocks, production and value; and decision-support tools to facilitate integrating FEGS into decision making.

Identifying, quantifying, and forecasting the stocks, production, and value of FEGS are important for demonstrating the relevance of natural systems to the public and policy makers, and they are fundamental to understanding the capacity of an area, or an ecosystem, to produce natural goods and services used by or useful to people. Characterization of FEGS is thus a valuable communication tool that provides support and added justification for ecosystem protection in decision making, including tradeoff analyses. At the same time, the identification of FEGS facilitates investigation into the social-ecological interactions between human actions, ecosystem condition, human well-being, a renewed reason to better understand how ecosystems function, and a basis for prioritizing which ecological attributes and processes should be incorporated into environmental management and policy.

The FEGS approach requires a subtle but important shift in perspectives in ecosystem services science, from identifying ecological goods and services that are important to human well-being (MEA 2005), to identifying what attributes of ecosystems people use, enjoy, or appreciate to fulfill a specific interest, and recognizing that the attributes that people use in pursuit of those interests differ across ecosystem types (Ringold et al. 2013). In other words, the FEGS approach is predicated on the fact that people use or depend on ecosystems in different ways, contingent on what they are doing or needing at a given moment and where those people are located (e.g., within what type of ecosystem). An important aspect of this beneficiary perspective is the inclusion of a wide range of human interests, including those related to culture and spirituality, as well as to health, economic, and overall well-being. By recognizing the importance of the human relationship to ecosystems, the beneficiary perspective addresses the concern that traditional resource management emphasizes goods and services (i.e., using metrics such as fisheries yield or recreational days) without explicitly considering the social context of a problem and the depth of the relationship between beneficiaries and the environment (Grumbine 1994).

In this chapter, we: (1) review the concept of the FEGS approach to human-nature interactions; (2) describe how FEGS are identified, organized, and measured using classification systems, and metrics and indicators; and (3) present examples of how the FEGS approach can be integrated into decision making, specifically to support Ecosystem-Based Management (EBM).

2 The FEGS Approach

The FEGS approach (Fig. 2) is motivated by the idea that identifying the biophysical attributes of ecosystems that are relevant to people will facilitate holistic benefit and economic assessment (Boyd et al. 2015). A key challenge for scientists is to identify

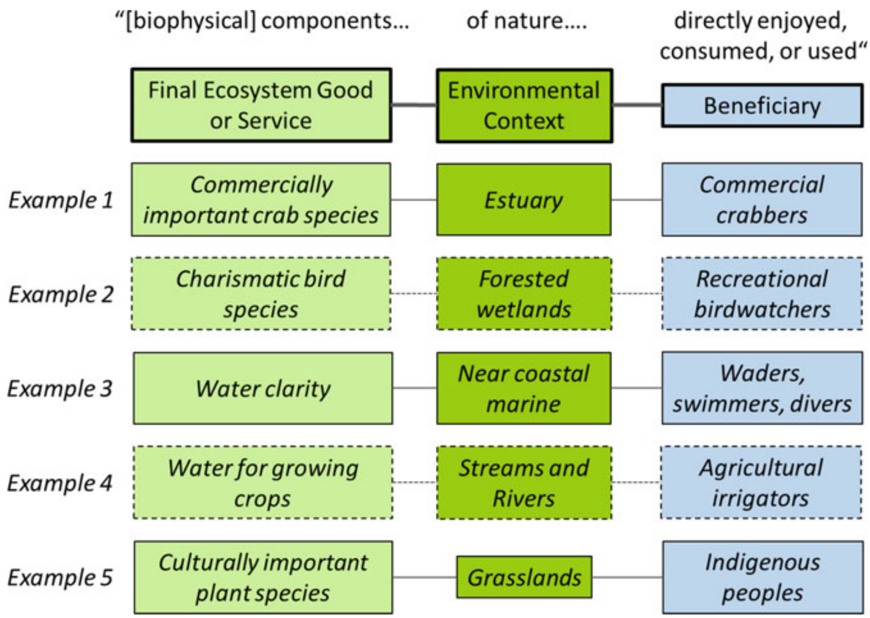

Fig. 2 Illustration and examples of the three elements needed to define FEGS. Note that many, if not most, beneficiaries depend directly on multiple biophysical attributes of multiple ecosystems rather than the single ones shown here

and measure biophysical attributes that are most relevant to human well-being (Boyd and Banzhaf 2007; Landers and Nahlik 2013; Boyd et al. 2015). In the FEGS approach, specification of the beneficiary must come before identification of which biophysical attributes should be measured. The benefits that groups of people obtains from nature are determined by how they use, enjoy, or otherwise depend on nature. To facilitate the identification of FEGS endpoints, people are grouped into beneficiary classes that describe their interests which drive the consumption, use, or appreciation of ecosystem goods or services (Landers and Nahlik 2013). Then, the ways that a given beneficiary class interacts with a given ecosystem to obtain those good and services determines the biophysical component of a specific FEGS. Note that FEGS do not include the ecological components or processes that are necessary to produce it, which are intermediate processes (Haines-Young and Potschin 2018). Additionally, it is important to acknowledge that FEGS for one beneficiary (e.g., water temperature for an aquaculturalist) may be an intermediate process for another (e.g., a recreational angler).

The FEGS approach can bring clarity to environmental management by translating intermediate ecological processes into FEGS by asking the following questions (Table 1):

- "*Who* are the beneficiaries?"
- "How and where (i.e., in what ecosystem type) do they use, enjoy or appreciate nature?"

Table 1 Examples of how starting with the beneficiaries can help bridge intermediate ecosystem processes with their beneficiaries (**bold**), the FEGS attributes they use or enjoy (*italics*), and the relevant environmental context (underline)

Intermediate processes	Beneficiary-oriented questions	FEGS approach
Habitat for fauna	Who are the beneficiaries and what do they use, appreciate, or enjoy about habitat?	• **Recreational hunters** hunt *game animals* when visiting forested areas in the region • **Recreational birders** want to see a *specific species of charismatic bird* and are thus drawn to wetlands in the region
Water quality regulation	Who are the beneficiaries and what do they use, appreciate, or enjoy about water quality?	• **Residents, dependent on drinking water aquifers**, are concerned about *water salinity* in groundwater • **Snorkelers** are concerned about *water turbidity* in popular coastal waters • **Commercial fishermen** are concerned about *contaminants in edible fish tissue* in the lakes they frequent
Water quantity regulation	Who are the beneficiaries and what do they use, appreciate, or enjoy about water quantity?	• **Municipal drinking-water plant operators** care about the *reliable availability of fresh water* from streams • Coastal **home owners** directly understand the value of shoreline protection through dunes and vegetation in *reducing the probability of property damage due to erosion by waves*

- "What ecological end-products (EEPs) do they use, enjoy or appreciate"
- "Where is the EEP located (i.e., ecosystem type) that is used, enjoyed, or appreciated?"

In a given lake, for example, while recreational boaters might be concerned about water clarity, odor, or having sufficient water depth in which to operate a boat, an industrial processor will be primarily concerned with water corrosiveness, presence of biofouling organisms, and the reliability of water quantity. Taking a FEGS approach helps ensure that the full range of benefits is considered by identifying meaningful biophysical indicators or metrics to be monitored, valued, and reported. A FEGS approach also facilitates communication about what matters to people by ensuring that key issues or stakeholders are not overlooked, and by allowing management decisions to include things people care about and understand (Yee et al. 2017).

2.1 FEGS Classification System

Several frameworks have been proposed for classifying ecosystem services (e.g., MEA 2005; Haines-Young and Potschin 2018; Landers and Nahlik 2013; US EPA 2015). Finisdore et al. (2018) describes a suite of classification systems and outlines the range of benefits of using classification systems. The goals of these classification systems are to develop a common, shared language in an interdisciplinary field and provide a consistent framework for identifying, organizing, and accounting for ecosystem services. Two classification systems for FEGS have been developed by the US EPA to identify the types of uses, needs, or desires that a beneficiary seeks to obtain from a given ecosystem type from specific biophysical attributes present there. A FEGS approach facilitates the creation of information useful for valuation and helps to minimize double counting and valuation problems regarding intermediate processes that are not clearly distinguished from FEGS (Ojea et al. 2012; Nahlik et al. 2012). The FEGS Classification System (FEGS-CS) was developed to help "determine those specific ecosystem attribute(s) associated with the specific FEGS that the beneficiary values" such that "these can directly lead to identifying appropriate metrics and indicators for FEGS" (Landers and Nahlik 2013). The FEGS-CS was created primarily to aid in organizing ecological metrics and indicators that would provide meaningful input to environmental benefit assessment and policy decisions. The National Ecosystem Services Classification System (NESCS) was developed by environmental economists (US EPA 2015). The NESCS comprehensively and uniquely identifies distinct categories of FEGS to support analysis of how changes in ecosystems affect human well-being by applying, adapting, and combining the principles underlying existing economic accounting systems for market goods and services, primarily for use in environmental accounting (Russell et al. 2020). The NESCS defines the ecological attributes of an environment that flow as inputs to human uses (both market and non-market) to both emphasize human reliance on these flows and to illustrate how changes in policy could affect those flows and the well-being they provide to human users of those environments. The NESCS also links to standard accounting systems (such as the North American Industry Classification System, NAICS; https://www.census.gov/eos/www/naics/) and more directly to existing economic valuation practices than does the FEGS-CS (US EPA 2015). The US EPA is nearing completion of a merged classification system (NESCS Plus) to leverage the best of both systems. The NESCS Plus will reduce confusion caused by parallel classification systems and be consistent with prior systems so that it remains relevant for audiences of FEGS-CS and NESCS.

2.2 FEGS Metrics and Indicators

The metrics and indicators associated with FEGS are important for incorporating definable benefits that people receive from nature into elements of decision-making

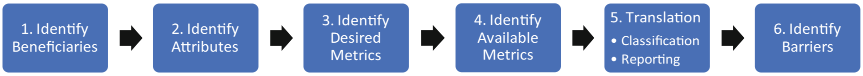

Fig. 3 Developing FEGS metrics and indicators for a given environment type is a 6-step process starting with beneficiaries and the attributes of nature that they value

processes (e.g., issue identification, options analysis, communication), as well as overall decision-making processes such as EBM and Structured Decision-Making (SDM).). In particular, the ecosystem-based framework underpinning FEGS-based classification systems allows analysts to view comprehensively the biophysical attributes (e.g., wild food, drinkable water, specific organisms) and resulting human benefits (e.g., nutrition, recreation, improved health, spiritual enrichment) provided by an ecosystem (Landers and Nahlik 2013). Further, FEGS-based classification systems identify classes of beneficiaries or users (e.g., commercial harvesters, recreational anglers, boaters) that potentially benefit from FEGS when they interact with nature. A simultaneous comparison of the different FEGS that might benefit each beneficiary or user group affected by a management option provides a complete summary and representation of the ways in which people may be impacted by management decisions. Synergies and trade-offs among groups of beneficiaries are expected. That is, an increase in a good or service of value to one group could either support a related increase to another beneficiary group or result in a reduction of a good or service to another group. For example, an increase in clean water for swimmers might also benefit recreational anglers, but an increase in complex reef structure that provides superior habitat to produce fish may be desirable to scuba divers (i.e., tourism), but undesirable for commercial and recreational fishing (e.g., due to gear entanglement).

Developing metrics and indicators of FEGS for a given ecosystem type begins with identifying which beneficiary groups directly interact with that ecosystem, then considers what specific components of nature are directly used, enjoyed, or appreciated by each beneficiary group, and moves to considering how those components could be measured directly (Fig. 3; Ringold et al. 2013). Typically, each beneficiary group will directly use, appreciate, or enjoy multiple biophysical attributes within a given ecosystem, with each attribute represented by one or more metrics. Those attributes may be identified by considering what individual people directly perceive or interact with from the environment. For example, an individual partaking in recreational angling within an estuary may care directly about the taxa and size of the fish she might catch, whether the fish she catches are safe to eat, whether the conditions at the site are safe, and/or whether the aesthetics of the site are appealing (Table 2). That is because an individual can use different attributes of an ecosystem depending which one of many "roles" she is playing (e.g., catch and release fisherman, subsistence fisherman, experiencer of natural place, etc.) while doing an activity as complex as recreational angling. Direct measurement of some attributes may be difficult or expensive (e.g., angler success in fishing), necessitating the use of surrogate metrics that approximate the suite of valued attributes (e.g., percent

Table 2 Developing FEGS metrics and indicators for recreational anglers in estuaries based on the attributes of nature that anglers care about. For illustrative purposes of a full suite of "what matters" topics, two non FEGS rows are also included (marked with **)

Ecosystem	Beneficiary	What matters directly to this beneficiary?	Attribute category	Attribute	Sub-attribute	Ideal metrics	Currently available metrics & indicators
Estuaries	Recreational anglers	Is it safe to go out?**	Water	Water movement	Wave intensity	Wave height, speed & direction	Wave height, current speed & current direction
			Weather	Wind strength/ speed	Wind intensity	Wind speed & direction	Wind speed & wind direction
		Will I catch fish I want to eat?	Fauna	Edible fauna	Taxa & presence	Game fish species presence or abundance, size, distribution, diversity	Game fish presence/ absence, catch per unit effort, harvest (biomass)
		Is there a chance I will become sick from eating caught fish?	Fauna	Edible fauna	Concentration of pathogens/ toxins/contaminants/parasites	Biotoxins, pathogens, parasites, chemical contaminants	Alerts from the FDA, fish consumption advisories
		Will my hook or net get fouled? **	Soil & substrate	Substrate quality	Structure	Marine hazards, shoreline/ benthos complexity	Marine hazards, local reports
		Is this area aesthetically enjoyable?	Environmental aesthetics	Scents, viewscapes, soundscapes	Water quality	Water color, algae taxa, water clarity & smell, presence of hypoxia or anoxia, sound environment	Monitoring data for water quality, sound, or odors, local reports
					Viewscape	Natural shoreline landscapes, unobstructed waterscapes	Percent natural land cover on shoreline, percent of water surface area as built structures (from aerial or satellite imagery)

of natural land cover along shorelines) while acknowledging the limitations of using surrogates. An important distinction here is ideal metrics versus available, or surrogate, metrics. While we need to specify the ideal metrics, we must then quantify them within budget, time, and available data constraints.

Identifying those metrics that closely represent how beneficiaries use attributes of nature facilitates the translation of data obtained with those metrics into information that analysts, stakeholders, and the public can use in their decision processes (Wainger and Mazzotta 2011). Balanced against that ideal is the pragmatic desire to use both existing metrics (i.e., to minimize methodology development) and existing data. Wainger and Mazzotta (2011) suggest that ideal FEGS metrics are:

- Easily understood by non-experts
- As close to the FEGS as possible (e.g., the taxa, size, condition, and abundance of game fish, as opposed to the total number of all fish at a location)
- Readily available (e.g., existing methodology and data sets)
- Available for large areas
- Available at a user-appropriate scale as defined by the management or user question(s) and by the beneficiary perceptions (e.g., a recreational angler might want data on the scale of an individual fishing spot, while a fisheries regulator might need information at a statewide or regional scale).

The number of FEGS metrics that need to be measured depends on the problem requirements. The list of all potential FEGS metrics for any ecosystem type can be quite extensive if the interests of all potential beneficiaries are considered. From an operationalizing perspective, however, analysts, stakeholders, or managers may determine that the classification of beneficiaries or ecosystem types (e.g., for each of the FEGS classification systems) may be too general for the problem at hand. That could drive the need for a narrower specification of beneficiaries, ecosystem sub-types, or ecosystem attributes, and consequently, a refined set of relevant metrics. The FEGS Scoping Tool (Sharpe et al. 2020) was developed to help stakeholders or managers collaboratively and transparently identify common interests and prioritize the attributes of nature that are most valued across beneficiary groups. Analysts can draw from existing sets of FEGS metrics or build from those to identify metrics relevant to quantifying and sustaining the stakeholders' most valued benefits of nature.

3 Operationalizing the FEGS Approach

3.1 Role of FEGS in Ecosystem-Based Management (EBM)

The FEGS approach can provide several important tools for EBM. First, the categorization of beneficiaries of FEGS helps to specify who the stakeholders are whose wants, needs, desires, and perceptions need to be addressed (Ringold et al. 2013; Landers et al. 2016). Those beneficiaries may live within or adjacent to an ecosystem (e.g., coastal communities) or distantly, and may come from diverse socio-economic groups. The potential for heterogeneity in their values needs to be considered when assigning weights to biophysical outcomes. Second, a FEGS-based classification system informs the selection of metrics or indicators useful for identifying, communicating, and quantifying the ecological attributes for application in assessing or monitoring FEGS, their production, or their use or appreciation by people. Third, identification of beneficiaries and associated FEGS informs the selection and development of models to forecast production of FEGS (i.e., ecological production functions; Wainger and Mazzotta 2011; Bruins et al. 2017), the delivery of benefits to people (i.e., benefit functions; Wainger and Mazzotta 2011; Villa et al. 2014; Bousquin and Mazzotta 2020), and integrated ecological-social well-being frameworks (Schlueter et al. 2012).

Identification of the beneficiaries of FEGS within an area of interest, and the ecological attributes that the beneficiaries use, helps decision makers understand the types and magnitudes of the tradeoffs involved in policy options, how beneficiaries will be affected by changing conditions within the ecosystem, the biophysical features of the ecosystem that are important to those beneficiaries, where and how beneficiaries experience those biophysical features, and where and how those biophysical features are produced. For example, the US EPA's Remediation-to-Restoration-to-Revitalization (R2R2R) program for contaminated-site clean-up actions around the U.S. Great Lakes inherently uses a beneficiary-centric, FEGS approach by working towards each community's vision for their desired human-nature outcomes (i.e., revitalization), and incorporates those outcomes into the goals of each phase of the clean-up and restoration (Williams and Hoffman 2020). In combination, this information can inform prioritization and local management relative to ecosystem goals and help to align decision-making with local values.

3.2 Integrating FEGS Into a Structured Decision Making (SDM) Framework, and Relevance to EBM

Structured Decision Making (SDM) provides an organizing framework to formally integrate FEGS, or any other approaches and tools into EBM (Gregory et al. 2012; also see Sharpe et al. 2020). The use of SDM places a strong emphasis on problem structuring by clarifying the problem, identifying objectives (i.e., separating those objectives to accomplish at the end from those objectives that are important ways to reach end objectives), and developing meaningful measures (Marcot et al. 2012; Maseyk et al. 2017). The use of SDM is an alternative to technical assessments or cost-benefit analysis, which may be done along with SDM, but are not required. Without clarifying "what really matters" upfront, resources can be wasted collecting the wrong information for the wrong problem, leading to irrelevant or misleading assessments (Carriger et al. 2013). A focus on what stakeholders' value, in contrast, can lead to more creative and effective outcomes. The FEGS approach can facilitate management decisions and actions that have a higher likelihood of acceptance across a variety of stakeholders because they are based on the stakeholders' priorities.

An SDM process includes a series of steps similar to other decision processes (Table 3) and can be used to identify where FEGS concepts can be integrated into an EBM decision process even while using other decision frameworks, such as AQUACROSS (Piet et al. 2017, 2020), an EBM policymaking framework (Cormier et al. 2017), or integrated ecosystem assessment (Foley et al. 2013). Each decision framework has a unique set of steps, but FEGS concepts can still be integrated at many places within each process (Table 3).

The FEGS approach advances the ability to identify, articulate, measure, and assess the potential role of relevant ecosystem goods and services in a given decision context. Using FEGS metrics or indicators, for example, may provide more relevant assessment endpoints for EBM and enable communication of benefits to humans better than endpoints that are difficult to link to human use (e.g., total primary production, pH, species diversity, etc.). The use of FEGS metrics should also be useful for regulatory purposes, such as risk assessment endpoints (Munns et al. 2017). Additionally, having a consistent approach for defining EBM terminology (Arkema et al. 2006) and clear articulation of EBM principles (Delaclamara et al. 2020) will help practitioners identify how to incorporate ecosystem services and decision strategies for a given EBM context. Further, the FEGS approach and tools presented here help EBM practitioners identify users or beneficiaries who will be affected by environmental changes at a site due to changes in the specific ecological attributes that those groups derive benefit from, for a given decision context (Fig. 4).

Within a larger SDM framework, the FEGS approach can also assist EBM practitioners identify and prioritize stakeholders to bring into the decision process (i.e., using the FEGS Scoping tool; Sharpe et al. 2020). Because FEGS are the link between biophysical condition and socio-economic benefits to people, this approach is compatible with socio-ecological systems frameworks (Elliott and O'Higgins 2020; Piet et al. 2020). By connecting ecosystem services directly to human health and well-being endpoints within a structured framework, the FEGS approach lends itself to systems analysis (including linkage frameworks, network analysis, and Health Impact Assessments (HIA)); Robinson and Culhane 2020; Williams and Hoffman 2020) for identifying and evaluating key stressors in the system or vulnerabilities to ecosystem-services supply that may need to be prioritized for management. Additionally, the FEGS approach helps advance efforts to develop classification systems useful for a range of EBM practitioners (Culhane et al. 2020). These classification systems can be leveraged to identify ecosystem-services focused metrics and indicators that can inform the evaluation of management alternatives.

Decision analysis tools, such as means-ends networks, direct ranking, swing weighting, or consequence tables (Gregory et al. 2012) can be used to identify management actions that might improve ecosystem services production or where ecosystem services may be a means to achieving other social or economic objectives. Because FEGS are closely linked to human beneficiaries, the FEGS approach fosters the identification and application of relevant tools across the SDM cycle (Fig. 4; Yee et al. 2017), such as for prioritizing and measuring economic benefits of ecosystem

Table 3 How a FEGS perspective can be integrated into other EBM decision frameworks. Acronyms: SDM (structured decision making); AQUACROSS (AQUAtic biodiversity & ecosystem services aCROSS European Union policies); EBM (ecosystem-based management)

SDM framework[a] *FEGS concepts*	AQUACROSS framework[b]	EBM policymaking process[c]	Integrated ecosystem assessment[d]
1. **Clarify decision context** *What FEGS might be impacted?* *What FEGS beneficiaries should be included as stakeholders?*	Phase 1: Societal goals • Key threats; policies; synergies and conflicts	1. Strategic goal-setting • Prioritize threats to ecosystems • Identify opportunities to improve socioeconomic and ecosystem status	1. Scoping • Identify goals and threats
2. **Define objectives and performance measures** *Which objectives are FEGS or their benefits?* *How will they be measured?*	Phase 2: Description of the Socio-ecological System • Ecological & social system criteria	2. Tactical objectives • Evaluations of ecological, cultural, social, and economic impacts • Indicators and performance measures	2. Indicators and reference levels • Identify measures for attributes of management interest
3. **Develop alternatives** *Are any FEGS means to achieve objectives?*	Phase 3: Planning an EBM response • Management strategy – Management measures – Policy instrument	• Development of management strategies	3. Risk analysis • Link indicators and threats • Identify overlaps between activities and impacts
4. **Estimate consequences** *What ecosystem services models or data are needed to estimate consequences?*	Phase 4: Implementation, monitoring and evaluation • Assessment of current state using functional relationships, time-series, spatial comparisons • Forecasting and scenarios to predict consequences	3. Management measures • Assess implementation feasibility • Evaluate ability of proposed management actions to achieve objectives	4. Management strategy evaluation • Assess options
5. **Evaluate trade-offs** *How much loss or gain in FEGS benefits is acceptable?* *Are FEGS beneficiaries differentially impacted?*	• Evaluation of management plans – Compare alternative strategies to "business as usual"	• Comparison of alternative management actions against weighted objective priorities	• Choose approach
6. **Implement, monitor, review** *Did the decision lead to measurable change in FEGS?* *Were there unforeseen impacts on FEGS to consider next time?*	• Evaluate effectiveness, efficiency, and equity of outcomes	4. Adaptive Management • Effective monitoring plan • Evaluate management effectiveness and alternative options • Recommendations to improve scientific input	5. Monitoring and evaluation • Track indicators and assess change

Table modified from Yee et al. (2017). (a) Yee et al. (2017); (b) Piet et al. (2017); (c) Cormier et al. (2017); (d) Foley et al. (2013)

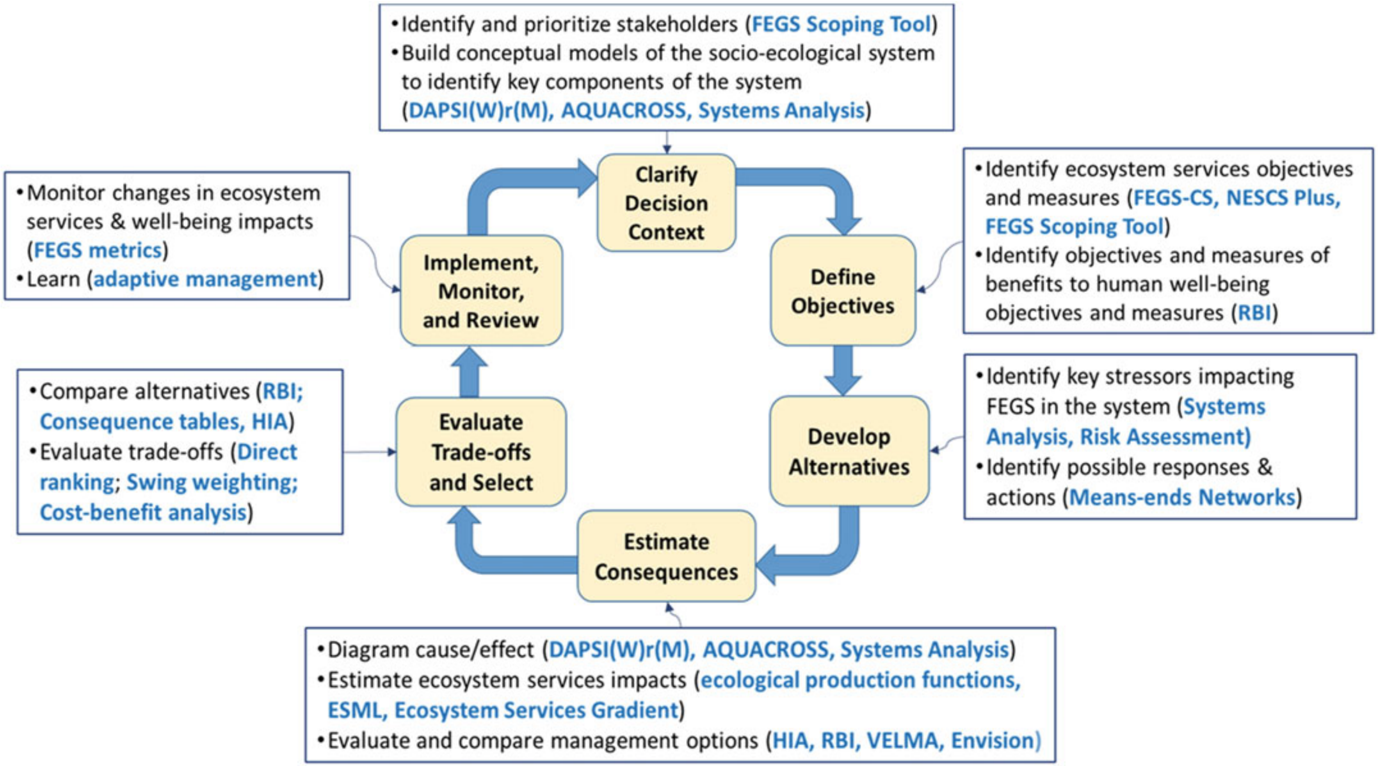

Fig. 4 Examples of tools (in bold blue; references in the text) that support integrating FEGS (Final Ecosystem Goods & Services) into different phases of Structured Decision Making (SDM). Tool acronyms modified from Yee et al. (2017). Acronyms: AQUACROSS (AQUAtic biodiversity & ecosystem services aCROSS European Union policies); DAPSI(W)r(M) (Drivers, Activities, Pressures, State, Impacts (on Welfare), Responses (as Measures); ESML (Eco Service Models Library); FEGS-CS (FEGS Classification System); HIA (Health Impact Assessment); NESCS (National Ecosystem Services Classification System); RBI (Rapid Benefits Indicators); VELMA (Visualizing Ecological Land anagement Assessments); see text for explanation

services using the related Rapid Benefits Indicators (RBI) approach (Bousquin and Mazzotta 2020).

Ecological production functions (e.g., models useful for estimating ecosystem services) can be utilized in an EBM context to quantitatively predict changes in ecosystem services in response to changes in a system, management alternatives for a decision context (Fulford et al. 2020; Lewis et al. 2020). In some cases, expert knowledge and synthesis of available data may be sufficient, or used in combination with quantitative modelling, to communicate how usage of natural resources changes in response to changes in ecological conditions (e.g., ecosystem services gradients; Yee et al. 2020). Different models may be integrated and applied within decision support systems (e.g., Envision, VELMA; McKane et al. 2020) to evaluate and compare the impacts of alternative management options on ecosystem services or other relevant objectives.

4 Summary

This chapter outlined the suite of concepts, methods, and tools that comprise the FEGS approach and demonstrates steps taken to incorporate the approach into decision-making for EBM. The FEGS approach focuses on advancing both the field of ecosystem services science and the utilization of decision-making frameworks to the field of EBM (Delacámara et al. 2020). One important application of the FEGS approach is to aid in guiding the development of ecological endpoints and metrics useful for assessment of stocks or site/system conditions for EBM. The foundational research across elements of the FEGS approach has been made operational across a suite of case studies and decision contexts, including examining alternatives in coastal forest management in the U.S. Pacific Northwest (McKane et al. 2020), cleanup of contaminated sites in the Great Lakes (Williams and Hoffman 2020), resiliency planning following natural disasters (Myer and Johnston 2020), restoring large ecosystems such as the Everglades (Gibble et al. 2020), and examining ecosystem management practices among different future climate scenarios in the Lower Mekong Basin (Johnston et al. 2020).

The FEGS approach can also be useful to EBM practitioners through the development of strategic communication messages (Harwell et al. 2020) for stakeholders and the public regarding the benefits associated with pending or implemented decisions. In particular, the FEGS approach aids in communicating which people within a system (i.e., beneficiary groups) will be affected by changes to the condition of the environment owing to how those changes will affect the benefits (i.e., health, economic, or social) people obtain from nature (Fig. 1). For example, recent work has advanced the field of biological condition gradients used to characterize and communicate the status of an ecosystem in relation to thresholds of meaningful change and potential management actions to incorporate ecosystem services (Yee et al. 2020). Overall, the approach of starting with human beneficiaries and asking what they use, appreciate, or enjoy about ecosystems holds promise for advancing ecosystem goods and services science to support environmental decision making, particularly ecosystem-based management.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Part III
Tools and Techniques

Ecosystem-Based Management and Natural Capital Accounting

Marc Russell, Charles Rhodes, George Van Houtven, Paramita Sinha, Katherine Warnell, and Matthew C. Harwell

Abstract Natural capital includes the physical and biophysical components of an ecosystem working together to produce a flow of services to the economy and to society that support human well-being. An ecosystem service thus represents a type of transaction between natural capital and humans and can be considered within tools to support Ecosystem-Based Management (EBM) decision making. A natural capital perspective can be a useful way to put the value of ecosystems on par with other socioeconomic values in an EBM decision context. Further, the application of structured classifications for ecosystem components, human beneficiaries (users), and a suite of flows of final ecosystem services helps EBM practitioners organize information for a given decision context. This chapter explores the utility of natural capital accounting as a tool for EBM, outlines a standardized framework for natural capital accounting, and summarizes an ecosystem services classification system for natural capital accounting that can be used as an EBM tool, especially relevant for the decision-making step of evaluating management options (e.g., scenarios).

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Lessons Learned

- The organizational structure of the National Ecosystem Services Classification System (NESCS) allows for the determination of environmental and valuation measures and metrics using classes and sub-classes specific enough that they minimize the possibility of double counting specific ecosystem service flows, an important criterion for establishing the creditability of ecosystem services assessments.
- The format of a NESCS code is **WW.XX.YY.ZZZ**, where each set of digits refers to the Environment, Ecological End-product, Direct Use/Non-use, and Direct User classes and subclasses, a useful feature for aggregating items in natural capital accounting tables, representing a unique potential pathway through which changes in Natural Capital may affect human welfare.
- Natural capital accounting efforts are directly relevant to the structured decision-making step "Estimate Consequences" in answering the question, "What ecosystem services models or data are needed to estimate consequences?"
- Natural capital accounting helps EBM practitioners organize: information for a given decision context; ways to approach the identification and valuation of the final ecosystem services for EBM decision making, especially relevant for evaluating management options (e.g., scenarios); and the standardized tracking of specific final ecosystem services over time.

Needs to Advance EBM

- As a new potential tool in the EBM toolbox, EBM practitioners need to examine natural capital accounting principles as part of efforts to analyze different scenario options as part of a decision context.
- Several natural capital accounting approaches (e.g., NESCS and InVEST) could be applied to the same decision context to inform the value-added benefits of natural capital accounting in EBM decisions.

1 Introduction

Management of ecosystems is a complicated affair (Delacámara et al. 2020). Not only do managers have to consider the many interactions in the ecology of a system, they are also mandated to manage the system for human-established goals. These goals might be conservation or restoration focused and oriented for ecological health and integrity or designated uses by humans. In either case the manager needs a way to break down a system into its component parts, quantify them in some way, link those parts to management goals, and assess trends over time, so they can adapt their management accordingly (Arkema et al. 2006; Guerry et al. 2015). Since physical structures, plants, and animals are easier to track than functions or processes, managers often rely on quantification of habitats, species, and abiotic factors monitored on a reoccurring basis. These factors can be thought of as natural capital (Costanza et al. 1997; Millennium Ecosystem Assessment 2005).

Natural capital is the physical components of an ecosystem working together to produce something of value to people (Costanza and Daly 1992). Ecosystem-based managers are focused on natural capital and on what that capital produces in their jurisdiction. Natural capital, just like typical economic capital, can be thought of as the machinery or structures that function to produce goods. In the case of natural capital, these goods are biophysical components that, when used or appreciated by humans, produce a flow of services to the economy and to society more generally, thereby supporting human well-being. Natural capital might act alone or may interact with other natural capital in a series of production functions—relationships between one feature of the environment and the production of another—to produce that which is directly valued, used, or otherwise consumed by humans. That biophysical component of nature that is used directly by humans has been referred to as a final ecosystem good (Landers and Nahlik 2013; DeWitt et al. 2020) or an ecological end-product (Boyd and Banzhaf 2007; Finisdore et al. 2019). These terms denote that the useful biophysical component is a result of production by natural capital. The Ecosystem Service (sometimes referred to as an ES in the literature) then represents a type of transaction between natural capital and humans. At that point, the Final Ecosystem Service (sometimes referred to as FES in the literature) is a transaction that adds value to economic production processes (which generate economic goods and services), or that directly contributes to human well-being (such as from the inspiration provided by a natural landscape).

Value is placed on the flow of ecosystem services produced when humans interact with ecosystems, as natural capital, through their use or appreciation of a final ecosystem good or ecological end-product (Farber et al. 2002). A natural capital perspective of ecosystems may help managers align ecological production and resulting flows of ecosystem services in a way that best supports or enhances human well-being. This chapter explores elements of natural capital, a standardized framework for natural capital accounting, ecosystem services classification in natural capital accounting, and the use of natural capital accounting as a tool for Ecosystem-Based Management (EBM).

2 Elements of Natural Capital

A natural capital perspective lends itself to the application of already accepted socioeconomic frameworks, tools, and approaches, and thus can be a useful way to put the value of ecosystems on par with other socioeconomic values in the system being managed (Wackernagel et al. 1999; Hein et al. 2016). An accounting framework, for example, can be applied to natural capital production to help differentiate what and how things produced by ecosystems are used, by whom, and whether supply is being maintained at levels that satisfy demand. Natural capital accounting is practiced by several countries to quantify and track natural resources such as in water (e.g., Hoekstra 2009), minerals (e.g., Lange 2004), and land accounts (e.g., Weber 2007). Specific ecological production in ecosystems is also being tracked using timber (e.g., Gundimeda et al. 2007), fisheries (e.g., Lange 2004), and wildlife accounts (e.g., Anderson 2003). These accounts have been called for by countries and regions experiencing scarcities, such as in droughts, or in areas that rely heavily on natural resources as the base of their economies. They are becoming more commonplace as the World Bank and the United Nations call for more standardized practices of ecosystem reporting and management (United Nations 2014). A natural capital accounting framework also allows one to tie ecosystem production to the economy by providing the structure to quantify biophysical supply and use by specific ecosystems and users, thereby providing the information needed for valuation and possible translation of that biophysical use into monetary terms (Guerry et al. 2015).

Natural capital accounting requires three types of classifications to work together. First, a list of ecosystems, natural areas, or other geographic groupings that separate out the various production areas with no overlapping areas is needed to cross reference to ecosystem production or supply from natural capital located within the boundaries of the area. Second, the supply from natural capital in the system needs to be apportioned to a list of users. Common to both the supply and use tables within an account, and tying them together, are the various ecosystem services. The quantified ecosystem goods, or ecological end-products (sometimes referred to as EEP in the literature), and their use by users (beneficiaries) for a given time period of a given natural capital account serves as the measure or count of ecosystem service flows that will populate the table cells. Within an accounting framework, for the accounts to balance, the total supply from ecosystems must equal the use by users. It is the list of these ecosystem goods or ecological end-products that is the third type of classification that is needed to construct natural capital accounts.

3 Standard Framework for Natural Capital Accounting

A standard framework for natural capital accounting called the System of Environmental-Economic Accounting, or SEEA, has been developed by the United Nations and partner organizations (Hein et al. 2016). Tangible environmental assets including land, water, minerals, and several resources such as timber and fish are part of the Central Framework, which makes up the core of SEEA, and is designed to quantify natural resources and their contributions to the economy in biophysical and monetary terms (United Nations 2014). The SEEA also houses Experimental Ecosystem Accounts (SEEA-EEA) which are tasked with tracking the extent and condition of ecosystem assets (ecosystems represented by spatial areas; e.g., forests, wetlands, cropland) and the flows of various ecosystem services that these ecosystem assets provide to people and the economy (United Nations 2014). These Experimental Ecosystem Accounts are needed to account for flows from complex ecosystems to people, flows that have not been treated in national accounts as natural resources and traded as commodities in the market. Ecosystem service flows in the Experimental Ecosystem Accounts are by nature difficult to translate into monetary contributions to the economy, as is done with the environmental assets of the Central Framework.

The SEEA-EEA definitions are laborious, detailed, and undergoing updates through 2020, but were constructed with extreme care to meet national accounting needs. Ecosystem assets are a type of capital, similar to machines in a factory, and yield a flow of ecosystem services, just as the factory machines yield a flow of services (e.g., they might stamp thick metal sheets into useful shapes faster and more accurately than humans with hammers could). The SEEA-EEA ecosystem services are "final" as in the definition in Boyd and Banzhaf (2007), and can be distinct from "benefits," where benefits may be the FES contribution to economic production or the FES may go directly to the users without further economic processing or inputs.

Within the SEEA-EEA framework of accounts, figures proposed as frames for "ecosystem services supply and use tables" list "ecosystem services" as rows in the supply-and-use tables (Fig. 1). The supply tables connect ecosystem services and products (in rows) to the types of assets that produce them (in columns) while the use tables connect them to the types of users that use them (in columns). The dark shaded shells in these tables represent "null sets" meaning that no supply or use connection exists for that row-column combination.

Ecosystem services supply-and-use tables are required to balance in accounting. This means that supply of an ecological end-product cannot be larger or smaller than the amount of that ecological end-product used in ecosystem service transactions, nor can it be missing for an ecosystem service to exist. In accounting, a supply with no use cannot represent an ecosystem service because there is no balancing entry in the corresponding use table. A nation cannot have 1000 km of swimmable shoreline and use only 100 km of it in the same accounting year and be able to correctly say they had more than 100 km of shoreline supplied and used as an ecological end-product within an ecosystem service. By the strictures of accounting, the other 900 km cannot be counted as an ecological end-product or part of a flow of ecosystem services during that year, because they were not used. Rows with no entries thus cannot represent ecological end-products, because the existence of transactions (depicted with entries in the intersecting cells) verifies final use, linking supply and demand within the flow of an ecosystem service. The numbers in the cells are a way to quantify particular ecosystem services as a transaction or flow between the supplying ecosystem asset and the human user.

Without use, there is not an ecosystem service, and thus proposed ecological end-products that ultimately do not have direct human use or appreciation (e.g., a wolf or eagle that lives and dies hundreds of miles from any human, say in remote reaches of Alaska, and is not valued specifically for its existence or bequest value) should not be included in accounting supply-and-use tables. Nonuse values are harder to derive in a transaction-value-based framework, which the SEEA is, so are currently not allowed. However, those same wolves or eagles not directly used might still be counted in another type of EEA account other than the supply-and-use tables, perhaps reflecting the condition of a distant ecosystem asset in a condition table or providing a service only between ecosystem assets. Thus, it is critical to define ecosystem services with a use and user component, and not just by identifying potential supply. Ecosystem services classification systems before those developed by the USEPA may have presumed, but did not directly incorporate, a use/user component. Without actual use, things classified as part of an ecosystem service are just ecosystem characteristics and processes that cycle through the environment, and thus not ecological end-products or part of final ecosystem services.

ECOSYSTEM SERVICES SUPPLY TABLE

		Type of economic unit	Type of Ecosystem Unit
Ecosystem services Provisioning services Regulating services Cultural services		A	B
Products		C	D

ECOSYSTEM SERVICES USE TABLE

		Type of economic unit	Type of Ecosystem Unit
Ecosystem services Provisioning services Regulating services Cultural services		E	F
Products		G	H

Fig. 1 SEEA EEA ecosystem services supply-and-use table structure. (Adapted from Table 4.4, December 2015 draft of United Nations 2014)

The U.S. natural capital accounting workgroup (Warnell et al. 2020) recognized the narrowness of the accounting definition and the difficulty of measuring actual ecosystem services and put important ecological measures and measures of known precursors of ecosystem services into an ecosystem condition account. For example, eagles and beaches that humans do not interact with might be tracked in a condition account. Accounts that are balanced, totaled, and specifically slated for integration with Standard National Accounts (SNA, the international accounting structure that provides a common economic foundation before environmental accounting is applied) need to have safeguards against double-counting. The supply-and-use tables are intended to be monetized and to be integrated with the SNA accounts in common terms. Condition accounts are not currently proposed to be integrated into the SNA or to be translated into dollar values, so can represent characteristics, processes and stocks of things (e.g., eagles) in the environment that are not directly used as part of ecosystem services.

4 Classification of Flows of Ecosystem Services for Natural Capital Accounting

The US EPA's National Ecosystem Services Classification System (NESCS) offers all three of the classification elements (i.e., types of supplying ecosystems, types of ecological end-products supplied and used, and types of users with different use demands) that are needed to do natural capital accounting (USEPA 2015). The NESCS and associated approach were designed to help identify and reference flows of services from ecosystems to human beings in a mutually exclusive way, which is critical for natural capital accounting.

The NESCS has a four-part structure (Fig. 2), with each part populated by a hierarchical set of classes and subclasses, which are intended to be as non-overlapping and comprehensive as possible for identifying distinct ecosystem service flows.

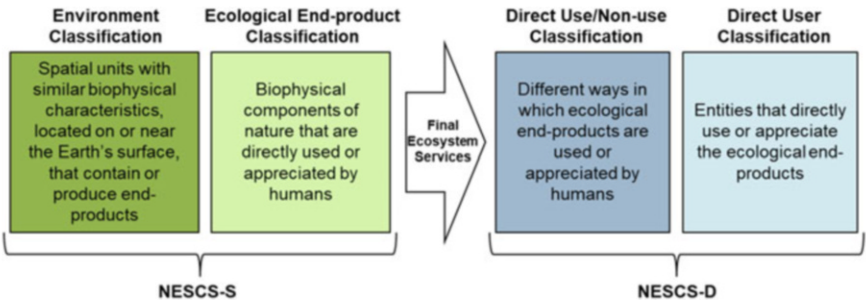


Fig. 2 NESCS four-part structure (from USEPA 2015)

The four parts of NESCS can be divided into two sides that are relevant for accounting: the supply-side (NESCS-S; Fig. 2 left) and demand-side (NESCS-D; Fig. 2 right) of the system. The NESCS-S side represents the source of ecosystem service flows. It characterizes different types of natural capital as environmental classes and represents the output of their ecological production as classes of ecological end-products. The NESCS-D side represents the recipients of the ecosystem service flows, by classifying the human users of the ecological end-products and how they are used. Taken together, the four parts hold the information necessary to designate individual flows of ecosystem services from natural capital to humans.

The NESCS adopts a nested hierarchical structure so that each part can be represented at multiple levels of aggregation or detail. The initial sets of classes presented by NESCS are meant to provide a high-level classification that provides a mutually exclusive partitioning of classes, each with its own set of subclasses. These initial classes and hierarchical structure were developed to meet the design requirements not found in other classification systems. The NESCS is intended to provide a broad and comprehensive structure capable of covering all the different ecosystem services that humans value from nature. In this regard, it is consistent with the total economic valuation (TEV) approach (e.g., Jewhurst and Mazzotta 2016) that is often used as a framework for valuing natural resources and environmental benefits. The NESCS is also expandable enough to identify specific classes and subclasses for determining environmental and valuation measures and metrics. It has classes that are intuitively separate from each other and thus specific enough that, when combined to represent flows of ecosystem services, they minimize the possibility of double counting specific ecosystem service flows, an important criterion for establishing the creditability of ecosystem services assessments (Fu et al. 2011). Moreover, by focusing on ecosystem services and the ecological end-products produced by natural systems, the NESCS structure also minimizes the possibility of double counting the contribution of "intermediate" ecological production processes in estimates of ecosystem service values (Landers et al. 2016). This is because the value of intermediate processes should be embedded within, and thus fully captured by, the value of final ecosystem services.

Each unique ecosystem service can also be easily referenced and identified by a NESCS code. The general format of the code is **WW.XX.YY.ZZZ**, where each set of digits (e.g., WW, XX, etc.) refer to the Environment, Ecological End-product, Direct Use/Non-use and Direct User classes and subclasses, respectively, as described below (Fig. 3). One example from Fig. 3, the WW.XX.YY.ZZZ NESCS code "41.12.11.2ZZ" would thus represent household extractive use of liquid water from a deciduous forest, as for a hiker taking a drink of water from a mountain stream below the tree line. Digits can be added or removed from each part of the four-part code to represent any further breakdown into more detailed sub-classes or to represent the roll up of subclasses into larger classes. While the ability to roll up classes into fewer classes is a useful feature for aggregating items in natural capital accounting tables, it may be less so for other uses of NESCS not covered here, such as the identification of metrics for each Flow of Ecosystem Service, mapping of areas of supply, or setting up scenario variables, which would primarily take advantage of

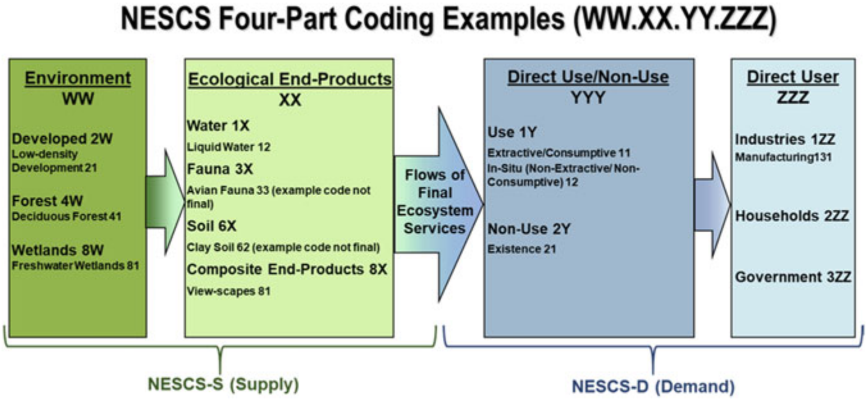


Fig. 3 NESCS coding examples (WW.XX.YY.ZZZ)

the flexible yet comprehensive nature of the hierarchical classification system. Each unique combination of one class or subclass from each of the four parts of NESCS defines a separate Flow of Ecosystem Service. In other words, it represents a unique potential pathway through which changes in Natural Capital may affect human welfare. The ability to define different combinations of classes allows the NESCS structure to be flexible and comprehensive. For example, the same Ecological End-Product category may be used in multiple ways or a single use category can be linked to multiple different user categories.

The first part of NESCS is the Environment classification (Fig. 3; far left box; "**WW**" part of the code), which is currently based on the system specified by Landers and Nahlik (2013). It spatially divides the Earth into non-overlapping areas with similar biophysical characteristics that, when taken together, can completely cover the surface of the Earth. The NESCS currently designates Environment classes down to a second-level of hierarchy, including a two-digit numeric coding structure, which provides a short-hand notation for the hierarchy and a numeric identifier for each element within each part. Example subclasses for the environment might include deciduous forests, freshwater wetlands, or low-density developments. When used within a four-part code designating a final ecosystem service, the environment classes and subclasses specifically refer to the environment in which the relevant ecological end-product is used or appreciated.

The second part of NESCS is the Ecological End-products classification (Fig. 3; second to left box; "XX" part of the code). End-products represent the components in nature that humans most directly use or appreciate (Farber et al. 2002). In its most aggregate form, classes of ecological end-products include Fauna, Flora, Water, Soil, Air, etc. As with Environment classes, these are subsequently broken down into a second-level hierarchy of subclasses, with codes designated for each one. Examples might include liquid water, avian fauna, or clay soils. One of the challenges in constructing this end-product classification is defining mutually exclusive categories while also recognizing that there can be substantial complexity and diversity in what people care about in nature. In addition to individual end-products, people often care about combinations of them. For example, people may value an entire landscape as more than the sum of individual value for the flora, fauna, water, etc., that are parts of the landscape. The NESCS includes a category called "Composite End-products" with sub-classes representing different types of natural features or phenomena that directly matter to humans but can be thought of as combinations of the other end-products. An example might be view-scapes. It is important to recognize that the joint nature of composite end-products may be problematic when applied to an accounting framework where there can be no overlaps between entries in the supply-and-use tables.

The third part of NESCS is the Direct Use/Non-Use classification (Fig. 3; second to right box; "**YY**" part of the code). The classes and sub-classes in this component describe distinct ways in which end-products can be directly used or appreciated by humans, again with the objective of providing categories that are non-overlapping and as comprehensive as possible. Examples include extraction of natural resources, such as wood, for transformation into economic products, such as timber, or non-extractive in-situ use associated with outdoor recreation, such as birdwatching. Consistent with the TEV framework, it includes separate "use" and "non-use" categories which make up the first hierarchical level. These use and non-use classes are then further subdivided into mutually exclusive extractive and in-situ use classes at the second-level of the hierarchy. Non-use use classes include existence, bequest, or other uses where humans do not have direct contact or physical use of the ecological end-product but might have a value associated with knowing that something exists or that something will be around for their descendants to enjoy.

The fourth and last NESCS part is the Direct User classification (Fig. 3; far right box; "**ZZZ**" part of the code). This component defines the separate economic sectors though which people directly use or appreciate end-products. Following established classification structures adopted by the U.S. Census Bureau and United Nations, the first level includes broad sectors of the economy, here: Industry, Households, and Government. To further subdivide the industry class, it adopts the existing North American Industrial Classification System (NAICS) and coding system, which is the standard used by U.S. federal statistical agencies in classifying business establishments (United States 2017). An example is the Manufacturing Industry sector which has a three-digit code, one digit for industry and two additional digits for the sector. Unlike commercial establishments, which tend to specialize in certain productive activities and can therefore be assigned to individual NAICS categories, households and governments do not specialize in the same way. For this reason, they are not yet divided into sub-classes as are NAICS categories. They currently are presented as first-level hierarchical classes, with further designation to subclasses remaining an option. One way to differentiate the many ways households and governments interact with nature is through the combination of the household or government user class with the different previously described use/non-use classes.

The NESCS approach helps guide the construction of supply-and-use tables within SEEA-EEA in that it clearly separates ecological end-products from ecosystem services by defining ecological end-products (i.e., components in nature that humans most directly use or appreciate) as part of a flow of ecosystem services (i.e., "transactions" that include human use of those components by a specific user). Without this specificity in terminology, confusion of, and mismatches between, what ecologists and other natural scientists measure as that which is supplied by an ecosystem asset and what social scientists measure as that which is used by a user could slow or stall efforts to generate useful estimates of ES for accounting.

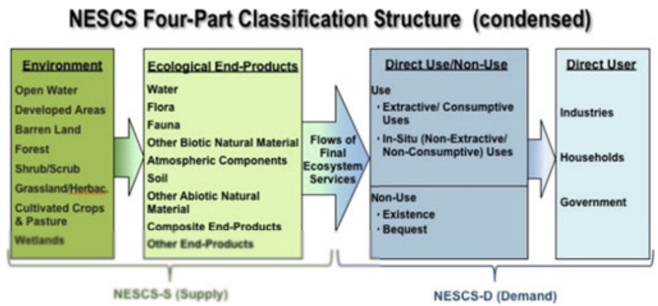
The NESCS identifies and classifies components of final ecosystem services according to both the environmental supplier and human user of the service in order to identify where certain metrics best fit within the natural capital accounting structure (Fig. 4a). The NESCS can help distinguish between ecosystem services eligible for inclusion in supply-and-use accounts in standard statistical frameworks, such as SEEA-EEA. By viewing the standard statistical frameworks through the NESCS lens, one can separate ecosystem characteristics and processes that some have considered to be ecosystem services from those ecosystem services that are allowable by the structure of supply-and-use accounting (finality in accounting is proven by direct use; Boyd and Banzhaf 2007; Landers and Nahlik 2013; Landers et al. 2016; USEPA 2015; United Nations et al. 2017). The NESCS offers a practical and flexible structure and set of rules for naming ecosystem service flows as the central object of measure in the supply-and-use account.

Using the four component NESCS classification, "ecological end-products" may be a more functional label for types of SEES-EEA supply and use table row names than the label "ecosystem services." (Fig. 4b) Whether the rows in a SEEA-EEA supply-and-use table are named ecological end-products or ecosystem services however, cells in the rows sit at the intersection with columns that either designate supply of an ecological end-product by an ecosystem asset, or designate use of an ecological end-product by a particular user. Since an ecosystem service can be thought of as a transaction between nature and humans, row names must match in both the supply-and-use tables since an individual cell represents the common interaction point, or transaction (i.e., ecosystem service), between the two tables. For any one ecosystem service type defined by ecosystem assets, ecological end-products and types of (use and) users, supply must match demand for the account to balance. Quantities in cells within EEA supply and use tables should, thus, be measures of ecosystem services in that they simultaneously represent both the supply and use of a particular ecological end-product (i.e., the row).

5 Natural Capital Accounts as Tools for Ecosystem-Based Management (EBM)

Frameworks and approaches to develop ecosystem services classification and accounting systems focus on developing common, shared language and consistent approaches for identifying, assessing, and accounting of ecosystem goods and services for specific human benefits (DeWitt et al. 2020). These approaches are

(a)



(b)

ECOSYSTEM SERVICE SUPPLY TABLE

		SEEA-EEA: Type of Economic Unit (columns)	SEEA-EEA: Type of Ecosystem Unit (columns)
		NESCS: Direct User*	NESCS: Environment
SEEA-EEA: Ecosystem Services (each type or class is a row)	NESCS: Ecological End-Product	A: no data here, Economic Units cannot supply ES	B: cells indicate supply of final ES by Ecosystem Unit; per NESCS, supply of Ecological End-Product by Environment to Direct User = supply of Final Ecosystem Service
SEEA-EEA: Economic Products (rows)	no NESCS equivalent	C: cells indicate supply of economic products by Economic Units; no ES supply or use here	D: no data here, Ecosystem Units cannot supply economic products

ECOSYSTEM SERVICE USE TABLE

		SEEA-EEA: Type of Economic Unit (columns)	SEEA-EEA: Type of Ecosystem Unit (columns)
		NESCS: Direct User*	NESCS: Environment
SEEA-EEA: Ecosystem Services (each type or class is a row)	NESCS: Ecological End-Product	E: cells indicate use of final ES by Economic Unit; per NESCS, use of Ecological End-Product by Direct User = use of Final Ecosystem Service	F: cells indicate use of ES by Ecosystem Units, not people or institutions, which the UNSD et al. 2017 defines as "intermediate ecosystem services"
SEEA-EEA: Economic Products (rows)	no NESCS equivalent	G: cells indicate use of economic products by Economic Units; no ES supply or use here	H: no data here, Ecosystem Units cannot use Economic Products

Fig. 4 Alignment of SEEA EEA ecosystem services supply and use tables with the National Ecosystem Services Classification System. (Adapted from Warnell et al. 2020). (**a**) National Ecosystem Services Classification System structure (adapted from USEPA 2015). (**b**) NESCS structure (**a**) superimposed on SEEA EEA ecosystem services supply and use table structure (Fig. 1) to show alignment. ∗ Direct Users could be associated with Direct Uses by nesting or dividing columns under Direct User (Economic Unit), so an Industry, Household, or Government might have Extractive or In-Situ uses of an Ecological End-Product

designed to be used primarily by scientists but developed to be useful for resource managers and ecosystem-based management practitioners. DeWitt et al. (2020) presents a crosswalk examining ecosystem services from a structured decision-making perspective with assorted EBM frameworks, including AQUACROSS (Piet et al. 2017; Delacámara et al. 2020). Natural capital accounting efforts are directly relevant to the structured decision-making step "Estimate Consequences" in answering the question: "What ecosystem services models or data are needed to estimate consequences?" This is translatable to Step 4 in the AQUACROSS framework, "Implementation, Monitoring and Evaluation," which focuses on either assessment of current state, or the application of forecasting and scenario tools to examine/predict consequences among alternative management decisions. This potential application of natural capital accounting also maps onto the step of "Evaluating Management Measures" (Cormier et al. 2017) and "Scenarios" (Foley et al. 2013) in other EBM frameworks. The InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) is one example of a modeling approach to utilize natural capital accounting for decision-making purposes (Daily et al. 2009); for further discussion on InVEST in this volume, the reader is directed to Fulford et al. (2020) and Lewis et al. (2020).

Overall, the natural capital accounting framework provides several important tools for EBM practitioners. First, the application of structured classifications for ecosystem components, human beneficiaries (users), and a suite of flows of final ecosystem services helps EBM practitioners organize information for a given decision context. Second, a natural capital accounting framework perspective informs the effort to identify and value the final ecosystem services for EBM decision making and to track them over time in a standardized way. Finally, the structured nature of natural capital accounting frameworks lends itself to supporting important EBM steps focused on evaluating management options among a suite of EBM alternatives for a given decision context (sensu DeWitt et al. 2020). With this perspective, EBM practitioners are encouraged to learn more about natural capital accounting in general, the meticulously detailed organization of the United Nation's SEEA-EEA (United Nations 2014), and the U.S. EPA's approach to natural capital accounting using NESCS (USEPA 2015), so they might add these tools to their EBM toolbox.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Establishing a Common Framework for Strategic Communications in Ecosystem-Based Management and the Natural Sciences

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Abstract There is a need for a generalized framework and guidance for developing strategic communication efforts for interdisciplinary practitioners of ecosystem-based management to ensure engagement and communication efforts focus on effective science-society dialogue. Too often, however, developing and implementing communication strategies is viewed as separate from the research and not undertaken until the research is complete. Developing a strategic communication plan involves outlining and articulating a project's goals and objectives, identifying communication goals, defining messages, audiences, and vehicles, characterizing the different types of communication flow paths (both internal and external), and developing clear metrics that will allow for evaluating the success of the communication plan. A strategic communication matrix provides an organizational and operational structure for implementing a strategic communication plan. Here, we offer specific guidance tailored to scientists, stakeholders, and decision makers for developing strategic communication efforts. This tailored framework is then examined through a case study application in the field of ecosystem-based management.

Lessons Learned

- There is a lack of peer-reviewed literature on the development and implementation of strategic communications (focusing on *message*, *audience*, *vehicle*) with the ecosystem-based management and natural sciences literature.
- Efforts to develop and implement strategic communication plans are more effective when there is holistic buy-in from organizations.

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- It is key to have a proactive communication effort from the beginning of a scientific effort, and it is just as important to develop evaluation and feedback methods to understand the effectiveness of the specific *messages* presented to particular *audiences* using the chosen communication *vehicles*.
- An organized framework for strategic communication moves general science communication from a tactical “list of tasks” to a more comprehensive strategy to communicate the relevance of the science.
- A strategic communication matrix can be used to effectively organize *messages* for specific *audiences* using specific *vehicles* to address overall communication goals for a given effort.

Needs to Advance EBM

- There is a need to implement strategic communication frameworks, ideally from the beginning of a project life-cycle, to advance both principles of EBM and case-study applications of EBM into future studies.
- There is a need to study previous EBM communication efforts in both the peer-reviewed and grey literature to determine their effectiveness and help improve communication efforts moving forward.
- Documentation of strategic communication efforts can inform assessing the effectiveness and success of communication and be used to help inform future communication efforts.

1 Introduction

Over the last few decades, the use of strategic communication has become increasingly prominent and valued across many disciplines, including Ecosystem-Based Management (EBM). Strategic communication is a three-element process that involves specific efforts to get the proper *message* delivered using the correct form (*vehicle*) of communication, to the intended *audience*, at the appropriate time (Braus 2009). A strategic communication approach is a useful tool to tackle sensitive topics, define and prioritize target groups, and standardize communication processes (Ekebom et al. 2008). In addition to communicating information, principles of strategic communication have been used to achieve a variety of goals ranging from persuasion (Halloran 2007) to coordination (Murphy 2008) to behavioral changes (Cabanero-Verzosa and Elaheebocus 2008; Mortenius 2014).

A strategic communication approach asks decision makers and stakeholders to think holistically about their communication efforts, looking beyond information sharing to think purposefully about what they want to achieve by sharing information. For practitioners of EBM, the importance of a meaningful science-policy dialogue is paramount to the effectiveness of using an EBM approach to bring science into the discussion and decision-making process for socio-ecological decisions (e.g., Long et al. 2015; Mattheiß et al. 2018; O’Higgins et al. 2020).

A recent case study analysis of multiple EBM efforts by Mattheiß et al. (2018) concluded that better communication strategies are needed to improve effectiveness of EBM efforts. In this chapter, we demonstrate why this approach is worth taking and articulate a generalizable framework for scientists, decision makers, stakeholders, and the larger EBM community of practitioners. We argue that it is not just the field of science that should embrace a strategic communication philosophy, but the full suite of EBM practitioners, including decision makers and stakeholders, and that this approach should not begin once results are ready for dissemination or publication but be incorporated throughout the decision process.

Ecosystem-based managers who develop conservation plans, projects, and policies work to understand people's perceptions while promoting habitat conservation (Goldberg et al. 2016). A number of chapters in this text acknowledge the important role of strong communication to maximize effectiveness of EBM efforts (e.g., Myer and Johnston 2020; Williams and Hoffman 2020). Long et al.'s (2015) analysis of core principles of EBM recognise the importance of communications as part of EBM implementation (Long et al. 2015).

Stakeholder engagement and involvement, a focused effort on bringing the appropriate groups of people together to discuss, and engage, on aspects of a given decision, is recognized as another core principle in EBM (Long et al. 2015). The goals of a given communication, or dialog, effort often go beyond simple transfer of information. For example, the Uganda Nutrition and Early Childhood Development Project identified behavior change as their ultimate goal and incorporated two-way dialog with stakeholders to ensure project objectives aligned with needs and demands of their project's beneficiaries (Cabanero-Verzosa and Elaheebocus 2008). By understanding beneficiaries' attitudes, beliefs, and practices, they improved the health and nutritional status of their clients, improved stakeholders' knowledge and practices, increased a demand for community health services and schooling, and enhanced local and social capacity within the community (Cabanero-Verzosa and Elaheebocus 2008). Including audiences' attitudes and perceptions in management considerations increases the likelihood of success (Goldberg et al. 2016).

A strategic approach to communication recognizes that true communication is a two-way process and that it is important to understand what the identified audiences are looking to get out of these interactions (USFWS 2016). A strategic communication program moves beyond limitations of most common communication models (e.g., "one size fits all," "presenting everything and letting the audience decide what is important," or "thinking that communication ends once information has been presented") and specifically focuses on building a communication framework composed of three interlinked pillars—*message*, *audience*, and *vehicle*—resting on the common foundation of clearly articulated communication goals. In addition to serving as an organizational framework, the physical structure of a strategic communication plan shows an audience where they fit into the larger picture. From an EBM context, this aligns with the core principle of "recognise coupled social-ecological system" (Long et al. 2015). Additionally, a robust strategic communication plan incorporates context-specific metrics for determining communication effectiveness and success. As described later, metrics can focus on one or more communication elements to inform whether the communication goals were met. As a whole, these important elements are relevant to applied science, stakeholder engagement, and decision making.

The National Academies of Sciences, Engineering, and Medicine (NASEM) find that it is important to communicate science effectively because people need to be able to integrate accurate scientific information into their personal values and life decisions (NASEM 2017). Traditional science communication approaches often assume that the main goal is to address the deficit of scientific understanding among non-scientists (Groffman et al. 2010; NASEM 2017). This approach assumes that once an audience is educated on the topic, the work of communicators is done (Kellstedt et al. 2008; Bubela et al. 2009; Groffman et al. 2010; NASEM 2017). However, studies show that while audiences do not lack the knowledge to understand, they interpret and use information they receive in different ways (e.g., Hansen et al. 2003). The difference between a science communication deficit model and a strategic communication model is that the latter is "a purposeful use of communication by an organization to fulfil its mission" (Hallahan et al. 2007) that does not make inherent assumptions about the audience's level of knowledge about a topic and encompasses goals beyond information transfer.

Liang et al. (2018) propose the term "Strategic Environmental Communication" (SEC), which combines the concepts of environmental communication, strategic communication, and persuasion research. Practitioners and scientists can use SEC to design strategic environmental campaigns using well-designed messages and science-based strategies (Liang et al. 2018). Strategic environmental communication takes the concept of strategic communication and applies it to increase the effectiveness of environmental campaigns used to promote pro-environmental attitudes, behaviors, and investments (Liang et al. 2018). Whether focusing solely on strategic communication, or adopting the concept of SEC, practitioners and scientists can benefit from a strategic approach to communication research.

Scientific results are not always available or accessible to the public and decision makers and questions addressed by scientists do not always speak directly to stakeholder concerns. We argue that strategic communication is a way to help bridge the gap between the two (de Bruin and Bostrom 2012; Winterfeldt 2012; Jones et al. 2013). By taking a strategic communication approach throughout the lifecycle of a research project, scientists can tailor their approach to respond to stakeholder priorities or develop buy-in and participation from stakeholders and collaborators, making communication a well thought-out, long-term process rather than a reactive exercise or one constructed after the project's completion (Odugbemi and Mozammel 2005). This helps ensure that the science meets the need of management within communities, and vice versa (DeLauer et al. 2012). We argue that the use of a strategic communication approach thus can be useful in efforts to achieve another core EBM principle, "decisions reflect societal choice" (Long et al. 2015), or at the minimum, a societal-choice-informed decision adequately considers the science elements involved. By studying and researching strategic science communication, one can better understand how effective communication might influence audience interpretations and reactions, as well as how audience responses might influence next steps in science programs and management decisions (Barker 2006; Jones et al.

2013). Through an analysis of strategic science communication research to date, we have developed a generalizable framework that can be applied across a range of scientific disciplines.

There is wide consensus that natural sciences would benefit from putting more effort into strategic communication efforts (Barker 2006; Hobbs 2006; Groffman et al. 2010), but most natural scientists, decision makers, and stakeholders have not been trained in strategic communication, and traditional science communication training has typically focused on information transfer clarity after the research has been completed (e.g., Turbek et al. 2016). In order to present a framework based on analysis of the available literature and tailored guidance for facilitating a strategic communications approach for natural scientists, decision makers, and stakeholders, this chapter is organized in the following manner. First, we conduct a literature review on strategic communication in the natural sciences. Second, we discuss the eight steps of developing a generalized framework for strategic communication in the natural sciences. Third, we discuss how to develop, implement, and track a strategic communication plan through a hypothetical ecosystem services case study.

2 Literature Review

A natural sciences literature review was conducted to characterize published efforts to organize, develop, and utilize strategic communication. The search focused on examples of strategic communication within the natural sciences to identify examples that included a clearly articulated strategic communication plan or framework. The primary search was on peer-reviewed literature related to the natural sciences. Additionally, we examined other science disciplines (e.g., health, political, and social sciences), non-science examples (business, military, and education), and communications field examples (public relations, technical/social media, and customer service). Utilizing multiple literature search engines, including Web of Science, Google Scholar, and Publish or Perish, the term "strategic communication" was searched by itself, as well with the following terms: "natural resources management," "ecosystem restoration," "adaptive management," "structured decision-making," "habitat conservation," and "ecosystem based management." While "strategic communication" approaches were not explicitly identified in many EBM examples, a large EBM case study analysis by Mattheiß et al. (2018) concluded that "the better the communication strategy the likelier the demand for scientific knowledge from the social system."

Potential articles were identified and subsequently filtered by title and abstract to determine whether papers were relevant with applicable information looking for title key words such as "communication" and "strategic" or articulation of strategic communication elements in the abstract. All remaining articles were then examined for information applicable to identifying elements of strategic communication yielding articles for further, in-depth analysis. The remaining articles were examined for three criteria: key words throughout the text such as message, audience, or vehicle; a clearly articulated strategic communication plan, goal, or a framework; and, whether examples of implementing a strategic communication plan were included (e.g., case studies). A separate, additional literature analysis examined specific journals such as *Science Communication*, the *International Journal of Strategic Communication*, and the *Proceedings of the National Academy of Science* for any missing sources or materials.

Additionally, we identified examples from U.S. federal agencies and environmental non-governmental organizations (ENGOs) in the grey literature on strategic communication efforts in the natural sciences. The Google search engine was used to identify relevant natural science federal agencies and ENGOs using the keywords "strategic communication plan," and websites for each relevant source were then searched using the same keywords. The natural sciences peer-reviewed publications and ENGO sources are cited in the Appendix, along with strategic communication elements identified for each source.

Common elements and important process steps in developing a strategic communication plan were identified through this literature review. As no generalizable framework for developing a strategic communication plan was identified from this literature review, we compiled elements of strategic communication and developed a strategic communication framework presented here (Fig. 1). While none of the individual framework elements are novel within the field of science communication, developing a generalizable overall framework tailored to the needs of scientists represents an important advancement in strategic communication for EBM practitioners, and the larger suite of natural sciences in general. In addition to outlining this framework, we present one approach for operationalizing strategic communication through development of a Strategic Communication Matrix.

3 Generalized Framework Development for Strategic Communication in Ecosystem-Based Management and the Natural Sciences

Three interlinked pillars of message, audience, and vehicle, built on clearly articulated goals, form the core of a generalizable framework (Fig. 1). The first step in the generalized framework is establishing project goals and objectives (Step 1). Once project goals are identified, communication goals (Step 2) can be derived and help drive the communication process and aid in accomplishing project goals and objectives. Many natural science papers identified establishing communication goals as one of the first steps in a strategic communication plan (Bronson 2004; Dayer and Meyers 2012; Timm et al. 2016). When identifying communications goals, decision makers and stakeholders must ask themselves what they are trying to achieve with their communication efforts. All too often, decision makers and stakeholders fall into the trap of communicating information for the sake of the information itself. In the framework presented here, when setting a communication goal, the question

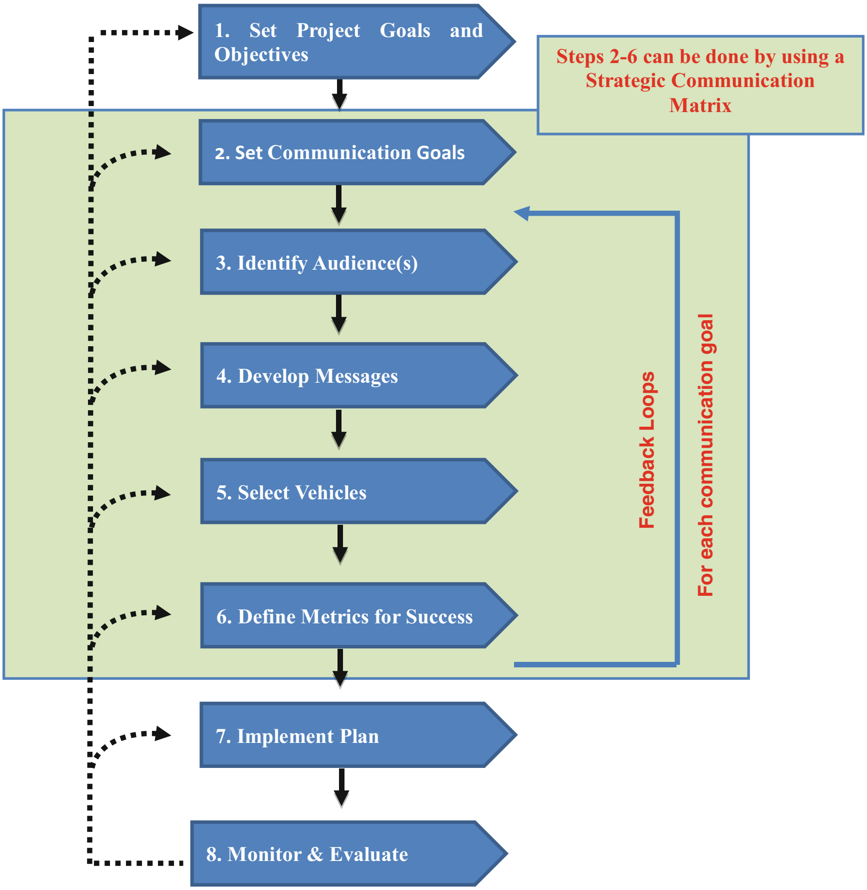

Fig. 1 Generalizable Strategic Communication Conceptual Framework using a Strategic Communication Matrix

decision makers and stakeholders must ask themselves is, "*What* are they trying to achieve?" The decision makers and stakeholders in this example may have several goals obscured under the more nebulous idea of "communicating the importance" of a science topic or decision. These goals could range from wanting to share their results or decisions with fellow stakeholders, to wanting to garner support for future work from stakeholders, to wanting to encourage external decision makers to incorporate scientific findings into their decision-making processes.

Using an example of ecosystems services to inform EBM, a more outcome-focused version of a communication goal could be "demonstrate how using the concept of ecosystem services could improve decisions related to human health and well-being." This type of goal could be identified as part of a larger effort to introduce the concepts of applying ecosystem services to inform decision making in general (e.g., DeWitt et al. 2020) as a way to present the desire to apply a similar approach in another decision context. In this text, examples include the use of ecosystem services assessment mapping for communicating future decision scenarios in Long Island (Myer and Johnston 2020), the articulation of ongoing ecosystem services consideration in EBM of the Vouga estuary (Lillebø et al. 2020), or the development of new applications of ecosystem services concepts, such as the ecosystem services gradient framework (Yee et al. 2020).

Identifying a project's communication *audience* (Step 3) includes identifying what the audience(s) already knows, what communicators would like them to know, how the audience gets information, and what the communicators would like those involved in the project to know (Groffman et al. 2010). As the communication plan is being developed and implemented, strategic communicators will gain more information about the knowledge base and values of different target audiences because a two-way communication approach, critical for effective EBM practices (Long et al. 2015), allows for feedback and evaluation. The selection of audiences, those groups who should be targeted to achieve those goals (USFWS 2016), organically follows identification of communication goals. Driscoll et al. (2012) examined a series of Long Term Ecological Reserve case studies that successfully built relationships between science and policy by focusing on engagement and distillation of results through media that met the needs of diverse audiences. Their case studies showcase programs that engage with decision makers on various issues through different communication approaches, which are determined by the audience of decision makers and the context and issues (i.e., messages) being addressed (Driscoll et al. 2012).

In an EBM context, the importance of two-way communication has been long established, including historical connections to the conservation literature, focusing on communication among relevant stakeholders as needing to be interactive and continuous. To learn more about the role of stakeholder engagement in socio-ecological decision-making contexts such as EBM, the reader is directed to examples such as Newton and Elliott (2016) and Lillebø et al. (2020). Recent advances in approaching prioritization of stakeholders in natural sciences are identified in Sharpe et al. (2020).

Developing appropriate *messages* for each goal and audience (Step 4) is more detailed and less organic than audience identification. Audiences approach a message with their own backgrounds, ideas, attitudes, and beliefs that must be acknowledged in developing successful messages (Barker 2006; Halpern et al. 2012). Developing clear messages for different stakeholders or audiences is important for promoting the responses intended by communicators and ensuring that identified goals and objectives are being met (USFWS 2016). Several natural science papers highlighted establishing effective, multi-party communication among decision makers, scientists, and community members, which led to more successful science and management (Leong et al. 2008; DeLauer et al. 2012). Message development must be done with the audiences' perspectives in mind. For example, if a communications program goal is to get recreational fishermen to comply with a licensing requirement, messaging that urges compliance should include language on how compliance helps achieve fishing community goals (e.g., licensing leads to more accurate estimates of fish populations which leads to more sustainable fishing, allowing fishing to be available to future generations).

Identifying specific communication *vehicles* (Step 5) will depend on what is known about the audience and is selected based on communication goals. The USFWS (2016) discusses selecting tactics, tools, and channels based upon the target audience, and Hobbs (2006) reminds the reader to "choose your medium carefully." The potential effectiveness of a communication vehicle can also be used to steer a science communication effort. Within the Tonle Sap Region of Cambodia, the Participatory Natural Resources Management Team recognized their target communication audiences were made up of different ethnic groups, languages, and religions, and they found meetings, workshops, posters, and environmental educators were the most effective communication vehicles for success of their communication goals (Thompson 2006). Additionally, Thompson (2006) recognized that vehicles which did not involve community members in development or interpretation process had very limited impact.

When outlining the messages/audiences/vehicles for each communication goal, *metrics of success* (Step 6) can be identified (e.g., NOAA 2009; Sea Grant 2003; NASEM 2006; USFWS 2014; NPS 2016) and used during implementation of a strategic communication plan and in efforts to monitor and evaluate the strategic communication plan. Metrics can focus on a specific aspect of communication (e.g., effectiveness of a given presentation to a particular audience), or a more comprehensive aspect (e.g., metrics that should be tracked to learn whether overall communication goals are met).

The approach for implementing a strategic communication plan should be tied back to overarching communication and project goals and may include establishing a shared understanding between science communicators and the audience. Operationally, the strategic communication plan is implemented with both content development and delivery in different forums (Step 7). Recognizing that what is persuasive for one audience may not be for another, a single project can have multiple messages, but targeted differently. Targeted messages could be as simple as having different sets of fact sheets for different regions, each highlighting regionally specific work. Additionally, different aspects of a larger project may have targeted messages for a particular audience. A Strategic Communication Matrix is one way to organize elements of a communication plan to help track and implement more sophisticated approaches to communications (e.g., more components or more complex sets of messages/audiences/vehicles) around a larger project. We present a hypothetical example in Table 1 demonstrating using a matrix to understand how different elements within the plan relate to, and rely upon, one another. More complex Strategic Communication Matrices can be operational in nature, including capturing additional metadata and tracking information.

Evaluations and feedback loops are critical to a strategic communication plan (Step 8). Interacting with intended audiences and/or stakeholders on a regular basis is important to ensure that messages are being received and having the intended impact. Further, it is important that communicators understand how audiences are responding to messages and whether adjustments are needed. These feedback loops

Table 1 Example template of a Strategic Communication Matrix

<table>
<tr><td>Project goal</td><td colspan="4">Insert Project Goal 1 here. This template can be adjusted to fit your project needs based on the identified project goal.</td></tr>
<tr><td>Project sub-goals</td><td colspan="4">Insert Project Sub-goal 1 here. This is the first sub-goal necessary in aiding and accomplishing project goal 1.</td></tr>
<tr><td>Communication goals</td><td colspan="2">Insert Communication Goal 1 here. This is the first communication goal necessary in aiding and accomplishing Sub-Goal 1. Ask "What are you trying to achieve?"</td><td colspan="2">Insert Communication Goal 2 here. This is the second communication goal necessary in aiding and accomplishing sub-goal 1. Ask "What are you trying to achieve?"</td></tr>
<tr><td>Audiences</td><td>Insert Audience 1 here. This is the first group targeted to achieve Communication Goal 1.</td><td>Insert Audience 2 here. This is the second group targeted to achieve Communication Goal 1.</td><td>Insert Audience 1 here. This is the first group targeted to achieve Communication Goal 2.</td><td>Insert Audience 2 here. This is the second group targeted to achieve Communication Goal 2.</td></tr>
<tr><td>Messages</td><td>Insert list of messages here. These messages are appropriate in aiding and accomplishing Communication Goal 1 and are specific to the targeted group identified as Audience 1.</td><td>Insert list of messages here. These messages are appropriate in aiding and accomplishing Communication Goal 1 and are specific to the targeted group identified as Audience 2.</td><td colspan="2">Insert list of messages here. These messages are appropriate in aiding and accomplishing Communication Goal 2 for both Audience 1 and Audience 2 and are specific to the targeted groups identified as Audience 1 & 2.</td></tr>
<tr><td>Vehicles</td><td>Insert a list of vehicles here that is specific to Audience 1 and their messages.</td><td>Insert a list of vehicles here that is specific to Audience 2 and their messages.</td><td colspan="2">Insert a list of vehicles here that is specific to Audience 1 & 2 and their messages.</td></tr>
<tr><td>Metrics</td><td>Insert a list of metrics for success. These metrics aid in monitoring and evaluating the success of communicating Communication Goal 1 with Audience 1.</td><td>Insert a list of metrics for success. These metrics aid in monitoring and evaluating the success of communicating Communication Goal 1 with Audience 2.</td><td colspan="2">Insert a list of metrics for success. These metrics aid in monitoring and evaluating the success of communicating Communication Goal 2 with Audience 1 & 2.</td></tr>
</table>

This matrix can be expanded or collapsed based on project needs to include as many communication goals, audiences, messages, vehicles, and metrics are necessary to aid in accomplishing a project goal are iterative in nature and ideally occurring throughout the project (e.g., getting feedback on each communication effort as it occurs). It is only by providing mechanisms to learn how a given audience responds that practitioners can refine and improve efforts (Hartman and Lenk 2001; LTER Network 2010; Okaka 2010; USFWS 2013; Ferguson 2015). The framework in Fig. 1 presents the need for both iterative development (e.g., NOAA 2016) and feedback loops throughout the strategic communication process (Table 1).

4 Ecosystem Services in an Ecosystem-Based Management Case Study

Ecosystem services, also referred to as the "benefits of nature," involve the identification and valuation of ecosystem attributes that benefit humans. Changes in *final* ecosystem goods and services (FEGS, or Final EGS), those services that directly benefit people (Landers and Nahlik 2013; DeWitt et al. 2020), can translate into changes in human health and well-being. An example Strategic Communication Matrix illustrates how to implement and track communication goals and messages (Tables 2 and 3) for an ecosystem services application in an EBM context.

In our example, the project goal is to examine and quantify how the supply and benefits of FEGS are delivered to different populations within a community as it relates to informing a specific EBM context (de Jesus Crespo and Fulford 2018). The studies within this goal involve identifying community-based preferences and values for natural resources, conducting quantitative modeling of FEGS and their benefits for human health, and exploring relationships between ecosystem services and human health, all in the context of the EBM decision at hand. This overarching project goal has multiple sub-goals, each having its own set of communication goals (Table 2). For example, the first project sub-goal focused on demonstrating the value of the concept of beneficiaries to stakeholders. To achieve this goal, several separate communications goals have been articulated, each aimed at achieving a different purpose with their respective target audiences (Table 2):

1. Clearly explain the concept of beneficiaries (those that receive the benefits provided by the ecosystem good or service) in EBM decision contexts. This goal is one of information transfer, in particular, supporting the establishment of a shared understanding of terminology.
2. Demonstrate how using the concept could influence existing EBM decision-making processes. This goal is aimed at building support for the work, getting buy-in among target audiences (in this case, the EBM stakeholders), and recognition of the coupled socio-ecological system (Long et al. 2015).
3. Understand what, if any, concepts are currently being used in place of the one proposed. This goal is aimed at expanding scientists' understanding of how their work is being received; the use of scientific knowledge in an integrated management context are both core EBM principles (Long et al. 2015).

Table 2 A hypothetical example of a Strategic Communication Matrix—project goals, sub goals, communication goals, and audience identification

<table>
<tr><td>Project goal</td><td colspan="6">Goal X: Examine how the supply and benefits of Final EGS (DeWitt et al. 2020) are delivered to different populations (beneficiaries) through studies involving community-based preferences and values for natural resources, conducting quantitative modeling of Final EGS and their benefits to human health outcomes, and providing a beneficiary perspective on advancing ecosystem to human health relationships.</td></tr>
<tr><td>Project sub goal</td><td colspan="6">Demonstrate the value of the concept of beneficiaries to stakeholders.</td></tr>
<tr><td>Communication goals (EBM principle)</td><td colspan="3">Increase awareness, understanding, and appreciation for the concept of beneficiaries in EBM decision contexts.
(stakeholder involvement; Interdisciplinarity)</td><td colspan="2">Demonstrate how using the concept of beneficiaries can change decisions/reframe EBM decision contexts.
(coupled socio-ecological system)</td><td>Elicit and understand what concept(s) and methods are being used currently in place of a beneficiary concept.
(use of scientific knowledge; Integrated management)</td></tr>
<tr><td>Audiences</td><td>*Agency leadership*</td><td>*Scientific collaborators*</td><td>*Community-level decision makers*</td><td>Scientific collaborators</td><td>Regulatory agencies/community-level decision makers</td><td>Regulatory agencies/community-level decision makers</td></tr>
</table>

The three audiences in *italics* are expanded upon in Table 2. To aid the EBM practitioner, we identified those core EBM principles from Long et al. (2015) most related to each of these example communication goals (in underlined text)

Table 3 A hypothetical example of a Strategic Communication Matrix—Audience identification, messages, vehicles, and metrics

Audiences	Agency leadership	Scientific collaborators	Community-level decision makers
Messages	• Beneficiaries are "the interests of an individual (i.e., person, organization, household, or firm) that drive active or passive consumption and/or appreciation of ecosystem services resulting in an impact (positive or negative) on their welfare." • Identifying beneficiaries allows researchers and decision makers to solicit input from groups that may be affected by changes in ecosystem goods and services, and to target beneficiary groups of interest. • Future work on beneficiaries will be used to identify how the benefits of Final EGS are delivered to different populations through EBM-related studies involving community-based preferences and modeling of Final EGS and their benefits to human health outcomes.		• Beneficiaries are "the interests of an individual (i.e., person, organization, household, or firm) that drive active or passive consumption and/or appreciation of ecosystem services resulting in an impact (positive or negative) on their welfare." • The concept of beneficiaries is useful in EBM decision contexts because it directly connects those who benefit from the environment to the ways in which they benefit
Vehicles	• Internal reports • Newsletters • Presentations to leadership	• Peer-reviewed journal articles • Conference presentations	• Presentations or webinars to community planning groups • Plain language web page
Metrics	• Sharing project results up the chain • Publicizing project results • Continued support for the research to continue	• Citations of work on beneficiaries • Publications of work using the beneficiaries concept	• Recognition and comprehension of the concept of beneficiaries • Agreement that the concept of beneficiaries would be useful in EBM-related decision making

This example is for the first communication goal identified in Table 2

In this example, the first goal focuses on providing foundational information. While the first goal does not directly focus on the science, communications may have to go beyond the bounds of the specific research in order to set the stage properly. The importance of setting the stage is underscored in the ecosystem services example, where the general public may not be aware—or supportive—of ecosystem services per se but does see the value in the concepts (and language) of the "benefits of nature" (Metz and Weigel 2013). The second goal focuses on messages specifically related to the science. The third goal gives an example of two-way communication that should be fostered in a strategic communication plan so that scientists are able to refine their scientific research and messaging to reflect the audience's needs and values. As these are done for a specific EBM context, communication goals may change as the EBM context changes.

For the first of these communication goals, three specific audience types were identified—agency leadership, scientific collaborators, and community-level decision makers (Table 2). In developing the actual plan, these audiences may be broken down further for more specific identification and targeting of messages. In our example Strategic Communication Matrix, we built upon the first communication goal in Table 2 and different messages were identified for the various audiences (Table 3). In this example, for this communication goal, one set of messages was aimed at agency leadership and scientific collaborators, and another set aimed at community-level decision makers. Although there is overlap between the sets of messages, they are differentiated by what they are attempting to achieve. When focusing on agency leadership and scientific collaborators, project communicators hope to build an understanding of the concept of beneficiaries, so it is considered in future planning efforts. With community-level decision makers, the messaging is aimed at showing how the concept of beneficiaries would specifically be useful to them in their work.

All messages are aimed at clearly explaining the concept of beneficiaries, but they are crafted with targeted audiences in mind. In this scenario, incorporating ecosystem services into EBM decision processes is done through the use of a "FEGS approach"—one that focuses on beneficiaries and their role in defining and articulating relevant ecosystem services for a given decision context (DeWitt et al. 2020). This represents only one of several ways to present ecosystem services information into the decision process. Other approaches, such as those focused on capturing the supply of ecosystem services (e.g., Lillebø et al. 2020; Myer and Johnston 2020) might have different communication goals related to identifying stakeholders or recipients of nature's benefits. A strategic communication approach can be useful for capturing information about those different approaches, focused on different target audiences, allowing for different target audiences to learn more to answer their different questions. For a discussion on the overall FEGS approach, the reader is directed to DeWitt et al. (2020); for more discussion on the value and approach of engaging with beneficiaries with this approach, the reader is directed to Sharpe et al. (2020).

Different sets of vehicles and metrics are identified for each targeted audience (Table 3). Even though the same set of messages is being targeted at agency leadership and scientific collaborators, the strategic communication plan recognizes that peer-reviewed journal articles and conference presentations that are successful at spreading their message to scientific collaborators would be far less effective in communicating to agency leadership. Similarly, success in communicating the concept of beneficiaries for consideration in future work looks very different to these two audiences. For agency leadership, successful communication is measured, in part, by continued support for the science, whereas successful communication with scientific collaborators is measured by seeing them use the beneficiaries concept in their own work on EBM decisions. When designing metrics for specific communication goals, the strategic communication plan approach asks the planner to consider what outcomes they are hoping to see overall and with each targeted audience.

It is easy to see how the complexity of strategic communication planning can increase exponentially when considering a large project in its entirety, underscoring the importance of a strategic communication plan and the value of a well-designed Strategic Communication Matrix. In our example, communicating the science on ecosystem services and its role with targeted beneficiaries within a larger EBM decision context through the use of a Strategic Communication Matrix allows advances in strategic communications for: (1) considering the full set of communication goals in a coordinated fashion; (2) finding opportunities for coordination across project goals; (3) finding ways to combine messaging associated with separate communication goals aimed at the same audience; and (4) leveraging limited resources into a more focused and outcome-driven communication effort.

5 Conclusion

Traditionally, the work of strategic communications has been done by individuals and organizations other than those conducting the research and making the decisions. We suggest that research projects and management plans benefit when scientists and decision makers proactively engage in the communication process from the beginning. This engagement allows them to focus the science and decisions in ways that resonate with targeted audiences and to share the work in ways more likely to have an impact; this aligns strongly with the core EMB principle of recognizing the coupled socio-ecological nature of the system. Although scientists and managers may have not traditionally been expected or trained to participate in communication work, the strategic communication framework laid out here (the three interlinked pillars of *message*, *audience*, and *vehicle*, resting on a common foundation of *communication goals*) can be used as a template. To be most effective, there needs to be buy-in, from an organizational perspective, on the effort required to build and implement a plan. The core EBM principle of stakeholder involvement calls for development of a strategic communication element in an EBM program.

An example in the interdisciplinary field of ecosystem services science was presented to demonstrate how to develop a clear and simple matrix to provide a visual roadmap for communication and to help coordinate efforts across a project. The use of a generalizable framework and a strategic communication matrix allows science and decision communications to be pursued using a systems-thinking approach. The EBM core principle of the use of scientific knowledge calls for practitioners to share their results or decisions with fellow scientists and stakeholders, garner support for future work from funding institutions, or encourage external decision makers to incorporate findings into decision-making processes. All scientists, decision makers, and stakeholders have project goals related to multiple audiences. The framework and matrix laid out here provide a pathway to help those scientists, decision makers, and stakeholders take a strategic approach in using communication efforts to help achieve their communication goals for their audiences. This approach expands scientific communication from the old deficit model, in which the primary goal was to make information available, towards a model that recognizes communication can be an invaluable tool in achieving science and management goals.

While the idea of strategic communication exists in the literature, the practice seems to be underutilized by scientists, decision makers, and stakeholders in general, including EBM practitioners. Those scientists, decision makers, and stakeholders who might be using the concept of strategic communication are not discussing their efforts in the peer-reviewed literature. Those that are publishing it in the literature are typically proposing strategic communication plans for future projects, rather than following up with the results of implementation and analysing whether or not communication improved. Therefore, our assumption of a lack of strategic communication plans/frameworks used among scientists, decision makers, and stakeholders may be overly simplistic. We hope that this chapter can encourage EBM practitioners to be more open/transparent about their science, decision making, management, and communication efforts in order to help the scientific community better communicate results. For strategic communication efforts that may have already taken place, but are unacknowledged in the peer-reviewed literature, a useful next step could be an effort to survey scientists, decision makers, and stakeholders to examine the question of their communication practices. This approach could give a more accurate picture of the state of strategic communication work being done by EBM practitioners.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

Appendix

Search results from both the literature (top table) and government and ENGO reports (bottom table)

Citation	Elements (G = Goals; M = Message; A = Audience; V = Vehicle; T = Two-way Communication; S = Metrics for Success)
Bronson, D. (2004). *Engaging Canadians: Building Professional Communications in Parks Canada. Communicating Protected Areas* pp. 61–68.	G, M, A, V
Day, B.A. and M.C. Monroe. (2000). *Environmental Education and Communication for a Sustainable World: Handbook for International*	G, M, A, V

(continued)

Citation	Elements (G = Goals; M = Message; A = Audience; V = Vehicle; T = Two-way Communication; S = Metrics for Success)
Practitioners. Academy for Educational Development, Washington, DC. 141 pp.	
Dayer, A. and R.M. Meyers. (2012). *Appalachian Mountains Joint Venture Strategic Communications Plan 2013–2017*. Communications Report 2012–01, Skaneateles, NY. 62 pp.	G, M, A, V, T
DeLauer, V., S. Ryan, I. Babb, P. Taylor, and P. Di-Bona. (2012). *Linking Science to Management and Policy through Strategic Communication. Advancing an Ecosystem Approach in the Gulf of Maine*. Stephenson, R.L., J.H. Annala, J.A. Runge and M. Hall-Arber (eds.). American Fisheries Society, Symposium 79:89–101.	M, A, V, T
Driscoll, C.T., K.F. Lambert, F.S. Chap–in III, D.J. Nowak, T.A. Spies, F.J. Swanson, D.B. Kittredge, and C.M. Hart. (2012). Science and society: The role of long-term studies in environmental stewardship. *BioScience, 62*(4):354–366.	G, M, A, V, T, S
Ekebom, J., J. Jäänheimo, J. Reker, M. Kindström, C. Lindblad, A. Mattisson, A. Sandstrom, and V. Jermakovs. (2008). *Towards Marine Spatial Planning in the Baltic Sea. BALANCE Technical Summary Report 4*(4).	G, M, A, V, S
Ferguson, D.B. (2015). *Linking Environmental Research and Practice: Lessons from the Integration of Climate Science and Water Management in the Western United States*. 2015 AGU Fall Meeting.	M, T
Goldberg, J., N. Marshall, A. Birtles, P. Case, E. Bohensky, M. Curnock, M. Gooch, H. Parry-Husbands, P. Pert, R. Tobin, C. Villani, and B. Visperas. (2016). Climate change, the Great Barrier Reef and the response of Australians. *Palgrave Communications, 2*:15046.	M, A, V, T
Groffman, P.M., C. Stylinski, M.C. Nisbet, C.M. Duarte, R. Jordan, A. Burgin, M.A. Previtali, and J. Coloso. (2010). Restarting the conversation: Challenges at the interface between ecology and society. *Frontiers in Ecology and the Environment, 8*(6):284–291.	A, V, T
Halpern, B.S., J. Diamond, S. Gaines, S. Gelcich, M. Gleason, S. Jennings, S. Lester, A. Mace, L. McCook, K. McLeod, N. Napoli, K. Rawson, J. Rice, A. Rosenberg, M. Ruckelshaus, B. Saier, P. Sandifer, A. Sholtz, and A Zivian. (2012). Near-term priorities for the science, policy and practice of Coastal and Marine Spatial Planning (CMSP). *Marine Policy, 36*(1):198–205.	M, A, V, T

(continued)

Citation	Elements (G = Goals; M = Message; A = Audience; V = Vehicle; T = Two-way Communication; S = Metrics for Success)
Hesselink, F., W. Goldstein, P.P. van Kempen, T. Garnett, and J. Dela. (2007). *Communication, Education, and Public Awareness (CEPA): A toolkit for National Focal Points and NBSAP Coordinators*. Secretariat of the Convention on Biological Diversity and IUCN, Montreal, Canada. 310 pp.	M, A, V, T
Kellam, D. (2004). *New Hampshire Estuaries Project: Strategic Communication Plan*. PREP publications. 17 pp.	G, M, A
Lawas, T.P., M.S.C. Tirol, V.R. Cardenas, and S.B. Jamias. (2010). Communication resource mapping for coastal resources management of Barangay Malabrigo, Lobo, Batangas, Philippines. *Journal of Environmental Science and Management, 12*(2):38–56.	G, M, A, V
Leong, K.M., K.A. McComas, and D.J. Decker (2008). Formative coorientation research: A tool to assist with environmental decision making. *Environmental Communication, 2*(3):257–273.	A, S
Long-Term Ecological Research (LTER) Network. (2010). *LTER Strategic Communication Plan: Bridging to Broader Audiences*. 55 pp.	G, M, A, V, S
Okaka, W. (2010). Developing regional communications campaigns strategy for environment and natural resources management policy awareness for the East African community. *Research Journal of Environmental and Earth Sciences 2*(2):106–111.	G, M, A, V, T
Smith, D. C., Smith, A. D. M., Dichmont, C., Steele, W., & Webb, H. (2014). *Towards a strategic relationship between CSIRO and FRDC*. Fisheries Research and Development Corporation. 36 pp.	A, V, S
Thompson, J.S. (2006). Strategic Communication in Community-Based Fisheries and Forestry: A Case from Cambodia. In G. Bessette (Ed.), *People, Land, and Water: Participatory Development Communication for Natural Resource Management*. International Development Research Centre (IDRC), Ottawa, Canada.	G, M, A, V, T, S
Timm, K., R. Hum, and M. Duckenmiller. (2016). *Using Communication Theory and Strategy to Communicate Science and Build Stakeholder Relationships in the Arctic*. 10 pp.	G, M, A, V, T
Velasco, M.T.H. (2006). *Management and Implementation of Communication Programs for Natural Resources Management in Agriculture*. In: Information and Communication for Natural Resource Management in Agriculture: A Training Sourcebook. College of Development Communication, University of the Philippines Los Banos pp. 85–96.	G

(continued)

Citation	Elements (G = Goals; M = Message; A = Audience; V = Vehicle; T = Two-way Communication; S = Metrics for Success)
Vidal, R.M. and G. Lucia (2004). *Strategic Communication Planning for a National System of Protected Areas, Mexico. Communicating Protected Areas*. Commission on Education and Communication, Switzerland and Cambridge, UK, pp. 69–86.	A, V, T
Wiggill, M.N. (2014). Communicating for organizational legitimacy: The case of the Potchefstroom Fire Protection Association. *Public Relations Review 40*(2):315–327.	G, A
Winterfeldt, D.V. (2012). Bridging the gap between science and decision making. *Proceedings of the National Academy of Sciences 110*(3):14055–14061.	M, A, T

Citations are included and strategic communication elements are identified for each of the citations. This literature review identified the extent to which strategic communication is studied and incorporated into projects within the field of natural sciences, and allowed published efforts to organize, develop, and utilize a strategic communication plan in ecosystem-based management to be characterized

Citation	Elements (G = Goals; M = Message; A = Audience; V = Vehicle; T = Two-way communication; S = Metrics for Success)
Africa Biodiversity Collaborative Group (ABCG). (2004). *Working together to help conserve Africa's biodiversity—revised communications strategy*. 13 pp.	G, M, A, V
International Union for Conservation of Nature. (2016). *Communications strategy Sri Lanka—global forest governance project: Strengthening voices for better choices*. 38 pp.	G, M, A, V, T
National Academies of Sciences, Engineering, and Medicine. (2006). *Review of the Marine Recreational Information Program (MRIP)*. The National Academies Press, Washington, DC. https://doi.org/10.17226/24640.	M, A, V, T, S
National Marine Sanctuaries. (2003). *Cordell Bank, Gulf of the Farralones and Monterey Bay National Marine Sanctuaries—strategic communication plan*. 10 pp.	G, M, A, V
National Oceanic and Atmospheric Administration (NOAA) Office of Chief Information Officer. (2009) *Communications plan: Reliable and consistent information exchange across NOAA's information technology community*. 21 pp.	G, M, A, V, S

(continued)

Citation	Elements (G = Goals; M = Message; A = Audience; V = Vehicle; T = Two-way communication; S = Metrics for Success)
National Oceanic and Atmospheric Administration (NOAA) Fisheries. (2016) *NOAA Fisheries communications implementation plan for the Pacific Islands*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Services. 8 pp.	G, M, A, V, T
National Park Service (NPS). (2016). *Wildland fire communication plan centennial edition: 2016–2020*. 18 pp.	G, M, A, T, S
Sea Grant (2003) *Positioning Sea Grant—An Integrated National Communications Plan*. 2003–2006. 24 pp.	G, M, A, V, S
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Prioritizing Stakeholders, Beneficiaries, and Environmental Attributes: A Tool for Ecosystem-Based Management

Leah M. Sharpe, Connie L. Hernandez, and Chloe A. Jackson

Abstract Successful Ecosystem-Based Management (EBM) approaches have advanced both a socio-ecological approach to systems thinking and the application of principles of structured decision making. This chapter presents a scoping tool designed to help decision makers in the early stages of their efforts by providing a transparent, repeatable, defendable approach for identifying and prioritizing stakeholders, the ways in which they use the environment (their beneficiary roles), and the most relevant environmental attributes for those uses as part of a set of decision criteria within a larger decision context. This scoping tool is a multi-criteria decision analysis approach that uses formalized criteria in stakeholder prioritization, along with the theoretical framework of the Final Ecosystem Goods and Services (FEGS) Classification System, to translate those prioritized stakeholders into the language of ecosystem services. The FEGS Scoping Tool is predicated on the idea that the decisions being made in a community can be complex, and that relevant and meaningful environmental decision criteria, let alone ecosystem services decision criteria, can be hard to identify and incorporate into the decision-making process.

Lessons Learned

- The EBM field lacks a clear pathway to prioritize stakeholders, develop a beneficiary profile, and focus management decisions on the environmental attributes most meaningful to a community.
- Transparent stakeholder prioritization provides clarity over who is included and why, facilitating a decision process that connects more directly to shared values,

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uses, and experiences, ultimately leading to increased legitimacy of the final decision.
- Development of a beneficiary profile allows managers to directly connect the ecosystem to the community's array of benefits, creating a holistic view of people's interactions to identify commonalities.
- Identification of key environmental attributes on beneficiary uses allows managers to focus their decision objectives on the most relevant metrics when evaluating tradeoffs.
- The FEGS Scoping Tool elucidates which attributes of the environment are highly valued based on the intersecting and overlapping interests of stakeholders and the beneficiaries they represent, which may lead to improved EMB design and buy-in.

Needs to Advance EBM

- There is a need to advance EBM practices through a transparent, repeatable, defendable approach for identifying and prioritizing stakeholders, the ways in which they experience the environment, and the most relevant environmental attributes for those uses.
- There is a need to identify how the FST users feel the tool influences their decision-making process and leads to improved EBM.

1 Introduction

Structured Decision Making (SDM) is a method of approaching a decision-making process in a formalized way that allows for more transparency and deliberation. Use of SDM is particularly valuable when the decisions being made are difficult ones—touching on a variety of issues, impacting a wide range of stakeholders, surrounded by uncertainty, or involving competing values (Gregory et al. 2012). Particularly difficult decisions are known as "wicked" problems. Some of the characteristics of what makes a problem "wicked" include problems that are essentially unique every time, with no clear stopping rule, with solutions that are neither "right" nor "wrong," with no clearly defined set of existing solutions, and where every solution is essentially a "one shot operation" (Conklin 2006). Ecosystem-Based management (EBM) decisions, with their need to include social, economic, and political considerations as well as complex ecological issues, are a perfect example of the type of "wicked" problem that could benefit from the use of an SDM approach (Van Bueren et al. 2003).

There are a wide range of SDM approaches (Gregory et al. 2012), but most contain some version of the same generic steps (Fig. 1). These steps are as follows:

1. Clarify the decision context—understand the context for the decision and why you are making it;

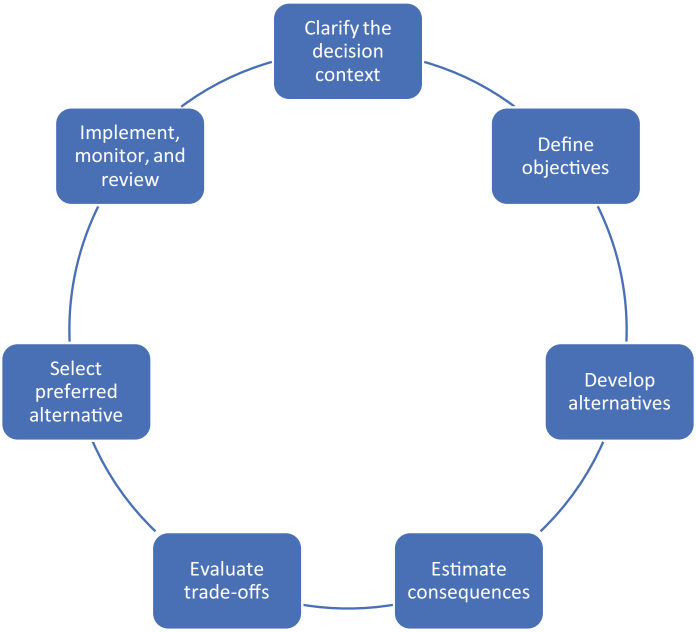

Fig 1 The decision steps in a generic structured decision-making process

2. Define objectives—clearly identify what decision makers or stakeholders value in the context of this decision and metrics for assessing how well alternatives meet those objectives;
3. Develop alternatives—identify possible alternatives for consideration;
4. Estimate consequences—estimate how well each alternative meets the decision objectives;
5. Evaluate trade-offs—examine trade-offs in how well the alternatives meet the decision objectives relative to one another;
6. Select preferred alternative—select an alternative; and
7. Implement, monitor, and review—monitor how well that alternative meets those objectives after implementation and whether any information from its real-world performance could be used to inform future decisions.

Although working through these steps can seem like an increased burden on decision makers, essentially, SDM is a formalization of the steps already being used in an *ad hoc* or unconscious fashion when making any decision (Gregory et al. 2012). For example, when deciding what to have for breakfast, explicitly clarifying the context, developing performance measures, and carrying out the remaining steps is unnecessarily burdensome. If, however, you were to explain to someone else how you decided upon your breakfast choice, you would see how you rapidly went through each step in the SDM process:

- I wanted to eat breakfast before I left for work (clarifying the context);
- I wanted something that was quick and filling (defining objectives and metrics);
- I had cereal and eggs in the kitchen (developing alternatives);
- I didn't have time to make eggs and I am trying to eat more fiber (evaluating trade-offs);
- So, I had cereal (selecting alternatives).

For more complex decisions, such as in EBM, explicitly working through the steps of a decision-analysis process has many benefits, but these steps will be worked through even without a formal process. The benefits of a formal process include improved guidance for information gathering activities, improved communication with stakeholders, increased opportunities for stakeholder engagement and involvement, improved documentation with a clear and transparent record of what happened during a decision process, and increased creativity in alternative development (Yee et al. 2017).

Decision Support Tools (DSTs) are tools that support decision-making processes. They can be powerfully effective methods for incorporating an SDM approach and increasing the transparency and repeatability of the decision-making process. The concept of decision support systems was developed in the 1970s and came out of the fields of organizational decision making and interactive computing systems (Keen and Morton 1978). By providing a mechanism for conducting one or more of the SDM steps and guiding the user through them, DSTs can facilitate incorporation of SDM thinking into a decision-making process. The tools themselves also provide users with clear and impactful ways to communicate with stakeholders (Fedra 1995).

Both the SDM approach and the use of DSTs support increased engagement with stakeholders and help decision makers identify multiple points in the decision-making process where stakeholder involvement could take place. Stakeholder involvement in making these difficult decisions is important for a number of reasons. Not only do they have the right to participate in making the decisions that impact their lives, stakeholders can also contribute valuable local knowledge that may otherwise be overlooked, and their involvement can lead to a better informed, and ideally, more legitimate (i.e., a "fair") decision process that considers representative perspectives (Cash et al. 2003; Fiorino 1990; NRC 1996). Despite its benefits, stakeholder engagement can be challenging. The process can be time consuming and expensive; it can be difficult to identify the complete set of stakeholders, and even if all stakeholders can be identified, it may not be feasible to include all of them in the decision-making process (Reed et al. 2009; Luyet et al. 2012), for one of a number of potential reasons (e.g., regulatory constraints, temporal constraints, or willingness or ability to engage). An example from this volume where not all stakeholders were fully engaged in an EBM effort is presented in O'Higgins et al. (2020).

These challenges are heightened in the complex context of EBM decisions. The effects of different management decisions on the environment can be highly uncertain and the ways in which stakeholders benefit from the environment can be easy to overlook. The concept of ecosystem services was developed to better quantify those benefits (MEA 2005). This concept was further refined as Final Ecosystem Goods and Services (FEGS), the "components of nature, directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf 2007). Although the language of FEGS provides stakeholders with a way of identifying and articulating the benefits they receive and value, the concepts are ones that require introduction and education (DeWitt et al. 2020).

The concept of ecosystem services is one that can be viewed as having a "supply" side—the goods and services the ecosystem is capable of producing—and a "demand" side—the goods and services that humans are interested in enjoying, consuming, or using (Culhane et al. 2020). Many approaches to ecosystem services focus on the question of supply (e.g., from an ecological standpoint, how large a fish population can be sustained). These questions rely on answers from the biological and ecological realms. In order to incorporate FEGS into decision making as effectively as possible, however, it is valuable to have a clear understanding of the "demand" side of that equation. This includes a clear understanding of who is benefiting from the ecosystem services (Culhane et al. 2020) as well as the specific aspects of the environment necessary to realize those benefits (DeWitt et al. 2020). These questions require answers from society and require engagement with stakeholder groups and community decision makers.

2 Stakeholders as Beneficiaries

The suite of FEGS are the attributes of the environment from which humans directly benefit, such as fish for food, property protection (i.e., protection from storm surge or wave action) provided by coastal habitats, or drinking water from a stream. The concept of FEGS is useful for decision making because it serves as a foundation for defining, classifying, and measuring ecosystem services (Landers and Nahlik 2013). The FEGS concept helps avoid ambiguity, minimizes double counting of a good or service (from a valuation perspective), bridges natural and social sciences to facilitate communication and collaboration, and is beneficiary-specific so it is directly connected to what people value (Landers and Nahlik 2013; Russell et al. 2020). Beneficiary roles are the ways in which an individual or group enjoys, uses, consumes, or interacts with some aspect of the environment. Beneficiaries are those who directly benefit from a FEGS (e.g., people who eat the fish, who own property that is protected by coastal habitats, or who drink the water from the stream) (Landers and Nahlik 2013). Defining a beneficiary helps identify the specific FEGS and connects them to human well-being (Landers and Nahlik 2013; DeWitt et al. 2020). This helps decision makers involved in EBM projects make decisions based on what matters, is directly valued, and directly benefits community members, ultimately improving human health and well-being.

Stakeholders are interested and affected parties. Stakeholder groups result from the roles the individuals within them play in society and the community. For example, a sporting club representing recreational anglers and boaters could be considered a stakeholder group. There is a lack of data on how people use the environment, and the ways in which individuals use a particular part of the environment is highly variable (e.g., one individual might choose a particular fishing spot because they went with their family as a child, while another individual might choose the same fishing spot because it is close to a car park). By identifying the beneficiary groups within stakeholder groups, decision makers are better able to identify and articulate the ways in which those in the community benefit from the environment. The individual members of the sporting club act as representatives of the club, but the members of the sporting club may benefit from the environment in many different ways and therefore would be composed of several beneficiary groups covering the different aspects of their interaction with the environment (i.e., fishing, boating, swimming, appreciation of views, etc.).

Both stakeholder and beneficiary concepts are valuable for community-level decision making and EBM applications because stakeholder groups make up the groups that should be consulted in the decision-making process and may include those most affected by the decision action. They are also the groups that community decision makers are used to thinking of and considering when making management decisions. Their roles as stakeholders, however, do not necessarily explicitly connect to how they are engaging with and benefiting from the environment. Beneficiary categories, on the other hand, are not necessarily a useful place to start when engaging in community consultation activities as groups and individuals are not used to thinking of themselves in these roles. By using both concepts, decision makers can connect how community members identify themselves within the community to how they benefit from the environment.

3 FEGS Scoping Tool

3.1 Identifying and Prioritizing Attributes Relevant to Stakeholders

Although there are repeated calls for inclusion of ecosystem services in environmental decision making (NRC 2005; Ruckelshaus et al. 2015), they are often less influential in the decision-making process than other social or economic considerations (MEA 2005; NRC 2005; Ruckelshaus et al. 2015; Yee et al. 2017). Oftentimes this is because the ecosystem services metrics that are used in decision making are ones that are easy to measure or commonly thought of (Yee et al. 2017) but may not be relevant or meaningful to decision makers or the communities they serve (Wasson et al. 2015; Yee et al. 2017). DeWitt et al. (2020) presents an argument on how the FEGS approach allows a decision maker to focus in on those ecosystem services most relevant to stakeholders and Nahlik et al. (2012) states that FEGS are more easily understood by the general public because the FEGS are determined by beneficiaries. By identifying more relevant ecosystem services, the services can be more influential in the decision-making process. Ecosystem services thinking, however, may not be intuitive for many decision makers, and long lists of potential ecosystem services (e.g., MEA 2005; Haines-Young and Potschin 2012) alone may provide insufficient guidance. Leaving ecosystem services out of the discussion entirely or selecting less relevant services can lead to decisions that omit commonly shared benefits (i.e. derived from recreational, cultural, and existence values), disconnect from what matters to people, and undermine biodiversity, human well-being, and social goals (Chan et al. 2012). For example, conservation and economic development programs in Papua New Guinea that neglected to incorporate culturally-based ecosystem services in restoration design undermined triple bottom line goals, leading to undesirable changes to impacted communities and the cultural values attached to the forest (Chan et al. 2012).

The FEGS Scoping Tool (FST) was designed to help decision makers identify and prioritize stakeholders, beneficiaries, and environmental attributes in a structured, transparent, repeatable process. The relevant and meaningful attributes can then be used to evaluate decision alternatives. The FST uses an SDM approach and the FEGS framework to identify the environmental attributes most relevant to the decision and valued by stakeholders in a transparent and structured fashion. The goal of the tool is to identify the most relevant environmental attributes for inclusion in the larger decision process so that valued FEGS are represented alongside other decision criteria. The level of stakeholder involvement in this or any other part of the decision-making process is entirely in the decision makers' hands and beyond the scope of this tool.

The FST uses a specific type of SDM, known as Multi-Criteria Decision Analysis (MCDA), with steps that mirror the generic steps of SDM. A MCDA is a formal decision-making framework that aims to represent decision goals in terms of explicitly evaluated criteria (Stewart 1992). This framework informs decision making in a transparent fashion by formalizing key criteria, explicitly stating priorities, and supporting easy replication and justification of results.

There are a range of MCDA approaches. The FST uses the method of ranking the alternatives on the sum of weighted criteria, which has been used in a variety of participatory environmental decision-making contexts (Ralls and Starfield 1995; OST n.d.). There are two main elements of an MCDA: (i) the decision alternatives that decision makers are considering, and (ii) the decision criteria used to prioritize those alternatives. In the FST, the decision criteria are a set of stakeholder prioritization criteria that were developed from the literature across a range of fields (Sharpe et al. under review) and the decision alternatives come from decision maker inputs and the National Ecosystem Services Classification System Plus (NESCS Plus) (DeWitt et al. 2020).

The benefit of using an MCDA approach for prioritizing amongst decision alternatives is that it focuses the discussion on the importance of stakeholder values rather than just the components of individual options. It does this by beginning the conversation with what decision makers are trying to achieve in meeting their goals (i.e., identifying the objective (MCDA Step 1 in Table 1)) and then identify what

Table 1 The MCDA steps in each of the three parts of the FEGS Scoping Tool. The colored boxes indicate how the output of one tier is used as an input in the next. The output from Tier 1 (red box) is used as an input in Tier 2 (red box). The output from Tier 2 (black dashed box) is used as an input in Tier 3 (black dashed box). (*modified from* Sharpe and Jenkins 2018)

MCDA Steps	Tier 1: Stakeholders	Tier 2: Beneficiaries	Tier 3: Attributes
1. Objective	Prioritize stakeholders	Prioritize beneficiaries	Prioritize environmental attributes
2. Decision criteria	Used when prioritizing stakeholder groups – supplied by the tool	Used when prioritizing beneficiaries – the stakeholder groups themselves are used as these criteria (i.e., which beneficiary groups are relevant to stakeholders)	Used when prioritizing attributes – the beneficiary groups are used as these criteria (i.e., which attributes are relevant to beneficiaries)
3. Metrics & Value functions	Used to score each stakeholder group for each criterion – supplied by the tool	Used to score each beneficiary group for each stakeholder group – this is done in the Beneficiary step of the tool when users are asked to identify those beneficiary categories found within each stakeholder group	Used to score each attribute for each beneficiary group – this is done in the Attribute step of the tool when users are asked to identify attributes of concern for each beneficiary group
4. Alternatives	Stakeholders identified by tool users	Beneficiary list from the FEGS Classification System and NESCS Plus	Attribute list from the FEGS Classification System and NESCS Plus
5. Weighting	Done by users in the first step of the tool	Stakeholder MCDA values from the output of Tier 1 are used as weights in this step	Beneficiary MCDA values from the output of Tier 2 are used as weights in this step
6. Score alternatives	User input at the Stakeholder step	User input at the Beneficiary step	User input at the Attribute step
7.Calculate value	Output at Stakeholder step	Output at Beneficiary step	Output at Attribute step

criteria are necessary for meeting them (Step 2). It then uses metrics (Step 3) to determine as objectively as possible how well each decision alternative (Step 4) meets those objectives (Step 6). This is separate from the question of the relative importance of those criteria (Step 5) to the decision makers. When choosing a car, for example, two people may disagree on whether storage capacity is more or less important than fuel efficiency, but they can easily agree on how each of those options score on those criteria. Together, how well each alternative scores on the criteria and the relative importance of each criterion to the decision makers are used to determine the MCDA value of each alternative and their overall prioritization (Step 7).

3.2 *Tiers of the FEGS Scoping Tool*

The FST has three tiers: (1) a stakeholder prioritization; (2) development of a beneficiary profile; and (3) identification of key environmental attributes, with each tier feeding into the next (Fig. 2a). Because the FST has three objectives—prioritizing the stakeholders, the beneficiaries, and the key attributes—each of the MCDA steps is run through three times in succession in a tiered MCDA approach,

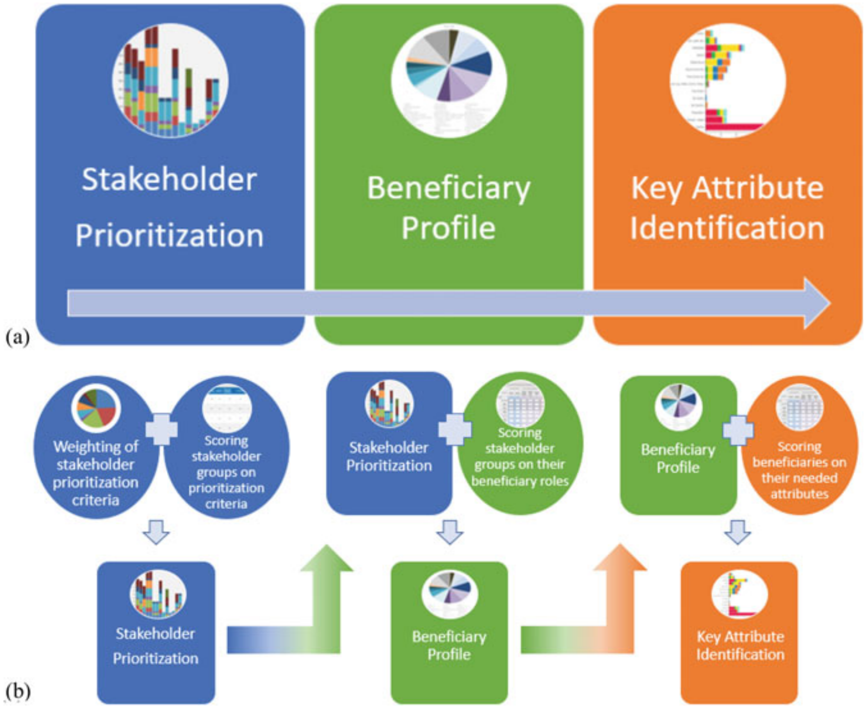

Fig. 2 Overview of the FEGS Scoping Tool. The FST has three tiers (a) —stakeholder prioritization, development of a beneficiary profile, and identification of key environmental attributes. Each tier feeds into the next. MCDA weights and scores decision alternatives on a range of criteria and uses the combination of weights and scores to prioritize those alternatives. In this schematic (b), circles are the user inputs and squares are the tool outputs. In the first tier, users weight and score the prioritization criteria to rank the decision alternatives (the stakeholder groups). In the second tier, the stakeholder prioritization (first tier output) is used as the weights and users score each of the stakeholder groups as to their beneficiary roles. These weights and scores are used to prioritize the decision alternatives (the beneficiary groups). In the third tier, the beneficiary profile (second tier output) is used as the weights and users score each of the beneficiary groups as to key attributes needed to realize those benefits. These weights and scores are used to prioritize the decision alternatives (the environmental attributes)

with the output from Tier 1 feeding into Tier 2 and the output from Tier 2 feeding into Tier 3 (Fig. 2b and Table 1). Specifically, the output from the stakeholder prioritization (Tier 1, Step 7) is used to weight the influence of those stakeholder groups in the development of the beneficiary profile (Tier 2, Step 5). Continuing that approach, the output from the beneficiary profile (Tier 2, Step 7) is used to weight the influence of those beneficiary groups in the identification of key environmental attributes (Tier 3, Step 5).

The FST was designed to be relatively simple for decision makers to incorporate in their existing decision-making processes. It does not require decision makers to collect specific data about the stakeholder groups beyond knowledge of the community and familiarity with the stakeholder groups. The more iterative and participatory a process is structured, the more likely the process will lead to results that are defensible and well-founded. The intent of the tool is to provide a simple, transparent process for prioritizing stakeholder groups and letting that inform a prioritization of the benefits they receive from the environment and the elements of the environment necessary for receiving those benefits. It is anticipated that tool users will often be the decision makers, but this is not necessarily the case. Non-decision makers, such as those interested in evaluating or analyzing a decision made by others, might find the FST valuable as well.

3.2.1 Stakeholder Prioritization

During the stakeholder prioritization, decision makers are asked to review and weight the stakeholder prioritization criteria found in Table 2. This is a question of values. In this step, decision makers must ask themselves which of these decision criteria are most meaningful to them when looking at the stakeholder groups they are prioritizing. Criteria weighting is a key element of MCDA and the FST. Priorities

Table 2 Stakeholder prioritization criteria used in the FST

Stakeholder prioritization criteria	Definition
Level of interest	The amount of interest a stakeholder group has in the decision-making process or the decision outcome
Level of influence	The amount of influence a stakeholder group has over the decision-making process
Magnitude of impact	The degree of potential impact to the stakeholder group as a result of the decision
Probability of impact	The likelihood of potential impact to the stakeholder group as a result of the decision
Urgency/temporal immediacy	The degree to which a stakeholder group would like to see a decision made or an action taken
Proximity	How frequently a stakeholder group comes into contact with the environment for which a decision is being made
Economic interest	Whether a stakeholder group's livelihoods or assets could be impacted by the decision outcome
Rights	Whether a stakeholder group has legal, property, consumer, or user rights associated with the decision-making process, the decision outcome, or the environment for which the decision is being made
Fairness	Whether the exclusion of a stakeholder group from the decision-making process would lead to the process being viewed as unfair by the community
Underrepresented/under-served populations	Whether a stakeholder group includes any underrepresented or underserved populations

will differ from community to community as well as among groups within a community. For example, if a business development group is using this tool to incorporate FEGS into their decision making, they are likely to weigh the criterion of economic interest substantially higher than a non-profit group focused on social justice. In this step, decision makers are asked to identify the criterion most important to them and give that criterion a weight of 100. After that most valued criterion has been identified, all other criteria are weighted relative to that criterion on a 0–100 scale. Those criteria that are not considered by decision makers should be given a weight of 0. The FST provides a visual aid to allow decision makers to see the relative impact each criterion will have on the prioritization process.

Current methods of stakeholder analysis in environment decision making focus on stakeholder identification (who is/should be considered a stakeholder), categorization (often focused on distinguishing groups based on level of engagement), and relationship analysis (using tools like social network analysis to understand how the different groups relate to and influence one another). Stakeholder prioritization is discussed in the fields of business, management, and public relations, but the concerns of researchers in those fields are imperfectly analogous to environmental decision making. Therefore, a proposed set of ten stakeholder prioritization criteria was developed specifically for the field of environmental decision making and this tool (Table 2) (Sharpe et al. under review).

These criteria are not entirely independent from one another; however, each captures some element that has been found to be useful or important in stakeholder analyses and it is critical to include all criteria that could be relevant for decision makers (Sharpe et al. under review).

Once the criteria have been weighted, the decision makers are asked to identify all stakeholder groups relevant to the decision context. There is no step or process embedded within the tool itself that ensures that all possible stakeholder groups are being included in the decision process. That is, this tool, just like any stakeholder engagement effort, relies upon the good faith efforts of decision makers to cast a wide net when it comes to stakeholder identification and inclusion. However, by having a record of which stakeholder groups were considered and by using the FST in a transparent and iterative fashion, there will be opportunities for decision makers and their constituents to identify missing stakeholder groups and include them in the process. After the stakeholder groups have been identified, users will then score them on each of the decision criteria. The tool itself lays out specific scoring metrics for each criterion with the goal of making the scoring as objective as possible. Although different decision makers could disagree on how important economic interest is in making a stakeholder group a priority, it should be clear whether (or not) a given group has an economic interest in the decision.

Once the decision criteria have been weighted and the decision alternatives (e.g., the stakeholder groups) have been scored on those criteria, the FST calculates a value for each alternative by summing the weighted scores for each alternative. The value, $y(i)$, of an alternative, i, is calculated as:

$$y(i) = \sum\nolimits_{m=1}^{M} w_m z_i$$

where M is the number of possible metrics for which i can be scored, w_m is the weight given to each criterion, and z_i is the score of alternative i on metric m. The value, $y(i)$, is then normalized, $n(i)$, to a 0—100 scale by dividing $y(i)$ by the sum of all weights:

$$n(i) = \frac{y(i)}{\sum\nolimits_{m=1}^{M} w_m}$$

This results in an output of a prioritized list of stakeholders with each group given a "value." This value is only meaningful in describing the relative differences in priority for a collection of stakeholders that have been evaluated in a single exercise. It provides comparative information about the relative priority of different groups but has no meaning beyond that.

3.2.2 Beneficiary Profile

In the second part of the tool, users develop a beneficiary profile of the decision context to better understand the ways in which the community benefits from the ecosystem under consideration. It helps decision makers take a more holistic view of various groups' interactions with the environment and identify commonalities among them. In this step, users are asked to segment each stakeholder group into its component beneficiary groups by percentage, for a total of 100%. Once this has been completed, the FST will once again calculate a "value" for each beneficiary group using the same calculations as in the stakeholder prioritization. In this calculation, however, the output values, $n(i)$, from the stakeholder prioritization are used as the weights, w_m, for the beneficiary profile.

The beneficiary categories in this step come directly from the Final Ecosystem Goods and Services Classification System (Landers and Nahlik 2013) and the National Ecosystem Services Classification System Plus (NESCS Plus) (DeWitt et al. 2020). At this step, tool users should ask themselves how each stakeholder group benefits from, uses, or values the ecosystem under consideration. For example, a stakeholder group consisting of a fishing club could benefit from the ecosystem through a waterbody that can be navigated by their fishing boats (the "Boaters" beneficiary), fish that can be caught (the "Anglers" beneficiary), and a pleasing view when traveling to and from the fishing site (the "Experiencers/Viewers" beneficiary).

3.2.3 Key Attribute Identification

In the final portion of the FST, users build upon the previous steps to identify the key environmental attributes of the decision context. These key environmental attributes are those attributes that are necessary for the stakeholders to receive the benefits that they value. In this step, users are asked to identify, by percentage, the environmental attributes that are necessary for each beneficiary group to succeed in using and benefiting from the environment. After the environmental attributes of concern have been identified for each beneficiary group, the FST will calculate a "value" for each attribute using the same calculations as in the stakeholder prioritization and beneficiary profile. In this calculation, however, the output values, $n(i)$, from the beneficiary profile are used as the weights, w_m, for key attribute identification. This step allows the user to see what environmental attributes are relevant for evaluation of decision alternatives and provides a clear explanation of why those attributes are relevant.

The list of attributes used in the FST was developed for NESCS Plus (DeWitt et al. 2020). To continue the earlier example, the "Anglers" beneficiary would likely care a great deal about "Charismatic fauna" (i.e., fish that are of interest to anglers) and "Edible fauna" (i.e., fish that are safe to eat). Attributes related to the fuel, fiber materials, or fungal communities, for example, would likely not have any impact on their ability to realize this benefit.

The prioritized set of environmental attributes are the set that should be considered when evaluating different management options. The combination of prioritized beneficiaries and attributes provides the decision makers with guidance on the appropriate metrics to use when evaluating those options. The metric(s) used to assess the management options should be ones relevant to the beneficiaries that care about that attribute. The NESCS Plus has released a report on the development of national metrics and indicators for a number of ecosystem/beneficiary/attribute combinations and would provide useful guidance for developing sets of metrics for local-scale decisions (DeWitt et al. 2020; US EPA under review).

3.3 Using the FEGS Scoping Tool

The FST was designed to be used at an early stage of decision making, when decision makers are aware a decision needs to be made, but before any actions are taken (e.g., step 1 of the SDM process—clarify the decision context). Once those key environmental attributes have been identified, they can be included as objectives for the decision under discussion (e.g., step 2 of the SDM process—define objectives). These FEGS-related objectives can be used later in the decision-making process alongside other, non-environmental attributes, such as cost of the alternatives or job creation associated with each alternative, to estimate the overall consequences for each alternative (e.g., step 4 of the SDM process—estimate consequences). The FST itself is not designed to work through the entire SDM process for a given management decision. Rather, it is designed for use in step 1 and for its outputs to be applied in subsequent steps. Beyond this, the FST provides no additional guidance in conducting those steps, or in making a final decision.

The FST was designed to be used during the scoping phase for any decision with an environmental context by community-level decision makers who are involved in articulating the overall decision objectives and choosing amongst various decision options. Ideally, the FST is used in a participatory, iterative fashion with input from stakeholders, but can be applied in a variety of ways depending on the community's existing decision-making processes and requires no technical expertise or data collection beyond familiarity with the community and its stakeholder groups.

Decision makers prioritize stakeholders, either consciously or subconsciously. This tool formalizes and records the stakeholder prioritization process and makes those priorities transparent. The stakeholder prioritization results are then used to systematically identify environmental attributes most relevant to the prioritized stakeholders. The FST is predicated on the idea that the decisions being made in these communities are complex and that relevant and meaningful environmental decision criteria, let alone ecosystem services decision criteria, can be hard to identify. Thus, the use of a structured stakeholder prioritization approach results in the ability to provide a transparent, repeatable, and defendable approach for selecting the more relevant environmental attributes for use as decision criteria in that larger decision.

4 FEGS Scoping Tool Applications for Ecosystem-Based Management

Along the coast of Oregon (Fig. 3), communities are inextricably interconnected with estuarine wetland ecosystems and upland watersheds. The region provides highly viable industries (e.g., dairy, agriculture, timber, and fishing) and recreation centered around natural resources, and communities affect ecosystem functions through recreation, resource use, urban development, and dredging for shipping and port access. Many small communities along the Oregon coast have experienced an increase in retirement age migration and are seeing the tourist service industry, residential and resort developments, charter fishing, and whale watching increase in importance to the local economy while extractive industries in timber and fishing have declined (Radtke and Davis 1994; Ackerman et al. 2016). Additionally, historic natural disasters and climate change are straining the ability for communities to solely depend on these resources and seek support in mitigating adverse impacts and restoring ecological functions.

Because the health of Oregon's coastal habitats is vital to the safety and well-being of its human and wildlife communities, federal, state, non-governmental organizations, academic, and private institutions in Oregon have invested heavily to research and restore impacted ecosystems. We are going to walk through a hypothetical example of using the FST at the beginning of a wetland restoration project to more clearly illustrate the FST process. This example is a simplified version of the prioritization that might happen at a site that is being restored back

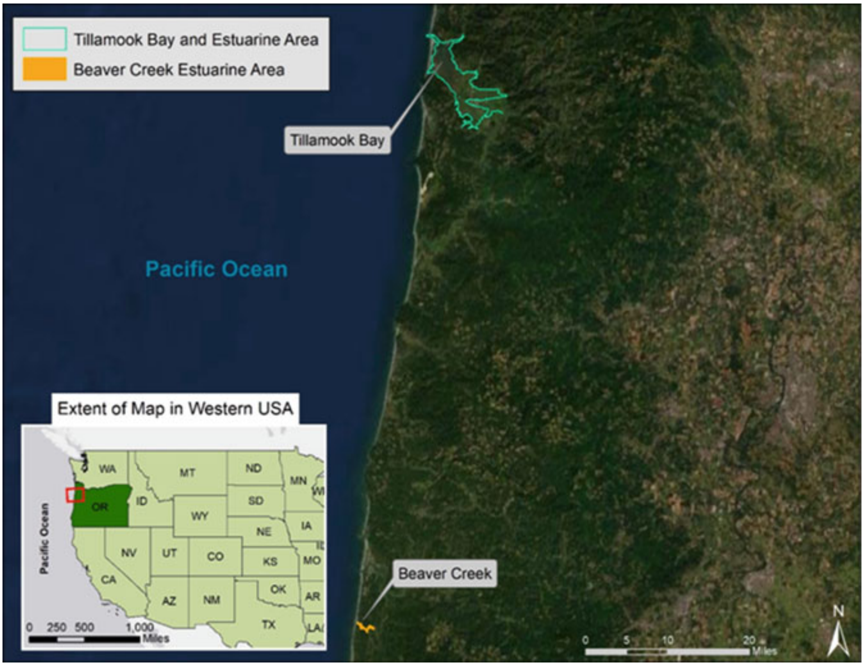

Fig. 3 Location of Tillamook Bay and Beaver Creek and their tidally influenced wetland areas. Extent area data layer was retrieved from Oregon Spatial Data Library on 11/14/2019, the estuaries data set from Adamus Resource Assessment Inc. (2004)

to a wetland, and does not include all likely stakeholders, beneficiaries, or attributes that would be included in a real ongoing project.

In the first tier of the tool, Stakeholder Prioritization, decision makers must assign weights to each criterion and then score the stakeholder group for those weights. When assigning weights, the decision makers might highly weight criteria such as Level of Influence and Level of Interest, if they are interested in making sure that influential and interested groups were prioritized and they might highly weight Underrepresented and Underserved Representation if they are interested in environmental justice concerns. Decision makers might also give Proximity moderate weight if they are interested in prioritizing those groups most likely to come into contact with the area and give Economic Interest moderate weight if they are interested in prioritizing those who may see an economic impact, either positive or negative, as a result of the project. For the purposes of this example, all other criteria are being considered unimportant to the decision makers and given a weight of zero. Figure 4 shows the user input form for the weights as well as the visual representation of the weights relative to one another.

After the weights are input into the tool, users identify the relevant stakeholder groups and score them, using provided metrics, on those criteria. In this case, some of the stakeholders sitting at the decision-making table might include Funding

Color	Criterion	Weight
	Magnitude & Probability of Impact	0
	Level of Influence	100
	Level of Interest	100
	Urgency & Temporal Immediacy	0
	Proximity	50
	Economic Interest	50
	Rights	0
	Fairness	0
	Underrepresented & Underserved Representation	100

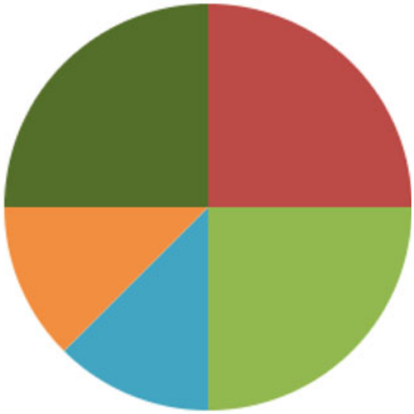

Fig. 4 The FST user input form for the weights (left) and the visual representation of the weights relative to one another (right)

Organizations, an NGO Conservation Trust, County Government Agencies, and Neighbors & Landowners. The metrics vary from criterion to criterion. In the case of Underrepresented and Underserved Representation, users are asked a yes or no question as to whether the stakeholder group contains any underrepresented or underserved groups. In this hypothetical example, only the Neighbors & Landowners group does. For the criterion of Proximity, however, users select from a range of scores based on how frequently the stakeholder group is in contact with the area in question or adjacent spaces. Once users have input the weights and the scores, they are combined to produce the stakeholder prioritization (Fig. 5). In this example, the Neighbors & Landowners are likely to include underrepresented or underserved groups and therefore have a higher prioritization given that this criterion was weighted higher than Economic Interest, which the other three stakeholders have. The County Government is prioritized the lowest because this stakeholder has less of an influence over the decisions being made than the other three stakeholders. This

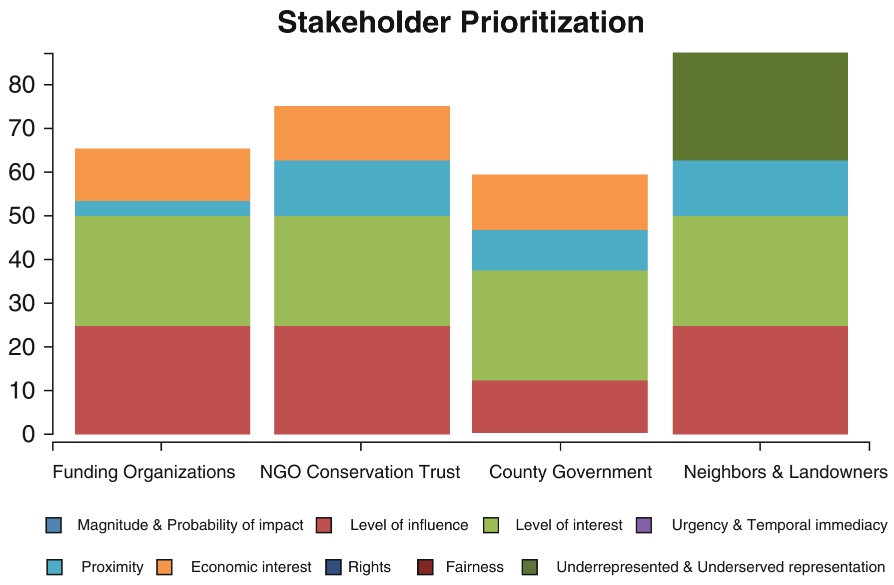

Fig. 5 Tier 1 output of the FST: Stakeholder Prioritization

would be a good time to communicate with the stakeholders and ensure that a key stakeholder group was not left out in the process.

In the second tier of the tool, Beneficiary Profile, users are asked to identify the ways in which each stakeholder group is benefiting from the area. In this example, decision makers might find that the Neighbors & Landowners group, while principally benefiting through owning the area or land adjacent to it, also benefit through recreational opportunities such as fishing (angler beneficiary category) and hiking (experiencer/viewer beneficiary category). These beneficiary scores are then combined with weights arising from the stakeholder prioritization to lead to a beneficiary profile (Fig. 6). The beneficiary profile allows decision makers to identify commonalities among stakeholder groups. In this case, we can see that all four stakeholder groups have common ground in that they all care about the continued existence of a healthy wetland ecosystem.

In the third tier of the tool, Key Attribute Identification, users are asked to identify the ecosystem attributes necessary for each beneficiary group to receive their benefit. In this case, a recreational angler beneficiary would likely care about water quality, water quantity, flora community, fauna community, edible fauna, viewscapes, and the ecological condition of the site. These attribute scores are then combined with weights arising from the beneficiary profile to lead to identification of key environmental attributes (Fig. 7). This final output allows decision makers to identify the key environmental attributes that should be considered and measured when contemplating restoration options for the site. It is unlikely that the entire suite of attributes can be considered, but if the user focuses on the attributes related most directly to sustaining the availability of the FEGS that anglers care about, it is more likely

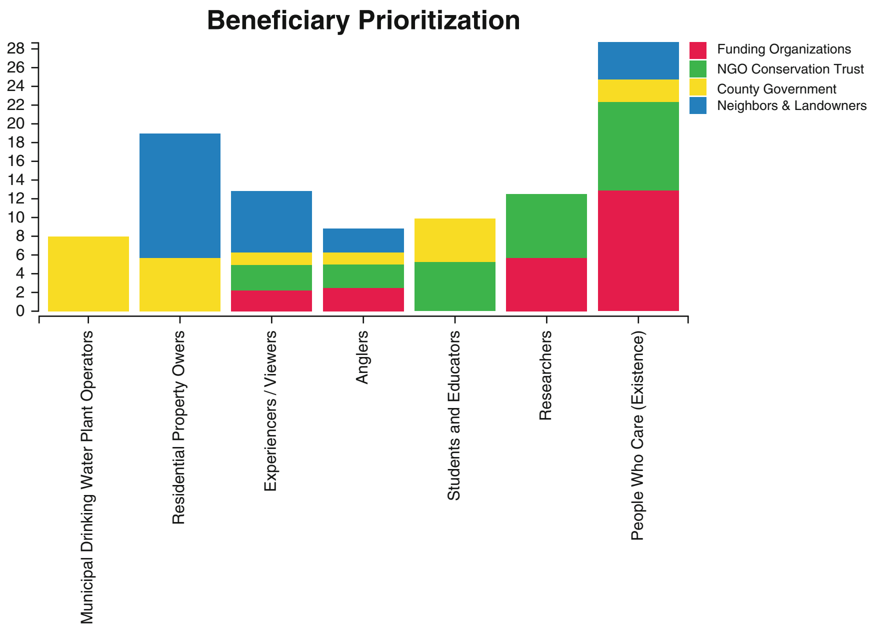

Fig. 6 Tier 2 output of the FST: Beneficiary Profile. Depicted as a bar chart showing the influence of each stakeholder group on the beneficiary group

that the ultimate decision will include ecosystem service metrics relevant to the stakeholders and the ways in which they benefit from the site.

While this example is hypothetical, there are numerous restoration efforts that have taken place in estuarine systems along Oregon's Northern Coast, such as the Southern Flow Corridor (SFC) project and the Beaver Creek watershed restoration project (Fig. 3). The SFC project was implemented by the Tillamook Estuary Partnership (TEP), one of EPA's 28 National Estuaries Programs that was established to conserve and restore estuaries and watersheds in Tillamook County (TEP 2019). The Tillamook Estuary Partnership first implemented a Comprehensive Conservation and Management Plan in 1999, and work was done to encourage broader community and stakeholder participation to identify values relevant to the estuary, prioritize conservation goals, and specify resource management actions (Gregory and Wellman 2001). The focus of the SFC project was restoring tidal wetland habitats and ecological function at the deltas of the Wilson, Trask, and Tillamook rivers (SFC 2019). The SFC produced a Project Effectiveness Monitoring Plan in which stakeholders and decision makers outlined four flood mitigation and restoration goals and expected ecological and economic benefits (Brophy and van de Wetering 2014). Interventions included a conditions assessment and various infrastructure changes to ditches, levees, dikes, floodgates, and buildings (Brophy and van de Wetering 2014). The project is currently in the monitoring phase to determine if the flood attenuation and ecological function goals were met (Brown et al. 2016;

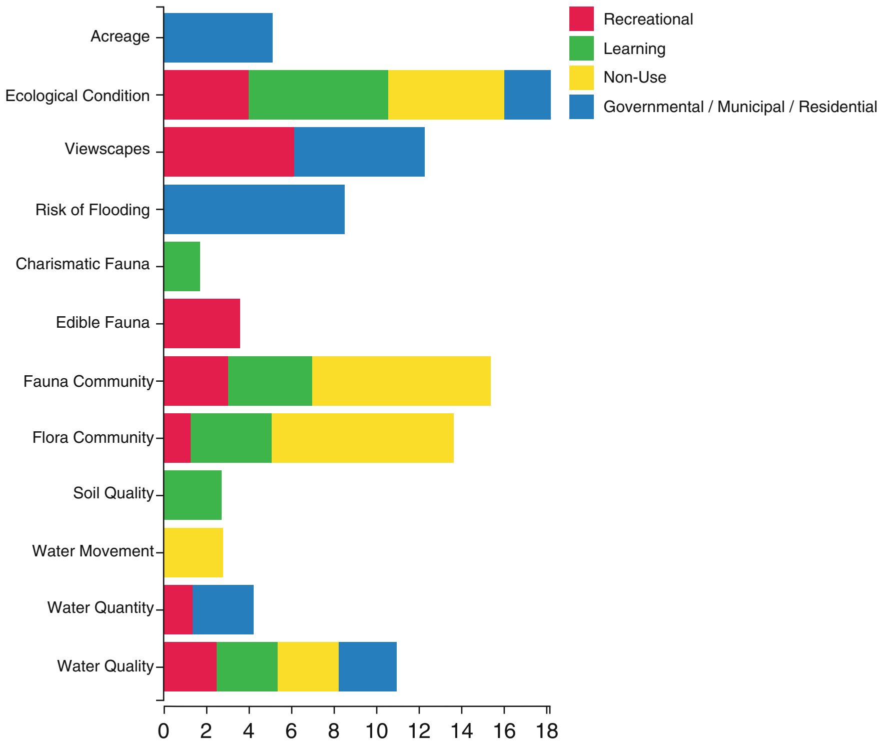

Fig. 7 Tier 3 output of the FST: Environmental Attribute Prioritization

SFC 2019). The conservation and restoration being done in the Beaver Creek watershed addresses critical watershed restoration issues (The Wetlands Conservancy 2018) and aims to simplify instream habitat, move roads that are too close to the stream, and increase the number of riparian trees and shrubs (TEP 2019).

Given that restoration is so important to the state of Oregon, it is important to assess the effectiveness of restoration relative to the achievement of a project's restoration goals that are meaningful to adjacent communities. By using locally-relevant ecosystem metrics and indicators derived from stakeholder and beneficiary goals, the local ecology, and the NESCS Plus, restoration effectiveness can be evaluated by assessing a restoration site based on its capacity to produce and deliver priority nature-derived benefits (i.e., FEGS). The set of FEGS will vary with, and be dependent on, community values and priorities. The FST is important to this process because it can be used in the initial stages of the restoration to identify priority FEGS, beneficiaries, and benefits for a particular restoration site. For example, the FST could be a useful approach to use at a nascent project site along the Tillamook River where a parcel of land that is currently farmland has been acquired and designated for restoration and enhancement. Dairy farms, forest, and rural homes surround the site, along with a road that often floods from the river. The driving interests of stakeholders at the site include the construction of flood mitigation structures, habitat restoration to support salmonid growth, waterfowl hunting, and a designated shooting range, while aligning with the county's policy of no-net farmland loss.

Prioritizing stakeholders in restoration projects such as this is necessary because of potential conflicting stakeholder interests. Although there is no mechanism in the FST to ensure that all stakeholder groups are considered and represented, the SDM approach it builds on supports the inclusion of as wide a range of stakeholder groups and interests as possible. All engagement efforts rely on the good faith of the decision makers. This tool is one way for them to be transparent about who is considered, who is most relevant to the decision and the community impacted by it, and why ultimately not everyone's interests may be feasibly incorporated into decisions and goals. Additionally, an iterative and participatory approach can be taken in the stakeholder characterization step to allow for additional stakeholders to be identified and incorporated into the process. Using the FST to identify a broad range of potential stakeholders functions as a documented structure to prioritize stakeholders, beneficiaries, and ecological attributes and can help site planners decide on which type of site to target, on which FEGS to focus, and evaluate available options towards achieving restoration goals—whether that be salmon for fishing, waterfowl for hunting, or habitat for flood protection. From there, it can be determined how local FEGS metrics and indicators may be used to assess progress towards achieving those desired benefits.

5 Conclusion

Ecosystem-based management is a field requiring complex tradeoffs for its decision makers. In addition to the difficulty and uncertainty surrounding ecosystem forecasting, weighing socioeconomic concerns against environmental ones can be challenging. The concept of FEGS helps provide managers with language that more directly connects environmental concerns to the community's values. Using an SDM approach like the FST provides a clear pathway to prioritize stakeholders, develop a beneficiary profile, and focus management decisions on the environmental attributes most meaningful to the community, all of which help facilitate effective communication of the value of proposed work. It is rarely feasible to include all possible stakeholders in a decision-making process. Stakeholder prioritization provides clarity over who is included and why, enabling managers to be transparent about the perspectives being given weight in the decision process and, ultimately, leading to increased legitimacy (as defined as a "fair" decision process that considers representative perspectives; Cash et al. 2003) of the final decision. Development of a beneficiary profile allows managers to directly connect their community to the array of uses it has for the area under discussion. This allows managers to find common goals across beneficiary groups as well as potential points of contention. The holistic view of these uses also lowers the chance that valued uses are overlooked and then impacted by the decision. Identification of key environmental attributes of concern based on these uses then allows managers to focus their decision objectives on the most relevant metrics when evaluating tradeoffs. Ultimately, inclusion of the FST in an EBM process could lead to identification and consideration of more relevant ecosystem service metrics as a result of a more deliberate approach to stakeholder engagement and an improved understanding of community priorities.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Linkage Frameworks: An Exploration Tool for Complex Systems in Ecosystem-Based Management

Leonie A. Robinson and Fiona E. Culhane

Abstract A key barrier to achieving Ecosystem-Based Management (EBM) is dealing with complexity of social-ecological systems (SES). SES incorporate ecological, social and economic factors that interact within and between each other. Carrying out quantitative analyses to aid decision making in these systems is often too complex and/or limited by data. We describe a complementary approach, the use of Linkage Frameworks, that can be used to explore EBM. Linkage frameworks are essentially networks of elements or nodes found in a system, with links representing the interactions between those nodes. In an EBM context, nodes might include human activities, their pressures, biodiversity components, the ecosystem services supplied in that ecosystem, and the users or beneficiaries of the activities and services supported. Interactions could highlight, for example, which activities introduce which pressures, which biodiversity components are linked to which human activities through their pressures, and which ecosystem services are supplied by each biodiversity component. This approach can help to structure systems conceptually, allowing consideration of complex systems in decision making and facilitating communication between, for example, scientists, ecosystem managers and stakeholders. We discuss the strengths, assumptions and limitations of the tool, drawing on examples from aquatic ecosystems across Europe.

Lessons Learned

- Although dealing with the complexity of SESs can be challenging, there are advantages to persevering. We show, using a linkage framework approach, that retaining the complexity of the system in analyses to inform decision making, can provide different perspectives on EBM questions.
- Linkage Frameworks are specified for specific systems and specific issues or problems. It is important that users are aware of this as the specification will affect the understanding of the system.

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- Decisions for EBM cannot be made on the LF approach alone. However, exploration of LFs can be used to highlight where complementary detailed, data driven studies are needed.

Needs to Advance EBM

- Effort needs to be made to include analyses that retain the complexity of the SES so that we can move forward in understanding indirect links and unintended consequences of EBM management decisions. These do not need to be data driven but do need to be supported by detailed, data driven studies to support the assumptions made.
- Non-linearity needs to be included in the risk assessment approaches that are supported by weighted Linkage frameworks. To date much of the existing work assumes that risk accumulates linearly as you 'add up' components that act together within a SES. In reality some cumulative effects may be antagonistic, whilst others may be synergistic, and the type of interaction may also vary in space due to environmental conditions.

1 Introduction

Complexity is a defining issue in dealing with environmental problems, sometimes called 'wicked' problems, where diverse interests and conflicts from ecological, economic and social elements meet (Rittel and Webber 1973; Game et al. 2014). Dealing with complexity in Ecosystem-Based management (EBM) is an ongoing barrier at all stages from identifying drivers of ecological issues to implementing management measures (see G. Piet et al. 2020 for further discussion of this complexity). While complexity is sometimes seen as a problem that needs to be solved or avoided in EBM, systems thinking tells us to embrace it (Mitchell 2009). Indeed, by retaining complexity in our assessments, we can discover emerging properties of social ecological systems (SES). This can help to overcome the traditional problems of environmental management, where narrow thinking can lead to poor decision making and unintended consequences that are often the result of these emerging properties (Yodzis 2001).

Retaining complexity in EBM does have its limitations. Data and resource needs are often much higher and uncertainty in the outcome can limit interpretation. In order to move forward pragmatically with a systems approach, it is necessary to start at the broadest level with a description of the entire system that includes both high level, often qualitative or semi-quantitative, analyses with detailed analyses of specific parts of the SES, as has been described in many chapters of this volume (e.g. see Elliott and O'Higgins (2020)).

In this chapter we use the construction of Linkage Frameworks (LF) to describe SESs, defining all relevant SES parts and the links between them. We explain what an LF is and explore how it can be used to underpin Ecosystem-Based Management (EBM). Linkage frameworks are essentially networks of elements (or nodes) found in a system, with links representing the interactions between those elements. In an

EBM context, nodes might include human activities, their pressures, biodiversity components, the ecosystem services supplied in that ecosystem, and the users or beneficiaries of the activities and services supported. Interactions could highlight, for example, which activities introduce which pressures, which biodiversity components are linked to which human activities through their pressures, and which ecosystem services are supplied by each biodiversity component. This approach can help to structure systems conceptually, allowing consideration of complex systems in decision making and facilitating communication between, for example, scientists, ecosystem managers and stakeholders. We will discuss the strengths, assumptions and limitations of the tool, drawing on examples from aquatic ecosystems across Europe and covering the following topics:

- Constructing linkage frameworks—elements, typologies and links
- Linkage frameworks as a visual tool for EBM
- Exploring the system—linkages, connectivity and modularity
- Weighting links—categorical and numerical approaches
- Linkage frameworks and Risk Assessment for EBM

2 Constructing Linkage Frameworks—Elements, Typologies and Links

Constructing a linkage framework requires defining the elements of the SES, the typologies within each element, and the links between these. The elements are the high-level building blocks of the framework that set out the *relevant* architecture for the EBM assessment being undertaken. For example, Box 1 illustrates a LF consisting of six major elements, all of which are important for exploring EBM options around achieving Europe's Marine Strategy Framework Directive (MSFD) (EC 2008). Box 2 considers a simpler assessment, where there were only three key elements in the LF explored. Here, the objective was to understand the dependencies of ecosystem services on marine species and habitats (Fiona E. Culhane et al. 2018).

Each component of the elements making up the LF can then be further defined for the system in question using a typology (for example, 'fishing' might be one component of the 'Activity' element of the LF). The typologies used should be *complete* for each element of the system, but the level of detail required will reflect the scope of the assessment being undertaken (e.g. whether to work with specific species or broad biotic groups). Typologies may build on existing work, and this may be especially important where the questions being explored have a context set out in policy, or where a structure is needed that is relevant to information or data collected at a specific level of detail (F. Culhane et al. 2020).

Box 1 Choosing *Elements* of the Linkage Framework *Relevant* for the Objectives of the Assessment

In the project Options for Delivering Ecosystem Based Marine Management (ODEMM), a linkage framework approach was used as the starting point for implementation of Ecosystem-Based Management (EBM). The main goals of the EBM assessment here were related to achieving the objectives of Europe's Marine Strategy Framework Directive (MSFD), i.e. Good Ecological Status (GES) for 12 key descriptors, which include, for example: '*seafloor integrity ensures functioning of the system*', '*elements of food webs ensure long-term abundance and reproduction*', and '*biodiversity is maintained*' (EC 2008). We were interested in the elements of the system that would affect achievement of these objectives (*sectors* (covering the major human activities affecting marine ecosystems in the study system), the *pressures* introduced by any one of those sectors, and the state of *ecological components*), how different *management options* acting on these increased likelihood of achieving GES, and how achievement of GES for those 12 descriptors might affect *ecosystem services* supported by the ecosystem. Thus, in this context, six key elements formed the overall structure of the system, whilst governance was defined for each EBM scenario explored (Fig. 1) (see further elaboration on this in Culhane et al. 2020).

For the example in Box 1, the MSFD lists activities, pressures, and ecological components relevant for European marine environments, as well as policy objectives as a list of 12 key descriptors (EC 2008). These lists formed the starting point for typologies (classifications or lists of components) of four of the seven key elements depicted by the boxes in Fig. 1 (see White et al. 2013). A typology of ecosystem services was developed in the absence of a suitable marine typology (Böhnke-Henrichs et al. 2013), and a typology of management options was devised to select relevant EBM measures (see Fig. 1 in Piet et al. 2015). For the Box 2 example, the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin 2013, 2018) was the starting point for the development of a marine ecosystem service typology. In addition, a holistic typology of service-providing units, built on existing EU typologies of habitats and biotic groups, was constructed. The explicit aim being to document how the combination of a specific habitat and biotic group supports the capacity of marine ecosystems to supply services (see detail in Culhane et al. 2018).

Having established typologies for each element within the LF of interest, the links between all components specified can be identified, initially in a qualitative way, simply showing where a link exists. Linkages are usually assigned based on expert judgement, underpinned by evidence from published studies where possible, and a set of matrices developed showing all links in the study system (e.g. https://www.odemm.com/content/linkage-framework).

Fig. 1 Blue boxes represent the major elements that were considered in this EBM study system where achievement of Good Ecological Status (GES; EC 2008) was the overarching objective, and examples of the components included in the underlying typologies for key elements are given. This diagram shows that a linkage framework is non-linear, as represented by the centre circle: linkages between elements are specified dependent on the issue being tackled at the time. All of the elements lie within the relevant governance setting, which determines the policy drivers, legal obligations, who is involved, and who makes the decisions (from Robinson et al. 2014)

In the Box 1 example, links between sectors and pressures were identified, where evidence suggests a pressure (such as noise pollution) can be introduced by a sector, whilst links between pressures and ecological components indicate that the pressure in question can affect that aspect of biodiversity (e.g. a specific habitat type). Links were specified between management options and sectors, pressures and ecological components, indicating where management might act (e.g. to reduce activity of a sector or restore state of an ecological component). Links between pressures, ecological components and GES descriptors indicated that the state of those pressures and/or ecological components can affect the potential to achieve the objective of a linked GES descriptor. Finally, links were specified between ecological components and ecosystem services, where the state of an ecological component is known to underpin the supply of a service.

In the Box 2 example, links between habitats and biotic groups indicate that the biotic group in question (e.g. demersal fish) would spend some or all of its life in a specified habitat. Whilst links between biotic groups and ecosystem services signify that there is evidence that the functioning of that biotic group in some way contributes to the supply of the linked ecosystem services.

Box 2 Service Providing Units: The Importance of *Typologies* That Reflect the Objectives of the Study

This example focuses on the work required to develop typologies within a linkage framework that best describe the functioning of marine ecosystems in terms of how they supply ecosystem services. Culhane et al. (2018) developed a typology of ecosystem components, or service providing units (SPUs), that fulfilled the following criteria:

1. SPUs reflect that biodiversity provides services through its functioning i.e. they explicitly specified and included biota and not just habitats. For example, while saltmarsh or mudflats provide erosion prevention, this does not recognise that it is the plants, tubes of invertebrates and films of microphytobenthos and microorganisms that actually supply the service. This point is important to recognise to both understand how the service is supplied and to understand how to protect or restore the service.
2. SPUs reflect that biota can vary in functioning between habitats and locations i.e. they explicitly included habitats to give SPUs a location and an abiotic identity. For example, floating clumps of macroalgae and attached algae both provide habitat for juvenile fish, however, only macroalage that forms belts around the coast also supplies coastal protection. This point is important to recognise the spatial aspects of ecosystem service supply.
3. SPUs reflect differences in vulnerability to human pressures i.e. vulnerability varies between biota and between habitats for the same biota. For example, epifauna is more sensitive to fishing pressure than infauna within habitats, and deep sea habitats are less exposed but also less resilient to fishing pressure than shallow habitats are. In considering the sustainability of ecosystem service supply, it is important to be able to recognise vulnerabilities to human pressures (Fig. 2).

Considering these three criteria led to the development of SPUs that consisted of a habitat and a biotic group. These units could then be linked to the specific services that they supply.

Complex SES considered in EBM may be represented by many thousands of linkages in a LF approach. This alone can highlight the complexity of the system and can be utilised to identify pathways through the system (see next section). Holistic EBM is complex by nature (Piet et al. 2020); the linkage framework approach allows scientists and advisors to visualise and structure the landscape within which advice is required and decisions are made. This can be of great importance where there is a lot of information or data available for some aspects of the SES in question

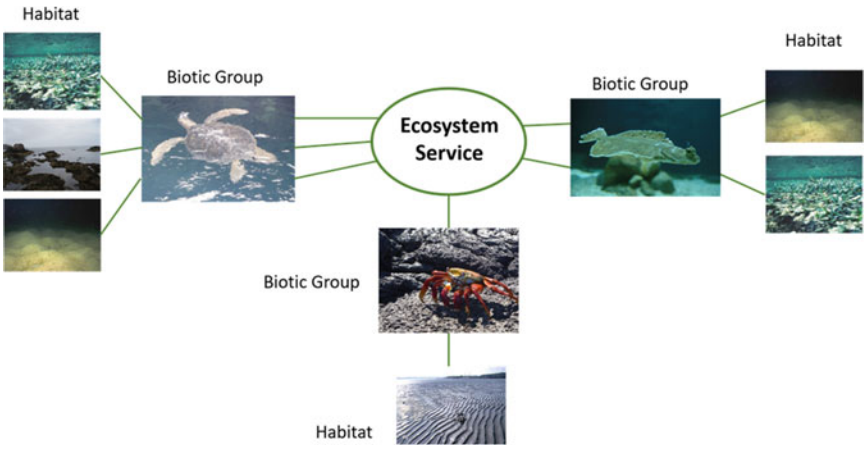

Fig. 2 Illustration representing a number of marine service providing units (SPUs), where each is made up of a habitat and a biotic group and lines represent the linkages between ecosystem services and the SPUs that supply them

and little for others. For example, Robinson et al. (2019)) used a LF approach to structure a stakeholder workshop. All activities operating and all relevant components of biodiversity in a study system were considered in terms of influence on stakeholder goals, rather than just those well known to stakeholders. This allowed barriers and opportunities in EBM to be identified.

It is important to note that the elements chosen, typologies defined within these, and basis on which linkages are defined must be clearly articulated in terms of the scope and rationale used. Ultimately, a LF provides a specific definition of a study system and it is critical that it is understood in this context.

3 Linkage Frameworks as a Visual Tool for EBM

The linkage framework approach provides a powerful tool for visualisation of the complex systems underlying EBM. One can start by illustrating the overall system (e.g. Fig. 1, Box 1), continuing to illustrate how the SES can be interrogated to reveal relevant linkages dependent on the topic of interest. Expanding on the example in Box 1, Fig. 3a highlights those elements of the marine SES that would be relevant in terms of considering status of, and threats to the MSFD's GES (Good Ecological Status) policy descriptors, whilst Fig. 3b builds on this, showing the linkages from management options that could act to reduce threat to GES. By working through a series of such illustrations, the framework within which EBM options sit can be contextualised for experts and non-experts alike, highlighting the interdependencies of the system.

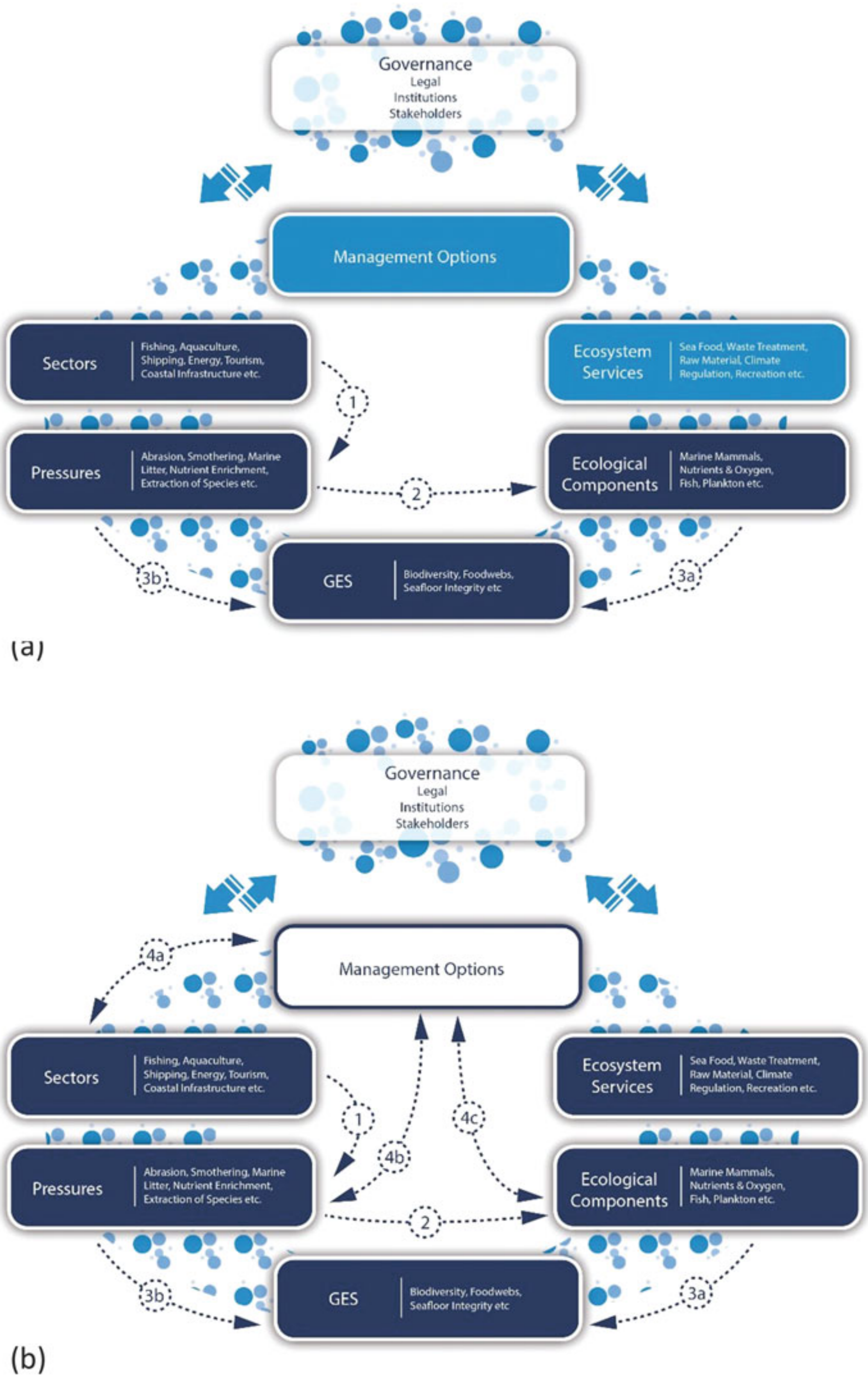

Fig. 3 The SES designed to encapsulate relevant elements that could be considered in EBM scenarios around achieving the EU's Marine Strategy Framework Directive (MSFD) objectives for Good Ecological Status (GES). In Fig. 3(**a**) elements (boxes) shown in dark blue are those that

From this broad starting point, the detailed linkages between the full typologies underlying the elements of the system can be further visualised in a linkage web diagram. For example, Fig. 4 shows the links between the sectors, pressures and ecological components for the North Sea, with 4(a) showing all documented links and 4(b) only those links relevant to the pressures introduced by the fishing sector. These diagrams highlight that an activity like fishing introduces multiple pressures that ultimately can impact all parts of the ecosystem, but also that there are several pressures that are introduced by many sectors but not by fishing. These linkage web diagrams are a powerful way to illustrate the connections in systems that may be otherwise ignored, unseen or neglected. They can also be used to highlight that focussing on a single sector or pressure may not result in successful management, because there are other sectors and pressures that are concurrently having impacts on the same parts of the ecosystem (emphasizing the need for holistic EBM).

Drilling down further, participatory approaches have been developed whereby illustrations of the individual components of the typologies represented in the linkage framework are used to help engage stakeholders in a holistic exploration of the SES (e.g. Fig. 5; Robinson et al. (2019)). In a series of workshops based around European marine regional seas held in 2013, illustrated cards of the full typology of marine ecosystem services developed in Böhnke-Henrichs et al. (2013) were explored with stakeholders. Feedback suggested that many stakeholders had not previously been aware of the number and broad diversity of ecosystem services supported by marine ecosystems (Robinson et al. 2014). Stakeholders further confirmed that their views on how EBM should be implemented had changed markedly because of the visualisations provided of the full SES for their regional sea.

4 Exploring the System—Linkages, Connectivity and Modularity

Once the system has been defined and visualised in terms of its elements, underlying typologies and linkages, analysis of the framework can be carried out in a number of ways, using approaches borrowed from system analysis including connectance and modularity. These techniques allow simple exploration of the elements and their links in the system. Yet, useful information can emerge from this high-level consideration. For example, Knights et al. (2013) showed how network analysis of a LF covering activities, pressures and ecological characteristics could be used to identify

Fig. 3 (continued) are fundamental in terms of the assessment of the status of, and threats to, GES descriptors. Arrow 1 indicates linkages from sectors that introduce listed pressures, arrow 2 shows links from those pressures to ecological components, and arrows 3(a) and (b), those pressures and ecological components whose status is relevant to GES. Figure 3(**b**) adds linkages from management options to illustrate that they can act on sectors, pressures and/or on ecological components (arrows 4 a–c)

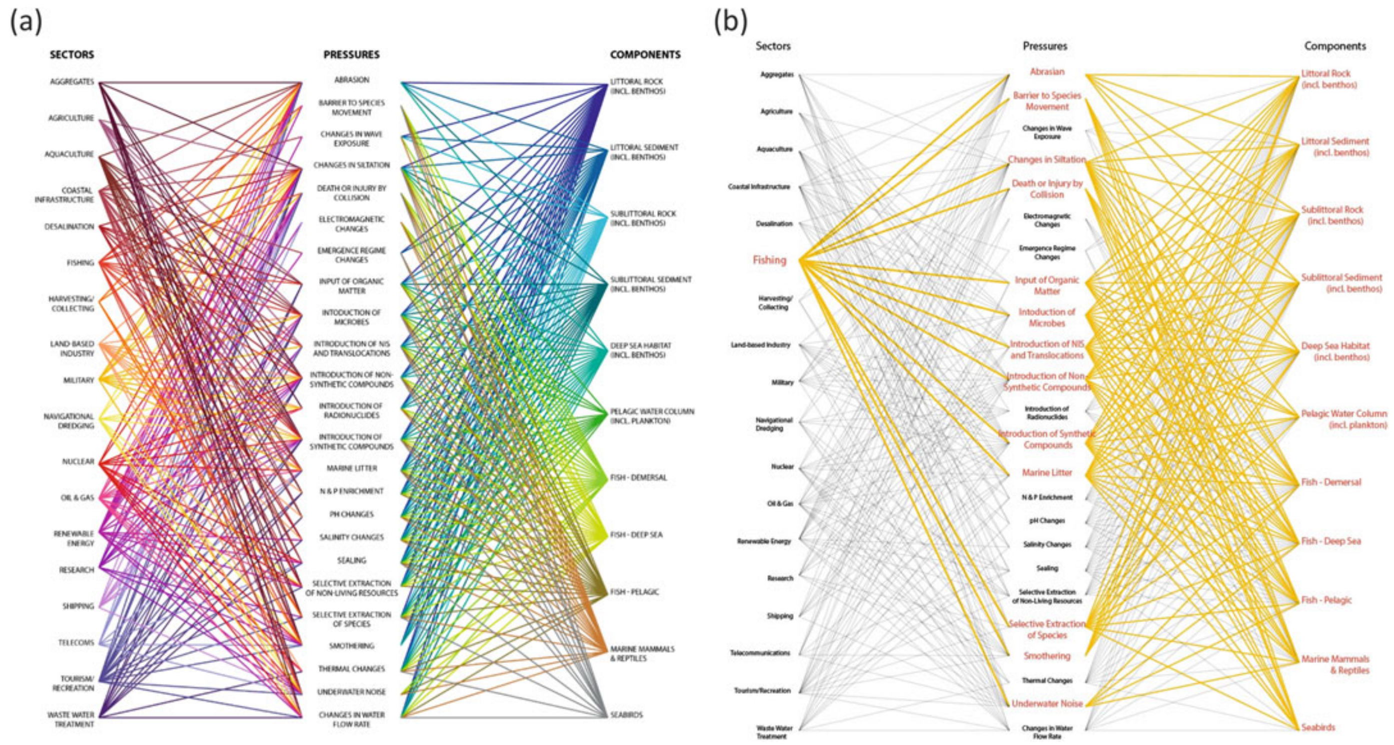

Fig. 4 (**a**) The sectors, pressures and ecological components of the North Sea and the linkages between them (taken from Robinson et al. 2014), and (**b**) the subset of that system highlighting linkages from Fishing to the ecosystem through the pressures introduced by that sector alone

Fig. 5 An example of three cards designed to illustrate individual components of some key elements of the Lough Erne socio-ecological system in Northern Ireland (from Robinson et al. 2019)

groupings of impact chains to aid in simplification when considering EBM options. Culhane et al. (2018) went on to consider the links between marine ecosystem components and ecosystem services. They calculated connectance (Poisot and Gravel 2014), or the number of links per biotic group supplying a service as a proportion of the total number of links in the full network (Fig. 6). This revealed the importance to service supply of less charismatic species in the network e.g. bacteria. It also revealed that remote habitats are important for supporting mobile species that supply services elsewhere. This suggests that management to protect the sustainable supply of services must consider broader areas than just those where services are used.

Another technique used to explore linkage frameworks is modularity (Beckett 2016). This way of visualising links in a network highlights groups, where certain components of the system share more properties than they do with others. For example, Robinson et al. (2019) used modularity to highlight sub-sets of stakeholder goals that have similarity to each other on the basis of their interactions with activities and biodiversity that occur in Lough Erne (see also O'Higgins et al. 2020) (Fig. 7). This approach highlighted activities such as conservation, scientific research, tourism, and other recreational activities that have strongly positive associations with stakeholder goals related to biodiversity, living landscapes, and heritage (see Module A.B, Fig. 7a). On the other hand, modularity highlighted a sub-group of biodiversity components made up of invasive species perceived to negatively influence the same stakeholder goals (Module B.C, Fig. 7b) (also see

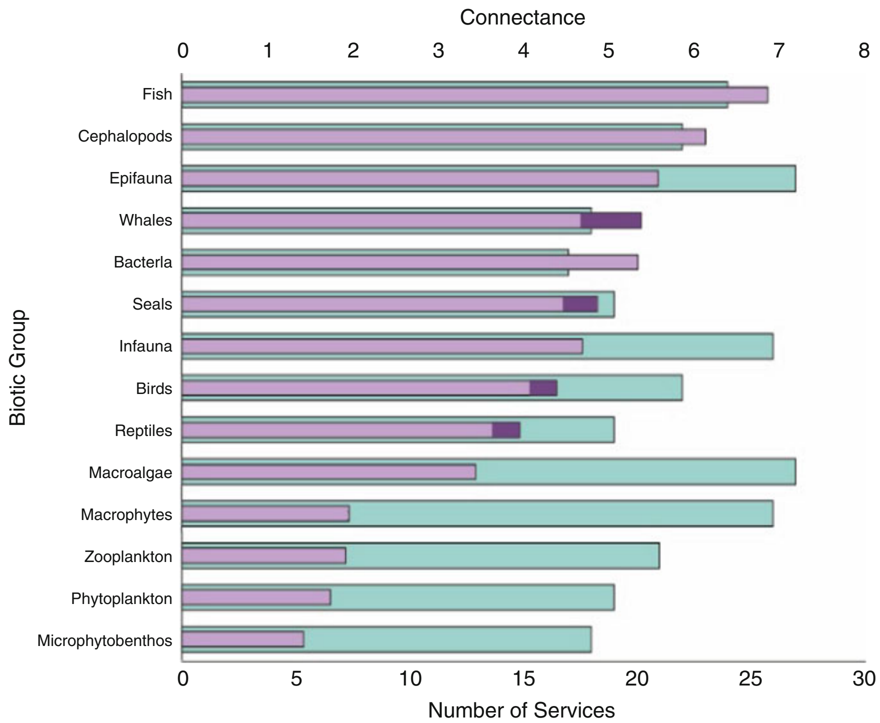

Fig. 6 The connectance (purple) and number of services (blue) supplied by each biotic group. Dark purple indicates indirect links, where a biotic group is being supported by a habitat remote from where it supplies services. (Taken from Culhane et al. 2018)

Fiona E. Culhane et al. 2018). This approach demonstrates that, although linkage frameworks are complex and contain many elements, meaningful patterns can emerge, and these patterns can be used to foster stakeholder discussion or to inform management decisions.

5 Weighting Links—Categorical and Numerical Approaches

Further development of the qualitative linkage framework approach is to make it semi-quantitative. Each link in the framework can be weighted according to its particular properties and how it interacts with the system. In the last example described, stakeholders in the Lough Erne system weighted typologies of activities and biodiversity, in terms of their perception of how each would affect their individual goals, using scores ranging from strongly positive to strongly negative (Robinson et al. 2019; see Fig. 7). In other approaches, weighting may be based on

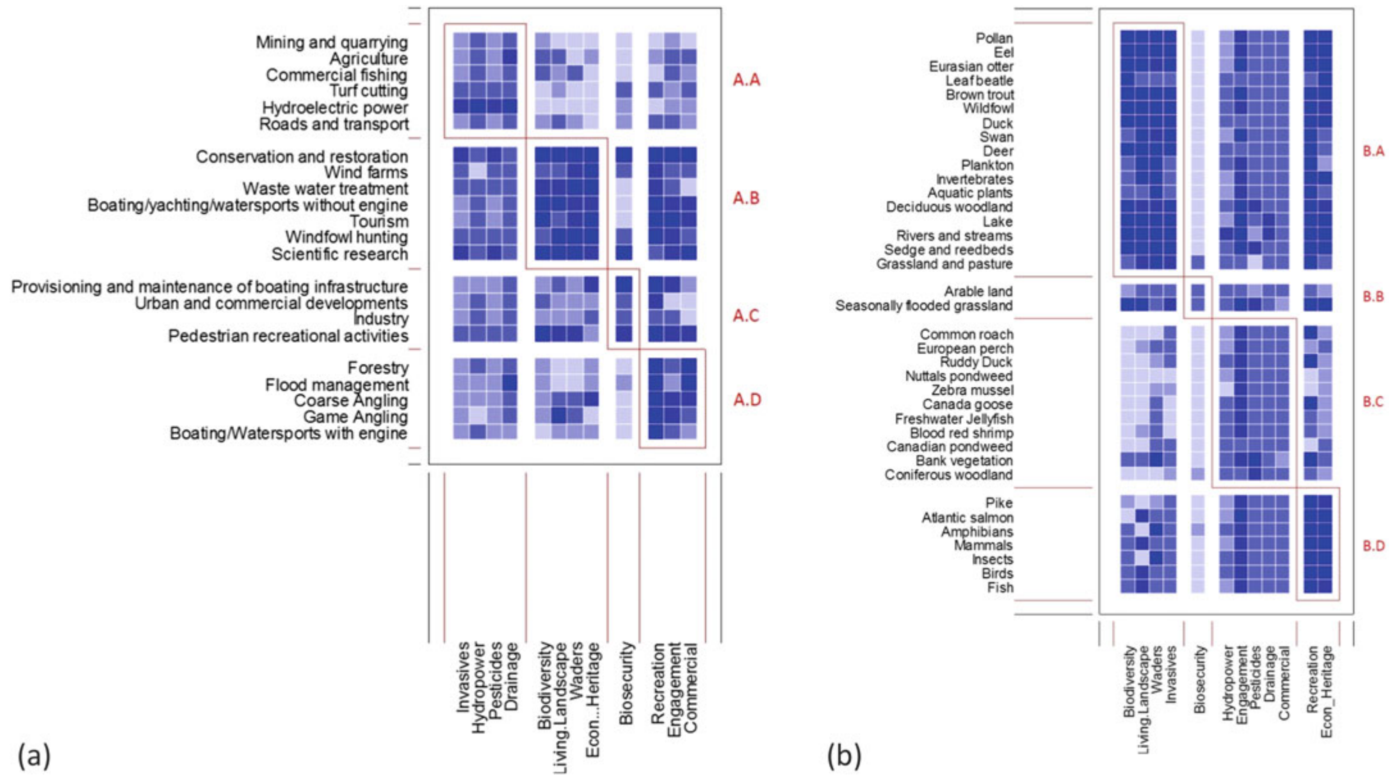

Fig. 7 Modular sub-sets of (**a**) activities and (**b**) biodiversity components, with the stakeholder goals they affect, highlighted according to the type of interaction; darkest blue squares indicate strongly positive interactions and lightest blue strongly negative. (Reproduced from Robinson et al. 2019)

unique criteria developed to define the interactions signified by specific linkages. For example, in the ODEMM LF of European regional seas (see Figs. 1 and 3) more than 6000 linkages were identified; expert weighting of these using five categorical criteria in a structured pressure assessment approach (see Robinson et al. 2013) allowed a relative comparison of threats in the system.

A categorical approach allows exploration of the nature of interactions in the SES, because the categories used can help to shed light onto the amount and types of interactions seen. For example, Robinson et al. (in prep) were able to show that almost 10% of threats (unique linked elements, e.g. a sector, pressure and ecosystem component affected; sensu impact chains in Knights et al. 2013) in European regional seas were of the most severe threat type, and of these, just under a third are threats associated with recovery times of >200 years. Linkages associated with high threat characteristics can be identified and the categorical criteria used to identify, for example, sets of threats that might have greatest management potential.

Weighting can also be given a numerical value. For example, Potts et al. (2014) and Burdon et al. (2017) weighted different protected habitats and seabirds, respectively, in terms of their importance for supplying different services. Teixeira et al. (2019) built on this approach, weighting the links between parts of the ecosystem and ecosystem services based on: (1) expert judgement, taking into account how important a habitat is in supplying a particular service in a particular region (supply potential); (2) the area of that habitat (supply capacity); and (3) the condition the habitat is in (supply condition). Together, these three aspects were considered to influence the capacity for habitats to supply ecosystem services and were combined into one overall score for each habitat (Fig. 8).

As discussed earlier, the complex SES underlying EBM can be structured and visualised with a linkage framework. Weighting the LF can then be used to explore relative importance among elements in the LF, according to the criteria used, whether that is how well a habitat supplies services, or how much an activity impacts an ecological component.

6 Linkage Frameworks and Risk Assessment for EBM

Finally, we go on to cover how a weighted LF can be used to explore risk in a SES. Risk consists of both the exposure of the ecosystem to threats and the consequence of those threats on the ecosystem. That is, how much is an ecological component exposed to activities that introduce pressures; and what is the magnitude of those activity-pressure interactions—in terms of characteristics of the pressure and how resistant the ecosystem component may be to pressure impacts. Combining these, exposure and consequence, allows the relative degree of risk to be assessed (Knights et al. 2015). Here we particularly focus on the risk to the ecosystem and to the supply of ecosystem services, which may then go on to impact economic activities or affect other social aspects of the system in an EBM context.

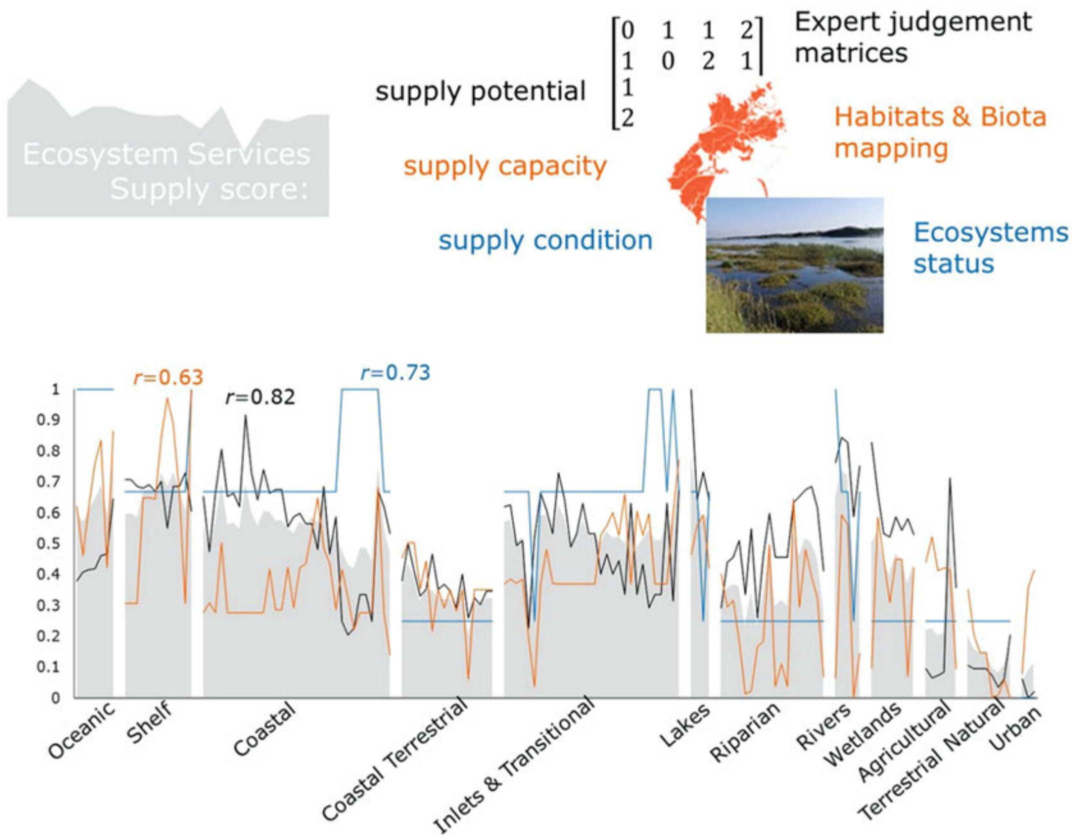

Fig. 8 Three different weightings—supply potential based on expert judgement, supply capacity based on habitat area, and supply condition based on ecosystem status (each scaled between 0–1, see y axis)—are given to links between habitats and ecosystem services to produce an overall ecosystem services supply score in aquatic ecosystems (reproduced from Teixeira et al. 2019). The bottom graph shows each of these scores over all habitats in each different aquatic ecosystem (e.g. Shelf), as represented by the black, orange and blue lines, respectively. The solid grey area represents the overall ecosystem services supply score, which combines the three individual aspects of service supply. R values are the Pearson *r* correlation coefficient of each individual weighting with the final overall ecosystem supply score

In marine environments, some risk assessment approaches have been implemented with spatial data, where habitat maps can be overlaid with maps of activities. Habitats can be given a score for their sensitivity and activities a score for the degree of impact they introduce. For example, Arkema et al. (2015) identified the relative degree of risk of coastal habitats in Belize and tested how different scenarios of future management options might result in impacts to different ecosystem services in low, medium and high risk habitats. This approach allows risk assessment to be linked to specific locations and management options. However, spatial data is often lacking in marine environments, especially at larger scales. An alternative approach is to base the risk assessment on a linkage framework, assigning scores to the weighted links, to produce a semi-quantitative risk assessment by combining scores through the LF. In this way, assessments of even large regions can be carried out. This was the approach used by Knights et al. (2015) in European regional seas and Halpern et al. (2015) in broad global assessments of risk to marine habitats.

The risk to habitats can also be determined relative to other habitats or locations. A recent example of this comes from Borgwardt et al. (2019), who looked at the impact risk across seven different European aquatic systems ranging from freshwater lakes and rivers to large marine regions (Fig. 9). They scored activity-pressure combinations for all habitats defined in an extensive linkage framework, based on five criteria: extent, frequency, persistence, severity and dispersal potential, to come up with one overall risk score for each habitat type. They reported both average and summed risk, where summed risk is greater when a habitat has more activities introducing more pressures to it.

Once the risk to habitats has been established, this can be extended to consider the risk to the supply of ecosystem services, under the assumption that ecosystem service supply is reliant on the state of the ecosystem. Using the same large, regional data, Culhane et al. (2019) linked impact risk from activities and pressures on ecosystem components with service supply potential from those ecosystem components to come up with a risk to service supply score. Overall, they found that risk is greatest in those habitats that have the greatest potential service supply (Fig. 10).

Finally, the risk scores assigned in a LF based assessment approach can then be interrogated to explore risk reduction under different EBM strategies. Piet et al. (2015) explored how different EBM measures reduced the risk to the Northeast Atlantic ecosystem, finding that measures performed differently, dependent on whether one focused on reducing past damage, present or future threats. They were able to utilise the underlying regional sea linkage framework to identify where to best target management, dependent on the focus and type of measures available.

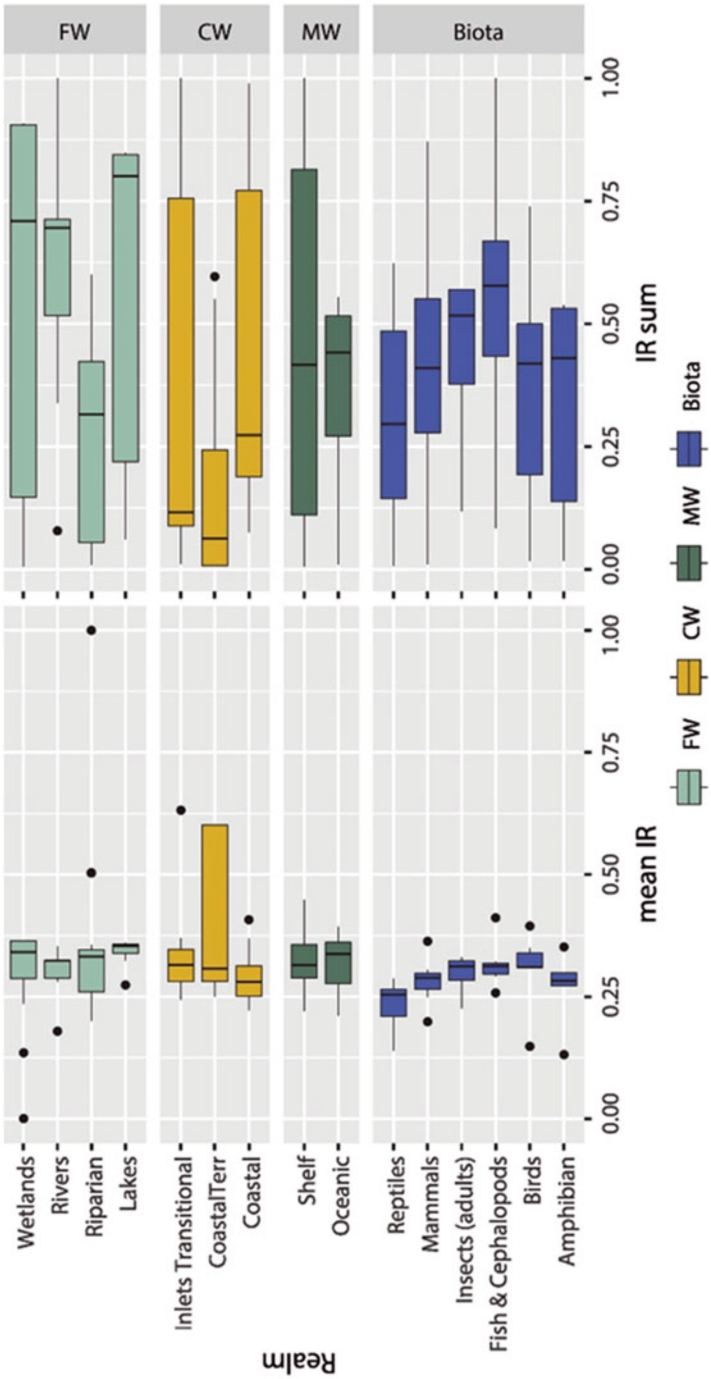

Fig. 9 Mean and summed impact risk (IR) across habitats within different European aquatic ecosystems (Realm, y axis) based on a risk assessment approach. (Reproduced from Borgwardt et al. 2019)

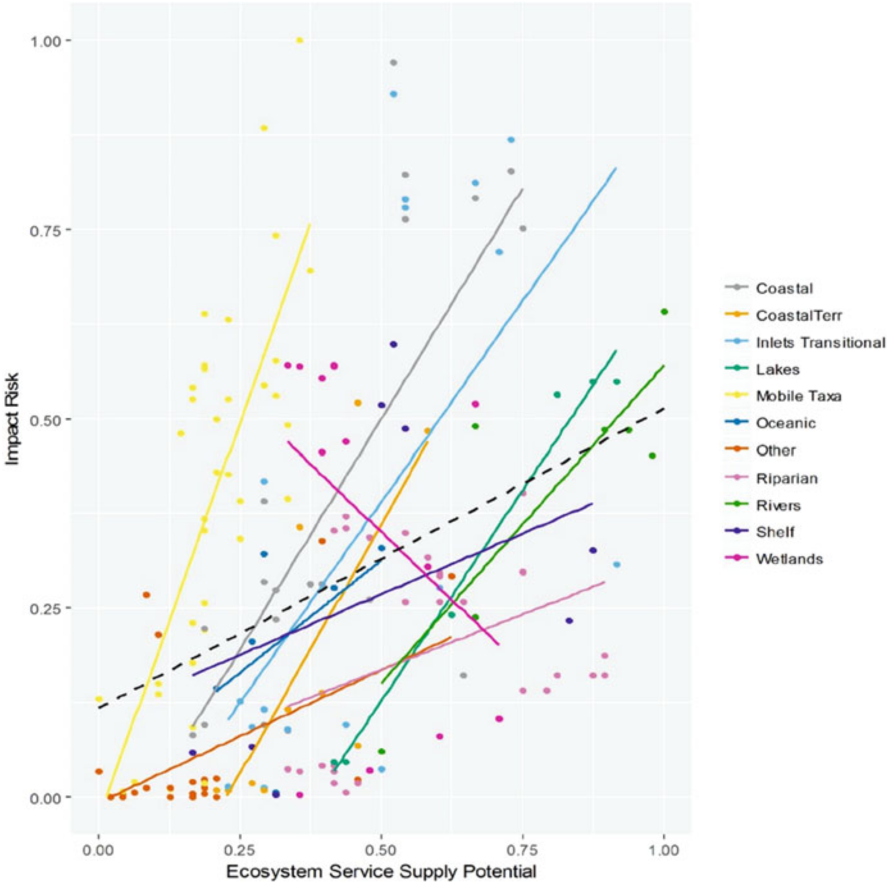

Fig. 10 The relationship between impact risk and service supply potential across different aquatic ecosystem types (shown in the key) in Europe based on a risk assessment approach. Black dashed line represents the overall relationship. (Reproduced from Culhane et al. 2019)

7 Summary and Conclusion

Linkage framework approaches allow a large amount of complex information, often based around expert judgement, to be formalised into an understandable structure. The LF can then be explored in various ways to answer or to provide different perspectives on EBM questions. While always considering the full complexity of SES is not possible, an LF approach goes some way to retaining this complexity and providing an overview of an SES without oversimplifying it. The approach is not data driven so can rely on a body of expert knowledge that exists amongst scientists or in the broader scientific literature. Yet, because the approach is not data driven and requires an expert knowledge base, it needs to be complemented with detailed, data driven studies to support the assumptions made.

It is important to clearly communicate how the scope and objectives of any study has influenced the linkage framework designed, in terms of the elements included, detailed typologies used, and linkages defined. It is equally important to acknowledge that the architecture of the linkage framework used and the summation methodology approach taken in any quantitative analysis of an LF (such as in a risk assessment) will influence the results (see Piet et al. (2017) for further exploration of this). A transparent and clear explanation of these aspects helps to engender confidence in users and to generate meaningful and reliable advice for EBM decision-making.

In the examples explored in this chapter, only direct interactions are included in the SES studied. Further work is required to account for indirect links, which can be as, or more important, in terms of understanding unintended consequences of management actions. Likewise, analysis of weighted LFs to date, has assumed linear interactions in terms of cumulative risk, and it is well-documented that non-linear responses can be expected in SES. Both indirect effects and non-linear responses can be incorporated going forward, but further research is required to fully account for these.

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Projecting Changes to Coastal and Estuarine Ecosystem Goods and Services—Models and Tools

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Abstract Coasts and estuaries provide an abundance of ecosystem goods and services (EGS) to humans worldwide. Models that track the supply, demand, and change in EGS within these ecosystems provide valuable insights that have applications in the context of land-use planning, decision-making, and coastal community engagement. However, developing models for use in coastal and estuarine ecosystems is challenging given the multitude and variability of potential input variables, largely due to their dynamic nature and extensive use. Models that can incorporate scenarios of environmental change to forecast changes in EGS endpoints are highly valuable to decision-makers, but only a minor proportion of available EGS models offer this utility. In this chapter, we describe the domain of models most useful to coastal decision-makers, present models at multiple scales that can predict EGS changes, and examine specific examples that epitomize this utility. We also highlight common difficulties in modeling coastal and estuarine EGS and propose suggestions for integrating EGS models into the coastal management decision-making process during times of increasing environmental change.

Lessons Learned

- Identifying the most suitable model(s) given the scale(s) of a particular question or goal is paramount in the modeling process
- Uncertainty is an inherent component of modeling that should be well-communicated by users to avoid misinterpretation of results

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- The complexity, as well as the time, cost, and data requirements, of many EGS models are barriers to widespread implementation

Needs to advance EBM

- To improve model relevancy and usability, resource managers and other stakeholders must be part of the model development process to identify important decision metrics
- Tiered or coupled models can allow for the identification and inclusion of multiple drivers as well as cumulative impacts on coastal and estuarine EGS, which is a critical need
- Development of a standardized framework for model implementation would increase the adoption of EGS models into coastal and marine planning processes

1 Modeling Changes in Coastal and Estuarine EGS

Coastal and estuarine ecosystems host some of the most dynamic and productive habitats in the world (Costanza et al. 1997; Costanza et al. 2014), including: seagrass beds, mangroves, coral reefs, salt marshes, sandy beaches, and dunes (Barbier et al. 2011). Each of these habitats provides a unique combination of benefits across the broad ecosystem goods and services (EGS) categories (i.e., provisioning, regulating, cultural, and supporting services) set forth by the Millennium Ecosystem Assessment (2005) (e.g., Marois and Mitsch 2015; Kassakian et al. 2017). As environmental changes propagate through these ecosystems, the provision of important EGS may also be affected. Changes can result from natural or anthropogenic impacts and come in a myriad of shapes and forms, including: sea-level rise, pollution, invasive species, storm events, and detrimental fishing practices.

Ecosystem alterations can be the product of a multitude of factors, occur at different spatiotemporal scales, and affect EGS of all types. Integrating these dynamic variables poses a challenge when modeling EGS endpoints. However, the quantity of models and tools that explicitly predict changes to EGS based on alterations to input variables make up a minor proportion of all models applied in coastal and estuarine ecosystems. Some of this disparity may be due to the relatively recent development of the EGS concept, as well as the inherent complexity and variability of these models. In this chapter, we present a suite of models that exemplify the approach of predicting EGS changes at different scales, outline the domain of models that may offer the most utility to coastal decision-makers, present examples epitomizing this utility, and highlight common difficulties across coastal and estuarine EGS models. We conclude with suggestions for integrating EGS models into the coastal management decision-making process during times of increasing environmental change.

Developing models for application in coastal and estuarine ecosystems can be challenging given their dynamic nature and the multitude of potential inputs and outputs; however, models provide valuable tools that can be used for many purposes, such as visualizing species distribution, assessing chemical fluxes, explaining habitat-species relationships, or identifying spatial patterns. The utility of models increases substantially for resource managers when environmental changes can be simulated within these tools to predict changes to EGS.

Although there are many EGS models (Bagstad et al. 2013b; Turner et al. 2016; Gret-Regamey et al. 2017; Little et al. 2017), this chapter focuses on models that provide predictions of how EGS may be affected as coastal ecosystems undergo change. We further narrow this focus to models that demonstrate a high degree of utility to decision-makers based on relevant spatial scales and endpoints. Within these criteria, we present example models that are well-established and demonstrate these traits with different degrees of complexity and quantities of modeled services to show the utility of this approach across a diverse spectrum of models.

Coastal and estuarine EGS models can range from simulating services provided by a specific site or habitat (Mendoza et al. 2017; Harris et al. 2018), to estimating global EGS values (Costanza et al. 1997; Boumans et al. 2015). Here, we limit our focus to models with spatial scales ranging from local (e.g., estuary or bay) to regional (e.g., U.S. Pacific West Coast), as they are the most likely to provide useful information when making resource management decisions (Turner et al. 2016). These models also vary greatly in their final output or endpoint. Some estimate relative changes in the degree of EGS provision (Hanson et al. 2012; Harris et al. 2018), while others estimate monetary values of EGS (Carr et al. 2018). The example models we discuss in this chapter all provide outputs that are quantitative and informative (but not necessarily monetary), as these provide the most benefit to decision-makers at local and regional scales (Ruckelshaus et al. 2015).

Within these established bounds of endpoint and spatial scale there still exists a wide range of model complexity. The spatial and/or temporal resolution, system dynamics considered, and type of EGS simulated each affect model complexity. In the following sections, we explore five example models representing different combinations of spatial scale, quantity of EGS, and complexity, each occupying different niches within the domain of EGS models that can predict change and inform decisions. The first model, HexSim, is a mechanistic model that describes living populations by tracking individuals; the second, XBeach, is a mechanistic model for estimating shore protection; the third, Atlantis, is a whole system model used in fishery management; the fourth, InVEST, contains many sub-models that can predict delivery of a suite of EGS; and the fifth, ARIES, uses machine learning to trace ecosystem service flows to beneficiaries.

2 HexSim Model

Ecosystem services under management are frequently tied to changes in both the behavior and demographics of living resources. Predicting these changes is an important part of decision making, and object-oriented and Individual-Based Models (IBMs) play an important role, both in making these predictions and in operationalizing the answers as a tool for policy. IBMs are mechanistic models that have typically been used to evaluate the movement, growth, and mortality of living populations, by tracking individuals rather than the population as a whole (DeAngelis and Rose 1992). IBMs are complex by nature because they describe an individual's state, and therefore the spatial resolution of any input data must match individual behavior (e.g., meter), while ensuring the spatial scale of the model remains relevant to decision making (e.g., whole estuary). IBMs are inherently data intensive because the mechanistic descriptions of movement, feeding, growth, and mortality are all necessary to properly track a group of individuals. Yet, given sufficient input data, IBMs have proven useful for predicting population-level change, particularly in cases where average population conditions are not sensitive to management.

IBMs track the state and response of individual agents, then combine the outcomes into a population level distribution for the chosen response variable (e.g., total biomass, individual size). Agents are defined by the context of the question but are generally individual organisms of a population or demographic group (e.g., individual anglers). In spatially explicit cases, the agent can be a spatial grid cell as is the case in the HexSim IBM (Rustigian et al. 2003; Fulford et al. 2011). Such spatial models are optimized for the study of population distribution in response to heterogeneous landscapes. Input data must include initial agent state for key variables, as well as function parameters describing how agents respond to environmental variability and management-based change. IBMs are by nature an iterative suite of interconnected forcing functions. Model output is temporally and spatially explicit as the model tracks individual deviations from an initial state, which can be summarized at any time during a model simulation.

The HexSim model (Schumaker et al. 2004; Schumaker and Brookes 2018) was used to predict the impact of seagrass management on fishery resources in Tampa Bay (Fulford et al. 2016). Seagrass restoration was combined with a suite of other habitat components, including water temperature and hypoxia, to predict how individual habitat selection and subsequent growth were impacted. Management decisions were evaluated in the context of multiple environmental factors based on their cumulative impact on growth and production of aquatic resources that provide ecosystem services. The IBM also informed the delivery of ecosystem services to stakeholders in that individual angler behavior was modeled in response to changes in fish distribution and apparent availability of habitat as fishing grounds. This latter bioeconomic component of the model was based on angler preference data (Fulford et al. 2016).

HexSim in Tampa Bay was a coupled movement and bioenergetics model for fish and a coupled movement and fishing success model for anglers. Input data included fish distribution prior to habitat change, habitat-specific fish growth and movement functions, angler distribution, and an angler choice function based on distance from access points and daily fishery return. Fish moved first, and their growth and mortality were predicted based on habitat choice. Anglers chose fishing locations based on distance from access points and knowledge of optimal fishing habitat and their catch rate was predicted based on a probability model and predicted overlap between anglers and catchable fish (Fulford et al. 2016).

IBMs are most useful in cases where the mean output value is insensitive to manageable change in the system (e.g., habitat restoration). A good example is the mean fish growth rate in the Tampa recreational fishery in response to seagrass habitat restoration (Fulford et al. 2016). Compensatory behavior and the multi-faceted nature of habitat selection greatly ameliorated the mean fish growth response, even with a large change in seagrass coverage. As a result of this insensitivity, the impact of successful seagrass restoration on fishery harvest was highly difficult to identify. As an alternative, an IBM approach was used to track individual behavioral response to habitat distribution, then bioenergetics sub-models were used to translate fish distribution into population production.

An IBM approach, such as the HexSim model described here, is both data-intensive and complicated enough that managers rarely attempt its application without assistance. That limits the utility of this approach to environmental decision making in data rich contexts (Rose et al. 2010; Rose et al. 2015). However, the availability of data and expertise for IBM use is higher than it has ever been and growing. These models require a high level of expertise to apply, yet the subtle nature of management related change and the confounding influence of other environmental variables make the use of IBMs much more informative and highly desirable for decision making.

3 XBeach Model

Models that can simulate the link between an ecosystem's structure and its ability to deliver ecosystem services are key to bridging the gap between ecology and economics. These types of models are often highly mechanistic and can occasionally be adapted for integration into tools for estimating coastal EGS (Bruins et al. 2017). XBeach is one such model that has been used to predict the ability of coral reefs to deliver the service of coastal protection under various conditions (Quataert et al. 2015; Pearson et al. 2017; Harris et al. 2018). It is a one-dimensional, morphodynamic model that simulates how ocean waves travel across complex, near-coast topographies and propagate up to the shore (Roelvink et al. 2009). XBeach was originally designed for wave propagation on beaches, but with some modification it has been expanded to simulate accurate reef hydrodynamics (Pomeroy et al. 2012; Van Dongeren et al. 2013). These capabilities allow it to estimate the extent to which coral reefs reduce wave run-up (how high a wave rises above mean sea level as it reaches the coastline) at local and regional scales. XBeach can also predict changes in the provision of reef shoreline protection under scenarios of environmental change such as sea level rise (Harris et al. 2018).

If a coral reef ecosystem is significantly diminished in size or health, its valuable coastal protection service could be completely lost (Sheppard et al. 2005). These protective services can also be reduced if coral reefs are not able to grow vertically at a rate comparable to rates of sea level rise (Harris et al. 2018). Within XBeach, reef degradation scenarios can be simulated by altering coastal bathymetry and surface roughness inputs, while sea level rise can be simulated by increasing offshore wave-height drivers. Resulting changes in the model output of wave run-up can then be interpreted from a coastal hazard perspective, with higher run-up values leading to a greater risk of damage to coastlines, particularly during storm events that further increase wave heights (Quataert et al. 2015).

Harris et al. (2018) demonstrated the utility of XBeach in predicting coastal protection services, by applying the model in reef sites in French Polynesia. Model parameters were calibrated using measured data from pressure loggers in cross-reef transects. They varied model inputs of sea level, reef vertical accretion or erosion, and reef surface roughness (corresponding to reef structural complexity). Monte Carlo simulations, in which the model was run multiple times with inputs varying randomly across a range of values, were used to address variability that was not captured by XBeach. Overall, results showed that the combined effects of sea level rise, reef erosion, and reduction in reef structural complexity led to wave heights that were 2.4 times greater than those under present conditions (Harris et al. 2018). These findings not only quantified the amount of protection offered by reefs under current conditions, but also provided coastal land managers with predictions of how waves may impact coastal areas in the future should the extent of coral reefs decline.

XBeach model output primarily consists of an estimated reduction in wave height, which by itself may not be informative enough for coastal decision-makers to draw worthwhile conclusions. However, when mechanistic models like XBeach are incorporated into larger modeling frameworks, its output can be used as an input for another model predicting levels of ecosystem service provision (Bagstad et al. 2013b). For example, the reduction in wave height could be converted into a reduced frequency of flooding, which may be more informative. Further, this reduction in flooding frequency could be converted into monetary value using methods similar to those described by Barbier (2016). This use of well-established and validated mechanistic models to deliver predictions of coastal EGS can be effective for incorporating EGS considerations into coastal management decisions and policy.

4 Atlantis Model

In whole-system (i.e., end-to-end) models (Travers et al. 2007), biophysical, socio-economic, and industrial components and processes are considered, along with associated feedbacks and interactions between components (Fulton et al. 2011; Kaplan et al. 2012; Weijerman et al. 2015; Marshall et al. 2017). One such model is Atlantis, developed for use in Management Strategy Evaluation, it supports Ecosystem-Based Management (EBM) of multiple fisheries on a regional scale (Link et al. 2010; Fulton et al. 2011). Atlantis is a complex, spatially explicit, hierarchical model containing interconnected submodels to evaluate potential management actions, policy changes, and ecosystem tradeoffs under various scenarios (Kaplan et al. 2012; Fulton et al. 2014; Marshall et al. 2017).

A biophysical submodel is the primary submodel within Atlantis, which follows nutrient flows (primarily nitrogen and silica) and distribution through the food-web to large marine mammals via ecological processes (e.g., production, predation, recruitment) and physical features (e.g., hydrodynamics, seabed types, water properties) (Link et al. 2010; Fulton et al. 2011). Output data from the biophysical model feed into an exploitation submodel that considers human uses and impacts, such as fishing, pollution, development, and other environmental changes (Link et al. 2010; Fulton et al. 2011). A monitoring and assessment submodel then utilizes outputs from the biophysical and exploitation models to simulate scenarios, from which adaptive management options and the associated uncertainty can be assessed (Link et al. 2010; Fulton et al. 2011). Finally, the monitoring and assessment output is fed into a management submodel, which consists of potential management rules defined by the user (e.g., restrictions, quotas, limits) that respond to the inputs from the assessment model (Link et al. 2010; Fulton et al. 2011).

Endpoints of Atlantis can include metrics such as biomass, concentration, catch, effort, or revenue (Link et al. 2010; Fulton et al. 2011; Kaplan et al. 2012; Marshall et al. 2017). Marshall et al. (2017) built upon previous applications (Kaplan et al. 2012; Weijerman et al. 2015) and utilized Atlantis to project the impacts of ocean acidification scenarios on fisheries EGS in the California Current ecosystem. To achieve this, future pH projections and biological pH sensitivities were plugged into the biophysical submodel; pH projections were derived from a separate model, whereas pH sensitivities were obtained from a meta-analysis of experimental results (Busch and McElhany 2016). Results demonstrated that a projected 0.2-unit decrease in pH negatively affected biomass, catch, and resulting revenue of most fishery management units considered. Atlantis predicted that state-managed functional groups would experience the greatest pH effects on revenue, largely due to the strong indirect effects on the valuable Dungeness crab fishery.

Although the full capability of Atlantis is impressive, utilization of the entire model without identifying key drivers can lead to over-parameterization (Fulton et al. 2011). There is also an inherent tradeoff between generality and precision with this broad ecosystem-based approach (Plaganyi 2007; Link et al. 2010). These limitations provide some context as to why Atlantis should not be used to determine specific management rules or actions, such as setting quotas (Fulton et al. 2011; Fulton et al. 2014). Atlantis researchers and developers have continued to improve and enhance the model however, to address some of model's weaknesses and create a more cohesive user community (Weijerman et al. 2016).

Properly implemented, end-to-end models like Atlantis can provide tremendous benefits to users, including the comparison of direct, indirect, and cumulative ecosystem effects across multiple fisheries under a variety of future scenarios (Fulton et al. 2011; Kaplan et al. 2012; Weijerman et al. 2015). Atlantis has been successfully implemented in regional-scale studies around the world (Weijerman et al. 2016), at scales of up to 1.475 million km^2 (Marshall 2017). These studies have demonstrated that Atlantis is a valuable tool to forecast changes in fisheries-related

EGS resulting from drivers of ecosystem change, including pollution, fishing, and climate change (Weijerman et al. 2015; Marshall et al. 2017). The scale and scope at which this spatially explicit model can be applied is consistent with the concepts of EBM, which is relevant for ecosystems not restricted by political boundaries, and thus important to the management of coastal and estuarine EGS.

5 InVEST Model Suite

Coastal decision-makers must often consider different types of EGS, thus models capable of analyzing tradeoffs between different types of EGS under different scenarios are valuable in the context of coastal planning. Recently, some 'decision support tools' have been developed to aid in this process by connecting a suite of EGS models within ecological and socioeconomic frameworks (Bagstad et al. 2013b; Gret-Regamey et al. 2017). In a recent review by Ochoa and Urbina-Cardona (2017), InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs), an open-source tool developed by the Natural Capital Project (NCP, www.naturalcapitalproject.org), was one of the most widely cited resources for spatially modeling EGS. InVEST was developed with the explicit purpose of integrating natural capital into decision making (Daily et al. 2009). InVEST started as a general EGS model for terrestrial and freshwater systems, but in recent years, several marine and coastal models have been incorporated into the toolset (Tallis and Polasky 2009; Guerry et al. 2012). Similar EGS decision support tools include MIMES, LUCI, and ARIES (Bagstad et al. 2013b), the latter of which is discussed in the following section.

InVEST models can be applied from the local to regional scale and provide a variety of endpoints, making them highly useful for land-use planners and other decision-makers. Most input data must be spatially explicit and can include maps of biophysical information (e.g., elevation, habitat, species distributions, etc.), as well as socio-economic indicators (e.g., population density, property values, industry costs, etc.) (Tallis and Polasky 2009; Guerry et al. 2012). Submodels use this input to predict changes in EGS such as coastal protection, fisheries, and recreation. Submodels employ several different approaches for estimating EGS, sometimes with an option to select the 'tier' of submodel (i.e., higher tiers being more complex) depending on the desired endpoints, data requirements, and effort investment (Guerry et al. 2012). Two submodels for habitat risk and water quality provide interconnections between EGS submodels and allow users to consider tradeoffs when prioritizing management across different EGS. The final submodel predicts the quantity of EGS delivered or applies valuation methods to provide monetary outputs, such as the value of sequestered carbon (Guerry et al. 2012). Individual EGS submodels can also be used independently if desired.

InVEST has been applied in many coastal planning contexts (Kim et al. 2012; Guannel et al. 2015; Oleson et al. 2017), though fewer coastal studies have taken advantage of the tool's submodel interconnections to analyze tradeoffs. The government of Belize partnered with the NCP in one such case study and used InVEST models to facilitate the development of an Integrated Coastal Zone Management (ICZM) plan (Arkema et al. 2015; Ruckelshaus et al. 2015; Verutes et al. 2017). With input from stakeholders and government agencies, the scientific modeling team identified three key ecosystem services (tourism, coastal protection, and the spiny lobster fishery) to analyze tradeoffs under three development scenarios. Simulating these scenarios in InVEST ultimately informed the development of an ICZM plan that was approved by the Belizean government in 2016 (Verutes et al. 2017).

The NCP continues to develop InVEST models in an on-going effort to improve connectivity with other complex models (e.g., Atlantis), enhance utility of submodels, and add new EGS models. Still, InVEST is most ideal for users with at least some spatially explicit data available from the area of interest. Gathering the spatial data required to run some of InVEST's coastal models can take considerable time and effort (e.g., see Bayani and Barthelemy 2016) and may not be feasible for all local decision makers. However, the ability to run core functions with less data and disable the more advanced model components helps to minimize this issue (Bayani and Barthelemy 2016).

6 ARIES Model Suite

A fundamental need to account for the spatial connectivity of ecosystem service supply and demand, with an explicit link to beneficiaries, led to development of the Artificial Intelligence for Ecosystem Services (ARIES) platform (Bagstad et al. 2013a; Villa et al. 2014a). ARIES has been refined through case studies, machine-learning, and technological advancements that now facilitates automated model customization to meet the needs of various decision makers (Villa et al. 2014a; Martínez-López et al. 2018).

As with InVEST, ARIES relies on spatially explicit input data and generates output that accounts for the location of beneficiaries, thereby closing the loop between ecological production functions that track resources (i.e., supply), to EGS of direct benefit to human users (i.e., demand) (Bagstad et al. 2013a; Villa et al. 2014a). ARIES accommodates both deterministic models and Bayesian networks to facilitate mapping of ecosystem service provision (Bagstad et al. 2014). Service Path Attribution Networks (SPANs) contain the ontologies for how services accrue in the system and are used to evaluate relationships between sources, sinks, use, and *'carriers'*. Carriers are the direct link between ecosystems and people that can facilitate provisioning benefits like drinking water, or preventive benefits like floodwater (Johnson et al. 2012; Bagstad et al. 2013a; Villa et al. 2014a, b). While not explicitly an IBM like HexSim, the flow of benefits is tracked using an agent-based approach via the discretized amount of a carrier from a source, to a use region or possible sink (Villa et al. 2014a, b). Carriers can be either physical (e.g., water) or informational (e.g., aesthetic views) and underlying SPAN models that vary based on the type of benefit, contain the rules dictating how a carrier moves in the system, including its absorption or ultimate delivery to an end user (Johnson et al. 2012; Villa et al. 2014a). Tangible benefits result from the accumulation of carriers by beneficiaries (Bagstad et al. 2013a).

The ARIES model suite, accessible through the open source software k.LAB (Knowledge Laboratory Integrated Development Environment), includes five baseline "Tier 1" ecosystem service models (Martínez-López et al. 2018). These base models (crop pollination, flood regulation, outdoor recreation, carbon storage, and sediment regulation) can be applied anywhere in the world without any new input due to existing datasets already built into the framework. In a basic model run, the user selects the spatiotemporal context, model resolution, ecosystem service of interest, and potential scenario conditions (optional). While users with data of greater spatial or temporal resolution can get further refined output, it is not necessary (Bagstad et al. 2014; Martínez-López et al. 2018). The supply component of the models quantifies the potential provision of a benefit, but it does not necessarily capture people's decisions to utilize ecosystem services (Villa et al. 2014a). Additionally, the models for crop pollination, outdoor recreation, and flood regulation estimate demand relative to other locations, thus output reflects ranked indicators, not biophysical values (Martínez-López et al. 2018).

ARIES models have been used to evaluate multiple ecosystem services including water provision, water quality, air quality, flood regulation, climate regulation, and aesthetic views (Bagstad et al. 2014; Ochoa and Urbina-Cardona 2017). For example, Bagstad et al. (2014) used ARIES to assess how aesthetic views translated to homeowner property values in Puget Sound, Washington (USA). Source features included mountains or waterbodies that added value, whereas view obstructions or blight areas were modeled as sinks (i.e., detracting from view quality). The flow models, which included distance decay functions, computed visibility along lines of sight between use locations and sources or sinks. Final results were a ratio between the values accrued by homeowners, relative to those of the entire landscape (Bagstad et al. 2014). A web-based ARIES 'explorer' is slated for release in 2020 and should facilitate greater accessibility and subsequent application in multiple settings (Martínez-López et al. 2018). The flexibility of the k.LAB software package and integration of more data-driven models (e.g., see Willcock et al. 2018) means that ARIES is easily customizable, and a growing user community may facilitate output more tailored to coastal beneficiaries and decision-makers.

Suites of models like ARIES and InVEST that have options to model multiple EGS over space and time, have indeed proven beneficial for coastal planning (Bagstad et al. 2014; Verutes et al. 2017). The true strength of these tools is their ability to take a comprehensive look at EGS and evaluate tradeoffs between potentially competing coastal EGS (Bagstad et al. 2013b). Integrating ecosystem service considerations into the coastal planning process can provide sustained benefits to coastal communities, and tools like ARIES and InVEST enable EGS to be more readily considered (Arkema et al. 2015). The ability to visualize EGS tradeoffs in a spatial context is beneficial to communicating the importance of ecosystems to stakeholders and this may help to optimize coastal planning by aligning scientific research with community priorities (Bayani and Barthelemy 2016; Verutes et al. 2017).

7 Common Difficulties, Emerging Issues, and Future Directions

Environmental changes are occurring at an unprecedented rate within coastal and estuarine ecosystems. Models capable of predicting alterations to coastal and estuarine EGS based on changes within these dynamic systems have much to offer scientists, resource managers, and decision-makers. Though these models vary in spatial scale, quantity of EGS modeled, and complexity, they share a number of common challenges.

7.1 Common Difficulties

Scale is important to understanding the context and applicability of model output. The importance and relevance of coastal and estuarine EGS span across the spectrum of spatial, temporal, and governance scales (Costanza et al. 2017). Identifying the best-suited model(s) given the scale(s) of a particular question or goal is paramount in the modeling process (Carpenter et al. 2009; Turner et al. 2016), as is utilizing data that corresponds to the same scale(s). Scale-dependence is tied to the accuracy and transferability of a model—some models can be applied across multiple spatial and/or temporal scales (Boumans et al. 2015; Francesconi et al. 2016; Lewis et al. 2019), whereas others are constrained to a single place and/or time. Difficulties can arise in interpreting outputs when existing models are applied at new scales or locations, an issue that is particularly relevant to coastal ecosystems, which can span a range of spatial and temporal scales (Swaney et al. 2012). Similar difficulties occur when gathering and applying model input data from a variety of sources and scales. Upscaling and downscaling are therefore common data manipulations in EGS modeling. For instance, results from small-scale experiments or processes will often be scaled-up for use in larger-scale models because of resource constraints that limit the scope of experiments and measurements (Craft et al. 2009; Peck et al. 2016). Downscaling, on the other hand, allows coarse output from global climate or systems models to be utilized in smaller-scale models (Peck et al. 2016). In both cases, scaling the original data introduces uncertainties (Andrew et al. 2015; Cheung et al. 2016) that impact model accuracy.

Uncertainty is an inherent component of modeling that should be acknowledged by users to avoid misinterpretation of results (Bagstad et al. 2013b; Peck et al. 2016). Communication of uncertainty to community partners and stakeholders is therefore vital (Guerry et al. 2012; Ruckelshaus et al. 2015); see Fulford et al. (2020) for a general discussion on communicating this uncertainty as an estimation of risk. Unfortunately, this uncertainty has not often been assessed systematically in coastal and estuarine ecosystem-scale models (Weijerman et al. 2015; Cheung et al. 2016). As the scale and complexity of a model increases, so typically does the uncertainty, which can be difficult to quantify for multiple interconnected submodels. Models that project changes in biological resources based on environmental alterations are also subject to uncertainties from model parameters, model structure, internal variability, and multiple scenarios (Arkema et al. 2015; Cheung et al. 2016; Marshall et al. 2017). Monte Carlo simulations, as demonstrated by Harris et al. (2018), are one approach that can provide deterministic models with an assessment of uncertainty due to errors in input data or parameter calibration. Tools based on probabilistic frameworks that facilitate a transparent characterization of uncertainty are also gaining traction (Bryant et al. 2018; Willcock et al. 2018). Although qualitative or bounded assessments of uncertainty provide some value to users, improved methods of assessment will need to be developed to progress toward deeper understanding of the uncertainty in applications of these complex models within coastal and estuarine ecosystems (Fulton et al. 2011; Cheung et al. 2016).

The complexity, as well as the time, cost, and data requirements, of many EGS models are barriers to widespread implementation (Chan and Ruckelshaus 2010; Plaganyi et al. 2011; Bagstad et al. 2013b; Willcock et al. 2018). A tradeoff exists between the complexity required for prediction accuracy and resource (i.e., time and cost) investment (Chan and Ruckelshaus 2010), the optimal balance varies among situations (see Fulford et al. (2020) for a general discussion on the tradeoff between parsimony and realism in regard to necessary complexity). The applicability of models requiring an extensive amount of input is limited for data-poor locations (Link et al. 2010; Bayani and Barthelemy 2016; Turner et al. 2016). A model's application can also be hindered in locations with ample data if a model's input data requirements are overly stringent (Bagstad et al. 2013b). As EGS models for coastal and estuarine systems become functionally more comprehensive and complex, the expertise needed to correctly apply them can become an obstacle to implementation. Improving a model's user interface may only superficially improve its usability, which may not increase the likelihood that it is ultimately applied correctly. The inclusion of adequate model documentation will aid in the application of the model by bringing transparency to its assumptions and limitations, while also providing adequate instructions and validation exercises (Bagstad et al. 2013b).

7.2 Emerging Issues and Future Directions

EBM emerged as an important and effective strategy for managing natural resources and EGS, in large part because of the explicit inclusion of humans within the ecosystem (Rosenberg and McLeod 2005; Guerry et al. 2012), which has increased the complexity and difficulty in ecosystems modeling. Although many EGS models have progressed, the consideration of human benefits that are difficult to monetize (e.g., cultural, existence, and subsistence values) is an area that is lacking (Chan and Ruckelshaus 2010; Plaganyi et al. 2011; Guerry et al. 2012; Liquete et al. 2013; Turner et al. 2016). The breadth of user groups and social behaviors considered needs to be increased to represent socio-ecological systems more realistically (Plaganyi et al. 2011), which will include potential interactions, feedbacks, and tradeoffs between groups and uses (e.g., see Fulford et al. 2016).

There is no single model that will fit the needs of every user, nor should there be; the best model will be the one most appropriate for the situation, data, and question at hand (Plaganyi et al. 2011; Peck et al. 2016; Turner et al. 2016). To improve model relevancy and usability, resource managers and other stakeholders must be part of the model development process to identify important decision metrics (Bagstad et al. 2013b; Costanza et al. 2017) and define the most practical management alternatives to test (Fulton et al. 2014; Turner et al. 2016). Decision-makers should have models available for different uses, scales of complexity, spatial resolution, and number of services, preferably models that can be coupled (Peck et al. 2016). Searchable inventories of EGS models like the EcoService Models Library (found at https://www.epa.gov/eco-research/ecoservice-models-library) may aid managers in the initial process of selecting the appropriate one. Perhaps the most optimal solution is a modeling framework that allows users to choose which components, groups, and interactions to include, similar to the modular designs of InVEST and ARIES (Guerry et al. 2012; Bagstad et al. 2014). With some modification, these tiered or coupled models can allow for the identification and inclusion of multiple drivers and cumulative impacts on coastal and estuarine EGS, which is a critical need (Carpenter et al. 2009; Chan and Ruckelshaus 2010; Little et al. 2017).

The concept of coupling models highlights another issue—error or uncertainty from input datasets can propagate through the model and significantly affect results. Validation provides users with a measure of confidence in model output, increasing the likelihood that a model's results will be accepted and used to inform the decision process (Bagstad et al. 2013b; Andrew et al. 2015). However, the availability of suitable data to validate and inform understanding is generally lacking (Mach et al. 2015; Plaganyi et al. 2011). For instance, ecological properties or processes within EGS models have often been parameterized with coarse data (Andrew et al. 2015), whereas many satellite-based land cover datasets have not been validated at all (Song 2018). Validation of outputs from the simulation of potential future scenarios is particularly difficult as the conditions simulated may not currently exist. The EGS model comparison study by Sharps et al. (2017), demonstrates the extensive validation (utilizing measured data for flow, water quality, soil carbon, and above-ground biomass) that is ideal when applying complex EGS models such as InVEST or ARIES.

While progress continues to be made on these issues, the lack of a standardized framework for implementation may continue to limit widespread adoption of EGS models into coastal and marine planning processes (Daily et al. 2009). In recent years, scientists working with decision-makers have been developing methods to formalize the inclusion of EGS in the planning process (Arkema et al. 2015; Ruckelshaus et al. 2015). Incorporating the estimation of impacts on EGS into the existing framework of Environmental Impact Assessments may be a promising route (Karjalainen et al. 2013). The DPSIR (Driver, Pressure, State, Impact, Response) is another, more conceptual, framework that has shown great progress incorporating EGS in decision making (Cranford et al. 2012; Kelble et al. 2013; Elliot and O'Higgins 2020). Starting in 2015, the European interdisciplinary research project AQUACROSS developed an assessment framework for the EBM of Europe's aquatic ecosystems and applied it in eight case studies (www.aquacross.eu). As more studies demonstrate the utility of EGS modeling within ecosystem-based management situations, knowledge and experience from these cases can be used as reference to further develop these frameworks (Forst 2009). The tools provided by the growing field of EGS modeling will undoubtedly aid scientists and decision-makers as they establish, validate, and apply innovative approaches to planning for estuarine and coastal change.

This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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An Integrated Multi-Model Decision Support Framework for Evaluating Ecosystem-Based Management Options for Coupled Human-Natural Systems

Robert B. McKane, Allen F. Brookes, Kevin S. Djang, Jonathan J. Halama, Paul B. Pettus, Bradley L. Barnhart, Marc Russell, Kellie B. Vache, and John P. Bolte

Abstract Simulation models offer a way to achieve a comprehensive understanding of the consequences of alternative community planning scenarios. For example, a community might want to understand how a particular decision—such as expanding an urban growth boundary into lands zoned for agriculture—will result in ecological, economic, and social tradeoffs for various stakeholder groups. This chapter explores the utility of ENVISION, a spatially-explicit decision support framework that integrates various ecological and human systems model "plug-ins" for informing Ecosystem-Based Management (EBM) options. While ENVISION already has a reasonably large tool box of such plug-ins, its usefulness could be further extended to address a wider range of community and ecosystem types. We specifically examine how a suite of existing U.S. Environmental Protection Agency decision support tools (VELMA, HexSim, CORESET, Coral PF and others) could significantly extend ENVISION's plug-in toolbox for coastal ecosystem EBM, inclusive of terrestrial-marine interactions and restoration goals of coastal communities dependent on marine ecosystem services.

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Lessons Learned

- Advances in applications of ENVISION over the past decade—e.g., the Willamette Water 2100 application—represent a major step forward for identifying EBM solutions to intertwined and seemingly insoluble (aka, wicked) environmental-economic-social problems.
- Depending upon the complexity and scope of EBM objectives, applications of ENVISION or similar human-natural systems modeling frameworks can involve substantial effort and cost.
- Costs/benefits of applications of ENVISION or similar tools should be weighed against alternatives, such as applications of stand-alone EBM models, reliance on purely empirical studies, or some combination.

Needs to Advance EBM

- Develop and apply model plug-ins that extend ENVISION's applicability to coastal ecosystems, particularly for identifying policies and EBM best practices for reducing sources and runoff of terrestrial pollutants to estuarine ecosystems.

1 Introduction

Communities invest significant time and resources in planning and want to understand the consequences of decisions they are considering. Those consequences can be complex and far reaching, affecting a wide range of stakeholders having different priorities. One way to attempt to understand the consequences of change is by using models. For example, a community might want to understand how forest harvest will affect a watershed's capacity to provide clean drinking water. For this context they might model the hydrology and biogeochemistry of selected areas to estimate how stream flow and nutrient loading may change as a result of logging. But the consequences do not end with flow and nutrients. How, where, and when the logging is done will also affect costs, jobs, revenue, fish and wildlife populations, and recreation inside and outside the harvested areas.

To gain a more comprehensive understanding of the ecological, economic and social consequences of any given land management decision option, the community would ideally use a framework capable of modeling these consequences in an integrated way, since each can directly or indirectly influence the others. Integrated modeling that supports Ecosystem-Based Management (EBM) can be very difficult and time consuming. To simplify the task, communities would benefit from a modeling framework with a tool box of plug-in models that can be tailored to address their specific needs.

Such an EBM framework would need to allow integration of varied plug-in models with a way for the models to share data. Much of the data will be spatial, so the framework must provide a shared GIS platform. In addition, there should be a way to test multiple scenarios and to provide optimization. Finally, there should be a way to provide constraints on what decisions can be made corresponding to a location's laws and policies (e.g., land use laws).

One such framework is ENVISION, developed primarily at Oregon State University (Bolte and Vache 2010; Santelmann et al. 2012; Spies et al. 2014; Bradley et al. 2016; Villarreal et al. 2017). ENVISION provides all the features listed above and is a mature, publicly available product (http://envision.bioe.orst.edu/Downloads.aspx). Using an extensible array of integrated plug-in models, ENVISION has been used to analyze ecological, economic and social tradeoffs in response to alternative future scenarios for coupled human-natural systems in a variety of locations. Some examples include the Puget Sound region in Washington, the Big Woods drainage in Idaho, and the Willamette River Basin and Tillamook Bay in Oregon. (http://envision.bioe.orst.edu/CaseStudies.aspx).

Goals of this chapter are to provide (1) an overview of ENVISION and an example of a regional-scale application; (2) an example of the process by which an existing spatial model can be integrated with ENVISION as a plug-in; and (3) recommendations for developing additional model plug-ins to further extend ENVISION's capabilities for integrated EBM planning. Our intent is to illustrate a path for integrating various existing EBM models within a well-established decision support framework that community planners and stakeholders can use to explore ecological, economic, and social tradeoffs associated with different decision options.

2 ENVISION—A Decision Support Tool for Ecosystem-Based Management

2.1 ENVISION Overview

ENVISION is a framework for constructing alternative future scenario applications concerning ecological, economic and social outcomes of interest to communities and regional planners. The framework consists of four main components (http://envision.bioe.orst.edu/About.aspx):

1. A dynamic spatial (GIS) engine for representing polygonal, network, point, and grid-based landscape characterizations.
2. A multi-agent modeling framework for representing values and behaviors of different decision-makers (agents) on the landscape.
3. A rich representation of policies guiding and constraining agent decision-making, and scenarios describing alternative strategies for landscape management.
4. An extensible plug-in architecture for including:
    (a) Any number of conformant autonomous process models describing landscape change dynamics;
    (b) Any number of "evaluative models" reporting landscape production metrics, typically measured in terms of scarcity along biophysical, ecological, social or economic dimensions;

(c) Any number of "visualizers" for visual representation of spatial data and inputs.

Figure 1 illustrates how these ENVISION components are integrated and feed back upon each other over time and space.

Other EBM decision support frameworks—notably InVEST and ARIES—share many of these capabilities (see Lewis et al. 2020). ENVISION stands out for its emphasis on the integration of ecological, social and economic models (plug-ins) with a sophisticated agent-based modeling subsystem. It also enables a rich representation of local and regional policies guiding and constraining actions of human decision-makers (agents) in landscape-scale simulations. Agents can be individuals, such as landowners and other citizens; or organizations and institutions, such as governments and businesses.

A main purpose of ENVISION's human-natural systems modeling approach is to facilitate discourse among different decision-makers and enable them to interactively play out and compare consequences of alternative management and regulatory choices.

2.2 *Example Application—ENVISION Willamette*

The ENVISION Willamette Water 2100 project—aka, ENVISION Willamette—is a good example of this framework's capabilities. ENVISION Willamette is a regional human-natural systems application that closely engaged community and regional decision-makers concerned with current and future supplies, usage and management of water resources within the 30,000 km^2 Willamette River Basin in the state of Oregon, USA (e.g., Bolte et al. 2011; Jaeger et al. 2017). The Willamette River is the 13th largest river in the USA. The river and its basin support a mosaic of agricultural, timber, recreational resources and several rapidly growing urban centers dependent on surface water supplies.

The ENVISION Willamette project involved a collaborative effort of Oregon State University, the University of Oregon, Portland State University, and the University of California–Santa Barbara. The project team used a structured decision-making approach to set up this ENVISION application (Fig. 2). Through a series of stakeholder engagement workshops, local, state, and federal stakeholders identified pressures and ecosystems services deemed important to their immediate and long-term ecological, economic, and social goals. Models of landscape processes relevant to these goals were then identified and plugged in to ENVISION. Plug-in models included forest ecosystem dynamics, land use change, human population growth, watershed hydrology, and storage of carbon within ecosystems (see next section for plug-in details). Relevant spatial datasets, policies and decision alternatives were then assembled to inform these models and build project scenarios.

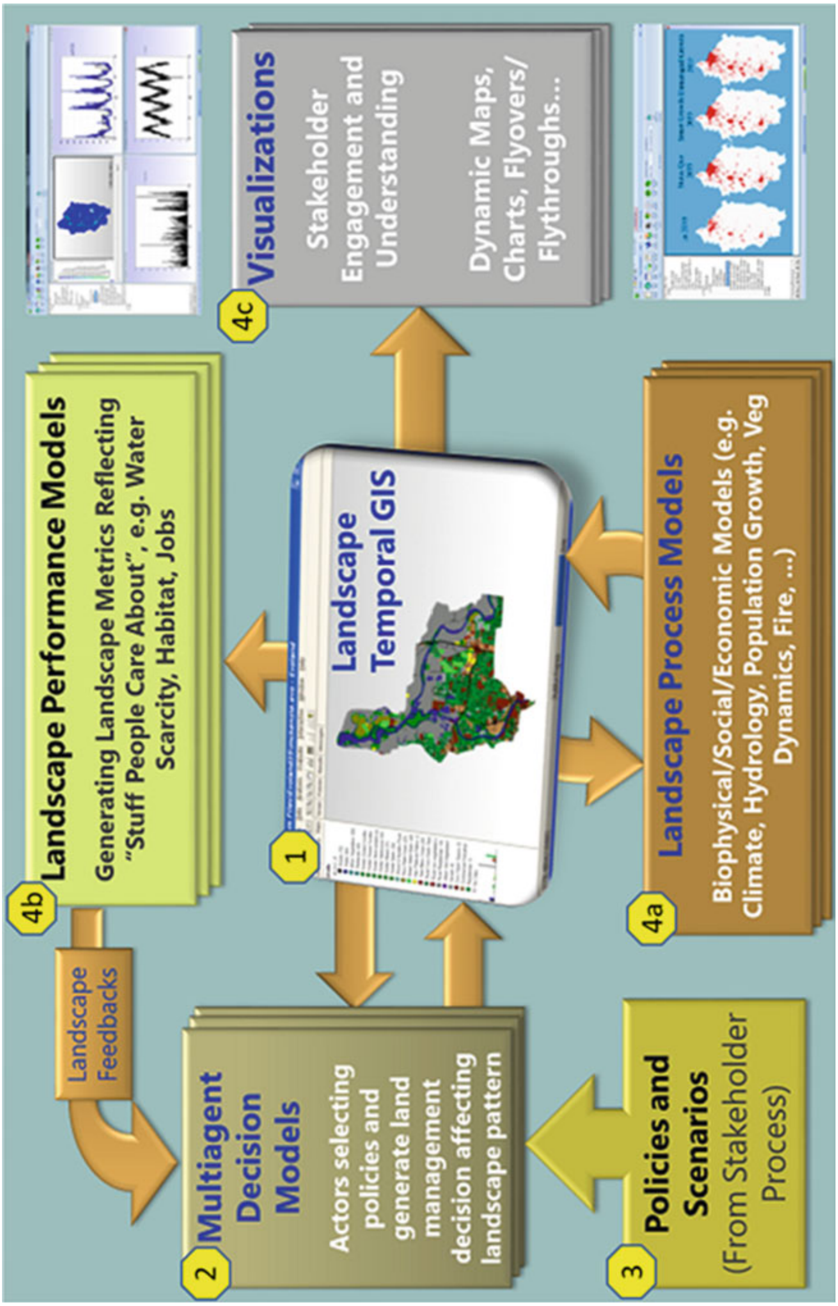

Fig. 1 ENVISION conceptual diagram. Numbers correspond to the list of four ENVISION components described in the text. (Reproduced from http://envision.bioe.orst.edu/About.aspx)

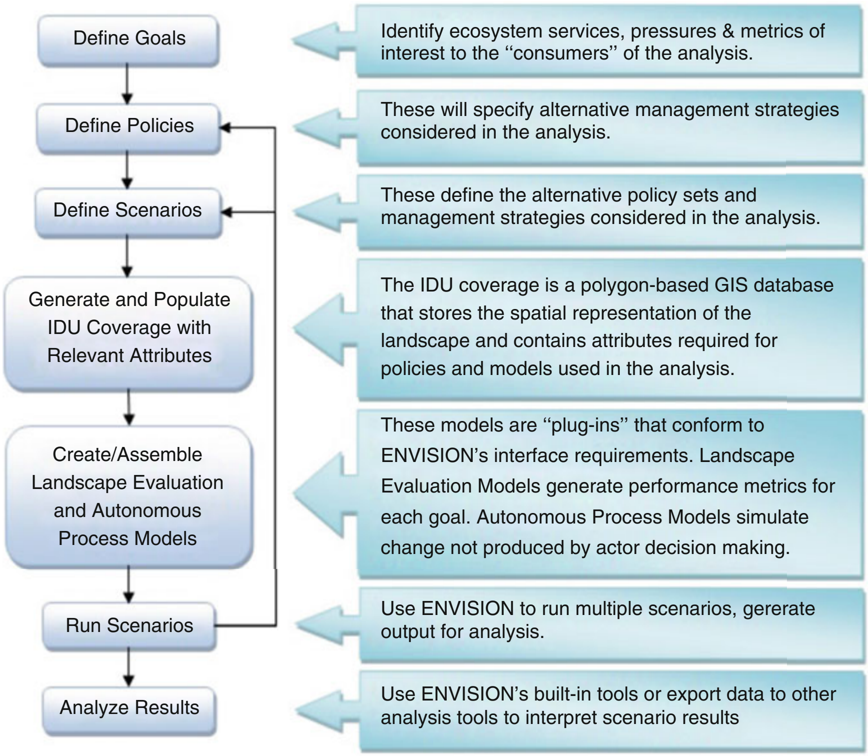

Fig. 2 Workflow for setting up ENVISION applications such as the ENVISION Willamette case study. “IDU” stands for Integrated Decision Unit, the user-defined landscape spatial unit. (Reproduced from http://envision.bioe.orst.edu/)

Modeled scenario results were evaluated, and iterative improvements were made, as necessary, to the workflow outlined in Fig. 2.

The resulting ENVISION Willamette application is a whole watershed model. Through its integration of various plug-in models, it attempts to represent the significant processes related to the supply and fate of water in the entire basin. These processes are both natural (e.g. precipitation, snow dynamics, infiltration, runoff, evapotranspiration) and human (e.g., reservoir operations, irrigation, municipal water use, crop choice). Application drivers include projections of climate, population, land use and income. The integrated ENVISION framework (Fig. 1) operates by simulating the processes across the entire basin per timestep—daily for some processes, annual for others (https://inr.oregonstate.edu/ww2100/analysis-topic/future-climate).

Key findings of the ENVISION Willamette Water 2100 case study address long-term (2010–2100) effects of an array of water regulations, land use alternatives, and climate and population scenarios on water scarcity, access to water, and associated ecological, economic and social tradeoffs.

A detailed summary of findings can be found on the ENVISION website (https://inr.oregonstate.edu/ww2100/key-findings) including several dozen peer-reviewed journal articles.

Briefly, ENVISION Willamette simulations broke out potential impacts for uplands (primarily forest) and lowlands (primarily urban and agricultural) during the twenty-first century. For example, upland snowpack is predicted to decline by 74–94% by 2100, limiting the region's main source of drinking water in summer, and contributing to as much as a ninefold areal increase in forest wildfires that, in turn, will reduce timber revenues and critical fish and wildlife habitat. In the lowlands, growing urban populations are predicted to increase water demand by up to 88%. ENVISION simulations also highlight water management implications and potential strategies for adapting to these complex and interconnected challenges.

3 ENVISION Plug-in Models

3.1 Existing ENVISION Plug-ins

This section reviews existing ENVISION plug-in models and their applications for the ENVISION Willamette project. In general terms, a plug-in is a software component that adds a specific feature to an existing software application, thereby increasing its functionality. Among the most familiar plug-ins are the innumerable "apps" that can be integrated into the operating systems of cellular phones to expand their functionality. Plug-ins can be very simple or quite complex depending on requirements.

ENVISION is distributed with a number of "standard" plug-ins (Table 1) that can be easily modified for specific applications. These are not required for ENVISION applications, but they have provided significant, commonly used functionality for many ENVISION applications.

Because of the complexity of the ENVISION Willamette application, a set of specialized plug-in models was also developed to achieve the project's hydrologic, ecological and socioeconomic goals. Examples include (1) *water system models* for simulating stream flow and temperature, and reservoir water management for the basin's stream/river network; (2) *ecosystem response models* for simulating effects of forestry, agriculture, fire and other disturbances on vegetation dynamics and stream habitat and fish populations; and (3) *socioeconomic models* for simulating changes in demand for drinking water and irrigation, and changes in land use, regulations, incentives and practices for managing rural and urban lands.

Additional details on ENVISION's standard and Willamette-specific plug-ins can be found in the ENVISION Developers Guide (http://envision.bioe.orst.edu/Downloads.aspx). The integration of these plug-in models within ENVISION provided a means for representing the complex feedbacks needed for estimating tradeoffs among ecological, economic and social processes and outcomes of concern to Willamette River Basin stakeholders. Characterization of such complex tradeoffs

Table 1 Standard ENVISION plug-ins

Plug-in name	Description
DynamicVeg	A sophisticated state transition model for vegetation in response to harvest, fire and other disturbances.
FLOW	Configurable hydrologic model representing water storage and movement within a landscape of user-defined polygons or a grid.
Reservoirs	Submodel of FLOW for defining individual reservoirs and their operations for holding and releasing water within a stream network.
WaterMaster	Submodel of FLOW that utilizes spatially explicit place-of-use, and point-of-diversion input data to simulate water right diversions as governed by a prior appropriated water-right system.
ModFlowAP	Implements the MODFLOW groundwater model.
Modeler	Allows users to define evaluative models for specified autonomous processes, including biophysical processes such as food and fiber production, water yield, etc.
ProgramEvaluator	Allows users to specify targets representing desired landscape condition. For instance, a community may want to preserve a certain percentage of the landscape in agricultural uses, or ensure some level of available buildable lands, or some other similar landscape statistic.
Trigger	Allows changes in one location to dynamically propagate changes into another location, once a specified threshold condition is met. For example, if a certain water quality regulatory threshold (TMDL) is exceeded at a given location, a remediation action such as implementation of riparian buffers can be triggered.
SpatialAllocator	Used for defining how processes (harvest, fire, fertilization…) are allocated spatially across the landscape to meet defined targets.
FlammapAP	Sophisticated fire behavior mapping and analysis program that computes potential fire behavior characteristics (spread rate, flame length, fire line intensity, etc.).
QuickFire	Simple fire model.
Target	Specifies a landscape variable whose total value across the landscape is represented as a trajectory.

within this human-natural system would not have been possible using the same or similar array of models applied in stand-alone mode.

3.2 Development of New ENVISION Plug-ins—VELMA Example

The list of ENVISION plug-ins is evolving as the framework is applied to new places and questions. In this section we provide an example of the general process by which an existing spatial model can be developed as a plug-in to extend ENVISION's capabilities for integrated EBM planning.

One of ENVISION's most pressing needs is a scalable, process-based biogeochemical model capable of addressing local and regional water quality and

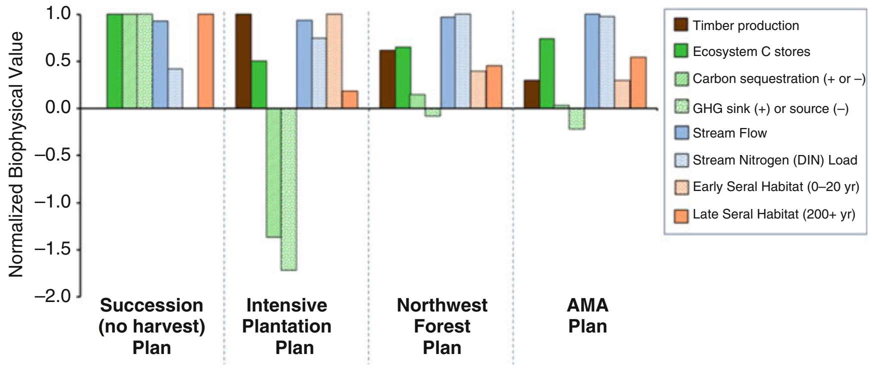

Fig. 3 VELMA results showing predicted ecosystem services and tradeoffs in the year 2200 when four alternative forest management scenarios (Cissel et al. 1999) were applied to present-day landscape conditions in the 230 km^2 Blue River watershed in Oregon. For each target ecosystem service—timber production, etc. (see legend)—simulated biophysical outputs (y-axis) across all four management scenarios are normalized with respect to the greatest simulated value for that ecosystem service. For example, timber production was at maximum for the Intensive Plantation Plan (y-axis value = 1.0), zero for the Succession plan (y-axis value = 0), and 62 and 30% of maximum, respectively, for the Northwest Forest Plan (NWFP) and the Adaptive Management Plan (AMA)

ecosystem services trade-off concerns. This gap could most easily be addressed by integrating a generalized landscape biogeochemical model plug-in with ENVISION's existing hydrologic FLOW plug-in (Table 1).

We chose the Visualizing Ecosystem Land Management Assessment (VELMA) model for this purpose. Developed by the EPA and Georgia Institute of Technology, VELMA is a spatially distributed, eco-hydrological model that links a land-surface hydrology model with a terrestrial biogeochemistry model for simulating the integrated responses of vegetation, soil, and water quality and quantity to interacting stressors. VELMA is applicable to essentially any terrestrial ecosystem (urban, agricultural, forest, grassland, prairie, wetland, etc.), as well as to mixed-use watersheds and regional basins (Abdelnour et al. 2011, 2013; McKane et al. 2014a, b; Hoghooghi et al. 2018; McKane et al. 2018a, b, c).

VELMA simulates how climate, land use, land cover and natural and engineered landscape features control the fate and transport of water, nutrients and toxics in watersheds, across scales ranging from small plots to large river basins, and from days to centuries. In addition to water quality assessments, VELMA has also been used to simulate climate and land use impacts on the capacity of ecosystems to provide a variety of ecosystem goods and services vital to human health and well-being—clean drinking water, flood prevention, food and fiber production, carbon sequestration, habitat for fish and wildlife, and others (e.g., Fig. 3).

3.3 Developing the ENVISION VELMA Plug-in

Using plug-in development procedures outlined on the ENVISION website (http://envision.bioe.orst.edu/Guides/CreatingPlugins.aspx), we developed a plug-in for VELMA's biogeochemical model, disconnected from VELMA's hydrologic model, and programmed to work interactively with ENVISION's existing hydrologic FLOW plug-in. FLOW provides many of the same functions as VELMA's hydrologic model, e.g., water infiltration, lateral surface and subsurface flow, stream routing, and evapotranspiration. Importantly, FLOW has the major advantage of being integrated with existing ENVISION plug-ins for informing management of the Willamette River basin's water, ecosystem and socioeconomic systems.

Figure 4 schematically illustrates the bidirectional exchange of information between the FLOW and VELMA biogeochemical plug-ins and, potentially, with ENVISION's standard plug-ins. All plug-ins are housed in a shared ENVISION library on disk. This library also contains plug-in input parameters (XML), environmental drivers (csv), spatial data (asc), and a project file (envx) that ENVISION uses to initialize a particular application.

The Flow XML file (HBV.xml in Fig. 4) describes a set of methods in the shared library that will be executed as part of the FLOW plug-in for each time step in the ENVISION application. The methods include (1) a hydrologic method that works on a specified landscape grid; (2) a built-in evapotranspiration method; and (3) the VELMA plug-in that is also placed in the shared library.

The VELMA plug-in is initialized using another XML file containing code that signals when the plug-in will be called, in this case at the beginning of each daily time step.

For testing purposes, a "BlueRiver_VelmaPlugin.xml" file was created and applied to demonstrate that ENVISION calls the VELMA plug-in, successfully engaging VELMA's biogeochemical model for the Blue River project application described above.

4 Potential Additional ENVISION Plug-ins for Coastal Ecosystem Applications

To further explore how ENVISION could be extended for EBM assessments, we compiled a list of existing and potential model plugins (Table 2) that are in current use by EPA and other researchers engaged in coastal ecosystem recovery planning.

Table 2 reflects our interest in the integration of models that have been or could be integrated for informing ongoing restoration activities across the EPA's National Estuary Program (NEP) (https://www.epa.gov/nep/local-estuary-programs). Many of these models have already been developed as ENVISION plug-ins for EBM projects at a number of NEP sites—Guánica Bay, Puerto Rico; Tampa Bay, Florida; and Puget Sound, Washington. Applications at these sites aim to assist community

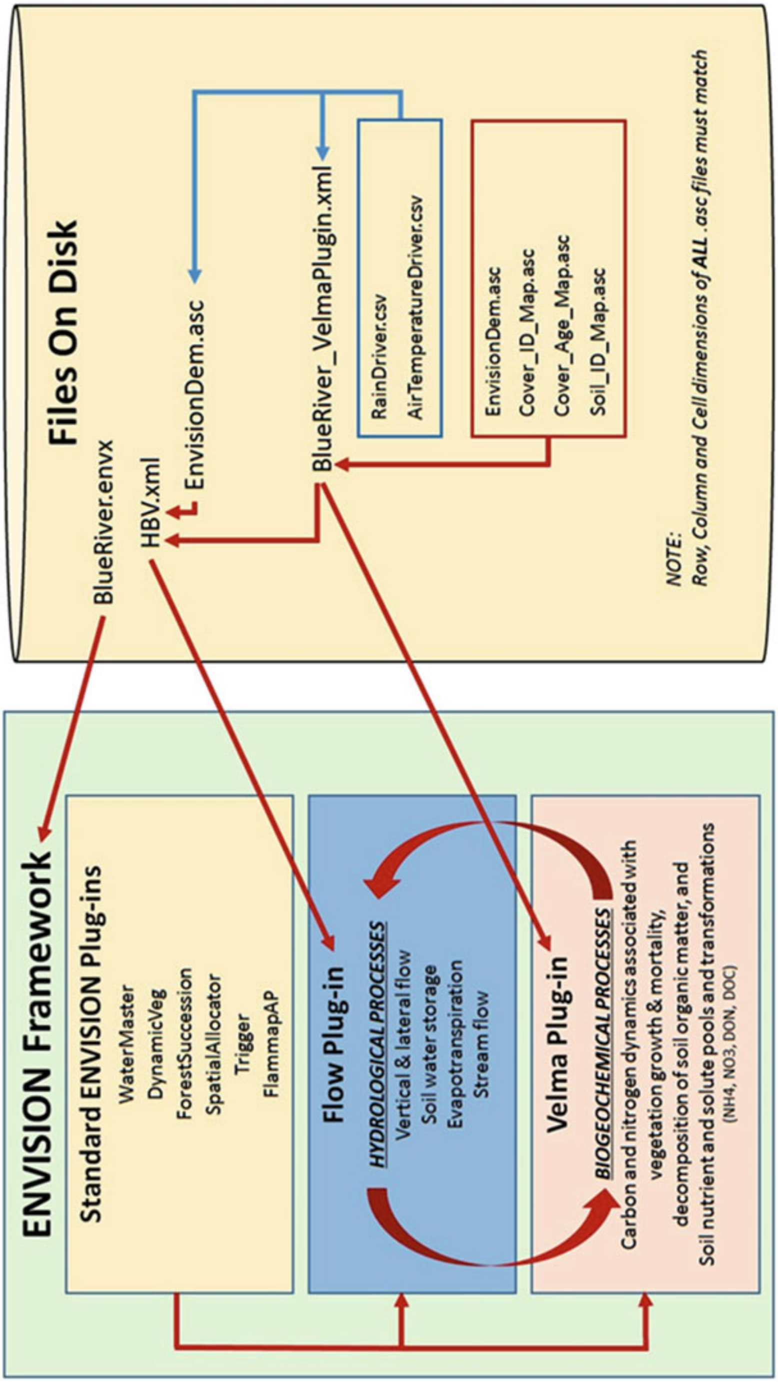

Fig. 4 Schematic showing data requirements and exchange of information among VELMA, FLOW and other ENVISION plug-ins

Table 2 Models currently in use by EPA for various community-based EBM projects designed to simulate ecological, economic, and/or human health outcomes for alternative decision scenarios

Model Name	Model Type	Landscape Unit, Scale	Timestep	Climate Regulation	Hazard Mitigation	Clean Air	Clean Water	Flood Prevention	Food, Fiber, Fuel	Carbon Sequestra-tion	Aesthetics, Recreation	Biodiversity	Human Health	Valuation
VELMA	Hydrology, Biogeochemistry	Grid, Flexible	Daily	√	√		√	√	√	√	√			
SMURF	Fish Habitat, Populations	Reaches, NHD	Seasons						√		√	√		
CORESET	Biodiversity Indicators	Grid, 500m	Yearly									√		
Coral PF	Coral Ecosystems	Flexible	Static		√							√		√
HexSim	Wildlife Habitat, Populations	Hexagons, Flexible	Flexible								√	√		
EPA H2O	Hydrology	NHD, Flexible	Static		√		√	√			√			√
HWBI	Human Well-Being Index	Flexible	Static										√	
Salish Sea Model	Estuarine Hydrodynamics, Biogeochemistry	Flexible	Variable		√		√		√		√	√	√	
CMAQ	Air Quality	Region, Continent	Sub-daily		√	√					√			
BenMAP-CE	Air Quality Health Impacts	Flexible	Daily			√							√	√

Existing ENVISION plug-in
Planned plug-in
No plug-in currently planned

Legend colors indicate whether an ENVISION plug-in has been or may be developed for a particular model. Listed models were developed in whole or in part by EPA, except the Pacific Northwest National Laboratory's Salish Sea Model. Model references: VELMA (Abdelnour et al. 2011, 2013; McKane et al. 2014b); SMURF (Snyder et al. 2019); CORESET (Melbourne-Thomas et al. 2011a, b); HexSim (Schumaker et al. 2014); EPA H2O (Russell et al. 2015); HWBI (Orlando et al. 2017); Salish Sea Model (Khangaonkar et al. 2018, 2019); CMAQ (Luecken et al. 2019); BenMAP-CE (Davidson et al. 2007; Berman et al. 2012)

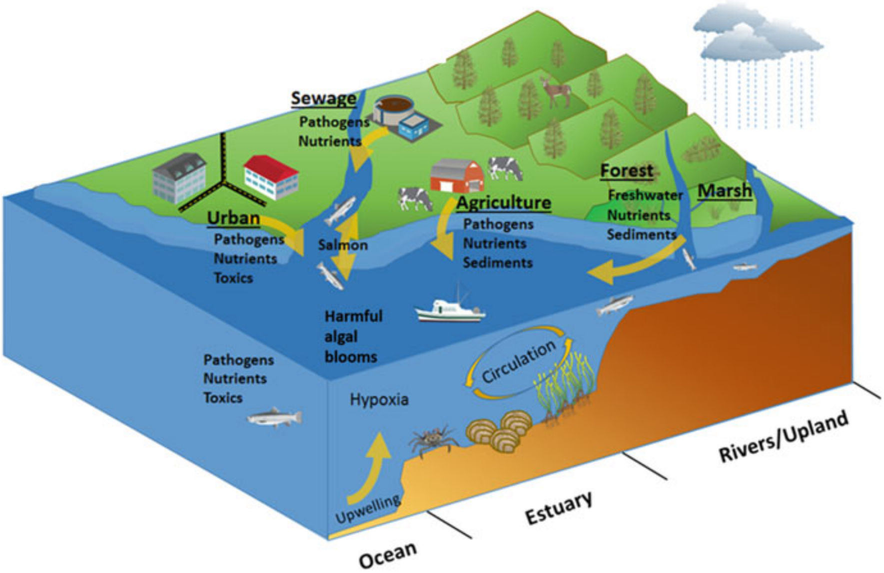

Fig. 5 Generalized representation of major terrestrial pollutant sources and transfers to marine waters in populated coastal ecosystems. Image: Darryl Marois

decision makers in balancing ecological, economic and social criteria over time-scales relevant to immediate needs and long-term planning goals.

A major challenge in modeling coastal ecosystems is the establishment of a coupled human-natural modeling framework capable of addressing transfers of terrestrial nutrients and toxic chemicals to marine waters, and consequent impacts on marine life and ecosystem goods and services. Figure 5 schematically summarizes typical coastal terrestrial-to-marine pollutant transfers, highlighting some major challenges for modeling EBM of NEP sites, such as Puget Sound's 31,000 km^2 basin and its mosaic of terrestrial, marine and human environments.

Characterizing such transfers and impacts is critical to estuarine restoration efforts in densely populated coastal areas. For example, in the Puget Sound National Estuary, killer whales (orcas) and their prey, Chinook salmon, have accumulated dangerously high levels of organic chemicals such as PCBs, PBDEs and PAHs (https://ecology.wa.gov/Water-Shorelines/Puget-Sound/Orca-task-force). This well-publicized situation has made these endangered species iconic indicators of the rapidly declining condition of Puget Sound's estuarine food web, perhaps foreshadowing a system-wide collapse such as those observed in Chesapeake Bay and other estuaries globally (Gelfenbaum et al. 2006).

We developed the ENVISION VELMA plug-in to better simulate EBM strategies for reducing transfers of terrestrial nutrients and contaminants to estuarine ecosystems in general, and to Puget Sound ongoing case studies in particular. Both ENVISION and VELMA have been extensively but separately applied to the terrestrial portion of the Puget Sound basin (e.g., Bolte and Vache 2010 for ENVISION; McKane et al. 2018a, b, c for VELMA). With VELMA's capabilities for estimating effects of land use and other disturbances on water quality, we anticipate that the ENVISION VELMA plug-in, when integrated with existing and proposed ENVISION plug-ins (Tables 1 and 2), can effectively extend ENVISION's functionality for addressing terrestrial and marine water quality and ecosystem service objectives pertinent to coastal ecosystem EBM goals. The State of Washington's Puget Sound Partnership has formalized such goals in terms of ~25 terrestrial-marine Vital Signs (ecosystem services), each monitored and managed under a specific Implementation Strategy (https://www.eopugetsound.org/articles/puget-sound-vital-signs-0).

Figure 6 conceptually illustrates ENVISION's flexibility for integrating existing and proposed plug-ins for coastal ecosystem EBM projects such as Puget Sound. This example includes a structured decision-making workflow involving development and application of stakeholder-relevant policies and decision scenarios, ecological production functions, ecosystem goods and services production functions, and benefit functions. The intent of such integration is to enable analyses of how alternative EBM options impact tradeoffs among ecological, socioeconomic and human health endpoints of concern to stakeholders.

In collaboration with Puget Sound community, tribal, state and federal partners, we have initiated development of such a human-natural systems framework, inclusive of coastal terrestrial-marine-human system interactions. This effort would merge the following:

- Existing Oregon State University ENVISION Puget Sound applications (Bolte and Vache 2010; http://envision.bioe.orst.edu/StudyAreas/PugetSound/).
- Ongoing EPA Puget Sound applications of VELMA (McKane et al. 2018a, b, c)
- Pacific Northwest National Laboratory applications of the Salish Sea Model (SSM) (Khangaonkar et al. 2018, 2019), coupled with National Oceanic and Atmospheric Administration applications of the Atlantis model (Levin et al. 2009), for simulating the circulation and fate of terrestrial pollutant inputs within the Puget Sound marine ecosystem and consequent impacts on water quality, fisheries and threatened food web species.
- Existing and new ENVISION plug-ins developed by EPA and others for extending ENVISION's applicability to coastal ecosystems (Tables 1 and 2).

Besides specific case study objectives, the overarching intent of these activities is to establish a generally applicable, coupled human-natural modeling framework that coastal ecosystem community stakeholders and restoration planners can use to anticipate and visualize how effects of EBM options in any particular location can propagate downstream and downcurrent with far-reaching benefits and tradeoffs for terrestrial and marine ecosystem services.

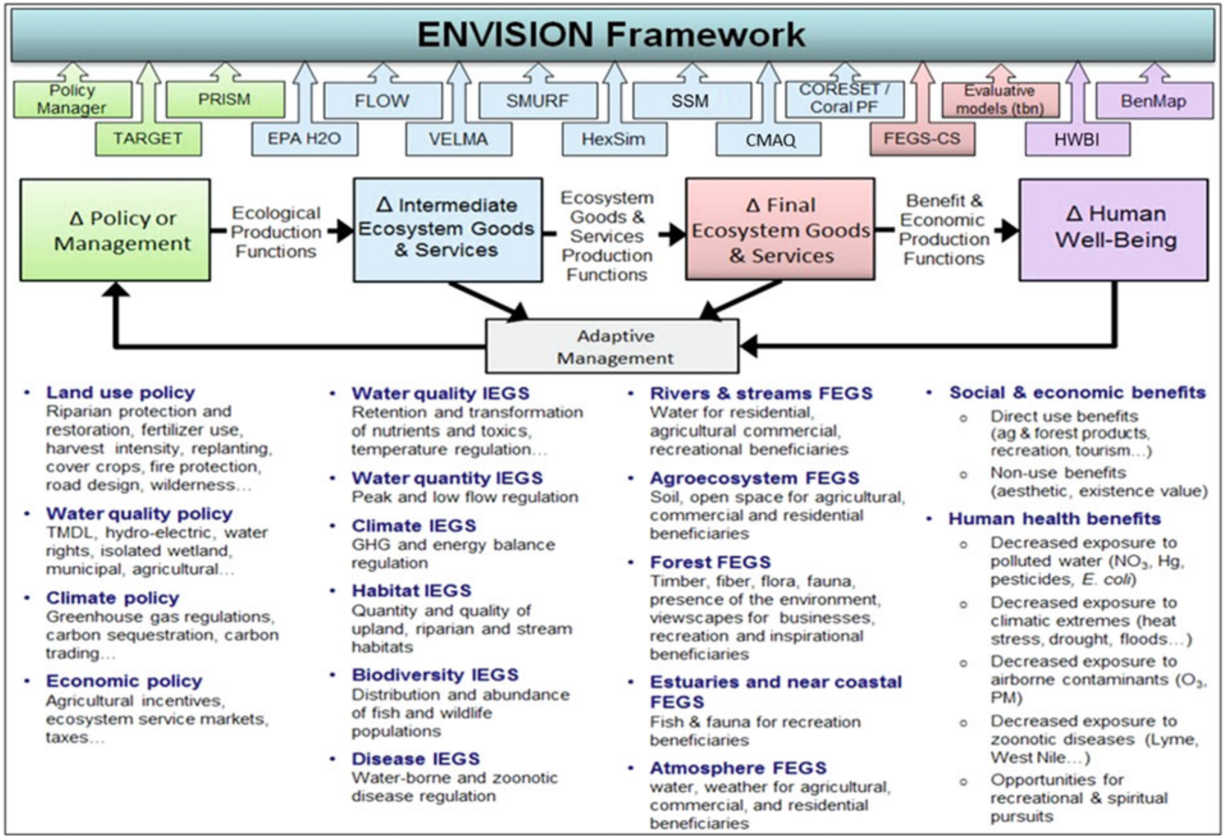

Fig. 6 Hypothetical ENVISION configuration for exploring the effects of alternative policies and land use decisions on intermediate ecosystem goods and services, final ecosystem goods and services, and human well-being for human-natural coastal ecosystems. This configuration includes existing and proposed plug-ins described in Tables 1 and 2, respectively. Integration of these plug-ins within ENVISION can potentially provide comprehensive tradeoff assessments for ecological, economic, social, and human well-being tradeoffs for alternative policies and EBM decision choices

5 Addressing Uncertainty Propagation within Multi-Model Frameworks

ENVISION and other multi-model decision support frameworks (Lewis et al. 2020) have emerged as important tools for informing EBM. A common concern about such frameworks is that uncertainties in one model can propagate and magnify as information gets passed from one model to the next, putting into question a framework's final outputs. Uncertainties in outputs can originate from uncertainties in input data that include measurement and processing errors; in individual models that include conceptual, mathematical, and computational errors; and in model-to-model incompatibilities such as differences in spatial and temporal scales that can propagate errors.

To reduce these sources of uncertainty, ENVISION employs a Monte-Carlo simulation approach that uses statistical descriptors of decision processes and, optionally, scenario-specific statistical descriptors of model inputs. This approach allows a given scenario to be run multiple times to produce a probabilistic distribution of possible outcomes (Bolte et al. 2012).

This uncertainty-based approach has additional advantages. For example, it is well-suited to the explore-then-test paradigm routinely emphasized in ENVISION applications to more effectively engage stakeholders in the design and analysis of alternative future scenarios (http://envision.bioe.orst.edu/CaseStudies.aspx). This approach also facilitates assessments of landscape vulnerability—the predisposition of a system to be adversely affected by stressors and lacking in capacity to adapt to environmental change (Bolte et al. 2012; IPCC 2014). And, when coupled with a multi-criteria assessment procedure (Hajkowicz and Collins 2007; Kiker et al. 2005), this approach can be used to estimate stakeholder weighting-of-importance for various ecosystem services.

6 Summary and Conclusions

This chapter has summarized key elements of the ENVISION decision support framework and how it has been used to help a wide range of communities develop and apply alternative decision scenarios in support of EBM planning. We used the ENVISION Willamette case study in Oregon to highlight key ENVISION features for this purpose. These include a dynamic GIS engine for representing changing landscape characteristics; a multi-agent framework for representing different decision makers; a rich representation of policies limiting decision maker's options; and an extensible array of model plug-ins for simulating ecological, economic and social tradeoffs in response to alternative future scenarios.

Using EPA's VELMA biogeochemical model, we demonstrated the relative ease of developing ENVISION plug-ins for enhancing water quality and ecosystem service assessments. We also demonstrated a hypothetical example and recommendations for developing additional model plug-ins to extend ENVISION's capabilities for integrated EBM planning in coastal ecosystems. This demonstration focused on our ongoing case studies in Puget Sound and other National Estuaries experiencing increasing terrestrial pollutant loads and consequent impacts on marine ecosystem goods and services.

Many of the recommended plug-ins already exist for extending ENVISION to simulate coastal terrestrial-marine interactions. The integration and application of these plug-ins via ENVISION would represent an important step in helping restoration planners and managers identify terrestrial pollutant sources and best practices for reducing terrestrial loadings to the marine ecosystem.

In conclusion, depending upon the scope of EBM questions and objectives, ENVISION and other coupled human-natural modeling frameworks (chapter "Projecting Changes to Coastal and Estuarine Ecosystem Goods and Services: Models and Tools") can require a substantial and expensive effort, potentially involving local, state, federal and academic organizations working in concert to assess how alternative decision scenarios may impact the capacity of local and regional ecosystems to sustainably provide vital ecosystem services.

That said, an argument can be made that the development and application of coupled natural-human modeling frameworks are a cost-effective means for discovering and guiding the implementation of EBM solutions to "wicked" ecological-economic-social problems. That is, problems that are "difficult or impossible to solve for as many as four reasons: incomplete or contradictory knowledge, the number of people and opinions involved, the large economic burden, and the interconnected nature of these problems with other problems" (Kolko 2012).

Obviously, no model can completely address these difficulties, but the integrative and analytical advantages of EBM frameworks such as ENVISION can at least provide informed and useful approximations based on best available data and science.

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Mathematical Modeling for Ecosystem-Based Management (EBM) and Ecosystem Goods and Services (EGS) Assessment

Richard S. Fulford, Sheila J. J. Heymans, and Wei Wu

"All models are wrong, but some models are useful"
George E.P. Box

Abstract There is a rich history in predicting ecological interactions in nature going back to the seminal work of Robert May. Historically, the use of models in ecological decision making has been centered on tools that require less data and are easier to communicate into policy. However, the explosion in the availability of ecological data, as well as the ready access to computer power, has opened the door to more detailed computation tools. This in turn has created a suite of questions about how and how much model-based projections can influence decision making in management of ecological resources. Here we address some of these issues particularly those created by model complexity, data quality and availability, and model acceptance by policy makers. Our goal is to use specific examples to critically examine these issues for both gaps and opportunities in how models can be used to inform decision making at the level of ecosystems and in the currency of ecosystem services to people.

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Lessons Learned

- Model complexity should fit the focal management question and the available data.
- The most useful ecosystem-based models are transparent with output that is easily translated into the language of policy. These endpoints are best accomplished with stakeholder engagement in model development.
- Multiple-model approaches, such as ensemble modeling, are better and more transferable than a single modeling tool.
- Model utility for Ecosystem-Based Management (EBM) decision making should be demonstrated through short-term forecasting efforts to improve the acceptance of models as a decision tool.

Needs to Advance

- Greater investment in ecological forecasting based on models as a testbed for model performance and improvement.
- Standardization of process for communicating model output in the language of policy including effective communication of model uncertainty as risk.
- Focus on improvement of existing models rather than new model development. This should be done in cooperation with policy makers early to improve model utility.

1 Introduction

Mathematical modeling has a rich history in ecology, but its foundations come from the physical sciences where well-characterized relationships such as reaction-diffusion are repeatable and well-validated (Getz 1998). The use of these relationships to address ecological questions has sprouted from multiple disciplines including population ecology (May 1974b), food web ecology (Whipple 1999), and nutrient dynamics (Clark and Gelfand 2006). What all these pathways have in common is the desire to use observable relationships to either predict or describe important but complex ecological outcomes. May (1974a) championed this concept in his seminal work using models to understand patterns in population dynamics. The work of May and others fostered debate on the utility of models vs. observation and fostered a modeling discipline in ecology that has grown steadily into a functional tool for management (Griffith and Fulton 2014; Nielsen et al. 2018; Piroddi et al. 2017).

Ultimately, we need models that are more than the sum of their parts and produce novel observations or theory useful for understanding nature. Nowhere is this more visible than in the effort to shift from management of single issues, such as nutrient load reduction, to a paradigm of ecosystem management (Heymans et al. 2018). Ecosystem dynamics are complicated and largely unobservable making models a pivotal tool for forecasting change (Heymans et al. 2016; Rose et al. 2010). The concept of ecosystem services further complicates the question, since social and economic dynamics not just ecological dynamics must also be considered (Austen et al. 2019). Models that incorporate all these dynamics are rare and largely unvalidated, yet the roadmap has been laid for such efforts and like the early days of ecological modeling the debate has shifted to incorporate models that predict ecosystem services (Nielsen et al. 2018; Zvoleff and An 2014). In this chapter we explore some of the issues related to ecosystem service modeling, first by defining it and then by exploring its possibilities and limits. This is not intended to be a thorough review of available ecosystem models, rather it is a discussion of where we are as a discipline and where we need to go to fully operationalize models for ecosystem decision making.

Ecosystem Goods and Services (EGS) are those elements of nature that directly benefit people. Models are well-suited to inform on an ecosystem good or service as the core value of models is to synthesize scientific information into a targeted endpoint. However, a focus on EGS as an endpoint for decision making requires socio/economic components, which are rarely included in ecosystem modeling (Bagstad et al. 2014; Fulford et al. 2015; Sanchirico and Mumby 2009). One example of interdisciplinary modeling for decision making is the integration of an economic model (Resource Investment Optimization System—RIOS) and an Ecosystem Services (ES) modeling tool (Integrated Valuation of Ecosystem Services and Tradeoffs—InVEST) to evaluate ES costs and benefits impacted by land use/land cover planning in the Thadee watershed in southern Thailand (Trisurat 2013, www.ipbes.net/resources/assessment reports). The more common approach is ecological modeling that forecasts EGS production without considering beneficiaries or assumes service delivery to be unrelated to changes in production (Craft et al. 2009; Redhead et al. 2018). Such limitations are not realistic and there is a need to broaden cross-disciplinary work in modeling to bridge this gap. The key stumbling block is to measure and communicate utility of models from different disciplines (Fulton 2010).

Model-based decision making is highly dependent on the accessibility and reliability of the chosen model. Accessibility encompasses ownership of the model framework, data availability as well as technical expertise, all which can affect a model's usability. Reliability considers the accuracy of model output, but also considers model acceptance based on how well the output is understood and the model's prior track record solving similar problems. Most of the issues to be discussed in this chapter find their origins in the need to demonstrate either model accessibility or reliability prior to application. There is typically a continuum of decision making from a purely policy-based to a data-based decision process. From the policy perspective, decisions are based largely on authoritative opinion. As the need for expert opinion grows, the decision begins to shift towards technical guidance and finally, when specific data are available, towards numerical decision rules. More complicated decisions are naturally shifted to the data end of the spectrum, however as problems become more complicated, so do the data. Ultimately the synthesis of that data into the language of policy can become a limiting factor. Here is where the value of models in decision making is most evident, as they are a powerful tool for synthesizing data. Yet, the process of data synthesis also yields a new challenge, that of communicating to non-technical decisions-makers. There is a balance to be struck between the amount of synthesis possible in any situation (e.g., how complicated is the problem?) and the ability to communicate the result well-enough to make it useful. In this chapter we take three examples of model applications to EGS in environmental decision making to illustrate how this balance was struck in each case.

The first example is taken from marine fisheries management and applies a population dynamics model to the challenge of maintaining a sustainable harvest in the context of both a variable fishery and a variable environment. Marine Fishery Stock Assessment (MFSA) has a long history of applying models to decision making. Stock assessment models such as the Stock Synthesis (SS) model (Methot 2009) are used to project stability of exploited populations in response to both fishery harvest and environmental variability (SEDAR 2016). Under MFSA models are usually combined with a formal process of stakeholder engagement to assure the right data and perspectives are utilized. The Gulf of Mexico Red Snapper (*Lutjanus campechanus*) stock (GMFMC 2018) provides an example. The sustainability of Gulf Red Snapper is assessed based on Maximum Sustainable Yield (MSY) i.e. the level at which harvest will not prevent the stock from replacing itself over the short term (~5 years). The SS model tracks population dynamics under different harvest scenarios and produces a prediction of MSY. Multiple model runs are used to account for several levels of uncertainty and model outcomes are combined to generate a best estimate of MSY, which is converted to a decision rule based on the uncertainty in the estimate and the level of acceptable risk of overfishing the stock. The models used in this assessment, as well as the input data for the model, are both well-validated and accepted for decision making (GMFMC 2018). In this example the ES is fishery harvest and the primary stakeholder (fishers) are engaged in the model-based decision process.

The second example comes from landscape management and involves a projection of how changes in land use practice in a watershed may impact the water quality of an estuary or other receiving water body (USDA-NRCS 2013). This example described the application of the Soil and Water Assessment Tool (SWAT) model (USDA NRCS; https://swat.tamu.edu/) to land conservation practices in Chesapeake Bay (CB) watershed USA. The SWAT model is a partially distributed hydrology model that plots water movement through a watershed to estimate the loading rate of nutrients and pollutants to the estuary. In the CB watershed, SWAT was used to examine changes in nutrient load between 2003–2006 and 2011 based on conservation practices in farm land mostly 'edge of field' retention of soil and nutrients. The model predicted a reduction delivered to streams in sediment (82%), nitrogen (44%), and phosphorus (75%). These results reinforce the value of land conservation on Bay water quality, but the information is not formally used to shape decisions. Rather the model output may be used to validate recommendations made to land owners for good conservation practice. In this example the ES is Bay water quality, which is an intermediate EGS leading to multiple benefits including recreational opportunities, shoreline land value, and aquatic biodiversity.

The third example comes from ecosystem management efforts in the Baltic Sea focusing on eutrophication over a large region using an ensemble modelling approach based on four different coastal eutrophication models (Skogen et al.

2014). The four models differed in coverage and approach but all projected coastal eutrophication and hypoxia responses to nutrient loading over time. The outcome was a map of eutrophication 'problem areas' in the Baltic Sea region that can be used to guide future conservation efforts. As in the case of the SWAT model, these results are not a part of a formal decision-making process but inform potential decisions aimed at reducing eutrophication. Ensemble models allow for different perspectives in modeling to be used and combined into a common recommendation (Skogen et al. 2014). In this example the ES is water qualtiy related to balanced primary production and a low incidence of hypoxia. These are intermediate EGS that contribute to fishery production, biodiversity, and human health.

Box 1 Central Questions in Making Model-Based Assessments Useful in Environmental Decision Making

1. Necessary complexity—How much detail is needed to address a management problem?
2. Operationalizing model output—Translating model output into the language of policy.
3. Transparent and transferable—Proper engagement with decision makers so they accept and use model-based information.

In this chapter we explore some of the dominant issues related to model-based assessments for environmental decision making based on ES assessments. We will use our examples from major areas of environmental decision making but we explore common ground by considering how models can be made more useful across important environmental issues. Three major issues exist in making models acceptable and reliable for model-based assessments of ES (Box 1). First models are meant to be simplified reflections of nature, but how much detail is needed? Modelers must address the question of how much model complexity is necessary for a given problem. Second, model output must be presented in a way that it is useful for decision making. This includes both translating model output into policy terms and properly communicating model uncertainty. The translation of model output and uncertainty into policy terms is called operationalizing the information. Graphic outputs and summarized metrics like ecological thresholds facilitate the operationalization process. Third, the model itself must be transparent and transferable to an issue and ecosystem. This means that not only is the model useful for a given problem, but it is also viewed as useful by policy makers and stakeholders, which may require early stakeholder engagement in model development. All these issues are necessary to make models useful for decision making, they are addressed here by examining some of the major issues in model choice and development.

2 Issues of Model Complexity

Quantitative models exist along a broad spectrum of complexity from simple empirical models (e.g. linear regression) to large mechanistic models that contain hundreds of parameters. By nature, simple empirical models are easier to conceptualize and fit as the parameters are simply components of a transformation from data to model output (X → Y). A good example is the exponential growth model (See Getz 1998), which describes exponential population growth and introduces the concept of carrying capacity in the case of limited resources. There is a tradition of parsimony in modeling that finds its roots in empirical models, since degrees of freedom are lost as fitting parameters are added (Getz 1998; May 1974a). In contrast, mechanistic models serve to recreate real pathways between data and model prediction in a way that allows for observable responses at the sub-model level but requires parameters that are themselves derived from empirical relationships. A simple example of a mechanistic model is the application of a functional response to predator-prey dynamics (Rosenweig and MacArthur 1963). The addition of data-derived parameters greatly adds to the complexity of the model, as well as the data requirements, but also provides a platform for examining the effects of real change that is not generally acceptable in empirical models where results cannot exceed the bounds of the data used to fit the model. If mechanistic models can be properly calibrated and validated for use, they become more useful for theoretical exploration of system change (Getz 1998). See Lewis et al. (2020) in this volume for additional examples of models across the complexity spectrum.

History of parsimony in modeling is derived largely from empirical modeling and its dependence on 'Goodness of Fit'. Empirical models are penalized based on inclusion of additional complexity if that complexity does not contribute significantly to the fit of the model (Burnham and Anderson 1998). However, that constraint is based on complexity that is not typically biologically interpretable. For instance, using a 3-parameter polynomial vs. a 2-parameter polynomial may improve fit, but adds very little to the interpretation of the results. This can be contrasted with mechanistic models commonly employed in ecological forecasting where new complexity may improve fit but also contributes to interpretation of the results (Scott et al. 2016). For instance, a well fit model predicting primary productivity in an estuary that does not consider plankton grazing is not *a priori* better as this is a significant loss term in real systems. In such a case, a modeler may accept a decline in the model fit to assure important dynamics are included. As data availability and computational power have increased, so has the attraction of increasing model complexity. Yet, neither of these improvements is a good reason to increase model complexity, rather that decision should be based on the trade-off between parsimony and realism (e.g., ARIES, Martínez-López et al. 2019). What rule should we use to justify model complexity in any particular situation?

The best approach is to match model complexity to the system and question in hand. Getz (1998) provided a review of model complexity relating it directly to the utility of models as a scientific tool. In that review, he highlighted the need to remain true to scientific principles of hypothesis testing to determine cause and the value of models for seeking a pattern that is believable and testable. Testability requires simplicity, but this goal is hard to apply in ecosystem science dealing with large complex systems. The question is how representative is the observed pattern in simplistic models? The second goal of predicting outcomes is much more aligned with ecosystem problems, but far less aligned with parsimony (Dietze et al. 2018). For example Bauer et al. (2019) assessed three models based on their ability to inform a comparison of fisheries management strategy, but their assessment was based not on information- theory comparisons of model accuracy (which reward simplicity) but on the range of disagreement among models in the predicted outcome. The review also considered model complexity in that the models differed both in approach and in the number of important issues each could consider (Bauer et al. 2019). The target was management guidance and the models were evaluated on how useful the results were for informing management. They concluded that complexity is needed where it applies to a known important dynamic in the ecosystem, such as competitive interactions between harvested and non-harvested prey.

The question of how much complexity is enough is a difficult one but is reminiscent of other threshold-based questions in computational science. The one common axiom is that you must pass over the threshold to identify it clearly (Hilborn et al. 1995). In modeling, the concept of necessary complexity has been suggested (Biebricher et al. 2012; Cahill and Mackay 2003) and we argue here that in ecological modeling it is necessary to overfit the problem at hand and then critically evaluate how much complexity is necessary to get a useable answer. Good examples of questioning simplifying assumptions comes from models that describe optimal behavior in animals (Alerstam 2011; Petchey et al. 2008). Response to change is a critical feature of forecasting ecological impacts of change in landscapes, water qualtiy, and food supply. Historically this type of modeling describes response to change based on the assumption that organisms will always respond to optimize their fitness in any novel situation (Doniol-Valcroze et al. 2011). Yet this approach is contrary to behavioral theory (McNamara and Houston 2009) and assumes an unrealistic level of knowledge regarding the spectrum of available resources (Fulford et al. 2011). Tools exist to predict non-optimal short-term behavior that both satisfy optimality theory and account for more realistic decision making in complex environments. Such tools pave the way for inclusion of additional complexity when necessary to the problem, such as for animal distributions in heterogenous landscapes (Rustigian et al. 2003) and mate choice predictions in fish reproductive models (Wooten 1984).

It is important to realize that the question of necessary complexity can be tied to the choice of model, such as with SWAT described above (USDA NRCS 2013). This mode-based analysis was intended to predict improvements in an ES (water quality) related to a management decision (land use conservation strategies) in the watershed. The model used was a semi-distributed hydrologic model with three effective vertical layers (surface flow, shallow ground water, and deep ground water). The conservation questions at hand all focused on surface flow, so loss to groundwater was a secondary element. In addition, the timescale was seasonal to annual and avoided examination of daily fluctuations in flow. This can be contrasted with fully-gridded models (e.g. Abdelnour et al. 2013), which are more detailed and optimized for measuring flow in multiple vertical layers at the scale of meters and days that might be needed to examine episodic storm events or annual forest harvesting strategies, rather than longer term landscape changes. The key is to approach any model problem by first critically evaluating the complexity that is needed to properly characterize the ecosystem and the issue at hand.

3 Communicating Model Uncertainty as Risk

A core objective of modeling applied to management is the meaningful assessment of risk. This is a key element of ES valuation and trade-off analysis among different competing services in the context of their impacts on social well-being (Spence et al. 2018). In the face of uncertainty and complex outcomes, models can and should communicate not just how the system will respond to change but also the probability and consequences of those responses in the context of policy. For example, in fishery management the goal is to reduce the probability of overfishing over a fixed time horizon (~5 years) (SEDAR 2016). Models are used to examine the probability of overfishing based on a suite of scenarios for future events (e.g., management limits, climate, recruitment variability), and these results are interpreted in the context of policy-driven risk thresholds (e.g., $P(\text{overfishing}) < 0.25$). Another example is the use of models in toxicological risk assessment to address the probability of adverse events (Forbes and Calow 2012). The challenge for traditional toxicological risk assessment is extrapolating from experimental data at the sub-organism level to population and ecosystem level effects. The use of models for toxicological risk assessment is controversial but shows promise as a method for extrapolation in cases where empirical data are unavailable (e.g., novel species) or unobtainable (e.g., ecosystems). The scientific community tends to vilify model uncertainty as a fault of the approach (Dietze et al. 2018), yet uncertainty is an intrinsic qualtiy of complex systems that necessitates a risk-based approach, and the quantifcation and communication of model uncertainty can contribute to the assessment of risk. For instance, in the fishery example from the Gulf of Mexico Red Snapper, the Stock Synthesis model is used to project the Over Fishing Limit (OFL) as the maximum allowable harvest rate for the stock. The goal here is to sustain the ES (fishery harvest) by managing fishing pressure in the context of other environmental influences (e.g., climate). Therefore, model output uncertainty is also reported and used to quantify the policy-based level of risk into a reduced maximum harvest value called Allowable Biological Catch (ABC). This value is the functional maximum rate used in management. See Lewis et al. (2020) in this volume for additional examples of quantifying uncertainty in specific models.

At the center of risk assessment based on model output is the proper quantification of model uncertainty. Model uncertainty can be parsed into uncertainty in data, parameter values, model functional choice, and underlying variability in the modeled system. Uncertainty within a model caused by input data and parameter uncertainty has received the most attention (LaDeau 2010; Nielsen et al. 2018). Several common tools exist for uncertainty assessment including Monte Carlo Simulation (Thorson et al. 2015; Zhang et al. 2012) and Bayesian Belief Networks (Schmitt and Brugere 2013). There is a strong need to standardize approaches for quantifying uncertainty so that non-modelers can better evaluate and compare results. Returning to the Gulf of Mexico Red Snapper example, the uncertainty analysis included examining sensitivity to parameter values, but also sensitivity to initial model conditions, timeseries dependencies from independent data, and model validation (SEDAR 52). These results were then converted to a cumulative estimate of uncertainty in the OFL value that was used to inform selection of ABC for the stock.

The key element for using models in decision making is the translation of model uncertainty into an estimation of risk. Risk has two components that must be considered in assessment of model output. First model output must be matched to a policy-based outcome (e.g., overfishing, HAB events). Frequently, this is the hardest step as model output is tied to the ecological dynamics not the policy-based objectives. For instance, in the case of Gulf Red Snapper the policy based risk component is standardized to a probability of 25% (SEDAR 2016). This value was derived from high level discussion of acceptable risk and all subsequent assessment outcomes must be reported in these terms by rule, which results in an operationalized model outcome. A comparable nutrient model might predict changes in nutrient concentrations through time and space but is less likely to synthesize those output into a time/space specific risk prediction. Policy makers must be engaged to make model output match the needs of management. In the nutrient case, that can be an estimate of the probability of exceeding Total Maximum Daily Load (TMDL) daily, which can be used to evaluate different strategies for reducing nutrient loading. Alternatively, a model might be adapted to estimate probability of harmful events given TMDL's are met, which is useful for assessment of different TMDL values. In either case, model uncertainty can then be evaluated as a component of estimating risk for the chosen outcome. This approach has been well-developed in fishery management where p(overfishing) has been defined so that it's incorporation into assessment models is straightforward. The same cannot be said for nutrient load management or habitat management, but this is largely a process limitation in that the tools exist but a mutually-acceptable process for inclusion does not.

4 Model Temporal and Spatial Scale

Model-based analyses are highly dependent on the choice of temporal and spatial scale and resolution. Models typically have an optimal resolution such as the SWAT model, which is a semi-distributed model designed to work best at medium resolution (e.g., Wellen et al. 2015) (seasons, km^2) vs. Ordinary Differential Equation Models designed to work at high temporal resolution but amalgamated over large spatial areas (e.g., Xu et al. 2011). The choice of a spatial and temporal resolution should be tied to the management question at hand and not the model. For instance, a fishery assessment of highly migratory coastal fishes does not require meter-scale resolution. In contrast, assessment of marine protected areas as a tool for increasing fishery recruitment may in fact require a high spatial resolution that can capture habitat features important to nursery production (e.g., Fulford et al. 2011). So, the choice to use a model also includes choices of scale that will impact the validity of the results. See Lewis et al. (2020) in this volume for additional examples of models optimized for particular spatial and temporal scales.

Simply considering both spatial and temporal variability in the same model is relatively novel in modeling as the issue of simultaneously quantifying both spatial and temporal uncertainty is challenging (Jager et al. 2005). For instance, in landscape management a wide variety of models with different spatial and temporal resolutions exist for examination of the impact of land use practices on nutrient loading into waterbodies. Historically, loading rates have been the focus of management, which do not require high spatial resolution (Hashemi et al. 2016). Yet, the impact of land use and the relative importance of surface flow vs. groundwater have greatly increased the need to examine nutrient loading at small spatial scales and in conjunction with short-term episodic events (Abdelnour et al. 2013). Models have had to keep up with this need. A potentially fruitful solution to scale issues is ensemble modeling or end-to-end modeling (Heymans et al. 2018; Lewis et al. 2020; Serpetti et al. 2017). Ensemble modeling was described earlier and end-to-end differs from ensemble modeling in that models are linked sequentially so that output from one model is input for another. Both approaches use multiple models and therefore open the door for examination of multiple temporal and spatial scales for a single system or problem. A good example of the application of multiple models to ES assessment is the ensemble model used to explore eutrophication problem areas in the Baltic Sea (Skogen et al. 2014). In this case, multiple spatial and temporal scales were involved in the collective analysis of eutrophication effects resulting from the interaction of nutrient loading and climatic forcing to generate common guidance for decision making. The result was the identification of 'problem areas' for eutrophication at several different scales, which makes the information more useful for decision making. As with other forms of complexity, spatial and temporal scale must be chosen deliberately to match the management problem of interest. This can be inhibited by fidelity to a particular model or limitations of available data for calibration and validation. Optimally, these three elements (model, problem, data) will be examined *a priori* to decide on what can be done so that operating at the wrong scale is not a criticism of the output. This is another opportunity for stakeholder engagement prior to any model-based analysis.

5 Connecting Science and Policy Objectives in Models

Models can be powerful tools for synthesizing scientific information to inform complex decision making if we can identify steps to operationalize model output (Box 2). Yet, for ecosystem-based management the use of models to justify decisions has not reached its full potential. If model-based assessments are needed to more fully inform decision making for ecosystem-based management, then modelers must meet the challenges described by making models more transparent and by communicating model output in the language of policy (e.g., ICES WGSAM 'Key run' model designation; https://www.ices.dk/community/groups/Pages/WGSAM.aspx). This will be greatly facilitated by early engagement with policy makers and other stakeholders in model choice and development, as well as formal processes for communicating uncertainty as a tool for risk management.

Box 2 Future Needs and Directions for Model-Based Decision Making

1. Standardize the process not the model—no model fits all problems and we should step away from promoting specific tools to promoting methods for tool selection. This includes a raised awareness of available tools and some standards regarding their evaluation.
2. Engage with policy makers early not late in the model development process.
3. Develop clear decision criteria that can be readily translated to and from model output.
4. Communication of model output should include uncertainty not as a limitation of the model but as a property of the decisional landscape useful for risk analysis.
5. Develop success stories based on ensemble modeling and provable short-term forecasting.

For models to be more widely used in decision making, scientists should also check the temptation to create new models. Rather, the focus should be on evaluating current models and providing practical suggestions on model selection for management decisions. Model libraries such as GULF TREE (http://www.gulftree.org/) and the EPA EcoService Models Library (https://esml.epa.gov/) provide search engines to help managers find the right tool for a specific problem. If new models are needed, decision makers should be involved in the development process and their opinions should be integrated into the modeling process (O'Higgins et al. 2020). Multiple model types can also be combined to improve usefulness of both predictive models, as well as utility models, such as neutral models to provide a benchmark for model performance and Bayesian Belief Networks, which can integrate qualitative information to make probabilistic predictions. The use of ensemble modeling holds particular promise for handling uncertainty, as it allows for multiple perspectives and the interactive exploitation of strengths of multiple models (Bauer et al. 2019).

There is also a need to develop proven track records for model-based assessments, which can only occur as a part of application to a practical problem. This can best be achieved through short-term forecasting exercises (Dietze et al. 2018) in real world situations. This approach demonstrates model abilities, but also provides a testbed for feedback and model improvement. Two key examples of this approach are in weather prediction (Bengtsson et al. 2019; Wu et al. 2019) and fishery stock assessment (GMFMC 2018). Future directions in model development for predicting ES should also include identification of such opportunities to engage in short-term forecasting in partnership with decision makers. Ultimately, the use of models in ecological decision making requires acceptance from policy makers and the public and that is where the most effort if needed.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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The Ecosystem Services Gradient: A Descriptive Model for Identifying Levels of Meaningful Change

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Abstract Characterization of ecosystem services can be a valuable element of Ecosystem-Based Management (EBM) in identifying meaningful measures of ecosystem change, understanding the natural resource gains or losses associated with changing ecosystem conditions, and communicating those benefits and tradeoffs to stakeholders in an intuitive way. Here, we introduce a descriptive model of the Ecosystem Services Gradient (ESG) that can be paired with the Biological Condition Gradient (BCG). The BCG is a conceptual framework that allows scientists and managers to characterize the status of an aquatic ecosystem along an anthropogenic disturbance gradient by describing and quantifying changes in biological or ecological condition with increasing levels of stressors. The ESG descriptive model builds upon the BCG approach by linking changes in ecosystem condition to effects on human health and well-being via changes in ecosystem goods and services. This involves identifying priority ecosystem services, defining them with metrics and indicators, and applying ecological production functions to translate levels of

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ecological condition to ecosystem services production. The ESG, through its structured approach to defining and enumerating potential changes in ecosystem services, allows decision makers to clearly assess and monitor the potential benefits, or related co-occurring benefits, of EBM, and significantly enhance how scientists and decision makers communicate these benefits to stakeholders.

Lessons Learned

- An Ecosystem Services Gradient (ESG) is introduced to describe the complete range of potential ecosystem services along a gradient of changing environmental condition
- The ESG approach leverages the concept of Final Ecosystem Goods and Services (FEGS) to identify metrics that are directly relevant to human beneficiaries
- An ESG can allow decision makers to describe meaningful and unambiguous measures that clearly communicate the potential gains or losses in ecosystem services
- The ESG facilitates a consideration of potential tradeoffs, or co-benefits, across multiple stakeholder objectives as part of EBM planning and implementation

Needs to Advance EBM

- Additional scientific research is needed to move from a narrative description of an ESG to a quantitative description that enumerates ecosystem services production with changing levels of condition
- Further development of the ESG approach is needed through case study examples across a range of ecosystem types and EBM applications

1 Ecosystem-Based Management Objectives and Tradeoffs

Ecosystem-Based Management (EBM) aims to maintain ecosystems in a healthy and resilient condition while providing the services that humans want and need (McLeod et al. 2005). However, ecosystems are complex, and layering on social and economic considerations can make operationalizing EBM seem intractable (Arkema et al. 2006; Link and Browman 2017). For successful implementation of EBM, there is a specific need to bound the scope of the problem by clarifying what really matters about a decision, including explicitly articulating how objectives will be measured and characterizing values-based tradeoffs among them (Gregory et al. 2012). To address this need, we propose a science-based descriptive model of ecosystem services production in response to changing environmental condition, the Ecosystem

Services Gradient (ESG). Scientific tools and approaches, like the ESG, can help to operationalize EBM in the decision-making process by identifying meaningful measures, defining reference points, communicating and monitoring the relevant social and economic impacts of actions, and evaluating tradeoffs across multi-sector objectives (Arkema et al. 2006; Cormier et al. 2017).

The conceptual foundation for an ESG follows that of the Biological Condition Gradient (BCG), developed over a decade ago in response to growing need to assess and effectively communicate levels of biological condition in a meaningful way (Davies and Jackson 2006; U.S. EPA 2016). The BCG leverages expert knowledge and biomonitoring data to describe ecological condition along a gradient from undisturbed to severely altered conditions. Our goal in creating an ESG framework is to build upon the original goals in developing the BCG: to create a common framework, based on measurable ecologically important attributes, that can be used to describe the complete range of condition, and provide a rational and consistent means for setting targets and communicating the consequences of different management choices.

The ESG leverages a number of practical strategies for integrating ecosystem services into decision-making, including: (1) prioritizing information and analysis to what is most important; (2) using the concept of *final* ecosystem goods and services (FEGS) to identify metrics that are unambiguous and directly relevant to human beneficiaries; (3) applying ecosystem services production functions (EPFs) to link changing condition to changes in ecosystem services; (4) understanding the range of potential outcomes; and (5) considering values-based tradeoffs across multiple, often competing, objectives (Yee et al. 2017). In this chapter, we present the conceptual foundation for the ESG as an analogy to the BCG and provide examples of how it is being developed to facilitate EBM.

2 Conceptual Foundation: The Biological Condition Gradient (BCG)

The BCG is a descriptive model that describes how attributes of biological condition change in response to increasing levels of anthropogenic stress (Fig. 1a; Davies and Jackson 2006). The BCG approach was developed to address a need for science-based approaches to more precisely and effectively communicate the existing and potential condition of aquatic resources for water quality management purposes under the U.S. Clean Water Act (United States Code title 33, sections 1251–1387). The biological characteristics, defined in the BCG as "attributes," include aspects of community structure, non-native taxa, organism condition, ecosystem function, and inter-habitat connectivity. The highest level of biological condition is Level 1, which represents natural or undisturbed biological communities and anchors the best condition for defining five levels of change or departure from this condition. Level 6 represents conditions most severely altered by anthropogenic

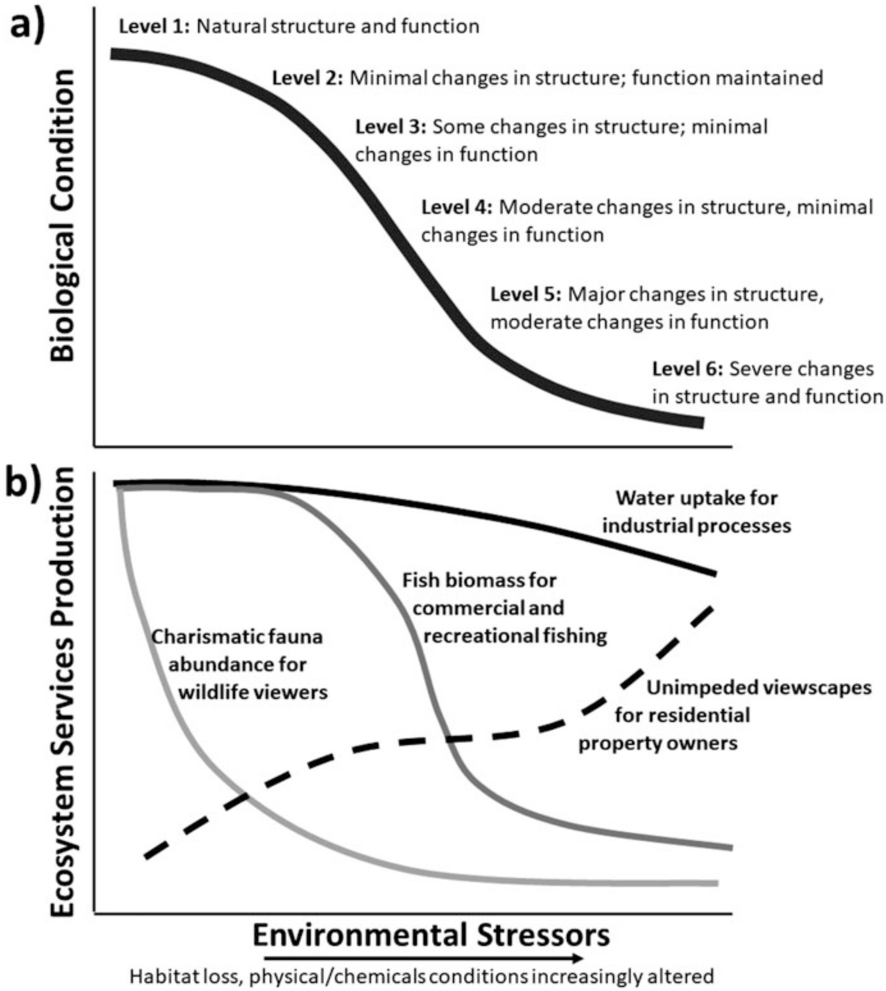

Fig. 1 The BCG model (**a**; top panel) of incremental changes in biological condition along a stressor gradient, and hypothetical changes in select ecosystem services (**b**; bottom panel) along the same gradient

stress. Each level is defined by an empirically-derived narrative description that can be consistently interpreted regardless of biology, location, or sampling method. A quantitative model is derived from narrative descriptions for each level and translated using metrics and measurable indicators to develop quantitative decision rules to identify thresholds to discriminate between BCG levels (for details see U.S. EPA 2016; Cicchetti et al. 2017).

In BCG development, a specific sequence of steps is undertaken for a given ecosystem to develop the BCG components (Table 1; U.S. EPA 2016). Because this structured approach is generalizable, BCG models can and have been developed for

Table 1 Steps in the process of developing and using a BCG

Biological Condition Gradient framework	Process
What biological attributes are relevant?	Identify and prioritize attributes
How will we measure them?	Identify metrics and indicators
What biological condition did we have?	Establish reference (natural) condition
What biological condition do we have now?	Collect and review bioassessment data
What biological condition do we want?	Set targets
How do we get there?	Identify management actions
What are the social and economic consequences?	*Conduct and communicate ecosystem services assessment (ESG)*

Adapted from Cicchetti et al. 2017
An additional step (in *italics*) indicates where an assessment of ecosystem services could supplement the process, and in conjunction with stressor and other data, inform management decisions

different regions and different ecosystems, including streams (Davies and Jackson 2006), estuaries (Cicchetti et al. 2017), and coral reefs (Bradley et al. 2014; Santavy et al. 2016). Though originally developed for aquatic ecosystems, the approach is applicable in terrestrial ecosystems as well.

The BCG can help precisely define biological condition, identify and protect high quality waters, evaluate the potential for improvement of degraded waters, select restoration targets, and clearly communicate the likely impacts of management decisions to the public. The additional step of assessing how ecosystem services change with corresponding levels of BCG (Fig. 1b) can help to communicate the social and economic benefits of protecting or restoring a site, or potential tradeoffs between different management scenarios (Cicchetti et al. 2017).

3 The Ecosystem Services Gradient (ESG)

3.1 Interpreting the ESG

Building on the conceptual foundation of the BCG, the ESG describes the complete range of ecosystem services along a gradient of biological condition from natural to severely altered. In environmental management situations where protecting biological integrity is the primary goal, directly pairing an ESG with a BCG can help decision-makers understand the potential co-occurring benefits and tradeoffs of management activities and communicate them to the public (Fig. 1). Furthermore, if a waterbody is designated for a particular use, such as recreational fisheries or contact recreation, an assessment of ecosystem services in conjunction with BCG can help identify the levels of biological condition that can be protected while still supporting desired levels of services (Davies and Jackson 2006).

Along a gradient of declining biological condition, ecosystem services may decline at different rates depending on the biological attributes providing those services (Fig. 1b). The quality of a recreational fishery, for example, may depend on the presence of uncommon taxa that are particularly vulnerable to stressors, whereas commercially-important fish species may be able to persist with some moderate degree of habitat degradation. Other ecosystem services may remain relatively unaffected along the gradient if the attributes that define biological condition are disconnected ecologically from the attributes providing the service. For example, the ecosystem service of water availability for use in industrial applications may be only partially influenced by ecosystem condition. Other ecosystem services might increase with declining biological condition. For example, depending on what local residents or recreational users consider to be aesthetically pleasing, charismatic species or unimpeded viewscapes may increase in value as presence of habitat or condition declines.

Operationally, an ESG may be directly paired with a BCG (Fig. 1). Table 1 describes an approach where an added step to the BCG process, ideally occurring during BCG development, could involve an assessment of ecosystem services. However, because the attributes that define biological condition in the BCG may not be the same attributes providing ecosystem services, EBM practitioners may prefer to develop an ESG independently of a BCG. Analogous to a BCG, the ESG would describe the full range of potential ecosystem services provisioning along a stressor gradient. Moreover, different biological attributes contribute to different ecosystem services, such that a suite of ESG curves may be needed for describing a range of different ecosystem services in a given system. However, the underlying approach in building the ESG is the same regardless, differing only in how the descriptive model is presented as either a gradient of decreasing biological condition or a gradient of changing service production (Fig. 1).

3.2 Steps for Developing an ESG

The steps to building and using an ESG closely parallel the steps to develop and use a BCG (Table 2). An important first step is working with decision makers and stakeholders to identify the relevant ecosystem services for the specific environmental management problem. A FEGS approach can help reduce ambiguity by explicitly and *directly* connecting biophysical indicators to the people that benefit from them (Fig. 2; Boyd et al. 2015; DeWitt et al. 2020). Along a continuum of ecological production, FEGS are distinguished from *intermediate* regulating and supporting ecological functions (e.g., habitat quality, water quality) that require additional steps to reach the ecological features (e.g., harvestable fish) directly experienced by human beneficiaries (Landers and Nahlik 2013).

Monitoring data on FEGS metrics form the basis for quantitatively describing levels of production from highest potential production to severely altered production along a gradient of changing environmental condition. While environmental

Table 2 Steps in the process of developing and using an ESG

Ecosystem Services Gradient Framework	Process
What *final* ecosystem goods and services (FEGS) are relevant?	Identify and prioritize FEGS
How will we measure them?	Identify FEGS metrics and indicators, and the bio-physical attributes that provide them
What FEGS could we have?	Establish potential availability under a range of bio-physical conditions
What FEGS do we have now?	Measuring, mapping, and ecological production functions (EPFs)
What FEGS do we want?	Evaluate co-occurring benefits and tradeoffs
How do we get there?	Identify management actions
What are the social and economic consequences?	Conduct and communicate benefits assessment using ecological benefit functions (EBFs)

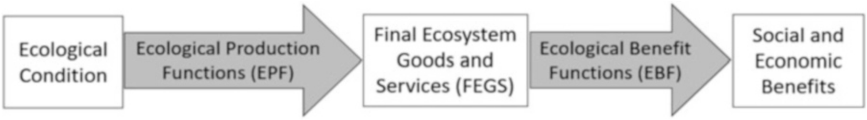

Fig. 2 Conceptual model illustrating the relationships between ecological condition, FEGS, and socio-economic benefits

assessments and monitoring often focus on collecting data on ecological condition, development of an ESG may rely on reasonable proxies where direct data or models are not available. Models, known as ecological production functions (EPFs), may be needed to translate environmental condition data to FEGS metrics (Fig. 2; Wainger and Mazzotta 2011; Bruins et al. 2017). This combination of expert judgment on meaningful metrics, collection of field data, and application of EPFs is used to first narratively and then, ideally, numerically describe incremental changes in ecosystem services provisioning along a stressor gradient to form the ESG. If ESGs are developed for more than one ecosystem service (e.g., Fig. 1b), then potential co-occurring benefits or tradeoffs can be examined alongside changes in ecosystem condition.

While FEGS represent the end product of what the environment provides to human beneficiaries, they require human input and interaction (e.g., a boat to collect the fish) for those services to be realized as actual benefits (Mazzotta et al. 2016). As such, a benefits assessment, using ecological benefits functions (EBFs) to translate ecosystem service supply into monetary, health, or other measures of benefit, could be an additional step for characterizing and communicating the benefits of EBM decisions (Fig. 2).

4 Example ESG Applications

4.1 Communicating Benefits of Coral Reef Protection

Healthy coral reef ecosystems supply multitudes of benefits on which many economies and societies rely (Wilkinson 2008; van Beukering et al. 2011), including recreation such as fishing, tourism, boating, SCUBA diving; education; coastal protection; and bioprospecting for novel pharmaceuticals and biochemicals (Moberg and Folke 1999; Principe et al. 2012). Marine coastal areas, including coral reefs, are exposed to increasing loads of nutrients, sediments, pollutants, and other materials originating from terrestrial sources that can deleteriously impact the ecosystem goods and services they provide and place them at risk of being lost (Harborne et al. 2017). Consequently, there is continuing urgency to develop tools to effectively communicate this information to improve public awareness of reef condition; understand what actions are most likely to protect these irreplaceable ecosystems; and provide a more robust process to inform management of the biological condition of coral reefs to ensure protection of high quality marine waters and their biological communities, and to develop restoration targets.

The framework used to develop the BCG model for freshwater streams, rivers, and lakes was adapted to incorporate coral reef attributes judged important to protect the biological integrity of tropical Caribbean and Western Atlantic waters, including marine coastal habitats such as mangroves, seagrasses, and coral reefs (Bradley et al. 2014; Santavy et al. 2016). From a preliminary narrative model of all coral reef assemblages (Table 3), two narrative BCG models were developed for Caribbean coral reefs, one for coral reef fish and a second for sessile marine assemblages, built primarily using attributes from scleractinian coral communities, but including algae, sponges, and octocorals (Santavy et al. 2016). For each, a numeric BCG model is being developed by eliciting expert knowledge in combination with bioassessment data and underwater videos, using mathematical fuzzy set theory to define decision rules for BCG levels (U.S. EPA 2016).

In order to build upon these coral reef BCGs and develop an ESG for coral reefs, the first step would be to identify the most relevant ecosystem goods and services for the particular decision context (Table 2), recognizing that ESGs would need to be developed on a site-by-site basis so that each ESG properly reflects the makeup of human beneficiaries at that site. For example, a coral reef that is situated within a Marine Protected Area that limits or bans many consumptive activities will provide a very different set of FEGS than a coral reef that has fewer use restrictions. Once the stakeholder groups associated with the reef have been clearly identified, the ways in which each of those groups benefit from the reef can be identified and clearly articulated. At this point, the FEGS necessary to achieve those benefits can be identified. This can be done in an ad hoc fashion or using a more structured approach such as the National Ecosystem Services Classification System (U.S. EPA 2015) or the Common International Classification of Ecosystem Services (CICES; Haines-Young and Potschin 2018) as a starting point to identify potential FEGS. Because

Table 3 Examples of narrative condition levels and associated attributes in coral reef BCG

Condition level	Physical structure	Corals	Fish, other vertebrates and invertebrates	Gorgonians, sponges, algae	Condition
Very Good Excellent BCG Level 1–2	High rugosity; very clear water; no sediment	High species diversity; includes rare, large, and old colonies	Balanced species abundances and sizes; large, long-lived species present	Low abundance of fleshy algae; sensitive species present	Low prevalence of disease
Good BCG Level 3	Moderate to high rugosity; water slightly turbid; low sediment	Moderate coral diversity; rare species absent	Noticeable decline in apex predators; large, long lived species absent locally	More fleshy algae than Level 1–2; highly sensitive species missing	Disease slightly above background level; some irregular tissue loss
Fair BCG Level 4	Low rugosity; water turbid; sediment accumulation	Reduced coral diversity; few or no living, large, old colonies; emergence of tolerant species	Absence of small reef fishes; large, long lived species absent locally; sensitive species conspicuously absent	Gorgonians replace sensitive coral and sponge species; abundant and diverse fleshy algae	Higher prevalence of disease and tissue loss
Poor BCG Level 5	Very low rugosity; very turbid water; thick sediment	Absence of colonies except highly tolerant species	No large fish; only tolerant species remain; high abundance of sediment dwelling invertebrates	Small and few colonies; highly tolerant species; high cover of fleshy algae	High prevalence of disease and high tissue loss

Simplified from Bradley et al. 2014

the potential list of beneficiaries and associated FEGS can get quite long, some prioritization may be needed to generate a manageable set for further consideration, based on greatest relevance to stakeholders or likelihood of impact by management (see the FEGS Scoping Tool; Sharpe et al. 2020).

Once the FEGS for a given coral reef have been identified, the next step is to develop metrics and indicators for each of the FEGS or the biophysical attributes that provide them. Table 4 below steps through this process for three common groups of human coral reef beneficiaries. The process begins by asking those benefiting from the coral reef what matters directly to them. That information can help identify the type of FEGS necessary to receive that benefit as well as the type of metrics that would be useful. The examples in Table 4 are generic, but more site-specific FEGS and metrics can, and should, be developed for site-specific ESGs. For example, fish diversity and abundance are suggested metrics for the beneficiary category of SCUBA divers. However, alternative or more specific metrics might be more

Table 4 FEGS metrics for a set of generic coral reef beneficiaries

Beneficiary	What matters directly?	FEGS type	Example metric
SCUBA Divers and Snorkelers	Is there sufficient visibility to be pleasurable to divers?	Water	Secchi disk depth (m)
	Is water quality safe for diving?	Water	Pathogen, contaminant, toxin concentrations
	Do these species attract the beneficiary?	Fauna	Fish diversity and abundance
	Do these species attract the beneficiary?	Fauna	Coral diversity and abundance
Recreational Anglers	Will I catch what I am expecting?	Fauna	Edible species abundance
	Will I catch something interesting?	Fauna	Charismatic species abundance
	Is it safe for boating?	Water	Wave intensity, surge height
	Is this reef aesthetically enjoyable?	Site appeal	Viewscape, sounds, smells
Coastal Property Owners	Will my property be damaged?	Water	Flood risk and coastal protection by the reef

appropriate, such as the presence or abundance of sea turtles, depending on site-specific factors such local ecology or cultural significance. Expert judgment can be used to determine the most appropriate substitute, depending on local factors as well as data availability. For example, it may not be feasible to collect daily Secchi disk depth readings, but local dive reports on water visibility may be easily collected.

After a complete set of beneficiaries and their associated metrics have been developed, these metrics can be compiled into models like those in Fig. 1 that demonstrate how changes in the level of environmental stress impact these prioritized FEGS. As shown in our coral reef example, there are multiple types of FEGS that may combine with one another to provide the overall benefit to a given beneficiary. For SCUBA diver beneficiaries, metrics related to coral diversity, fish abundance, and water visibility are all important for measuring the level of ecosystem service provided (Fig. 3a). Ecological production functions (EPFs) can be applied (e.g., Yee et al. 2014) to quantitatively link shifts in the level of environmental stress to shifts in the levels of FEGS provided to beneficiaries. One or more FEGS could be assessed for a single beneficiary (Fig. 3a), or those individual FEGS can be combined into an overall measure of realized benefit using ecological benefit functions (EBFs) (Fig. 3b, solid line). Either approach allows for a clear description as to how changes in environmental stressors directly impact different coral reef stakeholder groups (Fig. 3b, dashed lines).

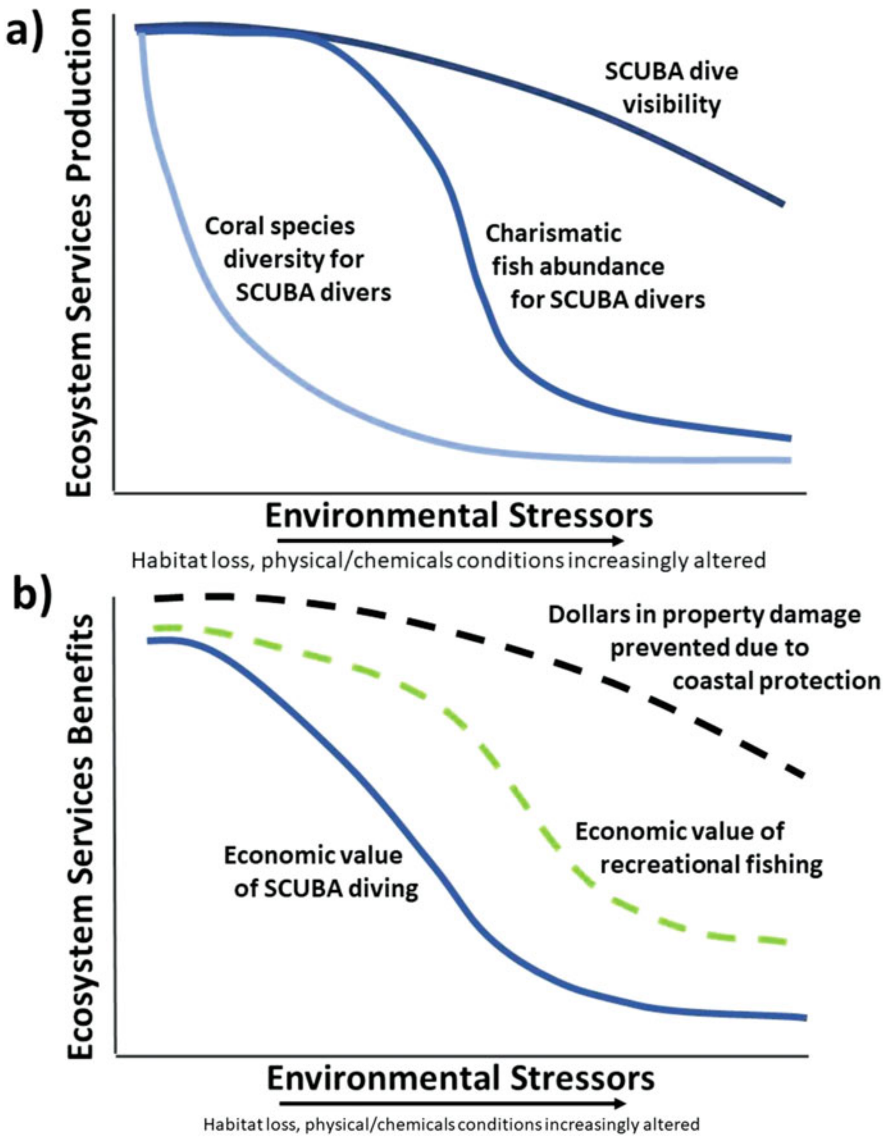

Fig. 3 A hypothetical ESG model showing changes in ecosystem service production and realized benefits in response to increases in environmental stressors. Top panel (**a**) shows changes in three individual FEGS for a single beneficiary (SCUBA divers) as developed through application of EPFs to changes in environmental attributes. Bottom panel (**b**) shows realized benefits to SCUBA divers (solid line) in addition to benefits to two other types of coral reef beneficiaries (dashed lines) as developed through application of EBFs to changes in EPFs

4.2 *Measuring Benefits of Habitat Restoration in Massachusetts Bay*

The Massachusetts Bays National Estuary Program (MassBays) is one of 28 National Estuary Programs (NEPs) across the United States, charged with developing and implementing a long-term plan to improve the waters, habitats, and living resources of their estuary(s). MassBays turned to the BCG framework as part of their plan to identify target conditions to manage 47 embayments along 1770 km of shoreline. MassBays is planning to use the BCG to communicate condition of key biological components that resonate with the public, including relevant invertebrate, fish, and habitat indicators with which local decision-makers can set public-supported targets. MassBays further realized that presenting community members with socio-economic information together with BCG biological information would greatly strengthen outreach and lead to better-informed decisions. In the course of updating their long-term plan (MassBays 2019), MassBays sought input on how people use the estuaries. Responses suggest the public cares about estuarine health, clean water and water quality, with favored activities that include shellfishing, swimming, and fishing. Developing a BCG classification scheme and a BCG/ESG approach (Fig. 1) will help to communicate the potential benefits of environmental improvement for both nature and people and to set appropriate, measurable, community-supported targets for restoration and protection.

As a proposed example of how MassBays could combine BCG and ESG, we focus on seagrass (*Zostera marina*), a habitat of restoration importance to MassBays. Seagrass beds throughout the MassBays system (and on the entire U.S. coastline) have lost significant area and function due to stressors including nutrient pollution, increasingly extreme weather events, and disease. Seagrasses support a diverse fauna and provide ecosystem services for many beneficiaries. To illustrate the approach, we selected recreational anglers, shellfishers, and birdwatchers as example beneficiaries. Fishing and shellfishing in seagrass beds are popular activities for those seeking striped bass, bluefish, scallops, crabs, and other species. Seagrasses are nursery habitats for many valued species and seagrass beds reliably sustain diving waterfowl for birdwatchers.

The BCG/ESG seen in Fig. 1 as two stacked graphs can also be presented as a Table to better show qualitative and quantitative thresholds of measures that define both the BCG levels of biological condition and the ESG measures of social and economic benefits. Table 5 provides an example of this for seagrass habitat, where the first column identifies each row with BCG level and the second column characterizes the seagrass biology defined by that level as narrative (which could most easily be quantified using seagrass acres as a proxy). The third column lists possible FEGS measures of valued fauna, and the fourth column shows measures of benefits to people. The last three columns align with the three boxes of the conceptual model of Fig. 2. To illustrate management application of Table 5, consider a hypothetical seagrass survey that shows only a few acres of sparse seagrass in a managed area. Sparse seagrass is a Level 5 (fourth row) BCG narrative measure (second column)

Table 5 A hypothetical example of a seagrass BCG/ESG set up as a table, with possible BCG measures (second column) linked to FEGS measures (third column) linked to benefit measures (fourth column)

BCG level	BCG narrative measures	Possible FEGS measures	Possible benefit measures
Level 1 to 2	Managed area has a large extent of abundant, dense, and healthy seagrass that supports diverse and abundant fauna	Fish/shellfish surveys and eBird (Sullivan et al. 2009) show valued fish species and scallops are very abundant and bird populations are very diverse	Observations and recreational fishing surveys show many people are fishing, scalloping, or birdwatching
Level 3	Some loss of acres from Level 1 to 2 (above); Abundant, dense, and healthy seagrass in some places; Thin and/or poor quality seagrass elsewhere; Diverse and abundant fauna in dense beds	Surveys and eBird show valued fish species and scallops are abundant and bird populations are very diverse	Observations and recreational fishing surveys show many people are fishing, scalloping, or birdwatching, comparable to BCG Level 1 to 2 above
Level 4	Moderate loss of acreage from Level 1 to 2 in managed area; Thin and/or poor quality seagrass in most places supports fewer and less diverse seagrass fauna	Surveys and eBird show valued fish species and scallops are moderately abundant and bird populations are diverse	Observations and recreational fishing surveys show a moderate number of people are fishing, scalloping, or birdwatching
Level 5	Major loss of acres from Level 1 to 2 in managed area; Sparse seagrass supports sparse fauna	Surveys and eBird show valued fish species and scallops are scarce and bird population diversity is only slightly elevated from adjacent non-vegetated areas	Observations and recreational fishing surveys show few people fishing, scalloping, or birdwatching
Level 6	No seagrass, shift to less diverse and productive non-vegetated faunal communities in managed area	Fish, scallop, and bird populations are comparable to those in non-vegetated areas	No more people are fishing, scalloping, or birdwatching than in local non-vegetated areas

with scarce valued fauna, and few people enjoying benefits (third and fourth columns). Presenting this information to the public together with descriptions of better environmental and socio-economic conditions at higher BCG levels could inspire a long-term vision of achieving, say, Level 3 conditions (second row) with abundant seagrass and fauna and many fishers and birdwatchers in some but not all places within the managed area. Once quantitative targets are set (e.g., for Level 3 acres) public-supported management actions (perhaps significant nutrient reductions) can be developed and implemented, then changes in BCG, FEGS, and benefit measures can be quantitatively monitored and reported back to the public.

The success of this potential approach for MassBays depends on the data and effort utilized to create a working BCG/ESG gradient that could apply at several scales: an overarching application to all 47 embayments; application to groups of embayments classified based on specific characteristic conditions; and to embayments at scales most relevant to management. Key elements of the effort are acquiring data and determining appropriate reference narratives and values for biological, FEGS, and benefit indicators. Comparability of the approach among embayments relies on a consistent identification of Level 1 or combined Level 1 / Level 2 condition.

To address data needs, MassBays worked with partners to synthesize a large amount of environmental, social, and economic data for these embayments. Reference condition for seagrass acres may be available through historic maps, charts, and surveys, or (as with all measures) by using best available current data, which may not represent Level 1 or 2, but can be interpreted in the BCG construct as Level 3 (or lower). It is likely that other existing MassBays data might serve as proxies for FEGS measures (here fishing, shellfishing, and birdwatching) based on methods in the literature, such as Rapid Benefits Indicators (e.g., Mazzotta et al. 2016).

Combining the BCG with an assessment of ecosystem services allows communication of environmental condition directly linked to the socio-economic benefits of environmental improvement (Cicchetti et al. 2017). This approach can resonate with people whose belief systems run the entire spectrum from those who most appreciate nature for its own sake to those who most appreciate the socio-economic benefits that nature provides to humans. Engaging a range of stakeholders invests more people in the value of environmental protection and is an important tenet of ecosystem-based management (Arkema et al. 2006). The BCG/ESG framework captures stakeholder input to develop goals using the approach of "what did we have, what do we have, what do we want, and how do we get there" to communicate both nature and benefits (Tables 1 and 2). This allows managers to set appropriate, measurable environmental targets that are supported by a diverse public.

5 Role of an ESG in Ecosystem-Based Management

The introduction of ecosystem goods and services advances the utility and applicability of the BCG framework for ecosystem-based management activities. Important action items within an EBM approach to decision making can include: the identification of objectives and performance measures to describe what really matters to stakeholders about a decision; the identification of management alternatives; the articulation of potential user conflicts or tradeoffs between management alternatives; and the articulation of potential direct ecosystem services benefits, or related co-occurring benefits, for a given decision context (Cormier et al. 2017). The ESG, through its structured approach to defining and enumerating potential changes in ecosystem services, allows decision makers to clearly articulate the elements feeding into each of these steps.

An important foundational principle of the BCG is science communication through BCG visualization and the accompanying tables that describe the technical aspects of each BCG level. Likewise, an ESG framework also lends itself to strong science communication served up in a strategic manner (*sensu* Harwell et al. 2020), allowing for communication of key messages to targeted audiences. Paired with a BCG, the ESG can allow decision makers to describe meaningful and unambiguous environmental objectives and their measures and clearly communicate the potential gains or losses in ecosystem services. The quantitative measures defined by the levels of an ESG might also indicate where biomonitoring can be used to assess whether actions are having the desired outcomes and what adjustments can be made to future actions as part of targeted adaptive management efforts (e.g., LoSchiavo et al. 2013).

Finally, the ESG approach can be helpful for identification of critical gaps in knowledge, helping EBM practitioners identify where resources may be needed to fill those gaps—in particular, what future scientific research is needed to move a narrative description for a given ESG level to a quantitative description. Future development of ESG principles include in-depth application to a suite of case study examples across a range of ecosystem types, both related to information and regulatory needs, such as condition assessments conducted for water quality management purposes under the U.S. Clean Water Act, and to broader EBM questions such as ecological protection, restoration or fisheries management.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Rapid Benefit Indicator Tools

Justin Bousquin and Marisa Mazzotta

Abstract Given the many interconnections between socio-economic and ecological systems, for Ecosystem-Based Management (EBM) to be effective, decision makers must consider metrics for both. Supply side tools and assessments characterize ecosystem condition, functioning, and potential to provide ecosystem goods and services (EGS). Demand side tools, including economic valuation, assess people's preferences for EGS and sometimes estimate the monetary amount people are willing to pay for a good or service. However, economic valuation is often omitted from assessments, due to lack of data or expertise; and economic valuation alone may not sufficiently capture all important aspects of some decisions. Benefit-relevant indicators have evolved as a way to measure the connection between goods or services that may be provided by an ecosystem, and people who may benefit from those services, while stopping short of valuation (Olander et al., Ecol Indic 85:1262–1272, 2018). Like economic valuation, benefit-relevant indicators can help assess trade-offs and compare alternative outcomes (National Ecosystem Services Partnership, Federal resource management and ecosystem services guidebook. National Ecosystem Services Partnership, Duke University, Durham, 2016). The Rapid Benefit Indicators (RBI) approach is an easy-to-use process for choosing a structured set of non-monetary benefit-relevant indicators for assessment (Mazzotta et al., Integr Environ Assess Manag 15:148–159, 2019). The RBI approach indicators are intended to be applied in conjunction with existing ecosystem service assessment approaches and tools, to connect changes in the availability of EGS to the locations where, and how, people benefit from those goods and services. Though developed for use with urban freshwater wetland restoration, the general RBI approach and indicator framework may be adapted and applied to other

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environmental changes or ecological systems. This chapter will detail the RBI approach and highlight how RBI tools can inform resource management decisions.

Lessons Learned

- When implementing EBM, it is not enough to simply maintain or restore the functioning of ecosystems; it is essential to also consider benefits from services that people want and need
- Socio-economic metrics must be linked to changes in ecosystems to be relevant to EBM policy and management questions
- Indicator-based methods inform decisions when direct measures of economic values are unavailable, overly complex, elicit resistance or are otherwise inadequate
- A structured process for selecting benefit indicators, such as the Rapid Benefit Indicators Approach, helps practitioners choose the right metrics
- Tools, such as the RBI checklist tool, the RBI spatial analysis tool and the RBI national catchment dataset, can make it easier for practitioners to evaluate benefits from services as part of their EBM approach

Needs to Advance EBM

- EBM methods and policy would benefit from more explicitly addressing and communicating benefits to people resulting from managing ecosystems
- EBM practitioners need tools that will allow them to evaluate the services and benefits provided by a wide array of ecosystems

1 Evaluating Benefits

Ecologists and economists have embraced the ecosystem services concept as an important support for understanding and managing social-ecological systems. Ecosystem-Based Management (EBM) seeks to protect, maintain or restore ecosystems and their functioning so that they can provide the services that people want and need (Piet et al. 2017; Delacámara et al. 2020). Decisionmakers using an EBM approach need to weigh trade-offs among the benefits and costs of different actions, and ecosystem service metrics can inform those trade-offs. Despite consensus around the importance of considering ecosystem services, it remains difficult for practitioners to choose the right metrics (Boyd et al. 2016; Olander et al. 2017).

There are metrics specific to each component of the ecosystem service framework (Fig. 1). Biophysical metrics are commonly used to model or monitor how an ecological system responds to human actions (the ecological outcomes). Ecosystem service metrics describe the ecosystem goods and services (EGS) produced by the system, based on its condition and resulting ability to perform functions needed to produce those EGS. To avoid double-counting of benefits to people, ecosystem service metrics should measure what is directly enjoyed, consumed or used by people, i.e. final ecosystem goods and services (FEGS; Boyd and Banzhaf 2007;

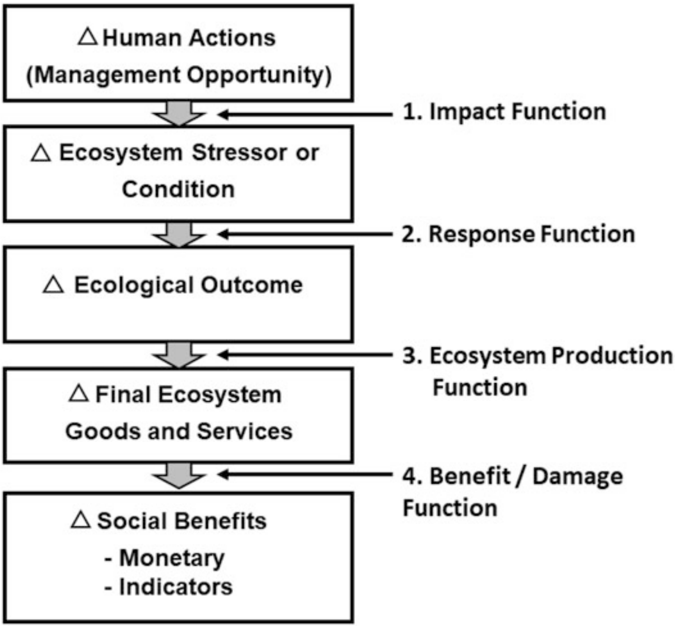


Fig. 1 Conceptual framework linking management actions to ecological change and to socio-economic benefits. (Adapted from Wainger and Mazzotta 2011)

Ringold et al. 2013). Changes in FEGS that result from human actions will lead to changes in human well-being, which may be measured in monetary terms or using indicators, as presented in this chapter.

There are many tools and metrics available to measure the functioning of ecological systems or their ability to produce ecosystem services, although these do not always measure the endpoints that are most appropriate for evaluating changes in human well-being. For example, there is often a bias towards, or over-reliance on, land use and land cover data (Tashier and Ringold 2019). Simply measuring an ecosystem's ability to produce goods and services does not provide evidence that those goods and services are used or enjoyed by people. Even when using the most precise metrics for ecological outcomes or ecosystem services, if these metrics cannot be linked to the socio-economic benefits that result from changes in the ecosystem, the assessment will be incomplete and risks being irrelevant to people. The closer metrics get to measuring the change in social benefits, the better those metrics can inform trade-offs.

Environmental decision makers, the people deciding between environmental management actions, often require or desire monetary measures of benefits to people. Economic valuation approaches monetize the value of ecosystem goods and services to people (Heal 2000b; Freeman et al. 2014). Valuation methods include the use of market prices, where available (e.g., for commercially-harvested

EGS such as fish or timber), but most EGS are not traded in markets and thus require the use of non-market valuation methods (National Research Council 2005; Champ et al. 2017). These include revealed preference methods that use people's behavior to infer values (e.g., the travel cost method, Parsons 2017); stated preference methods that use hypothetical questions or comparisons of choices to ask people to directly state their values (e.g. contingent valuation, Bateman and Willis 1999); and benefit transfer methods that apply values from existing studies to a new location and/or context (Johnston et al. 2015). Each of these methods is appropriate for different contexts and types of values. Whereas use values, where people directly interact with FEGS, may be measured by any of these methods; non-use values, where people do not directly interact with the service, can only be measured using stated preference methods or benefit transfer of stated preference studies. Each method has its advantages as well as shortcomings or pitfalls (Champ et al. 2017; Johnston et al. 2015; Freeman et al. 2014). These methods also tend to be resource-intensive, and location- and context-specific (Spash and Vatn 2006; Heal 2000a). Even when valuation studies are performed well, if the estimated values are not appropriately linked to available biophysical models or metrics, or if the appropriate biophysical metrics are not available, the estimated values will not be responsive to changes in ecosystems (Johnston et al. 2012; Schultz et al. 2012). Especially in the context of EBM, socio-economic metrics that can't be linked to changes in ecosystems are of limited relevance to policy or management questions.

In cases where assessments include metrics linking each component of the ecosystem services framework, decision makers may still face various difficulties in applying a comprehensive EGS assessment. Resource managers who must make decisions affecting EGS, such as state agencies, may not have in-house expertise to conduct an assessment, and may lack resources or support for commissioning an appropriate study. Olander et al. (2017) suggest that many methods are not responsive to policy and management changes, not generalizable enough to transfer from one context to another, or are too burdensome in terms of needed expertise, data or costs. Developing new metrics and assessment tools to fill gaps should involve decision makers to help ensure that their needs are met and that applications are feasible, with relevant outcomes (Ojo et al. 2018). However, tool developers must balance these factors against over-tailoring metrics and tools to be so location-specific that they are not transferable or overly burdensome to apply. For applied tools to be more widely employed, there often needs to be a shift in emphasis from accuracy to more general applicability and feasibility in terms of time, money, expertise and data availability. Practitioners have limited time and resources to learn new tools and to translate results to be relevant to their stakeholders; often, a less-precise but more easily-applied approach is sufficient for the types of decisions being made, especially when it will be used as part of a broader process of stakeholder engagement and adaptive management (Kline and Mazzotta 2012).

2 Non-monetary Benefit Indicators

Recognizing the limitations of monetary economic valuation, including uncertainties around value estimates but especially with regard to the expertise required and cost of primary studies, indicator-based studies offer an alternative (Boyd and Wainger 2002). Often study results are used primarily to stimulate public discussion or better direct further investigations (Thaler et al. 2014). In these cases, indicators may avoid some of the pitfalls and resistance that value estimates can elicit. Indicators can inform decisions when direct measures of economic value are unavailable, overly complex to estimate, or otherwise inadequate (Meadows 1998; Bossel 1999; Layke 2009). Desirable indicator variables have a strong relationship to the phenomena of interest yet are simple enough to be effectively monitored and/or modeled (Dale and Beyeler 2001).

Benefit indicators, in contrast to strictly biophysical indicators, provide information regarding the benefits and values of ecological changes to people. It is possible to use sound economic principles to formulate these indicators and capture the important aspects of value, while avoiding the burden of calculating dollar values (Mazzotta et al. 2019). Benefit-relevant indicators link biophysical outcomes to benefits for an identifiable group of people, in order to evaluate trade-offs and make decisions. An indicator is considered *relevant* when it captures something that directly alters beneficiaries' well-being, in units that are relevant to those beneficiaries (Olander et al. 2018).

The National Ecosystem Services Partnership (NESP) Guidebook presents an assessment framework that includes benefit-relevant indicators (NESP 2016). The guidebook suggests that conditions that influence values or preferences fall into five general categories (see also Wainger et al. 2001; Wainger et al. 2010):

1. Service Quality—quality of the service for its intended use
2. Capital and Labor—availability of capital and labor that complement ecological outputs in order to create goods and services
3. Number and characteristics of users
4. Reliability—reliability of the future stream of services
5. Scarcity and Substitutability—number of substitutes for the services provided

By incorporating these five conditions, benefit-relevant indicators complement ecological metrics with indicators more closely tied to what people value and prefer. Methods to measure benefit-relevant indicators can range from simple to complex and such metrics can be qualitative or quantitative. Though benefit-relevant indicators go a long way toward improving metrics, there is a great deal of flexibility and interpretation involved in defining the indicators for application. This level of flexibility makes benefit-relevant indicators broadly applicable; however, it also can result in inconsistency across applications, making them harder to compare or transfer across studies or locations with disparate contexts. While developing and applying benefit-relevant indicators is easier for decision makers than conducting economic valuation, it still requires economic or social science expertise.

3 Rapid Benefit Indicators (RBI) Approach

The Rapid Benefit Indicators (RBI) approach is an easy-to-use screening method for consistent and transparent site level assessment (Mazzotta et al. 2016, Mazzotta et al. 2019). The RBI approach is an extension of previous work on benefit indicators and parallels many benefit-relevant indicator concepts. The RBI approach uses a list of generic questions that capture important aspects of benefits and value to people, similar to the benefit-relevant indicators list of conditions. Practitioners answer each question to develop a set of non-monetary benefit indicators. Using a uniform indicator framework ensures greater consistency, while providing the latitude to select indicators best fitting the decision context. The RBI approach is intended to be used in conjunction with ecosystem service assessment approaches and tools to connect changes in ecosystems to changes in EGS and ultimately to benefits to people.

3.1 *Five Step Rapid Benefit Indicator Process*

The RBI indicators fit within a broader five-step process (Fig. 2) adapted from structured decision-making (Gregory et al. 2012) and in alignment with EBM (DeWitt et al. 2020). Each step in the process builds on the previous step. The **first step**, describing the decision context, includes identifying stakeholders and their objectives. Based on these objectives, the **second step** is to select the relevant ecosystem services and resulting benefits to assess, and to specify how those services and benefits are defined. Defining the ecosystem services and benefits is critical in order to be able to select appropriate indicators in step three. Identifying stakeholders, their values and how those values align with ecosystem services and benefits is essential and can be facilitated by FEGS approaches and tools (Sharpe et al. 2020).

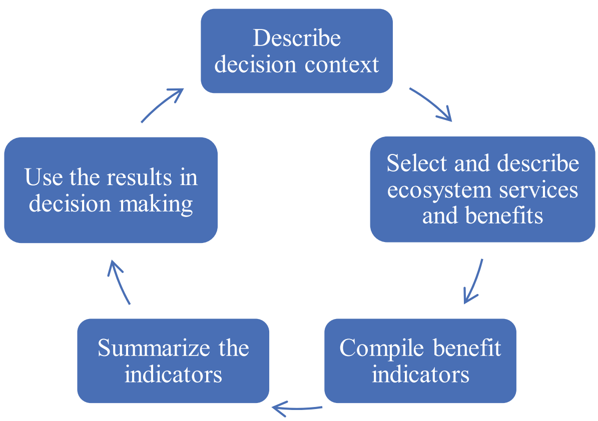


Fig. 2 The five-step process used to apply the RBI in decision-making (Mazzotta et al. 2019; adapted from Gregory et al. 2012)

The Rapid Benefit Indicators (RBI) approach Guidebook (Mazzotta et al. 2016) includes a case study where ecosystem services and benefits were selected from a list of those mentioned by resource managers during semi-structured interviews (Druschke and Hychka 2015). This list was further reduced to a set of ecosystem services that result in local benefits that are easily differentiated at a site scale. The RBI approach was designed to be applied at the site scale, which means that EGS that do not vary across sites do not need to be included in the assessment.

In **step three**, indicators are selected and compiled based on five questions, some with sub-questions. This is described in more detail below. In **step four**, compiled indicators are summarized to assist interpretation of results. **Step five** takes results summarized in step four and uses them in decision making.

Fostering adaptive management is a core part of EBM (Delacámara et al. 2020). To support adaptive management, the RBI decision process may be applied iteratively. After indicator results are used in decision making the practitioner may apply the same indicators over time to monitor both the results of actions and how changing conditions in the study area may suggest new priorities for action.

3.1.1 Rapid Benefit Indicator Questions (Step 3)

The five questions and their sub-questions are the core of the RBI approach. Some of these questions and sub-questions are optional, while others are required, as discussed further below.

Question 1: Can people benefit from an ecosystem service?

As an initial screening question, this helps determine whether people are able to benefit from an ecosystem service. It requires a site to currently meet, or to meet after restoration, three criteria:

1. It produces a final ecosystem good or service,
2. There are people who will benefit from the EGS, and
3. Complementary inputs required for benefits to reach people, if any, are available.

Sites that do not meet these criteria do not result in a benefit and require no further assessment for that EGS. A site fails the first criterion if it is not large enough or does not have high enough functioning to produce services of the required quantity or quality. For example, a site may be too fragmented to provide habitat for a bird species of interest to bird watchers. A site fails the second criterion if general demand for the service is lacking (see Russell et al. 2020 for similar requirements of Natural Capital Accounting). This is a more superficial precursor to question 2, assessing how many people benefit, and stops short of going into spatial relationships or quantification. For example, nearby flooding after recent storms is strong enough evidence of demand for flood-reduction services, without assessing where flooding occurred in relation to sites. For some ecosystem services, complementary inputs or conditions must be available for enjoyment of the service. A site fails the third criterion if such inputs or conditions are necessary but not available. Examples include infrastructure allowing physical access, or lack of institutional constraints such as regulatory harvest limits (NESP 2016; Olander et al. 2018).

Question 2: How many people benefit?

This question focuses on quantifying the pool of people who can benefit from an EGS at each site. In quantifying economic values, often the aggregate value of EGS is more sensitive to the number who benefit than the magnitude of individual values (Bateman et al. 2006). Thus, the number of beneficiaries provides an indication of the total benefits from an ecological change. The RBI uses a spatial approach to count beneficiaries, defining how services flow to people, defining the area where people may benefit, and quantifying the number of people who could benefit within the defined area. Services are generally produced in-situ and then either services flow to areas where people can access them, people travel to the site to access services, or both. RBI uses three general categories (Fig. 3) to characterize these spatial relationships (Fisher et al. 2009; Bagstad et al. 2013):

(a) Services are generated and must be enjoyed on site or within a geographic area (Fig. 3a), for example, through recreational uses of a site. To benefit, people must be located at or travel to that site or geographic area. When evaluating these services, the pool of beneficiaries depends on how far people will travel to the site.
(b) Services are generated on site and flow in all directions to a surrounding area (Fig. 3b), for example, birds or pollinators that use the site for habitat and move through nearby areas where people can benefit. People within the area where services flow will be able to benefit. When evaluating these services, it is important to consider how far services travel, whether those travel paths are blocked in any direction, and how those travel paths overlap with people who might benefit.
(c) Services are generated on site and flow in a single, or restricted, direction to a surrounding area (Fig. 3c), for example, downstream flood risk reduction from a wetland. People within the area where services flow will be able to benefit. This is true for services that flow downstream, such as water retention or purification. When evaluating these services, it is important to consider how far services

Fig. 3 The three categories RBI uses to characterize spatial relationships between where services are produced and where people access them (Mazzotta et al. 2019)

travel, whether anything impedes or assists that flow, in what direction, and how the travel path overlaps with people who might benefit.

Nonuse values for EGS that are important to people although they do not directly interact with them are a special case in which people do not have to be in any particular spatial relationship to the service, but simply need to be aware of the service and value it. These services can benefit people at varying distances without direct contact with people. For example, people may value the fact that a site provides habitat for a rare or unique species, although they do not see or interact with that species. The RBI approach is not tailored to the complexities of evaluating nonuse services, which are beyond the scope of this method. However, if a site is particularly rare or unique, or contains rare species, nonuse values may be significant and should be noted and evaluated using other approaches (Wainger et al. 2018; Richardson and Loomis 2009).

The spatial extent of relevant areas for assessment will vary based on local conditions and attributes of the ecosystem, landscape, and beneficiaries (Vajjhala et al. 2008). Distance decay may impact the service, causing service quantity or quality to decrease with distance from the source (Fisher et al. 2009; Bagstad et al. 2013). Services may also decrease in quantity or quality when they encounter "sinks," features on the landscape that absorb, degrade or deplete the service or the conduit transporting the service (Bagstad et al. 2013). Like services, benefits may experience distance decay, where people's values diminish with increasing distance from the area where services are accessed (Hanley et al. 2003; Bateman et al. 2006; Campbell et al. 2007; Campbell et al. 2008). One way to account for decreasing benefits with distance is to divide the number of beneficiaries into beneficiary pools based on distance bands and assigning lower weight to farther bands to help account for the decay in value. For example, when evaluating a recreational service, the number of beneficiaries within walking distance and driving distance may be estimated separately, with lower weight applied to those farther away.

Question 3: How much are people likely to benefit?

This question assesses the magnitude of benefits. How much people benefit from an ecosystem service is assessed by indicators answering four questions. Some of these questions may not be relevant to every ecosystem service, and those that are relevant may have one or more indicators. These questions are based on core concepts of economic theory of supply, demand, and value (Freeman et al. 2014; Nicholson and Snyder 2012). Economic theory posits that each of these factors, all else equal, will increase or decrease a person's value for a good or service through their effects on the demand function (the function that relates price or willingness to pay to quantity and quality of a good or service).

<u>a. What is the quality of the service?</u>

People benefit more from higher quality ecosystem services. Existing tools that measure the functioning of ecological systems or their ability to produce ecosystem services (e.g. Lewis et al. 2020; Culhane et al. 2020) may help inform assessment of the quality of ecosystem services provided by a site.

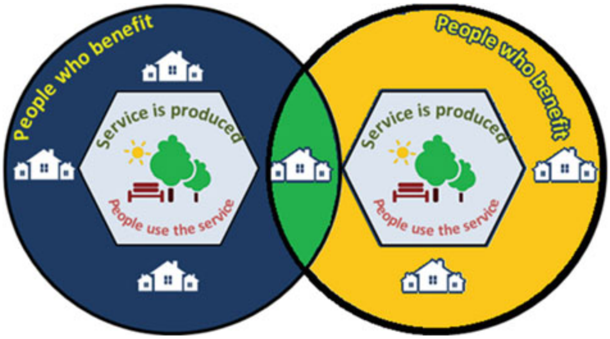


Fig. 4 In this illustration, the service areas shown in the blue and yellow circles represent recreational parks, and the location of individual beneficiaries is indicated by the house icons in each circle. The house within the green overlap area has access to two parks, where other homes depicted each can easily access only one park. Therefore, services provided by the parks are less scarce for the house in the green area and an additional park will have a lower incremental value for people in that location, than for those living in the other homes depicted, who have fewer substitutes (Yee et al. 2017)

b. Are there substitutes for the service or is the service scarce?

In general, people benefit less from each additional unit of a good or service they could receive. In other words, for a given beneficiary, the more units of a service that are available, the lower the value of more of that service or additional sources of that service (Yee et al. 2017; Fig. 4). Thus, fewer substitutes or greater scarcity lead to higher value, all else equal. The scarcity of ecosystem services already available to beneficiaries is informed by estimating the number of similar ecosystems and/or technological substitutes already providing those services. An increase in substitutes indicates lower value, so the direction of influence is reversed for this indicator as compared to the others.

c. What is the quality of complements to the EGS?

For some EGS, complements may be required for people to benefit, or may enhance benefits. For example, without a boat launch, people may not be able to access a waterway to benefit from recreational boating; and a higher quality boat launch will enhance benefits of boating. Thus, people benefit more when the quality of complementary inputs, other goods and services used with the ecosystem service, is higher. This is only important for services that are enhanced by complementary factors.

d. How strong are people's preferences?

People's strength of preferences influences their economic value (willingness to pay) for improvements in a good or service. In general, those with stronger preferences will benefit more from a given improvement in a good or service. For example, a more avid birder may have a higher value for improved bird habitat and resultant birding opportunities than someone who is less interested in birding. Characteristics of the beneficiaries that influence preferences are specific to the local context and the

EGS in question and may involve complex interactions among factors, and thus may be difficult to account for. In practice, this factor may often be omitted from assessment due to lack of data, or may be incorporated through more qualitative approaches such as opportunities for public comment.

Question 4: What are the social equity implications?

Determining social equity implications examines the population receiving the benefits and evaluates whether they are socio-economically disadvantaged. Benefits can be more important for vulnerable populations or people facing environmental concerns. These groups tend to have fewer resources to access ecosystem services yet may rely on them more (Norman et al. 2012). Similar to quantification of how many people benefit (Question 2), evaluating social equity implications involves characterizing the spatial relationship between where services are produced and where people access them. However, instead of quantifying the number of people who benefit, what is important for social equity is characterizing attributes of the people who benefit that make that population socio-economically disadvantaged.

Question 5: How reliable are benefits expected to be over time?

Determining the reliability of benefits over time explores the probability that some change will occur over time to inhibit the production of services or flow of benefits to people. When benefits are provided reliably over a longer period the total value of those benefits is greater. For example, benefits of a restored coastal marsh will diminish over time if the marsh becomes submerged due to sea-level rise. Features that impact the reliable delivery of services over time need to be site-specific but not necessarily benefit-specific.

3.1.2 Using Answers to Rapid Benefit Indicator Questions in Decision Making (Steps 4 & 5)

After developing a set of indicators and quantifying those indicators by answering the five generic questions, the next step, step four, is to summarize the indicator metrics. Summarizing metrics into a table can help when making comparisons across sites. These summary tables are analogous to the consequence tables used in structured decision making (see Gregory et al. 2012 for examples), where metrics evaluate the performance of each site. All tools developed to facilitate application of the RBI approach (described below) provide a summary table for this step. This summary table does not rank sites quantitatively or aggregate metrics.

The last step in the process, step five, is to evaluate the metrics in the summary table in light of the decision(s) to be made. There are many ways to weigh the different metrics and the resulting trade-offs when choosing among different actions for different sites. The indicator metrics may be used as is in disaggregated form as a basis for discussions, to inform participatory or consensus-type decisions; or they may be aggregated using Multi-Criteria Decision Analysis (MCDA) methods (Belton and Stewart 2002; Gregory et al. 2012). MCDA methods have successfully been used to aggregate RBI metrics (Martin and Mazzotta 2018a, b; Martin et al. 2018).

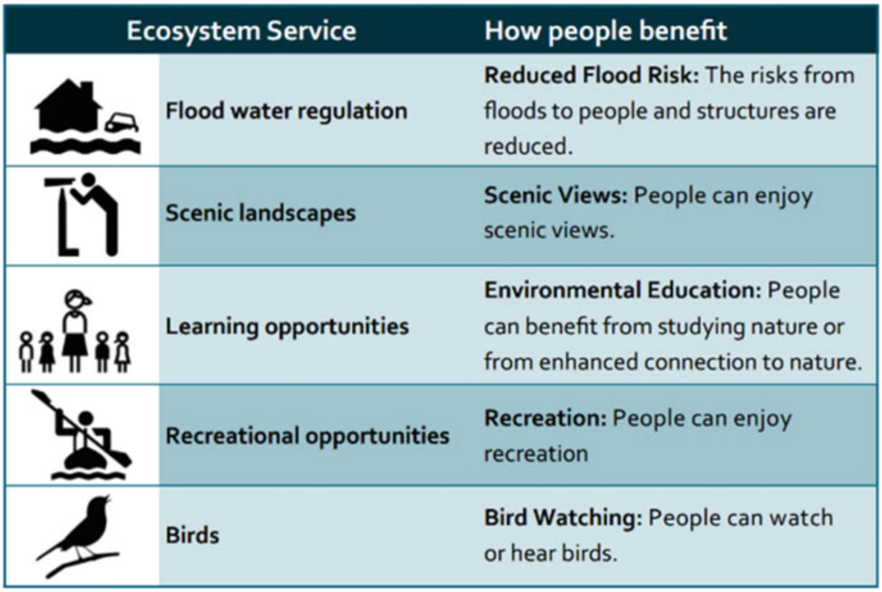
4 Tools for Applying the Rapid Benefit Indicator Approach

The RBI approach was originally developed for application to urban freshwater wetland restoration. The general approach and indicator framework will work with other types of environmental changes or within different ecological systems. Indicators for five benefits of urban freshwater wetlands (Fig. 5) have been previously developed and integrated into tools that help users more easily apply the Rapid Benefit Indicators approach.

The rest of this chapter focuses on three tools developed to assist in applying the RBI approach:

1. Checklist Tool
2. Spatial Analysis Toolset
3. National Catchment Dataset

These tools are available for download for those practitioners who may want to use them in their application, or use them as models that may be modified for other services and contexts. Each tool has unique aspects that make it more or less useful to specific audiences and applications (Table 1). For example, two of the tools produce a color-coded summary report for indicator results (Fig. 6).



	Ecosystem Service	How people benefit
	Flood water regulation	**Reduced Flood Risk:** The risks from floods to people and structures are reduced.
	Scenic landscapes	**Scenic Views:** People can enjoy scenic views.
	Learning opportunities	**Environmental Education:** People can benefit from studying nature or from enhanced connection to nature.
	Recreational opportunities	**Recreation:** People can enjoy recreation
	Birds	**Bird Watching:** People can watch or hear birds.

Fig. 5 The five ecosystem services and benefits that have previously developed indicators for freshwater wetlands restoration sites (Mazzotta et al. 2019)

Table 1 Differences between tools

Tool name	Software required	Custom benefits	Summary report	Guidance	Main advantage
Checklist tool	Excel/PDF	Y	Y	High	Document process
Spatial analysis toolset	ArcGIS	Limited	Y	Medium	Automated
National Catchment Dataset	GIS	N	N	Low	No data required

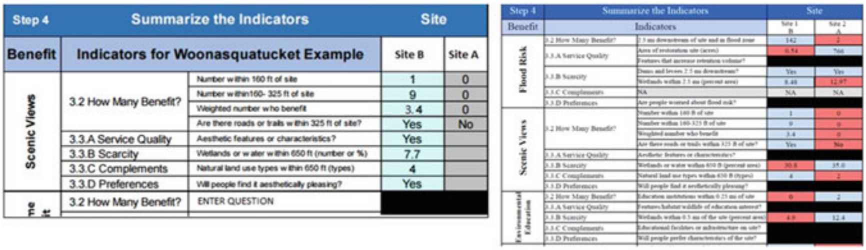


Fig. 6 Partial example color-coded summary reports produced by the Checklist Tool (Left) and Spatial Analysis Toolset (Right)

4.1 RBI Checklist Tool

The *Rapid Benefit Indicators (RBI) checklist tool*[1] is for recording results of manually-conducted analysis. Users can develop indicators using a variety of resources including paper maps, online maps, stakeholder engagement, or site visits. The checklist tool helps users go through the assessment process and provides space to document data sources, assumptions and other supplemental information.

The checklist tool is available in a pdf format as a low-tech solution that users can fill out by hand and print. The first two pages document the decision context and scope the ecosystem services and benefits being assessed. The next five pages of the pdf are each specific to one of the five ecosystem services and benefits that have previously developed indicators (Fig. 5). The last page is for summarizing results. In some cases, data entered in earlier pages of the pdf are automatically copied to their section in the summary table.

The macro-enabled Excel checklist tool has added functionality that walks the tool user through the step by step decision process. The Excel checklist tool uses responses entered in each form to dictate the next entry. For example, if a site fails to

[1] https://cfpub.epa.gov/si/si_public_record_Report.cfm?Lab=NHEERL&dirEntryId=331110.

meet criteria in Question 1, the tool skips forms for Questions 2–5 for that site. The Excel checklist has built-in forms for the five ecosystem services and benefits with previously developed indicators (Fig. 5). The Excel checklist can also document and assess newly developed indicators for different benefits, new ecological systems or different ecological changes using the five-question framework from the RBI approach. When site assessment is complete, the Excel checklist tool automatically summarizes indicators in a color-coded summary table (Fig. 6), where red indicates worse, blue indicates better, and gray indicates NA/neutral, relative to the average for all sites (for quantitative indicators) or based on characteristics increasing or decreasing benefits (for yes/no indicators). This summary does not rank sites.

4.2 RBI Spatial Analysis Tools

The *Rapid Benefit Indicators (RBI) spatial analysis tools*[2] compile indicators based on spatial data (Bousquin et al. 2017). Where datasets are available, this significantly expedites analysis. However, spatially-derived indicators are not adequate for answering all of the questions in the RBI approach. This tool is specific to the five ecosystem services and benefits with previously developed indicators (Fig. 5). This tool requires some familiarity with Geographic Information Systems (GIS). Results are summarized by site in table format, either spatially in the site dataset attribute table or as a pdf report (Fig. 6). Although the results are summarized spatially, the tool does not symbolize the results on a map or produce map products directly.

The *Rapid Benefit Indicators (RBI) spatial analysis tools* are an ArcGIS Python toolbox. This means users must have ESRI's desktop software, ArcMap® or ArcCatalog®, to open and use the tools (ESRI 2011). The toolbox does not require installation; users simply point ArcMap or ArcCatalog to it and then interact with it just like other toolboxes. Being written as a Python toolbox makes the tools more transparent and adaptable. Users who are familiar with the Python language can open the code and see all input handling and processes. This makes it easier to update inputs or processing to fit newly developed indicators.

The spatial analysis toolset includes seven individual tools (Table 2). The main tool, the *Full Indicator Assessment Tool*, runs a complete analysis of any or all of the five existing ecosystem services and their benefits. This is the fastest way to assess multiple ecosystem services and benefits. The additional tools in the toolset perform partial analysis. *Part* tools perform a variety of functions, including downloading data (e.g. *Flood Data Download Tool*), performing partial analysis using user updated parameters (e.g. *Social Equity of Benefits Tool* where the default buffer distance can be altered), or performing a specific part of the analysis that could be transferable to newly developed indicators (e.g. the *Presence/Absence to Yes/No Tool* can test for the presence of other spatial features near a site that could serve as indicators).

[2]https://cfpub.epa.gov/si/si_public_record_Report.cfm?Lab=NHEERL&dirEntryId=338471.

Table 2 Individual tools in the spatial analysis toolset

Tool name	Purpose
Full indicator assessment tool	Assess any/all of five benefit indicators
Benefit reliability tool	Assess benefit reliability (Question 5)
Flood data download tool	Download hydrologic data for flood risk indicators
Flood risk reduction tool	Assess reduced flood risk benefit indicators
Presence/absence to yes/no tool	Assess presence/absence for custom benefit indicators
Report generation tool	Summarize indicator results
Social equity of benefits tool	Assess benefit social equity implications (Question 4)

Table 3 National spatial datasets to use as default inputs

Data layer name	Source	Description
Restoration site polygons	EnviroAtlas[a]	Potentially restorable wetlands on agricultural land
Population raster	EnviroAtlas[1]	Dasymetric population
Address points	OpenStreetMap	OSM buildings[b]
Flood zone polygons	FEMA,[c] EnviroAtlas[1]	Flood zones Estimated Flood Plains
Dams/levees	USGS	National Hydrography Dataset
Wetland polygons	EnviroAtlas[1]	National Wetlands Inventory
Catchments	NHDPlusV2[d]	Catchments
Conservation lands	EnviroAtlas[1]	Protected Areas Database of the US (PADUS)
Landuse/Greenspace polygons	NLCD[e] NOAA[f]	National Land Cover Database (NLCD) Coastal change analysis program (C-CAP)
Trails	OpenStreetMap[g]	HikeBikeMap
Roads	TIGER/line,[h] OpenStreetMap[7]	Roads Roads
Educational institution points	DHS[i]	Homeland infrastructure foundation—Public schools, private schools
Bus stops	OpenStreetMap[7]	Buses
Social vulnerability	SVI,[j] EJScreen[k]	Social vulnerability index Demographic indicators

[a]https://enviroatlas.epa.gov/enviroatlas/interactivemap/
[b]https://osmbuildings.org/
[c]https://msc.fema.gov/portal/search
[d]http://www.horizon-systems.com/NHDPlus/NHDPlusV2_data.php
[e]https://www.mrlc.gov/data
[f]https://coast.noaa.gov/digitalcoast/data/ccapregional.html
[g]https://www.openstreetmap.org
[h]https://www.census.gov/geo/maps-data/data/tiger-line.html
[i]https://hifld-geoplatform.opendata.arcgis.com/
[j]https://coast.noaa.gov/digitalcoast/data/sovi.html
[k]https://ejscreen.epa.gov/mapper/

Although the way spatial analysis tools analyze data is predefined, there is flexibility in what the user can load as input datasets. Most of the inputs have nationally consistent datasets that can serve as a default (Table 3). This helps ensure the tools will not require data collection in most places. In many cases there are better alternatives to the national data. For example, national datasets may be incomplete (e.g. FEMA Flood zones; Bousquin and Hychka 2019), may lead to less precise results (e.g. EnviroAtlas raster population data when compared to address points; Bousquin et al. 2015), or may have higher resolution local datasets (e.g. statewide land use datasets like the one used in the Woonasquatucket watershed, RI case study; Martin et al. 2018).

4.3 RBI National Catchment Dataset

The *Rapid Benefit Indicators (RBI) catchment dataset*[3] allows a user to compare sites in different catchments based on a catchment characterization previously performed using a sub-set of indicator metrics (Bousquin and Hychka 2019). Rather than being a tool to help users analyze their data, this is a national dataset of results for two indicators of reduced flood risk benefits. The first indicator answers question two, *how many people benefit*. The second indicator answers the *is the service scarce* part of question three, *how much are people likely to benefit*. Using this dataset provides the least flexibility, as indicators and data inputs are predetermined.

Using this dataset requires some GIS experience, as comparing the data across restoration sites requires several steps. First, users can download these results as a comma separated values (csv) file for their region of interest. Next users will need to download the NHDPlusV2 catchment shapefile for their region. This catchment shapefile is the same one the *spatial analysis tools* use (Table 3). The csv file contains a "COMID" field to join the data to the NHDPlusV2 catchment shapefile's "FEATUREID" field. After joining the csv to the shapefile, GIS software (e.g. ArcGIS, QGIS, grass, R, etc.) can be used to visualize data by catchment. Catchment results can then be overlaid and compared to locations for restoration sites (Fig. 7). Comparisons can be visual, using site coordinates or imagery, or spatial, overlaying and summarizing indicators for a shapefile of sites.

Each catchment has 15 columns or fields. Most fields represent similar information but aggregated in different ways or determined based on different data. Each row presents values for a specific catchment. The dataset does not have the spatial resolution to differentiate sites within the same catchment.

For the first indicator, *how many people benefit*, beneficiaries are people in flood-prone areas downstream of a wetland restoration (Fig. 7). If restored, these people are in proximity to receive reduced flood-risk as a result. We recommend using values in either the "EA_pct_d" or "fld_pct_d" fields. These values represent the sum of the population in flood-prone areas in that catchment and catchments five km downstream. The difference between the two fields is in the data used to identify flood-prone areas. In the "EA_pct_d" field, an EnviroAtlas modeled layer defines flood-prone areas (Woznicki et al. 2019). In the "fld_pct_d" field, FEMA inland flood zones define flood-prone areas (FEMA 2018). Both estimates overlay the

[3] https://catalog.data.gov/dataset/nhdplusv2-catchment-rbi-data.

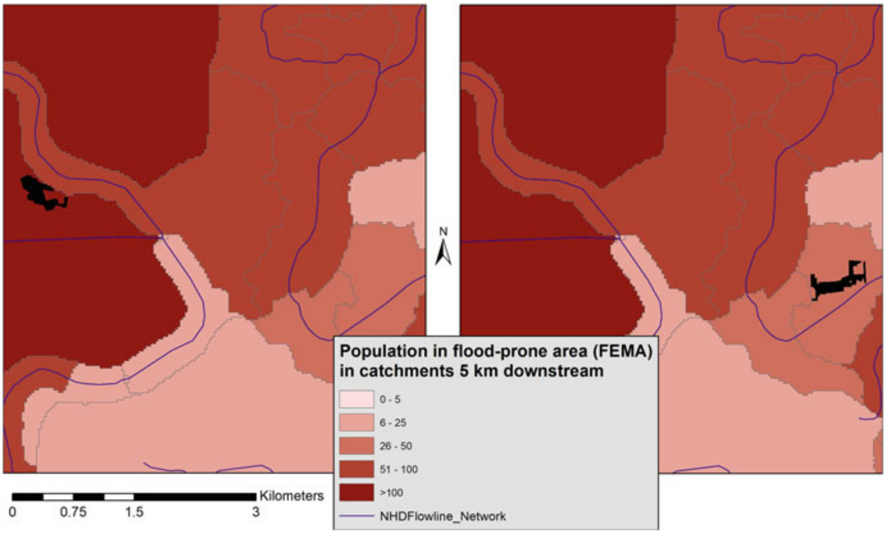


Fig. 7 Once catchments results from the csv file are joined to a spatial catchments dataset sites can be overlaid and compared based on characterizations for the catchment they fall within. This example shows a site (black) in a catchment with >100 people in FEMA flood-prone areas downstream of the site (Left) compared to a nearby site (black) in a catchment with 33 people in FEMA flood-prone areas downstream of the site (Right)

flood-prone areas with the same dasymetric 2010 population data and use the same methods to combine estimates from downstream catchments. Where there are more people in flood-prone areas downstream there is more demand for increased flood-reduction benefits. Thus, it is a maximizing criterion, and a higher value makes a catchment higher priority for restoration.

For the second indicator, *is the service scarce*, existing wetlands may already be providing flood-risk reduction services for the same beneficiaries. We recommend using values in the field "wet_pct", which represent the percent of that catchment that is currently wetlands. In catchments with abundant wetlands already providing flood-reduction services, added units of this service have less value than in catchments where wetlands providing flood-reduction services are scarce. Thus, it is a minimizing criterion, and a lower value makes a catchment higher priority for restoration.

The two indicator field values may need manipulation to inform decision making. Regions have physiographic differences that impact wetland suitability, flood-risk, and population density. As a result, the distribution of values within different regions is diverse. For example, a 10% difference in wetlands between catchments in west Texas is drastic but would be minor in coastal Minnesota. To account for this, Bousquin and Hychka (2019) suggest binning the indicators into discrete categories based on the distribution of regional values, and show a method using four quartiles, dividing catchment values into four categories of equal number. Different discretizations are better suited depending on thresholds, decision context and overarching objectives. Many GIS applications aid users in choosing discrete categories when visualizing data.

5 Summary

The RBI approach helps formalize a process for developing benefit indicators for site-level assessment. The RBI approach is a rapid screening assessment, meant to reduce the burden in expertise, data, and cost placed on practitioners using socio-economic metrics. The three tools described here were developed to help decision-makers apply the RBI approach indicators in a consistent and transparent way. These tools were developed for a subset of EGS and applied within the context of urban freshwater wetlands and could be modified or reproduced for other ecosystems and services. In a case-study application of the spatial analysis tool, we screened 65 candidate restoration sites within a watershed, using the RBI combined with MCDA methods, and identified four preferred restoration sites. These sites were further considered by a local watershed group, which subsequently proposed three of the sites for restoration, based on the sites' potential EGS values to local beneficiaries (Martin et al. 2018).

Each of the RBI tools addresses different user needs. The *checklist tool* is for recording results of manually-conducted assessments or for developing new indicators for other types of environmental changes or within different ecological systems. The *spatial analysis tool* is for users with some GIS expertise who have geospatial data to apply the existing rapid benefit indicators for urban freshwater wetland restoration sites. The *NHDPlus national catchment dataset* is for users only interested in assessing flood-protection benefits. Users of this dataset must be willing to accept reduced flexibility in exchange for not having to perform analysis on multiple input data sets.

We have presented a framework for consistently developing a set of benefit indicators, along with a set of tools for applying this framework. Alongside ecosystem service metrics, benefit indicators help ecological and social interactions be better considered within an Ecosystem-Based Management framework. When stakeholder values are used to select ecosystem services to assess and are reflected in benefits assessments selection trade-offs are better informed and social-ecological systems are better connected in Ecosystem-Based management decision-making.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Part IV
Governance

The Ecosystem Approach in International Marine Environmental Law and Governance

Sarah Ryan Enright and Ben Boteler

Abstract An ecosystem approach to the management of human activities in the marine environment began to feature as a normative concept in international instruments in the 1980s, beginning with the pioneering Convention on the Conservation of Antarctic Marine Living Resources. While an implicit basis for the ecosystem approach can be found in the 1982 Law of the Sea Convention, much of the additional conceptual development at the global level has occurred within the framework of the 1992 Convention on Biological Diversity. The subsequent widespread acceptance of the ecosystem approach has been described as a response to the failure of reactive and fragmented sectoral and zonal approaches to environmental protection and management. A consensus has emerged that a paradigm shift in thinking is needed, whereby traditional modalities of governance are replaced by proactive, integrative and holistic approaches involving adaptive management and greater cooperation between States, international institutions and other stakeholders in order to achieve effective and long-term, coherent implementation of policies across sectors. This chapter will discuss the origins and evolution of the ecosystem approach in international law, which can now be found in a wide range of international and regional instruments, including the regional seas conventions, fisheries management agreements, as well as the ongoing negotiations to develop an internationally legally binding instrument for the conservation and sustainable use of marine biodiversity beyond national jurisdiction. Finally, challenges to the operationalization of the concept in practice will be discussed.

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Lessons Learned

- There is no universally agreed definition of the Ecosystem Approach (EA) in international law.
- The Ecosystem Approach has thus far been developed largely as a set of non-binding soft law principles; therefore, its normative content remains weak and unclear in terms of its practical application and obligations on States.
- The Ecosystem Approach and adaptive management (AM) have received little legal scholarly attention in comparison to the closely associated precautionary principle.
- The Convention on Biological Diversity is a leader in the adoption of the Ecosystem Approach and has done significant work to elaborate its interpretation and application. The Malawi Principles and Operational Guidance remain relevant as a framework for action.
- It continues to be a challenge to operationalize the Ecosystem Approach in law and practice due to the uncertainties surrounding its meaning and potential approaches for implementation.

Needs to Advance EBM

- More practical guidance is needed on how the Ecosystem Approach is to be implemented in practice at global, regional and national levels.
- Adaptive Management has been deemed essential for the operation of the Ecosystem Approach, yet it remains controversial from a legal perspective. More practical guidance is needed on operationalising Adaptive Management.
- Cooperation and coordination are critical to the success of the Ecosystem Approach, yet they remain difficult to achieve. More political will is needed in order to make progress here.

1 Introduction

Despite the importance of biological diversity for life, it is now rapidly declining at alarming rates and marine biodiversity is no exception (IPBES 2019; Grooten and Almond 2018). The 2019 Global Assessment Report on Biodiversity and Ecosystem Services revealed *inter alia* that natural ecosystems had lost half their area, two thirds of the marine environment had been 'severely altered' by human activity and approximately one third of reef forming corals, sharks, and marine mammals are threatened with extinction. The ongoing degradation of ecosystems has forced an acknowledgement of the limitations of previous sectoral and species specific approaches to resource management and environmental protection, leading to the emergence of holistic governance alternatives, which emphasize connectivity and integration (Harrison 2017). The 2016 United Nations (UN) World Ocean Assessment (p. 9) emphasized that "the ocean is a complex set of systems that are all interconnected" and recognized that the development of ocean management had progressed from "no regulation to the regulation of specific impacts, to the regulation of sector-wide impacts and, finally, to regulation taking account of aspects of all relevant sectors." Out of an increased scientific understanding of the importance of ecosystems and ocean connectivity, the 'ecosystem approach' has emerged as a dominant paradigm in international ocean governance.

This chapter will trace the development of the ecosystem approach in international environmental law, from its origins in soft law instruments to becoming endorsed as the main framework for action under the Convention on Biological Diversity (CBD), and its subsequent widespread application in a marine governance context. Finally, challenges to the operation of the concept in practice will be discussed.

2 The Core Elements of the Ecosystem Approach

There is no universally agreed definition of the ecosystem approach in international law (UNGA 2006). The Secretariat of the CBD[1] has described it as being difficult to define in a simple manner (CBD 2004, p. 3), while de Lucia goes further calling it an "elusive, unstable and contested" concept (2015, p. 93) whose various articulations render the task of finding a meaningful common denominator challenging (De Lucia 2018). The ecosystem approach has been interpreted differently by various environmental institutions and regimes (Platjouw 2016), and is referred to interchangeably as 'Ecosystem-Based Management'[2] in international discourse (on definitions see further Delacámara et al. 2020). It is likely that the evolving nature of the ecosystem approach has been a contributing factor to the lack of clarity surrounding its meaning. It is a concept which continues to develop in parallel with scientific understanding of the nature of ecosystems and their core principles (Long 2012).[3] In fact, Morgera (2017, p. 71) has suggested that the translation of the scientific notion of the ecosystem into a legal construct has provided the basis for the normative development of the ecosystem approach, thereby having a "law-making effect".

Although it remains underdeveloped in comparison to related approaches such as the precautionary principle (Morgera 2017), an increasing amount of doctrine (see references for a comprehensive list) and technical guidance (e.g. FAO 2003; CBD 2004) has helped clarify the meaning and application of the ecosystem approach, as well as its core elements. Connectivity and integration are central to the idea. An early study by Brunnée and Toope (1994, p. 55) describe it as requiring:

> *consideration of the whole system rather than individual components. Living species and their physical environments must be recognized as interconnected, and the focus must be on the interaction between different sub-systems and their responses to stresses resulting from human activity.*

[1]Convention on Biological Diversity, June 5, 1992, 1760 UNTS 79.

[2]On Ecosystem Based Management, see *inter alia*, R Grumbine (1994), RD Long et al. (2015), SD Langhans et al. (2019).

[3]See *inter alia*, D Tarlock (2007, pp. 577–579), D Diz (2012, pp. 1–3), RD Long et al. (2015, pp. 54–56) for a brief history of the ecosystem concept.

Amidst the confusion surrounding its meaning, Trouwborst (2009) reminds us that the purpose of the ecosystem approach is the preservation and/or restoration of ecosystem health or integrity. He goes on to extract three strands of generic agreement (p. 28):

> *(1) The holistic management of human activities, (2) based on the best available knowledge on the components, structure and dynamics of ecosystems, (3) and aimed at satisfying human needs in a way that does not compromise the integrity, or health, of ecosystems.*

The work of the UN General Assembly has also been helpful in generating consensus on key components of the ecosystem approach. At the seventh session of the Open-ended Informal Consultative Process on Oceans and the Law of the Sea (UNICPOLOS) in 2006, the resulting report (ICP-7) provided a comprehensive list of elements including *inter alia*:

> (a) *Emphasize* ***conservation*** *of ecosystem structures and their functioning and key processes in order to maintain ecosystem goods and services;*
> (b) *Be applied within* ***geographically specific areas based on ecological criteria****;*
> (c) *Emphasize the* ***interactions*** *between human activities and the ecosystem and among the components of the ecosystem and among ecosystems;*
> (d) *Take into account* ***factors originating outside the boundaries*** *of the defined management area that may influence marine ecosystems in the management area*;
> (e) *Be inclusive, with* ***stakeholder and local communities' participation*** *in planning, implementation and management;*
> (f) *Be based on* ***best available knowledge****, including traditional, indigenous and scientific information and be* ***adaptable*** *to new knowledge and experience;*
> (g) *Assess risks and apply the* ***precautionary approach****;*
> (h) *Use* ***integrated*** *decision-making processes and management related to multiple activities and sectors*. (UNGA 2006, para. 6. Emphasis added)

Given that scientific understanding of ecosystems is incomplete, the ecosystem approach has been closely associated with the precautionary principle and adaptive management (Morgera 2017). The precautionary principle[4] entails taking early, preventative action in response to environmental threats, even in the absence of scientific certainty (Trouwborst 2009), and has been described as an "integral component" of the ecosystem approach.[5] Adaptive management offers a practical tool for dealing with law's apparent incompatibility with uncertainty. It provides a "flexible decision-making process that can be adjusted in the face of uncertainties as outcomes from management actions and other events become more understood through careful monitoring of these outcomes" (Williams et al. 2009).[6] It is often described as an iterative or ongoing learning process (Morgera 2017). The CBD has explained that the ecosystem approach requires adaptive management "to deal with

[4]Rio Declaration on Environment and Development (13 June 1992) 31 ILM 874, Principle 15.

[5]Declaration of the First Joint Ministerial Meeting of the Helsinki and OSPAR Commissions (Bremen, 26 June 2003) (OSPAR/HELCOM statement), Annex 5 ('Towards an Ecosystem Approach to the Management of Human Activities'), para 5.

[6]Referred to in Le Lievre (2019, p. 496), as the most recognized definition of adaptive management in the literature.

the complex and dynamic nature of ecosystems and the absence of complete knowledge or understanding of their functioning."[7]

Several international organizations have adopted working definitions of the ecosystem approach and attempted to make progress on elaborating its meaning and operation. The Conference of Parties (COP) to the CBD have defined it in light of the objectives of the Convention (Platjouw 2016)[8]:

> *a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way.*[9]

This definition is concerned with integration[10] and equity,[11] recognizing that humans are an integral component of many ecosystems.[12] Moynihan (2020) describes integration in the context of the ecosystem approach as meaning integration across sectors, between governance levels, between modern science and traditional methods and between different legal and management strategies. It is noteworthy that no particular spatial unit of scale is included in the CBD definition, rather the scale of analysis and action is to be determined by the problem being addressed.[13] The International Council for the Exploration of the Seas (ICES) adopted the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR)[14] definition (de Lucia 2018), which focuses on the management of human activities[15]:

> *The comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity.*[16]

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[7]CBD-COP 5 Decision V/6 'Ecosystem Approach' Doc UNEP/COP/5/23, (2000), A (4).

[8]CBD-COP 5 Decision V/6, A (1) states that the application of the ecosystem approach will help to reach a balance of the three objectives of the Convention: conservation, sustainable use, and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources.

[9]CBD-COP 5 Decision V/6 (2000), A (1).

[10]CBD-COP 7, Decision VII/11 'Ecosystem Approach' Doc UNEP/CBD/COP/7/21 (13 April 2004), para. A.3 referred to the ecosystem approach as providing an integrating framework for the implementation of the Convention's objectives.

[11]CBD-COP 5 Decision V/6 (2000), para. 6. Principle 1 states that ecosystems should be managed for their intrinsic values and for the tangible or intangible benefits for humans, in a fair and equitable way. The operational guidance contained in the same Decision at para. 9 promotes the fair and equitable sharing of benefits with the stakeholders responsible for managing ecosystems and supporting ecosystem services. See M Ntona and E Morgera (2018, p. 218).

[12]CBD-COP 5 Decision V/6 (2000), A (2). See E Morgera (2017, p. 72).

[13]OSPAR/HELCOM statement (2003), para 3.

[14]OSPAR is a regional mechanism by which 15 Governments and the EU cooperate to protect the marine environment of the North-East Atlantic. https://www.ospar.org/about

[15]Guidance on the Application of the Ecosystem Approach to Management of Human Activities in the European Marine Environment (2005) ICES Cooperative Research Report no. 273, 4.

[16]OSPAR/HELCOM statement (2003), para. 5.

The OSPAR Commission has stated that “the essence of the ecosystem approach is to allow sustainable exploitation of natural resources while maintaining the quality, structure and functioning of marine ecosystems.”[17] Long (2012) observes that the rationale for adopting such an anthropogenic approach is that while the ecosystem itself may not be managed, the human activities that interact with and impact upon the ecosystem may be managed with a view to conserving biodiversity. The UN General Assembly has also made it clear that ecosystem approaches “should be focused on managing human activities in order to maintain, and, where needed, restore ecosystem health.”[18] The anthropocentric focus is also illustrated via the deployment of the ecosystem approach in connection with the conceptual framework of ecosystem services (see further O’Hagan 2020),[19] seen by many as one of the core elements of the ecosystem approach (de Lucia 2015). Indeed, several definitions of the ecosystem approach refer explicitly to the ecosystem services they provide.[20]

3 Emergence and Development of the Ecosystem Approach in International Law

The ‘ecosystem approach’ as a normative framework is a relatively recent development. The first inklings of the ecosystem approach and of ecosystems becoming an object of conservation and protection in international law can be traced back to the 1970s (see further Long 2012; Platjouw 2016). Several non-binding soft-law instruments,[21] beginning with the 1972 Stockholm Declaration on the Human Environment, contained formative elements of what would become the ecosystem approach.[22] The adoption of the 1971 Ramsar Convention on Wetlands of

[17]OSPAR Commission Quality Status Report 2010, 9.

[18]Resolution 61/222 on Oceans and the Law of the Sea (20 December 2006), para. 119 (b); Resolution 62/215 (22 December 2007), para 99(b); Resolution 63/111 (5 December 2008), para 117(b). Cited in A Trouwborst (2009, p. 28).

[19]In simple terms, ecosystem services are the benefits humans obtain from ecosystems such as clean air, water, food, fuel, climate regulation, and recreation. See further the *Millennium Ecosystem Assessment 2005,* which provides a typology of four categories of ecosystem services: supporting, provisioning, regulating, and cultural services.

[20]For example, the definition adopted by the UN Environment Programme (UNEP) is similar to the CBD but replaces ‘conservation’ with ‘sustainable delivery of ecosystem services’. See UNEP (2016, p. 8).

[21]The use of the adjective ‘soft’ to describe the legal status of an instrument is intended to indicate that the instrument is not legally binding, regardless of its content. However, soft law instruments and the conferences and institutions that they create are very influential in international environmental law and have an important normative function. See further PM Dupuy and JE Viñuales (2015, p. 35).

[22]Principle 2 of the Stockholm Declaration states that “the natural resources of the earth. . .especially representative samples of natural ecosystems, must be safeguarded for the benefit of present and future generations through careful planning or management. . .” 10 years later, the

International Importance was also an important environmental milestone of this era.[23] The notion of 'wise use' is at the heart of the Convention and has been explicitly linked to the ecosystem approach.[24] The focus of the Convention has shifted over time from an original treaty on waterfowl habitat, to the protection of wetlands as an ecosystem, to the ecosystem services provided by wetlands (Dupuy and Viñuales 2015), illustrating the normative evolution of ecosystem protection. The 1973 Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)[25] and the 1979 Convention on Migratory Species (CMS) [26] also warrant a brief mention: while they are focused on the protection of species, they also refer to the importance of these species within their ecosystems (Platjouw 2016), which has an indirect effect of promoting habitat conservation and thus the conservation of ecosystems (Tarlock 2007). The ecosystem approach is currently taken into account in CITES practice.[27]

Beginning in the early 1980s, specific reference to the ecosystem approach began to appear in a number of international treaties concerning the marine environment (Long 2012). The 1980 Convention on the Conservation of Antarctic Marine Living Resources (CAMLR)[28] was one of the first instruments to utilize the ecosystem approach as a primary normative framework (Sands et al. 2018) and is generally regarded as a leader in its implementation (Fabra and Gascon 2008).[29] The CAMLR

UN General Assembly, in principle 4 of the World Charter for Nature (28 October 1982) A/RES/37/7 called upon States to manage ecosystems and organisms in such a way as not to endanger the integrity of those other ecosystems or species with which they coexist. For a more detailed overview, see A Trouwborst (2009, p. 29).

[23]Convention on Wetlands of International Importance especially as Waterfowl Habitat 996 UNTS 245.

[24]The definition of 'wise use' was updated in 2005, taking into account the widespread acceptance of the ecosystem approach: "Wise use of wetlands is the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development". Ramsar, Conference of the Parties 9 'A Conceptual Framework for the wise use of wetlands and the maintenance of their ecological character' (November 2005) Resolution IX.1 Annex A (2005), para. 22. The definition explicitly cites the ecosystem approach as developed by the CBD (COP5 Decision V/6) and that applied by HELCOM and OSPAR in their Joint Statement in 2003. See further, CM Finlayson et al. (2011, p. 191), E Morgera (2017) highlights an interesting circular evolution here whereby the ecosystem approach elaborated under the CBD built upon the earlier notion of 'wise use' contained in the Ramsar Convention.

[25]Convention on International Trade in Endangered Species of Wild Fauna and Flora 983 UNTS 243.

[26]Convention on the Conservation of Migratory Species of Wild Animals 1651 UNTS 333.

[27]CITES, Fifty-third meeting of the Standing Committee, Synergy between CITES and the Convention on Biological Diversity (CBD) (June 2005) SC53 Doc.8 (rev. 1). Cited in FM Platjouw (2016, p. 30).

[28]Convention on the Conservation of Antarctic Marine Living Resources, Canberra, 20 May 1980, 19 ILM 841.

[29]See pp. 575–581 for a detailed discussion of the implementation of the ecosystem approach in the CAMLR regime.

covers the entire Antarctic marine system[30] and has a broad mandate to conserve Antarctic marine living resources, which includes their 'rational use' (Arts. II (1) and (2)). This means that 'harvesting and associated activities' are permitted in the CAMLR area as long as such exploitation does not endanger the population levels of the harvested species or the ecological relationship as a whole between the marine living resources in the area (Art. II(3)).[31] Furthermore, the CAMLR prohibits changes to the marine ecosystem which are not potentially reversible over two or three decades (Art. II(3)(c)). The CAMLR is a good illustration of the ecosystem approach in action via its incorporation of basic principles of ecosystem ecology, its recognition of the importance of ecosystem interrelationships and its focus on the various components of the marine ecosystem (de Lucia 2015).

1982 heralded the adoption of the United Nations Convention on the Law of the Sea (UNCLOS),[32] which provides the overarching legal framework for the governance of the oceans. In contrast to the CAMLR, the ecosystem approach manifests itself in a more implicit manner in UNCLOS (Platjouw 2016). While it does recognize that "the problems of ocean space are closely interrelated and need to be considered as a whole",[33] and contains some elements of integrated decision making,[34] UNCLOS contains few explicit references to the concept of the ecosystem,[35] and promotes a zonal and sectoral approach to ocean governance (Scott 2015). A critical turning point was the adoption of Agenda 21 at the 1992 United Nations Conference on Environment and Development (UNCED)[36] which, via its explicit promotion of a holistic approach to oceans management, became a catalyst for

[30]Which it describes as 'the complex of relationships of Antarctic marine living resources with each other and with their physical environment' in Article I (3) CAMLR.

[31]See also R Long (2012, pp. 433–434), V de Lucia (2015, pp. 107–108), D Langlet and R Rayfuse (2018, p. 2).

[32]Convention on the Law of the Sea, Dec. 10, 1982, 1833 U.N.T.S. 397.

[33]Third Recital to Preamble of UNCLOS.

[34]See Articles 61 and 119 UNCLOS which in the context of fisheries require decisions to consider environmental, scientific, economic, and social factors and to consider the impact on associated or dependent species when establishing conservation measures. See further E Kirk (2015, p. 40).

[35]See Article 194(5) UNCLOS which requires parties to protect rare or fragile ecosystems and Article 145(a) which calls upon States to prevent interference with the "ecological balance of the marine effects of fishing on dependent or associated species".

[36]UNCED, Agenda 21: Programme of Action for Sustainable Development (1992) UN Doc A/Conf. 151/26. The 1992 Rio Declaration, *op cit*, also adopted at UNCED, recognised the "integral and interdependent nature of the Earth" in its Preamble. An important precursor to UNCED was the 1987 Brundtland Commission Report 'Our Common Future', which introduced the concept of sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" and linked it to conservation of ecosystems. See Report of the World Commission on Environment and Development, 'Our Common Future' 10 March 1987, Chapter 2.

further development of the ecosystem approach (Trouwborst 2009). Chapter 17 (para.1) of Agenda 21 underlined the importance of new approaches to marine management, at national, regional, and global levels, "that are integrated in content and are precautionary and anticipatory in ambit".

The parties to the CBD subsequently approved the ecosystem approach as the primary framework for implementation of its objectives in 1995,[37] making it the first international treaty to take a holistic, ecosystem-based approach to biodiversity conservation and sustainable use (CBD 2004). The CBD is considered a leader in the adoption of the ecosystem approach and has done more to elaborate the concept than any other regime (de Lucia 2018), capitalizing on previous legal developments in international environmental law such as sustainable forest management.[38] While the CBD contains a definition of an 'ecosystem' as "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit" (Art. 2), there is no explicit basis for the ecosystem approach in the text of the CBD.[39] Due to the lack of development of the notion at an international level, the CBD parties recognized the need to elaborate on its interpretation and application.[40] Thus, at their fifth meeting in Nairobi, Kenya in 2000, the COP agreed upon a definition (discussed in Sect. 2 above), recommended the implementation of 12 interlinked and complementary principles of the ecosystem approach, known as the Malawi Principles,[41] and also issued five points of Operational Guidance for their application.[42] At their seventh meeting in 2004, the COP confirmed that the establishment and maintenance of systems of protected areas play an essential part in implementing the ecosystem approach and achieving the objectives of the Convention.[43]

[37]CBD-COP 2 Decision II/8 (November 1995), para 1. CBD-COP 7, Decision VII/11, para. A.3.

[38]CBD-COP 7 Decision VII/11 (2004), para. 7 and Annex II; CBD Guidelines (2004), Annex III. See E Morgera (2017, p. 71).

[39]However, Platjouw points out that both the protection of ecosystems as well as the rehabilitation and restoration of degraded ecosystems are promoted in Articles 8(d) and 8(f) of the Convention. See FM Platjouw (2016, p. 32).

[40]In CBD-COP 4 Decision IV/1, B (1998), the need for a workable description and further elaboration of the ecosystem approach was acknowledged. See E Morgera (2017, p. 71).

[41]CBD-COP 5 Decision V/6 (2000), Section B.

[42]Ibid., Section C. See CBD-COP 7 Decision VII/11 (2004 and CBD Guidelines (2004) for detailed guidance on the rationale behind the Malawi Principles and their implementation.

[43]CBD-COP 7 Decision VII/28 (2004) UNEP/CBD/COP/DEC/7/28, para. 1.

Malawi Principles

1. The objectives of management of land, water and living resources are a matter of societal choice.
2. Management should be decentralised to the lowest appropriate level.
3. Ecosystem managers should consider the effects (actual or potential) of their activities on adjacent and other ecosystems.
4. Recognising potential gains from management there is a need to understand the ecosystem in an economic context.
5. Conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach.
6. Ecosystems must be managed within the limits of their functioning.
7. The ecosystem approach should be undertaken at the appropriate spatial and temporal scales.
8. Recognising the varying temporal scales and lag effects that characterise ecosystem processes, objectives for ecosystem management should be set for the long term.
9. Management must recognise that change is inevitable.
10. The ecosystem approach should seek the appropriate balance between, and integration of, conservation and use of biological diversity.
11. The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices.
12. The ecosystem approach should involve all relevant sectors of society and scientific disciplines.

CBD Operational Guidance for Application of the Ecosystem Approach

1. Focus on relationships and processes within ecosystems.
2. Enhance benefit-sharing.
3. Use adaptive management practices.
4. Carry out management actions at the scale appropriate for the issue being addressed with decentralization to the lowest level, as appropriate.
5. Ensure inter-sectoral cooperation.[44]

[44] CBD guidance describes inter-sectoral cooperation as a need to integrate the ecosystem approach into different sectors that impact biodiversity, including agriculture, fisheries and forestry and calls for increased communication and cooperation at a range of levels to achieve this e.g. through inter-ministerial bodies or information sharing networks. See CBD-COP 5 Decision V/6 (2000), para. 12 and CBD Guidelines (2004), Annex I.

After the ecosystem approach was endorsed by the parties to the CBD, it gained widespread recognition,[45] particularly in a fisheries management context,[46] where it has been termed the 'ecosystem approach to fisheries' (EAF) (UNEP 2016).[47] The Food and Agriculture Organization of the United Nations (FAO) has promoted the ecosystem approach as best practice.[48] For example, the 1995 FAO Code of Conduct for Responsible Fisheries recognizes the transboundary nature of aquatic ecosystems (Art. 6(4)) and its provisions have a broad scope to protect target and non-target species as well as the ecosystems associated with those species (Platjouw 2016).The ecosystem approach also became a key feature of the 1995 United Nations Fish Stocks Agreement (UNFSA),[49] which was designed to apply to fish stocks, regardless of their geographic location and therefore requires States to take into account the transboundary impacts of their decisions.[50] The precautionary approach is explicitly mentioned in UNFSA and is considered to be an essential component of the EAF.[51] UNFSA also created an obligation for States to cooperate through Regional Fisheries Management Organizations (RFMOs),[52] several of which also adopted the ecosystem approach (Sands et al. 2018). However, the actualization of the EAF in this context has been hampered by the fact that RFMOs do not cover the world's oceans and fishing resources in a comprehensive manner. RFMOs generally manage stocks either on a species specific or geographic basis, thus leaving many areas unregulated and many stocks and species unmanaged (Rayfuse 2016).

[45]E.g. The UN Convention on the Law of Non-Navigational Uses of International Watercourses (21 May 1997, entered into force 17 August 2014)) created an obligation for States to "protect and preserve the ecosystems of international watercourses", Arts. 20, 22 and 23. On the ecosystem approach and international water law, see further O McIntyre (2014, 2018), R Moynihan (2017, 2020). It was also endorsed in soft law by the 2002 World Summit on Sustainable Development in its Plan of Implementation, which emphasized the need to "develop and facilitate the use of diverse approaches and tools, including the ecosystem approach" in accordance with Chapter 17 of Agenda 21. See the Johannesburg Plan of Implementation of the World Summit on Sustainable Development (2002), UN Doc. A/CONF.199/20, para. 31 c.

[46]For example, the 2001 Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem recognized the importance of interactions between fishery resources and all components of the ecosystem, and the need to conserve marine environments and called upon States to develop best practice guidelines for introducing ecosystem considerations into fisheries management. See further EJ Molenaar (2002) and M Barange (2003).

[47]On the EAF, see generally, D Diz (2012) and FAO (2003).

[48]See e.g. FAO Code of Conduct for Responsible Fisheries 1995 and FAO International Guidelines for the Management of Deep-Sea Fisheries in the High Seas 2008.

[49]Agreement for the Implementation of the Provisions of the UN Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, 4 August 1995 (into force 11 December 2001) 2167 UNTS 3. UNFSA supplements UNCLOS and obliges coastal States and States fishing on the high seas to *inter alia* protect biodiversity in the marine environment and apply the precautionary and ecosystem approaches, with a view to conserving straddling and highly migratory fish stocks.

[50]Arts. 5 and 6 UNFSA. See E Kirk (2015, p. 40).

[51]Art. 5 (c) and Art. 6 UNFSA. See D Diz (2017, p. 131).

[52]Arts. 10, 11 and 12. On RFMOs, see generally R Rayfuse (2015).

Regional seas conventions (RSCs) are generally viewed as being more consistent with an ecosystem approach given that they have geographical as opposed to sectoral scope (Barritt and Viñuales 2016). However in practice they have not been as effective as hoped (Wang 2004); they have limited mandates, which only apply to States which are parties to the relevant treaty, and exclude many relevant human activities from their scope of application (Rochette et al. 2015). Also, most RSCs do not cover the high seas.[53] Different RSCs tend to emphasize different aspects of the ecosystem approach depending on the regional context (Langlet and Rayfuse 2018; Kirk 2015), however elements such as the precautionary principle,[54] recognizing the impact of transboundary activities,[55] the best use of scientific knowledge and advice,[56] and the involvement of stakeholders[57] can be found in several. The ecosystem approach has been explicitly endorsed by the parties to the Helsinki[58] and OSPAR[59] Conventions, with a recognition that the marine environment is both an ecosystem and interlocking network of ecosystems,[60] and it has been described as the 'overarching principle' in the OSPAR Commission's work.[61] The OSPAR scheme for implementing the ecosystem approach has been described as one of the most highly developed in international environmental law (Long 2012). It embraces an adaptive management approach via its use of a 'continuous cycle of steps' which involve setting and coordinating ecological objectives and associated targets and indicators, ongoing management, and regular updating of ecosystem knowledge, research, and advice.[62]

At the global level, the ecosystem approach has featured in the draft text of a new internationally legally binding instrument (ILBI) under UNCLOS on the conservation and sustainable use of marine biodiversity beyond national jurisdiction (BBNJ),[63] negotiations for which began in September 2018.[64] The BBNJ

[53]With the exception of OSPAR, Barcelona Convention, Noumea Convention, Lima Convention and CAMLR. See UN Environment (2017).

[54]E.g. Art. 3(2) Helsinki Convention; Art. 2(2)(a) OSPAR Convention.

[55]E.g. Art. 3 (6) Helsinki Convention; Art. 11 Barcelona Convention.

[56]E.g. Art. 13 Barcelona Convention.

[57]E.g. Art. 17 Helsinki Convention; Art. 15 Barcelona Convention.

[58]Convention on the Protection of the Marine Environment of the Baltic Sea Area, 1992 1507 UNTS 167.

[59]Convention for the Protection of the Marine Environment of the North-East Atlantic 1992 2354 UNTS 67.

[60]OSPAR/HELCOM statement (2003), para. 3.

[61]Preamble to Strategy of the OSPAR Commission for the Protection of the Marine Environment of the North-East Atlantic 2010–2020, OSPAR Agreement 2010–3.

[62]OSPAR Strategy 2010–2020, para 4.3.

[63]Draft text of an agreement under the United Nations Convention on the Law of the Sea on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction. Note by the President. Advance unedited version, 25 June 2019.

[64]Resolution 72/249 adopted by the United Nations General Assembly on 24 December 2017. The package of issues for negotiation is limited to: marine genetic resources, including benefit-sharing,

negotiations represent a recognition that measures by individual States or regional bodies are not sufficient to conserve the high seas due to the transboundary nature of the ocean. Furthermore, significant regulatory gaps in the existing international governance framework have prevented progress on addressing the increasing threats to high seas biodiversity.[65] Thus, the development of the ILBI can be viewed as a response to the previously sector specific and uncoordinated approach taken to govern the ocean, thereby demonstrating an endorsement of the ecosystem approach.

4 Operational Challenges

As can be seen from the above discussion, the ecosystem approach has been included in a wide range of ocean instruments. However, its application varies from treaty to treaty with none incorporating all aspects of the approach, likely a result of piecemeal and sectoral development to date (Kirk 2015). The CBD Secretariat has pointed out that there is no single way to implement the ecosystem approach as application will vary depending on the specific context, including local, national, regional, or global conditions (CBD 2004). Therefore, in practical terms, the ecosystem approach is a normative framework, which needs to be tailored to specific circumstances.[66] This results in a 'plurality of approaches' rather than a single 'true' version of the ecosystem approach (de Lucia 2015). In 2004, at COP 7, additional rationale and implementation guidelines for the Malawi principles were provided, whereby a mainstreaming of the ecosystem approach into national and regional biodiversity strategies, action plans, policy instruments, planning processes, and sectoral plans was promoted.[67] Despite these efforts, the principles have not been applied widely in practice as they are viewed as too complex or vague (Langlet 2018; Platjouw 2016).[68] They also allow much to be decided at a later stage, thus enabling action to be deferred (Kirk 2015).

area-based management tools, including marine protected areas; environmental impact assessments; and capacity-building and marine technology transfer.

[65]For a detailed discussion on identified gaps in high seas governance, see further KM Gjerde et al. (2019).

[66]CBD COP Decision IX/7, Ecosystem Approach (2008), UNEP/CBD/COP/DEC/IX/7, Preamble, para (a).

[67]CBD-COP 7, Decision VII/11 (2004), Annex 1, para 5.

[68]The EU, which is a party to the CBD, has embraced the ecosystem approach as a central theme in its marine governance legislation, including the Water Framework Directive 2000/60/EC, the Marine Strategy Framework Directive 2008/56/EC and the Maritime Spatial Planning Directive 2014/89/EU. However, challenges remain at the implementation level, especially in a fisheries context. See further, J Wakefield (2018), N Soininen and FM Platjouw (2018), D Langlet and R Rayfuse (2018, p. 449).

4.1 Scientific Uncertainty

Reasons for such inertia include the different interpretations of the concept by various actors, as highlighted earlier, and the difficulty in translating the evolving scientific understanding of ecosystems into law (Tarlock 2007). The ecosystem approach is underpinned by a comprehensive scientific knowledge base, however gaps in knowledge, scientific uncertainty, and dynamic multiple-scale ecosystem processes make it difficult to implement in a way that ensures legal stability and predictability (de Lucia 2018). In recognition of the fact that ecosystems change, parties to the CBD stipulated that the ecosystem approach must use adaptive management to anticipate and cater for such changes.[69] While appearing counter-intuitive at first,[70] adaptive management models, which enable new knowledge to be incorporated in a tailor made fashion as it becomes available, can provide solutions to the problems of scientific and legal uncertainty (Trouwborst 2009).[71] In this way, the implementation of the ecosystem approach is also in a constant state of evolution (Long 2012). Despite the allegedly 'limitless' legal options for implementing the ecosystem approach (Belsky 1985, p. 763),[72] Langlet and Rayfuse (2018) point out that the variety and complexity of both natural ecosystems and the institutional, legal, and administrative systems created for their management is what makes the effective implementation of the ecosystem approach so highly challenging. Given the context specific nature of the application of the ecosystem approach, it has been suggested that it is more constructive to view the Malawi principles as an overarching framework of understanding more than an explicit strategy (Langlet and Rayfuse 2018). Kirk (2015) has suggested that the lack of precise prescription as to how the ecosystem approach is to be implemented can be viewed positively, in the sense that it allows for tailored adaptation in response to the needs of particular ecosystems.

4.2 Institutional Fragmentation and Spatial Mismatch

Spatial mismatch between ecological boundaries and governance regimes has been a challenge for the effective operation of the ecosystem approach (Tanaka 2004; Kirk 1999).[73] The CBD envisages an ecosystem approach whereby the appropriate scale

[69] CBD COP 5 Decision V/6 (2000), Principle 9.

[70] BA Cosens et al. (2017, p. 16), observes that although law has often been viewed as a constraint on adaptation, it has proven highly adaptive over time.

[71] See also CBD-COP 5 Decision V/6 (2000), Section C. On adaptive management, see *inter alia*, JB Ruhl (2006), AJ Garmestani et al. (2008), DA Keith et al. (2011), E Raitanen (2017), Le Lievre (2019).

[72] Cited by R Long, 426. See the list of implementation options suggested by the UNGA (2006) at para. 7 as an example.

[73] On socio-ecological scale mismatch, see GS Cumming and others (2006).

of management action is to be determined by the problem to be addressed.[74] This is difficult to achieve on a global scale as the ocean is divided into areas under national State jurisdiction and the high seas, also known as areas beyond national jurisdiction (ABNJ), over which no State exercises unilateral control (Harrison 2017). The CBD is focused on the protection of marine biodiversity within the limits of national State jurisdiction,[75] thus leaving the high seas under the purview of the UNCLOS legal framework and other international and regional agreements.[76] This has resulted in major governance gaps, which the BBNJ negotiations are now seeking to redress. The challenges which arise due to the lack of spatial fit have been aggravated by the absence of a single overarching global body with the authority to adopt management measures for marine biodiversity conservation that apply to the entire ecosystem (Harrison 2017; Long 2012). As a solution, increased procedural cooperation and linkages between the various existing ocean regulatory regimes have been proposed (Tanaka 2004; Kirk 1999). Successful examples of inter sectoral cooperation on a global level include the work of the International Maritime Organization (IMO) and FAO on tackling Illegal, Unregulated and Unreported (IUU) Fishing[77] and in a biodiversity context, the close cooperation and coordination between the COPs of the CBD, CITES and CMS.[78] Regionally, institutional cooperation is taking place to coordinate fisheries activities in the North East Atlantic,[79] in relation to the identification and designation of marine protected areas (MPAs),[80] ecologically and

[74]Malawi Principle 7. CBD Guidelines (2004, pp. 20–21).

[75]In ABNJ the CBD only applies to processes and activities carried out under the jurisdiction and control of the Parties. CBD Art. 4 (b).

[76]These include regional seas agreements such as the Barcelona Convention, OSPAR, the Noumea Convention, CAMLR and the Antarctic Treaty, as well as RFMOs, CMS and the International Whaling Convention 1946.

[77]See e.g. Report of the Joint FAO/IMO Ad Hoc Working Group on Illegal, Unreported and Unregulated (IUU) Fishing and Related Matters, Document FIRO/R1124 (July 2007). A cooperation agreement between the IMO and FAO was entered into in 1965. See further J Harrison (2017, p. 279).

[78]See e.g. 1996 CITES-CBD MOU, 1996 CBD-CMS MOU and 2002 CITES-CMS MOU. J Harrison (2017, p. 278), Tanaka (2004, pp. 505–506). On the challenges of institutional linkage in a biodiversity context, see E Raitanen (2017, pp. 91–92).

[79]Memorandum of Understanding Between the North East Atlantic Fisheries Commission and the OSPAR Commission, 2008.

[80]E.g. the Parties to the Antarctic Treaty can only designate protected areas in consultation with the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) as the relevant RFMO in the region and vice versa. In the Mediterranean, cooperation between a regional seas body and a regional fisheries body is illustrated via the Memorandum of Understanding between the UNEP MAP-Barcelona Convention and FAO-GFCM (2012), Annex which includes collaboration on criteria to identify MPAs. See further J Harrison (2017, pp. 281–286).

biologically significant areas (EBSAs),[81] and large marine ecosystems (LMEs).[82] However, most examples of inter-sectoral and institutional cooperation tend to occur on an ad hoc basis without overarching coordination. These shortcomings have been recognised within the BBNJ process, and there is agreement on the need to address cooperation and collaboration among different institutions (Harrison 2017), however no clear consensus has yet emerged regarding modalities to achieve this. It also remains to be determined whether there will be a Conference of Parties with global authority as part of the new instrument, however it looks increasingly likely.[83]

5 Conclusion

Despite the challenges associated with the operation of the ecosystem approach, it has increasingly become a staple feature of modern marine management. However, given that most of the work done to flesh out how it can be implemented and applied has occurred on a soft law basis, the normative content of the ecosystem approach has been described as weak and unclear in terms of its obligations on States (Tanaka 2015). It is clear that a more holistic form of governance is a necessary corollary of the ecosystem approach, which will naturally require greater cooperation between States and international and regional institutions, integrated management across sectors, and planning on a variety of levels, including across boundaries (IPBES 2019; UNGA 2006).[84] Integrated management, with a long-term time frame (CBD 2004), is considered to be essential in order to ensure efficient coordination between organizations and compatibility between policies and activities.[85] However, its implementation has been hampered by the existing fragmented and decentralised institutional architecture of global ocean governance (Harrison 2017), as well as political and financial challenges (Scott 2015). Its meaning also remains obscure in

[81]The EBSA process, established under the CBD, has potential to play a useful role in facilitating cooperation in relation to the establishment of MPAs. It is not constrained by boundaries and works via regional workshops involving diverse stakeholder groups representing regional jurisdictions, intergovernmental bodies, non-governmental organizations and indigenous representatives. To date 279 EBSAs have been recognized, encompassing areas of the ocean both within and beyond national jurisdictions. See further DE Johnson et al. (2018).

[82]The LME concept was developed by the United States National Oceanic and Atmospheric Administration (NOAA) as a model to implement ecosystem approaches to assessing, managing, recovering, and sustaining LME resources and environments. Thus far, 64 LMEs have been defined globally. See further https://www.st.nmfs.noaa.gov/ecosystems/lme/; UNEP (2016), H Wang (2004), L Juda (1999). For critique, see J Rochette et al. (2015).

[83]See IISD *Summary of the Third Session of the Intergovernmental Conference on the Conservation and Sustainable Use of Marine Biodiversity of Areas Beyond National Jurisdiction: 19–30 August 2019* Earth Negotiations Bulletin Vol. 25 No. 218, available at http://enb.iisd.org/oceans/bbnj/igc3/

[84]UNGA (2006), para 7.

[85]Agenda 21, Chapter 17, para. 17.5(a). For a deeper discussion on integrated oceans management, see generally K Scott (2015) and J Harrison (2017), Chapter 10.

international law (Scott 2015; Tanaka 2004). Parties to the CBD have acknowledged that the full application of the ecosystem approach remains a 'formidable task', especially on a larger scale.[86] Nevertheless, the soft law developed by CBD parties, including the Malawi Principles and Operational Guidance, continue to remain relevant and applicable. Indeed, Morgera attributes the transformation of the ecosystem approach into a "fully-fledged system of soft law principles and guidelines" to this consensus based normative activity of the CBD parties (2017, p. 71). The BBNJ process represents a timely opportunity for States to tackle many of the challenges discussed in this chapter. While negotiations remain ongoing as of 2019, the design of the instrument and mode by which it provides or creates space for enabling elements (e.g. institutions, guidelines) will have a significant bearing on how the ecosystem approach is translated into practice in the future.

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[86] CBD-COP Decision IX/7 (2008), Preamble, para (f).

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Ecosystem-Based Management (EBM) and Ecosystem Services in EU Law, Policy and Governance

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Abstract Ecosystem-Based Management has become the dominant desired paradigm for environmental management globally yet what it entails and how it can be implemented presents many challenges. This chapter seeks to set out the legal bases for Ecosystem-Based Management (EBM) and Ecosystem Services in EU law and policy frameworks. It traces both concepts with a view to establishing how their legal status internationally and regionally has influenced their uptake within national governance frameworks. The rationale for EBM is to manage resources in a way that maintains the health of the ecosystem alongside appropriate human use of the marine environment, for the benefit of current and future generations, and accordingly is intrinsic to achieving sustainable development. EBM therefore represents two keys challenges for existing governance systems: firstly, the need to move away from sectoral based management and towards more integrated approaches and secondly, to embed the notion of 'healthy' ecosystems into all law and policy instruments. The chapter begins with the international and regional levels of governance and how both approaches have been incorporated. It then proceeds to a brief overview of the EU legal system, examines the key elements of biodiversity policy and conclude with a discussion and conclusions on whether these have enabled the implementation of EBM and ecosystem services.

Lessons Learned

- There is no definition of EBM or ecosystem services in EU law.
- EBM necessitates a move away from traditional sectoral focussed management as policy-makers need to manage for multiple ecosystem services that cannot be achieved if a single sectoral or policy 'lens' is taken.
- Regional Conventions have been instrumental in developing understanding of EBM and ES but can be limited by their individual remits.

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- In the EU, EBM is partially implemented and in a top-down manner giving Member States significant discretion and often resulting in extensive differences between countries.
- Many existing EU policies in their current form contradict the commitment to implementing EBM and preserving ecosystem services, such as measures and activities under Common Agricultural Policy and Common Fisheries Policy.

Needs to Advance EBM

- There is a need for clear definition and agreement on what EBM is and how it can be implemented in practice at EU and national levels.
- EBM also necessitates adaptive management in order to deal with dynamic ecosystems and the absence of complete knowledge or understanding of their functioning.
- There is a critical need for clarity on what EBM actually requires, on how implementation progress can be measured and sharing of successful 'better' EBM practices.

1 Introduction to Ecosystem-Based Management (EBM) and Ecosystem Services (ES) in International Law and Policy

Somewhat surprisingly, there is no universally accepted definition of Ecosystem-Based Management (EBM) or the ecosystem approach in international or EU law, yet this has not limited implementation efforts to date. Aspects of the ecosystem approach can be traced back to the Stockholm Declaration on the Human Environment in 1972, which called for cooperation on conservation, protection and restoration of the Earth's ecosystem (United Nations 1972). The two key international legal instruments of relevance to EBM and ES are the UN Convention on Biological Diversity and the UN Convention on the Law of the Sea.

1.1 UN Convention on Biological Diversity (CBD)

The Ecosystem Approach was adopted as the primary framework for the conservation and sustainable use of biodiversity under the Convention on Biological Diversity (CBD) in 1995 hence it is from the CBD that the most widely used definition is derived. The fifth Conference of the Parties describes the Ecosystem Approach as "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way" (CBD 2000). It went further to state that it "requires adaptive management to deal with the complex and dynamic nature of ecosystems and the absence of complete knowledge or understanding of their functioning". Implementation of the Ecosystem Approach is facilitated through 12 interlinked principles, known as the Malawi Principles (see Enright and Boetler 2020). Whilst the term 'ecosystem services' was not in use when the CBD was endorsed, the concept is implicit in the Convention text. Principle 5 goes some way towards explaining how the Ecosystem Approach and ecosystem services are connected: "conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach", with a focus on ecosystem functioning.

The Malawi Principles provide the context for further action and, at the tenth Conference of the Parties, a global Strategic Plan for Biodiversity 2011–2020 was adopted. The plan's mission is "to take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient *and continue to provide essential services*, thereby securing the planet's variety of life, and contributing to human well-being, and poverty eradication". This means that the CBD provides the global framework for regional and national action to protect biodiversity but also the services that biodiversity provides. The Strategic Plan includes the Aichi Biodiversity Targets on the management of protected areas and the conservation of all ecosystems through the application of the precautionary approach and the Ecosystem Approach. The Aichi Biodiversity Targets consist of five Strategic Goals each of which are accompanied by specific targets. The goal on improving the status of biodiversity by safeguarding ecosystems, species and genetic diversity, for example, is the basis for Target 11 on conservation designations: "by 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes" (CBD 2010). Ecosystem services are also addressed directly by Strategic Goal D: "Enhance the benefits to all from biodiversity and ecosystem services" which includes Target 14: "By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable."

1.2 United Nations Convention on the Law of the Sea (UNCLOS)

The Law of the Sea Convention defines the rights and responsibilities of nations with respect to their use of the world's oceans, protection of the marine environment, and the management of marine natural resources. There is no mention of the Ecosystem Approach in the Convention though debatably, the approach is included implicitly as it contains an absolute obligation to protect and preserve the marine environment (Article 192) and to adopt measures to protect and preserve rare or fragile ecosystems as well as the habitat of depleted, threatened or endangered species and other forms of marine life (Article 194(5)). Both these provisions, however, require further implementation mechanisms at national level in order to be achieved. The 1995 Fish Stocks Agreement[1] establishes an obligation for States to protect marine biodiversity via the protection of target and non-target species as well as the ecosystems associated with those species. The Agreement also includes a duty to cooperate in Regional Fisheries Management Organisations (RFMOs) or similar arrangements in an effort to recognise the transboundary nature of aquatic biodiversity and need for concerted coordinated action. Alas this remains a key challenge as the existing RFMOs do not cover all ocean space and organise their work according to geographic area and specific fish species meaning that many areas and species remain ineffectively managed. The links between the CBD and UNCLOS are recognised under CBD's Article 22(2) which provides that Parties will implement the CBD with respect to the marine environment "consistently with the rights and obligations of States under the Law of the Sea".[2]

1.3 OSPAR Convention

At the regional level, application of the Ecosystem Approach can be found in many of the Regional Seas Conventions and their associated mechanisms. The OSPAR Convention, for example, provides a framework for the regulation of almost all human activities[3] that have an adverse effect on marine ecosystems and biodiversity in the North-East Atlantic. The Helsinki and OSPAR Commissions adopted a statement on how the EA could be implemented under their respective legal instruments. This states that the Ecosystem Approach is "the comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity" (Helsinki and OSPAR Commissions 2003). The OSPAR Commission's work is guided by the Ecosystem Approach and it is implemented in the North-East Atlantic by means of the programmes and measures developed under OSPAR's six

[1]Agreement for the Implementation of the Provisions of the UN Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, 4 August 1995 (into force 11 December 2001) 2167 UNTS 3.

[2]An exception to this is provided in Article 22(1) whereby "CBD provisions shall not affect the rights and obligations of any Party deriving from any international agreement except where the exercise of those rights and obligations would cause a serious damage or threat to biological diversity".

[3]Except fisheries and pollution from ships.

thematic Strategies.[4] The International Council for the Exploration of the Seas (ICES), working with OSPAR and HELCOM, has been instrumental in developing a better understanding of the EA and providing guidance and recommendations on how it can be implemented (e.g. ICES 2005). This includes the identification of practical steps in applying the approach by those tasked with implementing marine policy in the EU and informed the design of the EU's Marine Strategy Framework Directive. Unfortunately, by design, the Regional Seas Convention bodies are limited in the actions they can take in implementing the EA due their pre-defined geographical area and remit for actions.

Essentially the rationale for EBM is that whilst the ecosystem itself may not be managed, the human uses and activities that interact and impact upon the ecosystem may be managed so as to conserve biodiversity and ensure sustainable development (Long 2012). Ultimately the aim is to preserve ecosystem structure and functioning so as to ensure the ongoing provision of products and services. Therefore, management of the impacts of human activities must focus on the entire ecological system and not its component parts. This necessitates a move away from traditional sectoral management approaches towards those that are integrated, adaptive and coherent across policy domains so as to take account of social, economic and environmental aspects. EBM recognises that new forms of valuation and assessment are needed, and that different sectors of society will view ecosystems from their own environmental, economic and societal needs. The role of ecosystem services therefore is to provide information on the values and services that flow from ecosystems to humans. Internationally, the Millennium Assessment (MA) and The Economics of Ecosystems and Biodiversity (TEEB) initiatives have sought to capture information on the value of and benefits from ecosystems in a format that can be used by policy and decision-makers.

2 EBM and ES in EU Law and Policy

Whilst EBM is not expressly mentioned in the European treaties, under Article 11 of the Treaty on the Functioning of the EU, there is a duty to integrate environmental protection into the definition and implementation of EU policies "in particular with a view to promoting sustainable development". Article 191 TFEU states that EU environmental policy should promote measures at international level to deal with regional or worldwide environmental problems, which could be interpreted to include EBM. Essentially EBM (or the Ecosystem Approach) is introduced and implemented in the EU in a top-down fashion through a wide range of Directives and policy documents, resulting in much national autonomy in terms of implementation. The EU Biodiversity Baseline found that only 17% of habitats and species and 11%

[4] Biodiversity, eutrophication, radioactive substances, hazardous substances, offshore industry and assessment/monitoring.

of key ecosystems protected under EU legislation were classified as being in a favourable state (EEA 2010). In an attempt to address reverse this loss and assist in becoming a more resource efficient and green economy, the EU adopted a biodiversity strategy in 2011.

2.1 Biodiversity Law and Policy

The **EU Biodiversity Strategy 2020** aims to implement the CBD's Strategic Plan for Biodiversity 2011–2020 and the Aichi Targets (European Commission 2011). It recognises the role that biodiversity plays in underpinning the economy and the services it provides. The strategy sets out six targets to achieve the over-arching 2020 target of "halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss". The targets relate to nature (target 1), ecosystems and their restoration (target 2), the sustainable use of Europe's nature, land and sea resources via agriculture, forestry and fisheries (targets 3 and 4), alien species (target 5) and the EU's global impacts (target 6). A mid-term review of the Biodiversity Strategy was conducted in 2015 and found that biodiversity loss and degradation of ecosystem services in the EU has continued and, whilst some progress has been made at the policy level, this has not yet halted the trend of degradation of ecosystems and services (European Commission 2015a). The review concludes by stating that the 2020 biodiversity targets will only be achieved if implementation and enforcement efforts become "considerably bolder and more ambitious." An associated Fitness Check of the Birds and Habitats Directive, as the cornerstones of biodiversity policy in the EU, evaluated these instruments in terms of their effectiveness, efficiency, relevance, coherence and EU added value (European Commission 2016a). In terms of effectiveness, the evaluation found that the general objectives of the Directives have not yet been met and that it was not possible to predict when the objectives would be fully achieved but acknowledged that improvements in the status of species and habitats occur where there are targeted actions at a sufficient scale.

The Biodiversity Strategy is complemented by a wide range of legal instruments including the **Birds and Habitats Directives**, the Water Framework Directive, the Marine Strategy Framework Directive and the EU Regulation on Invasive Alien Species (No. 1143/2014). These four Directives, and one Regulation, provide the legal basis to protect aquatic biodiversity across the freshwater—marine continuum and hence seek to enable implementation of EBM. Bastmeijer (2019) states that the Birds and Habitats Directive are not based on the ecosystem approach and Rouillard et al. (2018) state neither Directive explicitly mentions ecosystem services or takes them into account implicitly. Despite this they are essential in protecting certain types of biodiversity. The Birds Directive provides for the designation of sites for the protection of bird species listed in Annex I, along with the designation of sites designated for "regularly occurring migratory species not included in Annex I" under Article 3(2). This is complemented by the Habitats Directive which, under Article 3(1), requires Member States to select and designate "sites hosting the natural habitat types listed in Annex I and habitats of the species listed in Annex II". Together the protected sites designated under each Directive form the Natura 2000 network, the largest global network of protected areas, which seeks to achieve the objective of maintaining or restoring natural habitats and species of Community interest at Favourable Conservation Status. The Appropriate Assessment procedure under Article 6, determines whether a plan or project can be implemented without damaging a Natura 2000 site, through an examination of the implications of a proposed development for the Natura 2000 site and its conservation objectives. Though this can add to regulatory and consenting requirements, the majority of plans and programmes subjected to this assessment are permitted to proceed (European Commission 2016a).

The **Invasive Alien Species Regulation** (IAS, No. 1143/2014) is mentioned here as it is the only EU legal instrument to explicitly contain a definition of ecosystems services as "the direct and indirect contributions of ecosystems to human wellbeing" in Article 3(6), perhaps attributable to its relatively recent entry into force (2015) when compared to other biodiversity instruments. It consists of three types of measures: prevention; detection and eradication; and management measures, based on the list of IAS of Union Concern contained within the Regulation. This list is updated regularly commencing with a proposal from a Member State or the EC, supported by a risk assessment; followed by an expert evaluation of the available evidence and inputs from a range of stakeholders and the Member States. This proposal must then be approved by a Committee comprised of Member States' representatives before final adoption by the Commission. There are no timelines associated with this process. Bouwma et al. (2018) state the IAS Regulation requires that its effectiveness on biodiversity, ecosystem services and, human health and the economy, if applicable, is monitored though the development of concrete measures and comprehensive action plans to prevent the "unintentional introduction and spread of invasive alien species" is left up to Member States. There is no dedicated funding instrument associated with the IAS though actions have been supported by the Commission through Horizon 2020, LIFE+, the rural development programme and other funding programmes such as Interreg.[5]

The European Commission (2007) has stated that the implementation of the Natura 2000 network forms one of the legal components to implementing the ecosystem approach in the marine environment. The mid-term review of the Biodiversity Strategy, states that the Natura 2000 network now covers approximately 18% of land but marine coverage is much lower at 6%, well below the 10% Aichi target. This could be attributed to the fact that there are much less marine species and

[5]The Interreg IVA programme, for example, funded the INVEXO project to support joint management efforts on four priority invasive alien species in Flanders and southern part of the Netherlands.

habitats listed in the Annexes of both Directives.[6] Marine species are also more difficult to protect in the same manner as terrestrial species as the marine species concerned may have a wider geographic range, perhaps taking in multiple jurisdictions. The EEA (2015a) concluded that only 21% of the habitat assessments and 23% of the non-bird species assessments were at favourable conservation status and 52% of the bird species assessed were secure, based on the reporting required under both Directives between 2007 and 2012. The same report (p. 8) states that the most frequently cited pressures and threats for marine ecosystems are fishing, modification of natural conditions and pollution. This brings into focus the need for greater interplay between the nature conservation Directives and other thematic legislation, not only to better protect the environment, but to implement the ecosystem approach. This is echoed by the EC's mid-term review of the EU's Biodiversity Strategy (2015a) which notes that "a lot remains to be done to halt the loss of ordinary biodiversity outside the Natura 2000 network." A key consideration in this context is the costs associated with implementation. The Fitness Check (EC 2016a, p. 5) states that as Member States do not have to report on the costs and benefits of the nature conservation legislation, there is limited quantitative information available at the EU scale to underpin assessments on efficiency and that "compliance costs of designating, protecting and managing Natura 2000 sites have been estimated to be at least € 5.8 billion annually across the EU."

2.2 Water Law and Policy

Alongside the nature conservation Directives, the **Water Framework Directive (WFD)** seeks to prevent the deterioration of freshwater ecosystems and restore their good ecological status. The Marine Strategy Framework Directive (MSFD), adopted slightly later in 2008, aims to achieve good environmental status of EU marine waters by 2020. The Directives therefore provide an ecological continuum from a river basin to the sea and, by taking a cyclical approach to implementation, should enable an adaptive management approach, one of the fundamental principles of EBM. The WFD was the first EU legal instrument to adopt a holistic approach to aquatic regulation via a move away from management on the basis of administrative boundaries. Evaluation of water quality under Annex V of the WFD requires consideration of the quality of the structure and functioning of aquatic ecosystems (surface waters), the physical-chemical nature of the water and sediment, the flow characteristics of the water, and the physical structure of water bodies. Article 4 of the Directive contains exemptions from the ecological goals, either by way of *force majeure*, reasons of overriding public interest or lack of ability of the Member State to achieve the goals due to external factors (e.g. impacts from other States). Whilst

[6]There are nine marine habitat types and 16 species listed in the Habitats Directive, and 60 bird species listed in the Birds Directive.

elements of the Directive are based on physical, chemical and biological parameters, the River Basin Management Plans, required under the Directive must also consider the socio-economic environment of the region and all activities that might impact on the status of a water body. A fitness check of the WFD (EC 2012) recommended that greater consideration be given to ecosystem services in the Directive, its Common Implementation Strategy and other policies "so that they can be better reflected in the implementation on the ground."

The **Marine Strategy Framework Directive (MSFD)** primarily aims to prevent any further deterioration of the marine environment, recognising it is the basis of the blue economy or, in other words, recognising the ecosystem services the marine environment provides. The Directive tries to better integrate the concepts of environmental protection and sustainable use. The MSFD was also viewed by the European Commission (2005) as a way of addressing sectoral fragmentation in marine governance and attaining the international obligations the EU had in terms of biodiversity under the CBD and Regional Seas Conventions to which the EU is a party. The MSFD requires in Article 1(1) that EU Member States to "take the necessary measures to achieve or maintain good environmental status in the marine environment by the year 2020 at the latest." According to Article 3(5), this objective is to be achieved by applying "adaptive management on the basis of the ecosystem approach." It is clear, from the 11 descriptors used in the Directive, that the concept of "good environmental status" includes the conservation of biodiversity and the maintenance of ecosystem health and integrity. The descriptors have since been supplemented in an European Commission Decision (2010) which subdivided them into 29 criteria and 56 associated 'indicators' so as to determine more precisely what attributes of the ecosystem features should be considered when assessing environmental status, ultimately with the aim of achieving more uniform assessment. Ecosystem services are only indirectly included in the descriptors. Berg et al. (2015) explain that Descriptor 3, on commercially exploited fish and shellfish, defines criteria on fish demographics aiming to ensure that fish populations are able to be caught (ecosystem service: food provision) and still be viable and productive. They conclude that although the MSFD includes sustainable use of the marine environment as part of the definition of GES, the Decision does not include criteria targeting ecosystem services that can be used to inform on the aspect of sustainable use. Like under the WFD, Member States are required to produce Marine Strategies for marine areas under their sovereignty and jurisdiction which must contain a comprehensive assessment of the state of the marine environment, a definition of "GES" at the regional level, clear environmental targets and monitoring programmes.

Subsequent to the initial assessment, setting of targets and monitoring plan, Member States must design a programme of measures (POM) to deliver the targets, each measure being supported by a cost-benefit analysis. In designing their POM, Member States are obliged to consult competent authorities in the field of water and nature conservation policy but the involvement of other sectoral authorities is at the discretion of the Member State. Perhaps in recognition of the need to take a broader perspective to implement the Ecosystem Approach, the MSFD advocates that operational and implementation measures are adopted through the Regional Seas Conventions. The Programmes of Measures "shall include spatial protection measures, contributing to coherent and representative networks of marine protected areas (MPAs) adequately covering the diversity of the constituent ecosystems" such as SACs, SPAs and other forms of MPAs under the RSCs or other international agreements. Sites in the Natura 2000 network, with marine qualifying interests, are the single largest contributor to European MPAs in terms of coverage, though it is acknowledged that geographic coverage lessens further offshore: Natura 2000 sites covered 33.3% of nearshore waters, 11.3% of coastal waters and only 1.7% of offshore waters (EEA 2015b). The EEA differentiates between three types of MPAs in the EU: marine Natura 2000 sites, MPAs designated under Regional Sea Conventions, and individual national MPAs. The Commission (2015b) has stated that in order for MPAs to fully deliver their potential, they must include management measures and require effective monitoring and enforcement. MPAs will be an integral part of Maritime Spatial Plans, which are legally required by 2021. As the MSFD is intended to make marine regulation and decision-making more integrated in form and content it will influence both cross-cutting management approaches such as Maritime Spatial Planning and Integrated Coastal Management, as well as sectoral policies, such as the Common Fisheries Policy.

2.3 Sectoral Law and Policy

2.3.1 Common Fisheries Policy (CFP)

As outlined above, to achieve the over-arching objectives of the MSFD, particular marine activities will have to reduce their environmental impacts or plan to mitigate these. One example of this is the requirement to consider the effects of the Common Fisheries Policy (CFP) on GES.[7] Article 2 of the Basic Fishery Management Regulation (2371/2002) states that one aim of the CFP is "to minimise the impact of fishing activities on marine ecosystems and to ensure the progressive implementation of an ecosystem-based approach to fisheries management." The European Parliament and Council's 7th Environmental Action Programme (2013), in relation to the exploitation of marine resources, recognises that "care needs to be taken to ensure their exploitation is compatible with the conservation and sustainable management of marine and coastal ecosystems." For GES, Descriptor 3 requires that populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock. It is left up to Member States to decide how to achieve this. Fisheries policy with regard to resource exploitation is an exclusive competence of the EU. The MSFD sets out objectives to be achieved but is unable to subject the CFP to its terms.

[7] Recital 40, MSFD and Article 2(5)(j) of Regulation 1380/2013.

As such, the MSFD merely notes in Recital 39 that "measures regulating fisheries management can be taken in the context of the Common Fisheries Policy" . . . "with a view to supporting the achievement of the objectives addressed by this Directive". A significant weakness however is that there is no requirement for the CFP to conform to or harmonise with the environmental objectives of the MSFD, and no specific measures to ensure coherence, thereby limiting the potential for implementation of an Ecosystem Approach and consideration of ecosystem services in fisheries management.

Article 8 of the revised Fisheries Regulation (No.1380/2013) provides for the creation of fish stock recovery areas, "due to their biological sensitivity, including areas where there is clear evidence of heavy concentrations of fish below minimum conservation reference size and of spawning grounds".[8] In these areas fishing activities may be restricted or prohibited in order to contribute to the conservation of living aquatic resources and marine ecosystems.[9] These areas could also contribute to a coherent network of protected areas, as envisaged under the Biodiversity Strategy and CBD. Article 11(1) of the Fisheries Regulation (No.1380/2013) provides that Member States can adopt conservation measures within waters under their jurisdiction in order to comply with their obligations under the MSFD and other EU environmental law provided that such measures do not affect the fishing vessels of other Member States and are compatible with the objectives of the CFP. Where there is a serious threat to the conservation of marine biological resources, or to the marine ecosystem, based on evidence, Article 12 (No.1380/2013) provides that the Commission may adopt restrictive measures for environmental protection that apply to all vessels but, by definition, these are time bound in that they cannot apply for longer than six months and may only be renewed once. This 'supremacy' of fishing can best be explained by its prominence in the EU Treaties, under the agriculture provisions, and in practice means that all Member States would have to agree to any measures likely to impact on or modify fishing activity. The revised governance structure in the 2013 Reform Package aims to address this somewhat by giving Member States more of a role in customising regional conservation measures, specifically on recommendations for achieving the objectives of conservation measures, provided the Commission is of the opinion such recommendations are compatible with the relevant conservation measure and/or applicable multiannual plan.[10]

[8]Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC. Official Journal of the European Union, L 354, pp. 22–61.

[9]Article 8(1), Fisheries Regulation (No.1380/2013).

[10]Article 18(3), Fisheries Regulation (No.1380/2013).

2.3.2 Common Agricultural Policy (CAP)

The Common Fisheries Policy is largely modelled on the Common Agricultural Policy, one of the common policy areas of the EU, in operation since 1962. As a common policy area the aim is to ensure there is a level playing field and fair competition between farmers. The objectives of CAP are to improve agricultural productivity, ensure a fair standard of living for those involved in farming, stabilise markets, assure the availability of supplies and ensure that produce reaches consumers at reasonable prices.[11] Reforms to the CAP have expanded it to encourage farmers to provide public goods, enhance biodiversity and help address climate change but not in the official constitutionalised version from the late 1950s. The CAP is financed centrally by the EC through two funds: the European Agricultural Guarantee Fund (EAGF), which provides direct support and funds market measures; and the European Agricultural Fund for Rural Development (EAFRD) which finances rural development programmes. The budget for CAP is €362.8 billion, almost 40% of the total EU budget, for the period 2014–2020, of which €277.9 billion is foreseen for Direct Payments and market-related expenditure (Pillar 1) and €84.9 billion is for Rural Development (Pillar 2) (European Commission 2013a). Agricultural nutrient sources are a major contributor to the status of water quality across the EU and needs to be considered as part of an ecosystem approach to aquatic management. The Nitrates Directive, together with the WFD, requires Member States to monitor water quality and, in particular, to identify areas that are polluted or at risk of pollution due to agricultural activities. These areas are known as "Nitrate Vulnerable Zones" and Member States are required to create Nitrate Action Programmes in order to reduce and prevent water pollution. Measures under these programmes include limits in when fertilizers can be applied, requirements for storage of manure, conditions for fertilizer applications, and limits on the amounts of fertilizer that can be used.

The Nitrates Directive does not refer to the Ecosystem Approach or ecosystem services directly though the definition of "pollution" in Article 2(j) acknowledges that pollution can cause "... harm to living resources and to aquatic ecosystems, damage to amenities or interference with other legitimate uses of water", thereby implicitly recognising ecosystem services. Reform of the CAP in 2013 resulted in a new policy instrument of the first pillar (greening) and covers the provision of environmental public goods. The Green Direct Payment equates to 30% of the national direct payment envelope and recompenses farmers for maintaining permanent grassland, creating ecological focus areas and diversifying crops. A portion of the rural development programme budget (30%) must also be used for measures that are beneficial for the environment and climate change, such as agri–environmental-climate measures, organic farming, Areas of Natural Constraints (ANC), Natura 2000 areas and forestry measures. Whilst well-intentioned, these measures have been largely unsuccessful, in terms of biodiversity conservation (Pe'er et al. 2017),

[11]Article 39(1) Article 39 TFEU (ex Article 33 TEC).

mitigation of climate change (European Commission 2019), and wider public opinion on whether CAP actually does enough to address environmental degradation and climate change (European Commission 2016b). Pe'er et al. (2019) found that highest investments are made into the least effective greening (€789.9/ha), compared to a third as many payments for the more effective agri-environment climate measures (€247.2/ha) and direct payments continue to be 'coupled' to the production of certain crops and livestock including input-intensive systems such as beef fattening and vegetable production which undermines overall sustainability goals. To achieve these, significant reforms are required not only to the substantive provisions of CAP but also its overall governance and integration with other key policy areas.

2.4 *Cross-cutting Management Approaches*

Integrated marine governance has been a focus area of Commission work since the early 1990s in terms of Integrated Coastal Zone Management (ICZM) and more recently Maritime Spatial Planning (MSP). MSP is seen as essential to delivering the jobs potential provided by maritime sectors, protection of the marine environment and optimisation of the use of marine space. ICZM has a policy basis in the associated Recommendation dating from 2002.[12] This recommends that Member States protect their coastal environment based on an ecosystem approach "preserving its integrity and functioning". Recognising the need for more action on integrated management approaches, the EC proposed a draft Directive, aimed at creating a framework for both Integrated Coastal Management (ICM) and Maritime Spatial Planning, with a view to improving planning and management of the land-sea interface (European Commission 2013b). During the negotiation phase, however, ICM was dropped from its contents. No official explanation is available but significant concerns were expressed by the Committee of the Regions (2013), for example, as it was perceived that ICM impinged substantially on existing Member State competences relating to spatial planning policy and practice at regional and/or local levels. The MSP Directive (2014/89/EU) was adopted in July 2014 and necessitates Member States to establish their first Maritime Spatial Plans by 31 March 2021. Article 5(1) requires Member States to apply an ecosystem-based approach when establishing and implementing MSP. The substantive provisions of the Directive say nothing further on the ecosystem-based approach though Recital 14 could be said to add some clarity saying the aim of applying an ecosystem approach is to ensure that the collective pressure of all activities is kept within levels compatible with the achievement of GES under MSFD and "that the capacity of marine ecosystems to respond to human-induced changes is not compromised, while

[12]Recommendation of the European Parliament and of the Council of 30 May 2002 concerning the implementation of Integrated Coastal Zone Management in Europe. OJ L148, 6 June 2002, pp.24–27. http://data.europa.eu/eli/reco/2002/413/oj

contributing to the sustainable use of marine goods and services by present and future generations". The MSFD identifies MSP as a tool to support the ecosystem-based management of human activities in order to achieve GES, which is a point reiterated in Recital 22 (Preamble) of the MSP Directive.

The principles and procedures involved in implemented Integrated Coastal Management, Maritime Spatial Planning and the Ecosystem Approach are broadly similar and each have the ultimate aim of delivering sustainable development (see Le Tissier 2020). All advocate the need for more holistic, integrated and adaptive management moving away from sectoral approaches, increased stakeholder participation, better use of scientific data and knowledge and integrated monitoring to adapt management actions. The challenges arising are that all approaches can be applied at different scales, however, the ecosystem approach is more suited to the ecosystem scale whereas ICM and MSP will most likely to tailored to the scale of the management problems they seek to address which may not necessarily correspond to the ecosystem level. Whilst the principles are similar the approaches differ and accordingly there is a need for greater policy coherence that takes into account a wider range of economic, environmental and social aspects. The policies adopted need to secure the delivery of ecosystem and harmonise management and conservation objectives, both in the immediate and longer term. ICM is not a binding legal requirement on all EU Member States although the 2002 ICZM Recommendation remains valid. MSP is only in the initial stages of implementation in the majority of coastal Member States hence how the Ecosystem Approach and ecosystem services will be reflected in final plans remains to be seen. It is probable that the marine strategies developed by Member States to meet the requirements of the MSFD are more likely to be ecosystem-based than either MSP or ICM given their focus but this goes to reaffirm the need for policy coherence if the ecosystem approach is to be implemented across the entire aquatic area.

3 Discussion

The preceding sections demonstrate that the Ecosystem Approach and ecosystem services have been incorporated into EU law and policy to varying extents. At the international level this has also continued to progress but the variety of definitions and methodologies for implementation of the concepts at the global level has very real implications for other governance scales, including regional, national and local implementation which are more utilised for aquatic management under EU law. It has been long recognised (e.g. Scheiber 1997) that there is a critical need for clarity on what EBM actually requires and how progress on implementation can be measured. The commitment to implementing EBM at EU level requires more joint implementation of legal instruments to take account of the necessary move away from management according to administrative boundaries. This is central to the WFD and echoed in the MSFD yet despite having a Common Implementation Strategy there are significant disconnects between both instruments. For example, marine litter is one of the descriptors used for determining GES under MSFD but there is no equivalent in the WFD, even though the majority of marine litter derives from the land as scientific evidence confirms. From a governance perspective there is a need for institutional structures that facilitate cross-sectoral management and decision-making. Competences for aquatic management are split according to the land and sea, jurisdictional boundaries and also sectorally which goes against the principles of ecosystem-based management. The same is true for monitoring, enforcement and compliance actions.

Many existing EU policies in their current form contradict the commitment to implementing EBM and preserving ecosystem services, such as measures and activities under CAP and CFP. The Fitness Check (EC 2016a, p. 91) of the nature conservation legislation found that EU financial support applies to agriculture and forestry, the main land uses in Natura 2000 as well as to prevent damage caused by protected species (e.g. under rural development for large carnivores) or to compensate for such damages (e.g. under fisheries policy for fish-eating birds). Figure 1 illustrates the levels of agricultural, fisheries and conservation across the EU. The main beneficiaries of financial support in the EU remain the economic sectors of fishing and agriculture. The overall EU co-funding for Natura 2000 during the 2007–2013 period represented only 9–19% of the estimated financing needs and national co-funding was unable to cover the remaining gap. This could lead one to question the commitment to halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020. Alternatively, it could signal the need to better consider trade-offs in management processes. There is no ‘recipe’ for balancing conservation and development but it is essential that we understand who will benefit and who will lose out if ecosystem services change. This necessitates better understanding of ecosystem services. From a societal perspective, greater stakeholder participation is required so as to understand their interests and expectations. In theory, in this way more suitable interventions could be made to deliver sustainable development.

4 Conclusions

The European Commission has endorsed the United Nations’ Sustainable Development Goals (SDGs). Effective implementation of the SDGs will require fully coordinated policies that take into account the multiple relationships that exist between the different dimensions of sustainability, something that cannot be achieved currently given the preponderance of sectoral legislation, policies and associated institutional structures. The Marine Strategy Framework Directive has the ecosystem approach at its core but implementation of its objectives is largely left up to Member States themselves, whilst other legislation and policies refer to EA but it is not the overarching objective or priority for implementation. There will always be trade-offs between conservation and development and no form of decision-making will make this disappear. Attaining societal agreement on long-term goals

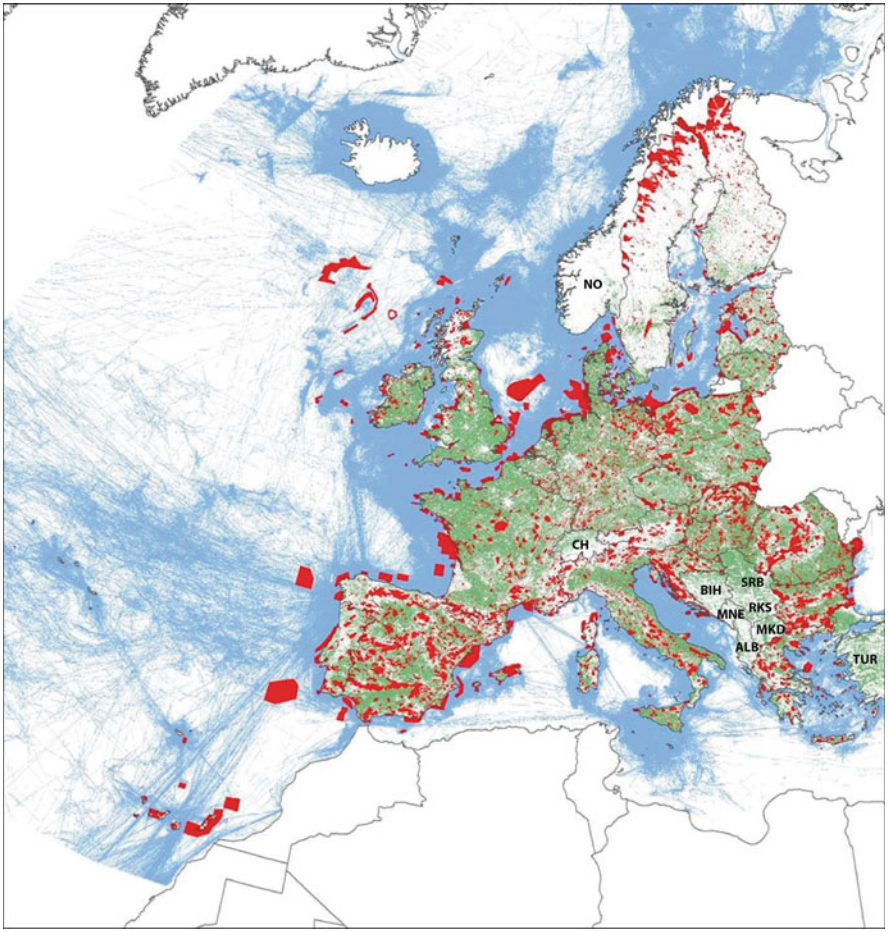

Fig. 1 Fishing vessel activity (blue), agriculture (green) and nature conservation (red) across the EU. Countries marked with ISO country codes are non-EU member states and do not participate in the EU Natura 2000 network, but have habitat data available under the Corine Land Cover inventory. Cartography by Tim O'Higgins

requires appropriate institutional mechanisms and these also need to acknowledge that societal goals change over time. Decision-making structures are still dominated by sectoral interests. This is clearly evident in the case of both the CFP and CAP. Reform of the CFP, for example, resulted in the creation of Advisory Councils as stakeholder bodies to address issues associated with participatory democracy (European Commission 2009). No measures affecting fishing activity can be adopted without reference to these Advisory Councils. Article 45(1) and Annex III (2) of Regulation 1380/2013 provides that 60% of the seats on the Advisory Councils are allocated to organisations representing the fisheries, processing and marketing sectors, with the remainder allocated to other interest groups. Wakefield (2019) states that the European Parliament had called for half the seats on the Advisory Councils to be reserved for interested parties outside the fishing industry but this was not accepted by the Member States in Council.

Like EBM, the conceptual basis for ecosystems services has been well expressed but implementation remains variable, perhaps explained by lack of agreement on definitions and uncertainties around the links between biodiversity and the services that flow from ecosystems. Each different ecosystem service also has a different legal status, some being public and others private. In effect, this means that rights will be held by individuals, groups, and the state, with further complexity arising between land and aquatic space and in transboundary contexts. Many of the SDGs are underpinned by the delivery of one or more ecosystem services, meaning policy-makers will need to manage for multiple ecosystem services, which cannot be achieved if a single policy 'lens' is taken. Neither EA or ecosystem services alone will be sufficient to deliver on all the SDGs, but will involve concerted efforts from the spheres of institutions, technology, science, politics and society generally. The effective implementation of EBM and ecosystem services is contingent on the necessary legal and regulatory frameworks being in place. Whilst there have been attempts to incorporate both concepts into the law and policy framework there is still a large degree of contradiction in terms of over-arching objectives and goals of many key instruments. Until these are resolved, EBM in Europe will continue to be an aspirational concept rather than a tangible and effective management approach.

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Ecosystem Services in U.S. Environmental Law and Governance for the Ecosystem-Based Management Practitioner

Donna R. Harwell

Abstract This chapter provides an overview of ecosystem services issues in United States (U.S.) law and governance for the Ecosystem-Based Management (EBM) practitioner. A brief overview summary of a suite of U.S. federal environmental laws where ecosystems services are relevant is presented along with a high-level overview of ecosystem services in federal and state agency regulations as it helps inform ecosystem-based management. As with the published science-based literature on ecosystem services, there is also a sizeable law-based literature available on ecosystem services. A HeinOnline law journal library focused search identified 1903 legal articles that contained reference to ecosystem services. Focusing on a snapshot of key literature, this chapter presents an overview of those articles that contained "ecosystem services" or "ecosystem based management" just in the article's title. From this survey across the breadth of law journals, a suite of ecosystem services topics related to EBM in environmental law are identified and summarized. Overall, the goal of this chapter is to present a high-level overview and direct the reader to resources to find more in-depth legal analyses of select ecosystem services topics.

Lessons Learned

- EBM practitioners need to have the large suite of federal environmental laws that impact EBM as a frame of reference.
- There is active legal scholar literature on the intersection between ecosystem services and environmental laws.
- The majority of the active legal scholar literature is focused on the core environmental laws, however, the summary table informs the reader of the potential applications of other legal and governance frameworks to ecosystem services and EBM.

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- The community of EBM practitioners should take advantage of the legal scholar literature; the chapter demonstrates to the reader the value of adding the intersection between ecosystem services and environmental laws into the information space for EBM practitioners.
- In the recent past, a few U.S. states have started to add ecosystem services language into their statutory and regulatory materials.

Needs to Advance EBM

- EBM practitioners need to add information from the environmental law literature to their background information as part of efforts to frame ecosystem services information in their EBM activities.
- Practitioners can look to the environmental law literature to identify examples where relevant information might be transferable to their specific scenarios, such as the examples of watershed-based services (e.g., Funk et al. 2020).
- The suite of traditional EBM practitioners needs to expand to include law and governance practitioners in order to merge and create a large overlap and cross-information exchange between the disciplines.

1 Introduction to Ecosystem Services and EBM in Law and Governance

There are multiple ways to present the intersection between ecosystem services and environmental law and policy as it informs Ecosystem-Based Management (EBM). One perspective involves considering how the authority of individual law or regulations may influence the access, condition, protection, and/or utilization of nature. Examples of high-level overviews of key environmental laws include Ruhl and Salzman (2007), Thompson (2008), Davis (2010), Ruhl et al. (2013), and Farber and Findley (2014). Another perspective might be from an ecosystem-type lens. For example, Ruhl et al. (2013) explores three examples of United States (U.S.) environmental laws and regulations from wetland, coastal, and forest resources protection perspectives. The full suite of U.S. statutes for natural resources are highly domain-specific in character (Scarlett and Boyd 2015); Ruhl (2005b) successfully argues that, "(U.S.) ecosystem management law is a cobbled-together body of law, if it can even be called that much."

There is a large breadth of U.S. federal environmental laws that can be broadly organized around overarching purposes of protection/conservation, restoration/remediation, and regulations focused on socio-ecological interactions (Fig. 1; note that acronyms are captured in Table 1's compilation of these laws). As an individual law is inherently complex (e.g., containing multiple goals and objectives), its placement on a Venn diagram showing the relationships between protection and conservation, restoration and remediation, and socio-ecological perspectives shown in Fig. 1 is overly simplistic.

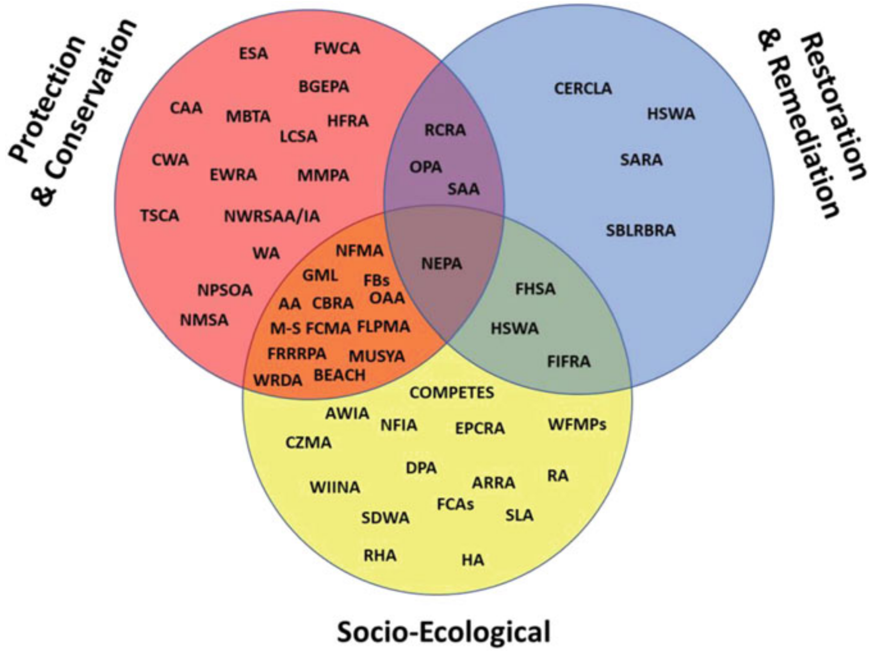

Fig. 1 Venn diagram identifying U.S. Federal environmental laws organized by three main perspectives. See Table 1 for acronyms used

Ruhl et al. (2013) highlights how the breadth of the U.S. environmental regulatory infrastructure from the 1970s–1990s became considered "top heavy." Hirokawa and Porter (2013) argue that, "the effort to integrate ecosystem services valuation into law has yielded complicated and unsatisfactory results." Further, legal scholars have called for the evolution of the application of environmental laws and regulatory tools to not only increase protection of ecosystems, but also the services they provide to people (e.g., Markell 2007; Ruhl et al. 2013). Examples of this perspective of characterizing how to protect ecosystem services through the use of regulations include Markell (2007), Davis (2010), and Pardy (2014).

The objectives of this chapter are two-fold: (1) to present an overview of the suite of U.S. Federal environmental laws and regulations with connections to ecosystem services; and (2) to present a survey of the legal scholar literature for a synopsis of ecosystem services issues in U.S. environmental law. These reviews are not intended to be fully exhaustive, but rather capture the broad suite of ecosystem services topics in U.S. environmental law and translated for EBM practitioners. For a recent overview of case law on ecosystem services, including U.S. examples, the reader is directed to Sharon et al. (2018).

Table 1 U.S. Federal environmental laws

Law & citation	Summary
Foundational laws over 100 years old	
Swamp Land Act 43 U.S.C. § 23 *et seq.*	The 1850 **SLA** provided legislation for giving Federal lands to the States in order to convert swamp lands into land for provision of agricultural and flood protection services.
Homestead Act 12 Stat. 392	The 1862 **HA** provided 160 acres of public land to homesteaders who paid a filing fee and lived on the land for five years before receiving the deed in order to promote westward expansion and the productive use (as a service) of the frontier. This was mostly repealed in 1976.
General Mining Law 30 U.S.C. § 22 *et seq.* (as amended)	The 1872 **GML** established that all valuable mineral deposits, and the lands where found, belonging to the United States were to be free and open to exploration and purchase for extractive services.
Organic Administration Act 16 U.S.C. § 551 *et seq.*	The 1897 **OAA** provided the authorizing legislation for the National Forest Service. The National Forest Service focuses on supporting forestry-based services.
Rivers and Harbors Act 33 U.S.C. § 407	The **RHA** of 1899 prohibited the construction over or in navigable waterways of the U.S. without Congressional approval and provided initial protection from water quality pollution.
Reclamation Act 43 U.S.C. § 391 *et seq.*	The **RA** was put into place in 1902 to set up water development (irrigation) projects in the U.S. west to support increasing westward settlement, including the productive use lands as a service, at the beginning of the twentieth century.
National Park Service Organic Act 16 U.S.C. § 1b *et seq.*	The 1916 **NPSOA** authorized the establishment of the U.S. National Park Service. The broad ecogeographic spectrum of National Parks encompasses a range of services associated with coastal, aquatic, and terrestrial services, along with public use and recreational-focused services.
Federal Migratory Bird Treaty Act 16 U.S.C. §§ 703–712	The 1918 **MBTA** is the federal enactment of the Migratory Bird Treaty (United States and Great Britain, acting on behalf of Canada; Mexico, Japan, and Russia subsequently signed onto this treaty) providing protections making it illegal to take, possess, sell or purchase any migratory bird (or parts) without a federal permit. The MBTA supports ecosystem services protection for recreational experiences and use for spiritual and ceremonial purposes.

(continued)

Table 1 (continued)

Law & citation	Summary
"Granddaddy" of U.S. Environmental Law	
National Environmental Policy Act 42 U.S.C. § 4321 *et seq.*	The 1969 **NEPA** law establishes the U.S.' goal to live harmoniously with nature by identifying where there is a federal nexus for the consideration of actions on the environment. It created the Environmental Impact Statement process that requires all federal agencies to review all potential actions for their impact on the environment. As a foundational piece of environmental legislation, there is a broad range of potentially relevant ecosystem services. (see Sect. 2.3)
Air Resources	
Clean Air Act 42 U.S.C. § 7401 *et seq.*	The **CAA** (including the 1970 Amendments) created a regulatory system to control most of the commonly produced and significant air pollutants. It set up air quality control regions and established National Ambient Air Quality Standards. Relevant ecosystem services may include air pollution removal and breathable air for human health and well-being. (see Sect. 2.3)
Water Resources	
Flood Control Acts 33 U.S.C. § 15 *et seq.*	A suite of legislation starting 1917, the **FCA**s provided authorizations for federal water control and flood protection services-based projects. The U.S. Army Corps of Engineers was established by the 1941 FCA with the authority to implement flood control policies such as the Flood Control and Coastal Emergency Act (Pub. L. No. 84–99).
Federal Water Pollution Control Act (**Clean Water Act**) 33 U.S.C. § 1251 *et seq.*	The 1972 **CWA** established that dumping in U.S. waters was not a right, that any person or corporation that wanted to dump anything into U.S. waters must have a permit, and that all wastewater must be treated, no matter the condition of the receiving waters. The CWA may involve many types of relevant ecosystem services, including pollution removal and supporting habitat condition for commercial and recreationally valuable species. (see Sect. 2.3)
Coastal Zone Management Act 16 U.S.C. § 1451 *et seq.*	The 1972 **CZMA** promotes development in the coastal zone area using a national perspective. but attempts to limit pollution arising from such development.
Safe Drinking Water Act 42 U.S.C. § 201 *et seq.*	The 1974 **SDWA** protects public drinking water supplies across the nation. It requires the U.S. Environmental Protection Agency to establish national primary and secondary drinking water standards to limit contaminants in drinking water, supporting the service of drinkable water.

(continued)

Table 1 (continued)

Law & citation	Summary
Coastal Barrier Resources Act 16 U.S.C. § 3501 *et seq.*	The 1982 **CBRA** was passed to protect and conserve coastal barriers, habitats that provide important flood protection and storm mitigation services. It prevents individuals who build in these zones from receiving any federal assistance, including federal flood insurance policies.
Emergency Wetlands Resources Act 16 U.S.C. §§ 3901–3932	The 1986 **EWRA** instructs the U.S. Fish & Wildlife Service to map the status and conditions of wetlands (and resulting services) across the U.S. and create a National Wetlands Priority Conservation Plan.
Water Resources Development Acts e.g., WRDA 2000 Pub. L. No. 106–541	**WRDA**s are a suite of laws focusing on a range of water resource management, protection, and utilization activities (and services) involving a federal nexus. For example, WRDA 2000 authorized most projects for the Comprehensive Everglades Restoration Plan in addition to other water resources protection plans across the country.
Beaches Environmental Assessment and Coastal Health (BEACH) Act Pub. L. No. 106–284	The Beaches Environmental Assessment and Coastal Health (**BEACH**) Act (2000) amends part of the Clean Water Act and focuses on monitoring and notifying the public about possible human health problems related to the ecosystem service of use for coastal recreation.
Water Infrastructure Improvements for the Nation Act Pub. L. No. 114–322	**WIINA** was passed in 2016 to focus on aspects of the U.S. drinking water infrastructure involving public health, supporting the service of drinkable water.
National Flood Insurance Act 42 U.S.C. § 4001 *et seq.*	The 1968 **NFIA** encourages states to create floodplain management programs that place restrictions on the location and type of construction, supporting the ecosystem service of flood protection. It also has a buy-out program component to take people out of floodplains to reduce rebuilding costs.
Deepwater Port Act 33 U.S.C. § 1501 *et seq.*	The 1974 **DPA** focuses on construction, operation, and decommissioning of deepwater ports (located beyond the U.S. territorial sea boundaries) and minimization of adverse impacts on the marine environment and the services they provide.
America COMPETES Reauthorization Act of 2010 Pub. L. No. 111–358	The 2010 **COMPETES** law instructs the Administrator of the National Oceanic and Atmospheric Administration to "identify emerging and innovative research and development priorities to enhance United States competitiveness, support development of new economic opportunities based on NOAA research, observations, monitoring modeling, and predictions that sustain ecosystem services." 33 U.S.C. § 893(b)(1)

(continued)

Table 1 (continued)

Law & citation	Summary
America's Water Infrastructure Act of 2018 Pub. L. No. 115–270	The 2019 **AWIA** law includes asking the National Academy of Science to examine how the U.S. Army Corps of Engineers approaches formulation, evaluation, and budget determination for water resources development projects, including "an analysis of whether such principles and methodologies fully account for all of the costs of project alternatives, including potential societal costs, such as lost ecosystem services, and full lifecycle costs for such alternatives." (Sec. 1103)
Land, Fish & Wildlife Resources	
Farm Bills 7 U.S.C. covers Agriculture	A suite of legislation starting in 1933, the **FB**s provided authorizations for, among other things, efforts on development and sale of agricultural products and other agroservices, research, and conservation. The Agricultural Act of 2014 (**AA**) called for an update to the strategic plan for forest inventory and analysis, with the implementation of an, "annualized inventory of trees in urban settings, including the status and trends of trees and forests, and assessments of their ecosystem services, values, health, and risk to pests and diseases." (Pub. L. No. 113–179; Sec. 8301)
Fish and Wildlife Coordination Act 16 U.S.C. § 661 *et seq.*	The 1934 **FWCA** law created the U.S. Fish and Wildlife Service, established the National Wildlife Refuge System, and requires the Secretary of the Interior to protect and conserve wildlife resources and authorize the public-use service of hunting of overabundant species.
Federal Wildland Fire Management Policies	The **WFMP**s are a suite of cross-Federal Agency wildland fire policies dating back to 1935 and have been revised following large-scale fire seasons (e.g., post-1998 fire season and the Yellowstone National Park fires). These policies have focused on supporting resource objectives in federal wildlands, including balancing the use of prescribed natural fires and ecosystem services (e.g., recreational access to nature) and potential negative ecosystem services effects such as smoke and stream sedimentation.
Bald and Golden Eagle Protection Act 16 U.S.C. § 668 *et seq.*	The 1962 **BGEPA** focuses on a suite of protections for two eagle species, including addressing issues of interference and abuse related to different aspects of shelter, breeding, nest abandonment, and feeding. BGEPA supports ecosystem services protection for recreational experiences and use for spiritual and ceremonial purposes.

(continued)

Table 1 (continued)

Law & citation	Summary
Multiple Use—Sustained Yield Act 16 U.S.C. § 528 *et seq*. (amended 1996)	The 1960 **MUSYA** established the system of National Forests for multiple outdoor recreation, range, timber, watershed, and fish and wildlife purposes (and services).
Wilderness Act 11 U.S.C. § 1131 *et seq*.	The **WA** of 1964 was established to preserve and protect, for current and future generations, certain wilderness lands in their "natural condition" with a number of relevant ecosystem services related to recreation and existence services.
National Wildlife Refuge System Administration Act 16 U.S.C. § 668dd	The 1966 **NWRSAA** is the U.S. Fish and Wildlife Service's "Organic Act" analog of the Park Service for management of the National Wildlife Refuge System for the purpose of protecting lands for the conservation of fish and wildlife, including threatened and endangered species, and puts boundaries on the ecosystem services of public access/use of refuge lands and waters.
Marine Mammal Protection Act 16 U.S.C. §§ 1361–1407	The 1972 **MMPA** represents the first legislation for ecosystem-based management for marine resources and was established to manage marine mammal species and population stocks as components of the ecosystems of which they are a part.
National Marine Sanctuaries Act 16 U.S.C. § 1431 *et seq*. (amended 2000)	The 1972 **NMSA** allowed for the designation and protection of special areas of the marine environment supporting a broad suite of coastal, recreational, and fisheries-related services.
Endangered Species Act 16 U.S.C. § 1531 *et seq*.	The 1973 **ESA** established protections for species, and their habitats, that have been listed as endangered or threatened. This law closed down the U.S. market in endangered wildlife, established heavy criminal penalties and fines for "taking" a member of an endangered or threatened species, and prohibits any federal actions that may impact the life or habitat of a listed endangered or threatened species. Habitats and supporting biodiversity preservation are often cited as the most relevant ecosystem services. (see Sect. 2.3)
Forest and Rangeland Renewable Resources Planning Act 16 U.S.C. § 1600 *et seq*.	The 1974 **FRRRPA** law gives authority to the U.S. Forest Service and U.S. Department of Agriculture to assess the Nation's renewable resources and develop and prepare a national renewable resources program for forestry and agriculture-based services.
Magnuson-Stevens Fishery Conservation and Management Act 16 U.S.C. § 1801 *et seq*. (amended 2007)	The 1976 **M-S FCMA** focuses on the valuable and renewable natural resources of U.S. fisheries across a range of geographic boundaries, ranging from anadromous species which spawn in rivers or estuaries, to migratory species of the high seas, and species in U.S. federal waters of the continental shelf. Ecosystem services related to fishery resources include harvesting for food supply and recreational opportunities.

(continued)

Table 1 (continued)

Law & citation	Summary
Federal Land Policy and Management Act 43 U.S.C. § 1701 *et seq.*	The 1976 **FLPMA** established the Bureau of Land Management, including inventories for present and future resource use.
National Forest Management Act 16 U.S.C. § 1600 *et seq.*	The 1976 **NFMA** Amends the Forest and Rangeland Renewable Resources Planning Act of 1974.
National Wildlife Refuge System Improvement Act Pub. L. No. 105–57	The 1997 **NWRSIA** updated the guidance for overall management of the National Wildlife Refuge System, including focus on maintaining the environmental health of the Refuge System while providing for determination of "compatible uses" of refuges for ecosystem services such as hunting and fishing, wildlife observation and photography, and environmental education and interpretation.
Healthy Forests Restoration Act Pub. L. No. 108–148	The 2003 **HRFA** was designed to minimize impacts of destructive wildfires on federal lands, including by allowing timber harvests on protected National Forests, and creating the ability for expedited NEPA review for projects under HFRA.
Chemicals	
Federal Hazardous Substances Act 15 U.S.C. § 1261 *et seq.*	The 1960 **FHSA** required labeling of hazardous household products to help consumers safely store and use products and allow for the ban of certain products that are dangerous or hazardous to human health. Ecosystem services are potential endpoints for a risk assessment process to examine the potential adverse effects of chemicals on the environment.
Federal Insecticide, Fungicide and Rodenticide Act 7 U.S.C. § 135 *et seq.*	The 1964 **FIFRA** law covers all chemicals manufactured to deal with pests in relation to agriculture and requires all of these chemicals to be registered with the U.S. Environmental Protection Agency prior to their use. Ecosystem services are potential endpoints for a risk assessment process to examine the potential adverse effects of chemicals on the environment.
Toxic Substances Control Act 15 U.S.C. § 2601 *et seq.*	The 1976 **TSCA** law requires the U.S. Environmental Protection Agency to keep a registry of chemicals used and introduced into everyday life. Ecosystem services are potential endpoints for a risk assessment process to examine the potential adverse effects of chemicals on the environment.
Emergency Planning and Community Right to Know Act 42 U.S.C. § 11001 *et seq.*	The 1986 **EPCRA** was created to help communities plan for chemical emergencies. It also requires industry to report on the storage, use and releases of hazardous substances to federal, state, and local governments. EPCRA requires state and local governments, and Indian tribes to use this information to prepare their community for potential risks. Ecosystem services are potential endpoints for a risk assessment process to examine the potential adverse effects of chemicals on the environment.

(continued)

Table 1 (continued)

Law & citation	Summary
Frank R. Lautenberg Chemical Safety for the 21st Century Act Pub. L. No 114–182, 130 Stat. 448	The 2016 **LCSA** is an update of the TSCA requiring the U.S. Environmental Protection Agency to have ongoing evaluations of chemicals registered under the TSCA using a risk-based standard. It also requires the EPA to impose fees on chemical manufacturers to pay for these evaluations. Ecosystem services are potential endpoints for a risk assessment process to examine the potential adverse effects of chemicals on the environment.
Environmental Remediation	
Resource Conservation and Recovery Act 42 U.S.C. § 6901 *et seq*.	The 1976 **RCRA** law regulates the waste cycle by defining wastes and dictating how they are to be treated. It aims to prevent the release of hazardous wastes into the environment. RCRA can support the protection or restoration of a suite of ecosystem services that are location specific.
Comprehensive Environmental Response, Compensation and Liability Act 42 U.S.C. § 9601 *et seq*.	The 1980 **CERCLA**, also referred to as Superfund, is an administrative system for removing hazardous materials from sites where they had been improperly dumped into the environment in years past. Sites are identified and placed on a National Priorities List, then assessed through the hazard ranking system. CERCLA can support the protection or restoration of a suite of ecosystem services that are location specific. (see Sect. 2.3)
Hazardous and Solid Waste Amendments Pub. L. No. 98–616, 98 Stat. 3221	The 1984 **HSWA** law requires the U.S. Environmental Protection Agency to develop criteria for identifying hazardous waste: ignitability; corrosivity; reactivity; and toxicity. It added stringent regulation of land disposal of hazard wastes to RCRA. HSWA can support the protection or restoration of a suite of ecosystem services that are location specific.
Superfund Amendments and Reauthorization Act Pub. L. No. 96–510, 94 Stat. 2767	**SARA** (1986) is the reauthorization of CERCLA and required that the hazard ranking system be updated and all identified sites were reviewed for possible water contamination due to run off. SARA supports the protection or restoration of a suite of ecosystem services that are location specific.
Oil Pollution Act 33 U.S.C. § 2701 *et seq*.	The 1990 **OPA** established a trust fund to clean up spills when the responsible party is incapable or unwilling to do so and outlines requirements for facilities (e.g., aboveground storage facilities) and vessels (e.g., oil tankers) to detail how they will respond to large discharges. OPA clean-up activities can support the protection or restoration of a suite of ecosystem services that are location specific.

(continued)

Table 1 (continued)

Law & citation	Summary
Small Business Liability Relief and Brownfields Revitalization Act Pub. L. No. 107–118, 115 Stat. 2356	The 2002 **SBLRBRA**, referred to as Brownfields, amended CERCLA to increase funding for cleanup at urban and suburban CERCLA sites. It focuses on cleanup of sites with petroleum or other hazardous waste contamination. Brownfields revitalization can support the protection or restoration of a suite of ecosystem services that are location specific.
American Recovery and Reinvestment Act Pub. L. No. 111–5, 123 Stat. 115	The 2009 **ARRA** updated CERCLA to add a large amount of stimulus monies to the Superfund in order to accelerate ongoing clean-up activities. As a result, ARRA can support the protection or restoration of a suite of ecosystem services that are location specific.
Supplemental Appropriations Act Pub. L. No. 111–212	The 2010 **SAA** included the call for an "ecosystem services impact study" by the National Academy of Sciences to, "conduct a study of the long-term ecosystem service impacts of the Deepwater Horizon oil discharge. Such study shall assess long-term costs to the public of lost water filtration, hunting, and fishing (commercial and recreational), and other ecosystem services associated with the Gulf of Mexico." (Sec. 2004)

The laws are grouped by thematic areas and year, and where the reader can find more information on an individual law via the full title, year, and legal citation information. U.S.C. = U.S. Code; § = Section; §§ = Sections; *et seq.* (*et sequentes*) = "and what follows"; Pub. L. No. = Public Law Number; Stat. = Statutes at Large

A high-level summary provides initial information about each law, its commonly used acronym (used for Fig. 1), and relevant examples to the field of ecosystem services

2 Ecosystem Services in U.S. Federal Environmental Laws

This Section presents a brief overview of a suite of U.S. federal environmental laws where ecosystem services and ecosystem-based management topics may be relevant. Here, this chapter expands beyond the list of what are considered "key" environmental laws to highlight the larger breadth of U.S. laws, and where the reader can turn to find more information, that may have relevance to ecosystem services and ecosystem-based management. Additionally, this chapter includes information on early U.S. federal legislation (over 100 years old), ancillary legislation not considered part of the "traditional suite" of environmental laws, and select Executive Orders that speak to environmental law related to ecosystem services.

2.1 Foundational Legislation

The foundational elements of U.S. environmental law predate the flurry of activities in the 1970s–1990s (Ruhl et al. 2013) and ultimately can be anchored in Roman law's "recognition that the general public had inalienable rights to access and use certain resources, namely the sea and seashore, rivers, and the air" (Connolly 2009). This is referred to as the "Public Trust Doctrine" (cf., Sax 1970; Ruhl 2005a; Ruhl and Salzman 2006); the first case addressing this in the U.S. occurred in 1842 (Smith and Sweeney 2006). Examples of 100+ year old U.S. Federal legislation that set the stage for identifying the importance of (protecting and valuing) ecosystems include: the Swamp Land Act (1850), the Homestead Act (1862); the General Mining Act (1872), the Organic Administrations Act (1897), the Rivers and Harbors Act (1899), the Reclamation Act (1902), the National Park Service Organic Act (1916), and the Migratory Bird Treaty Act (1918) (Donahue 2007; Tarlock 2007; Hirokawa 2011c; Cosens and Fremier 2014; Robbins 2018b).

2.2 U.S. Federal Environmental Laws—Overview

An overview of approximately 50 U.S. Federal environmental laws is presented in Table 1, including a description of each law's goals and purpose and an initial identification of which parts of a given law that may have relevance to ecosystem services and ecosystem-based management. Examples of U.S. Federal regulations that directly speak to ecosystem services are presented in Table 2. For a broader overview of the suite of U.S. environmental laws, the reader is referred to Farber and Findley (2014) and Salzman and Thompson (2003). At, or near, the "top" of the key list of U.S. Federal environmental laws are the National Environmental Policy Act, the Clean Water Act, the Clean Air Act, the Endangered Species Act, and the Comprehensive Environmental Response, Compensation, and Liability Act. The rest of this section briefly introduces these key laws and their intersection with ecosystem services.

2.3 "Key" U.S. Federal Environmental Laws

One of the primary U.S. Federal environmental laws with relevance to ecosystem services is the National Environmental Policy Act (NEPA), with the requirement that federal agencies evaluate a suite of alternatives (including a "no action" scenario) for developing pros/cons lists before a decision is made (Anderson 2011). Fischman (2001) argues for the direct utility of ecosystem services assessments as they may be "exactly the kind of assessment NEPA envisions, providing a means to inform the public and decision-makers about what we stand to gain or lose in several alternative

Table 2 Current U.S. federal regulations capturing "ecosystem services"

Agency	Code section	Summary	Citation
U.S. Department of Agriculture, Natural Resources Conservation Service	Healthy Forests Reserve Program Compensation for Easements and 30-year Contracts	This is part of the Healthy Forests Reserve Program to assist landowners to restore, enhance, and protect forestland resources on private land.	7 CFR § 625.8
	Grasslands Reserve Program Definitions	These definitions include conservation values which covers sustaining and enhancing ecosystem functions of grasslands.	7 CFR § 1415.3
	Wetlands Reserve Program Market Based Conservation Issues	Establishes the use of environmental credits for entities that implement conservation practices and activities.	7 CFR § 1467.20
	Agricultural Conservation Easement Program Environmental Markets	This section gives ecosystem service credits to landowners for conservation improvements to wetland reserve easements.	7 CFR § 1468.10
	National Forest System Land Management Planning Assessment; Sustainability; Multiple Use	This part deals with land management plans and the assessments of plan developments to include collaborative and science-based input so the lands involved are ecologically sustainable and have the capacity to provide ecosystem services to people and the community.	36 CFR § 219.XX (0.1; 0.6; 0.8; 0.10; 0.19)
U.S. Environmental Protection Agency	National Ambient Air Quality Standards Revisions to the Guideline on Air Quality Models	This Appendix provides guidelines for air quality modeling related to the derivation of Ozone National Ambient Air Quality Standards, noting that "emissions of NO_X, sulfur oxides, NH_3, mercury, and secondary pollutants such as ozone and particulate matter" can affect ecosystem services provided by forests and natural areas. (40 CFR Part 51, Appendix W)	40 CFR Part 51 Appendix W

CFR, Code of Federal Regulations; §, Section

scenarios." Fischman (2001) gives an example list of five types of NEPA-relevant decision activities:

1. Community-scale development activities with a federal nexus (e.g., highways; flood protection);
2. Development and use of renewable resource on public lands (e.g., logging and grazing);

3. Use (e.g., development, generation, and transmission) of renewable energy production, including coal, petroleum, and natural gas;
4. Use (e.g., development, processing, and transport) of non-energy mineral resources; and
5. Implementation of water projects, including permitting (e.g., wetland modification).

Cross-walking these examples with the Millennium Ecosystem Assessment's approach (Millennium Ecosystem Assessment 2005; Carpenter et al. 2009; da Silva and de Carvalho 2018) to classifying ecosystem services into four main categories:

- Provisioning (e.g., food/fiber; fuel);
- Regulating (e.g., water, disease);
- Cultural (spiritual; recreational; aesthetic); and
- Supporting (e.g., primary production; nutrient cycling)

it becomes clear that there is extensive relevance of applying NEPA to a range of ecosystem services that may be considered as part of NEPA consultations. Some example topics within NEPA's umbrella that are relevant include: property (Sect. 1.1); valuation (including cost/benefits analysis and markets; Sect. 1.2); development of alternative scenarios (including mitigation; Sect. 2); environmental impact assessments (not discussed here); and habitat evaluations (not discussed here). Recent legal scholar publications on NEPA and ecosystem services include Fischman (2001), Hirokawa and Porter (2013), Ruhl (2015). As a side note, the Millennium Ecosystem Assessment framework has been applied to environmental law issues around a number of topics (Thompson 2008; Ruhl 2015), including agrosystems (Ruhl 2008), public lands (Ruhl 2010a), aquatic resources (Ruhl 2010b), as well as the evolution of the ecosystem approach in international environmental law (Enright and Boteler 2020; Le Tissier 2020; O'Hagan 2020).

The Federal Water Pollution Control Act (commonly referred to as the Clean Water Act) includes provisions to protect aquatic ecosystems from human activities in order to protect a range of ecosystem services, including pollution removal (dilution and breakdown), providing habitat for wildlife (including those harvested commercially and recreationally), and assimilation and sequestration of nutrients (e.g., removal of excess nitrogen) (Salzman et al. 2001; Craig 2008; Ruhl 2010b; Smith et al. 2010). The Clean Water Act also includes provisions for mitigation banking, a mechanism of preservation, enhancement, or restoration of a specific natural resource area in order to provide compensation for the loss or degradation of another natural resource (see Sect. 2.1; Davis 2010). Additionally, there are several current topics of legal discussion with the Clean Water Act, including issues of jurisdiction (e.g., Craig 2008), setting Total Maximum Daily Loads (Ruhl 2010b), and filling wetlands (Ruhl et al. 2009) that are outside the scope of this chapter. As a side note, the Beaches Environmental Assessment and Coastal Health (BEACH) Act (2010), as an amendment to the CWA, is an example of additional legislation focused on the intersection between identification of human health issues and the ecosystem service of coastal beach use for recreation.

The Clean Air Act focuses on air quality protection and establishment of standards and intersects with ecosystem services in a number of areas, including, nutrient pollution removal (e.g., nitrogen, sulfur), and regulation of greenhouse gas emissions (e.g., Lazarus 2008; McGuire 2015).

The Endangered Species Act focuses on single-species management of threatened and endangered species, but with capacity to give attention to related habitats and for programmatic and multi-species consultations. Consideration of the intersection with ecosystem services is established in the literature for issues related to critical habitat (Salzman 1997; 2006), (indirect) protection of biodiversity (Thompson 2008; McGuire 2015), the use of Habitat Conservation Plans (Davis 2010), and the use of credits (Davis 2010).

The Comprehensive Environmental Response, Compensation, and Liability Act ("Superfund") has ecosystem services related connections to damage assessment (Wilson 2004; Desjardins 2014) as well as approaches to enhance cleanups (e.g., Green Remediation; Lipps et al. 2017) and redevelopment (Thompson 2008). See Sect. 4 on ecosystem services and remediation.

2.4 Non-Traditional Suite of Laws Related to Environmental Law and Ecosystem Services

There are other regulations not considered part of the suite of traditional environmental laws that are related to how ecosystem services are considered. For a land-use example, the National Flood Insurance Program, authorized by the National Flood Insurance Act, include the influences on, and distortion of, land prices that influence coastal and flood-plain development decoupled from other ecosystem valuation efforts for these important ecosystem landscapes (McGuire 2015). In contrast, the 1990 Conservation Reserve Program, established by the "Farm Bill," assesses (ranks) land parcels with the highest environmental benefits based on multiple criteria (Boyd et al. 2001; Davis 2010).

2.5 Executive Orders

Another suite of U.S. federal tools that can be used to examine related issues are Executive Orders (EOs), directives from the U.S. President to the Executive Branch of the government, including covering rulemaking for federal agencies such as the U.S. Environmental Protection Agency, U.S. Department of the Interior, and the U.S. Army Corps of Engineers. It is important to acknowledge that EOs represent policies, which are the operational applications of laws. That is, they capture different approaches to governing and interpretations of the execution of laws by the Executive Branch. This introduction to EOs is not intended to present an exhaustive survey of EOs relevant to ecosystem services, rather introduce this type of mechanism to the reader. For example, the National System of Marine Protected Areas was established in 2000 through EO 13158 ("Marine Protected Areas"). While EOs have extensive authority in that they are implemented at the same level as a regulation, they do not overrule an individual law, they are not legislatively approved, and they can be rescinded with the stroke of a pen by subsequent administrations.

Examples of EOs that explore further development of environmental-related cost-benefit analyses (Thompson 2008) include a suite of EOs on "Regulatory Planning and Review": EO 12866 (1993; 58 FR 51735), EO 13258 (2002; 67 FR 9385), EO 13422 (2007; 72 FR 2763), EO 13563 (2011; 76 FR 3821), and EO 13777 (2017; 82 FR 12285). This example suite of Executive Orders spans across multiple Presidential administrations.

In another example, EO 13547 ("Stewardship of the Ocean, Our Coasts, and the Great Lakes"; 2010; 75 FR 43023) explicitly referred to ecosystem services in providing guidance for coastal and marine spatial planning, specifically identifying those areas, "most suitable for various types or classes of activities in order to reduce conflicts among uses, reduce environmental impacts, facilitate compatible uses, and preserve critical ecosystem services to meet economic, environmental, security, and social objectives." This EO was revoked in 2018 and replaced by EO 13840 ("Ocean Policy to Advance the Economic, Security, and Environmental Interests of the United States"; 2018; 83 FR 29431) that did not reference the "ecosystem services" that nature provides to people but does reference the "benefits" the ocean provides the U.S. economy. As a side note, the reader is directed to Craig (2007) to learn more about coastal ecosystem services and environmental law and policy.

3 Themes in Ecosystem Services, EBM and Environmental Law

This Section presents a high-level literature review analysis of the existing legal scholarly literature on several current ecosystem services topics within U.S. federal environmental law. A literature search of abstracts, titles, and keywords published in the legal scholar literature was conducted using the HeinOnline law journal library search engine to identify potential peer-reviewed sources. The period of record for HeinOnline searches ranged from the date of inception for each legal journal in their database through March 2019. As a frame of reference, the HeinOnline search identified a total of 1903 legal articles that contained reference to ecosystem services. Focusing in on a snapshot of key literature, this chapter presents an overview of those articles that contained "ecosystem services" or "ecosystem based management" in the article's title. Although this search was not exhaustive, it provides a high-level snapshot of the current state of emphasis within the legal scholarly literature. The analysis presents a suite of ecosystem services themes in this literature, including fundamental elements (property and ownership; valuation, accounting and markets), conservation and protection (conservation and mitigation banking, public lands), and remediation on the "back end," including an introduction on natural resources damages.

3.1 Property and Ownership

Some aspects of ecosystem services may be subject to property rights law, that is, whether the value of an ecosystem service can be reduced to ownership (Hirokawa 2011c; Ruhl 2015). Pardy (2014) outlines one property premise related to developing approaches to protecting ecosystem services, namely that, "although some ES have no market value because they are not the subject of property rights and/or are not easily exchanged, all ES have an economic value that can be calculated by measuring their actual or potential importance to human well-being." Robbins (2018b) provides a general characterization that the case law for ecosystem services-based regulatory takings generally does not expressly treat ecosystem services as a property interest. And Hirokawa (2011c) argues that because ecosystem services may not have discrete boundaries, they could be considered property interests within another's property boundaries. One area of intersection between environmental law, property law, and ecosystem services is in "ecosystem energy services" (Hodas 2013). Ruhl (2005b) argues that the nuisance aspects of "common law" may be applicable for ecosystem services because the structure of this vehicle is flexible to handle changes, such as those encountered in the evolution of both the science of ecosystem services and its consideration in society (Hirokawa 2011c). The reader is directed towards Abrams (2007) for an overview on nuisance law and ecosystem services.

In the case of conservation easements, areas established to maintain essential habitat for species that can also provide ecosystem services, Cooley and Olander (2012) and Robbins (2018b) argue that because human value for ecosystem services can be extrapolated from easements, ecosystem-services related easements are considered property. Additional areas of development in environmental law and policy include the potential applicability of easements, and the services they may provide, for use in markets (see Sect. 1.2), such as for carbon credits, which require establishing a permanence of the market for credits (Ristino 2010). Easements, however, may not be permanent property instruments, and thus the ownership of the benefits (including delivery of ecosystem services) from a given easement, is an area of active development in the law (McLaughlin 2015).

From a technical perspective, there is a difference between an ecosystem good and an ecosystem service, namely that a good represents a market product (e.g., harvestable timber), while a service represents an ecosystem process or function (e.g., wetland filtering out water pollution) (Brown et al. 2007). Furthermore, the delineation of those ecosystem goods and services into *intermediate* (supporting products and processes not directly used by humans) and *final* (those used directly by humans) services to advance classification systems and environmental accounting (DeWitt et al. 2020; Russell et al. 2020) may also inform future discussions on ecosystem services and property law as it relates to matters of ownership. As a side note, there is continuing debate about biodiversity as an ecosystem service (Goble 2007), including whether it represents a final ecosystem service (DeWitt et al. 2020) directly benefiting people.

3.2 Ecosystem Services Accounting, Markets

The field of ecosystem services accounting and valuation is an ongoing area of scientific development. Pardy (2014) describes the three primary approaches for protection of ecosystem services as: (1) a regulatory approach (e.g., da Silva and de Carvalho 2018); (2) payments to protect ecosystem services (e.g., Hirsch 2007; Ruhl 2008; Benjamin 2013; Salzman et al. 2018), including investments in green infrastructure (Cosens and Fremier 2014; Salzman et al. 2014; da Silva and de Carvalho 2018); and (3) market-based approaches (e.g., Salzman 2005; Hirsch 2007; Glicksman and Kaime 2013; Kaime 2013). From an environmental law perspective, the authority for using valuation and accounting, and the range of potential approaches and methodologies themselves are all areas of ongoing development, case law, and legal debate. The primary legal spaces include natural resource damages, the consideration of compensation and mitigation, and the establishment of markets.

One primary approach for valuation includes "(focus) on a traditional, tort-like derivation of damages through per-unit calculations of past, present, and future damages" (Desjardins 2014). In one example, the Habitat Equivalency Analysis approach, a CERCLA provision using an accounting approach for habitat status/condition that is used to look at lost and restored services from a one-to-one comparison perspective (Ray 2009; Shaw and Wlodarz 2013), has received attention in a number of areas of environmental policy management, including natural resource damage assessments (NOAA 2000), NEPA projects (e.g., Ray 2009), and restoration decision making (Snyder and Desvousges 2013). Equivalence assessment approaches, including Habitat Equivalency Analysis and the related Resource Equivalency Analysis approach, are used for measuring losses and gains in habitat and biodiversity have been developed for a range of purposes (Desjardins 2014; Bezombes et al. 2017). Another suite of ecosystem services valuation focuses on the "willingness-to-pay" approaches, such as Contingency Valuation, the application of methodologies for natural resources that have no established market (Carson et al. 2001).

Ruhl et al. (2009) and Womble and Doyle (2012) explore mitigation banking in wetland and stream ecosystems resulting from the Clean Water Act and the 2009 Compensatory Mitigation Rule and its focus on market-based assessments of these ecosystems from a compensatory mitigation perspective. Further discussion on issues associated with geographic boundaries in environmental law and policy are outside the scope of this chapter, but the reader is directed to Womble and Doyle (2012) and Ruhl et al. (2009) for more information. Ruhl and Salzman (2007) and Salzman et al. (2018) present an overview of payments for ecosystem services, including both positive and negative incentives ("carrots vs. sticks"; Salzman et al. 2018) from a mitigation context.

A decade ago, there were more than 700 ecosystem services markets in the U.S. (Ristino 2010), with more than 2400 markets by 2016 (Bennett et al. 2016). Ruhl and Salzman (2007) provides examples of markets for forests. For carbon-based markets, one area of policy development is in carbon offsets, an accounting approach whereby the reduction in carbon emissions by one source could be used to offset the need for reduction in carbon emissions by another source. Carbon credits is one area of property law that is still in development (Ristino 2010; Glicksman and Kaime 2013; Ruhl et al. 2013). While this chapter does not explore current cap-and-trade issues, the reader is directed towards Glicksman and Kaime (2013) and Ruhl et al. (2013) to learn more. Likewise, the reader is pointed towards Brown et al. (2007) to learn more about technical and policy issues associated with measuring "carbon dioxide-equivalents" or the "social cost of carbon."

There are a number of environmental law and policy issues related to the development and implementation of markets, including property law, credits, banking, and accountability and oversight (e.g., Ristino 2010; Glicksman and Kaime 2013). One area of on-going property-based efforts is focused on real property instruments, the legal vehicles used to assign ownership of property (Ristino 2010). Another relates to the potential use of conservation easements (see Sect. 2.1), and whether easements can create permanency of both the credits themselves, and ownership of those credits, as it relates to how those credits are considered from a market or governance perspective.

3.3 Conservation, Protection & Mitigation Banking Tools

The Wilderness Act (1964) was primarily focused on the protection of public lands, including non-extractive services (Kammer 2013). The Federal Land Policy and Management Act (1976), focusing on the management of public lands, has a "no degradation" requirement that including taking necessary actions to prevent unnecessary or undue degradation of public lands (Donahue 2007). The 1978 Public Rangeland Improvement Act, implemented by the Bureau of Land Management, characterizes "less-than-potential production of ecosystem services, namely, 'wildlife habitat, recreation, forage, and water and soil conservation benefits,' is evidence of rangelands' 'unsatisfactory condition.'" (Donahue 2007). There is a large breadth of rangeland improvements called for by the Public Rangelands Improvement Act (and subsequent regulations), including soil resources, water resources, fish and wildlife habitat resources, and improvements for livestock and wild horse management Penderly (1997). Ecosystem services elements of forests are broad (Neuman

2007); Federal forest lands are "administered for outdoor recreation, range, timber, watershed, and wildlife and fish purposes." (16 U.S.C. § 528). Three U.S. Federal laws overseeing forestry services include the National Forest Management Act (1976), the Forest and Rangeland Renewable Resources Planning Act (1974), and the Multiple-Use Sustained-Yield Act (1960). From an environmental law perspective, attention in forestry ecosystem services includes advancing concepts of "payment for services" (Ruhl and Salzman 2007), balancing vegetation management plans, provisioning of forestry goods, and potential impacts on watershed function (Hirokawa and Porter 2013).

There are a number of federal "incentive programs," whereby the government pays private landowners to protect ecosystems and their services, including the Conservation, Wetlands, and Grasslands Reserve programs (Table 2), Environmental Quality Incentives Program, the Farm and Ranch Lands Protection Program, the Conservation Security Program, and the Forestland Enhancement Program (Brown et al. 2007; Ruhl 2008). In a related tool, the U.S. Department of Interior oversees the Land and Water Conservation Fund (1965), designed, in part, to "preserve ecosystem benefits for local communities" for both public and private lands (Land and Water Conservation Fund 2017). Areas of current attention in agricultural ecosystem services is in markets for carbon offsets (Davis 2010; see Sect. 3.2 for more on markets), and the U.S. Department of Agriculture's use of, "ecosystem service values as a basis for payments under traditional conservation program payments" (Ruhl 2015).

The U.S. government's Council of Environmental Quality's regulation on mitigation includes a section on, "compensating for the impact by replacing or providing substitute resources or environments" (40 CFR 1508.20(e)). Mitigation banking, an approach to protect, enhance, or create a habitat (particularly wetlands) as compensation for the impacts at other locations, is one tool used for providing compensation for ecosystem impacts. Examples of mitigation banking include those established through the authority of Sect. 404 of the Clean Water Act. U.S. federal agencies involved in mitigation banking include the U.S. Environmental Protection Agency, U.S. Department of Agriculture, the U.S. Fish and Wildlife Service, and the U.S. Army Corps of Engineers. Conservation easements are established to maintain essential habitat for species, where Cooley and Olander (2012) and Robbins (2018b) argue that human value for the ecosystem service can be extrapolated. Here, ecosystem services related easements are considered property. Discussions on environmental law and policy perspectives on ecosystem services and mitigation banking include Salzman and Ruhl (2000), Boyd et al. (2001), Hirsch (2007), and Robbins (2018b). A spin on wetland mitigation banking for use as a market for biodiversity offsets is presented in Spurgeon (2008).

3.4 Remediation on the "Back End"—Natural Resources Damages

There are a number of examples of natural resource damage provisions in U.S. Federal law that mandate valuation in response to a loss, or deprivation of ecosystem function and services, including the Deepwater Port Act (1974), the Oil Pollution Act (1990), the National Marine Sanctuaries Act (1972), and the Comprehensive Environmental Response, Compensation, and Liability Act (1980) (commonly referred to as Superfund) (Boyd et al. 2001; Wilson 2004; Smith et al. 2010). Salzman (1997) provides an early environmental law analysis of the need for information on ecosystem services information markets to feed the design of remediation strategies in Superfund. Another example of another ecosystem services related regulation that focuses on violation/penalties is the Natural Resources Damage Assessment that focuses on assessing compensation for injuries to natural resources (see Boyd et al. 2001 and Davis 2010 for overviews). In characterizing the U.S. Environmental Protection Agency's three-prong approach to enforcement (deterrence; fairness; swift resolution of environmental problems), Markell (2007) provides an overview of three tools: penalties for violations; injunctive relief (i.e., a court-driven order to address a problem); and Supplemental Environmental Projects (SEPs) as a form of relief in case settlements. A number of U.S. Federal environmental laws focus on prevention of ecosystem contamination (Table 1) using ecological risk assessment characterizations as an important tool. For an overview of efforts to advance ecosystem services as assessment endpoints in the ecological risk assessment process, the reader is directed to Munns et al. (2016).

4 Ecosystem Services and Environmental Law at Different Scales

One guiding principle of EBM involves the interaction across different scales (federal, state, and local) to address geographic-based management issues (Nugent and Cantral 2006; Green et al. 2014). This section presents an overview of where ecosystem services are captured at different scales, including state agency laws, and several examples at regional and local scales. Examples of current State environmental laws are introduced in Table 3. At present, just the three U.S. west coast states (Washington State, Oregon, and California) and Rhode Island have laws that explicitly refer to ecosystem services.

Examples of ecosystem services captured within U.S. regional-scale environmental law issues include:

- Ecosystem-Based Management of the western U.S. (e.g., Smith 1999);
- The use of the Endangered Species Act as an overarching framework for northwest Montana (Guercio and Duane 2009);

Table 3 Current U.S. state laws capturing "ecosystem services"

State	Code Section	Summary	Citation
Oregon	Public Health and Safety	Chapter 468 of the Oregon statutes deals with environmental quality. "It is the policy of this state to support the maintenance, enhancement and restoration of ecosystem services throughout Oregon, focusing on the protection of land, water, air, soil and native flora and fauna." Or. Rev. Stat. 468.583 (2018)	468.581; 468.583; 468.585; 468.587 (2018)
	Forestry and Forest Products	These sections of the Oregon statutes advance the continuation of the Forest Resource Trust to promote establishment and management of nonindustrial state forestland through the use of paying landowners for preserving ecosystem services.	526.695; 526.703; 526.705 (2018)
	Water Resources: Irrigation, Drainage, Flood Control, Reclamation	This Oregon statute establishes the Water Resources Department to develop and implement a holistic water usage plan for the entire state of Oregon. It includes ecosystem services as a point to consider when developing such plan.	536.220 (2018)
Washington	Forest and Forest Products	This Washington statute establishes a forest maintenance plan to wisely use timber resources and replenish such, including payments to forest landowners for ecosystem services provided to the public in preserving timber resources.	76.09.010; 76.09.020 (2019)
	Forest and Forest Products	This Washington statute discusses the Forestry Riparian Easement Program and reimbursement to small forest landowners for preservation of timber resources and ecosystem services supported by the program.	76.13.120 (2019)
	Public Lands	This Washington statute discusses the community forest trust program and that preservation of "ecosystem services such as clean water protection or carbon storage." Wash. Rev. Code s 79.155.030(2)(c)	79.155.030 (2019)
California	Fish and Game	This California statute defines "Ecosystem-based management" as "an environmental management approach relying on credible science, as defined in Section 33, that recognizes the full array of interactions within an ecosystem, including humans, rather	Cal. Fish & Game Code § 43 (West 2019)

(continued)

Table 3 (continued)

State	Code Section	Summary	Citation
		than considering single issues, species, or ecosystem services in isolation." Cal. Fish & Game Code § 43 (West 2019)	
	Fish and Game	This California statute gives definitions of words used in the state's advance mitigation and regional conservation investment strategies and includes incorporating the benefits of ecosystem services as part of the "regional conservation assessment" definition.	Cal. Fish & Game Code § 1851 (West 2019)
	Public Resources	This California statute instructs the Ocean Protection Council to support sharing of information between state agencies and making that information publicly available with respect to "social, economic, and cultural values, including the value of coastal and ocean ecosystems for providing ecosystem services." Cal. Pub. Res. Code § 35620(a)(2)(E) (West 2019)	Cal. Fish & Game Code § 35620 (West 2019)
Rhode Island	Health and Safety	This statute is the legislative findings for the Rhode Island Climate Risk Reduction Act of 2010 where the legislature states that "natural ecosystems and habitats, both coastal and upland, provide critical ecosystem services including, fisheries habitat, drinking water, and flood protection." 23 R.I. Gen. Laws § 23-84-2(6)	§ 23-84-2 (2019)

- The Northwest Forest Plan's framework (e.g., Neuman 2007); and
- Legal and regulatory authorities for managing the coastal resources of the Gulf of Mexico (Nugent and Cantral 2006), such as the use of the Manguson-Stevens Fishery Conservation and Management Act (1976) for red snapper and other fishery species (Pace 2009).

Importantly, Federal laws have spurned state and local laws, often looking at the scale of a watershed, the boundary of which may not necessarily align with political or governance boundaries. Looking at a broader suite of examples related to water protection, Greenwalt and McGrath (2009) explore the tenets of a pay-for-ecosystem-services (PES) model at a watershed scale. For a specific example, the Safe Drinking Water Act spurred New York City to implement local regulations on protecting the Catskills and Delaware watersheds providing the primary source of clean drinking water for its citizens (Thompson 2008; Salzman et al. 2001; Salzman

2011; Robbins 2018a, b). Other watershed-scale examples include water-based natural and engineered services in the Columbia River Basin (Cosens and Fremier 2014), and the production (Greenwalt and McGrath 2009), purification (Salzman et al. 2001), and apportionment of water for municipal and other uses (Ruhl 2003). Green et al. (2014) examine EBM issues at different legal scales for coral reefs, an example where upstream land-use decisions may not align with different scales and domains of existing environmental regulations on the downstream resource of interest.

Local-scale forestry examples in the legal literature includes the ecosystem management of Tillamook State Forest, involving a range of stakeholders, different scales of regulatory hierarchy in forest and adjacent lands, and a suite of forest-related ecosystem services (Neuman 2007 and citations therein). Other local-scale examples include urban forest planning (Hirokawa 2011a, b), land-use policies for agrosystems (Ruhl 2008), and salmon fisheries (Hirokawa and Gottlieb 2011). Other urban ecosystem services issues, also considered local scale, are outlined in Salzman et al. (2014).

5 Conclusions

EBM practitioners work in an interdisciplinary universe, spanning a range of science, engineering, and management/policy backgrounds and expertise. This chapter presents an overview of the large spectrum of U.S. Federal environmental laws, with particular relevance to the field of ecosystem services. Anchored by a review of the extant legal scholarly literature, this chapter presents a review of a broad suite of ecosystem services topics in U.S. Federal environmental law specifically translated for EBM practitioners as the primary audience, pointing the reader towards resources to learn more about individual elements presented throughout the chapter. This chapter provides EMB practitioners information from the environmental law literature to inform how they frame the legal context of ecosystem services information in their EBM activities. Finally, this chapter helps the reader identify examples where relevant information might be transferable to their specific scenarios, such as how policy and legal directives are framed in the watershed-based EBM example of the Danube Basin (Funk et al. (2020)).

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Unravelling the Relationship between Ecosystem-Based Management, Integrated Coastal Zone Management and Marine Spatial Planning

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Abstract Coastal zones are among the most productive areas in the world, offering a wide variety of valuable habitats and ecosystems services that have attracted humans and human activities over millennia. But equally coastal zones are also among the most vulnerable areas to climate change, natural hazards and other anthropogenic perturbations. The impacts of coastal change are far reaching and are already changing the wellbeing of coastal communities. It is essential to make use of long-term management tools to enhance the conservation of coastal resources whilst increasing the sustainability of their uses. Integrated Coastal Management (ICM) and Marine Spatial Planning (MSP) are both tools that attempt to override traditional sectoral approaches that lead to disconnected decisions and missed opportunities for more sustainable coastal development. Ecosystem-Based Management (EBM) describes the comprehensive integrated management of human activities based on the best available scientific knowledge to achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity. However, there is a degree of contradiction regarding the juxtaposition of EBM to ICM and MSP—does it underpin and coordinate the implementation of them or does ICM and MSP coordinate the application of EBM principles to management practices and goals or does the difference in terminology detract from the real challenge of achieving sustainability of the world's coastal and marine areas? This chapter provides insights into the juxtaposition of these concepts and suggests a promising future approach founded on Biodiversity Portfolio Analysis (BPA).

Lessons Learned

- Adherence to terminology rather than end goals can blur the emphasis and principles of processes needed to address environmental challenges in coastal and marine areas.

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- There is a need to recognise the coastal and marine environment as a portfolio of constituent elements that need to be managed as a cohesive whole.
- Principles and processes of EBM need to be juxtaposed with other methods and tools for understanding environmental challenges faced by society now and in the future.

Needs to Advance EBM

- Stronger inculcation of societal elements and their explicit inclusion in EBM principles and processes is a necessary requirement.
- Developing a coherent approach to link EBM with existing policy processes and outcomes is important.

1 Introduction

Scientific and policy communities have increasingly recognised that complex environmental problems are an existential threat to humanity that require integrative, interdisciplinary approaches to understand and manage the interaction between social and ecological systems (Binder et al. 2015; Defries and Nagendra 2017). Concurrently, there is also a growing recognition that environmental change is experienced at many inter-related scales from local to global with cause and impact also ranging from near to distant (Adger and Brown 2010). Whilst the sciences have quantified and documented environmental change and its consequences [op. cit.], research has also shown that human activities are largely behind the driving forces that are the proximate causes of environmental change (Stern et al. 1992).

Formal concepts of ecosystem management initially arose in the 1970s aligned to wildlife management and were first transferred to the marine environment as part of the United Nations Conference on the Law of the Sea (UNCLOS) (Forst 2009; Long et al. 2015). Environmental policies increasingly advocate a holistic approach to coastal and marine resource management (Buhl-Mortensen et al. 2017) that address an increasing degree of anthropogenic pressures on coastal and marine environments as well as conflicts between multiple users competing for space and resources. The Ecosystem Approach (EA) is a strategy that underpins the objectives of the Convention on Biological Diversity as "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way" (CBD 2000), and is widely referred to as Ecosystem-based Management or EBM (Long et al. 2015). EA is embedded in the concept of sustainable development, which requires that the needs of future generations are not compromised by the actions of people today, and puts emphasis on a management regime that maintains the health of the ecosystem alongside appropriate human use of the marine environment, for the benefit of current and future generations (ICES 2005; Defries and Nagendra 2017). Key features of EBM centre around notions that management should be holistic and not focused on single sectors or species [c.f., Garcia et al. 2003] and directed to managing human activities and their impacts on ecosystems rather than the ecosystem itself (Leslie and McLeod 2007). However, the reality has been that both scientific research and management programmes have applied concepts of EBM with very diverse variations in emphasis, definition of terms and application of principles (Yaffee 1999; Arkema et al. 2006; Sardà et al. 2014; Long et al. 2015). In large part this is because there has been a mismatch in institutional arrangements (Alexander and Haward 2019) where Integrated Coastal Management (ICM) and Marine Spatial Planning (MSP) have been employed to redress a traditional sectoral focus of marine environmental and resource management (Ehler and Douvere 2009; Freestone et al. 2010; UNEP 2010; Smith et al. 2017) and have stronger policy and governance as well as management components (Javier 2015). ICM and MSP have had greater policy presence because they are seen as providing a means to improve decision making by providing a framework to analyse competing human activities and managing their impact on the marine environment (Buhl-Mortensen et al. 2017).

2 Unravelling EBM, ICM and MSP

Degradation of coastal and marine ecosystems is an often cited impact of environmental change (e.g., Stern et al. 1992; Arkema et al. 2006; Alexander and Haward 2019), and is a consequence of post-1900 industrialisation and post-World War II economic and population growth (Yasuhara et al. 2012), which have increasingly seen both human migration to the world's coastal zones and exploitation of coastal and marine space and resources as demand has outstripped availability of land-based space and resources (Caddy and Grithiths 1995; Long et al. 2015). EBM has been advocated as a key pillar in the sustainable management of coastal zones and marine areas in conjunction with ICM and MSP respectively (UN Environment 2018; Langlet and Rayfuse 2019).

There is an ever-growing, largely academic, literature that debates the relationship between the definitions and roles of EBM with ICM and MSP (see for example, Haines-Young and Potschin 2011; NOAA 2011; Aswani et al. 2012; Celliers et al. 2016) in the context of whether one is subsumed within the other (Golitsyn 2010) and the legal basis and definitions of these terms in different jurisdictions are discussed in detail in this volume (See Enright and Boteler 2020; O'Hagan 2020; Harwell 2020). The aims of ICM and MSP whilst principle-based, in common with EBM, are primarily place-based in their implementation in order to:

- Reduce conflicts, and enhance synergies, between sectors and their activities, and
- Protect and conserve the environment and its resources upon which those activities are dependent.

ICM and MSP seek to achieve these objectives by:

- Integrating between levels of government and other management authorities (including across administrative boundaries),
- Integrating between disciplines, and
- Integrating across spatial and temporal scales.

Although EBM essentially includes much of these aims and objectives it differs in its sense of 'place', which is focussed on ecosystem units rather than sectoral activities and, whilst acknowledging that humans are part, are dependent on and occur with the ecosystem, places a different emphasis of management objectives. The consequence is that EBM can place an unequal weighting to the three tenets of sustainable development—environmental, economic and social equity—and lead to the view of human activity as impacts to ecosystem long-term viability and longevity. In contrast, MSP and ICM are essentially planning processes that seek to overcome the fragmentation in single-sector management approaches and analyse and organise human activities in coastal and marine areas to achieve economic and social objectives whilst safeguarding ecological integrity (Sandersen et al. 2013). Thus ICM and MSP have evolved with a greater emphasis towards designing rules and procedures for how to govern an area (Rodriguez 2017). This can lead to decisions that emphasises a 'balance' between protection of the environment and the maintenance and development of coastal and marine dependent economies: such a perspective is inherent to many coastal and marine strategies such as the EU integrated maritime policy (European Commission 2007). Therefore, whilst EBM has evolved to include many of the same principles as ICM and MSP (Haines-Young and Potschin 2011) it has a greater emphasis on conserving ecosystems and ecosystem services, which brings in additional management challenges over and above merging sectoral approaches to exploitation of coastal and marine space and resources (Golitsyn 2010). What this has meant is that, regardless of the terminology used, the practices of EBM by implementing authorities have tended to focus on applying an EBM approach to inform strategies and plans that seek to promote the sustainable growth of coastal and maritime economies, the sustainable development of marine areas and the sustainable use of marine resources. For instance the EU Marine Strategy Framework Directive as the environmental pillar of the EU's cross-cutting Integrated Maritime Policy, and aspects of the National Environmental Policy Act (NEPA) in the USA.

3 Current Coastal and Marine Management Regimes

The coastal and marine environment and its resources have been managed through fragmented and sectoral approaches (Altvater and Passarello 2018; Kelly et al. 2018) that fail to incorporate the complexity and interconnection of marine ecosystems and the cumulative pressures that different human activities have on species and habitats (see O'Hagan 2020 for discussion on the dominance of the Common Agricultural Policy and the Common Fisheries Policy over activities within the environment for

Europe). EBM is described as an integral component of management regimes, such as ICM and MSP (Kittinger et al. 2014; Robinson et al. 2014; Javier 2015), but has developed in a number of parallel approaches leading to a plethora of terminology and variation in the detail of principles depending on the ultimate management interest. Essentially the recognition of the importance of EBM within any management approaches is centred on acknowledging the relationship of ecosystem services to human welfare and emphasising the need for tools that encourage coordination and cooperation, participation, transparency, public interest, etc., to achieve the governance of these spaces. At the same time, the increasing interest in developing new uses and activities bring an important concern about its environmental consequences (NOAA 2011). A further benefit of EBM is its focus on natural boundaries and interconnections rather than the 'un-natural' administrative boundaries that characterise social and economic organisation (Paxinos et al. 2008; Environmental Law Institute 2009a; UNEP 2012). Therefore, EBM may be viewed as a process that implements the concepts of an Ecosystem Approach into planning management regimes for coastal and marine areas (Douvere 2008; Ansong et al. 2017).

The expectation that EBM with ICM and MSP can close the gaps between societal objectives and the state of the environment is great (Karlsson 2019). Many nations and regions have management regimes that seek to harmonise laws, policy, plans and strategies within prevailing institutional arrangements to achieve sustainable development of coastal and ocean space that balance different uses of the space and resources (Balgos et al. 2005). Increasingly, EBM has been applied to coastal and marine areas to incorporate both environmental and non-environmental factors into management regimes in order to inculcate human systems as components of the natural environment (Domínguez-Tejo et al. 2016). These initiatives seek to embed EBM as a procedure to ensure that management processes achieve an equitable and sustainable balance between conservation of the environment and persistence of ecosystem services into the future with the multiple demands on coastal and marine space and resources (Rodriguez 2017). The practical application of EBM within management regimes has recognised that there is a divergence from theory depending on political and socio-economic priorities (Jones et al. 2016) but that EBM provides opportunities to address challenges of functional metrics and indicators, spatial and temporal measures to address multiple and cumulative uses of resources, integration across sectors and adaptive management (Rodriguez 2017).

There is a growing awareness that management regimes, and the way they are informed by EBM approaches, need to be updated to account for contradictory environmental/societal and economic/stakeholder goals that can incentivise an economic and environmental perspective to act against each other (Breen et al. 2012). Furthermore, EBM, ICM and MSP need to evolve methodologies that better account for the consequences of current and future scenarios of development and uses of coastal and marine areas.

4 The Future

The academic debate over the supremacy and juxtaposition of EBM, ICM and MSP terms (and their variants) of one above the other perhaps misses the point that they all purport to have a primary objective oriented around concepts of sustainable development, which is principally about meeting the needs of the present without compromising those of the future (Brundtland 1987; Holden et al. 2014). To deliver 'sustainability', and develop an organisational methodology for the integrated management of natural resources requires approaches that complements but go beyond 'classical' conservation concepts such as endangered species or habitat protection and various types of protected area designations (English Nature 2003). Such considerations lie behind concepts that seek to extend ideas of humans as an integral component of the ecosystem (Millennium Ecosystem Assessment 2005) to those of humanity existing within planetary boundaries (Raworth 2012; Dearing et al. 2014) and ecosystem services. However, assessing the diversity of social relationships with coasts and marine space can prove difficult for scientists and practitioners in order to protect and conserve the services and benefits they provide (Kittinger et al. 2014). In particular outstanding challenges exist in regard to the spatial distribution of social, environmental and economic values; contrasting across different types of uses and their cumulative effects; addressing risk and uncertainty; and the juxtaposition of administrative and jurisdictional boundaries versus ecological boundaries (Domínguez-Tejo et al. 2016). This presents barriers to incorporating social dimensions of marine ecosystems into ecosystem-based management, which can in turn affect the success of planning and management initiatives encompassed by ICM and MSP.

Sustainable development is often addressed as a finite utopian vision of some 'perfect' future. However, sustainable development is increasingly viewed as a process—or transformative pathway—towards a largely unknown (or uncertain) future that can change over time and space whereby concepts of adaptation and resilience hold greater importance (Denton et al. 2014; Feola 2015; Romero-Lankao et al. 2016). Management of coastal and marine space and resources often focuses on "islands" of high value ecosystems, in terms of economic value or conservation (Hills et al. 2009). However, integrated management requires land and seascape-levels analysis of all ecosystem values from both monetary and non-monetary perspectives. In viewing sustainable development as a pathway the opportunity for the balance between different elements—environmental, economic and social—can vary over time and space. In actuality coastal and marine spaces are a mosaic of different interests vying for the 'best' return on activities whether they are conflictory or complementary, exploitative or conservationist. Furthermore, relative returns can vary over time and space. In practice, this means that successful EBM needs to incorporate analyses of risk and return to ensure that management outcomes lead to greater security across all elements—human and environment—of coastal and marine systems over short and long timescales.

The Biodiversity portfolio Analysis (BPA) method is derived from the logic used in share (equity) portfolio management in terms of balancing within a portfolio the returns with the risks (Figge 2001; Breen et al. 2012). Optimising the returns from a share portfolio, or a suite of ecosystems in a landscape, is dependent on the relationship between the units in terms of risk and return in time and space. This leads to an approach that more approximates with portfolio management that provides flexibility to adapt and adjust over time and space as situations change (Hills et al. 2009; Breen and Hynes 2014)—especially given the uncertainty associated with all three pillars of sustainability. It has been suggested that BPA as part of a holistic management approach could lead to strategies that 'favour' conservation as a longer term strategy maintaining future options in comparison with current management approaches that can be 'weighted' towards sustaining the status quo of existing coastal and marine use with less emphasis on future potential development and opportunities (Figge 2001; Hills et al. 2009; Breen et al. 2012). In practice this means that drawing distinctions between pillars of sustainability and assigning a hierarchy of one over the other is likely to be counterproductive. ICM and MSP influence the spatial and temporal distribution of human activities and, to attain sustainable development, can only be effective if EBM, with its focus on the costs and benefits that ecosystem services provide, runs concurrently (Douvere and Ehler 2006; Environmental Law Institute 2009a, b; Forst 2009; Soma et al. 2015; Hummel et al. 2017; Altvater and Passarello 2018). In this way sustainable development of coastal and marine spaces can be achieved through:

- Addressing the heterogeneity of coastal and marine areas to reflect the interdependency of ecology and human activities upon each other.
- Influencing the behaviour of humans and their activities over time in a way that respects ecological limitations and boundaries.
- Addressing the conflicts and compatibility issues that arise from different human activities targeting the same ecological resources and/or different ecological resources from the same space.
- Steering single-sector management to become integrated across multiple sector decision making.

The complexity of managing coastal and marine areas is ever increasing as growing populations identify more uses for the services and benefits provided by the world's coastal and marine areas, and build the capacity to exploit opportunities (Francis et al. 2019). As the number and variety of both possible uses of coastal and marine spaces, and interest groups pursuing uses, escalates it is important to be able to assess the effect of management regimes not only in the short term but over medium and longer term planning horizons. A portfolio tactic allows an assessment to be made of a range of possible scenarios the outcomes from an EBM approach to achieve a sustainable future, whilst minimising the risk to that objective.

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Part V
Case Studies

Models and Mapping Tools to Inform Resilience Planning After Disasters: A Case Study of Hurricane Sandy and Long Island Ecosystem Services

Mark Myer and John M. Johnston

Abstract In the aftermath of Superstorm Sandy in 2012, recovery and rebuilding efforts focused on resilience and diversified infrastructure that included consideration of the benefits that healthy ecosystems provide. County governments on Long Island identified a need for tools to map coastal and estuarine areas that may provide ecosystem services. Current methods of ecosystem service mapping often rely on complicated statistical models, labor-intensive site validation, or proprietary data. We examined a method of fast ecosystem services mapping that relies on publicly-available data, includes stakeholder input, and uses ArcGIS software that is ubiquitous in municipal planning. This chapter provides an example of ecosystem service mapping that generates easily explained visualizations suitable for non-scientific audiences with tools already available to municipal planning departments. We explain how to define indicators of benefit presence, obtain data, and create maps using examples from a collaboration with Nassau County, Long Island, New York.

Lessons Learned

- Some situations do not require an effort-intensive modeling approach to ecosystem services mapping—for these, a quick estimate serves the purpose.
- Consulting with stakeholders at every step of the process is essential. As researchers, our ideas of which ecosystems and benefits are important may not match theirs.
- It is possible to map the areas that may provide ecosystem services using publicly available data with a combination of expert consultation and inductive reasoning.

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Needs to Advance EBM

- Despite several contemporaneous efforts, there is no agreed-upon standard to rigorously define ecosystem services. This subjectivity introduces uncertainty into service mapping.
- Local partners may not immediately perceive utility in ecosystem-based management, underscoring the need for communication and outreach that enumerates its advantages over established paradigms.

1 Hurricane Sandy and the South Shore of Nassau County

Superstorm Sandy made landfall in New Jersey and New York on October 29, 2012, leaving behind a wake of destruction along the U.S. eastern seaboard. The storm caused billions of dollars in damage, killed at least 147 people in the United States (Blake et al. 2013) and dozens more overseas, and it left millions temporarily without electricity and fuel (Diakakis et al. 2015). In Nassau County, Long Island, - New York, voluntary evacuations were announced for the south shore's storm surge area in anticipation of extensive damage. On impact, Sandy's storm surge was nearly 14 ft above mean low tide, causing inundation of coastal areas and shoreline changes from erosion and accretion of sand and sediment (Hapke et al. 2013). Disruption to south shore bays from an influx of salt water and sediment was widespread, including a reduction in eelgrass (*Zostera marina*), which serves as a crucial habitat for local shellfish (Tinoco 2017). In the aftermath of the storm, New York City and the surrounding communities committed to rebuilding and adding infrastructure in ways that increased resilience to future natural disasters, with consideration of green infrastructure methods of reducing stormflow, increasing infiltration, and reducing nutrient runoff to improve ecosystem services (Interboro Team 2014; The City of New York 2013). As part of the Federal response to the disaster, the U.S. Environmental Protection Agency's Office of Research and Development and Region 2 (including New York, New Jersey and Puerto Rico) worked with the Federal Emergency Management Agency, New York Department of State, county policy makers, and others to identify projects that diversify the built and green infrastructure portfolio while increasing community resilience to natural disasters.

Hempstead Bay is in the western part of Long Island's south shore embayment, extending approximately from Far Rockaway in Queens to Massapequa in south-eastern Nassau County (Fig. 1). Long Beach forms the barrier between the Bay and the Atlantic Ocean on the cityward side, with Jones Beach continuing the chain of barrier islands to the east. The area supports a diversity of ecosystems, including freshwater streams, brackish streams, tidal wetlands and marshes, seagrass and open bay, and barrier islands. The Bay is integral to the lifestyle enjoyed by residents, both by providing ecosystem services directly and indirectly supporting others. Shellfishing for clams, oysters, and scallops is both a recreational activity and part of the local economy, with bay scallops alone contributing millions of dollars per year (Peconic Estuary Program 2015). Sailing and recreational boating are enjoyed

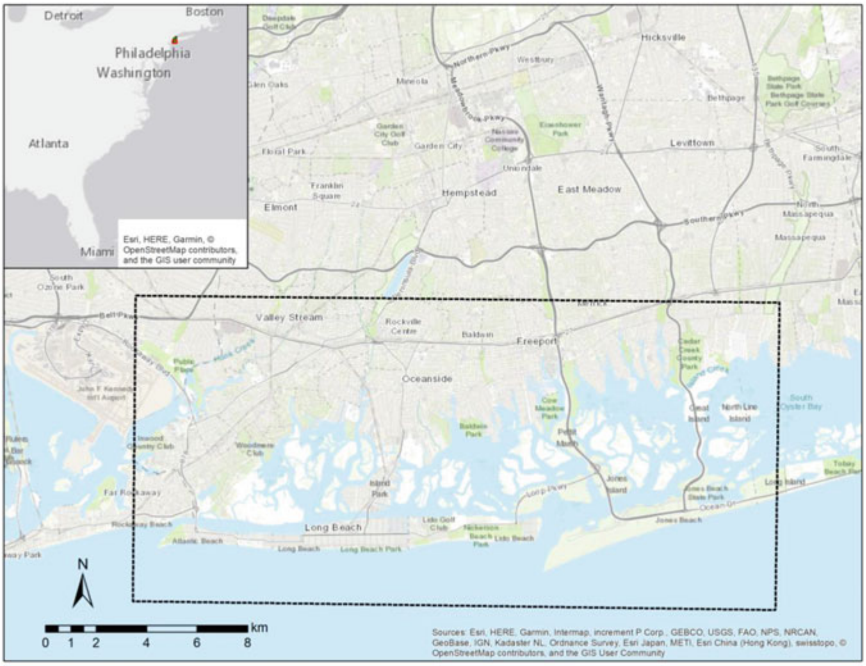

Fig. 1 Study Area: South shore of Nassau County in Long Island, NY. Hempstead Bay is located between the south shore and Long Beach, in the dotted box

in the warmer seasons. Several rare and endangered birds including the piping plover (*Charadrius melodus*) and yellow-crowned night heron (*Nyctanassa violacea*) reside in the area, which makes it a destination for birdwatchers (Cohen et al. 2006; Hodgman et al. 2015). Aquatic vegetation, both emergent and submerged, attenuates wave energy and decreases inundation from storm surge (Paul et al. 2012). The Bay ecosystem is recovering but faces pressures from development and pollution. The south shore experienced decades of stressors from development activities resulting in loss of submerged aquatic vegetation and coastal wetlands and increased nutrient loadings that impair coastal ecosystems (Hartig et al. 2002). Local commercial shellfish landings are far from historic highs because of habitat loss. Though storms are a normal dynamic of coastal ecosystems, areas stressed by human activity have less resilience, limiting their ability to recover from disturbance (Carpenter et al. 2001). The maps and methods described here were provided to the county planning office to help communicate the presence of and potential for enhancement of ecosystem services. Increasing awareness of local ecosystems and their services, especially through the use of intuitive maps, was a first step in building understanding of the relationships between wetland vegetation and coastal resilience and fisheries production.

2 Background on Nature's Benefits

We consider the term "nature's benefits"—the benefits of nature that people care about—to be synonymous with ecosystem services. Final ecosystem goods and services are provided or created by ecosystems and directly enjoyed or utilized by people. We acknowledge the distinction between final and intermediate services (Lamothe and Sutherland 2018; DeWitt et al. 2020); however, we don't address this further. Our goal was to help Nassau County visualize and communicate the location of benefits that people derive from the Bay, therefore we used the informal, intuitive concept of nature's benefits. We avoid confusing the concept by clarifying that any part of the Bay that is utilized or enjoyed and was not built by people is a nature's benefit.

The local government of Nassau County (Fig. 1) managers were interested in mapping nature's benefits to identify priority areas of the Bay that provide multiple services and to determine what benefits are located near areas of planned development. They wanted to identify, characterize, and describe locations that provide nature's benefits as part of public communication and outreach tools as part of overall efforts to mitigate negative impacts from human activity and to target efforts that support those benefits.

3 Geographic Information Systems—Utility of Arc ModelBuilder

The desired product for Nassau County was an illustrative map, suitable for inclusion in a handout or a poster for public-facing communication, that could be quickly produced without a long, data-gathering process. Inspired by the "service-providing area" maps of Angradi et al. (2016) used to visually characterize geographical areas that provide targeted ecosystem services, these are intended to quickly convey where benefits are likely to occur. Although they are simplifications of the natural world, a major strength is that they may not require fieldwork or an expensive monitoring program to generate. Nature's benefit maps outline the geographical areas that have the potential to provide ecosystem services, conveying those benefits through the use of a few, easily described indicators that outline the boundaries/presence of a given benefit. We considered cost, simplicity, and end-user ease-of-use as drivers of the overall design goals, and we targeted county and municipal planners (and their communications staff) as the intended users.

To meet these design goals, we used the ModelBuilder semi-automated map-building feature available in ESRI ArcGIS, the most popular and widely used Geographic Information System (GIS). GIS is used to work with spatial data and create maps and is considered an essential tool in municipal planning, environmental science, ecology, environmental economics, and many other disciplines. ModelBuilder is a visual representation of GIS operations in ArcGIS as a directed

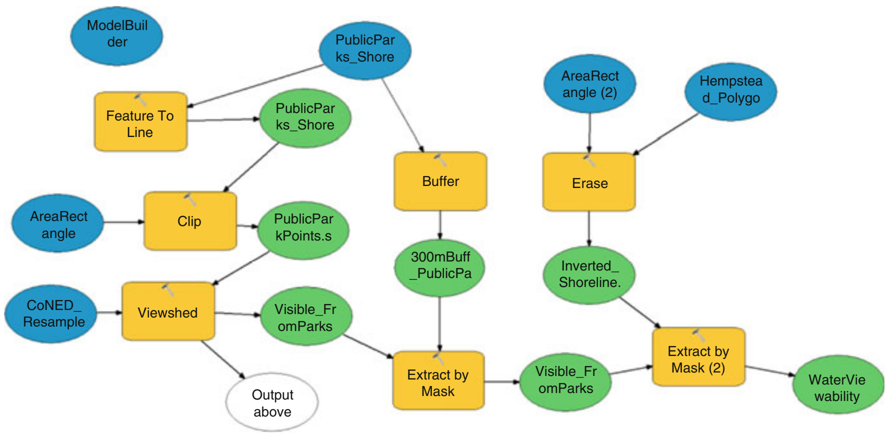

Fig. 2 A ModelBuilder directed graph (i.e., a GIS workflow) used to create a map of areas of the shore suitable for recreational viewing of aquatic animals. A blue oval is a map file, a yellow box is a GIS operation, and a green or clear oval is an output file

graph (Fig. 2). Shapes and colors are used to represent a map file, a table of input data, a mapping operation, or an output file. Following the arrows shows the GIS workflow as steps from input to output. For example, in Fig. 2, the input file PublicParks_Shore is a shapefile that contains areas of public parkland adjacent to the ocean and is represented as a blue oval. Following the two arrows from the PublicParks_Shore input, the map operations performed on the shapefile are *Feature To* Line, which turns the shape into a line, and *Buffer*, which outputs a polygon of the area a given distance from its input. Map operations are represented as orange rectangles. The output is represented as a green oval that can be used in other map operations, creating a continuous workflow. A ModelBuilder workflow can be saved as a portable file and shared for reuse and editing by other users with ArcGIS software. Benefits maps can be delivered with ModelBuilder files as a compressed archive with all required input data in a single package.

4 Steps to Generate a Nature's Benefits Map

Generating a nature's benefit map begins with local knowledge to determine which benefits are present and amenable to mapping. It is important to consult with residents, representatives of businesses that are associated with the local landscape (farming, fishing, or ecotourism for example), political representatives, and researchers including ecologists, hydrologists, and geologists. Consulting as many experts as possible strengthens the impact of the product and fosters inclusion and ownership. Soliciting the values and preferences of the various stakeholders also helps rank the priority order of the potential list of benefits (Sharpe et al. 2020). Meetings, surveys, or personal correspondence can all be used to determine which benefits are most important to a community. It may be necessary to brainstorm a preliminary list of benefits to jumpstart conversations, especially among audiences that have not been exposed to ecosystem services concepts. It is also possible to highlight the benefits of nature that may be overlooked or underappreciated by including these in a preliminary list for stakeholder consultation.

Amenability to benefits mapping means that a particular benefit should be present in a fixed location and there should be information available to determine indicators of its presence. Reliable indicator data must be available to determine the location of benefits so that GIS operations can be performed to identify their location across the county (i.e., area of interest). Benefits must occur in a discrete location that can be visualized on a map to be considered amenable. If a benefit rarely exists or occurs almost everywhere, it is unlikely to be a good choice for nature's benefits mapping as it may not help inform differences between management alternatives for a particular decision. For example, reduction of bay nitrogen pollution is a benefit related to the presence of denitrifying bacteria (Christensen et al. 1987). However, microscopic bacteria can potentially occur everywhere in the Bay, and their location and abundance are also in flux, so it is difficult or impossible to map this benefit.

5 Indicator Selection

Because many benefits cannot be directly observed or quantified, we use indicators to estimate their presence. In our case indicators are environmental (i.e., habitat) characteristics known to occur with the presence of a plant or animal species. We define an indicator as a mappable (fixed, measurable) quantity that spatially co-occurs with the benefit. As such, nature's benefits maps indicate where a benefit may be present but is not guaranteed. In other words, the presence of the indicator is necessary but not sufficient to ensure the presence of the benefit. Reliable indicators are crucial because a map based on faulty assumptions will be misleading or incorrect. The guiding question to ask is "what are the one or two characteristics that are almost always present when this benefit is provided?"

To illustrate indicator selection, we use the examples of hard clam gathering and shore fishing. For the hard clam example, the benefit is those clams that are harvestable, so metrics are needed to convert this benefit into rules for mapping the extent of this benefit. Because clam collectors can only reach so far underwater, even if they are using a specialized tool, one indicator will be water depth less than 2 m (about 6 ft) at mean tide, which will represent the area where a collector could reasonably reach the bottom of the Bay at low tide (Wells 1957). The second indicator represents areas where hard clams are likely to live. We researched the ecology of the hard clam and found they tend to live in areas with a sand or mud bottom (Wells 1957; Walker and Tenore 1984). Therefore, our indicators for hard clam collecting are areas of the Bay with a sand or mud bottom in two meters or less depth at mean tide, because that is where hard clams that people can reach are most likely to occur. It's important to note that only clams that people can harvest are considered a nature's benefit: if they can't be collected, they aren't considered a benefit. Because of this, most nature's benefit maps include at least one accessibility indicator.

Similarly, for shore-based fishing, indicators should reflect the potential of the public to catch fish while standing on a shore. The first indicator will be those areas identified as shoreline; the area must be adjacent to water, because an angler needs to be on the shore to catch fish. Our second indicator for shore-based fishing is public accessibility, for example state or local parks. Finally, an angler can reasonably cast a line about 30 m (around 90 ft) at the most, so benefits locations will be within 30 m of shoreline. Combining these indicators results in areas that provide the benefit of shore-based fishing.

6 Mapping Indicators

Once indicators that provide the mapping boundaries of a given nature's benefit are chosen, the next step is to find spatial data to represent the indicator, download it preferably from a publicly available source, and load it into a GIS platform. Indicators can be in almost any geodata format, from elevation and land cover rasters to wetland and soil type polygons and bathymetry contours. In the eight examples created for Nassau County, publicly available indicator data were used, avoiding the need for the use of proprietary or privileged information.

In the example of hard clam gathering, our first indicator was an indicator of accessibility, as water depth two meters or less at mean tide. A raster elevation map showing water depth of tidal zones is the indicator, and we used the Coastal National Elevation Database Project (United States Geological Survey 2019). The website contains a download link for the Topobathymetric Digital Elevation Model (https://gis.ny.gov/elevation/NYC-topobathymetric-DEM.htm), which shows the data availability for coastal U.S. waters. As an example of ensuring that indicator data are current and not outdated, we used a special report on the website that detailed how the topography and bathymetry model was adjusted after Hurricane Sandy (Stronko 2013). The second indicator was sand bottom type. Bay bottom substrate type was provided by the United States Fish and Wildlife Service's National Wetlands Inventory (NWI) (United States Fish and Wildlife Service 2019). This dataset includes wetland and estuarine ecosystems for the coastal United States. We obtained the Wetlands and Deepwater Code Diagram from the NWI to provide the bottom substrate of estuaries and bays (Fig. 3).

For shore-based fishing, obtaining all of the indicators was more challenging. For the first indicator, an internet search for "United States shoreline polygons" directed us to the National Oceanic and Atmospheric Administration's (NOAA) Office for Coastal Management shoreline website (National Oceanic and Atmospheric Administration 2019) where we ultimately chose the NOAA Composite Shoreline dataset, which the site says is for high-resolution cartographic work. We then clipped it to our study area using GIS to reflect the south Hempstead Bay area.

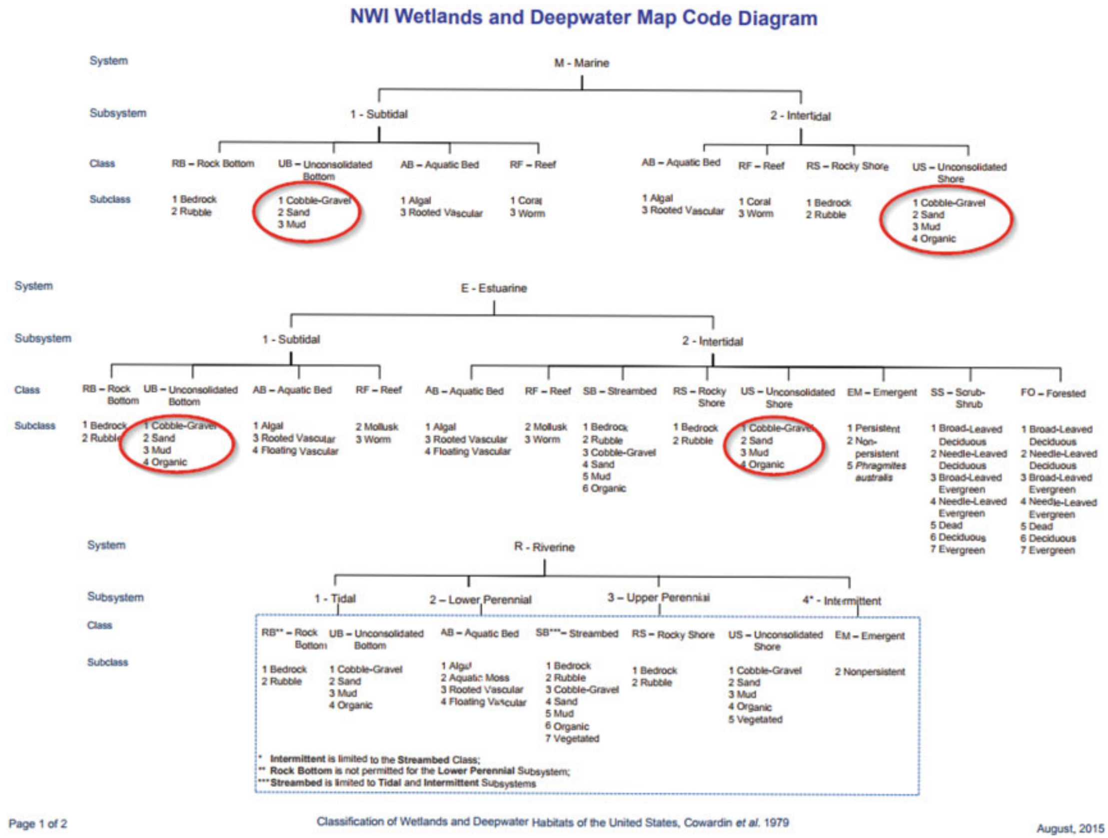

Fig. 3 The National Wetlands Inventory contains indicators of aquatic bottom types (cobble-gravel, sand, mud, organic). Sand bottom type, important for hard clams, is identified with red ovals

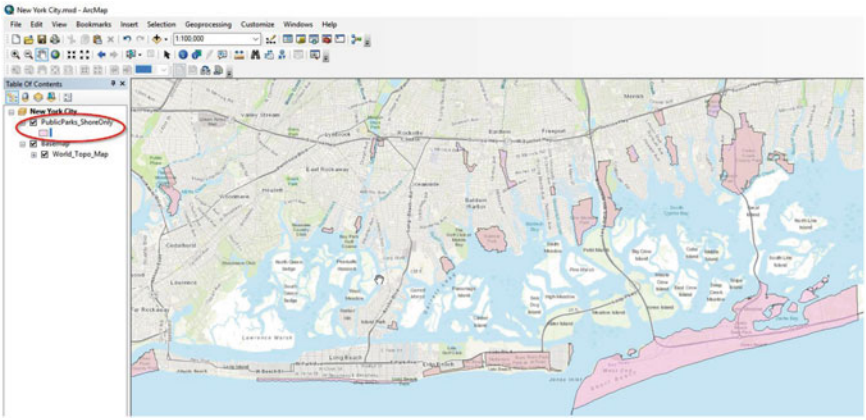

Fig. 4 Screen capture of ArcMap's Table of Contents for Nassau County shoreline fishing benefits mapping. Public parks indicator was created from other data

The second indicator, publicly-accessible areas, didn't exist for our area of interest, therefore, we created it. The ESRI World Topographic Map is provided with an ArcGIS Online subscription, and we traced the boundaries of all the areas labeled as public parks or beaches along the shore using the Editor tool to create a new polygon file (Fig. 4). Unlike other indicator data, these were not from a U.S. government source. Even though we did not use the original map, instead we referred to it as a guide to make a new polygon file, it is important to examine the type of license to ensure proper use and to credit the original source on any documents as demonstrated in Figs. 5(a and b).

The third indicator for shore-based fishing was any area of publicly-accessible shoreline within 30 m of shore. We used distance buffering within ArcGIS ModelBuilder to include only areas within 30 m, and there was no need to download another indicator dataset.

7 Nature's Benefits in Nassau County

Our collaboration with Nassau County started with an initial list of 20 nature's benefits. Soliciting input from the anticipated end users of the maps was essential in delivering a useful product. Some of our initial choices of benefits, such as SCUBA diving and guided boat touring, were ruled out as requiring excessively subjective judgments to choose indicators. Others, like waterfowl hunting and seal habitat, were discarded because they were deemed less important. After two rounds of deliberation, we chose the following benefits: bay scallop habitat, hard clam collecting, shore fishing, offshore striped bass fishing, summer flounder fishing, vegetative wave attenuation, aquatic animal viewscapes, and yellow-crowned night heron habitat.

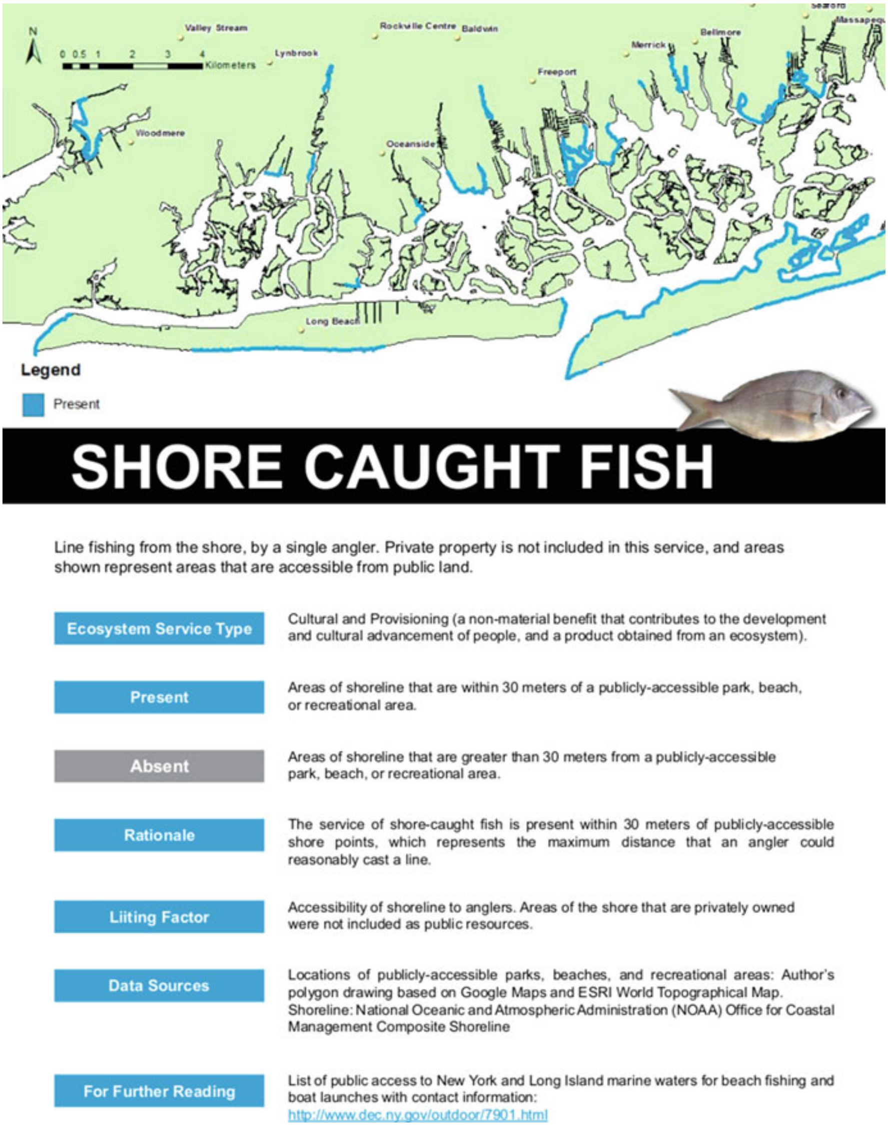

Fig. 5 (**a** and **b**) Nature's Benefits handouts for shore fishing and hard clam collecting

This stakeholder collaboration effort resulted in eight nature's benefit maps that illustrated a variety of benefits important to people who live on the Bay. We created 1-page handouts for each nature's benefit map (e.g., Figs. 5(a and b)), explained the indicators used to create it, and provided background information on the benefit along with further reading. Data sources were included with the example handouts, and a selection of the peer-reviewed literature that guided our choice of indicators

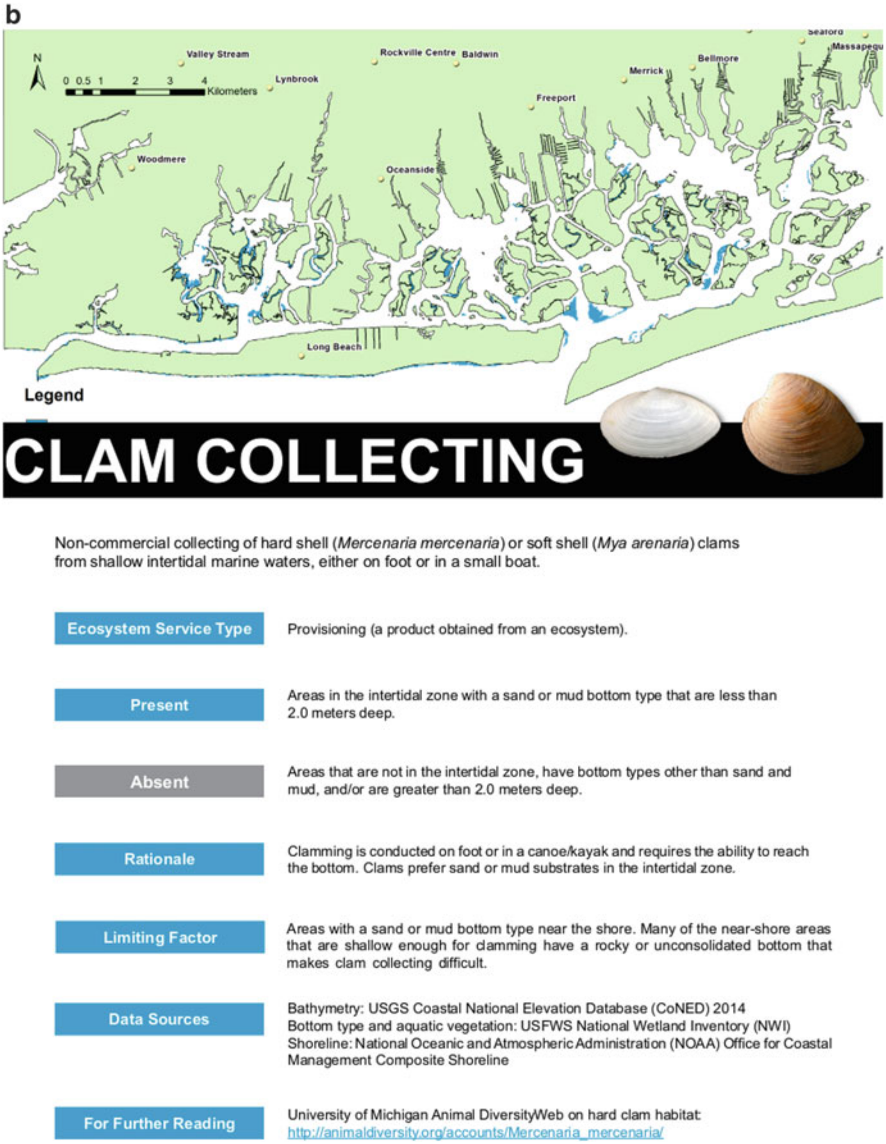

Fig. 5 (continued)

was listed in a "For Further Reading" section. We provided a comprehensive package to Nassau County that included the eight benefit handouts, all input files, Arc ModelBuilder files, a Further Reading document that included peer-reviewed literature supporting indicator choices, and two guidance documents. A summary was also provided with an illustrated step-by-step manual that guides a user through indicator election, creating new maps, and navigating the GIS software for making changes to the Nature's benefit maps.

8 Synthesis—Using Nature's Benefit Maps

The utility of nature's benefit maps is in their intuitive ease-of-use and their clarity of presentation. They are especially effective for communicating benefits to a non-scientific audience, including decision makers in local governments involved with planning and zoning where consideration of nature's benefits may reach a more sustainable and resilient solution. In our collaboration with the New York Department of State and Nassau County Department of Public Works, two use cases were prominent: (1) from a regional perspective, identifying areas of nature's benefit hotspots (i.e., multiple overlapping benefits); and (2) from an implementation or management perspective, identifying the most limiting factor determining the spatial extent of the benefit. Ecosystems, such as coastal emergent wetlands, that provide a high density of benefits can be managed for preservation or actively improved. For vegetative wave attenuation the limiting factor was the extent of emergent aquatic vegetation, rather than submerged aquatic vegetation or the extent of wetland, intertidal, and aquatic zones. Knowing the limiting factor allows planners to communicate how best to increase the amount of a nature's benefit, possibly increasing its spatial extent. Decision makers and the public both wanted to know where benefits were located and how to manage to potentially increase benefits. Nature's benefits mapping provides both information needs for decision support.

A moderate degree of skill with GIS is required. A user needs to know how to arrange files in the proper directories for ArcGIS, open and edit ModelBuilder, and must have some familiarity with GIS operations like clipping to be able to fully utilize the tool. ArcGIS is required to utilize ModelBuilder and replicate the examples here, but we decided this was acceptable because ArcGIS is in widespread use in municipal planning departments, such as the Nassau County government, who are the primary end-users. This approach is broadly transferable to other GIS platforms, including open-source software such as QGIS (https://qgis.org/en/site/). Maps are considered provisional, indicating the potential for benefits, unless fieldwork is done to confirm benefit presence and absence.

End-user guidance and feedback at each step (sometimes described as being "co-developed") was invaluable in delivering products the county could use. The handouts served as templates for other benefits, and we recommend that nature's benefit maps be used in the initial stages of planning, including developing and evaluating potential alternative scenarios for a given management effort. These modeling and mapping tools have applications in health impact assessments, environmental impact assessments, and municipal planning and zoning. Our emphasis was on making best use of publicly available data and translating these clearly for ease-of-use, including end-user modification and extension to serve a range of interests and needs.

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Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Ecosystem-Based Management to Support Conservation and Restoration Efforts in the Danube Basin

Andrea Funk, Timothy G. O'Higgins, Florian Borgwardt, Daniel Trauner, and Thomas Hein

Abstract Biodiversity and environmental integrity of river systems in the Danube catchment is threatened by multiple human alterations such as channelization, fragmentation or the disconnection of floodplains. Multiple human activities, including the construction of hydropower plants, expansion of agricultural use, and large-scale river regulation measures related to navigation and flood protection, are resulting in an ongoing loss of habitat, biodiversity and ecosystem service provision. Conservation and restoration of the systems biodiversity and ecosystem service provisioning is a key task for management but is challenging because the diversity of human activities and policy targets, scarcity of data compared to the complexity of the systems, heterogeneity of environmental problems and strong differences in socio-economic conditions along the Danube River hampers coordinated planning at the scale of the whole river basin and along the whole river from source to mouth. We evaluated three different implementations of an Ecosystem-Based Management (EBM) approach, which aims to support management efforts. This was done following the principles for EBM related to the resilience of ecosystems, the consideration of ecological and socio-economic concerns, the inclusion of multi-disciplinary knowledge and data addressing the ecosystem scale independent of administrative or political boundaries. This approach has been developed in the H2020 project AQUACROSS.

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Lessons Learned

- Coupled modelling frameworks are a useful tool for modelling biodiversity restoration measures
- Multiple policy targets can be harmonized with this approach

Needs to Advance EBM

- Continued international cooperation informed by costed measures

1 Introduction

The core principle of Ecosystem-Based Management (EBM) is to concurrently consider biodiversity and human society as integral parts of the ecosystem and manage the socio-ecological system as a whole (Domisch et al. 2019; Langhans et al. 2019). Delacámara et al. (2020) review the many ‘flavours’ of EBM to identify six characteristics or principles, which set EBM apart from other types of management:

1. It considers ecological integrity, biodiversity, resilience and ecosystem services
2. It is carried out at appropriate spatial scales
3. It develops and uses multi-disciplinary knowledge
4. It builds on social-ecological interactions, stakeholder participation and transparency
5. It supports policy coordination
6. It incorporates adaptive management.

While these EBM principles are not proscriptive, i.e. any particular EBM activity is not required to have all these characteristics, they may offer useful criteria by which EBM activities may be practically assessed.

The Danube River Basin (DRB) is the most international river basin in the world shared by more than 80 million people across 19 countries (Fig. 1). The Danube River connects with 27 large and over 300 small tributaries on its way from the Black Forest to the Black Sea, covering a catchment size of approx. 800,000 km^2.

As a result, a huge variety of human activities and related pressures affect this area and a number of major environmental issues threaten the ecosystems of the Danube. As Europe’s second longest river, the Danube has long been a major transport corridor. Today, it connects Europe’s largest port of Rotterdam with the Black Sea via the Rhine-Main-Danube canal. Physical modifications of the river morphology to accommodate transport and power production have altered flow regimes with serious consequences for ecosystems including the disconnection of the river from its natural flood plains. Agricultural activities along the Danube have resulted in pollution by nutrients and pesticides. The combined effects of these and other pressures have resulted in overall degradation of the freshwater ecosystems and severe declines in iconic species such as different sturgeon species. The

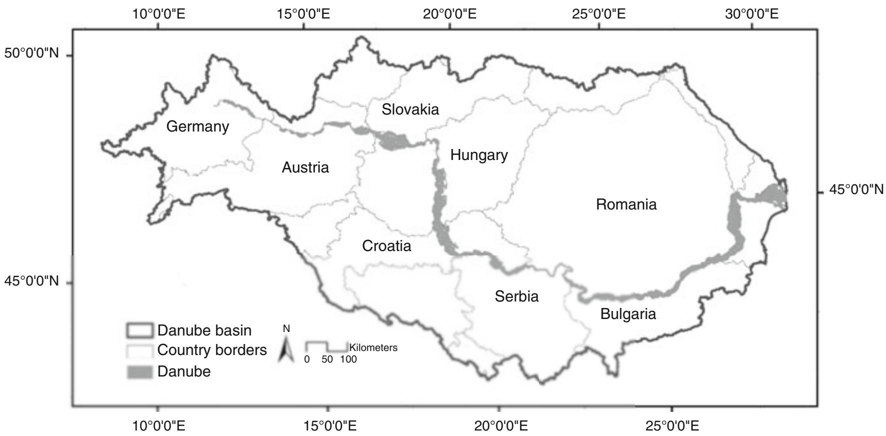


Fig. 1 The Danube River Basin and the corridor of the Danube river

International Commission for the Protection of the Danube River (ICPDR) provides a formal international mechanism for environmental management collaboration across the Danube Basin (detailed information on the many environmental issues can be found on their website (https://www.icpdr.org/main/).

Despite conservation efforts, ongoing and partially conflicting demands within and among the different neighboring countries, inconsistencies in legislation, high administrative and socioeconomic complexity as well as partially lack of on-site expert knowledge all hamper sustainable management (Hein et al. 2016, 2018; Habersack et al. 2016).

There are two major challenges for the management of the DRB. The multi-cultural setting makes transboundary issues extremely difficult and challenging. For example, the basin lies in the historical political border between capitalist and communist countries, which greatly influences the socio-economic situations, social behaviors, technical developments, as well as water uses and protections between the two former systems (Sommerwerk et al. 2010) and resulting in varying priorities towards, and capacities for, environmental protection (O'Higgins et al. 2014). In the DRB, this historical background is well reflected in the structural differences between the Upper Danube (capitalist countries) where hydro-morphological alteration is high but pollution is low, while in the Lower Danube (former communist countries) pollution is still a highly relevant issue but level of impact due to river engineering works is still relatively low (Sommerwerk et al. 2010). This phenomenon is also reflected in the ranking of stressors along the Danube River. Hein et al. (2018) found that for the Upper Danube hydro-morphological alterations due to hydropower generation, navigation, and flood protection has the highest importance followed by forestry, disturbance due to recreational activities, recreational fisheries and last by pollution, whereas the Lower Danube is mostly impacted by land use including forestry, agriculture and urbanization having an direct as well as an pollution effect on the system and last by hydro-morphological alterations of the river.

Table 1 Policies directives and initiatives with synergistic and antagonistic effects on conservation objectives in the DRB

Instrument type	Name	Targets and goals
Policy	EU Biodiversity Strategy	Full implementation of the Birds and Habitats Directives
	EU Strategy for the Danube Region	e.g. Sturgeon 2020 program for the protection and rehabilitation of sturgeon
Legally binding directives	Water Framework Directive (2000/60/EC; WFD)	Good Ecological Status—through implementation of the Danube River Basin Management Plan
	Flood Risk Directive	Danube Flood Risk Management plan
	Birds Directive (2009/147/EC)	Favorable conservation status (for selected species)
	Habitats Directive (92/43/EEC)	Favorable conservation status (for selected habitats)
	Renewable Energy Directive	Total of 20% of EU energy needs to be supplied by renewable sources (including hydro power).
Initiative	Trans-European Transport Network	Good navigability for important waterways, including the removal of obstacles

The second major challenge in DRB management is to establish synergies among multiple competing interests and policy targets including e.g. navigation, hydropower production, flood protection and nature conservation (Sommerwerk et al. 2010). Human stressors interact with the management goals of the Water Framework Directive (EC 2000) or Nature Directives (EC 1992) and the Biodiversity Strategy to 2020 (EC 2011), resulting in potential synergies and conflicts between the various management goals. The implementation of sectoral policies on hydropower (renewable energy), navigation, and flood protection may show significant synergies and antagonisms, and the interaction of their implementation significantly influences the actual type and extent of pressures on rivers. Table 1 lists some of the interrelated directives, policies and initiatives with specific relevance to the management of the Danube River and its associated ecosystems.

For example, the Flood Risk Directive (EC 2007) aims at reducing risk of flooding along water courses including natural water retention measures (e.g. dyke relocation to provide more space for rivers). Floodplains are therefore a key element of the EU Green Infrastructure Strategy (ICPDR 2016). Like-wise navigation projects might either have a synergistic effect on nature protection goals in already significantly altered river sections (if ecological restoration is supported within the project), or an antagonistic effect in intact river sections where every intervention may create a conflict with nature protection goals (DANUBEPARKS 2011). With a multitude of interacting environmental and other directives, management targets can have synergistic as well as antagonistic effects, which vary from place to place. Moreover, these interactions are complex and not sufficiently understood.

In this context, modern management concepts can neither exclusively focus on the mitigation of single pressures or stressors nor can they limit their measures to single ecosystem components, species groups or other single targets. In contrast, they have to consider complex interactions and feedback loops between the ecosystems and the society. Thus, for the future, explicit and well-defined ecosystem-based targets need to be formulated, and adequate measures need to be defined to achieve more resilient ecosystems, guarantee the provision of a broad range of ecosystem services, and increase the resilience against emerging stressors like climate change or invasive species (Hein et al. 2018). Given the need for holistic catchment scale management approaches (Hein et al. 2018; Seliger et al. 2016), EBM offers the potential to incorporate multiple objectives related to biodiversity, ecosystem services and socio-economic benefits into a single, harmonized management approach for the DRB. The Danube River, as one of the largest river-floodplain systems in Europe, is a highly complex, threatened and challenging socio-ecological system, and therefore an ideal system to test and apply an EBM approach. To this end, within the frame of the AQUACROSS research project a number of tools and techniques were combined and tested for application in the Danube catchment. In this paper we describe and discuss three different approaches and provide a qualitative assessment of how these methods relate to the EBM principles identified above.

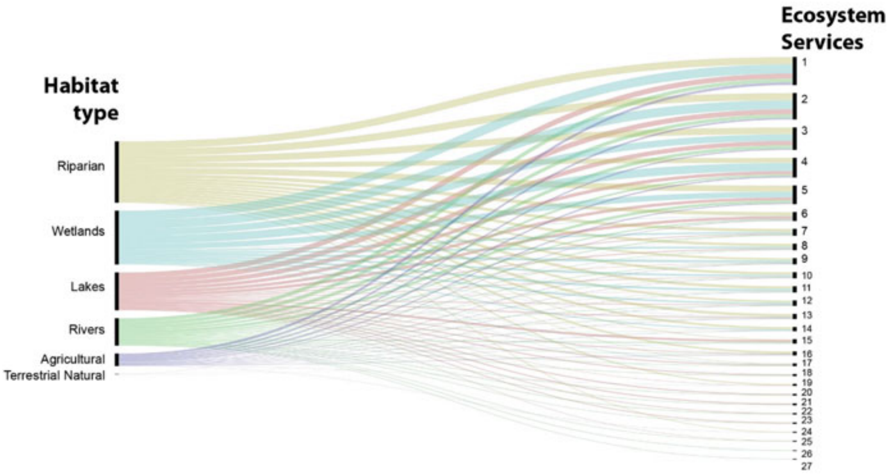
2 The Studies

Other authors in this volume (Fulford et al. 2020; Lewis et al. 2020) have addressed the challenges of model design and selection and the potential for combining models to address particular situations. We evaluate three different quantitative and qualitative approaches that have been applied at the Danube catchment scale to describe and model the socio-ecological system. A linkage framework approach (Borgwardt et al. 2019; Teixeira et al. 2019; Robinson & Culhane 2020) was used to assess the relationships between different activities within the catchment and their relations to biodiversity and ecosystem services. The potential of EBM was also tested within two quantitative studies following an EBM planning framework based on a generic model-coupling approach proposed by Langhans et al. (2019). The workflow consists of three elements a spatial (model-based) representation of (1) biodiversity, (2) ecosystem services (ESS), and (3) a combined spatial prioritization of biodiversity and ESS supply and demand.

Finally, Domisch et al. (2019) combined the ARIES (Artificial Intelligence for Ecosystem Services) modelling framework (Villa et al. 2014) with the application of MARXAN (Ball et al. 2009) to identify a range of spatially explicit management zones and options. Funk et al. (2019) combined Bayesian Belief Network Modelling with the ARIES model to identify river reaches maintaining multiple ecological functions and support multiple services to prioritize individual areas for conservation incorporating a range multiple restoration criteria.

2.1 Linkage Frameworks

A Linkage Framework (LF) for the Danube Basin (Fig. 2) identified 53 specific human activities (or Drivers) occurring in the catchment (Borgwardt et al. 2019). Furthermore, 35 different pressures in five different categories (biological, chemical, physical, energy, and exogenous/unmanaged) were identified, as well as 33 ecosystem components (27 habitats and 6 biotic groups). These components were linked to 27 ecosystem services (ESS) and abiotic outputs. Over 23,000 impact chains relating drivers-pressures and ecosystem components were identified and categorized. To investigate the impact chains, their connectance was calculated and linkages were also weighted in terms of the extent, frequency, dispersal, severity and persistence of interactions to increase their explanatory power. Analysis of the impact risk of



1. Scientific; education; heritage; aesthetic; entertainment
2. Existence; bequest
3. In-situ recreational activities involving biota
4. Lifecycle maintenance, habitat and gene pool protection
5. Mediation of waste by biota
6. Symbolic; sacred and/or religious
7. Atmospheric composition and climate regulation
8. Mediation of waste by ecosystems
9. Water conditions
10. Soil formation and composition
11. Mass flows (erosion protection)
12. liquid flows (flood protection)
13. Maintenance of physical and chemical abiotic condition by natural chemical and physical processes (e.g climager regulation)
14. Mediation of air flows by biota (shelter)
15. Provision of biomass for nutrition
16. Mediation of waste by physcial and chemical processes
17. In-situ recreational activities involving landscapes
18. Provision of biomass
19. Pest and disease control
20. Provision of drinking water water
21. Provision water for other purposes
22. Mediation of flows by natural abiotic structures
23. Spiritual and emblematic value of landscapes
24. Biomass based energy sources
25. Existence and beqquest values for landsacpes and phyical settings
26. Abiotic reneable energy (eg. hydroelectricity)
27. Abstraction of rocks and minerals

Fig. 2 Flow diagram of the linkage framework depicting impact chains from habitat type to ecosystem services pressures on ecosystem components revealed that physical change poses the highest threat to freshwater systems and to fish. Physical pressures are highly linked to environmental engineering and hydropower but also to the direct effects of land claim or land conversion activities (Borgwardt et al. 2019). Further along the impact chain, the ecosystem components within the Danube catchment were identified to have the capacity to supply 27 ESS (regulating and maintenance, provisioning, and cultural services and abiotic). Floodplains with their riparian forests and wetlands were the highest connected realms providing the greatest variety of ecosystem services.

2.2 Coupled Models: ARIES and MARXAN

Domisch et al. 2019 tested the EBM approach within the whole DRB by combining species distribution modelling for 85 fish species as a surrogate for biodiversity with four estimated ESS layers (carbon storage, flood regulation, recreation and water use) using the modeling platform ARIES. In a final step, multiple management zones were defined using the spatial prioritization tool Marxan with Zones to derive different spatially explicit management options for the whole region. In order to explore the transboundary challenges of the Danube catchment management the costs of establishing management zones were compared across nations using purchasing power parity (PPP) adjusted gross domestic product (GDP) per capita and the relative share of each country's area of the DRB. This approach therefore accounted for countries having limited financial resources (i.e. a proxy for social equity in the EBM approach) and less land area in the DRB as those might face additional challenges in financing EBM. Finally, they compared the spatial plan derived from an assumption where each country contributes equally to the EBM to one where the PPP-adjusted GDP and the percent area of each country in the basin were used as additional costs. The two analyses led to clear differences in the spatial configuration of management zones, in the GDP and percent area approach more conservation and critical management zones (with medium level of ecosystem service use) were allocated to the (wealthier) upper Danube region.

Domisch et al. 2019 used Marxan with Zones, to minimize the overall costs of a zoning plan, while ensuring that the predefined feature targets were met. Therefore four zones were characterized by different objectives and constraints (1) a "focal conservation zone", (2) a "critical management zone"—a buffer zone—, (3) a "catchment management zone" allowing for higher levels of ESS use potentially less compatible with protecting biodiversity (i.e., recreation), and (iv) a "production" zone with high use for ecosystem services (i.e., water use).

2.3 Coupled Models: Bayesian Belief Networks and ARIES

Funk et al. (2019) employed a coupled modelling approach at the scale of the Danube River to prioritize river-floodplain stretches of the navigable Danube for restoration and conservation, focusing on the river and its adjacent floodplains and riparian area (rather than the entire catchment). Bayesian Belief Networks (BBN: graphical models which represent the probabilistic relationships between different components of a system) were used to integrate different sources of information on Drivers and Pressures and their effects on environmental State (Elliott & O'Higgins, 2020). Open access GIS Datasets for Drivers and pressures included: land use data, potential riparian zone transport and navigation, and hydro-morphological pressures. This information was then used to inform weighting of the relationships within the BBNs.

Based on spatial information on conservation status based on the Habitats Directive reporting, BBNs were generated to spatially model likely species distribution in relation to the combinations of drivers and pressures for each of eleven indicator species representative of different habitat types (Table 2). The predictive power of these BBN models was tested statistically (using the R statistical computing package (see Funk et al for full details).

Table 2 Biodiversity indicators used by Funk et al. (2019)

Class	Species	Common name	Indicator
Fish	*Gymnocephalus baloni*	Danube ruffe	Fast moving waters
	Gymnocephalus schraetser	striped ruffe	Main stem larger river
	Rhodeus amarus	European bitterling	Stagnant water-connectivity
	Misgurnus fossilis	European weatherfish	Stagnant waters
	Zingel zingel	common zingel	Main stem large rivers, connected side arms
	Zingel streber	Danube streber	Main stem small to large rivers, connected sider arms
Amphibian	*Bombina sp.*	Fire-bellied toads	Fish free seasonal ponds
	Triturus dobrogicus	Danube crested newt	Temporary water bodies
Bird	*Haliaeetus albicilla*	White tailed eagle	Undisturbed wetlands
	Alcedo atthis	Common kingfisher	Active erosion and natural substrate
Mammal	*Lutra lutra*	Otter	Overall natural habitat conditions

Spatial mapping of ESS was conducted using the ARIES pollination, recreation and flood models submodels.

A spatial database combining the ARIES outputs with the outputs of the probabilistic species modelling was interrogated using clustering to identify multi-functional river and flood plain reaches supporting biodiversity and ESS supply. These multi-functional clusters were then mapped.

The model used a multi-objective optimization tool (e.g. Sacchelli et al. 2013), which enabled systematic optimization for different management objectives. One objective was to prioritize sections for conservation or restoration with a high remaining multi-functionality to reduce effort and costs, a second objective was to prefer sites with high reversibility (i.e. low level of human use) to increase probability of success, and finally to prefer semi-natural areas to reduce costs and loss of agricultural yield. Different weightings of the three objectives represent different possible management plans and therefore can be used as a basis for a more integrated and targeted planning. This process resulted in the development of a suite of potential target areas for restoration, conservation or mitigation efforts.

Consistent with other studies (Egoh et al. 2011; Maes et al. 2012), Funk et al. (2019) recorded a high overlap between areas important for biodiversity and areas important for ESS supply, pointing to a close interrelationship between biodiversity and ESS that is often greater in natural systems (Chan et al. 2011; Schneiders et al. 2012). Specifically, the multi-functionality approach tested by Funk et al. (2019) showed that in the study area, only natural and near-natural river-floodplain systems provided habitat for various aquatic species as well as multiple ESS.

In the study, sites with greater probability of restoration success, indicated by low level of driver intensity related to navigation, hydropower and flood protection constraints as well as sites with high level of remaining semi-natural area (compared to agricultural area) were prioritized. In this way the study addressed potential opportunity costs of restoration efforts across the entire Danube River. This approach afforded the ability to provide better cost-effectiveness in achieving large scale conservation and ESS targets at the catchment scale (Bladt et al. 2009; Egoh et al. 2014), and to potentially avoid conflicts with drivers.

3 EBM Principles

Overall the application of the LF to the Danube Basin, illustrated the complexity of interactions between human activities, ecosystem components and the ESS they provide, and is useful in identifying the most important ecosystem components with respect to ESS supply as well as the types of activities that most likely affect these components through pressures. With respect to the EBM principles, the LF can support the **first principle** in terms of communicating the links between ecological integrity, biodiversity (expressed at the habitat level) and ESS. The LF is not spatially explicit and can be transferred and adapted for use at in any similar system and applied to any spatial scale of interest thereby supporting the **second principle** (appropriate spatial scales) of EBM. The LFs are developed by 'experts' on a given location, who assess the activities and pressures, based on their knowledge. While LFs require an holistic view of a system, they do not necessarily integrate insights from a range of disciplines (**principle 3**) rather they characterize a suite of social-ecological interactions (**principle 4**). In its capacity to foster an understanding of the complexity of these links to promote understanding of policy synergies, they may also be used to facilitate and support policy coordination (**principle 5**). However, because the LF is a semi-quantitative and expert judgement based approach it is unlikely to carry sufficient confidence to justify any particular policy decision. Since the LF does not identify particular management options its current role in adaptive management (**principle 6**) is limited. Nevertheless, with its basis in the causal chain analysis of the DPSIR (see Elliott this volume) the linkages could potentially be extended to incorporate response options. For fully detailed accounts of development and analysis of the LF and comparison across regions, and aquatic ecosystem types, the reader is directed to Borgwardt et al. 2019, Teixeira et al. 2019, for a general description and discussion of the approach see Robinson and Culhane (2020).

The two integrated modelling studies (Domisch et al. 2019; Funk et al. 2019) exemplify how different holistic approaches can be used to identify management options which consider ecological integrity biodiversity resilience and ESS (**Principle 1**). Both implementations of the quantitative model coupling framework for EBM (Langhans et al. 2019), confirms how biodiversity and ESS estimates can be jointly simulated within the DRB given the availability of requisite data and models. It demonstrates that the method is very flexible and the criteria and models used are broadly applicable and the approach is transferable to other aquatic systems (Funk et al. 2019, Domisch et al. 2019).

Both approaches were spatially explicit and developed specifically to work at the appropriate spatial scales (**principle 2**). In the first study (Domisch et al. 2019) this included the entire catchment while the second study (Funk et al. 2019) had a more restricted focus specifically on rivers and the flood plain, nevertheless both studies worked across international borders which is a prerequisite for the work in the Danube.

Both model used a range of data sources, in particular Domisch et al. (2019) used truly multi-disciplinary, economic and environmental data (**principle 3**) to account for economic disparity, within the social part of the social-ecological system. This approach accounts for countries having limited financial resources (i.e. a proxy for social equity in the EBM approach) and land area in the Danube River Basin as those might face additional challenges in financing EBM in the basin.

In contrast, Funk et al. (2019) selected a method indirectly accounting for costs independent from country level's financial limitations, prioritizing sites with greater probability of restoration success at lower cost (i.e. indicated as lower loss of agricultural area). Therefore the multi-functionality approach accounts for the emerging view that ecological restoration requires restoring ecosystems for the sustainable and simultaneously provisioning of multiple goods and services such as water, flood protection, recreation, and biodiversity, among others to increase cost-effectiveness (Paschke et al. 2019).

One potential pitfall with both approaches is the stakeholder participation and transparency (**Principle 4**). Neither study directly used stakeholder input to inform the model building process, rather, the choices were made at the technical level by the modelling teams. To make the approach operational, participatory processes involving stakeholders across the catchment, member state and local levels would be a further important step. BBNs in particular are one promising technique which can be easily adapted to incorporate stakeholder input. It is possible to construct BBNs models based on stakeholder perceptions allowing co-design of modelling activities (see O'Higgins et al. 2020 for an example). In addition, the use of the AI approach included in the ARIES model may lack the transparency of more traditional deterministic environmental models, which may reduce the acceptability of model results. Elsewhere in this volume Fulford et al. 2020 discuss practical trade-offs inherent in model complexity.

Both the policy coordination potential (**principle 5**) and the adaptive management aspects (**principle 6**) are strong in both studies described above. Outputs from both models produced a suite of policy-relevant options enabling joint efforts to conserve the Danube.

Funk et al. 2019 accounted for this principle by using data and knowledge derived and used in the framework of different policies, directives and initiatives e.g. navigation and hydropower sector (e.g. TEN-T regulation), water management sector (Water Framework Directive), local data from protected areas (Birds and Habitats Directive) and spatial land use information. This includes a continuous hydro-morphological assessment for the navigable Danube River compliant with CEN standards (Schwarz 2014; ICPDR 2015), Land cover/Land use (developed to support e.g. EU Biodiversity Strategy to 2020) or sectoral data collected on the status of the waterway, critical locations for navigation and navigation class (Fairway 2016). Cause-effect relations within the network of interactions between driver, pressure and state variables along the Driver-Pressure-State chain were then analysed within a quantitative Bayesian Network approach. Therefore, the approach selected by Funk et al. 2019 provides the first large scale statistical proof of multiple relationships of biodiversity and human uses and pressures along the navigable stretch of the Danube River. Therefore, it has the potential to increase knowledge on the socio-ecological system across sectors and policies and is serving as a basis for a strategic and more integrated management approach.

The Domisch et al. (2019) study explicitly included consideration of regional inequalities and economic capacity and generated a more in-depth picture of the feasibility of particular conservation efforts, thus enabling the adaptation of plans to meet these real-world social constraints.

4 Conclusions

We developed and tested different qualitative and quantitative implementations of an EBM approach for a complex socio-ecological system, the DRB. The LF approach helped to understand the complex interaction within the social-ecological system and to describe the main human activities and pressures affecting the aquatic ecosystem components. The modelling approaches summarized in this paper have increased the consideration of ecological integrity and biodiversity, accounting for multiple species and different relevant ESS. These studies illustrate approaches considering cumulative impacts by multiple human activities including land use, navigation and hydropower and integrate this multidisciplinary data and knowledge. The prioritization approaches taken fosters integrated management planning across multiple policies by creating the opportunity to pursue different policy objectives simultaneously.

All three selected EBM application for the DRB were implemented at the ecosystem scale i.e. including the whole catchment or river independent of jurisdictional, administrative or political boundaries (Borgwardt et al. 2019, Domisch et al. 2019, Funk et al. 2019) and therefore have the potential to foster transboundary cooperation for a EBM of the DRB.

Both implementations of the quantitative model coupling framework for EBM (Langhans et al. 2019), showed how biodiversity and ESS estimates can be jointly simulated within the Danube River Basin given the availability of requisite data and models. This demonstrates that the method is flexible, the criteria and models used are broadly applicable, and the approach is transferable to other aquatic systems (Funk et al. 2019, Domisch et al. 2019). The EBM principles used for qualitative assessment of the modelling approaches may serve as a useful generic basis for the design of further EBM studies.

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Combining Methods to Establish Potential Management Measures for Invasive Species *Elodea nutallii* in Lough Erne Northern Ireland

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Abstract Lough Erne (Northern Ireland) is a heavily modified water body in a transnational catchment, straddling Northern Ireland (in the U.K.) and the Republic of Ireland. The lake has a long history of human modification from hydro-electrification, eutrophication and the introduction of non-native species. Most recently the proliferation of the non-native pond weed *Elodea nutalli* has adverse implications for recreational users of the lake. In order to establish management measures which might be acceptable to a range of lake, a number of methods, using a mixture of disciplines were combined. Fuzzy cognitive mapping exercises combined with formal goal identification surveys were conducted to establish consensus on the main environmental problems and conflicts. GIS was used to visualise potential management scenarios. Management scenarios were costed and presented to lake users to establish the preferred measures. The overall process promoted discussion and awareness of different uses and user perspectives to enable development of consensus.

Lessons Learned

- Loose coupling of models provided a useful means of analysing the system in a data poor situation
- Co-design of the models enabled the development of consensus
- Visual representation through maps and graphs enabled the communication of complexity
- Including the full suite of stakeholders is ideal but we were unable to compel unwilling stakeholders

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- Mechanisms to promote improved farm management have the potential to enable solutions where both farmers and the environment benefit.

Needs to advance EBM

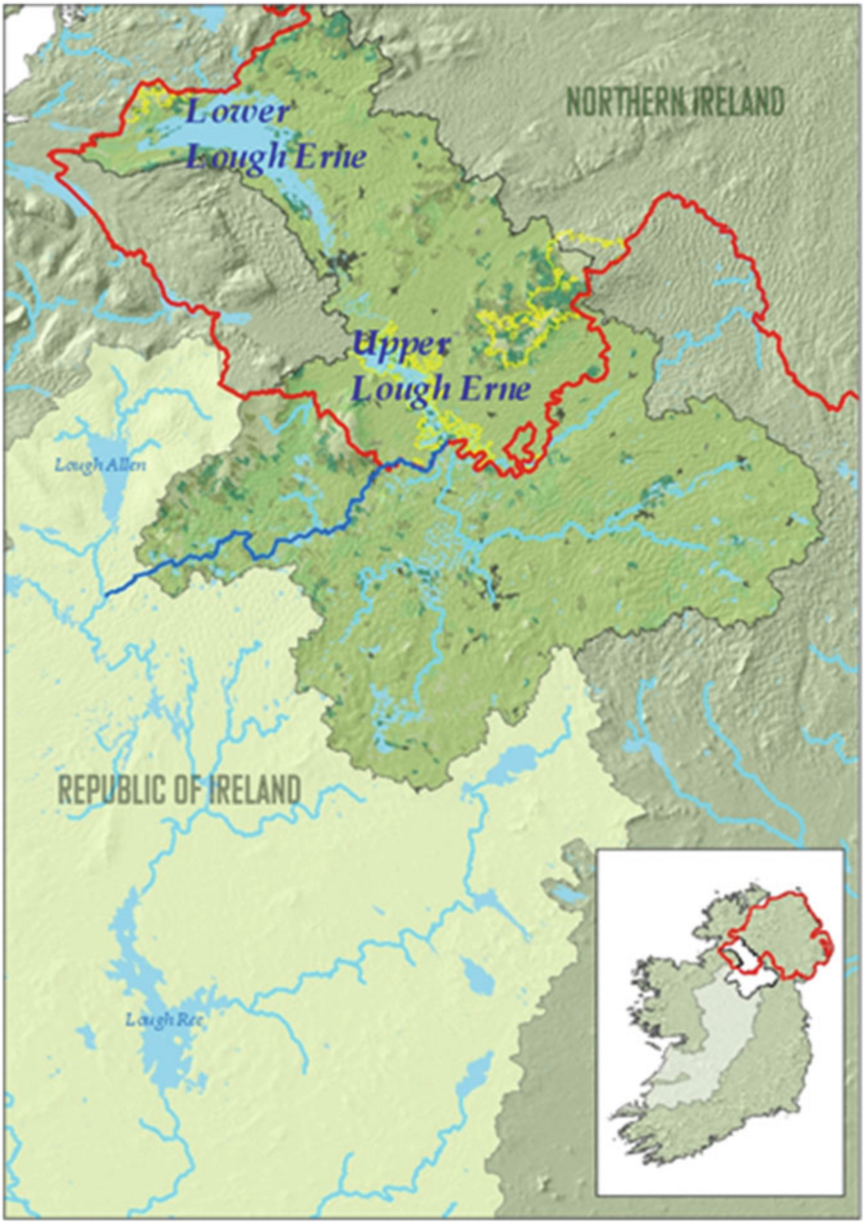
- In the context of the Erne as a transboundary system the importance of developing inclusive stakeholder fora to enable transboundary cooperation is one important requirement.
- More integrated quantitative modelling of the hydrological system and the nutrient fluxes could enhance the evidence base for action.
- Development of socially inclusive stakeholder processes.

1 Introduction

Ecosystem-Based Management (EBM) has been defined as an approach to management which "*integrates the connections between land air water and all living things, including human beings and their institutions*" (Mee et al. 2015), as such EBM in a particular location must incorporate considerations of both social and ecological aspects of that Social-Ecological System (SES). For any given system, what may be ecologically desirable to some, may be socially unacceptable to others, and different management options may result in costs or benefits to different sectors in society resulting in trade-offs or conflicts. In transboundary systems these costs and benefits may accrue in different jurisdictions, resulting in governance challenges and adding a further layer of complexity to effective management of a particular problem. In addition, achievement of socially desirable management end points may be constrained by the ecological properties of a given system. Very often there is a great degree of uncertainty about how effective a particular management measure might be, resulting in the need for adaptive management or a learning by doing approach (Holling 1978). Thus developing realistic and appropriate management strategies and targets relies on knowledge of both the ecological functioning of a system and the objectives of different users of a system, including the legislative objectives for environmental state and economic development.

The spread of Invasive Alien Species is considered a major threat to biodiversity the UN Convention on Biodiversity Strategic Plan for biodiversity (CBD 2014). In Europe, under the regulation of Invasive Alien Species (IAS) (EC 2014), a list of IAS of union concern has been drawn. The regulation forbids the transport or trade of the listed species. Member States are required to set out action plans on the pathways of invasion and to put in place surveillance systems and effective management measures for those species found to be widespread. Where an ecosystem is deemed to be "degraded, damaged or destroyed" the ecosystem must be restored unless costs of restoration are disproportionately high compared to the benefits.

Lough Erne, Co. Fermanagh Northern Ireland (NI), is comprised of Upper Lough Erne and Lower Lough Erne, both widened channels of Ireland's second largest river, the Erne. The lakes are in Northern Ireland but a substantial part of the catchment is situated within the Republic of Ireland (Fig. 1) and the region has a legacy of social and cultural division and conflict. The Erne is also connected to the



Map of Lough Erne showing the international border (red) as well as the Catchment (inset white) and the its connection with the Shannon catchment (inset light green)

Fig. 1 Map of the study area, showing Upper Lough Erne SAC and international border

Shannon river basin (the largest on the island) by the Shannon Erne Waterway. Due to this international connectivity, management of invasive species in Lough Erne is critical to the management of freshwater biodiversity across the entire island of Ireland.

The Lough Erne system has been settled since neolithic times and the ecology of the lake has been shaped by human society for millennia (Lafferty et al. 2006). References to the Erne fisheries date back to mythology and the oral tradition of early Christian times (Went 1945). In so far as a natural state can be determined for the Erne system, the fish fauna of the lake is naturally depauperate, comprised of post-glacial relic species (Salmonids, pollan and eel) supplemented by historically introduced species, including, bream, perch and pike (Rosell 2001).[1] Scientific records of non-native species date back to the late nineteenth century when Canadian pond weed (*Elodea canadenis*) was first recorded, (Moore and More 1866). Upper Lough Erne is particularly prized for its flora and fauna having several national and international environmental designations. Agriculture is also vitally important in the surrounding catchment and the lake has a history of eutrophication associated with agriculture in the catchment and associated fertilizer runoff (Battarbee 1986). It is currently considered moderately eutrophic under the EU Water Framework Directive (EC 2000). The lake supports a wide range of recreational activities, in particular boating and fishing are major contributors to the local tourist industry. Since the 1950s, the lake has been harnessed for hydro-electricity production. When constructed, the hydro power stations provided benefits to the government of the Republic of Ireland in their drive for rural electrification and to the people of Northern Ireland in the management of flooding. At the time, the Erne Drainage and Development Act (1950) which enabled the construction was an unprecedented example of cross-border cooperation.

There have been a number of recent invasive species introductions to the lake (Gallagher et al. 2015, Minchin et al. 2016) with the most significant being the zebra mussel (*Dreisenna polymorpha*) (Rosell et al. 1999; Maguire et al. 2006). The recent proliferation of Nutall's pond weed (*Eldoea nuttalli*) is a particular problem for tourism because it interferes with recreational boating and fishing. The growth of the pond weed is facilitated by the high nutrient levels of the lake waters and exacerbated by the high water transparency caused by the filter feeding of the zebra mussel (Kelly et al. 2015). This physical removal of the already established weed is costly (Kelly et al. 2013) and has proved ineffective in controlling its spread.

The aim of this paper is to describe the application of a flexible multi-disciplinary methodology in Lough Erne to demonstrate and communicate the utility of EBM practices to meet the needs of lake users and enable development and communication of Ecosystem-Based Management objectives and trade-offs which meet the needs of lake users and mangers within the system.

[1] Pike are known in the Irish language as "Gall Iasc" translating to "French" or "Foreign Fish" suggesting a Norman origin in Ireland's waters (ca. 1000 bp).

2 Materials and Methods

The overall approach to EBM was based on the "butterfly" assessment framework which is detailed in Elliott and O'Higgins (2020). An initial scoping meeting was held with stakeholders in the Lough Erne Invasive Species Working Group (LEISWG) (an informal collective of interested parties including several government agencies) and a second stakeholder workshop was held with participation from a variety of organisations from both Northern Ireland and the Republic of Ireland, including state agencies, national and local government as well as non-governmental organisations. The formal analysis of stakeholder goals at this meeting is described in Robinson et al. (2019) and the goals identified through that analysis formed the basis of a Fuzzy Cognitive Mapping exercise.

Fuzzy Cognitive Maps (FCM) are semi-quantitative models of system operation based on an individual's/individual stakeholder group perception of the structure and function of a given problem or system. An FCM is a diagraph or directed graph made up of variables (points, nodes or concepts), and relationships between these concepts (links or edges), or, put simply boxes and weighted arrows. Positive or negative values are assigned to these relationships and expressed as a fraction of one based on the perceived strength of the relationship. FCM has been widely applied to a range of situations; it can be used to build models of system behaviour based on expert opinion and can be used to build consensus amongst stakeholders, as well as to develop predictions for system function based on scenarios. Özesmi and Özesmi (2004) describe the mathematical aspects as well as a range of different approaches to developing FCMs.

Five separate Lough Erne stakeholder groups participated at a workshop to produce FCMs; these included, environmental NGOs and conservation groups, water managers, hydro-electricity producers and wildfowling groups as well as local government organisations. While there was high level participation by the Northern Ireland Department of Agriculture, despite invitation, farmers groups themselves chose not to attend. The DPSIR (see Elliott and O'Higgins 2020) was used as an organisational frame to elicit concepts and connections from participants. Each FCM was generated by starting with a particular Driver within the SES which interfered with the objectives of specific groups (see Robinson et al. 2019), the specific components were agreed by the groups and acted as a starting point for the FCMs. Each FCM was written on a whiteboard and relationships between all concepts identified were considered and assigned a positive or negative weight. Following the workshop the FCMs were photographed and then rendered electronically using Mental Modeller software (http://www.mentalmodeler.org/) before export. Matrices, output from Mental Modeller, from each group were combined (in Microsoft Excel) to develop a joint matrix representing the overall FCM of the whole group (called the JOINT FCM). The open source software GEPHI (https://gephi.org/) was used to visualise the data. Analysis of the FCM was carried out in R using the FCM modeller library (https://CRAN.R-project.org/package=FCMapper). In order to develop a consensus map, it was necessary to harmonise concepts within the maps. For example, concepts such as "*fish stock salmonids*" and "*game fish*" were amalgamated as were "*fish stock cyprinids*" and "*coarse fish*". To generate a final consensus map, the weight of each connection was determined by summing the weights of all connections from each contributing map. Only consensus connections from one or more groups contributed to the final map, reducing the complexity of the model from the 55 nodes of the original combined model to a final consensus model containing just eleven nodes. The baseline conditions general direction of change based on stakeholder understanding were determined by allowing the model to run to steady state. To assess the sensitivity of the systems to changes in different components 11 different model runs were performed, for each, one of the eleven model components was fixed to its initial state and the effects on all other components were examined (Table 1).

Based on the consultation process with the LEISWG, specific measures to manage the pond weed by manipulation of lake levels were identified. These measures were designed to manage the impact of pond weed proliferation on recreational activities within the lake, by controlling the amount of light available to the weed as well as increasing the draft between recreational vessels and the weed (Fig. 2). The spatial consequences of these measures were assessed using GIS. The EURODEM Digital Elevation Model (horizontal resolution 25 m) GIS was used to identify cells within a 5 km distance of the Lough with elevations marginally greater than the lake level. The Erne Drainage and Development Act (1950) stipulates strict limits for the lake levels during the summer season, between 150 ft. (45.7 m) and 154 ft. (46.9 m) above sea level), a range of approximately 1.2 m. The location and extent of potentially flooded lands was simulated at 5 increments from 0.2 m to a level of 1.2 m above the lake levels of the Digital Elevation Model, corresponding to the legally determined limits of lake water levels which are legally determined.

Table 1 Model components showing positive or negative changes >1% in the sensitivity analysis, affector components and the size of effect

Component	Affector	Effect
	Components	%
Water quality	Agriculture	−18
	Forestry	−15
	IAS	−14
	Habitats	11
	Tourism	−1
Habitat	Water quality	9
	Conservation	9
Biodiversity	Water quality	6
	Habitat	4
Tourism	Water quality	2
	Conservation	1
	IAS	−1
Agriculture	Conservation	2
	Flood management	2
Flood management	Habitat	3

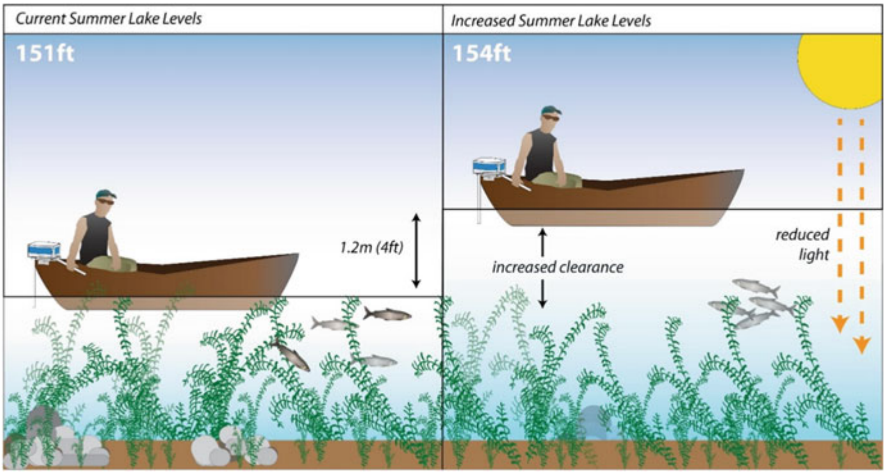


Fig. 2 Illustration of how raised lake levels could result in potential improvements for recreational boating

Economic valuation was used to appraise the costs of potential management measures. The costs to agriculture of raising lake levels were estimated in terms of annual standard output from the NI annual farm census and in terms of land value based on compulsory purchase price.

Based on the results of the fuzzy cognitive mapping exercise, valuation of a variety of agricultural measures to reduce nutrients was also performed, benefits transfer was based on Cuttle et al. (2007), these authors reviewed a range of farm Best Management Practices (BMPs) to reduce diffuse water pollution from agriculture, describing in detail the costs and technical effectiveness of each measure. Using a Cost Effectiveness Analysis (CEA) method, BMPs that could be implemented at least cost for the farmer while maximising potential Phosphorus (P) reductions were identified. The cost curve method was subsequently applied (Lago 2009) to estimate levels of abatement that could be potentially achieved as BMPs are sequentially added at farm level while considering their financial costs. The costs were calculated for two targets, 30% reduction in nutrient concentrations and 70% reduction in nutrients at the farm level under the assumption that the reductions would translate proportionally into improvements in water quality.

3 Results

Figure 3 illustrates the results of the joint FCM showing all 55 Drivers, Pressures and Ecosystem components considered by stakeholders to contribute to the functioning of the system. The concepts are scaled by the number of connections (known as density in the language of FCM). It is immediately apparent that agriculture, tourism

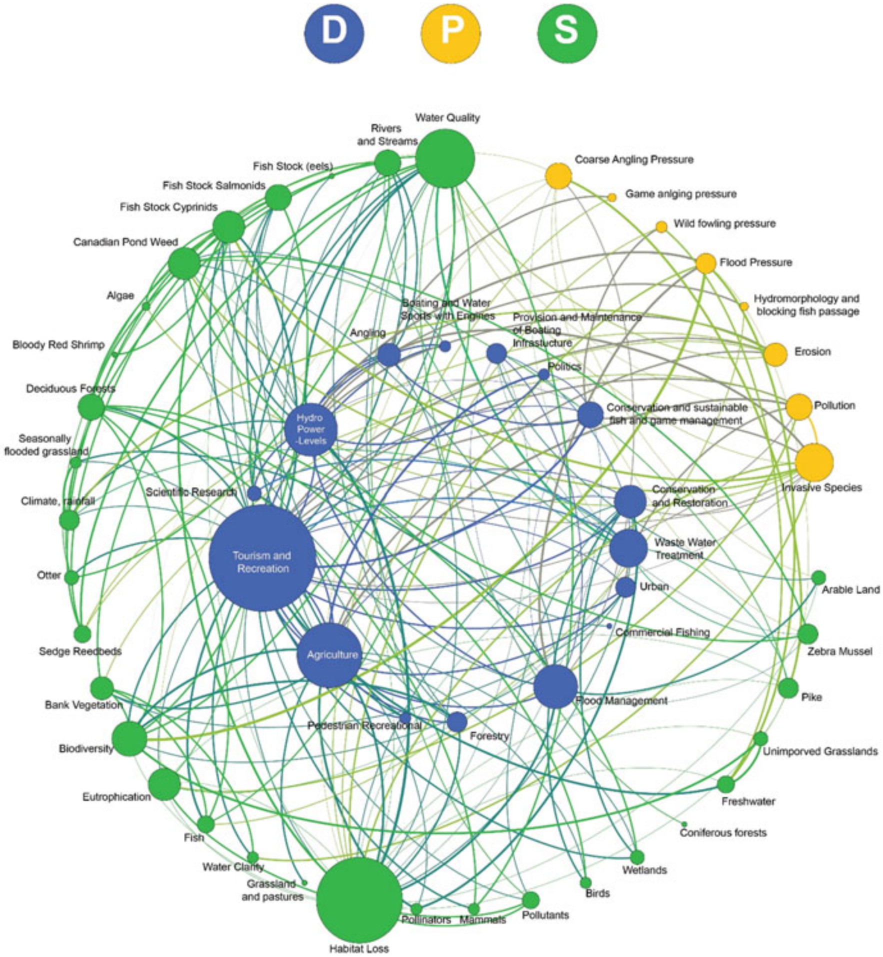


Fig. 3 Combined Fuzzy Cognitive map showing all concepts (*nodes*, *circles*) identified and relationships (*lines*, *edges*) between them based on the stakeholder workshop. Concepts are scaled according to the number of relationships with other concepts. Colours represent different components of the DPSIR framework. D = Drivers, P = Pressures, S = State (See Elliott and O'Higgins 2020)

and recreation and water quality are highly connected. Flood management, hydropower and invasive species are also seen to be highly connected to other components. Table 1 summaries the results of the sensitivity analysis conducted on the consensus FCM. Overall the model component which was most sensitive to alterations in the system was water quality showing relatively large responses to agriculture, forestry, IAS and tourism. The negative relationship between agriculture and water quality represents the strongest interaction in the whole model. Habitats had the strongest positive affect on water quality. In turn water quality had positive effects on habitat and biodiversity. The model illustrates the central importance of water quality to stakeholders, and the range of effects that water quality plays including positive effects on Habitat, Biodiversity and tourism, as well as its sensitivity to a range of drivers, but in particular agriculture. On this basis, the strong negative relationship between agriculture and water quality was used as the basis for identification of potential mitigation measures and the economic costings.

The adjustment of lake levels, which emerged as a potential management option for control of *Elodea*, would result in costs to farmers due to the inundation of productive agricultural lands and does not address the water quality of the Lough specifically. Inundation of agricultural land may also produce co-benefits in terms of biodiversity by increasing the area of semi natural riparian habitats. Maintaining the Lough at higher levels during summer may also result in benefits to the hydro-production sector enabling increased generation capacity. Figure 4 shows the area of land inundated by raising water levels by 1.2 m as well as the marginal changes in area of land inundated. The overall cost of compulsory purchase of the potentially inundated areas was just over £2 m. In the absence of compulsory purchase the total costs to farmers in terms of lost annual productivity due to inundation was under £0.5 m.

A range of nutrient abatement measures were also considered. There are many potential mechanisms to decouple agricultural activity from water quality impacts. The economic analysis summarised in Table 2 illustrates how a number of cost saving BMPs (negative costs) can save money to individual farms while also contributing to reduced nutrient loading. The target of 30% reduction in P loading can be met by implementing the first 6 measures sequentially with an overall cost of £15 m for the whole catchment.

4 Discussion

A range of techniques were employed to understand the Lough Erne SES, these included fuzzy cognitive mapping to combine individual stakeholder groups perception of the systems into an agreed and dynamic model of system behaviour, and more mechanistic GIS-based modelling approach to understand the effects of specific management measures on other activities within the catchment, as well as valuation of potential measures. In combination, these methods revealed a system which is highly complex and where incomplete knowledge is the rule. Nevertheless, the combination of simple techniques in co-design with stakeholders enabled the development of a consensus view of the system identifying eutrophication as a priority problem, with a range of potential measures for management of pond weed in the system also being identified. Ultimately the potential measures were reduced to a single metric of cost effectiveness to enable stakeholders in Lough Erne to consider the relative merits of the measures identified.

The problem of management of *Elodea* in Lough Erne is not a simple one. The problems of eutrophication are well understood to be an underlying cause for the

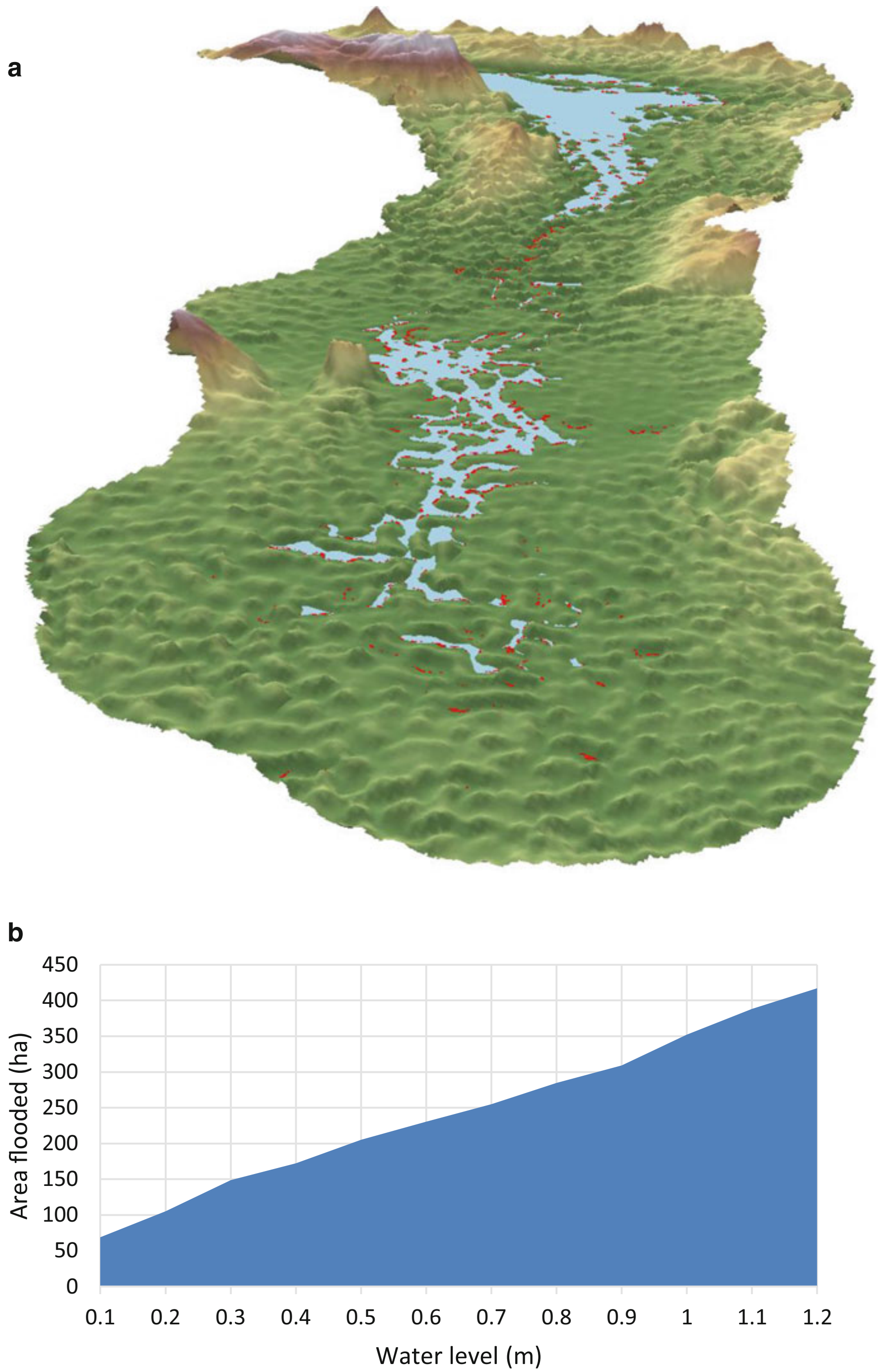


Fig. 4 (**a**) 3D visualisation of modelled areas of inundation based on raising the lake level by 1.2 m (**b**) cumulative area of inundation with increasing water level

Table 2 Cost Efficiency (CE) and percentage reduction in Phosphorus from a range of agricultural Best Practice Measures

Nutrient reduction measure		CE ratio[a]	P Loss[b]
1	Integrate fertiliser and manure nutrient supply	−472.44	0.10
2	Reduce fertiliser application rates; 20% Reduction P	−5.25	3.10
3	Do not apply P fertilisers to high P index soils	−2.62	6.00
4	Do not spread farmyard manure to fields at high-risk times	0.98	21.04
5	Do not apply manure to high-risk areas	2.25	26.57
6	Transport manure to neighbouring farms 5 km	2.69	56.68
7	Establish and maintain artificial (constructed) wetlands	2.86	74.87
8	Re-site gateways away from high-risk areas	3.17	75.63
9	Use a fertiliser recommendation system	5.25	76.36
10	Site solid manure heaps away from watercourses and field drains	5.25	77.07
11	Transport manure to neighbouring farms 20 km	5.57	86.47
12	Move feed and water troughs at regular intervals	5.85	88.36
13	Do not apply fertiliser to high risk areas	7.87	88.83
14	Site solid manure heaps on concrete and collect the effluent	22.87	89.16
15	Avoid spreading fertiliser to fields at high-risk times	27.36	89.38
16	Reduce field stocking rates when soils are wet	28.43	90.34
17	Fence off rivers and streams from livestock	38.40	90.53
18	Establish riparian buffer strips	38.40	90.72
19	Reduce overall stocking rates on livestock farms	54.05	94.15
20	Loosen compacted soil layers in grassland fields	85.04	94.21
21	Allow field drainage systems to deteriorate	177.85	94.27
22	Reduce the length of the grazing day or grazing season	255.90	94.33

[a]£/% Reduction in P loss/ha, NPV/ha over an 8-year period. Discount rate 3.5%
[b]Farm level per ha

proliferation of *Elodea* in the Erne system. Eutrophication is directly related to excess nutrients (P particularly) loading to the lake associated with fertilizer use by agriculture, one of the main socio-economic sectors in the region; fortunately, there are many well-known management measures which can be taken to reduce excess nutrient loading. The fact that some improvement in water quality can be achieved while also saving money for farmers provides a powerful justification for taking measures such as identified in items 1–6 of Table 2, thereby improving the efficiency of farms while also yielding environmental benefits for other stakeholders using or enjoying Lough Erne. However effecting change in agricultural practices is beset by governance challenges relating to the implementation of the European Common Agricultural Policy in two separate jurisdictions, where nutrient emissions occur in both the Republic and Northern Ireland but the environmental and social impacts of eutrophication effects are experienced disproportioantely in Northern Ireland. While there are clearly efficiencies to be achieved, the structuring of the Common Agricultural Policy and its single farm payments do not necessarily effectively promote this efficiency. While improved and cost-saving farming practices may reduce nutrient emissions to the Lough, the legacy effects of historic pollution combined with the recycling of nutrients by zebra mussel are likely to cause time-lags between measures and their effects.

By contrast, the proposal to manage lake levels, represents a relatively "quick-fix". However this proposal also represents an unknown quantity in terms of effectiveness and projected economic costs due to the consequent loss of farm land and productivity. The response of *Elodea* may not be as simple as reduced growth due to light limitation, and there is also potential for the existing weed to float to the surface resulting in continued nuisance for recreational activity. In reality the effectiveness of water-level management on *Elodea* are still uncertain, while the consequences to farm productivity are quite certain. Any future alteration of lake levels to control pond weed will therefore be an exercise in adaptive management or learning-by-doing.

While FCM highlighted clear and shared priorities for stakeholders, concerns for co-benefits, in terms of multiple ecosystem services that the lake ecosystem provides, did not emerge explicitly using the methods here. For example, many regulation and maintenance services are inherently valued by people but are often not prioritised compared to services linked to commercial concerns. However, participants did clearly value habitats and biodiversity. Increasing water levels, leading to an increase in the area of temporally flooded riparian habitats, or reducing nutrient inputs from agriculture, could enhance biodiversity and the supply of associated ecosystem services. The addition of holistic methods, such as the linkage framework approach proposed in (Robinson and Culhane 2020), could help to highlight other important services or elements of the SES that might influence the way management measures are evaluated.

5 Conclusions (Learning by Doing)

The current state of the Lough Erne SES results from a very long history of human use, conflict and alteration of lake and watershed ecosystems dating back for millennia. The modern Lough is a highly valued ecosystem which provides multiple benefits to humans yet also suffers from a range of chronic and acute environmental problems. European environmental legislative requirements for the Lough are not fully integrated into the management practices of the Lough, and the recent regulation on Invasive Alien Species adds an additional burden of management. Of the aquatic species listed in the regulation and found in the Lough Erne catchment, only one, *Elodea nutalli* (Nutall's pond weed) has had significant economic impacts to date.

A mixture of common and popular software as well as a range of free and open access tools was used to develop a bespoke methodology suited to a very specific local problem based on the perspectives, perceptions and using the language of interested stakeholders. Graphical output of the FCM was useful in communicating with stakeholders and in developing consensus on the main causes of the weed problem, while the resulting simplified model provided justification to focus the analysis on nutrients and eutrophication. Loose coupling of models allowed flexibility. Ultimately more complex ecological models including detailed biogeochemical components would be required to accurately predict the outcome of management measures. However, the utility of our simple approach is that it can promote understanding of trade-offs and represents a forum for interested stakeholder to contribute their knowledge to the management process. This is in itself a vital component of environmental management—since it is people who make the decision on what measures are to be taken. What our approach lacks in analytical complexity or data-driven robustness of more complex modelling techniques (many examples of which can be found in this book) it makes up for in terms of low-cost and ease of application with stakeholders when addressing a specific problem in a specific location. The co-design of the FCM with stakeholder enabled effective communication of complexity of the Lough Erne SES and trade-offs among environmental management options.

Integrated, ecosystem-based management approaches to the management of Lough Erne enable consideration of multiple primary activities and their pressures and provide a basis to meet multiple environmental as well as social and economic objectives. The transboundary nature of the Lough Erne catchment is a barrier to truly integrated management of the catchment, and the political boundaries between the two jurisdictions appear to be becoming more pronounced as the UK is set to leave the European Union. The Erne Drainage and Development Act (1950) (EDDA) was an early example of cross-border cooperation and succeeded because there were mutual benefits to be gained in the two jurisdictions. Changes to the management regime of the lake levels (within the legal limits of the EDDA) offer one opportunity for the management of the system which could continue to provide benefits to users of the Lough Erne SES on both sides of the border and could act as a focus for continued cross-border cooperation. While the inclusion of farmers within any of these management decisions is paramount, and these were notably missing from our stakeholder groups, our stakeholder approach did provide a valuable opportunity for cross border cooperation and collaboration which is one vital element in future management of the Lough Erne system.

While current EU environmental regulation provides a common cross-border framework for environmental management in the Lough Erne catchment across an international boundary, the future basis for such cooperation is unclear. The UK is currently presently in the process of leaving the European Union and the current political and economic basis for environmental regulation, as well as for enabling and subsidising agricultural production, is unlikely to remain as it is, while the potential future alternatives are largely unknown. Major changes in the social system comprising primary activities as well as the norms and values enshrined in environmental laws and regulations may be on the way. The effects of these changes on a social ecological system already characterised by overwhelming complexity cannot be foretold.

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Mitigating Negative Unintended Impacts on Biodiversity in the Natura 2000 Vouga Estuary (Ria de Aveiro, Portugal)

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Abstract This chapter presents the co-development of the Ecosystem-Based Management (EBM) planning process in the Vouga estuary for the mitigation of unintended impacts on biodiversity resulting from the 2019/2020 management plan. This estuary, part of Ria de Aveiro coastal lagoon located on the north-west coast of Portugal (40°38′N, 08°45′W), connects the Vouga river catchment area to the Atlantic Ocean. Ria de Aveiro, part of the Natura 2000 network, is characterised by high biodiversity and a wide range of ecosystem services. However, it is also a vulnerable territory that requires a management plan in practice for environmental protection, targeting threatened species and habitats, but also to enable socio-economic welfare. Framed by EBM principles, the stepwise planning approach aimed at identifying the governance boundaries and institutions, the policy objectives, synergies, and gaps relevant to managing biodiversity, and to promote participatory actions with local stakeholders and policy-makers to understand their objectives. These three first steps enabled us to understand the social-ecological system and to co-develop relevant EBM solutions. In the final step, the proposed

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EBM solutions were evaluated for effectiveness, efficiency, equity and fairness, and then compared to the present condition. The co-developed solutions target science, policy and stakeholders interfaces. Namely, scientific knowledge applied to restore saltmarshes and seagrasses, policy objectives harmonising monitoring across EU Directives and integrate territorial management instruments, and management process involving stakeholders throughout.

Lessons Learned The co-created EBM plan for the Vouga estuary Natura 2000 site is foreseen to support the further development of the Vouga Estuary Management Plan. To this end, it is also foreseen to support actions for a more comprehensive understanding of the social-economic implications of the provided ecosystem services in line with the Centro Portugal region strategy for smart specialisation (Portugal RIS3 Centro).

These are:

- *Continue to increase stakeholder participation*: stakeholders want to contribute to management and actively participate in the co-creation of adaptive management solutions;
- *Integrate and coordinate policies*: proceed with the development of the Vouga estuary management plan considering connectivity across water domains, landowners and users;
- *Promote adaptive management and acknowledge unintended impacts*: harmonise existing mandatory monitoring programmes to support regular evaluation and enable adaptive management involving stakeholders to respond to future management needs and challenges.

Needs to Advance EBM

- At the scale of Natura 2000 Vouga estuary, EBM plans should be co-created with input from local stakeholders and policy-makers. To protect biodiversity, managers should consider climate change projections and acknowledge uncertainty. For the successful implementation of the identified water and nature policies in places like the Vouga estuary, any actions need to ensure involvement of users and landowners.
- At a global scale, and particularly at European Union scale, it has been acknowledged that biodiversity protection is still deficient and that, at current trends, the EU Strategy for 2020 will fail to achieve its goal of halting loss of biodiversity. To this end, EBM, that encompasses any management or policy options intended to restore, enhance or protect the resilience of the ecosystem, appears as a valuable approach in support of EU Strategy beyond 2020.

1 Introduction

The United Nations (UN) 2011 declaration of the 2011–2020 Decade on Biodiversity brought to the forefront news the urgent need to halt the loss of biodiversity as well as its overall vision for 2050 of *"living in harmony with nature"*. It also made clear the fact that ecosystem functioning and the provision of ecosystem services essential for human well-being are supported by biological diversity. Within the UN Environment Programme, the Convention on Biological Diversity (CBD) developed The Strategic Plan for Biodiversity with a shared vision, mission, and set of strategic goals: the Aichi Biodiversity Targets.[1]

To this end, the European Union (EU) Biodiversity Strategy to 2020 aims at ensuring the existence and conservation of biodiversity and ecosystem services and at halting the loss of global biodiversity (European Commission 2011). Its main objective is to fulfil the implementation of nature protection legislation, with special emphasis in Natura 2000 sites with high biodiversity value. This strategy includes six targets focused on: better protection and restoration of ecosystems and their associated services; establishment of green infrastructure; development of sustainable agriculture and fisheries; control of invasive alien species; and an EU contribution to stop global biodiversity loss.

Action 5 is based on improving knowledge on ecosystem services. The use of maps helps to achieve this action by characterizing the spatial heterogeneity of ecosystems and services they supply, and the associated pressures and impacts. They also help to translate scientific evidence into information that is understandable for policy and decision making (Maes et al. 2016). Thus, mainstreaming values of biodiversity and ecosystem services into decision-making is expected to help increase awareness about the implications of further degradation and loss of natural ecosystems on human well-being (Teixeira et al. 2018, 2019).

An Ecosystem-Based Management (EBM) of aquatic ecosystems is more likely to support a timely achievement of the EU 2020 Biodiversity Strategy targets than isolated sectorial management initiatives (Piet et al. 2017; Martínez-López et al. 2019a). Such an integrative approach to ecological, social and governance principles sets an adequate context to apply socio-ecological concepts such as ecosystem services in practical management initiatives (Lillebø et al. 2019; Martínez-López et al. 2019b). The EBM planning process involves the coordination of policies, institutions and practices (Drakou et al. 2017; Piet et al. 2017; Rouillard et al. 2018), representing a holistic approach that aims to balance multiple interrelated dimensions of ecological integrity and human well-being (Gómez et al. 2016, 2017; Langhans et al. 2019).

Following Rouillard et al. (2018), the proposed approach aiming at mitigating negative unintended impacts on biodiversity in the Natura 2000 Vouga estuary

[1] Aichi Biodiversity Targets: https://www.cbd.int/doc/strategic-plan/2011-2020/Aichi-Targets-EN.pdf.

considers the following principles (Curtin and Parker 2014; Gómez et al. 2016, 2017; Martin et al. 2018):

- *EBM considers ecological integrity, biodiversity, resilience and ecosystem services;*
- *EBM is carried out at appropriate spatial scales;*
- *EBM develops and uses multi-disciplinary knowledge;*
- *EBM builds on social–ecological interactions, stakeholder participation and transparency;*
- *EBM supports policy coordination;*
- *EBM incorporates adaptive management.*

1.1 Study Site

Ria de Aveiro is a shallow coastal lagoon located on the north-west coast of Portugal (40°38′N, 08°45′W). The adjacent coast experiences strong seasonal upwelling, the designated North Atlantic Upwelling that supports high levels of productivity especially in summer (Lopes et al. 2014). The lagoon establishes the aquatic continuum between the upstream catchment area (3500 km^2) of the Vouga river that contributes with circa 80% of the freshwater inflow, and the Atlantic Ocean through a single connection (1.3 km length, 350 m wide and 20 m depth) (e.g., Lillebø et al. (eds) 2015; Stefanova et al. 2015; Sousa et al. 2016; Lopes et al. 2017). These hydrographical settings determine that the Vouga river estuary is located within the boundaries of Ria de Aveiro coastal lagoon.

Due to its valuable natural capital, listed under both the Birds Directive and the Habitats Directive, Ria de Aveiro is a classified site under the Natura 2000 network, entailing a Special Protection Area (SPA) that includes extensive saltmarsh habitats and the adjacent marine area. Since 2011, the lagoon is also an International Long-Term Ecosystem Research (ILTER) site. Within the lagoon watershed Aveiro city represents the major urban settlement with circa 60,000 inhabitants. Like other social-ecological systems, the Vouga estuary is subject to co-competing land and water uses (Lillebø et al. (eds) 2015). Previous trans-disciplinary studies acknowledged the importance of the Vouga estuary's geographic location combined with its natural capital, which has enabled the development of a wide variety of economic, cultural and recreational activities (Lillebø et al. (eds) 2015; Dolbeth et al. 2016; Lillebø et al. 2016; Sousa 2017; O'Higgins et al. 2019). However, this area often requires human intervention for protection, or to enable economic activities, due to anthropogenic pressures impacting the hydro-morphological conditions of the lagoon, the Vouga estuary, and the adjacent Baixo Vouga Lagunar freshwater section of the Vouga river, and natural pressures like ocean storm surges, coastal erosion, and torrential rain and flood events (Pereira and Coelho 2013; Lillebø et al. (eds) 2015; Dolbeth et al. 2016; Lopes et al. 2017; Luís et al. 2018).

1.2 Biodiversity Challenge in the Natura 2000 Vouga Estuary

Two management interventions, occurring during 2019/2020, will likely have negative unintended impacts on biodiversity (Lillebø et al. 2019; Martínez-López et al. 2019b):

- Dredging programme to enable hydrodynamic equilibrium and navigability in Ria de Aveiro coastal lagoon (APA 2018);
- Extension of a flood bank to disable surface saltwater intrusion into Baixo Vouga Lagunar agricultural areas, named '*Sistema de Defesa Primária do Baixo Vouga Lagunar*' (DGADR 2017).

The dredging programme's ultimate goal is to improve lagoon navigability and is expected to allocate part of its dredged sediments to reinforce the banks at lower elevation zones, threatened by surface saltwater intrusion from inundation, for the protection of infrastructures and goods. Additional dredged sand will be used for beach replenishment. The extension of the flood bank is expected to improve accessibilities, foster agricultural and livestock activities, and protect wildlife and other economic activities, namely ecotourism with bird watching tours, angling, and recreational activities at the upstream area of the flood bank. These two management options will cause negative, unintended impacts on biodiversity, including changes of the system's eco-hydrodynamics, including water current velocity, turbidity, and tidal prism that will impact seagrasses and saltmarshes (Lillebø et al. (eds) 2015, 2019; Dolbeth et al. 2016). Additionally, downstream saltmarshes will be subdued to "coastal squeeze" as the combined effect of the physical flood bank barrier with the increase tidal prism will result in longer submersion periods that saltmarsh species are not adapted to (Martínez-López et al. 2019b).

The goals of our EBM approach are to:

- Contribute to operationalising an EBM planning process in response to foreseen unintended impacts resulting from the present management options;
- Mitigate unintended impacts from a major dredging programme targeting the hydrodynamic equilibrium (APA 2018);
- Mitigate unintended impacts from the extension of a flood bank targeting surface saltwater intrusion into agricultural areas (DGADR 2017);
- Make use of the best available information in a trans-disciplinary context.

To reach these goals, the overarching policies, programmes, key governance institutions, and objectives relevant to managing biodiversity were identified. Stakeholders were engaged throughout the process in order to co-define the baseline, co-develop management scenarios, and co-create the EBM plan.

2 The EBM Planning Approach

To address the governance challenges in the frame of the Vouga estuary, the EBM planning approach followed a stepwise procedure.

2.1 Step One: Setting the Governance Boundary

In order to be policy relevant the governance boundaries for this EBM approach were set at 500 m from the aquatic realm boundary (Fig. 1) following the Vouga Estuary Management Plan under development by the Portuguese Environmental Agency.[2] This Plan links public administration and private sectors, and provides the basis to effectively manage Ria de Aveiro's natural capital, ecosystem services and associated socio-economic activities. It encompasses an integrated land-use management plan with appropriate measures to protect all wetlands habitats, e.g., transitional waters, mud and sand flats, seagrasses and salt marshes, as well social, economic and cultural development. As shown by O'Higgins et al. (2019), the 500 m boundary is well aligned with ecosystem service production and consumption boundaries of this resource system. In this way, the Vouga Estuary Management Plan policy boundary overlaps with the production and consumption of relevant ecosystem services. Furthermore, the Plan foresees the articulation of territorial management instruments, plans and programmes at different scales, from local to regional (Centro Portugal region) and national, covering appropriate spatial scales for the EBM approach.

2.2 Step Two: Identify Policy Objectives, Synergies, and Gaps

The most relevant national policy plans and programmes (and institutions responsible for implementation of the policy instruments), objectives for the planning process of EBM responses, as well as linkages to EU Policies, are presented in Table 1. These initiatives cover aspects from nature conservation, to water quality and management, to climate change adaptation and tourism. The later are key drivers for sustainable economic growth of the Centro Portugal region (Dolbeth et al. 2016). At local/regional scales, it is also important to consider sectoral plans and programmes that integrate operations, enable collaborative work among institutions, and promote articulation of environmental, economic and social factors. The most relevant national and regional institutions to be considered in the planning process of EBM responses are presented in Table 2.

[2]Agência Portuguesa do Ambiente, I.P.—APA, https://www.apambiente.pt/.

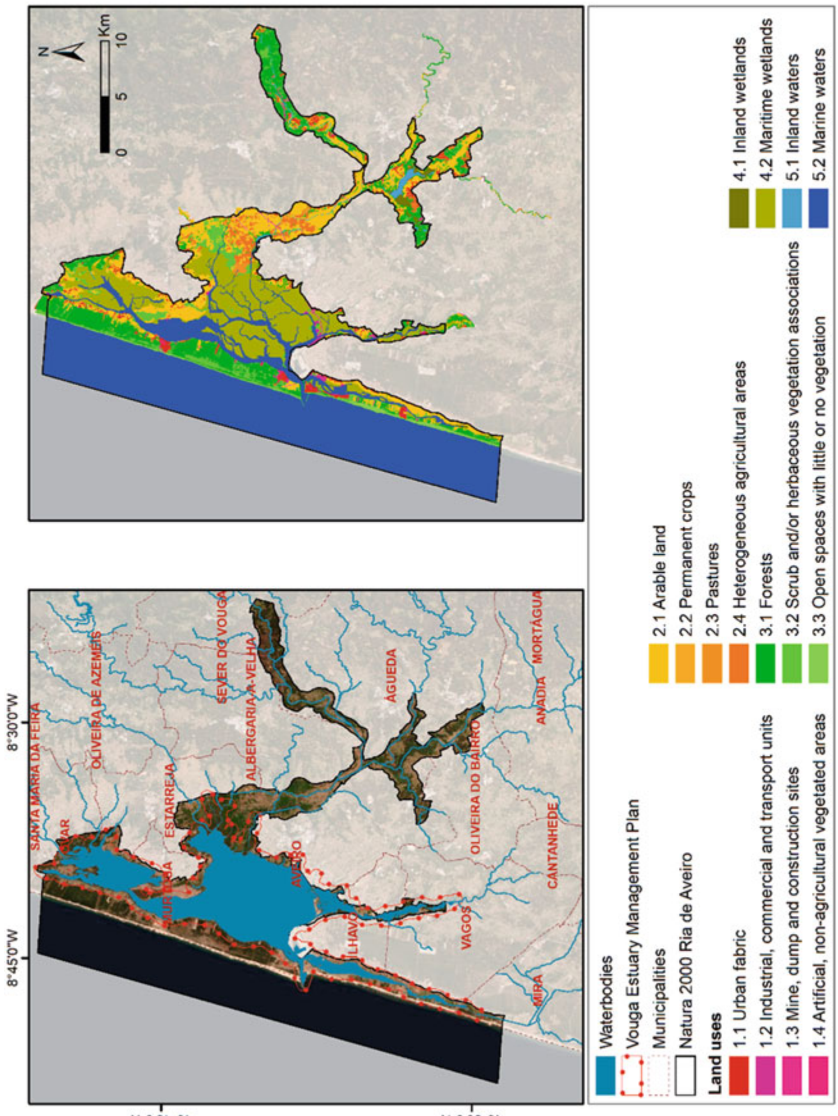

Fig. 1 The Ria de Aveiro coastal lagoon and boundaries of the Vouga Estuary Management Plan. Land uses are shown in the lower panel

Table 1 Identification of national relevant policy plans and programmes, and objectives for the EBM planning process at Ria de Aveiro: (a) link to EU policies and (b) link to regional/local policies

Policy plans and programmes	Objectives	(a) Link to EU policies
Sectoral Plan for Natura 2000 Network (PSRN2000) Institute for Nature Conservation and Forests (ICNF)	Territorial management tool for implementation of the national policy for conservation of biological diversity, aiming at safeguarding and enhancement of sites and SPAs of the continental territory, as well as maintenance of species and habitats in a favourable conservation status in these areas	Birds Directive (2009/147/EC); Habitats Directive (92/43/EEC)
National Water Plan (Decreto-Lei no. 76/2016) *Inter-ministerial Commission for Water management:* APA/ARHC; ICNF; Regional Directorate for Agriculture and Fisheries (DRAP); Directorate-General for Marine Resources (DGRM)	Governmental cross-sectoral management for the next 10 years: Increase water productivity and promoting rational use, with maximum respect for territorial integrity of the river basins; Protection, conservation and rehabilitation of water resources and associated ecosystems; Meeting needs of the population and country's economic and social development; Respect for relevant national and Community legislation and satisfaction of international commitments assumed by the Portuguese State; Access to information and participation of citizens in management of water resources	Water Frame Directive (WFD) (2000/60/EC) Floods Directive (2007/60/EC) Marine Strategy Framework Directive (2008/56/EC)
River Basin Management Plan (PGBH—RH4) Portuguese Environment Agency (APA/ARHC)	Outlines water planning for the tri-basin region of Vouga, Mondego and Lis, in accordance with WFD	WFD (2000/60/EC)
National Strategic Plan for climate change adaptation (ENAAC) Ministry of Environment	Establishes the need for adaptation. Contains the National adaptation strategy, and associated action plan, including reducing vulnerability and increasing response capacity.	EU Strategy on Adaptation to Climate Change (COM (2013) 216
National Strategic Plan for Tourism (PENT) Ministry of Economy and Innovation	Serves as basis for implementation of a series of initiatives aimed at fostering sustained growth of national tourism over the coming 10 years, and guiding activities of Portugal	EU strategy for a smart, sustainable and inclusive growth (COM (2014) 85 final, 2014/0044)

(continued)

Table 1 (continued)

Policy plans and programmes	Objectives	(a) Link to EU policies
	National Tourism Authority, as the key public body for the sector.	
		(b) Link to regional/local policies
Polis Litoral Ria de Aveiro APA/ARHC; ICNF	Integrated Operations of Rehabilitation and Recovery of Coastal Areas. Strong collaborative work between central administration and the Ria de the Aveiro Region Intermunicipal Community (CIRA)	Address the regional policy instruments Contribute to the Vouga estuary management plan Contribute to the Regional strategy for smart specialization (RIS3 Centro)
Coastal Zone Management Programme Ovar—Marinha Grande (POC-OMG) APA/ARHC	Reconcile the various conflicts of uses of the coastal zone, promoting articulation of environmental, economic and social factors related to coastal management.	
River Basin Management Plan (PGBH—RH4) Portuguese Environment Agency (APA/ARHC)	Outlines water planning for the tri-basin region of Vouga, Mondego and Lis, in accordance with WFD	

Source: Lillebø et al. (2019)

Each policy main objective (Table 1) identifies regional policy instruments contributing to the Vouga Estuary Management Plan, which aims at contributing to the Centro Portugal region strategy for smart specialisation (Portugal RIS3 Centro). Within this strategy, sea-related economic activities were selected as a strategic priority together with agriculture, forestry, tourism, information and communication technologies, materials, biotechnology, and health and wellness. The boundary for the Vouga Estuary Management Plan is presented and discussed in Fidélis and Carvalho (2015), and is considered as management boundary in O'Higgins et al. (2019) and in the proposed EBM approach (see Sect. 2.1). The Vouga Estuary Management Plan requires coordination with:

- Sectoral Plan for Natura 2000 Network (Institute for Nature Conservation and Forests; ICNF, I.P.), the territorial management tool for implementation of the national policy for the conservation of biological diversity;
- National Strategic Plan for climate change adaptation, following climate change projections, and containing the National adaptation strategy and associated action plan relevant in Centro Portugal coastal area.

The proposed EBM approach requires monitoring the policy impact of unintended pressures resulting from present management options. Although most of the information is reported in the frame of these EU water-related and Nature Directives, data sets are not harmonised. Therefore, the main gap identified concerns

Table 2 Identification of main institutions and policy domains for the EBM planning process

Institution	Policy domain	Additional information
Portuguese Environmental Agency (APA, I.P.) through the Regional Hydrographic Administrations (ARH Centro)	River Basin Management Plan (WFD) and Flood Risk Management Plan (Floods Directive) for hydrographic Region 4 (RH4) that includes Vouga, Mondego and Lis Rivers, and the foreseen estuary land use and management plans.	APA/ARH Centro is responsible for: water resources management; spatial planning of water resources, uses (including the economic analysis) and demands, and law enforcement; and for strategic and integrated planning of the coastal zone.
Institute for Nature Conservation and Forests (ICNF, I.P.)	Sectoral Plan for Natura 2000 Network (Habitats, Birds Directives; Biodiversity Strategy)	ICNF, I.P. is the national authority for nature conservation, biodiversity and forests; articulates and promotes integration of forest policy and conservation of nature and biodiversity in policies to combat desertification; to mitigate climate change and its effects; and to reduce country's energy dependence.
Centro Region Department of Agriculture and Fisheries (DRAPC)	Common Agricultural Policy (CAP) and Common Fisheries Policy (CFP)	DRAPC is a service of the Ministry of Agriculture, Forestry and Rural Development, whose mission is to participate in formulation and implementation of policies in agriculture, rural development and fisheries, as well as collaborate in policies in areas of forests, food security and plant health, in liaison with relevant central bodies and services within the framework of the efficiency of local management of resources.
Directorate general for Natural Resources, Safety and Maritime Services (DGRM)	Marine Strategy Framework Directive (MSFD) and Maritime Spatial Planning (MSP)	DGRM is a government entity of the Ministry of the Sea, a central office of direct administration of State, with administrative autonomy with the mission, under maritime administration functions, to implement and execute policies concerning maritime safety and prevention of pollution by ships and ensure regulation, inspection, national coordination and control of activities developed under these policies.

(continued)

Table 2 (continued)

Institution	Policy domain	Additional information
The Centro Regional Coordination and Development Commission (CCDRC)	Promote an integrated and sustainable development of Portugal's Centro region (NUT II)	CCDRC is tasked with coordinating and promoting at the regional level governmental policies with regard to Regional Planning and Development, Environment, Land Management, Inter-Regional and Cross-Border Cooperation and also support local government and inter-municipal associations. CCDR-C's fields of intervention also encompass management of regional operational programmes funded by the EU, and other regional development financing instruments.

the lack of harmonised monitoring programmes for the Water Framework Directive (WFD, 2000/60/EC) and Habitats Directive (HD, 92/43/EEC). In this context, one of the main challenges concerns the use of best available information.

2.3 Step Three: Understand Stakeholder Objectives

Vouga estuary Natura 2000 site governance involves a multiplicity of institutions, organisations and stakeholders, and involves articulation of programs and plans of local, regional and national levels (Teles et al. 2014; Fidélis and Carvalho 2015; Lillebø et al. (eds) 2015; Sousa et al. 2016; Sousa 2017; Fidélis et al. 2019). The Portuguese Environment Agency, through its Regional Hydrographic Administration for Portugal Centro Region (APA/ARH Centro) was engaged at a very early stage of the project, helping identify key management questions. Other stakeholders engaged at the kick-off stage of the work and contributed to the datasets that support scenario development include the Institute for Nature Conservation and Forests (ICNF, I.P.), Regional Directorate for Agriculture and Fisheries (DRAP Centro), Hydrographic Institute (IH), and Directorate-General for Marine Resources (DGRM). Stakeholder's participatory moments included two workshops (WS I and WS II) and a final seminar, where the co-created EBM plan was presented. All participants received a non-technical, open-access book in Portuguese detailing the entire EBM process and the main results. Approximately 70 stakeholders representing the four major groups, namely policy/governance, public administration, business, and non-governmental organizations, were invited to participate (Table 3).

Table 3 Identification of key stakeholders for the planning process of EBM responses

Policy/Governance	
Environment	APA/ARH Centro—Portuguese Environmental Agency ICNF—Institute for Nature Conservation and Forests
Fisheries and agriculture	DRAPC—Centro Region Department of Agriculture and Fisheries
Marine	DGRM—Directorate-General for Natural Resources, Safety and Maritime Services
Public administration	
Regional administration	CCDRC—The Centro Regional Coordination and Development Commission CIRA—Inter-municipal Community of the Aveiro Region
Municipalities within the Natura 2000 classified area	Águeda, Albergaria-a-Velha, Anadia, Aveiro, Estarreja, Ílhavo, Mira Murtosa, Oliveira do Bairro, Ovar, Vagos.
Parishes within the Natura 2000 classified area	E.g., Angeja, Avanca, Beduído & Vieiros, Bunheiro, Cacia, Canelas & Fermelã, Esgueira, Espinhel, Fermentelos, Gafanha Da Boa Hora, Gafanha Da Encarnação, Gafanha Da Nazaré, Gafanha Do Carmo, Glória & Vera Cruz, Murtosa, Óis da Ribeira, Ouca, Ovar Union of parishes, Pardilhó, Requeixo, Salreu, Santo André De Vagos, São Jacinto, São Salvador, Sosa, Torreira, Vagos & Santo António De Vagos, Válega.
Business	
Industry	Portucel—The Navigator Company
Tourism	Incrível Odisseia—Moliceiros boat rides Sterna—Solar boat tours and bird watching
Agriculture	ABBVL—Association of Beneficiaries of Baixo Vouga Lagunar ACRM—Association of Breeders of Marinhoa Breed ALDA—Association of Agriculture of the District of Aveiro
Fisheries	APARA—Artisanal Fishing Association of the Region of Aveiro
Aquaculture	APA—Portuguese aquaculture association
Services	APA—Port of Aveiro Administration (APA)
Other	
Local associations	AVELA—Sailing club ADERAV—Association for the study and protection of the Natural and Cultural Heritage of Aveiro Region CCPAV—Hunting and Fishing Club of Aveiro/Vouga
Non-governmental organizations (NGO's)	FAPAS—Fund for the Protection of Wild Animals GEOTA—Study Group on Spatial Planning and Environment LPN—League for the Protection of Nature SPEA—The Portuguese Society for the Study of Birds ASPEA—Portuguese Association of Environmental Education

Source: Lillebø et al. (2019)

2.3.1 Stakeholders' Perception and Spatial Multi-Criteria Analysis

At the first workshop (WS I), 17 stakeholders representing the four major groups participated, signing an informed consent agreement form, and were asked to identify the relevance of ecosystem services in Ria de Aveiro for building alternative management scenarios (Fig. 2). Participants were invited to express their opinion regarding expected beneficial effects and persistent concerns related to the current management options and contribute to the spatial multi-criteria analysis through prioritization of ecosystem services (Lillebø et al. 2019; Martínez-López et al. 2019b). This prioritization reflected stakeholders' social preferences regarding ecosystem services in order to find optimal management actions (sensu Villa et al. 2002; Martínez-López et al. 2019b). The method adopted ensures transparency of the participatory process, which is of paramount importance as different sectoral interests, such as conservationists, local users and from the business sector, like tourism, may express different priorities in relation to a set of ecosystem services of interest. This is crucial to make the participatory valuation of ES an opportunity for a more comprehensive, fair and integrative perspective for EBM (Martínez-López et al. 2019b). This socio-ecological approach illustrates how planned and structured co-developed solutions can effectively contribute and support adaptive management and conservation of coastal ecosystems (Lillebø et al. 2019).

2.3.2 Recommendations for EBM Implementation

At the second workshop (WS II), 15 stakeholders representing the four major groups, which signed an informed consent agreement form, were asked to identify the relevant issues that should be included in the adaptation strategy as well as the opportunities and constraints of implementation.

Participants were invited to join round-table groups (Fig. 3) following a 'world café' methodology to discuss three topics:

- Environment and ecosystem services (spatial distribution of EBM measures, identification of areas for remediation of marshes, benefits and constraints);
- Institutions and equity (identification of institutions involved, process coordinators, benefits and constraints);

Fig. 2 Overview of WS I participatory moments: habitats spatial distribution maps; presentation of WS I objectives; spatial multi-criteria analysis for ES valuation

Fig. 3 Overview of WS II participatory moments: The three 'world café' sessions considered

- Operationalization and sustainable development (identification of existing activities supported by the benefits provided by marshes, business opportunities, benefits and constraints).

Stakeholders were also invited to answer the question "*In which way the EBM methodology can be better or not in relation to the management approaches used until now*?"

2.4 Step Four: Understand the Social-Ecological System

The assessment of Vouga estuary's current state included the identification of habitats, specific public and private primary human activities, and respective pressures in the entire Natura 2000 territory. To address the ecological perspective of the system, data sources from scientific publications, projects (e.g., LAGOONS EU FP7; ADAPT-MED EU FP7 ERA-NET; LTER-RAVE FCT, AQUACROSS EU H2020), national agencies (e.g., above mentioned), online platforms (e.g., Copernicus datasets) and from national/regional official reports were integrated. In order to harmonise habitats classification, all data sets (mainly following Annex I of EU Habitats Directive; Sousa et al. 2016) were converted into the EUNIS habitat classification, following the official correspondence table available at the European Environmental Agency (EEA) portal (http://eunis.eea.europa.eu/habitats.jsp). Data sets on ecosystem services (mainly following CICES, V4.3; Lillebø et al. (eds) 2015; Sousa et al. 2016) were updated and classified following the latest Common International Classification of Ecosystem Services (CICES, V5.1; https://cices.eu/) (Haines-Young and Potschin 2017; O'Higgins et al. 2019). The potential of a given habitat to supply ecosystem services was attained using a lookup table on the contribution of each EUNIS habitat compiled based on expert judgment (Teixeira et al. 2019). The identified ecosystem services were aggregated into eleven ecosystem services in order to enable spatial multi-criteria analysis by stakeholders (Table 4). This table includes the correspondence code from CICES v4.3 to 5.1, considering the identified services for the considered territory, as well as selected aggregation of services used in the scope of the participatory moments in order to optimize communication and active participation of stakeholders.

Table 4 Assessment of ecosystem services provided by Vouga estuary: (a) provisioning; (b) regulation and maintenance; (c) cultural

ES code	Group	Class	v4.3	v5.1	Subclass Ria de Aveiro
(a) CICES section: Provisioning					
ES1 Biotic based energy sources	Mechanical energy	Animals reared to provide energy (incl. mechanical)	1.3.2.1	1.1.3.3	Physical labour provided by cattle supporting agricultural activities
	Biomass-based energy sources	*Cultivated plants* (incl. fungi, algae) grown as a source of energy | *Wild plants* (terrestrial and aquatic, incl. fungi, algae) used as a source of energy	1.3.1.1	1.1.1.3 & 1.1.5.3	*Not applicable at the selected management area*
ES2 Abiotic energy sources	Renewable abiotic energy sources	Freshwater surface water used as an energy source | Coastal and marine water used as energy source	N/A	4.2.1.3 & 4.2.1.4	*Not applicable at the selected management area*
ES3 Biotic materials	Biomass	Fibres and other materials from wild *plants/animals* for direct use or processing (excl. genetic materials)	1.2.2.1 & 1.2.2.2	1.1.5.2 & 1.1.6.2	Reeds seasonally harvested Worms collected in intertidal mudflats and used as bait Macroalgae are collected for *in-situ* aquaculture Seagrasses and macroalgae ("moliço") harvesting Sea rush used as cattle bedding and then as a fertilizer
ES4 Abiotic materials	Non-metallic	Mineral substances used for material purposes	N/A	4.3.1.2	Sand extraction
	Water	*Surface water/ ground water* (and subsurface) used as a material (non-drinking purposes)	1.2.2.1 & 1.2.2.2	4.2.1.2 & 4.2.2.2	The lagoon provides surface water for salt production and forest-fire control, and ground water for inland aquaculture, agriculture, livestock, urban and industrial purposes

(continued)

Table 4 (continued)

ES code	Group	Class	v4.3	v5.1	Subclass Ria de Aveiro
ES5 Nutritional biotic substances	Biomass	Wild plants (terrestrial and aquatic, incl. fungi, algae) used for nutrition	1.1.1.3	1.1.5.1	Wild glasswort *Salicornia* sp. harvested and sold as a gourmet product
		Wild animals used for nutritional purposes	1.1.1.4	1.1.6.1	Fish and shellfish: lamprey *Petromyzon marinus*, European eel *Anguilla anguilla*, allis shad *Alosa alosa*, clams *Ruditapes decussatus* and *Venerupis corrugata,* cockle *Cerastoderma edule* cuttlefish *Sepia officinalis*
		Plants cultivated by *in-situ* aquaculture grown for nutritional purposes	1.1.1.5	1.1.2.1	Macroalgae farming
		Animals reared by *in-situ* aquaculture for nutritional purposes	1.1.1.6	1.1.4.1	Aquaculture farms of marine fish and bivalves
ES6 Nutritional abiotic substances	Mineral	Mineral substances used for nutritional purposes	N/A	4.3.1.1	Marine salt extraction (salt pans)
	Water	Surface water for drinking	1.1.2.1	4.2.1.1	*Not applicable at the selected management area*
(b) CICES section: Regulation and maintenance					
ES7 Mediation of flows	Mass flows	Control of erosion rates ǀ Buffering and attenuation of mass movement	2.2.1.1 & 2.2.1.2	2.2.1.1 & 2.2.1.2	Overall, coastal dunes, salt marshes and seagrass meadows contribute to maintain the lagoon integrity
	Liquid flows	Hydrological cycle and water flow regulation (incl. flood control and coastal protection)	2.2.2.1 & 2.2.2.2	2.2.1.3	Seagrass meadows and salt marshes reduce sediment resuspension and turbidity in the water column, contributing to increase the light availability in the water column São Jacinto dunes, salt marshes and reeds provide resilience to extreme weather events and act as physical buffering of climate change

(continued)

Table 4 (continued)

ES code	Group	Class	v4.3	v5.1	Subclass Ria de Aveiro
ES8 Mediation of waste toxics and other nuisances	Mediation by biota | Mediation by ecosystems	Bio-remediation by micro-organisms, algae, plants, and animals	2.1.1.1 & 2.1.1.2	2.1.1.1	Bio-remediation by ecosystem components (e.g., halophytes); Decomposition/mineralisation processes of plant material mediated by microorganisms; Biological filtration by oysters, clams and mussels in aquaculture and by wild animals
		Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals | Dilution by freshwater and marine ecosystems	2.1.1.2 & 2.1.2.1 & 2.1.2.2	2.1.1.2 & 5.1.1.1	Bio-physicochemical filtration/sequestration/storage/accumulation of pollutants by the lagoon habitats; Adsorption and binding of metals and organic compounds in ecosystems, as a result of combination of biotic and abiotic factors; Hydrodynamic dilution of pollutants (tidal action)
ES9 Maintenance of physical chemical biological conditions	Lifecycle maintenance, habitat, gene pool protection	Maintaining nursery populations and habitats (incl. gene pool protection)	2.3.1.2	2.2.2.3	Maintaining nursery habitat for fisheries species and commercial invertebrates; Seagrasses, salt marshes including extended areas of reeds, intertidal mudflats, sand flats and salt pans
	Pest, disease control	Pest control (incl. invasive species) | Disease control	2.3.2.1 & 2.3.2.2	2.2.3.1 & 2.2.3.2	Maintaining the system in a healthy status (e.g., from alien species or diseases)
	Soil formation, composition	Decomposition and fixing processes and their effect on soil quality	2.3.3.2	2.2.4.2	Decomposition of biological materials and their incorporation in sediments
	Water conditions	Regulation of the chemical condition of salt waters by living processes	2.3.4.2	2.2.5.2	Water purification by tidal wetlands, including seagrass meadows and salt marshes

(continued)

Table 4 (continued)

ES code	Group	Class	v4.3	v5.1	Subclass Ria de Aveiro
	Atmospheric composition and climate regulation ǀ Gaseous/air flows	Regulation of chemical composition of atmosphere and oceans ǀ Maintenance and regulation by inorganic natural chemical and physical processes	2.3.5.1	2.2.6.1 & 5.2.2.1	Global climate regulation by greenhouse gas/carbon sequestration by seagrass meadows and salt marshes, water columns and storage in sediments and their biota; Transport of carbon into oceans
		Regulation of temperature and humidity, including ventilation and transpiration	2.3.5.2 & 2.2.3.2	2.2.6.2	Micro and regional climate regulation by the Ria de Aveiro lagoon water body that includes the Vouga estuary
(cont.) (c) CICES section: Cultural					
ES10 Physical & intellectual interactions with biota, ecosystems, land & seascapes environmental settings	Physical and experiential interactions	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	3.1.1.1	3.1.1.1 & 6.1.1.1	*In-situ* bird watching of: Resident birds, e.g., *Charadrius alexandrinus* Migratory birds, e.g., *Himantopus himantopus*
		Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions ǀ Natural, abiotic characteristics of nature that enable active or passive physical and experiential interactions	3.1.1.2	3.1.1.2 & 6.1.1.1	Walking, diving, biking, sailing, boating, kite surfing, windsurfing, kayaking, swimming, leisure fishing (angling) and leisure hunting.
	Intellectual and representative interactions	Characteristics of living systems that enable scientific	3.1.2.1	3.1.2.1 & 6.1.2.1	Ria de Aveiro is subject matter for research

(continued)

Table 4 (continued)

ES code	Group	Class	v4.3	v5.1	Subclass Ria de Aveiro
		investigation or the creation of traditional ecological knowledge ǀ Natural, abiotic characteristics of nature that enable intellectual interactions			
		Characteristics of living systems that enable education and training	3.1.2.2	3.1.2.2	Natural and cultural heritage of the lagoon are subject matter of education (e.g., guided boat tours in Ria, science activities in the summer with the support of the University of Aveiro, BioRia Environmental trails, Natural Reserve of São Jacinto Dunes, "Marinha da Troncalhada" salt pan ecomuseum, "Santiago da Fonte" salt pan (belongs Aveiro University), ship-museum "Santo André" (an extension of the Maritime Museum of Ílhavo); "Casa Gafanhoa" municipal museum (testimony of the rural livelihoods of Ílhavo municipality)
		Characteristics of living systems that are resonant in terms of culture or heritage	3.1.2.3	3.1.2.3	Archaeological sites (e.g., shipwrecks, ship hull, and other isolated findings); Traditional architecture (e.g., "Palheiros", "Casa Gafanhoa"); Traditional boats (e.g., "Moliceiro", "Bateira", "Mercantel"); Traditional activities (e.g., salt production)
		Characteristics of living systems that enable aesthetic experiences	3.1.2.5	3.1.2.4	Sense of place; Artistic representations of nature (e.g., ceramic tiles, painted shells); Inspiration for some painters and writers, interested in the history and heritage of the lagoon and its users

(continued)

Table 4 (continued)

ES code	Group	Class	v4.3	v5.1	Subclass Ria de Aveiro
ES11 Spiritual symbolic & other interactions with biota ecosystems & land seascapes environmental settings	Spiritual and/or emblematic	Elements of living systems used for entertainment or representation ǀ Natural, abiotic characteristics of nature that enable spiritual, symbolic and other interactions	3.1.2.4	3.2.1.3 & 6.2.1.1	*Ex-situ* experiences through local festivals related with the lagoon's products and activities (e.g., "Festa da Ria" summer festival with traditional "moliceiro" boats race; Cod fish festival; Eel and "ovos moles" from Aveiro; food festival; International marine salt festival; "FARAV" handcraft festival, Lamprey festival, Allis shad (*Alosa alosa*) festival
	Other cultural outputs	Characteristics or features of living systems that have an existence value	3.2.2.1	3.2.2.1	Enjoyment provided by salt pans, salt marshes, seagrasses and wild species
		Characteristics or features of living systems that have an option or bequest value	3.2.2.2	3.2.2.2	Willingness to preserve salt pans, salt marshes, seagrasses and wild species for future generations

Equivalence of CICES classification (Group v4.3 & Class v5.1) to aggregated ecosystem services (ES code) used at the participatory moments in the scope of the EBM approach. Note: the assessment included the biologically mediated process and the abiotic outputs
Adapted from Martínez-López et al. (2019b) and O'Higgins et al. (2019)

Analysis of ecosystem services valuations (Martínez-López et al. 2019a) revealed two major stakeholder opinion groups, stating within group more similar preferences regarding ecosystem services, and whose composition was heterogeneous and not related to specific stakeholder groups identified. Weights in the spatial multi-criteria analysis in the selected Natura 2000 area took into account the mean of the ecosystem services scores given by individuals in the same opinion group. However, in the absence of strong and significant differentiation among the two opinion groups' valuations, a compromise map was generated, representing the average prioritization of ecosystem services by all participants.

Overall, the stakeholders' valuation clearly revealed the importance attributed to ecosystem services directly provided by water (freshwater, transitional, and coastal/marine), with special emphasis on the lagoon ecosystem (Lillebø et al. 2019). However, different preference patterns may arise if the focus is set on smaller scales or in specific areas of the case study as also demonstrated by Martínez-López et al. (2019a).

The final ecosystem services valuation maps were then compared with areas that will be affected by the dredging programme and flood bank. The mapping process involved the combination of several lines of evidence. The spatial explicit information on habitat mapping, human activities, and identified ecosystem services (Teixeira et al. 2019) was considered along with stakeholder's ecosystem services prioritization, the ecosystem services provisioning risk assessment (Lillebø et al. 2018), and stakeholders' persisting concerns regarding the foreseen measures, crucial to effectively highlight the most critical areas for the implementation of EBM management measures.

2.5 Step Five: Specification of Relevant EBM Solutions (as Part of the EBM Cycle)

Specification of relevant EBM solutions requires a clear definition of each component of Drivers-Pressures-State relationships as well as their causal links (Gómez et al. 2017; Teixeira et al. 2018). The applied approach, after Gómez et al. (2017) links the socio-economic and the ecological systems by making a clear distinction between:

- *"the activities that benefit from the provision of natural goods and services for the production of final goods and services that are of direct concern for human welfare;*
- *the drivers of pressures affecting ecosystems, represented by the specific demands of naturally provided goods and services in the quantity, quality required at specific places and moments of time;*
- *the primary activities that (co-) produce goods and services provided by natural capital that are of direct concern to explain the pressures over ecosystems."*

The assessment of the current state included the identification of the specific primary activities and the respective pressures considering the identified habitats (see Sect. 3.4). The specification of the relevant EBM solution requested the following:

- Identify a baseline scenario, incorporating the considered management measures;
- Formulate objectives related to the unintended impacts on biodiversity;
- Screen measures and instruments to understand ecological and social components;
- Construct a narrative reflecting management measures, stakeholders' perception of ecosystem services valuation, and science-based knowledge generated, to support planning the EBM response;
- Evaluate proposed habitat restoration measures using EBM criteria, taking into consideration policies and feasibility, to show that compliance is achievable.

2.6 Step Six: Evaluate the EBM Solutions

After co-defining EBM management alternatives, the following step concerns the co-evaluation of proposed management alternatives, following Piet et al. (2017), by applying established pre-screening criteria: effectiveness (i.e., hitting the target); efficiency (making the most for human well-being); and equity and fairness (i.e., sharing the benefits). This allowed to determine the performance of the proposed EBM measures compared to a baseline situation "*in terms of environmental impacts, subsequent costs and benefits of human wellbeing at individual and collective levels, and the distribution of these impacts and costs throughout society*" (Piet et al. 2017).

3 The Co-created EBM Plan for the Vouga Estuary Natura 2000 Site

3.1 The Governance Boundary

As for other socio-ecological systems the Vouga estuary governance is complex, involving several institutions with multi-level and multi-spatial scales of governance (Lillebø et al. (eds) 2015; Sousa 2017; Fidélis et al. 2019), with different governance models applied for integrated water resource management (Teles et al. 2014; Fidélis et al. 2019). Fidélis et al. (2019) assessed alternative governance models for Ria de Aveiro considering "*the organizational settings established to accommodate the different policy priorities existing in an estuary, their decision-making tools and processes, responsibility boundaries, stakeholder involvement schemes, and the means to face the challenges of a dynamic and vulnerable system*". This analysis, built upon Teles et al. (2014), presents an in-depth discussion highlighting the need for a paradigm change that implies high levels of institutional reforms. The authors concluded that "*regardless of the model adopted, it is crucial to derive a stable collaborative framework of decision-making in order to integrate action plans and policies for integrated water resource management in estuarine areas*" which is in line with the proposed approach to specifically address co-development of EBM planning in the Vouga estuary for the mitigation of unintended impacts on biodiversity.

3.2 Policy Objectives, Synergies, and Gaps

The assessed EU water-related and Nature Directives showed that Natura 2000 network sites should be '*managed in a sustainable manner, both ecologically and economically*', involving local policy-makers and stakeholders. Priorities identified for improvement include:

- Harmonisation and integration of monitoring programmes of Water Framework and Habitats Directive in water-dependent Natura 2000 sites;
- Development of the Vouga estuary land use and management plan aiming to conserve and promote sustainable use of water resources, ecosystem functions, integrated management, and coordination between various territorial management instruments;
- Enhancement communication among entities and foster involvement and active participation of land users and landowners.

3.3 Understand Stakeholder Objectives

Stakeholders representing different sectoral interests or activities share key objectives to: foster sustainable development of economic activities and preservation of aquatic biodiversity; integrate territorial management instruments; enhance participatory management; and co-create adaptive management solutions. There were a number of aspects that stakeholders identified, both with respect to major beneficial effects and persisting concerns, regarding Ria de Aveiro and the Vouga estuary from the first workshop:

Ecosystems biodiversity—Stakeholders highlighted habitat richness as important and revealed concerns regarding impact of dredging on seagrasses, saltmarshes, and juvenile fauna due to changes in eco-hydrology and potential mobilization of contaminants due to dredging.

Water management—The need for targeted dredging (e.g., oriented for habitats, housing, and infrastructure) was acknowledged but concerns were expressed regarding changes at the system hydrology, specifically increase in tidal prism due to dredging. Consequently, low navigability in inner channels during low tide and the increase of ocean water volume in the lagoon during high tide are of concern.

Agriculture—Concerns were related to the loss of traditional agricultural activities that enable ecosystems and biodiversity maintenance, which could benefit from incentives and compensations. The stimulus for agriculture was acknowledged. To this end the need for the completion of the flood bank and the increase in agricultural land area was set forward.

Fisheries and aquaculture—Stakeholders highpoint the relevance of this coastal system to migratory species with high socio-economic value, such as sea lamprey (*Petromyzon marinus*), European eel (*Anguilla anguilla*) and allis shad (*Alosa alosa*). Concerns regarding these activities could be overcome through incentives and compensations for maintenance of traditional aquaculture activities that maintained ecosystems and biodiversity.

Tourism and recreational activities—Tourism was seen as an opportunity, namely marked walking trails, supporting recreational activities and ecotourism. The increased navigability conditions inside the lagoon after planned dredging will promote recreational boating and touristic activities, although some concerns remain as increases in water current velocity is expected to alter habitats used for other touristic and recreational purposes (e.g., loss of lagoon inner mud/sand-flats used either as beaches or preferential bird watching sites).

Transversal—Other beneficial aspects and persisting concerns were considered transversal to the previous issues, specifically benefits from development of different sectoral economic activities; as well as the recognised scientific knowledge on Ria de Aveiro natural capital. Main concerns were due to lack of communication, which is paramount for integrated management, the need for information and awareness in the municipal councils, as well as landowner involvement, and lack of regulatory surveillance of activities within Ria de Aveiro.

Considering the management measures to be implemented, stakeholder knowledge and perceptions supported baseline scenario development, formulation of objectives related to the unintended impacts on biodiversity, and narratives of possible futures to support planning the EBM response.

3.4 Understand the Social-Ecological System

To understand the impact of the dredging in the Ria de Aveiro, we identified key human activities (Fig. 4), resulting pressures, habitats (see Fig. 5 for EUNIS habitat types), and how these support valuable ecosystem services. Relevant activities are related to boating, diving, shipping, coastal defence, port facilities, saltworks, fishing, bait digging, aquaculture, agriculture, livestock and forestry.

The linkage framework for Drivers and Pressures considering the Vouga coastal watershed is shown in Fig. 6. It can be seen that transitional waters realm, which includes the EUNIS habitats type code A, is affected by several pressures resulting from specific human activities, including capital dredging and maintenance dredging. The linkage chain associated with these activities reflects the complexity of linkages relating activities with associated pressures that determine functions and services provided by the habitats they impact. Figure 7 highlights the linkage framework for Drivers-Pressures-Ecosystem Component-Ecosystem Function and Ecosystem Services in the Vouga estuary. The management options 'behind' the baseline scenario were plotted considering the aggregated primary activities of dredging (representing the dredging programme) and flood and coastal defence (representing the extension of the flood-bank) and ecosystems components, functions and services. These management options are also relevant for other activities, namely cultivation of crops and livestock. The extension of the flood bank will disable surface saltwater intrusion into Baixo Vouga allowing recovery of arable land for agriculture and livestock. Furthermore, they will also affect input of organic matter and litter into the aquatic environment. As such, changes in the mechanical and physical structuring will affect biogeochemical cycles and production (primary

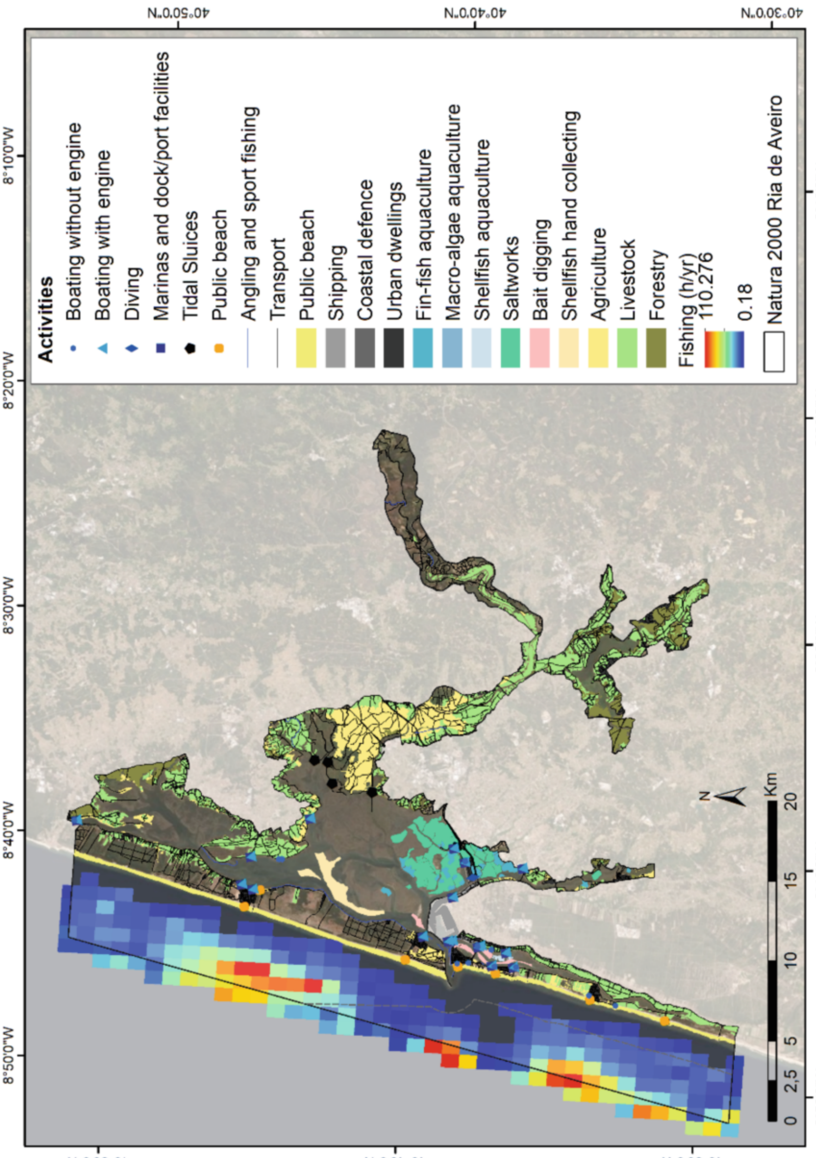

Fig. 4 The main activities identified at the Vouga river coastal watershed under classification of Natura 2000 network

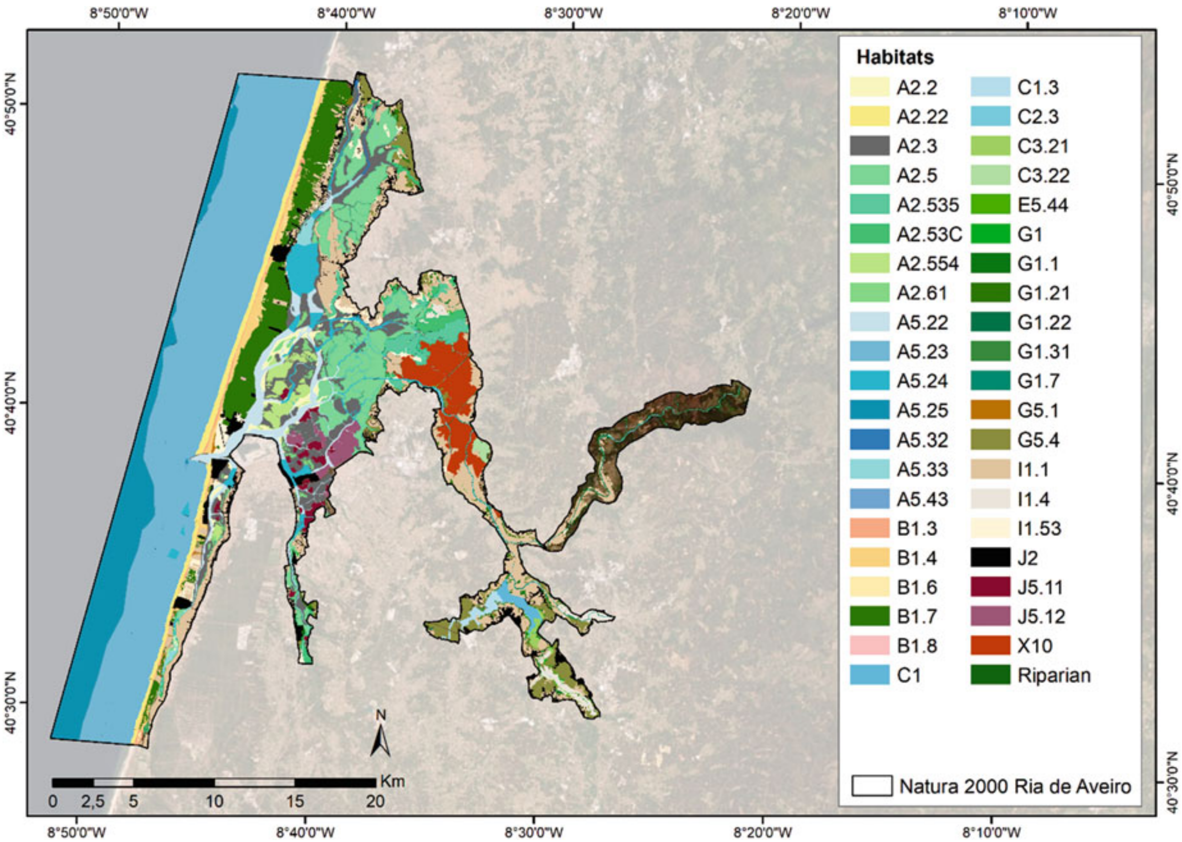

Fig. 5 The EUNIS habitats identified at the Vouga river coastal watershed under classification of Natura 2000 network. Code A relates to transitional waters domain, Code B to coastal/marine waters domain, Code C to freshwater domain, Code E to grasslands, Code G to riparian vegetation, Code I to terrestrial habitats, Code J to hypersaline habitats (e.g., salt pans), , Code X to habitat complexes, and Code X10 to Bocage

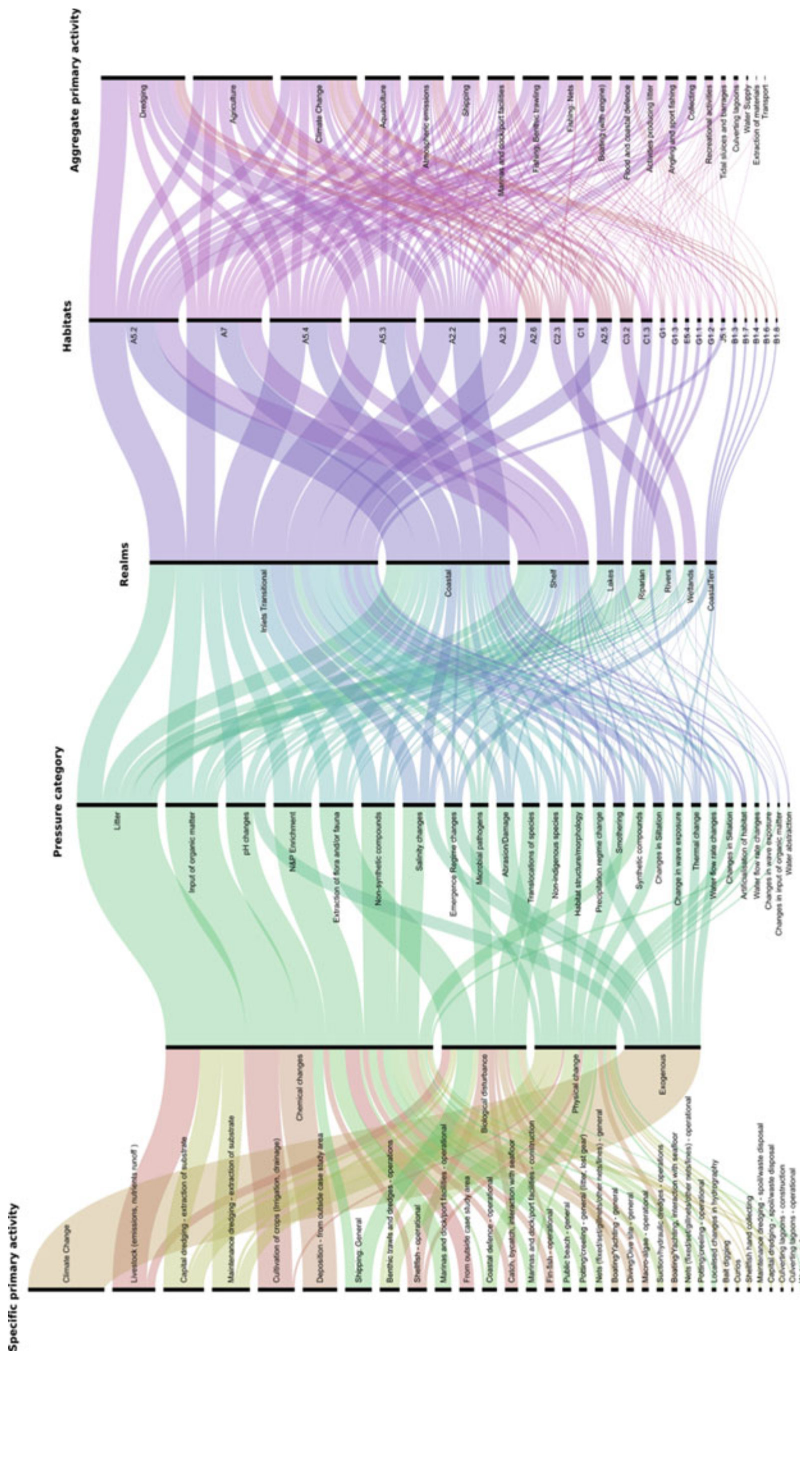

Fig. 6 Linking Drivers and Pressures in the Vouga river coastal watershed. (Plotted with Mauri et al. 2017)

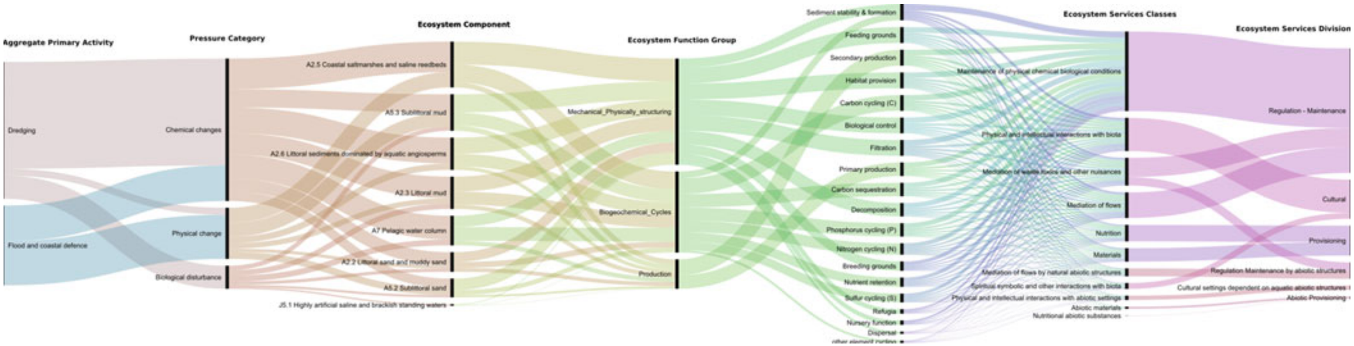

Fig. 7 Linkage framework for Drivers-Pressures-Ecosystem Component-Ecosystem Function and Ecosystem Services in the Vouga estuary, highlighting the management options, i.e., the dredging activities and the extension of the flood bank. (Plotted with Mauri et al. 2017)

Table 5 Description of proposed EBM responses supported by the prospective scenarios and considering the existing Sectoral Plan for Natura 2000 Network as well as the National Strategic Plan for climate change adaptation

	Seagrasses meadows restoration	Saltmarshes restoration
Main cause for mitigation measures	Compensate the loss due to changes in water current velocity and light availability.	Compensate the loss due to 'coastal squeeze' and the increase in tidal prism.
Ongoing science based knowledge to support mitigation measures	Intertidal *Zostera noltei* numerical modelling in Ria de Aveiro (e.g., Azevedo et al. 2013, 2017). Research projects BioPradaRia and Remoliço (PT MAR2020 funded) testing restoration techniques for *Z. noltei* populations *in situ* and under controlled laboratory conditions.	Running InVEST GIS-based modelling tool to support the selection of potential areas, as well as the restoration techniques. These might combine nature-based solutions to protect shorelines and actions to promote sediment accretion for salt marsh elevation.
Detailed restoration measures	(1) Protection of existing populations from fragmentation and increase resilience by enhancing sediment stability through application of coconuts fibber mats (e.g., Sousa et al. 2017a) (2) Transplantation of *Z. noltei* plots from selected donor sites within Ria de Aveiro (e.g., Suykerbuyk et al. 2016)	(1) Restoration of salt-marsh communities (Sousa et al., 2017b), namely *Juncus maritimus*, through revegetation of sheltered mudflats, considering submersion time. (2) Foster saltmarshes elevation through accretion.

Data source: Lillebø et al. (2019)

and secondary) as well as regulation and maintenance, cultural and provisioning ecosystem services and abiotic outputs.

Although dredging activities and extension of the flood bank are acknowledged as important for shipping and agriculture, concerns remain on the impact of dredging on seagrasses, saltmarshes and juvenile fauna due to changes in the ecosystem eco-hydrology.

Overall, stakeholders' sectoral activities (including public and private sectors), or spheres of interest (including direct and indirect users), generate conflicting interests that need to be considered in the context of co-creation of adaptive management solutions that consider better coordination among policies.

The specific policy plans and programmes aiming at mitigating negative unintended impacts on biodiversity in the Natura 2000 Vouga estuary will focus on restoration of tidal wetlands, namely seagrasses and saltmarshes (Table 5), development of the Vouga Estuary Management Plan, engagement of local users and landowners in restoration actions, and the promotion of the value of ecosystems services provided by tidal wetlands. Both measures, to compensate for the loss of seagrasses and saltmarshes, have as policy target the Water Frame Work Directive and the Birds and Habitats Directives. As well, the target policy instruments already in place are River Basin Management Plan and National Water Plan.

3.5 Evaluate the EBM Solutions

The EBM plan, co-created with stakeholders, is shown in Table 6. During the evaluation processes, special attention was given to seagrasses and saltmarshes restoration measures. These measures aim at recovering the ecological processes and services of these valuable coastal wetlands, being in this way ecologically sustainable, socially desirable, ethically defensible, and culturally inclusive.

Table 6 EBM plan alignment with EBM principles

EBM principles	EBM scenario
1. EBM considers ecological integrity, biodiversity, resilience and ecosystem services	The harmonised WFD and HD monitoring programmes will together with the proposed tidal wetlands restoration measures, and stakeholder participation, increase resilience and ecosystem services.
2. EBM is carried out at appropriate spatial scales	The EBM Plan considers the boundaries of the Vouga Estuary Management Plan and the interconnections with the Ria de Aveiro watershed.
3. EBM develops and uses multi-disciplinary knowledge	The Vouga Estuary Management Plan foresees the coordination between various territorial management instruments as well as stakeholders' sectoral activities, with support of science-based knowledge. The Vouga Estuary Management Plan should therefore involve complementary expertise between and within natural and social sciences, in a trans-disciplinary approach.
4. EBM builds on social–ecological interactions, stakeholder participation and transparency	The EBM plan was co-created with input from local stakeholders and policy-makers, and considers their perceptions, namely their concerns regarding the unintended pressures from the base-line scenario, their valuation of ecosystem services through spatial multi-criteria analysis and their recommendations regarding opportunities and constraints regarding implementation of the plan.
5. EBM supports policy coordination	The proposed EBM approach is timely to the Portuguese spatial planning and water planning systems, framed for the protection and management of estuarine systems. The EBM plan also proposes to harmonise Water Framework Directive and Habitats Directive monitoring programmes.
6. EBM incorporates adaptive management	The proposed measures, namely the habitats restoration measures, follow principles of resilience thinking and adaptive management, by considering ecology, management of natural capital and systems analysis.

Source: Lillebø et al. (2019)

Additionally, the relevance of these coastal wetlands as nursery areas, which support important economic activities in the region, is acknowledged by local populations (Dolbeth et al. 2016; Newton et al. 2018; Lillebø et al. 2019).

For the implementation, relevant EU funding instruments might be considered, namely R&I H2020 and the following Horizon Europe programmes, LIFE environmental programme, as well as Regional Development and/or Territorial Cooperation funds (Marino et al. 2014; UE 2016).

Both measures to restore tidal wetlands have the same policy target (i.e., Water Framework, Birds and Habitats Directives), are legally permissible, and are implementable using the same policy instrument (River Basin Management Plan; National Water Plan), therefore administratively achievable although it implies the commitment of several Institutions. This is foreseen with the proposed development of the Vouga Estuary Management Plan and is also effectively communicable and politically expedient for promoting the value of ecosystems services provided by tidal wetlands. In addition, effective implementation of proposed habitat restoration in the selected Natura 2000 area is consistent with the prevailing political climate and has explicit support of national political leaders. The main differences between the baseline and the proposed EBM solutions are shown in Table 7.

The performance of the proposed EBM measures is presented in Table 8. The baseline scenario corresponds to the unintended impacts on biodiversity, i.e., increase in tidal prism and water velocity; loss of coastal wetlands habitats (seagrasses and saltmarshes) and saltmarsh 'coastal squeeze' at the downstream area of the flood bank.

4 Vouga Estuary Natura 2000 Site Stakeholders' Evaluation and Feedback

Local stakeholders were supportive of the approach, "*ecosystem-based management allows for a 'correction' of less good results*" and appreciate that it is "*concerned with beneficiaries, as well as biodiversity*". Overall, stakeholders considered that:

- *The environment and biodiversity will be the main beneficiaries from tidal wetlands restoration;*
- *Some economic activities related to fisheries and ecotourism, which has a recognized potential, might benefit;*
- *Restoration actions need to ensure involvement of users due to conflicting activities and landowners, as most of the area is private property;*
- *Large interventions should include financing for implementation of the corresponding minimizing measures;*
- *There is a need for post-licensing supervision to ensure compliance with environmental protection obligations;*
- *There is a need for clear communication between institutions and enforcement of existing regulations;*

Table 7 Main differences between the baseline and proposed EBM solutions

Main differences	Baseline	EBM solutions
Environmental ambition/policy target	Protect biodiversity in line with Natura 2000 objectives. Whilst enabling economic and other activities in the area, aim to mitigate negative impacts of interventions and economic activity.	Same as in baseline.
Measures	Two measures are to be implemented in 2019/2020, which will have foreseen but unintended negatives impacts on biodiversity (dredging programme and the extension of a flood bank).	The same as baseline, but to minimise negative side effects, additional measures are proposed (see Table 5).
Policy instruments	Many policy instruments are implemented to achieve biodiversity goals, including protected areas.	Harmonise monitoring across water and environmental related Directives; Incorporate stakeholders into planning; Integrate territorial management institutions (and their multiple goals) into planning; Support development of Vouga Estuary Management Plan.
Sites	The boundaries of the Vouga Estuary Management Plan and the interconnections with the Ria de Aveiro watershed.	Same as in baseline; Seagrass and saltmarsh restoration sites will be selected considering multiple ecosystem services and with stakeholder input.
Governance/ Institutional context	Many separate; Limited, inconsistent stakeholder involvement in management.	Coordinated input from multiple institutions into integrated Vouga Estuary Management Plan. Ongoing, coordinated stakeholder engagement in management.

Source: Mattheiß et al. (2018)

- *There is a need for reinforcement of integrated management and development the Vouga Estuary Management Plan.*

As part of the co-creation process, stakeholders evaluated the produced maps, the ecosystem indicators' results, the proposed EBM solutions, and they discussed the benefits and constraints regarding its implementation. As a final remark stakeholders acknowledged that responses should be framed in the Sectoral Plan for Natura 2000 Network, and should consider climate change projections and the National Strategic Plan for climate change adaptation (Fig. 8 illustrates the spatial planning regulations to consider for climate change adaptation in the region).

At the Vouga river coastal watershed the Sectoral Plan for Natura 2000 Network establishes the strategic orientation and programme norms for the actions of central and local government, and the measures and guidelines provided therein should be transposed to the Municipal Planning of the territory and Special Plans. Thus, the management measures provided for the Sectoral Plan will only be binding measures when they are inserted in the Municipal and Special Plans. Within the considered

Table 8 Application of the pre-screening criteria: effectiveness, efficiency, and equity and fairness

Pre-screening criteria	EBM solutions
Effectiveness—hitting the environmental target	The proposed measures, i.e., saltmarshes restoration and seagrasses restoration clearly address the set of environmental targets defined under relevant policies.
Efficiency—making the most for human wellbeing	Although proposed measures were not supported by cost-effectiveness analysis or cost-benefit-analysis, they were supported by a spatial multi-criteria analysis performed by stakeholders in which they expressed preferences regarding the provided ecosystem services.
Equity and fairness—sharing the benefits	By integrating the proposed measures in the Vouga Estuary Management Plan, and by acknowledging the context of adaptation to climate change, strategy for Biodiversity and the Centro Portugal region strategy for smart specialisation (Portugal RIS3 Centro), stakeholders with very different interests participated actively and acknowledge the equity of benefits already achieved, or to be achieved, although they identified constraints regarding its implementation.

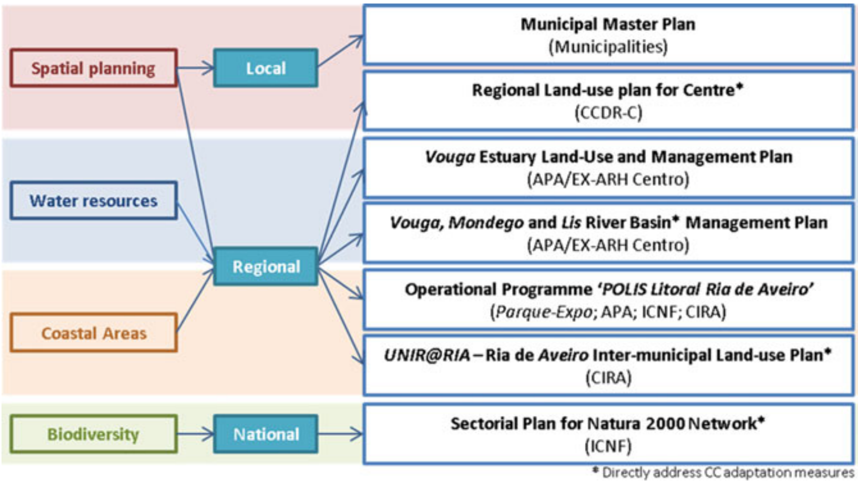

Fig. 8 Spatial planning regulations for climate change adaptation for Vouga river coastal watershed. (Source: ADAPT-MED 2015)

boundaries, this plan is implemented by the UNIR@RIA, which ensures articulation between the regional and municipal plans that are relevant for the Ria de Aveiro and associated protected areas.

The National Strategic Plan for climate change adaptation is framed on a Territorial Management System organized within a framework of coordinated interaction at three levels (Law no. 48/98, August 11): national, regional and local.

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Ecosystem-Based Management for More Effective and Equitable Marine Protected Areas: A Case Study on the Faial-Pico Channel Marine Protected Area, Azores

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Abstract Marine Protected Areas (MPAs) are increasingly employed as a tool to protect Europe's swiftly declining marine biodiversity. However, despite increasing coverage, MPA effectiveness and equity is considered highly variable. Concurrently, Ecosystem-Based Management (EBM)—that is, management that aims to protect, restore, or enhance the resilience and sustainability of an ecosystem to ensure sustainable flows of ecosystem services and conserve its biodiversity—is growing in prominence. We applied EBM in the Faial-Pico Channel, a 240 km^2 MPA in the Azores, Portugal, to assess whether EBM can protect biodiversity whilst meeting diverse stakeholder and policy goals. Collaborating with local stakeholders and policy-makers, this chapter documents the steps of EBM: identifying integrative policy and stakeholder objectives, understanding the social-ecological system, scenario development, and identification and evaluation of EBM measures and policies. We find that stakeholder co-creation and collaboration is a key strength of EBM and should be strengthened in the Faial-Pico Channel. We find that local stakeholders support effective and equitable EBM of MPAs by clearly identifying challenges and priorities, co-creating solutions, providing low-cost knowledge and expertise, and

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through ongoing monitoring, enforcement, and evaluation of the impact of management.

Lessons Learned

- Stakeholder engagement and participation supports long-term sustainable protection of biodiversity and equitable and effective management of MPAs
- Stakeholders can contribute at each stage of EBM: identifying social objectives, understanding the social-ecological system, identifying an EBM plan, and evaluating impact
- Stakeholders contribute by clearly identifying challenges and priorities, co-creating solutions, and generally by providing low-cost knowledge and expertise, as well as increasing societal acceptance.
- EBM is an appropriate framework for increasing effectiveness and efficiency of MPAs

Needs to Advance EBM

- Clear guidance on how to effectively engage stakeholders at each stage of the EBM process
- EBM has high environmental and socio-economic data demands. Guidance on how to apply EBM in low-data environments would support uptake.

1 Introduction

Globally, marine biodiversity declined by 49% between 1970 and 2012 (Tanzer et al. 2015). This rapid decline threatens the resilience of marine ecosystems and their ability to sustainably produce ecosystem services that humans depend on to survive and thrive (Cardinale et al. 2012). Policy makers have turned to Marine Protected Areas (MPAs) as a key tool to reverse marine biodiversity loss (Gill et al. 2017). Indeed, globally, the Convention of Biological Diversity's Aichi Target 11 and the UN Sustainable Development Goal 14 aim to "efficiently and equitably" protect 10% of coastal and marine areas within MPAs (UN 2016; Secretariat of the CBD 2011). However, the efficacy and equity of MPAs is questioned and considered highly variable (Gill et al. 2017).

Researchers, policy-makers, and environment managers are increasingly interested in the ecosystem-based management concept as a promising approach to more effectively, efficiently, and equitably manage aquatic ecosystems (see, e.g., Delacámara et al. 2020). Ecosystem-based management (EBM) is a principle-based management approach that aims to protect, restore, or enhance the resilience and sustainability of an ecosystem to ensure sustainable flows of ecosystem services and conserve its biodiversity (see Gómez et al. 2017; Rouillard et al. 2017). While there is increasing interest in ecosystem-based management, there are still relatively few practical examples worldwide, especially as applied to Marine Protected Areas.

This chapter presents a summarised excerpt from the more detailed AQUACROSS project case study report (McDonald et al. 2018), documenting the application ecosystem-based management (EBM) in the richly biodiverse Faial-Pico Channel, a 240 km^2 Marine Protected Area in the Azores, Portugal. We include it in this book as it illustrates in an integrated manner how each of the concepts developed in the AQUACROSS project can be combined to practically apply EBM to manage biodiversity. To apply ecosystem-based management, we collaborated with local stakeholders and policy-makers and follow the AQUACROSS Assessment Framework (Gómez et al. 2017). The chapter aims to: (1) demonstrate how the AQUACROSS Assessment Framework can be followed to practically apply ecosystem-based management; (2) identify how ecosystem-based management can protect biodiversity and improve social welfare in the specific context of the Faial-Pico Channel social-ecological system, and (3) understand how ecosystem-based management generally can support existing MPAs to become more effective and equitable.

2 The Faial-Pico Channel Marine Protected Area: Case Study Context

The Faial-Pico Channel is rich in biodiversity, and its complex of habitats, species, and ecological processes is recognised as one the most diverse and representative complex of habitats in the Azores archipelago (MarBEF Data System 2006; OSPAR Commission 2016). However, despite a 30 year history of increasing international, Azorean, and local protection for the area (Abecasis et al. 2015), biodiversity in the MPA continues to be lost, as indicated by falling population indices of target coastal species in the channel (Afonso et al. 2014).

Numerous human activities in the Channel place pressure on the ecosystem, especially fishing and tourism. Fishers and tourism operators (including diving operators) value the biodiversity hotspots within the Channel, but have different objectives for how they should be managed. It is important to balance these objectives, as both tourism and fisheries are important local industries for the 30,000 people who live on Channel's neighbouring islands. Commercial fisheries are a historically important driver of the local economy, and still employ 1.5–3.2% of the total working population (Ojamaa 2015; Statistics Portugal 2017).[1] Tourism has swiftly become central to the local economy, with the number of tourist nights in the Azores tripling from 1995–2015; in 2016, tourists spent 228,000 nights on the islands (SREA 2017). As one indicator of the sector's importance, in 2015, the

[1]Statistics Portugal: own calculations, Fishermen registered at 31 December 2015 in Azores. This compares to a rate of 0.6% for Portugal.

accommodation sector directly employed 2% of the total Azorean workforce.[2] This has supported economic growth, with GDP per capita growing at 2.7% per year since 2000 (currently at €16,000).[3]

The increased demand by tourists (and tourism providers) for eco-tourism in the Channel and declining biodiversity is leading to conflict between commercial fishers and other stakeholders as to how the Channel should be managed (AQUACROSS 2017). Managing the Channel is complicated by multi-level and overlapping responsibilities, with policy development and enforcement split across the local-level Nature Park of Faial and Nature Park of Pico, both under the mandate of the Regional Directorate for the Environment (Direcção Regional do Ambiente, DRA). Other relevant managing authorities include the Azores-level Regional Directorate for Sea Affairs (DRAM) and the Regional Directorate for Fisheries (Direcção Regional das Pescas, DRP), all who must consider local (i.e., Faial and Pico Island), Azorean, Portuguese, and EU policy targets.

In response to falling local biodiversity and to balance stakeholder competition for space, local authorities have extended Marine Protected Area to cover the Faial-Pico Channel. Parts of the Channel have been protected under local policy as a MPA since 1980, with this extended under NATURA 2000 protection in 1995, and OSPAR coverage in 2006, and consolidated under new Azorean Island National Park regulation in 2007 (Abecasis et al. 2015).

Dovetailing this government push for increased biodiversity protection, bottom-up stakeholder demands have driven Faial-Pico Channel management, resulting in an increase in stakeholder participation in MPA management. An early, nearby example was the Condor Seamount, which in 2010 following a stakeholder participatory process was designated a temporary MPA to facilitate marine research (Ressurreição and Giacomello 2013; Ressurreição et al. 2017). Following this and other Azorean examples, local government and scientists supported Faial-Pico tourism operators when they published an open letter calling for an extension of MPA coverage in the Channel to promote non-extractive recreational activities, instigating two stakeholder meetings to gather input on MPA management revisions. While these workshops lacked sufficient representatives from the tourism sector and no recreational fishing representatives, they represent more inclusive management of the MPA by local authorities and the resulting change in law (Ordinance 53 2016) increased protection for some of the high biodiversity zones in the Channel. Within this context—of falling biodiversity, increased competition for the Channel, and at the same time more inclusive MPA management—our application of ecosystem-based management aims to build on previous policies and approaches and identify how local authorities and stakeholders can increase the effectiveness and equity of Faial-Pico Channel MPA management.

[2]Eurostat: own calculations, SBS data by NUTS 2 regions and NACE Rev. 2 (2014–2016). This compares to a rate of 2.3% for Portugal.

[3]EUROSTAT: GDP at current market prices by NUTS2 region.

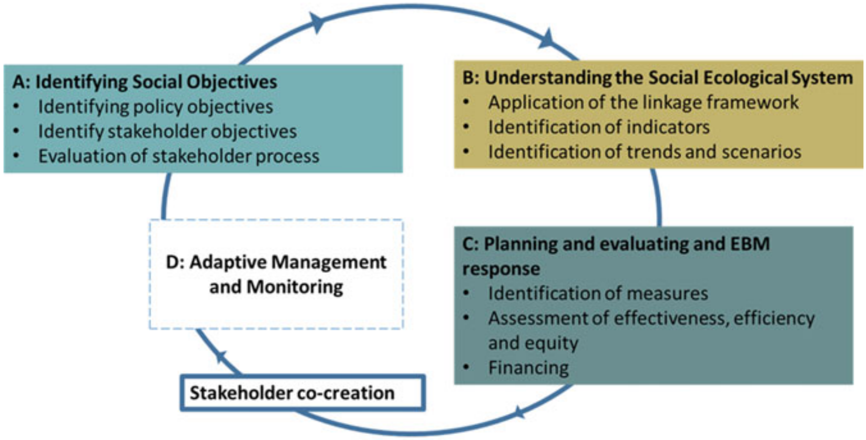

Fig. 1 AQUACROSS Assessment Framework, as applied in this case study

3 Methodology

To apply ecosystem-based management, we followed the AQUACROSS Assessment Framework (Gómez et al. 2017). As shown in Fig. 1, we applied this in three overlapping steps.[4] Below, we describe the different methodologies applied at each step, as well as how stakeholder co-creation supported the whole process.

3.1 Stakeholder Co-creation

Common to our methodology at all steps was co-creation with local stakeholders. Given EBM's ambition to reflect the complexity and multifunctionality of the Faial-Pico Channel, diverse representative stakeholder participation was required. We mapped stakeholder interest and influence, using snowball sampling to identify and recruit diverse stakeholders (following Reed 2008). Through phone and in-person semi-structured interviews and small meetings we gathered input and feedback from all key stakeholders including recreational and commercial fishers, diving operators, environmental NGOs, scientists, and representatives of all relevant policy ministries and departments (Regional Directorates). Stakeholders also identified issues, shared their views, and provided input and feedback at two workshops: (1) Stakeholder workshop 1—Horta—3rd of October, 2017*:* 31 local stakeholders discussed the current and future management of the Faial-Pico Channel MPA, and how science and local knowledge can support policy (AQUACROSS 2017);

[4]Due to the timing of the case study, we did not progress to applying the fourth step of adaptive management and monitoring.

(2) Stakeholder workshop 2—Horta—23rd of May, 2018: 18 local stakeholders collaborated on a concrete plan for stakeholder-based management of the Faial-Pico Channel MPA, and prioritised and developed measures to managed the Channel (AQUACROSS 2018).

(A) Identifying Policy and Stakeholder Objectives
To understand *policy objectives* we applied at a local level Rouillard et al.'s (2017) approach and reviewed relevant Faial-Pico and Azores regulations, laws and strategies related to the environment, fishing, and tourism, i.e., the sectors driving pressures on local biodiversity. We assessed key features, implementing measures, and governance of the most important local policies, and applied the Driver-Pressure-State-Impact-Response model to identify the expected pathway through which the management measures impact biodiversity in the Faial-Pico Channel, i.e., how the policy affects ecosystem state, pressures, or drivers. Finally, we identified synergies, conflicts, and gaps in relation to how local management and policy affects biodiversity in the Channel, and how biodiversity protection could be improved. To understand *stakeholder objectives* we relied on stakeholder interviews and the two workshops. To understand *stakeholder processes* and to identify how current stakeholders could better support MPA management, we used the development of a recent relevant policy Fishing Ordinance no. 53/2016 as a case study, evaluating how existing stakeholder processes could be adapted to the requirements for EBM.

(B) Understanding the Social-Ecological System
We applied the AQUACROSS Linkages Framework to understand the current Faial-Pico Channel socio-ecological system (Robinson and Culhane 2020). We mapped marine habitats present in the Channel and then used expert judgement, local scientific reports and economic and environmental data, and interviews with local scientists and regulators to identify drivers and activities, the pressures these place on habitats, and link these habitats to ecosystem-services production. Having identified key elements in the Channel's simplified social-ecological system (see Fig. 2), we then identified indicators and collected data on state and trends. We presented this to stakeholders at workshop 2 and co-developed future scenarios to identify future trends that would require integrative management and to identify potential trade-offs associated with different approaches for managing fishing, tourism, and biodiversity within the MPA.

(C) Identifying an EBM Plan
To identify the combination of management measures and implementing policies that make up the EBM plan, we collaborated with local stakeholders and policymakers. They suggested a long list of potential measures/policy instruments in interviews and at stakeholder workshop 1 (AQUACROSS 2017). At stakeholder workshop 2, stakeholders selected priority management measures and implementing policies and developed how these should be implemented in the Channel (AQUACROSS 2018). We then ensured the workability of these individual measures and policies and combined them into an EBM plan. Finally, we evaluated this EBM plan relative to a baseline of current management using three criteria:

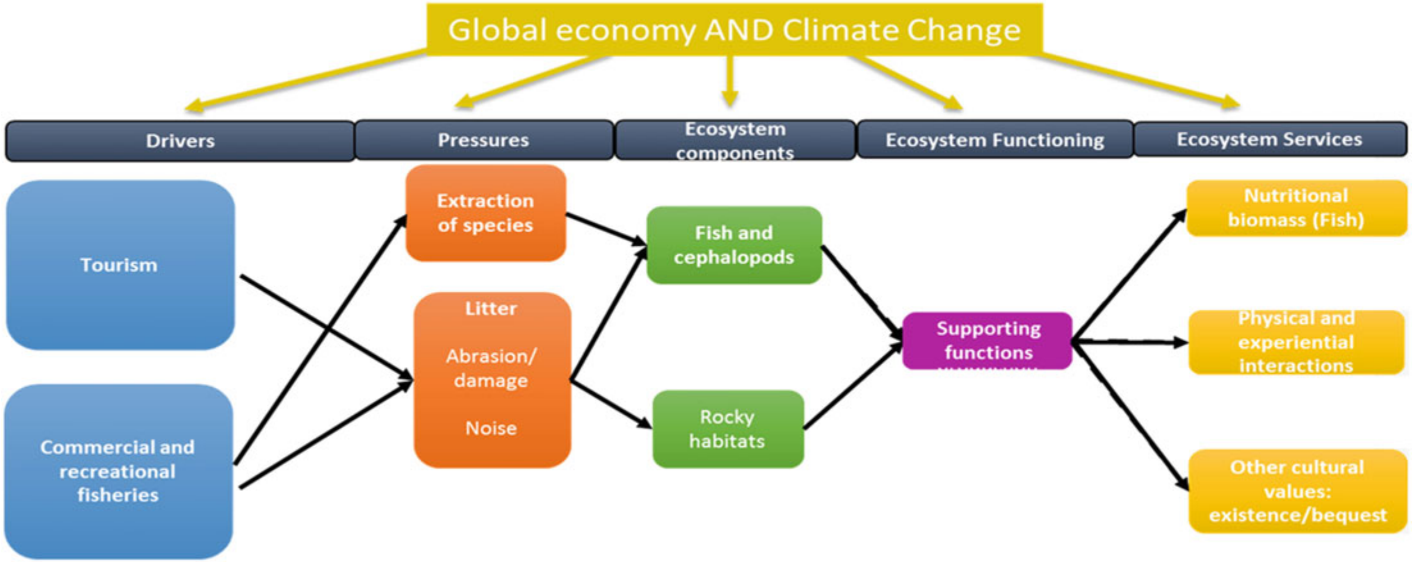

Fig. 2 Simplified linkage framework for Faial-Pico Channel, showing key drivers, pressures, ecosystem-components, ecosystem functioning, and ecosystem services

effectiveness, efficiency, and equity. Here, we drew on stakeholder and expert input and the AQUACROSS Linkage Framework to qualitatively assess direct and indirect impacts. To assess how the direct costs of the EBM plan could be financed, we interviewed participants and quantitatively assessed tax and levy impacts (following European Commission et al. 2017).

4 Results

4.1 Identifying Policy and Stakeholder Objectives

Policy Objectives

Biodiversity in the Channel is protected by environmental policies. However, as described in Rouillard et al. (2017), the positive impact of these policies can be undermined by sectoral policies, which support drivers (fishing, tourism) that place pressures on biodiversity. Together with local policy-makers and stakeholders, we concluded that, while local policy already targets sustainability, there are three policy gaps that should be priorities for improving MPA management:

- *Lack of coordinated management of the Channel limits synergies*—The current dispersion of responsibilities and management between environmental directorates (Faial and Pico Island Nature Parks, DRA, DRAM, and DRP) hinders integrated and coordinated management, implementation, monitoring, and evaluation of the Faial-Pico Channel. Leaders of the Island Nature Parks have reported lacking expertise and interest in non-terrestrial protected areas (AQUACROSS 2017). DRAM has the expertise and the mandate for coordinating and regulating the MPAs but is currently lacking operational means to implement monitoring or enforcement.
- *Issues of scale of marine resources not reflected in policy or governance*—The current split of the Channel into two separate Faial and Pico management units fails to recognise the Channel's interconnected ecosystem, and its links to the wider Azores marine ecosystem. A key benefit of MPAs are the potential positive spillover effects: MPAs elsewhere have been shown to increase species richness and catch rates in neighbouring waters (Russ and Alcala 2011). Negative spillover effects can also occur, where closure of one area increases fishing effort in boundary or neighbouring zones (Murawski et al. 2005). Managing the Channel as one integrated unit could help balance these competing spillover and network effects to meet local and Azorean biodiversity goals. In this way, the most recent MPA management regulation (Ordinance 53 2016) suggests a way forward: it was developed by DRAM in collaboration with DRP, who also manage the Azores Marine Park, thus better reflecting ecosystem scale.
- *A lack of monitoring data limits target setting and adaptive management*—Ecosystem-based management requires decision-makers to monitor policy impact and regularly revisit management tools if objectives are not being met

effectively, efficiently, and equitably (Rouillard et al. 2017). This requires monitoring and data at the appropriate spatial scale (i.e., Faial-Pico Channel), as well as clearly defined and spatially consistent policy objectives and targets. Ideally, this should include both ecological data (i.e., measures of biodiversity state, such as fish stocks) and socio-economic data (benefits and costs for society, e.g., fishing income, MPA visits). This data challenge is compounded by the issue of scale: policy objectives are set—and existing biodiversity and economic data collected—at the national (or, in some cases, island) scale, rather than at the Faial-Pico Channel-level. This makes it difficult to set and evaluate quantitative local targets. Additionally, Channel monitoring data is currently insufficient to manage biodiversity.

Stakeholder Objectives

EBM aims to maximise overall social welfare. Accordingly, it is important that as well as existing policy objectives, MPA management must consider other stakeholder goals. In the Faial-Pico Channel, there was considerable overlap between policy objectives and stakeholder priorities, but we did identify additional stakeholder objectives, some of which all stakeholder groups shared, and others where different groups were in conflict.

- *Shared stakeholder objectives*: Stakeholders all recognised that they share the Faial-Pico Channel MPA and come from the same community. Accordingly, all stakeholder groups share four central objectives: long-term sustainability, simplified and holistic management of the Channel, regular monitoring, and ongoing participatory management. (AQUACROSS 2017, 2018).
- *Conflicting stakeholder objectives*: The major stakeholder groups within the Channel also have conflicting objectives (AQUACROSS 2017, 2018). Additionally, as the Channel consists of many distinct habitats, stakeholders also place different value on different parts of the Channel (Schmiing et al. 2015; Afonso et al. 2014). For example, commercial fishers' prioritise access to fishery grounds, which can be in conflict with recreational fishers wish for extended catch limits and tourism operators' desire for expansion of the MPA to protect biodiversity and restrict extractive uses.

Enhancing cooperation and managing these conflicts relies on transparent and inclusive governance, which stakeholders believe could additionally decrease conflict, increase knowledge, and motivate greater environmental protection (AQUACROSS 2018).

Stakeholder Processes

Stakeholder processes are central to EBM, and given the gap we identified between policy objectives and stakeholder objectives and the presence of stakeholder conflicts, we evaluated existing stakeholder processes for integrating stakeholders into policy development. We found that while policy-makers' development of a non-technical scientific report (Afonso et al. 2014) and stakeholder workshops were positive steps in enabling stakeholders to contribute to policy design/

development, low participation from two key sectors—recreational fishing and tourism operators—meant the process was not representative. A second conclusion was that stakeholders should be involved throughout the policy cycle, not just in the policy development stage. Such adaptive management requires ongoing monitoring, evaluation, and, if necessary, adaptation of any management measures. This ongoing stakeholder engagement, for example through clear communication or regular workshops, would help ensure that decision-makers have full information on stakeholder objectives and priorities and feedback on whether current management is optimal or needs adjustment.

4.2 Understanding the Social-Ecological System

The second step of the AQUACROSS Assessment Framework is to understand the Faial-Pico Channel Social-Ecological System (SES). Effective management requires an understanding of how society affects the ecosystem, and how the ecosystem provides benefits to society, as well as the complex processes within the SES. We used the AQUACROSS Linkage Framework and developed indicators to understand the current state of the SES, and also used co-developed scenarios to identify potential future challenges and trends that would need managing.

Linkage Framework Analysis

Figure 2 presents a simplified social-ecological system for the Channel. We find that biodiversity in the Faial-Pico Channel is affected by the society that surrounds it: human activities like fishing and tourism place pressures on the Channel. These pressures affect the ecosystem's health and its ability to deliver valuable ecosystem services, such as fish and recreational experiences, which drive human activities and responses.

Our analysis shows that both the key sectors of fishing and tourism place many of the same pressures on the ecosystem, such as litter and noise. Unsurprisingly, fishing is most associated with the key pressure of extraction of fauna and flora. The linkage framework also assesses impacts over time: we find that fishing exerts more acute pressures, while tourism is associated with pressures that are more chronic. Accordingly, policies targeting fisheries will more swiftly decrease pressures than tourism-targeted policies.

We also used the Linkage Framework to assess which ecosystem components were most central to the Faial-Pico Channel SES. Fish are highly valued by all stakeholders. We find that rocky habitats support the most ecosystem functions and were associated with the most ecosystem services. This aligns with recent research on values of biodiversity indices around the Faial and Pico islands, which shows that the highest values were linked to rocky habitat, which provide refuge and substrate for various marine species, making rocky habitats important sites for fishing and diving (Schmiing et al. 2014). These insights suggest that management should prioritise protection of these central and valued ecosystem components.

Indicators
Our development and evaluation of indicators suggests that policy-makers can use indicators to understand the system, set quantitative targets, and monitor and evaluate trends and the impact of management measures. However, a key conclusion of this exercise was that a lack of quantitative Faial-Pico Channel data limits ability to apply EBM. The small scale and trans-boundary nature of the case study makes it difficult to use Azores-level data. Ecosystem-based management of the Channel calls for collecting and developing more specific Faial-Pico Channel data, especially to measure the current state of the ecosystem and its biodiversity, and on flows of key ecosystem services (fish for consumption, recreational experiences, and existence/bequest values).

Future Scenario Development
Scenarios are valuable as they provide a vehicle for incorporating diverse information into a comprehensive, actionable vision of the expected future (Gómez et al. 2017). Together with Azorean stakeholders and policy-makers (AQUACROSS 2018), we reflected on the understanding of the current SES, as well as our understanding of policies and stakeholder objectives, to develop identify what 2018–2050 is likely to bring to the Channel:

- **Climate change** will impact all sectors, increasing variability and uncertainty.
- The **global economy** will continue to drive ongoing—but fluctuating—growth.
- **Tourism** will continue to grow economically—with more visitors, income, and infrastructure.
- These changes mean **marine biodiversity** will be under increasing pressure in Faial-Pico Channel.
- **Commercial fisheries** and **recreational fishing** will remain central to local life, but sensitive to uncertain trends in fish stocks and biodiversity.

Developing this scenario clarified the gaps between current management (and the resulting expected future) and the future stakeholders and policymakers and stakeholders desired. Overall, we concluded that all stakeholders depend on a sustainable and resilient ecosystem. Given the large uncertainties and unknowns, stakeholders and policy-makers need to be adaptive—employing regular monitoring, evaluation, and if necessary, management changes.

4.3 *Identifying an EBM Plan*

Our final steps in applying ecosystem-based management in the Faial-Pico Channel was to reflect on identified objectives and policy gaps, and draw on our understanding of the current and future state of the SES to identify a set of priority management measures and implementing policies (the EBM Plan). We then evaluated the extent to which this EBM Plan would increase effectiveness, equity, and efficiency relative to a baseline of current management. We also investigated how regulators could finance the EBM Plan, which has important equity affects as well as being crucial for MPA effectiveness (Gill et al. 2017).

EBM Plan

We identified the following measures and policies as priorities for EBM management of the Faial-Pico Channel:

1. **Increased monitoring of biodiversity**
2. **Increased stakeholder participation through a Stakeholder Advisory Group** consisting of representatives of all sectors.
3. **Integrate and coordinate Channel management** through a Marine Protected Area management plan and policy coordination group.
4. **Clear communication and enforcement of existing regulations**—e.g., through simple information panels and surveillance cameras
5. **Implement a sustainability tax**—a tourism tax/diving fee.

Evaluation of the EBM Plan

Effectiveness: Due to data and methodological limitations, we are unable to decisively quantitatively assess how the EBM Plan will affect biodiversity (i.e., its environmental effectiveness). The EBM plan has direct impacts on biodiversity by increasing enforcement and awareness of existing fisheries/biodiversity regulation, which will increase compliance and decrease a key pressure on local biodiversity, extraction of species. The implementation of a sustainability tax will marginally decrease tourism and related pressures. The EBM Plan would also have indirect positive impacts on biodiversity by increasing scientific knowledge and financing to support management, policy integration, and stakeholder cooperation. Stakeholders believe that a stakeholder advisory group would result in greater environmental protection and increases in biodiversity (AQUACROSS 2018).

Efficiency: Assessing economic efficiency of the EBM Plan requires an understanding of its direct and indirect costs and benefits. However, given the indirect, supporting nature of the majority of elements of the EBM Plan, we cannot quantitatively assess this. Using the AQUACROSS Linkage Framework, we find that there is uncertain impacts on the value of fish caught to be eaten; increases in the existence/bequest value of the system; and likely increases in the value of experiential/physical interactions with the ecosystem. Alongside this qualitative assessment, evidence of efficiency is provided by the fact that each of the policy instruments that form the EBM plan were co-created with local stakeholders, whose selection of the plan, who believe that the benefits of the plan will outweigh the costs (AQUACROSS 2017, 2018).

Equity: A key focus of the EBM Plan is to increase stakeholder involvement and ownership of MPA management in such a way that the EBM Plan recognises and balances the costs and benefits to different stakeholder groups, and focusses on synergies and a shared commitment to environmental sustainability. Indeed, all stakeholders prioritised this cooperative, participatory element of the EBM Plan, arguing that it would decrease conflicts between different users and policy entities through better communication, and the promotion of multiple uses of the Marine

Protected Area (AQUACROSS 2018); all evidence of greater equity under the EBM Plan than under current management.

Financing: The first four elements of the proposed EBM plan place costs on fishers (who will face increased enforcement and compliance costs), while tourists, tourism operators, and other local stakeholders benefit (both from exclusive access to diving locations and positive environmental impacts). Financing can be used as a way to share the costs between those who benefit and those who bear cost. Our assessment of two financing options (a per dive fee levied by tourism operators and a per night occupancy tax) suggests that even at low rates of €2 per dive or €0.25 per night, either of these options could cover the likely direct costs of the EBM Plan and share the costs between different stakeholder groups to improve equity.

4.4 Local Policy Recommendations

Overall, our co-development of an EBM plan for the Faial-Pico Channel with stakeholders resulted in the following set of complementary management measures and policy instruments: (1) increase scientific monitoring, (2) implement stakeholder co-management with a Stakeholder Advisory Group, (3) increase integration and coordination of Channel management (e.g., by means of a coordination group of fishing, tourism, and environment Regional Directorates and island national parks); (4) communicate and enforce existing fishing and biodiversity regulations, and (5) finance biodiversity protection and share costs. This plan would better protect Channel biodiversity, whilst also ensuring economic and social sustainability. A key element of this plan is extending the stakeholder participation and policy cooperation that was evident in the EBM process and in existing local government stakeholder engagement efforts. In light of the Azores government's strategic goal of increasing MPA coverage, to ensure their success, we encourage continued engagement of stakeholders in planning, implementation, and evaluation. This, along with increased scientific knowledge and cross-sectoral policy coordination, will enable adaptive management in the Channel, reduce stakeholder conflict, and can improve effectiveness and efficiency of management, delivering benefits to the whole community into the future.

5 Conclusion and Discussion: How Can Ecosystem-Based Management Support Effective and Efficient Management of Marine Protected Areas?

We conclude that the Faial-Pico Channel case study provides evidence that ecosystem-based management and the AQUACROSS Assessment Framework can support decision-makers to manage Marine Protected Areas more effectively, so that they equitably meet biodiversity goals, both in the specific case of the Faial-Pico Channel and more generally in existing MPAs.

Our key conclusion is that stakeholder engagement and participation is beneficial for long-term sustainable protection of biodiversity and equitable and effective management of MPAs, and that ecosystem-based management's placing of representative stakeholder participation at the centre of ecosystem management is its key strength. Stakeholder engagement and participation has value in its own right. Reed (2008) reviewed stakeholder engagement literature and found that it promotes active citizenship, increases public trust, empowers stakeholders through co-generation of knowledge, improves public perception of policy, promotes social learning, and can reduce conflict between stakeholders and lead to creative solutions to environmental problems. In addition, stakeholder engagement is one of the defining principles of EBM (Long et al. 2015; Gómez et al. 2017). Stakeholder co-creation within this case study increased the relevance, acceptance, and quality of the management plan, and, as recognised by stakeholders, promotes synergistic solutions that provide multiple benefits, reducing stakeholder conflict, as well as improving knowledge and justifying more biodiversity protection (AQUACROSS 2018). It can be challenging involving stakeholders: for example, we found some stakeholders are harder to involve than others, and the process can be time-consuming, focused on discussion rather than action. However, on balance, we believe that the benefits of stakeholder co-creation outweigh these costs. This conclusion aligns with recent participatory management initiatives within the Azores, such as Condor seamount (Austen et al. 2019) and the Azorean fisheries regulation (Ordinance 53 2016) that increased protection for some high biodiversity areas in the Faial-Pico Channel. Our case study built on these initiatives and underlines the importance of integrated and representative management as a way to cope with the complexity and interlinkages of marine social-ecological systems.

Our experience also identified other strengths and challenges of ecosystem-based management for managing Marine Protected Areas. We found that ecosystem-based management provides a framework for integration of diverse stakeholders and objectives (biodiversity/environmental and sectoral). This integration clarifies the interconnectedness of the social-ecological system, and strengthens understanding of and arguments for collaborative, sustainability-focussed long-term ecosystem management. Key challenges that we faced were that while the interdisciplinary work of ecosystem-based management results in more useful and impactful policy, it requires diverse expertise and sometimes challenging cross-sectoral and cross-disciplinary collaboration and communication. Additionally, the newness and apparent complexity of the interdisciplinary work can make it challenging to get buy-in from sectors and policy makers. Finally, while EBM's emphasis on science-informed management are likely to support effective biodiversity protection, data and methodological limitations were a challenge in our case study.

It is too soon to evaluate the impact of the Faial-Pico MPA EBM process, though we conclude that the process had stakeholders' support and that it contributed to sustainable marine policy development in the Azores. Stakeholders demonstrated their support for the EBM process through their participation and positive comments in the workshops (AQUACROSS 2017, 2018). In particular, stakeholders supported EBM's commitment to representative stakeholder participation in policy development (AQUACROSS 2018). Alongside concurrent Azores projects and policy development, the case study and resulting EBM plan support ongoing MPA policy development and increasing stakeholder involvement in Azores marine policy, as evidenced by current processes to update Azorean MPA policy.

Overall, The Faial-Pico Channel EBM Plan, and its development and evaluation, provide evidence of how ecosystem-based management can support existing and future marine protected area management. The results are relevant in the Azores, where the government is committed to expanding MPA coverage, and globally to meet international MPA coverage targets. This study provides valuable information on how participatory management can support effective and equitable MPAs through clear identification of challenges and priorities, creative co-creation of solutions, low-cost knowledge and expertise, and ongoing monitoring, enforcement, and evaluation of the impact of management.

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Using Stakeholder Engagement, Translational Science and Decision Support Tools for Ecosystem-Based Management in the Florida Everglades

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Abstract Managing water for competing human and environmental demands in the Greater Everglades is a multi-dimensional challenge that includes managing a complex ecological system while providing water supply and flood control for widespread high-density urban communities and nationally important agricultural lands. Ecosystem-Based Management (EBM) in the Florida Everglades is examined at multiple spatial and temporal scales. There is a corresponding increase in the number and diversity of stakeholders involved as the temporal and spatial scales of management across the landscape increases. Therefore, translational science, decision support tools, effective stakeholder engagement, and communication are paramount. This chapter provides a case study of EBM in an aquatic system facing ecological challenges, such as eutrophication and non-indigenous species management, which are framed by complex social, cultural, and political contexts. A framework for navigating multi-agency governance models and competing stakeholder visions using socio-ecological science (i.e., science of interlinked human and natural systems) to address practical and theoretical challenges for managing freshwater wetlands is discussed. By examining best practices in stakeholder engagement and linking translational science with multiple, science-driven decision support tools, important lessons learned can be carried forward in an effort for continually improved governance and collaboration for ecosystem management and restoration.

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Lessons Learned

- Real-time monitoring data, frequently updated and accessible modeling output and stakeholder communication are key to successful EBM in complex socio-ecological systems, such as the Everglades.
- EBM recommendations should consider agency-specific missions and goals across the key stakeholders involved.
- Coordinating short, intermediate, and long-term recommendations at both local and regional scales may improve EBM outcomes.
- Incorporating real-time monitoring data from across the landscape, along information from model output, enhances EBM.
- Decision support tools that integrate monitoring data with spatial habitat and wildlife models enhance assessment of conditions and development of recommendations that support EBM.

Needs to Advance EBM

- Additional development of the decision-making process framework and further integration of decision support tools for use in multiple spatial and temporal scales.
- Increased/enhanced incorporation of EBM approach into existing governance models/mandates, and methods of communication of recommendations to managing agencies.
- Perhaps most importantly, increased connection of operational decisions to measuring the resulting ecosystem responses to support adaptive management will further enhance the success of the EBM approach in the Everglades.

1 Introduction

Wetlands are productive ecological systems that provide habitat to many species that form complex and interdependent communities. Wetland systems collect water and sediment from across the landscape and regulate hydrologic cycles that provide ecosystem services such as water filtration and supply, flood control, coastal protection from storms, carbon sequestration, natural products (e.g., shellfish), and recreational opportunities. Humans often alter these systems by draining them to provide fertile farmland or to support development. The hydromorphological alterations resulting from these socio-ecological interactions often lead to impacts such as decreased wildlife populations and reduced ecological service. However, when adaptively managed, wetlands can be sustainable and provide a range of ecosystem services to humans while providing crucial habitat to wetlands.

This chapter outlines key socio-ecological interactions across the landscape and the Ecosystem-Based Management (EBM) approach used to promote restoration of the natural function of the Everglades system while also providing numerous ecosystem services, such as water supply and flood control. This approach integrates stakeholder engagement with translational science and decision support tools to inform multipurpose water management operations. Communication between various groups (e.g., stakeholders, managers, etc.) is critical to the EBM process and, in the Everglades, is based on a framework comprised of formal and informal requirements. Throughout this chapter, the various components and linkages that make up the EBM approach in the Everglades are presented as a case study to illustrate the practical application of EBM across a landscape.

2 Everglades Ecosystem

2.1 Overview of an Ecosystem in Trouble (Through 2000)

The historical Everglades were 9307 km^2 (2.3 million acres) of a vast wetland that began at a chain of lakes in the Kissimmee basin that flowed into Lake Okeechobee and stretched to the end of the Florida peninsula (Douglas 1947). Lake Okeechobee was historically 1891 km^2 (730 mi^2) in size and water depths fluctuated between 3–4 m (10–20 ft.) deep (McVoy et al. 2011). The lake acted as a natural reservoir storing water from the Kissimmee Chain of Lakes and during periods of high water from tropical cyclones. When full, the water spilled over into pond apple forests causing a wide swath of slow moving water in the form of sheetflow (Fig. 1) (McVoy et al. 2011). Water depths south of the lake in the sawgrass dominated Everglades ranged from 5–100 cm (2–40 inches) deep. The slow moving water with occasional periods of high pulse flows created a ridge and slough landscape that included tree islands as crucial habitat for wildlife and plant species. (Frederick and Ogden 2001). The slope of the land was so gradual at 5 cm per 1.5 km (2 inches per mile), that water only moved southward at 30 m (100 ft) per day (National Research Council 2010). This water eventually discharged into the mangroves of Florida Bay and the Ten Thousand Islands in southwestern Florida.

2.2 Drying of the Marsh for Agriculture by Compartmentalization

In the late 1800s through the 1930s, settlers sought to dry out the swamp south of Lake Okeechobee in order to use the rich muck and peat to grow crops. In 1947, major floods occurred over South Florida with over 2.5 m (100 in) of rain causing the United States Congress to authorize the Central and South Florida (C&SF) Project in 1948, which was intended to provide drainage and flood control for the croplands and the outlying communities (Fig. 2) (Light and Dineen 1994). In spite of the public outcry against compartmentalization, 1600 km (100 mi) of levees, 1160 km (720 mi) of canals, and 200 water control structures (Light and Dineen 1994) were designed.

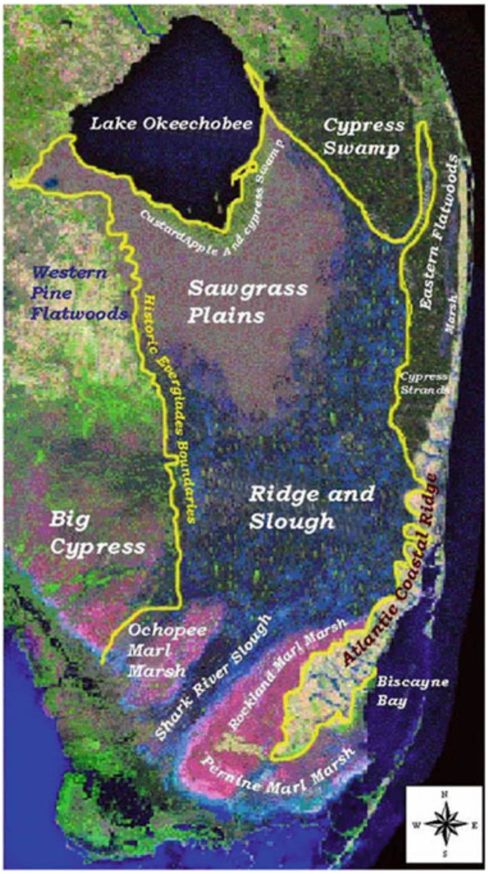

Fig. 1 The historical and present Everglades. (Reproduced from the National Academies of Sciences, Engineering, and Medicine 2018)

Although too late to stop damage to the Everglades, Marjory Stoneman Douglas published "*The Everglades: River of Grass*" in 1947 arguing for saving the Everglades.

The C&SF project, built during the 1950s and 1960s, completely disconnected the historical Everglades. The historical sheetflow lost its natural headwaters and became a complicated water management system designed to provide flood control and continual drainage of the system for agriculture (National Research Council 2010). The drained area south of Lake Okeechobee became known as the Everglades Agricultural Area (EAA) and is approximately 2850 km^2 (1100 mi^2) (Snyder and Davidson 1994). Multiple Water Conservation Areas (WCAs) covering approximately 3500 km^2 (1350 mi^2) were developed for managing water in the open areas of the Everglades.

The C&SF Project had multiple direct, but adverse, hydrological impacts on the Everglades (US Department of the Interior 1994):

Fig. 2 C&SF project compartmentalization of the Everglades into the Everglades Agricultural Area, Water Conservation Areas, preserves, and parks. (Reproduced from Musser 2010)

- The Everglades were reduced in size by more than fifty percent (50%) to approximately 4000 km^2 (one million acres).
- Loss of an annual average of 2.7 billion m^3 (2.2 million acre feet) of fresh water that flowed into the coastal estuaries along Florida's east and west coast due to lack of conveyance and storage to send the water south into the Everglades. The lack of sheetflow and the resulting altered hydroperiod in the remaining Everglades changed native vegetation and habitat.
- Lack of variability in water delivery drastically altered the seasonal patterns of high and low flows to the remaining Everglades.
- The Everglades continued losing its unique ridge and slough landscape, along with tree islands, causing a change in wildlife population abundances and distribution.

In the 1970s, the larger U.S. environmental movement brought more attention to the Everglades. By the late 1980s, state and Federal agencies, citizen groups, and the Tribes began focusing on restoring the Everglades ecosystem and protecting its species and habitats. Currently, there are 68 threatened and endangered species in the Everglades (USFWS 1999). The most noted species change has been in the population, distribution, and habits of wading birds native to the Everglades (Ogden 1994). Even the most basic component of the Everglades—water—is at risk. Water depths, duration, and overall distribution across the landscape changed with drainage, creating a suite of ecosystem changes (Kushlan 1987; Ogden 2005). Water quality has been consistently deteriorating due to agricultural chemicals, urban runoff, and animal waste from ranchlands and dairies upstream in the EAA, Kissimmee Chain of Lakes, and urban areas such as Orlando. Successful restoration of the Everglades ecosystem requires the appropriate interaction between the quantity, quality, timing, and distribution of water (ENP 2015).

2.3 An Ecosystem Managed for Multiple Purposes (2000–Current)

At present, the remaining greater Everglades ecosystem is a human-shaped environment that is managed for multiple purposes, including: flood protection, water supply, water flows for the environment, and habitat supporting a variety of flora and fauna. Water flow is a foundational element of most Everglades ecosystem management, with two primary water management agencies, the U.S. Army Corps of Engineers (USACE) and the South Florida Water Management District (SFWMD), making and implementing operational water management decisions. Smaller government entities and utilities, such as the Lake Worth Drainage District and the Everglades Drainage District, influence hydrology at smaller scales. Additionally, a mosaic of federal, State and Tribal lands exist throughout the greater Everglades Ecosystem Management decisions in these land units are made by the U.S. National Park System, U.S. Fish & Wildlife Service, Florida Fish and Wildlife

Commission, the Miccosukee Tribe of Indians of Florida, or the Seminole Tribe of Florida. Overall, the greater Everglades landscape is a complex, socio-ecological system involving a range of governance models (Ankersen and Hamann 1996). Each governing body is operated under a different suite of legal and policy mandates, with different levels and types of stakeholder engagement. Juxtaposed against this complex range of governance models is a large body of scientific research underscoring the interconnectedness of the landscapes and ecosystems. From this research, an increased understanding of the system has led to the development of a diverse array of tools to evaluate, assess, and predict system responses to management operations and restoration projects.

3 Current Water Management

The USACE monitors and manages the multi-purpose operations of spillways, locks, pump stations, culverts, canals, reservoirs, and water conservation areas (USACE 2019) and is considered a federal partner to the State of Florida's water management districts. Among other activities, such as water quality monitoring and scientific research, the SFWMD manages water supply, flood control, and is the State partner in Everglades Restoration. The SFWMD operates approximately 2100 miles of canals and 2000 miles of levees/berms, 77 pump stations and more than 600 water control structures and 620 project culverts across central and southern Florida (SFWMD 2019). This extensive network of infrastructure (Fig. 3) encompasses three water conservation areas (WCAs), large wetland areas that are compartmentalized by berms and levees and receive and discharge water through water control structures. The WCAs make up the Greater Everglades and are located upstream of Everglades National Park (ENP). Historically, the Everglades were compartmentalized into WCAs to prevent catastrophic flooding as witnessed before the C&SF project was authorized in 1948. The WCAs store rainfall, Lake Okeechobee flood releases, and excess water runoff from the EAA, as well as recharge aquifers, reduce seepage into urban areas, and protect against salt water intrusion from rising sea levels. The WCAs also provide flow-through capacity of water cleaned (primarily by removal of phosphorus) by stormwater treatment areas (STAs). STAs are constructed wetlands that remove and store nutrients through plant growth and the accumulation of dead plant material. STAs are comprised of parcels of land utilizing different types of emergent (e.g., cattails, pickerel weed and bulrush) and submerged (e.g., hydrilla, southern naiad and chara) plants that take phosphorus directly from the water in STAs (SFWMD 2019). STAs are critical for providing environmental benefits to the Everglades landscape, species, and their habitats. (USGS 2013).

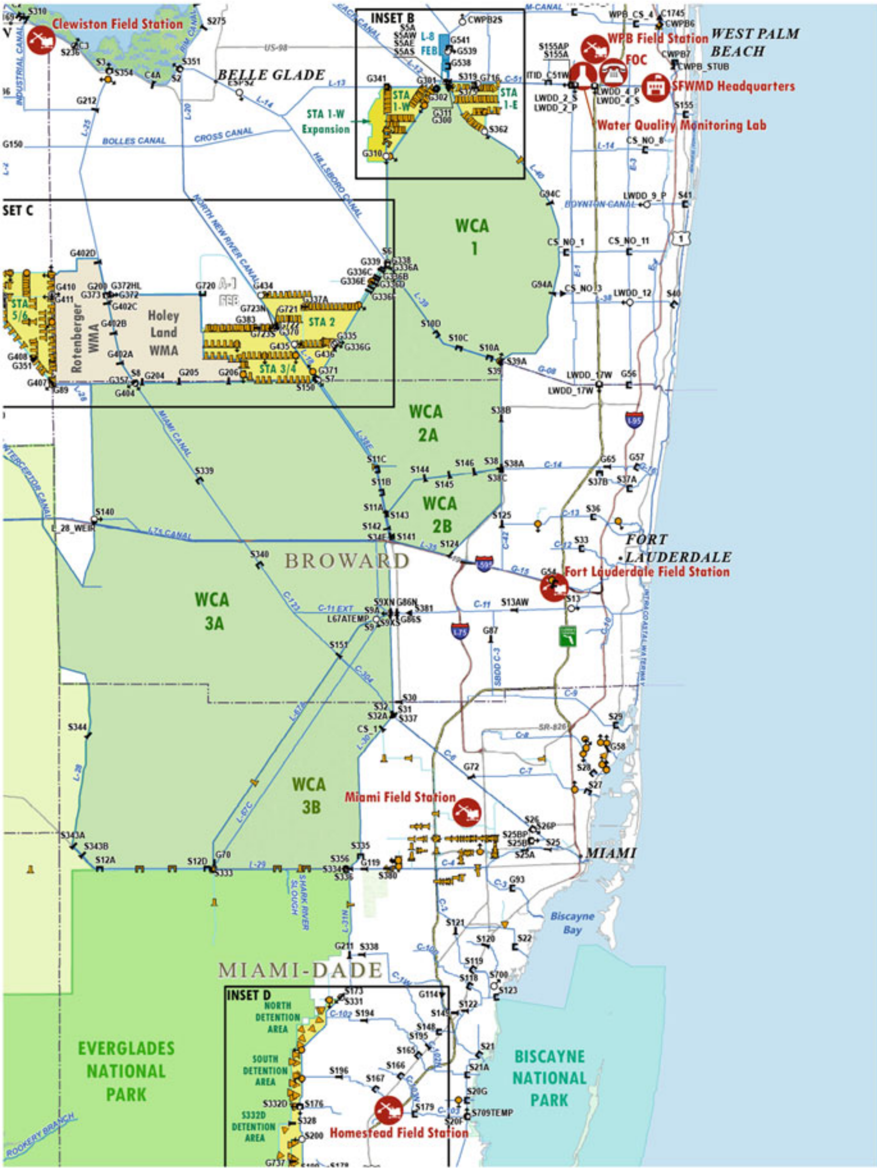

Fig. 3 Facility and infrastructure location index map indicating canals, pumps, weirs, spillways, and stormwater treatment areas in yellow. (Reproduced from SFWMD 2016)

3.1 Rainfall, Regulation Schedules, and an Altered Ecosystem

Rainfall drives the hydrology of the Everglades. However, water management actions also influence hydrologic conditions throughout the ecosystem. Lake Okeechobee and the WCAs are managed by Water Regulation Schedules, which are a set of rules based on antecedent conditions, rainfall formulas, monthly and seasonal rainfall based water management plans, and regulatory requirements for flows into ENP. Regulation schedules (e.g., Fig. 4) provide recommended operational guidelines for maintaining target water level ranges in each WCA, which are monitored through a complex network of gauges. They also provide recommended water levels for the beginning of the dry season (November 1) and for the beginning of the wet season (June 1). Water managers can implement operational changes that deviate from a regulation schedule during and after an extreme rainfall event. This "deviation" can exceed normally recommended water discharges to get water levels lowered more quickly to acceptable levels.

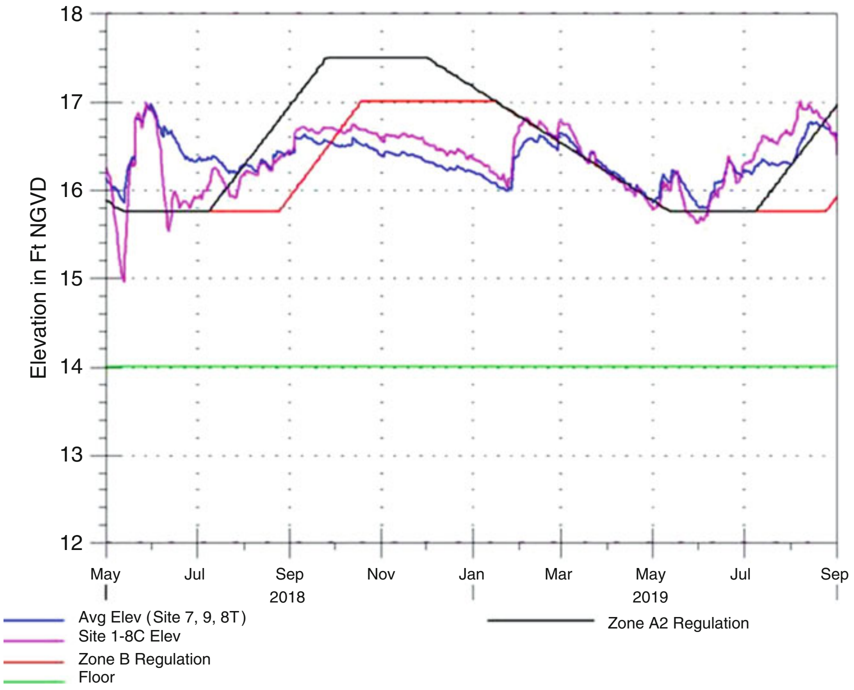

Fig. 4 Regulation schedule and water levels in 2018 for Water Conservation Area 1 within the Greater Everglades. Red and black lines indicate surface water elevation targets throughout the year. Purple and blue lines indicate actual surface water elevations resulting from water management operations and rainfall in Water Conservation Area 1 (USACE 2019)

At times, water management rules can be in conflict with the environmental needs of species and their habitats, although the intent of Everglades restoration projects is to mimic the natural hydrology of the Everglades. Hydrologic and ecological studies have shown that some rules could be altered to better mimic the Everglades' natural hydrology. This has been particularly evident with some of the State and federally listed threatened and endangered species (e.g., Snail Kites (*Rostrhamus sociabilis plumbeus*), Wood Storks (*Mycteria americana*), wading birds, and Cape Sable Seaside Sparrows (*Ammodramus maritimus mirabilis*) (USFWS 2016). Some of the habitat these species depend upon has become altered due to unnatural water depths, shorter- or longer-than usual hydroperiods, and altered rates and patterns of flow. The succession of short-hydroperiod marl prairie grasses to long-hydroperiod sawgrass in naturally occurring high ground in ENP is a prime example of areas becoming unnaturally wet due to water management operations. Management operations of water levels in the Kissimmee Chain of Lakes, Lake Okeechobee, and the Kissimmee River have also led to the deterioration of Snail Kite habitat in lake littoral zones and river floodplains (Cattau et al. 2008). Snail Kites breed and nest in the littoral zones and floodplains, which are also home to their primary prey, the apple snail. It has been documented that water management operations have at times reduced water levels too rapidly, causing damage to the habitat and forcing the apple snail to move into deeper water and leaving apple snail egg clusters without necessary water levels to survive (Bennetts and Kitchens 1997). For Wood Storks and wading bird colonies, 41 cm (16 in) of water depth is required to support their fish prey. Water management operations can, and have, either flooded or quickly dried out, core foraging areas for wading birds (USFWS 2014).

Because of the potential negative effects to listed threatened and endangered species, water management in central and southern Florida has occasionally been changed and amended by federal regulatory documents, such as a USFWS Biological Opinion (BO). A recent BO mandated that water managers coordinate with ecologists and biologists from state and federal government agencies, non-governmental organizations, and other interested parties prior to changing particular structural operations due to potential upstream or downstream ecological effects. These BOs are also responsible for changing the rate and timing of structural flows to better mimic natural rainfall and hydrology to protect certain habitats like the marl prairie in ENP (USFWS 2016). However, it was recognized in the 1999, and all subsequent BOs, that there would be times when unseasonal rainfall could and will overwhelm the water management system. During those times, it is critical to "share adversity" among stakeholders and thus, closely coordinate among agencies to provide the best operations for human health and safety, the Everglades landscape, and the biology and ecology of the system (USFWS 2016).

3.2 Monitoring and Current Status of the Ecosystem

Changes in hydrology (hydromorphological alterations), water quality, and water management are the principal stressors that affect the ecosystem (e.g., Walker 1999). Monitoring hydrology includes maintenance of hydrologic monitoring gauges, collection of data, and processing and dissemination of hydrologic data. The monitoring station network (Fig. 5) within the ecosystem is comprised of almost 300 gauges (USGS 2009) that measure water stages and water quality and are operated by the Big Cypress National Preserve (BCNP), ENP, SFWMD and United States Geological Survey (USGS) (2009).

These gauges are used to model surface water elevations and depths across the landscape, often as related to wildlife, (e.g., Fig. 6), and can be used for planning purposes. New gauges may occasionally be needed in strategic areas due to topographic variability. However, the need for new gauges is weighed against habitat impacts resulting from installation (USFWS 2016).

3.3 Planning for Future Water Management Operations

As a federal partner, USACE water managers monitor gauges, water depths, and regulation schedules of water conservation areas. These activities include analyzing past and predicted rainfall events and considering the requirements of a multitude of stakeholders, including, but not limited to, navigation, ecological needs, agriculture, and recreation. The SFWMD uses risk analysis to evaluate the "present condition" of the system, on which to base its water operations recommendations. The purpose of the risk assessment is to evaluate water resources and the risks associated with operational decisions (Hirsch 1978). This evaluation is accomplished by estimating the probability distribution function of select variables, conditional on the current, or otherwise specified, state of the system (SFWMD 2019). The SFWMD also provides historical rainfall on a monthly and seasonal basis. Other agencies provide products and input to operational planning including the Climate Prediction Center (CPC) with rainfall predictions out to a year, the National Weather Service (NWS) with quantitative precipitation forecasts out to seven days, USGS and ENP with EDEN (Everglades Depth Estimation Network) ecosystem-wide water depths, and USFWS with the Species Climate Outlook that focuses on the expected climate out to 12 months, but includes general weather expectations for species across the State, projected out to 2100. Each source of data, including model projections, provides value added information to the larger discussion that was not systematically incorporated into the historical discussion on how to best manage water operations.

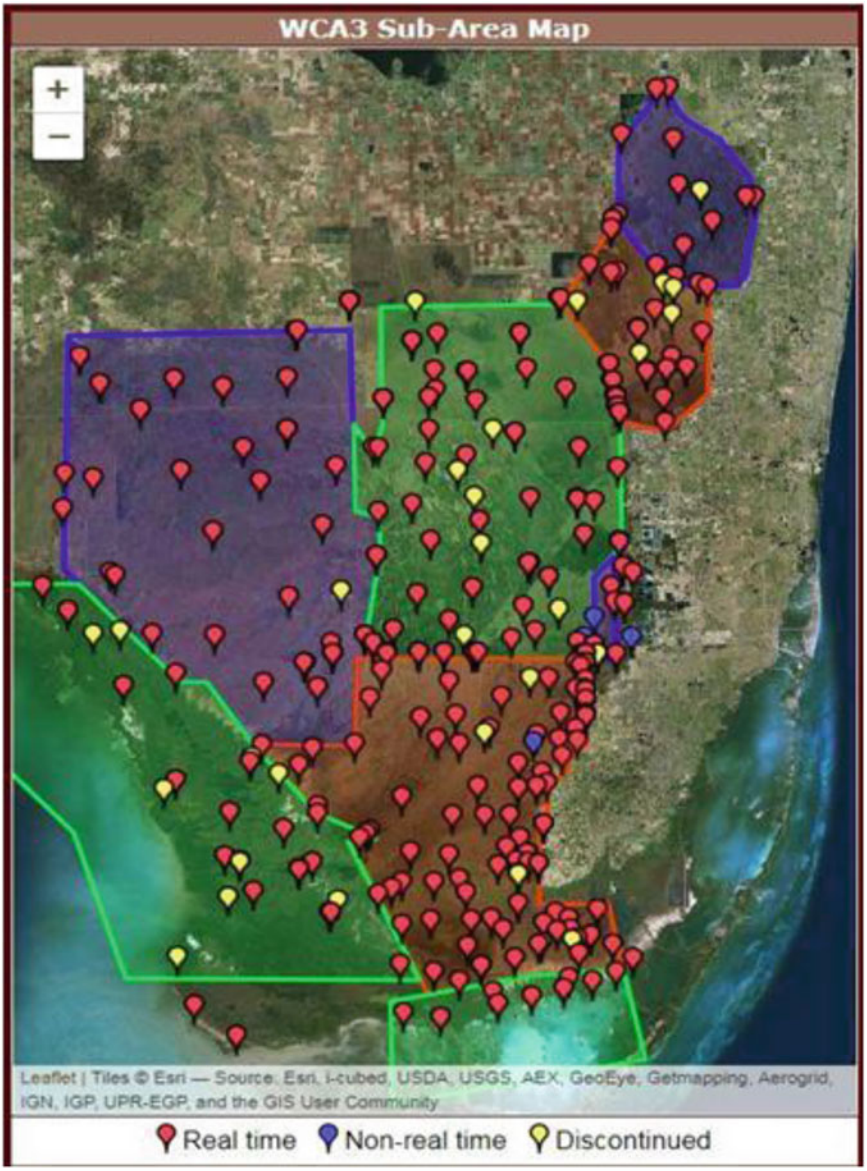

Fig. 5 Gauge locations for monitoring water levels and water quality in the Everglades. (Reproduced from USGS 2009)

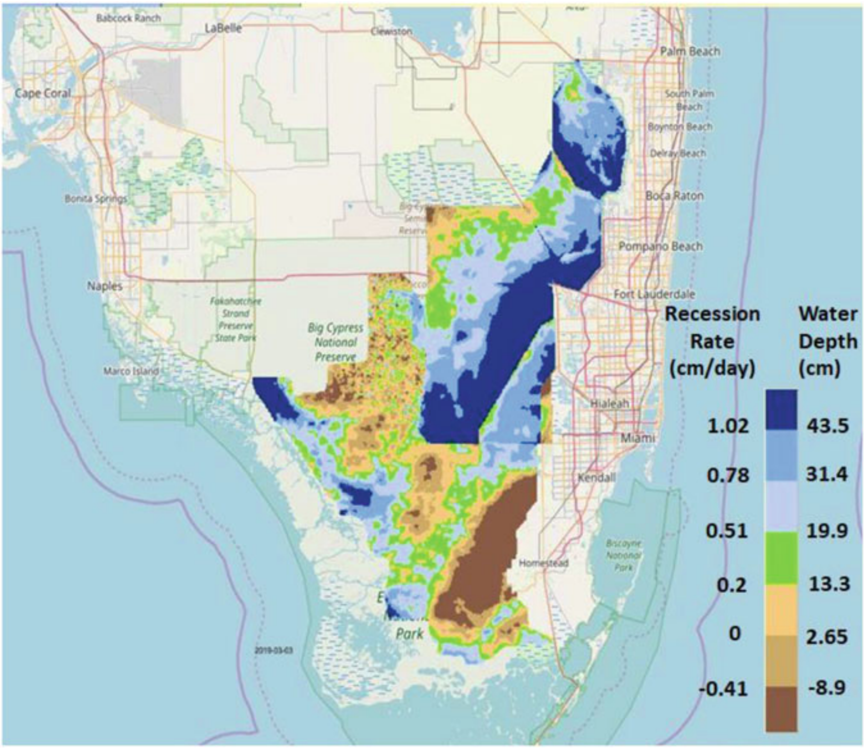

Fig. 6 Example of a water depth map, taken from the Wading Bird Depth Viewer. (Reproduced from USGS 2009; https://sofia.usgs.gov/eden/wadem/)

4 Translational Science

Translational science focuses on the importance of communicating scientific information to "connect end-users of environmental science to the field research carried out by scientists" (Schlesinger 2010). The use of a strategic communication approach (Harwell et al. 2020) thus can be useful in efforts to achieve effective EBM based on the EBM principle that, "decisions reflect societal choice" (Long et al. 2015). While "strategic communication" approaches were not explicitly identified in many EBM examples, a large EBM case study analysis by Mattheiß et al. (2018) concluded that "the better the communication strategy the likelier the demand for scientific knowledge from the social system." The translation of science within the context of ecosystem management is key to a broad understanding of both social and ecological systems and their interlinkages, which promote the development of innovative tools and management approaches to sustain biodiversity and the long-term delivery of ecosystem services (Piet et al. 2017).

As a complex socio-ecological system, the Greater Everglades involved numerous stakeholders from backgrounds including government agencies, universities,

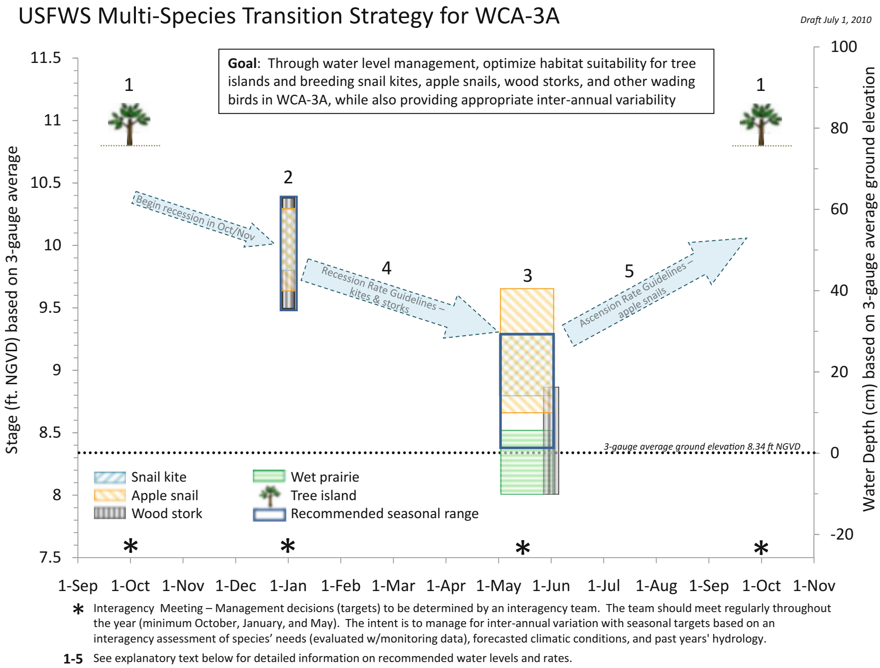

Fig. 7 USFWS Multi-Species Transition Strategy for Water Conservation Area 3A. Strategy includes recommending ranges and targets for species and habitats likely to be impacted by Everglades Restoration projects. *Denotes timing of intended interagency coordination meeting. (Reproduced from USFWS 2010)

non-profit conservation organizations, and tribes that rely on receiving ecosystem services. A variety of managing local, state, and federal agencies provide those services through ecosystem management of the adjoining Everglades. These same agencies, plus universities and other organizations, conduct research, collect monitoring data, and fill key roles in water management decisions and operations. Due to this complexity, communication among stakeholders that can be impacted by water management decisions, managers that make operational decisions and scientists collecting research and monitoring data is central to effectively applying EBM to an ecological system, particularly at the landscape scale. For the purposes of EBM of the Everglades, the need to strategically communicate the elements of translational science (e.g., Harwell et al., 2020) have been acknowledged for decades for both ecosystem management (Kushlan 1979) and ecosystem restoration purposes (Harwell 1997). Here, a translational science framework for EBM in the Everglades (Fig. 7), involves the communication of information among scientists, water managers, and stakeholders. Much of this EBM framework is foundationally defined in regulatory and/or planning documents related to Everglades Restoration (e.g., Biological Assessments under the Endangered Species Act (ESA), Environmental Impact Statements under the National Environmental Protection Act) and implemented at a high level through related programs such as the Restoration

Coordination and Verification (RECOVER) and the Congressionally mandated South Florida Ecosystem Restoration Task Force, as well as through field-level coordination between scientists and land managers. As such, these stakeholders helped define and develop the operational EBM framework presented here. We do recognize that other EBM frameworks have been developed for larger spatial and/or governance scales, such as the AQUACROSS framework (Piet et al. 2017).

At a field level, science generated by agency scientists (both associated with the operational management of the system and other agencies' supporting science) is translated into status and condition information for both ecosystem components (e.g., wading birds, tree islands) and the underlying hydrology and environmental conditions. This information is fed to stakeholders, including water managers and operational decision makers through a suite of communication forums (e.g., weekly and quarterly coordination meetings) and media (e.g., model output, informational graphics, narrative and numerical assessment statements). Water managers and operational decision makers translate this information into the context of agency mandates, goals, and operational constraints to make decisions that change the water management of the system. The resulting ecological outcomes, part of the monitoring effort for determining success, is folded back into the status and condition information as part of a larger adaptive management cycle. While water management decisions are still made by sector-focused operational managers, the Everglades EMB framework creates both the mechanism and the opportunity for other Everglades socio-ecological system goals and information to be served up for consideration.

This framework allows for real-time integration of operations data from discrete structures, knowledge of hydrodynamics throughout the system, modeled surface water elevations, and information on ecological envelopes (e.g., boundary conditions) for various species to develop water management recommendations for the best ecological outcome on multiple temporal and spatial scales. Recommendations are made with other uses, constraints, and regulations in mind but focus on ecological outcomes by aiming to identify where the system could use more/less water or; faster/slower water level ascension/recession rates, as well as highlight ecologically sensitive areas and/or species and habitats for a given operational decision.

5 Managing Eco-hydrology in the Everglades

Species typically used as indicators of Everglades ecosystem health and restoration success represent a range of habitats, behavioral characteristics, niches, and conservation status (Doren et al. 2009). Different species are used as indicators at different scales depending upon their response time to changes in environmental conditions, data availability, and the utility of relevant tools (Doren et al. 2009). Common indicator species include alligators, wading birds, Snail Kites, and apple snails. Typical habitats, such as tree islands, sawgrass ridges, and sloughs provide the basic structure of the Everglades and are sensitive to changes in water management.

All wildlife and habitats in the Everglades are adapted to annual patterns of rainfall and regional flooding. Typical Everglades habitats are characterized by a range of conditions largely driven by water levels and hydroperiods, which vary widely throughout the system because of a north-south elevation gradient, variations in landscape micro-topography that drive water levels and hydroperiods, and seasonal rainfall patterns that result in a typical dry season (November–April) and wet season (May–October).

Species that have evolved with the Everglades are able to survive in the highly dynamic system characterized by low nutrients and extremely variable intra- and inter-annual water levels using a variety of strategies, such as synchronization of breeding seasons with periods of suitable water levels and prey availability. In general, wading birds are colonial nesters and under typical conditions largely use the same flooded areas for nesting from year to year. Some species, such as Wood Storks, travel long distances to forage if conditions in their typical nesting spots are not ideal. However, other species, such as the non-wading Snail Kite, are more nomadic and select annual nesting sites based upon where conditions in the system are most conducive to successful nesting (i.e., appropriate nesting materials available and water levels that are 20–80 cm (50–200 in.) deep (Bennetts et al. 1988). These water depths protect nests from land-based predators and provide suitable habitat for their primary prey, apple snails, which prefer depths less than 50 cm (125 in.) (Darby et al. 2002). Species such as the endangered Cape Sable Seaside Sparrow nest near the ground yet require dry habitat to breed, and are therefore dependent upon areas of higher elevation with shorter hydroperiods. This diversity in habitat requirements are largely provided throughout the Everglades ecosystem by changes in ground surface elevation and extensive micro-topography, which results in a mosaic of habitats encompassing a range of water depths and hydroperiods across the landscape.

The Everglades ecosystem is heavily impacted by socio-ecological pressures, such as efforts to provide flood control and water supply to surrounding urban and agricultural development. Water managers strive to operate existing water control structures across the landscape in a coordinated fashion with the overall goal of providing conditions similar to those historically driven by rainfall and natural sheetflow, while providing crucial ecological services such as flood control and urban/agricultural water supply. Creating natural conditions in this highly impacted and managed system requires the complex integration of operational constraints, regulations, and policies, with the varied habitat requirements and population status of the suite of indicator species. For example, tree islands and ridge and slough habitat types, as well as the underlying peat substrate, are sensitive to hydrologic patterns such as water depth and hydroperiod. Science-based thresholds for water depths and hydroperiods are used to inform recommendations so that management is protective of these habitat features.

Because of the inextricable link between wildlife and water in the Everglades, consideration of past, current and future water conditions, habitat conditions, and wildlife population status at short, mid- and long-temporal scales are crucial for making water management recommendations that are protective of sensitive wildlife and habitats. The earliest formalized effort to summarize and integrate suitable

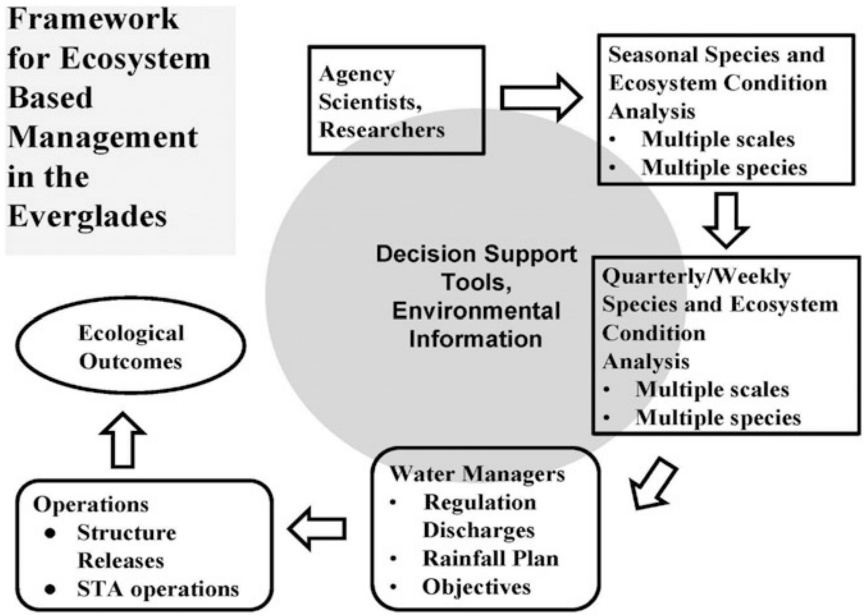

Fig. 8 Framework for Ecosystem-Based Management in the Greater Everglades ecosystem. Decision support tools and environmental information (*shaded circle* in background) provide an overarching anchor for EBM

conditions for a range of species was the Multi-Species Transition Strategy (MSTS) for WCA-3A in the Everglades Restoration Transition Plan (ERTP) developed by the Ecological Services branch of the USFWS as part of ESA Section 7 consultation (Fig. 8) (USFWS 2010).

The MSTS plan includes descriptions of the typical hydrologic ranges and timing of suitable conditions for multiple species considered indicators of Everglades Restoration. The plan, available to inform operations, management, and restoration decisions, directly compares and illustrates tolerance ranges for different sensitive habitat types and species, allowing identification of overlapping conditions and potential conflicts for management of these species. The information in this plan is combined with current, past, and projected future habitat conditions to inform daily/monthly/seasonal ecological recommendations for water management, with a particular focus on areas impacted by ongoing restoration construction projects by the USACE.

The ability to develop effective recommendations is dependent upon accurate, real-time data regarding habitat conditions and the ability to predict the likely response of wildlife populations to management operations. Much of the available research and/or monitoring efforts are driven by a given species' conservation status, population status, and/or inclusion in USFWS' species recovery plans, the Monitoring and Assessment Plan (RECOVER 2009), and indicator species identified by the

Department of Interior's South Florida Ecosystem Restoration Task Force, which is part of the Comprehensive Everglades Restoration Plan (CERP) (USACE and SFWMD 1999). Species designated with official conservation status, such as species protected by the ESA, or designated as a Florida Species of Special Concern, may have additional regulatory or recommended guidelines available to incorporate into a decision-making framework.

There are a variety of available tools and reports for determining and assessing antecedent, current, and potential future conditions such as USGS' EDEN, a landscape-scale surface water depth model, the National Oceanographic and Atmospheric Administration (NOAA) long- and short-term climate outlook predictions (Quantitative Precipitation Forecasts; QPFs), past rainfall trends, and the USFWS' Species Climate Outlook report, which characterizes forecasted conditions based on the requirements and tolerances of select species. A suite of scientists from universities, local, State, and federal agencies, Tribes, private consultants, and non-profit organizations provide wildlife population status and habitat requirement updates from ongoing research and monitoring efforts.

In addition to driving research and enabling partnerships, the CERP provides a critical framework for incorporating science, including new research and monitoring data, into management decisions using an adaptive management strategy (Loschiavo et al. 2013) creating opportunity for applying Ecosystem-Based Management (EBM) approaches to achieve restoration goals. A significant amount of resources have been invested in using results from scientific investigations to develop additional decision support tools to inform water operations and habitat management across the entire Everglades landscape (Table 1). These tools combine routinely collected environmental data with wildlife data and habitat condition information that can be used to assess past, current, and likely future conditions across the landscape to predict and evaluate potential impacts/benefits of water management and operations to wildlife and habitats.

Government and university partners developed the most easily accessible and frequently updated spatial modeling tools. These tools are used for making daily, weekly, and seasonal ecological recommendations. Many tools include maps that integrate known habitat preferences of a suite of wading birds, including the federally threatened Wood Stork (e.g., preferred water depths, rate and direction of change in water levels) with current and/or future conditions to indicate different levels of habitat suitability across the Everglades landscape (Wading Bird Depth Viewer, WADEM; Table 1). A similar tool based on recommended hydroperiod and water depths is available for the federally endangered Cape Sable Seaside Sparrow (Sparrow Viewer; Table 1), as well as other species such as Snail Kites and apple snails. While these tools and species updates are available to be individually considered by water managers when making operational decisions, most available tools do not provide a high-level integration of information or provide specific ecological recommendations.

Table 1 Names, descriptions, and sources of some of the spatial modeling tools used to develop multispecies water management recommendations in the Everglades

Species	Status	Time-scale	Tool description	Translational science information and source
Snail Kites[a]	Federally Endangered	Real-time; Long-term	EVERKite	Generates spatial maps of conditions for Snail Kites, either current or simulated under different hydrologic scenarios; specifically refers to targets defined in USFWS Biological Opinion. https://www.jem.gov/Modeling/EverKite
Apple Snails[a]	Least Concern; Primary prey of endangered Snail Kite	Long-term	EVERSnail	Generates estimated population size under different hydrologic scenarios. https://www.jem.gov/Modeling/AppleSnail
Wood Storks[a]	Federally Threatened	Real-time; Long-term	WADEM	Generates spatial maps of conditions for wading birds, including Wood Storks, either current or simulated under different hydrologic scenarios; specifically refers to targets defined in USFWS Biological Opinion. https://www.jem.gov/Modeling/WADEM
Wading Birds[b]	Varied	Real-time; Long-term	WADEM	Generates spatial maps of conditions for wading birds, including Wood Storks, either current or simulated under different hydrologic scenarios; specifically refers to targets defined in USFWS Biological Opinion. https://www.jem.gov/Modeling/WADEM
Alligators[b]	Threatened due to similarity of appearance (USFWS)	Long-term	Alligator Model	Generates spatial distribution maps of environmental condition related to Habitat, Breeding, Courtship & Mating, Nest Building, Nest Flooding, Overall Suitability, either current or simulated under different hydrologic scenarios. https://www.jem.gov/Modeling/Alligator
Cape Sable Seaside Sparrow[a]	Federally endangered	Real-time; Long-term	Sparrow Viewer	Generates maps of current or simulated conditions as related to Cape Sable Seaside Sparrow; either current or simulated under different hydrologic scenarios; specifically refers to targets defined in USFWS Biological Opinion. https://sofia.usgs.gov/eden/csss/index.php

For more on translational science aspects, the reader is directed to Sect. 4
[a]USFWS Multi-Species Recovery Plan (USFWS 1999)
[b]2009 Revised CERP Monitoring and Assessment Plan (RECOVER 2009)

6 Integrating Information Into Recommendations

Integrating relevant information into ecological recommendations at multiple temporal and spatial scales occurs through a series of seasonal, weekly, and/or daily meetings that occur at key times throughout the year. The scope of discussions and recommendations become more narrowly focused with increased meeting frequency. The utilization of available tools and monitoring information allow the characterization and assessment of species-specific habitat conditions as well as the potential impact of operational decisions on indicator species. Recommendations focus on addressing ecological needs of the Everglades more than specific operational decisions, although incorporating information about the feasibility of operations and regulatory guidelines strengthen recommendations. Some specific operational recommendations can be made, such as recommending preferred volumes and rates of inflows/outflows at specific structures, but only in areas that fall within existing regulatory frameworks and guidelines that can be feasibly considered (i.e., without formally updating regulatory guidelines).

Seasonal meetings generally occur at the beginning of the dry and wet seasons, as well as one meeting during the transition between wet and dry seasons (October, January, and May). These meetings focus on assessing conditions, characterizing desired ecological outcomes for the upcoming season, and defining water management guidelines for achieving those desired outcomes. Meetings are typically full-day, in-person workshops that include species updates (monitoring and research), overviews of current, past, and expected future climate conditions, a summary of ongoing and planned operations, and short- (7–10 days) and mid/long-term climate outlook discussions to develop recommendations for achieving desired ecologic (30–90 days) al outcomes. Meeting participants include key stakeholders such as agency, university, and non-governmental organization scientists. Factors such as short- and long-term stakeholder/managing agency goals and inter-annual variability of indicator species population dynamics are also considered when making recommendations. Seasonal recommendations are based on the overall assessment of conditions (past, present, and future) and typically include the identification of priority species and areas within the system, the characterization of desired ecological outcomes for the upcoming season, and the development of seasonal targets for rates of change in water depth and hydroperiod based on current conditions as indicated by monitoring data and modeled species habitat suitability. Recommendations can also include generalized recommendations such as to retain water where storage is available (per regulatory guidelines), to avoid particular operations that can have deleterious habitat impacts (e.g., degraded water quality), and/or suggestions that foster more natural (e.g., applying a ramping approach when adjusting structure inflows/outflows). Ecological thresholds and/or targets for other species and habitats, such as alligators and tree islands, also inform recommendations to benefit the greatest number, and/or highest priority (most sensitive or imperiled) of species across the landscape. Seasonal meetings tend to have the highest number of participants because a broader range of stakeholders, (many of which are identified

26.1.3), including non-governmental organizations, local municipalities, natural resource management agencies, and researchers from a range of organizations, are interested in providing species and habitat updates and input towards longer-term recommendations. Accounting for variability throughout the system, as well as agency-specific missions, directives, and/or priorities, recommendations are provided at multiple spatial and temporal scales as appropriate.

Once seasonal recommendations are developed, a smaller, core group of species experts and agency biologists evaluate, update, and communicate recommendations in real-time. Weekly meetings of a core group of primarily managing agency scientists develop habitat condition updates as well as any necessary updates to seasonal ecological recommendations. Weekly ecological recommendations typically include assessment of current and expected (short- and mid-term) conditions of habitat and wildlife populations, recent operations, and projected rainfall trends. Ultimately, the group identifies areas with sensitive habitats/wildlife populations (e.g., nesting wading bird colonies), and/or localized areas that could ecologically benefit from more/less water.

Weekly and/or daily coordination teleconferences typically take place during the dry season and coincide with wading bird, Snail Kite, and Cape Sable Seaside Sparrow nesting seasons. These coordination meetings partially focus on these indicator species to assess current conditions and short-term climate predictions to make targeted, short-term recommendations, often resulting in updating and refining seasonal recommendations to promote desired outcomes. The scope and focus of weekly and/or daily meetings tends to be narrower than in seasonal meetings. These meetings are often followed up with daily or semi-daily coordination meetings with agency scientists and managers during the occurrence of major events such as significant rainfall or changes in operations.

Group recommendations and updates are communicated to operations managers at the SFWMD and USACE through written reports developed by the group and presented during weekly operations meetings by agency scientists that participate in both the development of ecological recommendations and agency water management operations meetings. Routinely providing recommendations in other forums, such as regulatory Periodic Scientist Calls, which are monthly public coordination calls mandated by ESA consultation and hosted by USACE, as well as intra-agency management communication chains support coordinated recommendations across management agencies.

Available tools are currently integrated within the recommendation-making process to various degrees. Further integration of real-time population distribution and conditions would enhance the ability to understand and communicate short-term ecological needs. As additional decision support tools are developed and implemented, a combination of current conditions, past conditions, and past climate conditions over a long period can be used to determine the most likely wildlife response to water trends across the Everglades system for the upcoming season. These likely scenarios can be evaluated, and recommendations made, for individual species or a select group of species (e.g., wading birds). One highly anticipated tool currently under development is a species-forecasting tool (USGS) that considers habitat requirements and potential habitat suitability in the upcoming season for an entire collection of indicator species based on expected water level trends. Water trend scenarios can then be evaluated for their benefits to the greatest number or highest priority indicator species, which can help define priorities and potential outcomes. An early version of this tool is currently being tested and studied to determine the best way to integrate it into decision-making processes.

An infographic depicting information about current ecological conditions and species distributions, specifically designed for operations managers, is currently in development. This graphic is intended to provide a visual summary of recommendations and the ecological conditions that influenced them in order to more effectively communicate ecological needs and document the decision process to operations managers. An additional in-depth annual review of how recommendations influenced water operations and promoted desired ecological outcomes would allow further understanding of how ecological components are incorporated into the decision-making process and support refinement of how recommendations are made to maximize EBM effectiveness. Finally, additional benefits could be gained from the available tools and communication strategies if they were incorporated into existing and future regulatory guidance.

7 Conclusion

A suite of EBM activities support multi-purpose management of the Everglades ecosystem to provide ecosystem services as well as support plant and wildlife communities. With a focus on stakeholder engagement, communication, and the development/use of tools to integrate a wide range of conditions, operations, climate, and wildlife population data, the EBM framework presented here promotes consistent and effective management and restoration of the Everglades to meet a wide range of complex goals, needs, and ecological targets.

Stakeholder participation and communication is key to the effectiveness of this multiple element, decision-making EBM process in a complex socio-ecological system. Engagement by interested parties, researchers, and managing agencies enable integration of local and regional priorities to support healthy wildlife and habitats, as well as provide critical ecological services (e.g., flood control, water supply). Documenting the process and results for developing recommendations promotes communication with stakeholders and provides a record that can be used for adaptive learning.

This multi-agency/stakeholder approach to using integrative tools and real-time monitoring data for coordinating and developing comprehensive and effective water management recommendations is superior to previous approaches because this method is inclusive, transparent, comprehensive, and provides a landscape context to recommendations for individual management areas. The coordination of ecological recommendations among scientists from the various land management agencies across the landscape integrates research, monitoring, and stakeholder interests, and provides water managers with a more holistic and cohesive set of recommendations for supporting wildlife and habitats, even when there is not full consensus regarding ecological recommendations among the agencies due to area/agency-specific goals and objectives. A dedicated focus on stakeholder engagement facilitates the inclusion of local expertise (representing those stakeholders identified in Sect. 2.3) provided by agency biologists, who manage individual areas, with information provided by university and agency scientists that monitor regional conditions and wildlife. Stakeholder engagement allows for the incorporation of valuable insights from these and other stakeholders into the development of recommendations that integrate specific needs of individual areas with the needs of the Everglades ecosystem as a whole.

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Remediation to Restoration to Revitalization: Engaging Communities to Support Ecosystem-Based Management and Improve Human Wellbeing at Clean-up Sites

Kathleen C. Williams and Joel C. Hoffman

Abstract Remediation to Restoration to Revitalization (R2R2R) is a framework to identify ecological and policy-based relationships between large-scale aquatic sediment remediation projects, subsequent habitat restoration projects, and waterfront revitalization. A defining feature of R2R2R is that it possesses three essential feedback loops: a translational ecology feedback loop, an adaptive management feedback loop, and a project management feedback loop. The R2R2R framework builds on Ecosystem-Based Management (EBM) theory by addressing the role of humans through these feedback loops, and by recognizing the ability of communities to learn and make choices that improve the environment through translational science. In this framework, translating ecological changes from remediation and restoration projects to public benefits (e.g., swimmable water, potential for urban greenspace) using the concept of ecosystem services is critical to support decision-making. In practice, community perceptions and uses of the remediated and restored ecosystem or habitat are central to EBM. We use the Great Lakes Area of Concern program to illustrate how R2R2R exemplifies EBM for large, complex sediment remediation and aquatic habitat restoration projects.

Lessons Learned

- The Remediation to Restoration to Revitalization (R2R2R) framework is integrative of diverse interests through ongoing opportunities for engagement and a synthesis of input to inform research and project alternatives
- Consideration of translational ecology and adaptive management, in addition to the project, create distinct opportunities for engagement with the community, stakeholders, and project implementers

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- Health Impact Assessment can create science-based, community-relevant, and decision context relevant recommendations

Needs to Advance EBM
- Identify relationships between a positive change in environmental stressors, such as sediment contamination and habitat degradation, and improvements in human health or quality of life
- Case studies inclusive of a broad range of environmental management contexts that contribute to our social capacity for inclusive, equitable decision-making in social-ecological systems

1 Introduction: The R2R2R Framework

Remediation to Restoration to Revitalization (R2R2R) is a framework to identify ecological and policy-based relationships between large-scale sediment remediation projects, subsequent habitat restoration projects, and community revitalization. Ecological outcomes of remediation and restoration may be defined in terms of either ecological quality (e.g., a sediment quality target) or quantity (e.g., acres of wetland restored; Krantzberg 2003). Revitalization outcomes promote human-wellbeing, including social equity, while protecting or improving natural capital (Angradi et al. 2019). However, as a social-ecological system, the connections and feedbacks between remediation, restoration, and revitalization are not well-understood. In the Great Lakes region, remediation and restoration projects along urban waterfronts changed people's interactions with urban, aquatic ecosystems. For example, after projects were complete, increased use of trails and waterways, and changes in economic activity and land uses, improved people's quality of life (Krantzberg 2012; Hartig et al. 2019; Williams and Hoffman in review). However, a framework to identify the connection and feedbacks that led to the change in quality of life has not been available to researchers or managers. This impedes achieving a broader goal of maximizing ecological outcomes and social benefits to human well-being. Therefore, our goal was to develop this framework, recognizing that it should incorporate principles of Ecosystem-Based Management (EBM).

We developed R2R2R in the context of the Great Lakes Area of Concern (AOC) Program. The U.S.-Canada Great Lakes Water Quality Agreement (Annex 1, 2012) defines Areas of Concern as "geographic areas that fail to meet the general or specific objectives of the agreement where such failure has caused or is likely to cause impairment of beneficial use of the area's ability to support aquatic life." When the program was initiated in 1987, 43 AOCs were identified, 31 of which were entirely or partly in U.S. waters. Most AOCs are located near population centers, are within the Great Lakes coastal zone, and are degraded by legacy contaminants including heavy metals and persistent organic pollutants (Hartig et al. 2019). The program recognizes 14 distinct beneficial use impairments (BUIs). Nearly all AOCs have multiple impaired beneficial uses, which arise from multiple causes including sediment and water contamination, habitat loss, excess nutrients and sediment inputs, and improperly functioning storm or sewer systems (Hartig and Zarull 1992). It is common for Superfund sites to be located within an AOC, and for brownfields to be located nearby. The AOC program's goal is to remove BUIs through contaminated sediment remediation, aquatic habitat restoration, or both. In AOC communities, revitalization is primarily targeted towards urban waterfronts, and is inclusive of policies or actions on waterfronts or in adjacent aquatic areas.

Our objective is to describe the R2R2R framework and demonstrate how it builds upon EBM theory. We anticipate the R2R2R framework can be applied broadly to the problem of remediation, for example in brownfields or Superfund programs (Lipps et al. 2017) and recognize that important details of implementation will depend on each program's requirements and legal authorities. For example, under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, also known as Superfund), the definition of natural resources and the criterion for future use (e.g., residential) will influence remediation, restoration, and revitalization decisions (Burger 2008). For the purposes of this chapter, we will speak of R2R2R in general terms, and provide examples specific to the Great Lakes AOC program. We conclude with a case study which illustrates that integrating social and ecological knowledge is possible by integrating a formal decision-support system (Health Impact Assessment) designed to consider diverse knowledge and values into the R2R2R framework.

2 R2R2R Framework as a Decision-Support System

The R2R2R framework describes a process for achieving management goals and project objectives within a social-ecological system (Fig. 1). Generally, R2R2R sites involve substantial engineering to address ecological stressors causing identified ecological impairments. Community members, stakeholders, and agencies choose the acceptable ecological or human risk at the remediation site, the type and quantity of habitats to be restored (within biological constraints), and the future uses and activities that will occur at the site (Krantzberg 2003). Because it is an engineered project within a community, the ecosystem undergoing remediation and restoration is embedded within a social system which may be organized by neighborhoods, user groups, municipalities, and state, federal, and tribal agencies. It is a flexible, adaptive system that is built upon, and responsive to, diverse community members and organizations. As such, R2R2R can also be described as an adaptive co-management system (Folke et al. 2002). As with adaptive co-management, R2R2R focuses on specific places (AOCs) and environmental stressors (BUIs), and emphasizes learning through management actions, adaptively evolving management activities and governance (Dietz et al. 2003; Olsson et al. 2004).

A defining feature of R2R2R is its three feedback loops: a translational ecology (TE) feedback loop, an adaptive management (AM) feedback loop, and a project management (PM) feedback loop (Fig. 1). As illustrated, the R2R2R framework is cyclical and iterative, and thereby explicitly recognizes ecological and social

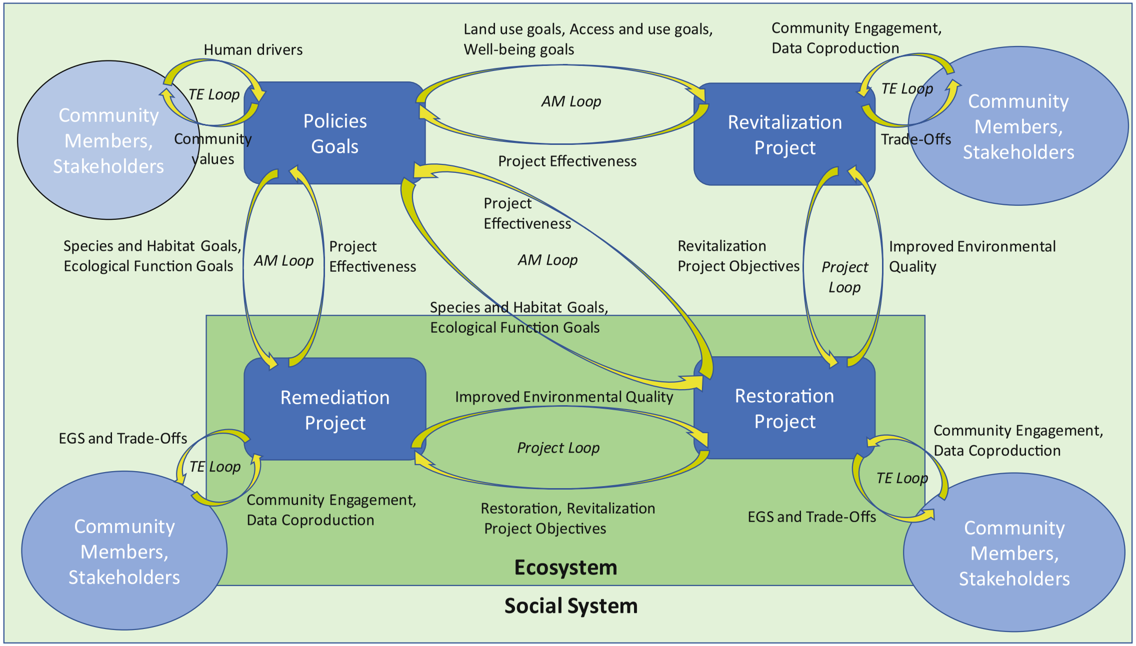

Fig. 1 Social-ecological system diagram of the Remediation to Restoration to Revitalization (R2R2R) framework. The framework includes four discrete stages (dark blue boxes): policy and goal setting, remediation projects, restoration projects, and revitalization projects. An adaptive management (AM) loop connects each of the project types to the policies and goals. The loop is based on the application of relevant goals to the projects and the measurement of project effectiveness to assess the project against the goals. The project loop connects each project, recognizing that the projects are dependent on each other such that the level of environmental quality or ecological integrity obtained is linked to the objectives of the other projects. A translational ecology (TE) loop connects each stage to the community and stakeholder through the application of local knowledge via data coproduction and community engagement to the project and the flow of ecosystem goods and services (EGS) from the project (and related EGS trade-offs when evaluating project alternatives) to benefit community members

complexity and interactions. For example, the cycle could begin with a plan for revitalization, which in turn may spark community, stakeholder, or decision-maker interest in remediation and restoration. These feedback loops are linked. For example, the TE and AM feedback loops intersect during community engagement as the community helps to define the project ecological and social goals.

In translational ecology (TE), scientists and community members collectively identify ecological and social goals, and address problems related to both environment and society (Enquist et al. 2017). Translation is built on knowledge exchange between scientists and stakeholders, and thereby promotes mutual learning. This approach is shared with EBM (Slocombe 1998) and is inherent to the design the Great Lakes AOC program (Krantzberg 2012). The R2R2R framework requires an iterative, participatory TE process that is dynamic and adaptive because the underlying ecological system is complex and the social system includes agencies, non-government organizations, and socio-economically diverse communities. For AOCs, R2R2R is inherently translational; when their original Remedial Action Plans (RAPs) were created in the late-1980s, state agencies were advised by scientists, local governments, and community members (Botts and Muldoon 2005; Hartig and Zarull 1992). The TE feedback loop ideally continues through four discrete stages of the R2R2R process (policies and goals, and remediation, restoration, and revitalization projects; Fig. 1). Scientists engage with community members and stakeholders to gather knowledge and co-produce scientific data, which are translated into potential ecosystem goods and services (EGS), with associated benefits or losses (trade-offs) identified based on project alternatives (e.g. Martin et al. 2018; DeWitt et al. 2020). This information is provided to decision-makers, and their decision in turn affects researchers and community members (Daily et al. 2009; Wall et al. 2017). In TE, a wide range of stakeholders and community members are included. To address related environmental justice concerns, extra effort may be required to engage under-represented and vulnerable populations that are directly or indirectly affected by the contaminated site (Geller et al. 2016).

In the adaptive management (AM) feedback loop, periodic evaluations based on project-effectiveness metrics determine whether the project is meeting identified targets relative to either goals (program or policy scale) or objectives (project scale; Slocombe 1998). If those goals are not met, decision-makers must modify the project to address identified shortcomings, and the cycle of evaluation and modification continues (e.g., McLain and Lee 1996). If decision-relevant endpoints are chosen that convey value to the public regarding the ecosystem services gained or lost through the remedy (e.g., water of sufficient quality to use, fish of sufficient quality to safely consume) or restoration (e.g., wetlands of sufficient quality to support ecologically sensitive, charismatic species), metrics can potentially be used for both public communication and in a translation ecological context (Allan et al. 2015; Angradi et al. 2016; Olander et al. 2017; Wall et al. 2017). Ideally, within the AOC program, AM for remediation and restoration projects begins with discrete management actions to support project implementation, followed by project implementation, monitoring, and evaluation, and ultimately (if impairments are successfully removed) AOC delisting. In between the time when all identified management actions are complete and delisting, there is opportunity for the AOC community (including state agencies, citizen advisory committee, and potentially stakeholders and community members) to address additional remediation and restoration needs as necessary.

Remediation goals are developed by evaluating the risk or impact to the ecosystem from chemical exposure, which can be accomplished formally though ecological risk assessment or through other evaluations to determine ecosystem impairments (Burger 2008). Remedy effectiveness is an approach to determine whether remediation goals are met, wherein physical, chemical, and biological lines of evidence (LOE) are measured prior to, potentially during, and after remediation (e.g., Meier et al. 2015). Multiple LOE such as sediment or porewater contaminant concentration, toxicity tests, contaminant bioaccumulation, indices of biotic integrity, or bioassays are measured to evaluate the ecosystem response. Ideally, at least some LOE have associated targets (i.e., a desired value post-remediation).

Restoration effectiveness is an emerging concept within the R2R2R framework. We assume here that the remediation project has at least some ecosystem restoration goals. The purpose of restoration is to address ecological stressors and in the context of the AOC program, which is focused on riverine, coastal, and riparian habitats, these stressors include aquatic habitat loss or degradation, sedimentation or erosion, and invasive species (Hartig and Zarull 1992). In R2R2R, identified ecological goals include multiple species and habitats, and may address supporting ecological functions (e.g., nutrient cycling or fish spawning habitat). The topic of developing habitat restoration targets has been addressed at-length (e.g., Ruiz-Jaen and Aide 2005; Palmer et al. 2005). Some targets, such as water quality criteria, must specifically address program needs. Other targets which are ecologically relevant (i.e., an index of biotic integrity or metric of stress) and devised for various levels of biological organization (species to ecological community) may support AM decisions.

Revitalization effectiveness is a relatively new concept in Great Lakes AOCs. Remediation has economic benefits, which have been measured in AOCs by hedonics and economic impact analysis (e.g., Isely et al. 2018). For R2R2R, revitalization is best understood as a positive change in community wellbeing (Krantzberg 2003). Community-level social and economic changes can occur in concert with remediation and restoration efforts (Krantzberg 2012). For example, AOC practitioners associate remediation with positive social changes, including increased recreational use of the waterfront and changes in land use or waterfront business (Williams and Hoffman in review). While environmental, economic, social, and governance metrics have been proposed (Angradi et al. 2019), few formal studies exist that demonstrate broad-based changes in community wellbeing or quality of life (Krantzberg 2012).

The PM feedback loop recognizes that the community and scientists may identify project-specific goals that require certain ecological targets to be met at the preceding step. Often, risk assessment, remediation, and restoration are conducted separately and independently; however, integration can achieve efficiencies in terms of time and energy (Burger 2008). In the R2R2R framework, we postulate that integration can yield a greater ecological and social impact than would be achieved by conducting remediation, restoration, and revitalization projects separately. However, we are not aware of studies evaluating the relative success of integrating these elements.

3 R2R2R and Ecosystem-Based Management

As a framework, R2R2R shares foundational elements with Ecosystem-Based Management (EBM; Grumbine 1994; Arkema et al. 2006; Ruckelshaus et al. 2008). These qualities include that it is normative (reflects specific values), principled (aims to improve public good), integrative of different interests by synthesizing a wide range of information and knowledge garnered through engagement and participation, accommodating of complexity and change through feedback loops, is explainable to a wide group of people through the translational ecology loop, and is adaptive (Box 1; Slocombe 1998). In R2R2R, project workflow is relational and directional, such that the outcomes of the last stage (revitalization) are linked to both the initial conditions and success of prior stages (remediation and restoration). As such, ideally, discussions regarding all three stages and the associated projects.

Box 1 Principles of the Remediation to Restoration to Revitalization Framework

Ecological integrity and sustainability

To restore ecological integrity, the program adheres to a sustainable relationship between humans and ecosystems recognizing that the capacity of ecosystems to support life has been substantially diminished because of human actions (i.e., ecological impairments). An important outcome is the creation or restoration of the cultural, spiritual, or experiential relationship between people and the river.

Spatial planning

The spatial distribution of people, resources, and ecosystems is critical to R2R2R. Remediation and restoration are spatially-specific activities that occur within a landscape mosaic of human and ecological communities. Sediment contamination and aquatic habitat loss or degradation are spatially discrete and heterogeneously distributed, though stressors may occur at larger spatial scales. The built environment is amenable to spatial planning.

Effectiveness metrics

Effectiveness metrics inform whether remediation, restoration, and revitalization project objectives are met. Adaptive management can occur in ecological and social domains, and therefore metrics and pre- and post-project monitoring should occur in both domains.

(continued)

Box 1 (continued)

Remediation, restoration, and revitalization adaptively linked

Objectives and outcomes of remediation, restoration, and revitalization projects flow from one project to another, and thus success is interlinked. While remediation and restoration objectives are often established by government agencies, because revitalization objectives are community-based, community values and use objectives can inform remediation and restoration objectives.

Agency of people

Where science and policy knowledge are produced together, the process undertaken to engage stakeholders and the public matters. In the context of decision-support (e.g., health impact assessment), collaborative group processes facilitate mutual learning. Trust and legitimacy, which are founded on facilitation and knowledge sharing, are important for project success.

Social system integral to the framework

The translational feedback loop integrates community values and knowledge with scientific knowledge to inform project objectives, which in turn aims to improve provisioning of ecosystem services for community wellbeing. Recommendations to achieve ecological and social goals arise from community members, stakeholders, and scientists, and are responsive to policy goals (e.g., removal of ecological impairments).

Participatory process that integrates different kinds of knowledge

Translational ecology is used to integrate scientific, local, and traditional ecological knowledge using participatory processes that adhere to principles of democracy, collaboration, communication, and equity. The participatory process is organized as a cooperation among individuals, nongovernment organizations, municipalities, and state, federal, and tribal agencies working at neighborhood, city, reservation, and state scales.

In R2R2R, as with EBM, ecosystem services provisioning is fundamental to sustainability over time (Levin and Lubchenco 2008). Ecosystem goods and services (EGS) are outputs of nature that contribute to human wellbeing when consumed, used, or enjoyed (Bruins et al. 2017). Coastal wetlands and river mouth estuaries (Larson et al. 2013), the most common habitat impacted in AOCs, provide a diverse array of ecosystem services (Sierszen et al. 2012). Beneficial use impairments in AOCs such as fish consumption advisories, beach closings, or dredging restrictions represent ecosystem services loss. Moreover, EGS respond directly to alteration of the biophysical state of the AOC (Yee et al. 2020), provide a consistent and comprehensive set of benefits for consideration, and can be mapped or quantified to illustrate trade-offs to the public (Angradi et al. 2016).

It is important to recognize in Fig. 1 the relationships between ecological integrity and ecosystem services on the ecosystem side, and wellbeing and equity on the social system side (Schoeman et al. 2014; Piet et al. 2020). An accounting of ecosystem services can provide measures of community benefits that may not be easily recognized (Olander et al. 2017). By embedding EGS as the connection to wellbeing and equity, we recognize trade-offs are central to decision-making (Angradi et al. 2016; Martin et al. 2018). As such, recommendations can be made to ensure that impacts do not disproportionately affect under-represented, disadvantaged, and vulnerable groups. Here again, we need decision support processes that integrate different kinds of knowledge, including traditional ecological knowledge (Berkes et al. 2000), recognize the importance of social inclusiveness, and provide recommendations to improve equity.

It is also important to recognize that the restored ecosystem and community are embedded within larger systems, such as a watershed and state, respectively. This external relationship may present concerns related to resiliency, reversibility, source control, long-term stewardship, or other factors that arise from outside the control of the project area or community and which need to be addressed as part of the solution (Adger et al. 2005; Levin and Lubchenco 2008; Palumbi et al. 2008). The Great Lakes AOC program can address AOC-specific structural goals (habitat amount, cultural features) and organizational goals (ecological productivity or connectivity, human use and development), but can only contribute to process goals (biodiversity and evolutionary complexity, quality of life; Slocombe 1998), which are driven by large-scale stressors (e.g., Allan et al. 2015).

One of the animating questions in EBM, and in R2R2R, is *who* will do this work? Substantial challenges exist with respect to our understanding of social-ecological systems for implementing integrated solutions. For example, social dimensions of social-ecological systems remain poorly defined (Brown 2009, 2014), and when considered are limited to studies of scale, governance, and institutions (Brown 2014; Turner 2014). Moreover, the focus on "the functionality of institutions and …normative issues as outcomes," instead of as integral parts of the system (Cote and Nightingale 2012) minimizes the attention given to relations of power, diverging interests, and social identities (Brown 2014; Turner 2014) that often challenge the creation of sustainable solutions. In R2R2R practice, we therefore ask, "what is the role of scientists, decision-makers and managers, public advisor groups, and citizens?" Are roles clarified, aligned, and sufficiently supported for R2R2R to be successful? How do we address ongoing conflict between government processes that traditionally rely on technological knowledge versus community processes that rely on integrated knowledge? Current environmental management practice often treats these different interests as competing, but an agonist approach (Mouffe 1999) suggests that finding ways to treat these different types of knowledge as equal can be productive (Cote and Nightingale 2012). To address the underlying drivers of environmental degradation, we need to engage community morals and values, as well as promote shared benefits of revitalization (Groenfeldt and Schmidt 2013; Daigneau 2015). This is where TE principles (collaboration, engagement, commitment, communication, process, and decision-framing) are critical to connect research and practice (Lawson et al. 2017; Wall et al. 2017).

R2R2R can address potential shortcomings with respect to community engagement that can occur in EBM (Krantzberg 2003). It does so by stipulating community engagement and translational science to occur at each part of the process, recognizing that different community groups may be engaged with and impacted by each part of the process differently. We postulate that for R2R2R to be successful, there must be a translational component that facilitates learning given that remediation and restoration projects generally rely on scientific or technological knowledge (Cote and Nightingale 2012; Partidario and Sheate 2013). To do so requires a decision-support process that follows TE principles and is based on constructivist learning theory, such as impact assessment (e.g., health impact assessment; Partidario and Sheate 2013) or structured decision-making (Sharpe et al. 2020). For implementation, R2R2R requires concepts and strategies from social science that aim to span the boundary between scientific knowledge and decision-making (Mollinga 2010; Williams 2015). Next, we demonstrate how using a constructivist approach can yield a broad range of potential political solutions and recommendations through community engagement (Cote and Nightingale 2012; Evans 2011). The following case study explores how this engagement may produce an integrated solution.

4 Implementing R2R2R Case Study

The U.S. Environmental Protection Agency designated the St. Louis River Area of Concern (AOC) in 1987 owing to historical degradation including discharge of untreated wastewater and debris from industrial and municipal facilities (MPCA and WDNR 1992). The AOC includes the lower 63 km of the St. Louis River. The port cities of Duluth, Minnesota and Superior, Wisconsin are situated at the river's mouth where it flows into Lake Superior. With respect to sediment contamination, chemicals of concern include polychlorinated biphenyls (PCBs), polychlorinated dibenzodioxins (PCDDs, or dioxins), polychlorinated dibenzofurans (PCDFs, or furans), polyaromatic hydrocarbons (PAHs), and heavy metals, all of which are present at multiple locations in the AOC (Crane et al. 2005). State (Minnesota, Wisconsin) and tribal (Fond du Lac Band of Lake Superior Chippewa) agencies coordinate sediment remediation and habitat restoration projects. In Duluth, the neighborhoods adjacent to the river were once home to numerous saw mills, coal tar processing facilities, and steel mills, but today are suffering from poverty and poor health outcomes (Williams et al. 2019).

With sediment remediation underway, the City of Duluth is revitalizing the "St. Louis River Corridor" through the development of active recreation opportunities including trails, regional parks, and improved access to the river in these same neighborhoods (City of Duluth 2018). The AOC and community revitalization processes intersect at the Kingsbury Bay and Grassy Point habitat restoration project. Kingsbury Bay is adjacent to a city park and campground that is also a historical Native American camp, along which a riverfront trail will be improved. Grassy Point is city-owned green space along the riverfront; it is a wetland complex and has an unimproved boat launch and a wetland boardwalk with a fishing pier that are in disrepair. Kingsbury Bay is impacted by long-term excess sedimentation that resulted in aquatic habitat loss. Grassy Point was the location of two former saw mills, resulting in sediment contamination from both wood wastes and dioxins. The two sites are about 2 km apart, and the combined project covers about 1 km^2 of river and involves dredging 268,000 m^3 of sediment. The restoration will beneficially use sediment dredged from Kingsbury Bay to build habitat at Grassy Point Park, while remediation work as Grassy Point includes wood waste removal and covering contaminated sediments. The project will improve river access in economically-disadvantaged neighborhoods with poor access (USEPA 2019). Project objectives include adding a river trail, a swimming beach, canoe and kayak landings, board-walks, fishing piers, and interpretative signage. The project area also has potential for supporting plants including wild rice that are culturally important to indigenous people.

Recognizing the complexity of design decisions, project implication for the health of adjoining neighborhoods, and a diversity of community members and stakeholders, we conducted a health impact assessment (HIA) to create evidence-based recommendations to inform the Minnesota Department of Natural Resources project design (i.e., sediment remediation and aquatic habitat restoration) and the City of Duluth park planning (i.e., site access and amenities). Health impact assessment (HIA) is a systematic process that uses a variety of data sources and community and stakeholder consultation to ascertain potential health impacts of a policy change or decision, and to make recommendations that mitigate negative health impacts or enhance positive health impacts (National Research Council 2011). Health impact assessment is built upon democratic participation, health equity, and a comprehensive approach to health (i.e., community wellbeing; Quigley et al. 2006). Further, HIA assumes the community is a stakeholder (Human Impact Partners 2011), which is important given a lack of consensus regarding long-term plans for redevelopment along the river (Johnston et al. 2017).

The inclusion of community input early in the process ensures community values are considered at a point when values can influence recommendations provided as part of the HIA (Iroz-Elardo 2014). In the Kingsbury Bay-Grassy Point habitat restoration project, we started the process with two introductory meetings for stakeholders, one for the general public and one for group representatives (i.e., city planners, non-government organizations, and natural resource managers). The meetings were nearly identical and included an introduction to HIA, an overview of the project, and opportunities for discussion and input. To ensure opportunities for meaningful inclusion, we captured different types of knowledge and experience. Participants in both meetings were invited to share their experiences with the two sites by placement of sticky notes on project maps, and to explain how the changes would impact their daily lives or experiences (Boschmann and Cubbon 2014; Johnston et al. 1995). The submitted comments reflected a variety of personal and professional experiences and conveyed individual perspectives regarding sense of place and identity. Comments included concern about park maintenance, safety, access, traffic, invasive species management, disturbance of an adjacent Superfund site, and identified opportunities for social gathering, birding, fishing, and boating.

Health impact assessment is designed to mediate power relations through an inclusionary and iterative process. In the Kingsbury Bay-Grassy Point habitat restoration project HIA, all submitted comments were analyzed to identify the social and environmental determinants of health and wellbeing valued by community members and stakeholders (Marmet et al. 2008). This community-based information was used to construct health pathways, which identify how project elements are likely to effect health outcomes. We identified impacted ecosystem services to relate the proposed biophysical changes at the site to public-friendly endpoints such as swimmable water or edible fish, and then to health benefits. Transparency was ensured throughout the process through ongoing communication and opportunities for input. Detailed meeting notes that include all comments, questions, and participants were distributed after each meeting, and community members and stakeholders could make suggestions or corrections to any set of notes produced. Community members and stakeholders were invited to participate in technical and community committees that advise the HIA through each step.

In total, the HIA yielded 77 unique recommendations to improve project health outcomes. Recommendations addressed trail safety, environmental quality, cultural resources, and social places for gathering. Recommendations also addressed the ecological quality of the project, including to protect existing high-quality wetlands and to implement invasive species controls. Importantly, the process was widely endorsed by community members, stakeholders and decision-makers as positive and constructive. We found that HIA is helpful in the AOC context because it provides a means to explore how sediment remediation, habitat restoration, and park management are interrelated, and how this will collectively impact community health as defined by community members and stakeholders. In this case study, the HIA informed both final landscape design for habitat restoration and will be included in future park master planning.

Broadly, HIA integrates with the three loops of R2R2R, and therefore may be widely useful. In this case study, HIA served as a formal approach to decision support that facilitated translational science. Other decision support approaches might also be used to support TE (e.g., Sharpe et al. 2020). The TE loop included the community through engagement and data co-production; community knowledge was foundational to the science-based pathways for impact analysis. The AM loop related HIA recommendations to restoration projects and city park plans, which changed in response to research and collective discussions about the value of Kingsbury Bay and Grassy Point Park. Finally, the PM loop linked revitalization outcomes such as safe swimming beaches and birdwatching opportunities to ecosystem services that were necessary to restore through remediation and restoration.

5 Conclusions

It has long been recognized that successful implementation of an ecosystem approach to integrate environmental and social decision-making requires attention to both the substance and process of the approach (Slocombe 1993; McLain and Lee 1996). The R2R2R framework builds on EBM theory by addressing the role of humans (e.g., natural resource agencies, community members) through multiple feedback loops, and recognizing their ability to learn and make choices that improve the environment through translational science. We demonstrate that integrating social and ecological knowledge is possible by utilizing the R2R2R framework with a specific process (HIA) designed to consider diverse knowledge and values. Ecosystem-Based Management is difficult because the benefits humans and society derive from ecosystem processes cannot be viewed as objectively existing "out there," but as entangled in social and political processes (Ernston 2013). The R2R2R framework provides interlinking loops of translational ecology, adaptive management, and project management as a system for integrating these diverse processes. As a community of practice, R2R2R is relatively new, and we anticipate that ongoing remediation and restoration efforts in the Great Lakes and elsewhere will contribute to our social capacity for inclusive, equitable decision-making in social-ecological systems.

Disclaimer This chapter has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the author(s) and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

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Predicting Future Vegetated Landscapes Under Climate Change: Application of the Environmental Stratification Methodology to Protected Areas in the Lower Mekong Basin

John M. Johnston, Robert J. Zomer, and Ming-cheng Wang

Abstract There are 176 protected areas within the Lower Mekong Basin (LMB), comprising 183,000 km^2, almost 30% of the LMB. Climate change poses challenges to their management, because changes in the timing and amount of rainfall and maximum temperature from current (baseline) conditions alter vegetation growth and composition. The prediction of future climate, i.e., the pattern of temperature and rainfall expected 30–60 years from present, is accomplished using Earth System Models (ESMs). However, future ecosystem structure, including dominant vegetation, is less well studied. A successful approach is Environmental Stratification (EnS), involving statistical analysis of climate variables to identify relatively homogeneous spatial climate patterns (zones and strata) that are robust predictors of vegetation associations. Our objective was to predict changes in the spatial distribution of bioclimatic conditions across the LMB by the year 2030 and 2060, based on downscaled (1 km^2 resolution) ESM projections. Five major bioclimatic zones and twenty-five bioclimatic strata were identified using EnS, ranging from extremely hot and xeric at the lower elevations, to warm temperate and mesic at higher elevations. The largest expanse of area is extremely hot and moist (50% of total area), followed by extremely hot and xeric (24%), and extremely hot and mesic (18%), with mean annual temperature for the various zones ranging from 18.1 to 27.2°C. More than 9% to 29% of all protected areas are projected to shift to a different bioclimatic zone by 2030, and from 7% to over 77% by 2060.

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Lessons Learned

- Up to 29% of protected areas will shift to a new bioclimatic zone by 2030, necessitating adaptive management to adjust the current boundaries and their level of protected status.
- Ecosystem-based adaptation (EbA), a type of ecosystem-based management (EBM), provides a holistic approach to habitat and species protection under climate change.
- Coordinated management efforts across national boundaries will be important in the future of protected areas in the Lower Mekong Basin (LMB).
- Both environmental stratification (EnS) and the analysis of projected change in spatial distribution of bioclimatic conditions provide reliable information for climate change adaptation planning.

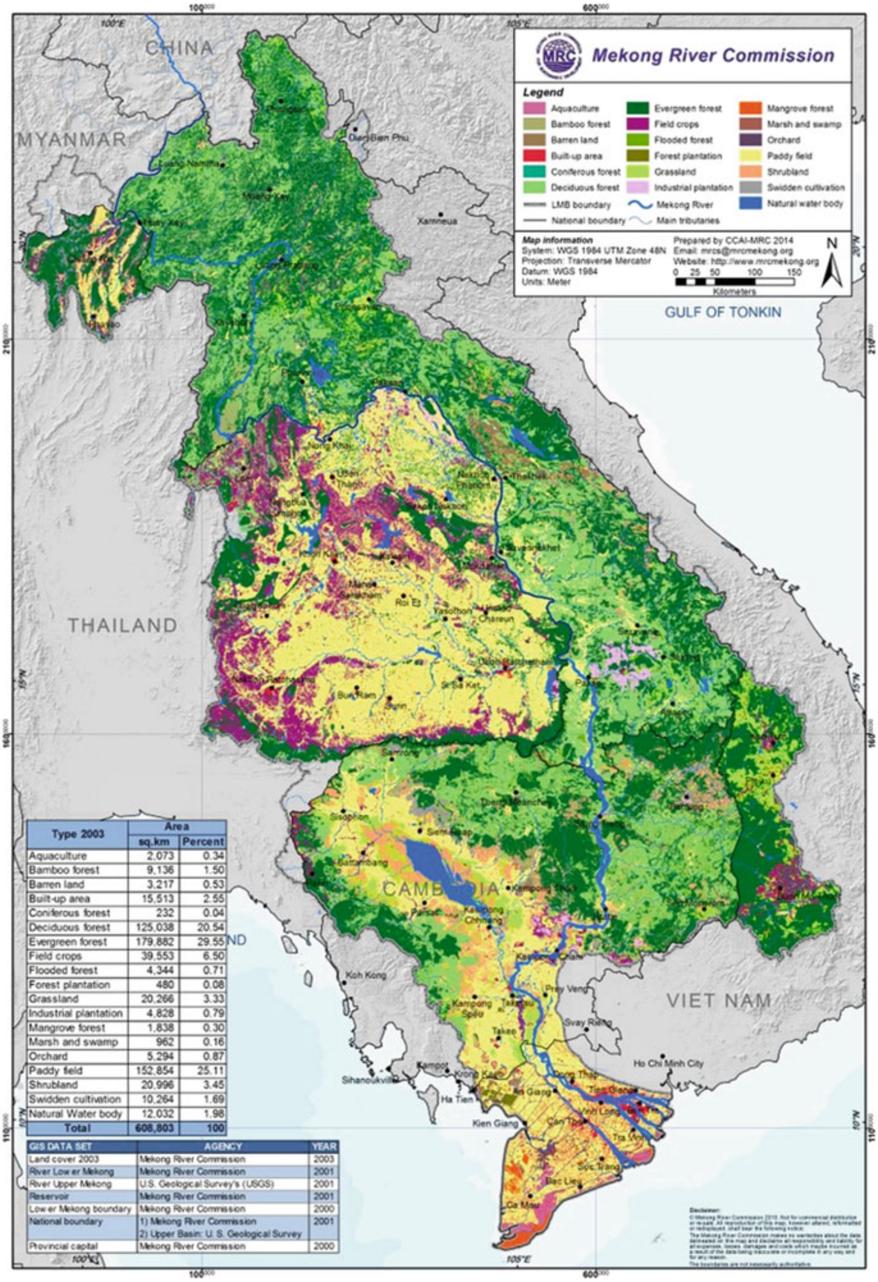
Needs to Advance EBM

- Conservation research and monitoring efforts in the LMB have focused more on freshwater ecosystems and species, and a greater emphasis is needed in the future on terrestrial ecosystems and vegetation assemblages as provided by EnS.
- EBM of terrestrial ecosystems, including protected areas, must account for human uses and co-existence, possibly through emphasis on non-timber forest products for food and livelihoods, permitting sustainable harvest of renewable resources by local populations.
- The interaction of aquatic and terrestrial systems in the LMB is a needed research area to achieve greater sustainability of the food-energy-water nexus.
- EbA strategies are needed in agricultural areas of the LMB, especially the Khorat plateau, where temperatures are expected to increase and rainfall decrease, leading to crop stress and failures, increasing food insecurity and further stressing protected areas.

1 Introduction

1.1 Lower Mekong River Basin Context: Ecosystem, People and Challenges

The Mekong River Basin beginning in Lao People's Democratic Republic (PDR) is referred to as the Lower Mekong Basin (LMB), and it supports over 60 million people (MRC 2011c). The LMB includes portions of the riparian countries of Lao PDR, Vietnam, Cambodia and Thailand. It is estimated that over 60% of the LMB population is connected to the Mekong River for its food and livelihood (Fig. 1). Lao PDR and Cambodia plan to graduate from least developed country status by 2020, with Vietnam achieving middle-income status by 2030. Aquaculture is forecast to double to 4 million metric tons in 20 years, with dry season irrigation expansion planned to increase by 50% to 1.8 million ha, including mainstream water transfers planned by Thailand to alleviate drought conditions in the Northeast (MRC 2011a).



Type 2003	Area sq.km	Area Percent
Aquaculture	2,073	0.34
Bamboo forest	9,136	1.50
Barren land	3,217	0.53
Built-up area	15,513	2.55
Coniferous forest	232	0.04
Deciduous forest	125,038	20.54
Evergreen forest	179,882	29.55
Field crops	39,553	6.50
Flooded forest	4,344	0.71
Forest plantation	480	0.08
Grassland	20,266	3.33
Industrial plantation	4,828	0.79
Mangrove forest	1,838	0.30
Marsh and swamp	962	0.16
Orchard	5,294	0.87
Paddy field	152,854	25.11
Shrubland	20,996	3.45
Swidden cultivation	10,264	1.69
Natural Water body	12,032	1.98
Total	**608,803**	**100**

GIS DATA SET	AGENCY	YEAR
Land cover 2003	Mekong River Commission	2003
River Lower Mekong	Mekong River Commission	2001
River Upper Mekong	U.S. Geological Survey's (USGS)	2001
Reservoir	Mekong River Commission	2001
Lower Mekong boundary	Mekong River Commission	2000
National boundary	1) Mekong River Commission 2) Upper Basin: U. S. Geological Survey	2001
Provincial capital	Mekong River Commission	2000

Fig. 1 Lower Mekong Basin study area. Land cover types are provided basinwide for all watersheds contributing to the Mekong River

The food, water, and energy needs of the population are therefore expected to increase pressure on LMB ecosystems, with climate change an additional long-term driver to be considered. The Mekong River Commission (MRC) was created in 1995 by the riparian governments of the LMB to promote sustainable development for the greater good of all countries (MRC 2011c).

Although the overall amount of water is not considered limiting in the LMB, the Southwest Monsoon results in strong seasonality of rainfall, with very little precipitation occurring in the dry season from November through May. Dry season food cultivation is dependent on irrigation from surface waters of the river and adjacent ecosystems. Because of their interdependence, water, energy, and food must be managed concomitantly, with careful consideration of tradeoffs that directly and indirectly affect the nexus (Linke 2014; Rasul 2014). Downstream ecosystems and national and local community dependence on flows from the northern basin are transboundary issues that require integrated planning and coordination. As a type of Ecosystem-Based Management (EBM), attention is being given to ecosystem-based adaptation (EbA) strategies for managing climate change impacts (Munang et al. 2013; Sierra-Correa and Cantera Kintz 2015; Vignola et al. 2013; Wamsler et al. 2014). An EbA strategy balances infrastructure development with investments in natural capital for more resilient and sustainable solutions. Protection of LMB watersheds contributing flow and material input to the Mekong is a primary EbA strategy in consideration (ISPONRE 2013; MONRE 2013).

These promising examples of national-level initiative within the LMB for EbA to climate change impacts were developed for Vietnam and Lao PDR. Each provides a national level operational framework for a participatory process of mainstreaming ecosystem services concepts and practices, outlining a fairly detailed process with example tools and methods to facilitate replication of the framework. A guide for decision makers is also recommended as a companion document with more detailed descriptions and examples of the concepts (Ranganathan et al. 2008). An important and necessary complement to the national level is a basin-wide strategy for transboundary adaptation planning in the LMB. Ecosystem services of water, nutrient, and sediment provisioning provided by upstream forest and wetland ecosystems are critical to maintaining delta fisheries and agricultural productivity and protection from saltwater intrusion, because many of the impacts of climate change cannot be managed effectively at the national level. Only when the suite of transboundary adaptation strategies are fully addressed will the MRC vision of “an economically prosperous, socially just, and environmentally sound Mekong River Basin” be realized (MRC 2010).

Within the MRC, the Climate Change and Adaptation Initiative (CCAI) has the mission of climate change impact assessment and adaptation planning and implementation within the LMB. CCAI has the objective of guiding climate change adaptation planning and implementation through improved strategies and plans at various levels and in priority locations throughout the LMB, which includes a mix of scoping, synthesis, outreach, and capacity building (MRC 2011b). A critical role of the MRC CCAI is to provide a consistent, basin-wide approach for evaluating transboundary climate change adaptation. The CCAI is dedicated to developing basin-wide datasets and assessment methods that can be used at multiple spatial scales (e.g., nationally and locally) whether data rich or data limited. This is an essential role of CCAI: no other group is committed to providing the basin-wide context for national studies and the ability to compare results consistently across the LMB. Outputs include basin-wide datasets, verified ecosystem modeling methods and knowledgebase for reuse by member countries, and the identification of data gaps and priorities for reducing uncertainty. The CCAI is responsible for providing data and modeling methods that are well documented (e.g., user's guides, peer-reviewed literature), freely available, and could be used at multiple spatial scales, from basin-wide assessments to province-level if needed within member countries. Forecasting the impacts of climate change on ecosystems and biodiversity was a focus of the CCAI (Trisurat et al. 2018), with multiple ecosystem service forecasting models in consideration for potential use in the LMB.

1.2 Landcover Change and Ecosystem Models for Scenario Analysis

Landcover data provide an efficient means of working with large geographic regions, and a number of models have been developed to utilize these data to characterize ecosystem services (Bagstad et al. 2013a, 2013b; Feng et al. 2011; Jackson et al. 2013; Vigerstol and Aukema 2011). The majority of these ecosystem services models utilize landcover data as input to quantify services and their value using biophysical models with a statistical or empirical formulation, a necessary strategy when working in data limited areas (Bagstad et al. 2013a; Daily et al. 2009). Attributes that differentiate the approaches, making some models more suitable than others, include availability, maturity, flexibility, and ease of use. The InVEST model developed by The Natural Capital Project is freely available, well documented, includes economic valuation methods, and has been applied around the world at multiple spatial scales to evaluate the impacts of land use and climate change. Examples include the impact of land use change on water yield, carbon storage, nutrient retention, sediment retention and biodiversity provisioning (Leh et al. 2013), the impact of hurricane and typhoon disturbance on water yield, water purification, soil conservation, carbon storage and biodiversity (Chiang et al. 2014), timber production and carbon sequestration losses due to hurricane damage (Delphin et al. 2013), sediment retention and water yield under land use and climate change (Su and Fu 2013), the impact of climate change on water yield for a low flow regime in a river basin (Marquès et al. 2013), and to assess the combined effects of land use and climate change scenarios on water yield, nutrient and sediment retention in 2050 (Hoyer and Chang 2014).

The InVEST model has also been used to evaluate payment for ecosystem services schemes and the value of hydropower energy production (Fu et al. 2014) and the economic value of wetland ecoservices water purification and carbon sequestration under three land use change scenarios (Harmáčková and Vačkář 2015). Water-related ecosystem services have received the most attention compared to other services, and Vigerstol and Aukema contrasted the InVEST and ARIES models for freshwater services provisioning with the watershed hydrologic models SWAT and VIC (Vigerstol and Aukema 2011). Sánchez-Canales et al. performed a sensitivity analysis of the InVEST water yield model and found it most sensitive to potential evapotranspiration and rainfall (Sánchez-Canales et al. 2012). Bai et al. used InVEST to evaluate the correlation and spatial overlap of sediment and nutrient retention, water yield and biodiversity services for possible management and restoration of regional hotspots (Bai et al. 2011). The InVEST model had been applied successfully within the LMB to address climate change impacts (ISPONRE 2013; MONRE 2013; Rosenthal et al. 2013). However, none of these models or approaches provides a forecast of future vegetation composition. Often, the most recent landcover data are used, held constant over the analysis period, or assumed to have a future composition as in scenario analysis. In this chapter, we explore Environmental Stratification (EnS) to develop bioclimatic strata and major bioclimatic zones for analyzing potential changes in LMB protected areas under different climate scenarios. EnS was selected by the CCAI to forecast future bioclimatic zones and the related vegetation structure in 2030 and 2060 and used to support the protected area analysis basinwide.

1.3 Environmental Stratification

Environmental Stratification (EnS) is a statistical technique for analyzing climate variables to identify relatively homogeneous spatial climate patterns (i.e., strata) that are in turn strongly associated with plant species and can serve as robust predictors of vegetation associations. Metzger et al. proposed using EnS to create a consistent, national to global biodiversity observation network (Metzger et al. 2013a, 2013b), based on a methodology involving maximum likelihood analysis and clustering (Zomer et al. 2014, 2015). Metzger et al.'s global bioclimatic map has been used to develop regional maps of climate associations and also as a reference for comparison with regional data (Zomer et al. 2014). The MRC and CCAI maintains basinwide climate and landcover data for the LMB that includes dominant vegetation types; these include, for example, coniferous, deciduous, and evergreen forests that are the necessary inputs for the EnS methodology (Fig. 1).

1.4 Biodiversity and Protected Areas in the LMB

Effective management of LMB ecosystems and their services requires a baseline assessment of status and ongoing monitoring and evaluation to assess the outcomes of decisions. This provides the necessary information for EbA—a key EBM strategy for navigating an uncertain future. As an example, forest ecosystems provide services related to wild foods and non-timber forest products (NTFPs). Households with access to productive forestlands have adaptation strategies not available to the population living in urban areas when food becomes scarce or expensive. Livelihoods are also supported from NTFP harvesting that brings income. The overall climate change adaptation strategy is one of decentralization to provide greater access and opportunity to the population. This is particularly important to communities in rural areas that have less resources and may have fewer options compared to those living in cities. The implication is that food security should consider the contribution of forest ecosystems to unconventional food supplies and subsistence economic opportunities. This is reinforced by a recent survey of biodiversity in Lao PDR: "The plants and animals harvested in the Mekong and its surrounding habitats provide an important source of food and income for the people who live along its banks" (IUCN 2013). Others have gone further to state that shifting agricultural practices in forest ecosystems are inherently more sustainable because they are less intensive and co-exist with forest ecosystems (Yokoyama et al. 2006). Numerous international experts recommend that "ecosystem-based approaches can contribute to climate change mitigation and adaptation and to sustainable development more broadly. Spatial planning for ecosystem services at international, national and local levels will be an important component of ecosystem-based approaches" (Leadley et al. 2010). The initial focus was on evaluating landscape change basin-wide and the use of these data to characterize selected ecosystem services for water provisioning and nutrient and sediment retention (Trisurat et al. 2018). Although protected areas and biodiversity had been evaluated at the provincial level, e.g., Nan Province, Thailand, it did not include climate impacts on bioclimatic zones that support terrestrial ecosystems (Trisurat et al. 2019). Changes in rainfall and temperature will influence vegetation patterns, potentially shifting species ranges to higher elevations and favoring new dominant and co-dominant plants compared with current communities (Zhang et al. 2014). Protected areas that are unchanging will likely fail to provide the range of habitats and corridors necessary for species to adapt to climate change (Monzón et al. 2011).

2 Study Area and Methods

The Mekong River in Southeast Asia drains an area of 795,000 km^2 that is referred to as the Lower Mekong Basin (LMB, Fig. 1). Annual mean precipitation ranges from approximately 1000 mm in northeast Thailand to over 3,500 mm in northern Lao PDR (Trisurat et al. 2018). This vast aquatic ecosystem flows over 4,800 km from its source in the Tibetan Plateau to the South China Sea below Ho Chi Minh. Of the approximately 475 km^3 annual average discharge, 12% or 60 billion m^3 are withdrawn for agricultural, industrial and other consumptive use. Hydropower energy potential of the LMB is estimated at 30,000 MW with only 10% developed; however, 26 hydropower projects are underway, with 12 mainstream and 30 tributary

Table 1 Protected areas in the Lower Mekong Basin

Lower Mekong Basin				
	Protected area		Area	
IUCN category	No. of PAs	% of PA	km^2	%
Ia	20	11.4	10,054	5.5
II	56	31.8	35,683	19.5
III	1	0.6	130	0.1
IV	27	15.3	36,524	20.0
V	8	4.5	6,431	3.5
VI	22	12.5	37,267	20.4
Not Applicable	10	5.7	39,069	21.3
Not Reported	33	18.8	17,914	9.8
Grand Total	*176*	*100.0*	*183,071*	*100.0*

IUCN categories provided by number (No.) of protected areas (PA), % of total PA, area in km^2 and % of total area for each category
Source: UNEP-WCMC (2019)

dams planned over the next 20 years (MRC 2011a). Ecosystem services provide food, water, and energy security to the extent they can be maintained in the face of a growing population and increasing demand for food and energy (MEA 2005). This includes services for habitat for protected species, with landcover and protected area data provided by the MRC (MRC 2011c). According to the World Database on Protected Areas (UNEP-WCMC 2019), there are 176 protected areas within the Lower Mekong Basin (LMB), comprising 183,000 km^2, almost 30% of the total LMB (Table 1). Excluding IUCN (International Union for Conservation of Nature) categories “Not Applicable” and “Not Reported”, protected areas still account for over 20% of the LMB.

Baseline climate was derived from data averaged from 1980 through 2010, while 2030 represented a 30-year average of the period 2015 to 2045, and 2060 represented the period 2045 to 2075 for terrestrial ecosystems using ESM ensemble climate forecasts from the SimCLIM dataset (Trisurat et al. 2018). The EnS methodology used the SimCLIM set of selected ESM and emissions scenario combinations, which included a multi-model ensemble ($n = 13$) of Coupled Model Intercomparison Project 5 (CMIP5, https://esgf-node.llnl.gov/projects/cmip5/) ESMs applied across three representative concentration pathways (RCP 2.6, 4.5, 8.5), to assess climate change impacts on vegetation growing conditions. Consistent with the Fourth National Climate Assessment (USGCRP 2018), the RCPs span a range of lower to higher greenhouse gas (GHG) concentration, corresponding to increased radiative forcing of 2.6, 4.5, and 8.5 watts per square meter (W/m^2). In other words, scenario RCP8.5 involves annual GHG emissions increasing throughout the twenty first century, leveling off by 2100, while lesser RCPs represent decreased GHG emissions (USGCRP 2018). Most global climate modeling experts agree that the biosphere is already experiencing a trajectory consistent with RCP8.5, unless substantial GHG emission mitigation steps are taken immediately, worldwide. Climate-envelope models identify constraints on species geographic ranges related to aspects of temperature, precipitation, relative humidity, including the minimum and maximum values (Lassiter et al. 2000). Bioclimatic envelopes characterize to the extent feasible the complete set of conditions conducive to ecosystems, species, and biological interactions, and are therefore used to gauge the impacts of climate change across the LMB protected areas.

3 Results

Five bioclimatic zones comprised of 25 bioclimatic strata were found within the LMB (Fig. 2), with zones ranging from extremely hot and xeric at the lower elevations to warm temperate and mesic at higher elevations. Mean annual temperatures are typically inversely correlated with average elevation. Additionally, 92.5% of the bioclimatic zones in the LMB are classified as extremely hot at baseline, and this percentage is expected to increase in 2030 under RCP8.5 (Table 2). Of particular note in 2030 is the reduction of the extent of the warm temperate and mesic zone from 1.5% to 0.2%. This represents a substantial, near-term loss of an already rare bioclimatic zone. There is also an increase in extremely hot and xeric (dry) from 11.4% to 17.1% (Table 2). By the year 2060 the zones appear to be limited to two primary types, extremely hot and mesic and extremely hot and moist, comprising 93.2% of the total area. Baseline LMB climate generally provides moist to mesic conditions conducive to forested ecosystems and rainfed agriculture; however, the warm temperate and mesic zone within protected areas diminishes under all scenarios and nearly disappears under RCP 8.5 by 2060. Bioclimatic zones for baseline and future climates in 2030 and 2060 across all emissions scenarios are illustrated (Fig. 3) and summarized for year 2060 across all emissions scenarios and compared to baseline and the entire LMB (Fig. 4).

The mean elevation of the bioclimatic zones within protected areas shifted upwards within mountainous terrain and across large plains or plateaus with elevation gradients under higher emission scenarios (Table 3). The extremely hot and mesic zone decreased in elevation; however, this is an artifact of the dramatic increase in extent of this zone across the LMB. The lower elevation, extremely hot and xeric zone shifted only marginally upslope in the highest emissions scenario by 2060, reflecting the small extent of this bioclimatic zone.

To emphasize the magnitude of bioclimatic change accompanying climate change, the percentage of the total area of each protected area that shifts to a different bioclimatic zone is shown (Fig. 5). Percent shifts in bioclimatic zone and strata are also summarized for the LMB and across all protected areas for both 2030 and 2060 and across all emissions scenarios (Table 4). Greater than 11% to almost 38% of the LMB is projected to shift to a different bioclimatic zone by 2030, and 9% to almost 88% will shift to a new bioclimatic zone by 2060 under RCP 8.5 (Table 4). Basinwide results were slightly greater than compared to protected areas, with 9% to 29% projected to shift to a different bioclimatic zone by 2030 and from 7% to more than 77% by 2060.

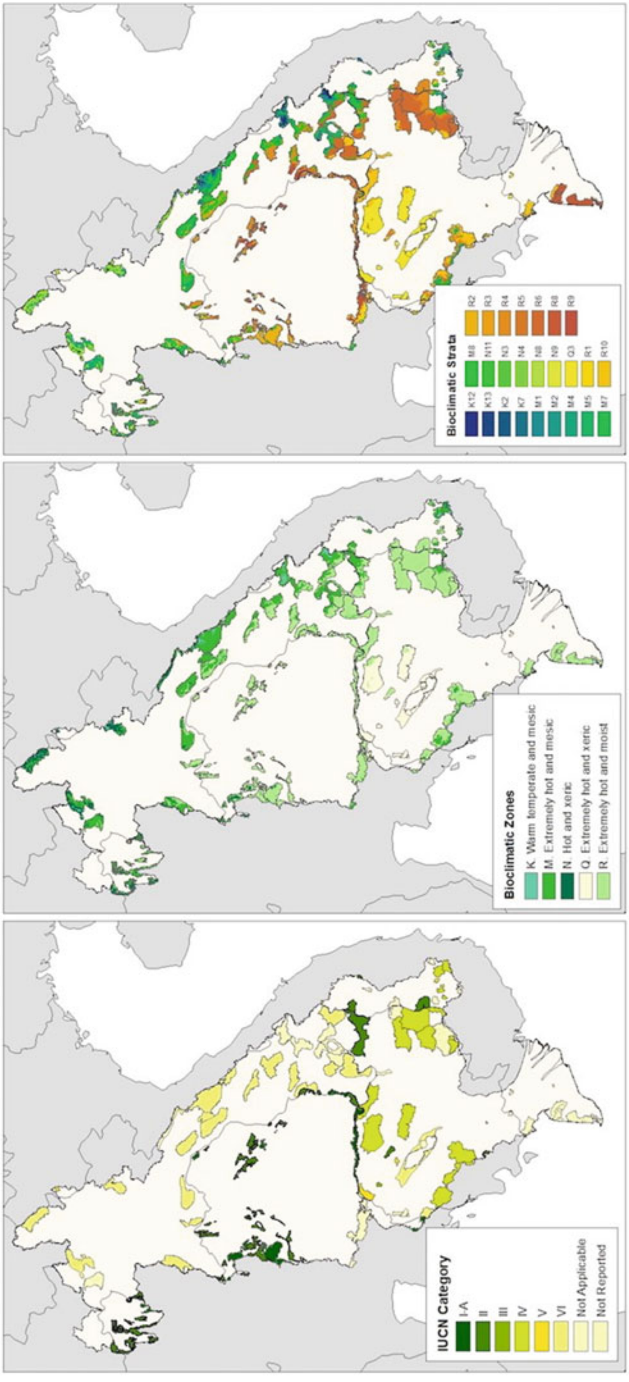


Fig. 2 Protected areas within the Lower Mekong Basin, also bioclimatic zones and strata within protected areas. (Source: UNEP-WCMC 2019)

Table 2 Areal extent (in km^2) and percent of total area (bottom) of bioclimatic zones within Lower Mekong Basin protected areas

Bioclimatic zone	Zone	1995	2030			2060		
			RCP_2.6	RCP_4.5	RCP_8.5	RCP_2.6	RCP_4.5	RCP_8.5
Area of bioclimatic zones within protected area within the Lower Mekong Basin								
Warm Temperate and Mesic	K	1,695	1,131	734	246	1,265	222	12
Extremely Hot and Mesic	M	29,136	26,615	25,145	27,596	27,189	29,487	79,688
Hot and Xeric	N	6,471	4,686	3,527	2,072	5,062	1,974	203
Extremely Hot and Xeric	Q	12,507	15,579	17,934	18,758	14,726	17,787	7,165
Extremely Hot and Moist	R	60,021	61,819	62,490	61,158	61,158	60,360	22,782
Percental of all protected area within the Lower Mekong Basin within bioclimatic zones								
Warm Temperate and Mesic	K	1.5	1.0	0.7	0.2	1.2	0.2	0.0
Extremely Hot and Mesic	M	26.5	24.2	22.9	25.1	24.8	26.8	72.5
Hot and Xeric	N	5.9	4.3	3.2	1.9	4.6	1.8	0.2
Extremely Hot and Xeric	Q	11.4	14.2	16.3	17.1	13.4	16.2	6.5
Extremely Hot and Moist	R	54.6	56.3	56.9	55.7	56.1	55.0	20.7

Source: UNEP-WCMC (2019)

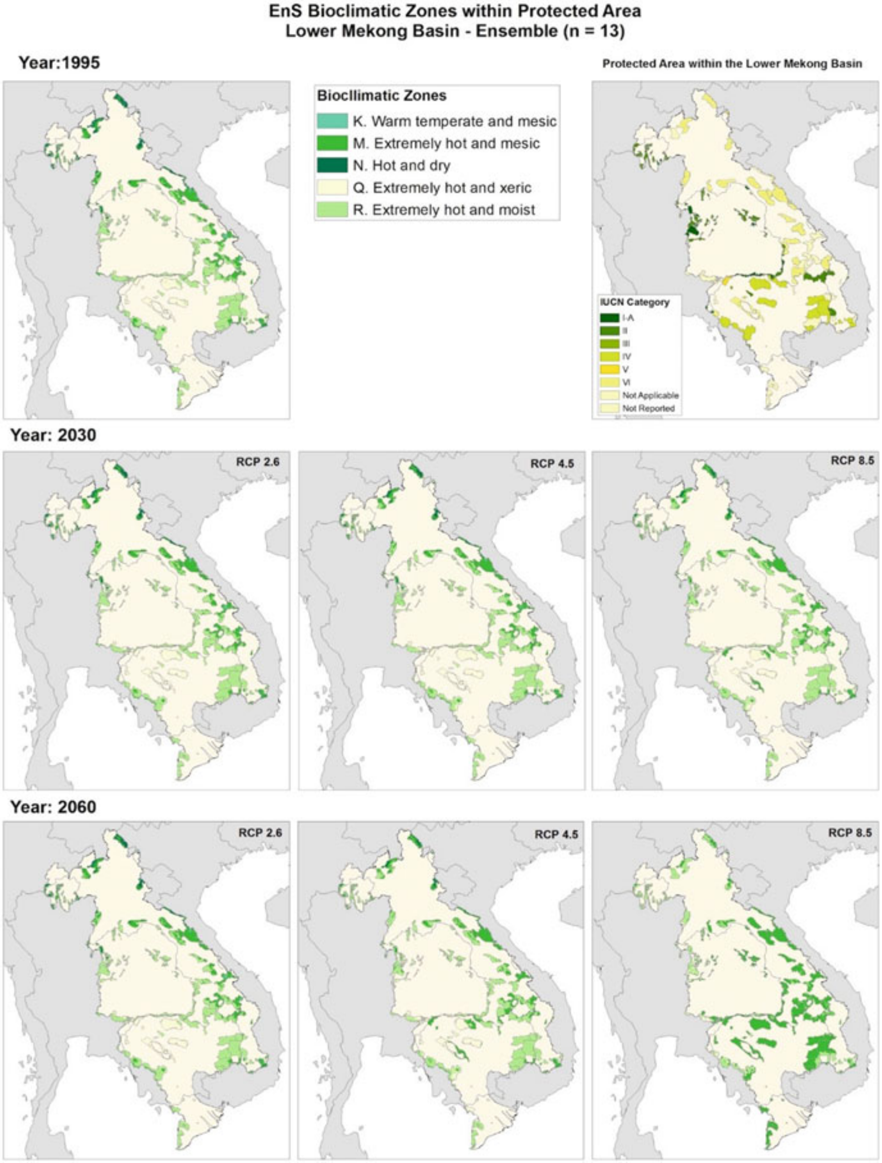


Fig. 3 Environmental Stratification (EnS) bioclimatic zones within protected areas in the Lower Mekong Basin, for baseline (1995), and 2030 and 2060 under emission scenarios RCP 2.6, 4.5, and 8.5

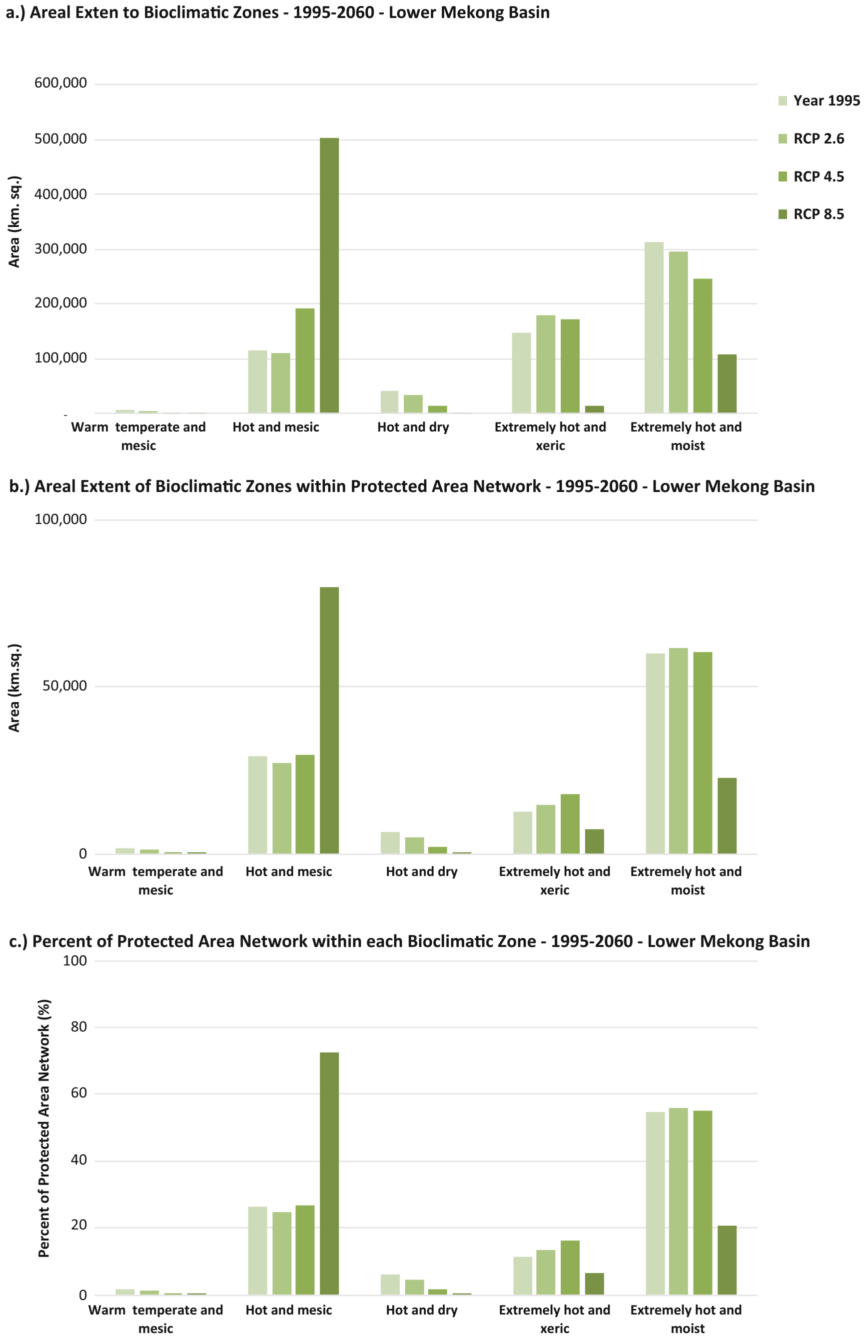


Fig. 4 The extent of bioclimatic zones in the Lower Mekong Basin (**a**) basinwide, (**b**) within protected areas, and (**c**) as percent of the total protected area in bioclimatic zone, from baseline (1995) to 2060 under three emission scenarios RCP 2.6, 4.5, and 8.5

Table 3 Projected change in the mean elevation above sea level (m asl) of bioclimatic zones in protected areas within the Lower Mekong Basin by 2030 and 2060, under three emission scenarios RCP 2.6, 4.5, and 8.5

Bioclimatic zone	Zone	Elevation				Elevation shift		
		1995	RCP 2.6	RCP 4.5	RCP 8.5	RCP 2.6	RCP 4.5	RCP 8.5
		m asl				m		
Projected change in mean elevation of bioclimatic zones and their upward shift by 2030—LMB								
Warm temperate and Mesic	K	1,535	1,601	1,666	1,823	66	131	288
Extremely hot and Mesic	M	721	783	824	564	62	103	**(157)**
Hot and Xeric	N	1,130	1,206	1,263	1,364	76	133	235
Extremely Hot and Xeric	Q	96	106	112	129	10	16	32
Extremely Hot and Moist	R	219	252	279	332	33	59	113
Projected change in mean elevation of bioclimatic zones and their upward shift by 2060—LMB								
Warm Temperate and Mesic	K	1,535	1,584	1,837	2,211	50	303	676
Extremely Hot and Mesic	M	721	769	496	268	48	**(225)**	**(453)**
Hot and Xeric	N	1,130	1,188	1,373	1,694	58	243	564
Extremely Hot and Xeric	Q	96	104	134	504	8	38	408
Extremely Hot and Moist	R	219	244	344	733	24	125	513

Numbers in bold indicate a negative shift

4 Discussion

As indicated by the roughly one-third (year 2030, RCP8.5) and two-thirds (2060, RCP8.5) of protected areas shifting to a new bioclimatic zone from baseline, climate change will have a substantial impact on protected areas in the LMB. Bioclimatic zones correspond to major vegetation assemblages (i.e., dominant and co-dominants) and translate broadly to landcover types such as coniferous, deciduous and evergreen forest. Zonal transitions are more profound than strata regarding ecosystem change, because a shift to a different bioclimatic zone indicates novel bioclimatic conditions with direct consequences for the biota and ecosystem function of that protected area. Changes in bioclimatic strata correspond to the level of vegetation species and can be investigated further if foundation species are of interest. Understanding the nature and direction of these changes provides crucial information for adaptation planning and management. In the near term (year 2030, RCP8.5) protected areas in central and northern LMB require attention to adaptation planning (i.e., average zonal shifts of 60% or greater). By 2060, under emissions scenario RCP8.5, protected areas all across the LMB are at risk. Only a few protected

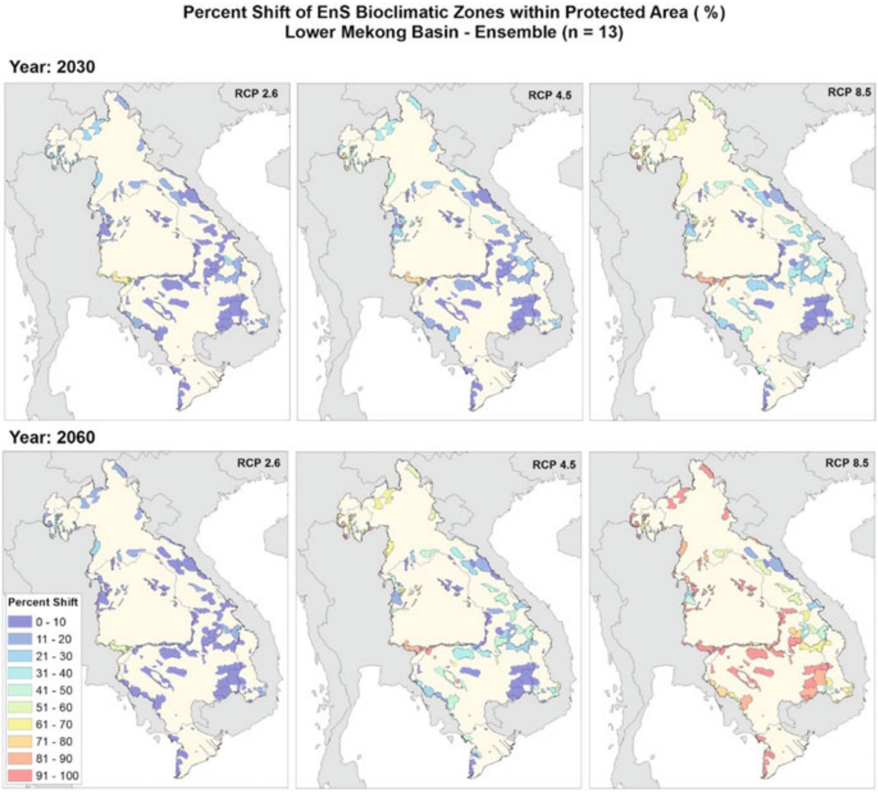


Fig. 5 Average percent of protected areas within the Lower Mekong Basin that shifted to a different bioclimatic zone from baseline (1995) as projected for 2030 and 2060 across three emission scenarios RCP 2.6, 4.5, 8.5

Table 4 Percent of total area for the Lower Mekong Basin (LMB) and protected areas that shifted to a different bioclimatic zone or strata by 2030 and 2060, under three emission scenarios RCP 2.6, 4.5, and 8.5

Percent of area shifting to a different bioclimatic zone or strata						
	2030			2060		
	RCP 2.6	RCP 4.5	RCP 8.5	RCP 2.6	RCP 4.5	RCP 8.5
Bioclimatic zones						
LMB-all	11.8	19.6	37.4	9.2	42.2	87.7
LMB-Protected area	9.3	16	29.1	7.1	31.7	77.6
Bioclimatic strata						
LMB-all	24.6	38.2	57.8	19.4	62.5	94.8
LMB-Protected area	32.1	50.8	68.2	24.9	70.4	90.2

areas are expected to experience minimal zonal shifts (i.e., average shift of 0–20%), and these areas are mainly located along the mountainous border of Lao PDR and Vietnam. Moreover, climate change is a landscape-level disturbance that is expected to substantially impact the effectiveness of the protected areas and conservation efforts across the LMB. Coordination will be needed across provincial and national borders to compensate for the projected bioclimatic shifts.

The need to synthesize existing data and knowledge of LMB ecosystems and their services is fundamental to EBM. A comprehensive, basin-wide characterization of the natural capital of the LMB is the foundation of an accurate accounting of the costs and benefits of sustainable development decisions to society: "To date there has been no comprehensive overview of the value of ecosystem benefits in the region, leading to a serious undervaluing by both politicians and even many local communities. A full review of Mekong ecosystem services is urgently overdue" (WWF 2013). Ideally, this should also include a comprehensive review of beneficiaries of final ecosystem goods and services (Ringold et al. 2010).

According to various international experts (Leadley et al. 2010), the greatest drivers of ecosystem and biodiversity change are "land use change, modification of river flow, freshwater pollution, and exploitation of marine resources" with "climate change and ocean acidification increasingly important drivers during the twenty first century." This suggests that the freshwater ecosystems, fisheries, and forests of the LMB are most threatened in the near term. Furthermore, "if greenhouse gas emissions continue along current trajectories, several Earth System models project that this will result in far greater climate-induced transformations of terrestrial biomes and marine biota than projected in earlier global biodiversity assessments." There is reason for optimism though, as the authors point out that "new socio-economic scenarios point to plausible development pathways of low greenhouse gas emissions and low land conversion that could lead to much lower biodiversity impacts." These scenarios of future development may be optimistic, but they are consistent with sustainable development goals and "require fundamental changes in development paradigms, but are coherent with known constraints on economics, resource use and human development goals" (Leadley et al. 2010).

There are challenges though to a complete characterization of LMB ecosystems. There is no single authoritative source, and there are gaps in the data which require further research for both freshwater and forest ecosystems, including habitat quality (WWF 2013). An exception is the ecological study of the Mekong upper floodplain and wetlands above Vientiane (IUCN 2013). This study is also exceptional for its companion socioeconomic study of community livelihoods along the river and surrounding forest that describe human uses of natural capital for both food and household income (Singer 2013). The Critical Ecosystem Partnership Fund provided a comprehensive review of hotspots of conservation potential in the larger Indo-Burma region that includes the LMB (CEPF 2011). Their strategic review addressed not only biodiversity but also the important social, legal, and political dimensions that must be considered when allocating resources for conservation of natural capital. However, the scope of their review is broad and provides little information for ecoregion characterization.

The World Wildlife Fund, with an ecosystem services emphasis, conducted a study of forest cover change in the Greater Mekong and compared their results to other data sources (WWF 2013). They provided maps contrasting two alternative scenarios that spanned a green energy, conservation-balanced future with business-as-usual exploitation of natural capital. They also addressed freshwater ecosystems, identifying 13 distinct but connected aquatic systems along the Mekong. The authors caution that information on ecosystem quality is particularly lacking. For example, remotely-sensed landcover data products typically do not discriminate between primary forest and disturbed, fragmented forest cover, although primary forest is known to be scarce in Lao PDR and Thailand (WWF 2013).

Overall, our understanding of social, economic and environmental system interactions and their dynamics is in its infancy, and experts argue in favor of a precautionary approach. The combined effects of time lags between drivers of ecosystem response and the measured indicators, thresholds of system change that result in nonlinear, accelerated response, and tipping points of irreversible ecosystem change, mean that "the impacts of global change on biodiversity are hard to predict, difficult to control once they begin, and slow and expensive to reverse once they have occurred" (Leadley et al. 2010). Fortunately, EBM and EbA provide useful, practical strategies for dealing with these uncertainties.

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