The Knowledge Status of Coastal and Marine Ecosystem Services - Challenges, Limitations and Lessons Learned From the Application of the Ecosystem Services Approach in Management

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The concept of ecosystem services (ES), first introduced in 1970’s, gained mainstream attention in 2005, when the Millennium Ecosystem Assessment formally proposed a definition for it. In spite of this attention, many aspects about the ES concept have remained controversial to date, i.e., their classification, value, generation, link to human well-being, and supportive role as management tool. This review explores the knowledge status of ecosystem services, focusing on those services generated in coastal and marine environments (CMES). A knowledge gap and an underdevelopment of tools to assess CMES is evident in the literature, especially when compared to the progress done in the assessment of land ES. Possible explanations reside on the yet small proportion that the research done on CMES represents for the ecosystem service framework (ESF), in part due to the intrinsic challenges of researching the marine environment, also due to the limited availability of spatial data on marine ecosystems. Nevertheless, the ES concept is getting more attention toward policy-makers and stakeholders, leading to the implementation of an ecosystem services approach (ESA) to the management and protection of CMES. Six lessons are rescued from the literature to improve the ESA: (1) integration of the ESA in a science-policy process; (2) more simplicity for the CMES prediction models; (3) move toward empowering of stakeholders; (4) integration of the value pluralism of CMES with less focus on money; (5) the link of ES to Human Well-being must not been forgotten; and (6) communication of results and social literacy are key.

Keywords: coastal and marine ecosystem services, ecosystem service approach, ecosystem services framework, human well-being, policy making
INTRODUCTION

Since the start of the twenty first century it has become clearer to scientists and decision-makers that nature is, and will be, affected by our activities (Costanza et al., 1997; Daily, 1997; Rockström et al., 2013; Rockström, 2015; Steffen et al., 2015). The issue is then how we preserve our limited resources while obtaining what we need from the ecosystems without destroying them in the process. Perhaps no other quote embodies this urgency better than Theodore Roosevelt’s quote during the Deep Waterway Convention (Roosevelt, 1907): “The conservation of natural resources is the fundamental problem. Unless we solve that problem, it will avail us little to solve all others.”

Paradoxically, the importance of many ecosystems is often recognized after they are lost, as was the case of the mangrove ecosystems, following Hurricane Katrina (Chambers et al., 2007); or the Lake Erie, which was declared as dead in the 70’s (Burns, 1985). At the end of the 1960’s major concern arose, mainly triggered by a series of high-profile ecological disasters, including the Torrey Canyon oil spill off the coast of England (Wells, 2017), and the burning of the Cuyahoga River (Adler, 2002), inspiring the green movement and ultimately the environmental rights revolution (Boyd, 2011), calling action from governments and politicians to protect nature.

Most of the argumentation to protect nature, in the early years of this environmental revolution, was based on ethics and the intrinsic value of nature (Soulé, 1985), which generated divided attention from the public and a fierce dichotomy between intrinsic-nature supporters and instrumental users of nature (Kloor, 2015). In the last 20 years, scientists have called attention toward a shift in the paradigms of conservation (Kareiva et al., 2011; Kareiva and Marvier, 2012), and the use of holistic approaches which do not choose between nature or development, but gather the best of both (Tallis et al., 2014).

In this regard, an emergent concept has grown since 1970 when a report by The Study of Critical Environmental Problems (SCEP) entitled ‘Man’s Impact on The Global Environment’ first introduced the term ecosystem service (SCEP, 1970). In the academic literature two publications ignited the interest on ecosystem services (ES), the book by Daily (1997) entitled “Nature’s Services. Societal Dependence on Natural Ecosystems”, and the article by Costanza et al. (1997) published in Nature: “The value of the world’s ecosystem services and natural capital”. However, it was only in 2005 that ES found a well-structured definition which mainstreamed the concept globally. The Millennium Ecosystem Assessment Report defined ES as the “benefits people obtain from ecosystems” (MEA, 2005:26).

Although the ES concept was initially criticized for namely price-tag nature, an important body of knowledge on ES has accumulated over the past twenty years, collaborating to the establishment of an ecosystem services framework (ESF). The ESF (Haines-Young and Potschin, 2010) is essentially a discursive tool which has reinvigorated the environmental agenda at the global level (Muradian, 2017), by focusing in two core topics: the classification of the ES and the evaluation of the connections that those services have to human well-being. In spite of this progress authors like Liquete et al. (2013); Patterson and Glavovic (2013); Costanza et al. (2017); Torres and Hanley (2017) have argued that coastal and marine ecosystems services (CMES) initially received less attention from the scientific community, thus, creating a progress gap and underdevelopment of assessment tools when compared to land ES.

Marine ecosystems, whether coastal or off-shore, benthic or pelagic, are unique environments that need to be protected mostly due to three reasons: they are vitally important, not only ecologically but also economically; they are being degraded by economic activity, habitat destruction, land conversion, pollution impacts, overfishing (Barbier, 2012); and they are the focus for policy and management interventions from global to local levels (Patterson and Glavovic, 2013). Even though for long time marine science and coastal management have tended to go separate ways, generating a different set of research questions, publishing in different journals, thus, establishing separate research communities (Ostrom, 2009). Recent trends have reverted this situation, shifting toward interdisciplinary collaboration and even transdisciplinarity for the study of marine ecosystems.

In this context, a novel management tool has emerged as an alternative to incorporate the ES concept in economic decision-making, gathering from both the ecosystem-based management approach and the ideas of the ESF. This tool is known as the ecosystem services approach (ESA). The ESA (Martin-Ortega et al., 2015) seeks to put into a social context the worth of natural ecosystems, aiming to better manage the complex socio-ecological systems in which the ES are generated and delivered. However, and even though ESA has been highly recommended by a group of scientists (Daily et al., 2009; Maes et al., 2012; Costanza et al., 2017), other authors (Egoh et al., 2007; Koschke et al., 2012; Laurans et al., 2013; Stocker, 2015) agree on the inappropriate or inconsistent application of the ESA for supporting management and policy, especially when used to manage CMES.

It is impressive although, that in less than 20 years since its mainstreaming by the MEA (2005), the ES concept has expanded and dominated most of the conservation agenda (Muradian, 2017), in part due to the decline of the previous dominant paradigm, but also due to its seemingly simple message: to pay attention to the importance of human reliance on nature. Braat and de Groot (2012) have called this a new paradigm in conservation, an ecosystem services paradigm (ESP), which follows, both in academia and policy, the same basic steps: first identify ES, second assess and estimate them, and third, capture and manage values to incorporate them into decision-making.

In spite of the widespread dominance of the ESP, many concepts related to ES remain highly controversial, e.g., the definition of ES itself (Boyd and Banzhaf, 2007; Fisher et al., 2008; Freeman et al., 2013), the use of ecosystem valuation in the actual management of marine ecosystems (Laurans et al., 2013; Torres and Hanley, 2017), the link between the delivery of ecosystem services and the generation of human well-being (Costanza et al., 2007; Raudsepp-Hearne et al., 2010; Fisher et al., 2013), and the valuation of non-market ecosystem services, such as cultural and social ones (Fish et al., 2016; Bullock et al., 2018). The first part of this review is focused precisely on the body of literature...
addressing all those controversial issues concerning the ESP, and starting with the search of consensus for what definitions are best fit to cope with ES regarding coastal and marine ecosystems.

Although the ES concept emerged as an out-of-the-box ecological idea, mostly a reactionary view to the traditional intrinsic-conservation movements, its merit has been linked to the interdisciplinarity of its origins, especially the collaboration with economics. Hence, it is vital to elaborate on the economic perspective of ES and the implications it brings to their study and management. The second part of this review focuses on the actual management of CMES. With a fast-growing body of literature on this topic, this section will be explored from a different perspective, drawing attention over what has been recognized as key lessons (Ruckelshaus et al., 2015; Beaumont et al., 2017; Drakou et al., 2017) to generate better knowledge that can be used on the implementation of policies and the process of decision-making regarding the marine environment.

WHAT IS AN ECOSYSTEM SERVICE?

The Issue of Definition

The Millennium Ecosystem Assessment (MEA, 2005) defines ES as the benefits people obtain from ecosystems. Under this definition, ES are goods, experiences, and products derived from ecosystems. Such definition has some intrinsic issues, the one of intermediate ecosystem services, or also known as supporting services (Potschin-Young et al., 2017), and the ES as natural products without human intervention (Boyd and Banzhaf, 2007). To put into perspective, consider the case of fishing. According to the MEA (2005), captured fish is considered an ecosystem service. However, Boyd and Banzhaf (2007) pointed out that for the fish to be caught and transported to land two main inputs are needed; the presence of a harvestable fish population and, the combination of fishing time, gear quality and fishermen skills. Therefore, captured fish is neither a pure product of the aquatic ecosystem nor it should be considered an ecosystem process necessary to maintain good water quality standards as well as a healthy aquatic food web in the sea water. Under closer inspection, however, the nutrient cycling does not provide a direct benefit to humans, it rather represents an intermediate step necessary to obtain usable clean water. Subsequently, a beach requires the combination of both ecological services, such as nutrient cycling, and human input, i.e., roads and other kind of infrastructure, which turns it into a leisure area. Therefore, we end up with two conflicting views, according to Polasky and Segerson (2009) both the beach and the nutrient cycling should be considered ES, while for Boyd and Banzhaf (2007) they should not, because one requires additional human input to provide the ES attached to it, and the other is an intermediate step necessary to have a proper and healthy beach, but which does not provide a direct benefit to people.

Finding a Middle Ground

Costanza et al. (2017) have acknowledged the prolonged debate caused by contrasting points of view about what an ecosystem service is or should be. Although it might seem trivial, it is important to remark that the problems arising from an unclear definition could subsequently built up on bigger issues such as misuse of indicators, inadequate measurement units, double counting of services during valuation, and weak incorporation of the framework and methodology into an operational decision-support system (Nahlik et al., 2012; Austen et al., 2019). Lack of clarity on the ES definition, also makes more difficult the communication of its relevance to the public (Saarikoski et al., 2015). Therefore, a clarification becomes necessary. In this regard, the definition provided by Fisher et al. (2008), which is largely based on the pioneer review by Boyd and Banzhaf (2007), provides a stronger ground about what ES are, “the end products of ecosystems utilized actively or passively to produce human well-being”.

Under this definition, four main characteristics can be attributed to any ecosystem service (Guerry et al., 2012; Freeman et al., 2013). Firstly, they provide a benefit to people. ES are entirely linked to their contribution to human well-being, they cannot be defined independently and the benefits obtained by people could either be direct or indirect, conspicuous or unconscious, ergo their consideration as end products of nature (Braat, 2013). Secondly, ES are biophysical components of the ecosystem, but they are not ecological processes or functions. The latter can contribute to ES; however, they are not synonymous. To exemplify this, we can think about carbon sequestration in water bodies. For this ecosystem service to be achieved, it requires the conjunction of many processes and ecosystem functions to work properly, i.e., filtration of suspended matter, primary production, microbial transformation among others (Armoskaitė et al., 2020), which will remove carbon from the water to be incorporated into biomass. But these functions are not exclusive of carbon sequestration. In fact, they can also collaborate to the
generation of other ES such as macroalgae material, wild fish, or even good water status for enjoyment of spiritual experience (Teixeira et al., 2019).

Thirdly, they can, and should, be measured as discrete units rather than as rates. To understand this we can take again the previous example. On one hand we have an ecosystem function such as primary production, which is a continuous process that changes rapidly on time. Thus, it is measured as a rate, usually of carbon, against long time periods (years), or short time periods, i.e., days (Kaiser et al., 2005; Sigman and Hain, 2012). On the other hand, carbon sequestration aims to measure fixed concentrations of carbon which are incorporated into living organisms such as macroalgae, phytoplankton biomass, seagrass, and others, whose values have low short-term variation. Lastly, they are purely natural components without any combination with human production. An example is fishing, which under this definition is mostly a benefit obtained by the combination of an ecosystem service (wild fish population of any targeted species) with common goods, such as a net, boat, fishing time (Boyd and Banzhaf, 2007).

The rationale with this definition is to make ES visible, comparable, and accountable for economic production, in a way that their value is not masked and forgotten in the process of accounting. Therefore, the focus on differentiating ES from intermediate services or ecosystem functions, as well as from the immediate analogy to benefit (SEEA, 2020), is of high relevance and demands explicit clarification when assessing ES. Nevertheless, under certain circumstances the use of the term ES could refer broadly to intermediate and final services, as long as the valuation of the ES accounts for this (U.S. Environmental Protection Agency [U.S. EPA], 2009:12–13).

This flexibility in terminology could collaborate to avoid a potentially harmful polarization within the ES user’s community (Nahlik et al., 2012; Muhar et al., 2017), at the time it promotes the institutionalization of the ES concept into decision and policy systems (Steger et al., 2018). Hence, a call for a ‘guiding pluralism’ (Hermelingmeier and Nicholas, 2017) that embraces this range of perspectives still existing around the ES definition, is suggested as possible way to address the issue as long as those perspectives involve the integration of the concept of sustainability as a guiding principle (Norgaard, 2010; Ainscough et al., 2019).

What is intended by introducing the problematic definition of ES in the beginning of this review, is to raise awareness about the ongoing and fast evolving development of the ES concept. This is a discipline with just two decades of formal scientific structuring and working (Gómez-Baggethun et al., 2010; Costanza et al., 2017), which has had as many critics (e.g., Thompson and Barton, 1994; Toman, 1998; McCauley, 2006; Simpson, 2016) as supporters (e.g., Costanza et al., 1997; Freeman, 2003; Armsworth et al., 2007; Pearce, 2007). Perhaps the words of Costanza et al. (2017) explain better what the ES concept seeks to explain: “If anything, the ecosystem services concept is a ‘whole system aware’ view of humans embedded in society and embedded in the rest of nature.” As it was also identified by the Antwerp declaration, the major purposes of the ES concept must be to serve as an awareness raising tool, scientific approach, and decision-making aid (Ainscough et al., 2019).

### Defining Coastal and Marine Ecosystems Services

According to the MEA (2003), the marine ecosystems encompass those marine areas deeper than 50 meters, while coastal ecosystems are areas located between 50 meters below mean sea level and 50 meters above the high tide level. Coastal ecosystems could extend over the continental shelf and inland up to 100 km from the shoreline. These categories do not represent single ecosystems. In fact, they are formed by a variety of ecosystems. However, they share a group of biological, climatic, social, and cultural factors whose consideration is useful for analyzing changes in their capacity to provide ES and the expected consequences for human well-being.

Torres and Hanley (2017) proposed a further division of the two main categories of the MEA (2003). Based on a mixed approach between management applicability and the Water Framework Directive classification of aquatic ecosystems (WFD, 2000/60/EC), they identified 8 broad ecosystem types, as shown in Table 1. The table also depicts the specific ecosystems whose services are object of valuation within each ecosystem type, and the management area each type could contribute to. Most of these ecosystem types are not mutually exclusive.

It should be stated, nonetheless, that such classification does not seek to create borders or segregation during the assessment or valuation of these ecosystems. On the contrary, it looks for an integrative approach while researching on ecosystems types which form part of a continuous seascape (Barbier, 2012; Torres and Hanley, 2017), allowing at the same time

| Broad ecosystem types | Specific ecosystems | Management areas |
|-----------------------|---------------------|-----------------|
| Coastal Ecosystems    | Wetlands, mangroves, marshes and swamps | Wetland management |
|                       | Beaches             | Beach management |
| Coastal areas         | Coastal protected natural areas, capes, peninsulas and barrier islands | Coastal area management |
| Inland and transitional waters | Rivers, streams, canals, lakes, reservoirs, deltas, estuaries and catchments | River basin management |
| Marine Ecosystems     | Bays, gulfs, sounds, fjords, inland seas and sea waters near the coast | Coastal water management |
| Coral reefs           | Coastal coral reefs | Coral reef management |
| Deep sea              | Deep sea, open ocean (including cold-water corals) | Deep-sea waters protection |
| Marine protected areas | Marine conservation zones, marine parks, marine reserves, marine sanctuaries and marine critical habitat units | Marine protected area policy design |

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**TABLE 1** | Coastal and marine ecosystems types and management areas. Modified after Torres and Hanley (2017).
to assess the trade-offs and synergies which would emerge during the decision-making process (Howe et al., 2014). In fact, key functional linkages arising from the connectivity across an entire seacape of marine habitats influence the provisioning of ecosystem goods and services (Pittman et al., 2011).

Take for example the mangrove-seagrass-coral reef seascape; all three habitats interact to provide support for marine fisheries, sediment control, and storm protection (Barbier, 2016b; Gillis et al., 2017). The connectivity among habitats mutually reinforces the support of coastal and marine fisheries because adult fish use the coastal habitats for breeding or as nurseries, with the resulting cohort migrating to coral reefs and open waters where they become part of a fishing stock. On the other hand, the sheltering effect of reefs may enhance the ability of seagrass beds, marshes, mangroves, and other coastal habitats to attenuate waves and buffer winds; whereas sediment control by mangroves and seagrass beds may also protect coral reefs, thus, enhancing their goods and services (Barbier, 2017).

**COASTAL AND MARINE ECOSYSTEMS SERVICES UNDER THE ESF**

If we look at the current state of the ESF, a bold but realistic statement can be drawn. Much of the work done has been preponderant for terrestrial ecosystems (Costanza, 1999; Beaumont et al., 2007; Chan and Ruckelshaus, 2010; Guerry et al., 2012; Maes et al., 2012; Patterson and Glavovic, 2013; Townsend et al., 2018). This focus is counterintuitive, considering that Costanza et al. (1997), in the first global assessment of the value of the world ecosystem services, estimated the contribution of marine ecosystems in about 63% (US$ 20.9 trillion/year) of the total US$ 33 trillion/year worth in services provided by global ecosystems. An unprecedented value which, despite controversies about its accuracy (Costanza et al., 2014; Pendleton et al., 2016), has highlighted for the first time the economic importance of marine and coastal ecosystems for modern societies.

In spite of their value, coastal and marine ecosystems are among the most threatened and degraded ecosystems in the world (Martínez et al., 2007; Barbier, 2017), mainly due to their alteration as consequence of increasing human settlement in coastal areas, which has been reported to reach approximately a third of the world’s human population (Small and Nicholls, 2003). A surprising fact if we consider the relatively small area that coastal zones represent for earth’s land (4% of total surface). As society’s demand for coastal and marine environments increased, so did the pressure over these ecosystems caused by human activities such as fisheries, accounting for around 90% of catches only in coastal areas (Worm et al., 2009), aquaculture, shipping, recreation (MEA, 2005; Halpern et al., 2008; Diop et al., 2016; Visbeck, 2018), generating unseen levels of degradation. Moreover, global stressors, such as global warming, are expected to cause further changes and losses of mangrove forests, coral reefs, wetlands (IPCC, 2019), with the subsequent loss of those CMES delivered to society (Cooley et al., 2009; Doney et al., 2012).

Yet, the amount of data and the feasibility of methods to assess the provision of CMES are much more limited for marine ecosystems when compared to the terrestrial ones (Costanza, 1999; Barbier, 2012; Townsend et al., 2018). It is necessary to ask, why in spite of the extreme value of coastal and marine ecosystems (Martínez et al., 2007; Barbier, 2012; Barbier, 2017; Gaylard et al., 2020), their contribution to human well-being through the delivery of CMES (Costanza et al., 1997; Pérez-Maqueo et al., 2007; Van Der Meulen et al., 2008; O’Garra, 2009), and the acknowledgment of human increasing pressure and perturbation over marine ecosystems (Diop and Scheren, 2016), there is still a significant gap in the development of tools and methods to better evaluate CMES (Barbier et al., 2011; Patterson and Glavovic, 2013; Stocker, 2015; Lau et al., 2019).

**Why Did a Knowledge Gap Develop?**

A first possible reason would lead to consider that the inherent challenges of studying the marine environment have imposed certain limitations to scientists (Barbier et al., 2011; Barbier, 2012; TEEB, 2012). As consequence, CMES in general have not received enough attention. Certainly, this was true between 1997 and 2006, when an average of 2.5 papers per year on CMES were published (Liquete et al., 2013). The release of MEA (2005) brought the topic to the academic mainstream, changing for good the attention of many scientists and funding agencies toward the topic (Gómez-Baggethun et al., 2010). Liquete et al. (2013) found that the average rate of publications related to CMES increased to 23 papers published per year after 2006. However, the ES concept continues to be driven forward mainly by research on terrestrial ecosystems (TEEB, 2012; Milon and Alvarez, 2019). Apparently, within the ES scientific community, researching on CMES is a relatively new theme (Liquete et al., 2013). Costanza and Kubiszewski (2012) found a total of 2400 papers listed in the ISI Web of Science journals related to the subject of ES, but Liquete et al. (2013) detected only 279 papers specifically addressing CMES in the same searching engine. According to Townsend et al. (2018), studies on marine ecosystems represent only 9% of the ES literature accumulate over time.

Yet, united to the lack of attention, a bias focus toward certain coastal and marine ecosystems (Barbier et al., 2011) could have accentuated the delayed development of CMES assessment tools. In their review of 145 selected papers, Liquete et al. (2013) found that the most frequently analyzed study area was the coastal zone (43%) or the coastal and marine area together (28%). On the contrary, the open sea was the focus on around 18% of the articles. In fact, three coastal ecosystems - coral reefs, salt marshes, and mangroves - emerged as the best researched ecosystems in terms of ES assessment (Yeo, 2002; Barbier, 2007; Costanza et al., 2008; Chen et al., 2009; Davy et al., 2009; Hicks, 2011; Zhang and Smith, 2011). Other marine ecosystems, like the deep sea (Thurber et al., 2014) or the open ocean (Stocker, 2015), are recognized highly important to humanity, but hold few or none specific valuations of their ES (Jobstvogt et al., 2014; Culhane et al., 2018). Another explanation points toward the marked differences between terrestrial and marine ecosystems in terms of their
physical environments, ecological processes, and contemporary patterns of human impacts (Carr et al., 2003). Key characteristics of ocean ecosystems such as openness (Denny, 1993), decoupling of local off-spring (Carr et al., 2003), connectivity of the seascape (Townsend et al., 2018), and three-dimensionally of pelagic habitats (Carr et al., 2003; O’Higgins et al., 2019) have imposed restrictions to the applicability of valuation methods (Mendelsohn and Olmstead, 2009) and assessment techniques (Polasky and Segerson, 2009), originally developed to assess land ES, when applied to coastal and marine ecosystems (Maes et al., 2013; Eghol et al., 2012). Consequently, underestimation in valuation estimates (Costanza, 1999; Barbier, 2012; Costanza et al., 2014; Simpson, 2016), oversimplification of the production of services (Guerry et al., 2012; O’Higgins et al., 2019), and lack of integration of value-diversity for governance schemes (Townsend et al., 2018) are common problems when assessing CMES.

Additional complications have raised due to the scarcity of spatial data for marine ecosystems (Guerry et al., 2012; Townsend et al., 2014; Nahuelhual et al., 2020), which has limited the elaboration of CMES provision and demand maps. Although scarcity of spatial data is a usual issue in natural sciences, terrestrial ecosystems have largely benefited from the development of remote sensing technology, as opposed to marine ecosystems. With the exception of seagrass, kelp forest, and coral reefs to depth of 40 m (Chauvaud et al., 1998; Poursanidis et al., 2018), most coastal systems, especially benthic habitats, are unreachable for most satellites or have accuracy constrains due to high turbid coastal waters (Townsend et al., 2018). But even for those ecosystems in which maps were developed, the focus has centered on mapping single ES (Söderqvist et al., 2005; Barbier et al., 2008; Guerry et al., 2012). Lioy et al. (2013), found in their review that at least half of the articles (48%) studied just one service, whereas only (13%) assessed six or more services, thus, simplifying the production and delivery of CMES which most of the time are generated by different ecosystems and at different spatial scales (Raffaei and White, 2013).

Lastly, there has been an excessive focus on local assessment of CMES (Lioy et al., 2013; Townsend et al., 2018). In their review, Lioy et al. (2013) found that 48% case studies were focused at a local scale, with only 8% reaching supranational, 4% continental, and 9% a global scale. Most of these case studies have been carried out by researchers in the global north (Costanza and Kubiszewski, 2012; Schaafsma and Turner, 2015), with the global south misrepresented or absent, as it can be noted in regions such as central and south Africa or West Asia (Figure 1). This has led to many authors underestimate the context specificity of many ES (Needham et al., 2011; Chamberlain et al., 2014; Townsend et al., 2018), and the risks of over or under valuation when applying benefit transfer between regions (Hicks, 2011; Van den Belt and Cole, 2014) or to larger scales (McGlashan and Firn, 2003; Lozoya et al., 2011). Although for some CMES, such as coastal protection or nursery ground, the assessment of highly localized patches of the ecosystem might be enough to assign proper value (Sandilyan and Kathiresan, 2012; Narayan et al., 2016), global services such as climate regulation, have a scale of assessment at the level of square kilometers, or larger, before the area becomes meaningful to service delivery (Bouillon et al., 2008; Lovelock, 2008).

THE ECONOMICS BEHIND ECOSYSTEM SERVICES

Barrett (2003), in his book 'Environment and Statecraft: The Strategy of Environmental Treaty-Making', provided information on more than three hundred international conventions which related to the environment. This much international cooperation seems to show the concern of the worldwide community with the possible collapse of ecosystems, loss of biodiversity, and increasing extinction rate of species, among other ecological issues (Pearce, 2007). At the same time, there seems to be an apparent mismatch between the people's willingness to pay for ecosystem conservation (Woodward and Wui, 2001; Horton et al., 2003), running into a magnitude of trillions of dollars, and the actual expenditures on ecosystem conservation, at best ranked at a few billions of dollars. In this regard, Pearce (2003) has argued that there is a global deficit of care to preserve both biodiversity and ecosystem services, with Deutz et al. (2020) estimating such deficit on biodiversity financing as a gap of between 598 and 824 billion US$ per year.

But how well do expenditures for conservation efforts represent the measurement of care for the ES? If the money flowing to conservation policies is the main measurement of success, there are many pitfalls to be aware of. For example, secondary effects might be difficult to quantify after a policy measure has been set, due to amplification of the cash flow or lack of follow-up by the responsible agencies. Also, investing into one policy does not necessarily make it a good policy. Bad policies are tremendously abundant and could outweigh the effects of good ones (Pearce, 2007). Considering that research in ES has grown steadily since 2006 (Fisher et al., 2008; Gómez-Baggett et al., 2010; Lioy et al., 2013; Ruckelshaus et al., 2015). If ES are to be understood and effectively used, they need to be framed into a language policymakers can understand (Beaumont et al., 2007). Needless to say, ES need to be framed into economics.

Ecologists have long been reluctant to approach ecosystems and their processes with terms and methods traditionally used by economists (Beder, 2011; Baveye et al., 2013). Nonetheless, new frameworks, such as the cascade model (Haines-Young and Potschin, 2010), have received increasing recognition due to its flexibility to represent the flow of ES between the natural environment and the social, human-made world (Jacobs et al., 2016). Box 1 provides a further explanation about the model’s core idea and applicability to the ESF.

An alternative, and possibly better exploration of the economic details of the ESP is offered by the ecological production theory (Boyd and Krupnick, 2013). The ecological production theory in a nutshell is the adaptation of the economic production theory to the analysis of natural systems. Under this view, ecosystems are formed by commodities that interconnect
with each other through a variety of physical-chemical processes. The major challenge that Boyd and Krupnick (2013) faced by using this approach was to keep the consistency with ecological sciences, while presenting meaningful insights to their social science counterparts. A struggle which Freeman et al. (2013) conceptualized best into what they called functions. To them, functions do not strictly connect to their mathematical meaning, but rather represent a transformation, a link between the natural commodities (Boyd and Krupnick, 2013), the ecosystem services (Fisher et al., 2008), and the changes in human welfare (Haines-Young and Potschin, 2018).

The Ecological Production Function (EPF)

All the ES are the result of a bundle of processes, reactions which transfer energy between the ecosystem components, all of them occurring naturally (Austen et al., 2011). Take for example the provision of seafood. To obtain a certain amount of seafood, e.g., fish, crabs, seaweed, it was necessary to combine primary and secondary production, nutrient cycling, predation, and transfer of energy from lower levels of the food web, ultimately reaching the desire level from where the extraction, ergo seafood, was obtained. Thus, the EPF refers to all those processes that occur in nature which enable the generation of ES (Freeman et al., 2013).

In this scenario, the EPF copes with complex, invisible, and sometimes still poorly understood processes (Costanza et al., 2017) which, nonetheless, can be avoided. Understanding the variation in ES production is mostly done by monitoring ecological indicators, or bioindicators. According to Freeman et al. (2013), an indicator is a proxy for complex phenomena and can be used to reflect on the provision of a service and how it changes over time. Finding the right indicator is key for defining the EPF. Costanza and Daly (1992) and Freeman et al. (2013) have stated that a possible good way to find the right indicators is through the concept of ecosystem service providers and the ecosystem service providers efficiency (Kremen, 2005).

An example in which these concepts can be explored is by analyzing the ecosystem service of coastal erosion protection. In coastal settlements, the presence of certain habitat formations (seagrass, mangrove forest, dunes, saltmarshes) provide protection to the coastline from wave and tide action. Therefore, the extension, quality, and resilience of those structures is what we can consider an ecosystem service provider. An example frequently mentioned in literature is that of mangrove forests in tropical regions (MEA, 2005; Fisher and Turner, 2008; Liquete et al., 2013; Barbier, 2016a; Narayan et al., 2016). Here, the use of bioindicators can extend from measuring the forest area, to the average extension of the mangrove forest over critical areas, the percentage of managed areas of the forest, and the rate of deforestation and logging, among others (Valiela et al., 2001).

However, sometimes, the selection of indicators cannot be as straightforward as with this example. Many CMES are global and non-proximal (Costanza, 2008), meaning that location does not matter nor does the proximity of the location to the human beneficiaries. Likewise, many species are highly mobile, therefore,
giving different relevance to various locations at different times of the year or at a particular stage in an organism’s life cycle, all affecting the provision of ES (Carr et al., 2003; Hicks, 2011). Feld et al. (2009) found that most indicators are measured at regional or local scales, with functional indicators and indicators reflecting temporal differences rarely measured, irrespective of scale. Indicators need to consider both the spatial and temporal scales at which their measurement can be effective, but such task remains yet to be done (Olander et al., 2018) and likely to be context specific (Hummel et al., 2019).

Although many researchers (Barbier et al., 2011; Hattam et al., 2015; Lillebø et al., 2016) have tried to identify a series of indicators which better work for some CMES, practical guidelines for selecting indicators are still missing (UNEP-WCMC, 2011; Van Oudenhoven et al., 2012), especially for the coastal and marine environment. Nevertheless, some researchers have proposed a series of criteria which can be used to ensure the feasibility of the selection of indicators and their utility to describe the EPF. These approaches vary between three basic criteria (Link et al., 2010): measurability (“Are data available?”); sensitivity (“Can they detect change?”); and specificity (“Is the change in the indicator a response to the pressure of interest as opposed to natural variability?”). Among more exhaustive approaches, Dale and Beyeler (2001) proposed five alternative criteria, that indicators respond to stress in a predictable manner; are anticipatory (i.e., signify impending change); predict change that can be averted by management; are integrative (i.e., can indicate change over key gradients across an ecological system); and have low variability in response.

The Economic Demand Function (EDF)

The ecosystem which provides a service is usually named ‘natural capital’ (Costanza and Daly, 1992). This definition conveys a parallel between ecosystems and the monetary-economic capital, mainly in two points: ecosystems produce a flow of services which are attained to temporal variability and they usually interact with other kinds of capital (social, human, or built capital) to generate a benefit to human well-being (Costanza et al., 2017). The connection between ES and the benefit we humans obtained from them represents the economic demand function (EDF). In simple words, it is the realized enjoyment of an ecosystem service (Hattam et al., 2015).

This human enjoyment placed into economic terms means welfare. A common mistake here is to assume that human welfare is referred exclusively to monetary welfare, like dollars in a bank account. Money is certainly the most common metrics to measure welfare, but welfare can rather be approached as a wide range of metrics when using the EDF, such as satisfaction, number of people benefited, among others. Contrary to the EPF, the processes which generate an EDF are often discernible, closer to society, but perhaps not less challenging to represent. The challenge resides now in how to best represent the variation of human welfare according to the ES dynamics.
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The cascade model is formed by five elements, shown in Figure 2, which are constituents within the natural system (environment) and the social and economic system. In the natural system, supporting or intermediate services—which are formed by biophysical structures or processes and functions—give rise to the final service, whereas this final ecosystem service provides a good and/or benefit to humans, which is ultimately valued according to context, culture, or economic reasons (TEEB, 2010, 2012). Box 2 for further explanations about the meaning of value in ES. This model suggests a tight relationship between people and nature embodied in the ES concept. It argues the need to identify both the functional characteristics of ecosystems that give rise to services, and the benefits and values that they support (Potschin and Haines-Young, 2011, 2016) to fully grasp their production-delivery relationship.

Nevertheless, many publications (Potschin and Haines-Young, 2011, 2016; Potschin-Young et al., 2017) have expressed that the intention with the cascade idea is to highlight the essential elements that have to be considered in any ecosystem service assessment, and the kinds of relationships that exist between them. The assertion is that no individual component should be locked in isolation, but rather from an inter- and even transdisciplinary perspective.

In this regard, it is necessary to argue about the convenience of using the cascade as a model (Prousevell et al., 2010) or rather a flexible conceptual framework (Potschin-Young et al., 2017). The Intergovernmental Platform of Biodiversity and Ecosystem Services – IPBES (UNEPI, 2014; Díaz et al., 2015) has highlighted the advantages of frameworks as tools which simplify thinking, structure work, clarify issues, and provide a common reference point. Moreover, the EU-funded OpenNESS Project found that the cascade model has been used as an organizing framework (Pagella and Sinclair, 2014; Tolvanen et al., 2016), a tool for reframing perspectives (Spangenberg et al., 2014a; b; Brink et al., 2016), an analytical template (Boulton et al., 2016; Guisado-Pintado et al., 2016), and as an application framework (Chapman, 2014; Daw et al., 2016; Gissi et al., 2016).

In spite of controversies or preferences, the cascade model has helped answer important questions about the ESF, including whether there are critical points: ecological, socio-cultural, and economic ones. De Groot et al. (2010b) found this distinction necessary due to the multiple levels at which benefits from ecosystems satisfy the needs of different groups of people, hence, the subjective value they acquire. They used the example of fishermen and their income in different cultures to elaborate in this idea. Artisanal fishing is not as profitable now as it used to be decades ago (Belhabib et al., 2016; Barange et al., 2018); however, for some cultures, it still represents a source of identity, a bond with nature. For the new generation of fishermen, the value of fishing could reside more in the social aspect rather than in the economic profit (Chan et al., 2016).

The first category, ecological values are crucial for human survival since they play key roles in the maintenance of essential life-support processes (MEA, 2005). Among them we find functional integrity, health, and resilience of an ecosystem to sustain life (De Groot et al., 2010a). These are critical values for keeping the provision of ES (TEEB, 2010) and to guarantee the sustainability of ecosystems (Wood et al., 2018). Bronnião et al. (2010) described the second category, socio-cultural values, as culturally shaped and derive from three value domains: ‘intrinsic value,’ which is the value of an ecosystem on its own, regardless of what people could obtain from it (O’Neill, 2002); ‘instrumental value’, the contribution to the beneficiary’s wellbeing (Kenter et al., 2015); and lastly, ‘relational values’, meaning the concerns associated with relationships and responsibilities between people or between nature and people (Chan et al., 2016). Economic values, the third category, express the importance of ES in monetary terms. This monetary value encompasses both, use value (direct or indirect) and non-use value. The accuracy with which these economic values can be measured varies and it seems to be method-dependent (De Groot et al., 2010a).

The IPBES has also developed a conceptual framework which identifies three inclusive elements in the interaction between human societies and the non-human world, among others: (1) nature; (2) nature’s benefits to people; and (3) a good quality of life (Díaz et al., 2015). IPBES assumes a value pluralism perspective by acknowledging the diversity of worldviews and values, thus, leading stakeholders to a different iterative approach regarding the identification of policy objectives and instruments. The IPBES approach emphasizes that values are fluid and sometimes cannot be placed rigidly into one category (e.g., instrumental or relational), illustrated by the color gradient of Figure 4. As Arias-Arévalo et al. (2017, 2018) concluded, the narrative is an important element which is sometimes overlooked when approaching the valuation process and usually supports the following three value typologies: living for nature (intrinsic in the sense of the direct moral consideration of non-human subjects of a life), gaining from nature (instrumental), and living in nature (relational).
In this regard, authors like Boyd and Banzhaf (2007); Pearce (2007); Fisher et al. (2008); or Boyd and Krupnick (2013) have offered perspectives about how to represent this dynamic. In the simplest way, an EDF can be visualized as a typical demand-supply curve (Pearce, 2007). Imagine we have a two-axis plot, in which the x-axis represents the ES supply level, while the y-axis represents the valuation level given by humans. Thus, as it happens in economics, the variation in the y-axis is bounded to the valuation given by humans if obtaining an extra unit of the ES represented in x-axis. The key word to understand here is value (in Box 2 an explanation is given about the complexity and subjectivity this word has for the ESP); therefore, for the EDF only the economic values hold a relevance, whether those are use values (direct and indirect) or non-use values.

In Figure 3, a conceptual analysis based on Pearce (2007) and Fisher et al. (2008) is explored. These authors recognized both the different EDF curves when assessing use-value (marketed ES) and non-use values (non-marketed ES). ES associated with markets in which formal exchange takes places have a negative slope curve ($D_{ES(M)}$). High provision of an ES is valued less since there is not major lack or unsatisfied demand, whereas as the ES becomes scarce, moving left in the x-axis, the valuation cost steeps up in the y-axis. We can easily imagine this scenario when thinking about fish, fuel, food, wood which are sold in a market with price tags attached.

On the other hand, ES which have been for long considered abundant or impossible to trade in conventional markets skip the first curve. Therefore, a second curve which merge both marketed and non-marketed ES ($D_{ES(MNM)}$) is established. Although this curve presents a downward trend too, it bends sharply when the provision of ES is low. Moreover, the $D_{ES(MNM)}$ curve lies somewhere above the $D_{ES(M)}$ (Pearce, 2007). Farley (2012) argues that the reasons for the differences between the $D_{ES(M)}$ and the $D_{ES(MNM)}$ are the apparent abundance of non-marketed ES in human history, which led them to be outside of economic analysis and ultimately to become externalities in production chains; as well as the impact that a shortcut provision, meaning a sharp and sudden scarcity of ES, would have over the willingness to pay of humans.

As for the supply of ES, a positive rising curve is defined. This curve could be understood as the supply curve of the EDF, but is called here the marginal cost of ES ($MC_{ES}$). Pearce (2007) understood this curve as comprised by the cost of managing ES, or, in other words, of conserving them, plus the opportunity cost, i.e., the cost forgone due to conserving a given ecosystem service instead of using it now. The $MC_{ES}$ fulfills the expectation of the increasing cost which brings the provision of more ecosystem service (Small et al., 2017).

**Implications of an Economic Framework for CMES**

In Figure 3, there are three details that need further consideration. Firstly, the interception of $D_{ES(MNM)}$ and $D_{ES(M)}$, with the $MC_{ES}$ originates two points known as $ES_{MIN}$ and $ES_{OPT}$. While the $ES_{MIN}$ depicts the point where only marketed services are provided, the $ES_{OPT}$ considers also those services which are not traded in markets. Although it might seem complicated at first, the idea is very insightful. Consider the case of fisheries stock assessment. As we saw previously, the ecosystem service provided here would be limited to the part of the fish population with the preferred size at the right time of the year. However, if what is solely considered in the determination of the fish stock is the fish needed to satisfy human demands, fish as food (i.e., $ES_{MIN}$), we are overlooking the role that same fish is playing as part of the ecosystem either as food for other species, as a key species in the ecosystem or as intermediate link in the production of another ES ($ES_{OPT}$). Therefore, the risk of considering only the $D_{ES(M)}$ is to generate an under-provision of ES, when $ES_{MIN} < ES_{OPT}$ (Fisher et al., 2008).

Secondly, ecologists have predicted that perturbations in the functioning of ecosystems would limit their capacity to provision services (Chapin et al., 2000; Loreau et al., 2001; Villamagna et al., 2013; Villasante et al., 2016). Hence, it is necessary to consider a safe minimum level in which the ecosystem’s components (including biodiversity, populations, and interactions) are enough to maintain the functions that give rise to the various CMES. This level is represented in Figure 3 as the Safe Minimum Standard (SMS). The issue relies on where exactly this line should be drawn, as the uncertainty is currently great due to the nature of each ES and the state of the ecosystem under assessment (Dobson et al., 2006).

Thirdly, in spite the validity of $D_{ES(MNM)}$ to capture the real benefits of ES provision, its application is limited under current context because our inability to capture the value of non-marketed ES (Fisher et al., 2008). In other words, in strictly economic terms, those services which are not traded in markets or lack at all of one, cannot be attribute a monetary value.

**FIGURE 3** | The graphic representation of the economic demand function (EDF). The $D_{ES(MNM)}$ curve represents the variation of ES traditionally traded in markets, thus, with use value, whereas the $D_{ES(MMM)}$ curve represents all those services which hold a non-use value. At some level of degradation, both curves behave differently, but both run into a problem of uncertainty and possible collapse. Reproduced from Fisher et al. (2008).
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FIGURE 4 | Diverse values related to nature, nature’s contributions to people and a good quality of life. The grading in the colors indicates that both instrumental and relational values can be ascribed to the value of nature’s contributions to people, and to highlight that it is intertwined with nature and a good quality of life. Reproduced from Pascual et al. (2017); open access under the Creative Commons Attribution-Non-commercial-No Derivatives License - CC BY-NC-ND 4.0.

The ESP provides many tools to further enhance the application of ES for nature conservation, ecosystem restoration and sustainable management. More information available at https://www.es-partnership.org/.

THE MANAGEMENT OF CMES

In 2008, the Ecosystem Service Partnership (ESP1), an international community of more than 3000 scientists, policymakers, practitioners, stakeholders, and end-users of ecosystem services, started a series of annual conferences aiming to gather its members to work on identifying the accomplishments and gaps of the ES concept to date. Under the title of “Solutions for Sustaining Natural Capital and Ecosystem Services,” the conference of 2010 produced a major message for the world, which is known as the Salzau Message.

Burkhard et al. (2012) reproduced a fragment of this message: “The human population of earth is likely to increase to 9 billion people by the end of the century, the global climate is being transformed, biodiversity loss continues, and conventional, fossil-based economies are no longer a viable option. Business as usual is a utopian fantasy. If we are to improve the sustainable well-being of humanity […] a precautionary approach to decision-making should […] be adopted. We believe that solutions to providing a sustainable and desirable future require broad recognition of the basic facts about ecosystem services and natural capital, and advances in two key areas: (1) integrated measurement, modeling, valuation and decision science and; (2) adaptive management and new institutions.”

The MEA (2005) made evident that human use of ES is expanding, at the same time that the status of most of those services show a decreasing trend (Figure 5). Alarming is the decline in regulating services, whose deterioration foreshadows future declines in other ES. Indicators show that future trajectory will continue to be unfavorable unless society acts to combat the adverse trends (Leslie and McLeod, 2007; Carpenter et al., 2009).

(Freeman et al., 2013; Small et al., 2017). This scenario will progressively contribute to the undermining of ecosystems and induce limitation of policy-makers to conserve them (Sagebiel et al., 2016). Pearce (2007) proposed the creation of a dual process of economic valuation by capturing those values under some sort of market creation, whilst Fisher et al. (2008) proposed keeping distance from monetary valuation, since it is not necessary but always generates issues due to its imprecision and difficulty (Kahneman et al., 1993; Bateman et al., 1997a,b). What turns out relevant from the application of this framework is that capturing the benefits of those non-marketed ES is essential for a meaningful future of the ESP (Schuhmann and Mahon, 2014).
Like the MEA in 2005, the “Salzau Message” acknowledged the inevitability of change in the status quo if we want to achieve sustainability, and proposes the ES concept as a tool to guide the process of decision-making and the management of the Natural Capital. This strategic integration of the ES concept into the ecosystem approach (CBD, 2000) is known as the ESA.

The ESA is, in short, a strategy which acknowledges the complexity of human-nature interactions and the need for interdisciplinarity to manage the complex social-ecological systems in which ES are generated, delivered, and ultimately enjoyed by humans (Ostrom, 2009). The ESA could take various forms and include numerous methods, but according to Martin-Ortega et al. (2015), it is based on four common aspects: it requires a valuation of ES based on the benefits humans get from them; ES are the product of ecosystem processes, a relationship which should be explicitly described; interdisciplinary collaboration together with stakeholder engagement at various levels is required; and the outcomes are susceptible to be incorporated into environmental policy and management decision making.

The ESA has been increasingly recommended to inform environmental management and planning (Daily et al., 2009; Maes et al., 2012; Börger et al., 2014), and efforts to apply it at different governmental scales are notably more apparent (Beaumont et al., 2017), being the establishment of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) in 2012, a major milestone in the pursuing of this effort. However, many authors agree on the inappropriate or inconsistent application of the ESA for supporting management and policy-making of CMES, with reasons including its overwhelming complexity (Koschke et al., 2012; Collie et al., 2013), the relatively poor number of success stories of CMES assessments included in decision making (Laurans et al., 2013), to the lack of clear guidelines about which of the many possible CMES could and should be quantified (Egoh et al., 2007), or other issues related to governance systems (Olsen et al., 1997).

Daily et al. (2009) argued that for the ESA to gain more weight in the decision-making process, two fundamental changes need to occur. First, the science of ES needs to advance rapidly; second, the ESA must be explicitly and systematically integrated into decision making by individuals, corporations, and governments. The first change is already happening (Boyd and Banzhaf, 2007; Fisher et al., 2009; De Groot et al., 2010a; Potschin and Haines-Young, 2011; Barbier, 2012). Definitions, metrics, frameworks of assessment, and future directions of the field are clearer now than in 2009. In spite of this growing body of literature and interest on ES, the effective application of the ESA for coastal and marine ecosystem management and its integration into the planning, management, and design of policies is still at an early stage (Boulton et al., 2016; Rivero and Villasante, 2016; Robinne et al., 2018).

Despite the incipient state of the ESA, the fact is that its application has had valuable lessons to build upon (Richardson et al., 2015; Beaumont et al., 2017; Steger et al., 2018). However, the management of CMES is not a typical ‘forward problem’ that could be solved by following a set of predetermined steps; it could rather be considered an ‘inverse problem’. An inverse problem consists in using the results, or observations, to infer the values of parameters characterizing the investigated system (Tarantola, 2006). Translating this concept to the ESA means using the literature available, the success and failure stories, to find the weaknesses of the system that would allow to improve the same system used during the application of the ESA (Tarantola, 2006). Following Tarantola (2005), we need to ask ourselves how does the newly acquired data modify our previous information? What is proposed in the following section does neither intend to focus on single case studies nor to review in detail each single framework proposed by a particular author. Instead, I propose a “review of reviews”.

What the contribution at hand calls a review of reviews is simply to focus on the documentation of lessons learned when the ESA was applied in a real decision-making context. These reviews are not abundant (Ruckelshaus et al., 2015; Beaumont et al., 2017; Drakou et al., 2017). However, they are extremely valuable in terms of giving an insightful perspective on the application of the ESA, since, in most of the cases, the authors did not just compile a group of case studies, but rather participated totally or partially in the implementation, execution, and posterior analysis of results of those cases. By focusing on the lessons learned, we can turn back to the ESA system model and take a look at what needs to be improved for future applications, considering the particularities of the given contexts.

Moving From Knowledge to Real-World Decision Making

Based on the review of the work of Ruckelshaus et al. (2015); Beaumont et al. (2017), and Drakou et al. (2017), six key lessons were identified. These lessons are recognized as milestones for improving the ESAs application. They aim to increase the potential use of the ESA for the management of CMES and the generation of greater impact (Posner et al., 2016) on management schemes at different governance levels and over a wide variety of countries (Sattler et al., 2018). Although the three reviews did not base their findings on the same case studies, or even came to the same conclusions, they mostly focused on similar key points of the ESA process. Therefore, it is precautionary for the reader to be aware of the limitations of this review, which, except for Ruckelshaus et al. (2015), had their case studies located mostly in the global north. Therefore, contextualizing is extremely relevant if replication in other realities is intended (Pendleton et al., 2015).

Lesson 1 - Impacts Are Greater When the ESA Is Part of a Science-Policy Process

Ruckelshaus et al. (2015) asked this key question during their research: “What kind of information is needed to create useful, credible science and change decision process and outcomes?”. They found it is hard to know unless an interactive transdisciplinary science-policy process is implemented, in which
FIGURE 5 | Trends in human use (upper) and condition of ES (lower). Provisioning, regulating, and cultural ecosystem services are shown in the left, center, and right side, respectively. Length of black radial lines shows the degree of change in human use or condition of the service. Reproduced from Carpenter et al. (2009). Copyright (2009) National Academy of Sciences.

scientists, local experts, stakeholders, and decision makers are involved. In other words, for CMES to be operatively used to inform decision-making, science cannot be a simple provider of information, but rather a process of knowledge integration should take place (Posner et al., 2016; Drakou et al., 2017). Likewise, the ESA should help to clarify which CMES are susceptible to be quantified, with what tools, and ultimately if the application of the ESA would help clarify a particular policy decision (Pendleton et al., 2015; Steger et al., 2018).

The process to accomplish this integration is, as expected, highly complex and time consuming (Bagstad et al., 2013). However, in places like Belize, it certainly has created a significant impact (Arkema et al., 2015). In 2010, Belize's Coastal Zone Management Authority and Institute (CZMAI) started a collaboration with The Natural Capital Project (NatCap)3 and World Wild Fund for Nature (WWF) to create a national coastal zone management plan. This collaboration brought interesting synergies to both sides, allowing NatCap to gather critical local knowledge that was later used to co-develop alternative zoning schemes, producing scenarios that were relevant to local stakeholders and provided just the necessary level of input detail to the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) models for exploration of likely CMES outcomes (Coastal Zone Management Authority Institute [CZMAI], 2012).

However, once the importance of this integration is acknowledged, how should it be framed? Daily et al. (2009) proposed a framework in which the ESA is outlined into a linear sequence formed by five stages: Institutions, Decisions, Ecosystems, Services, and Values. Each stage is linked to a process which involves the flow of information in the form of models, by providing clear and credible ES information for decision makers. More information available at: https://naturalcapitalproject.stanford.edu/.

3The Natural Capital Project (NatCap) was formed in 2006 with the primary goal of transforming decisions affecting the environment and human well-being.
valuation or incentives. This framework resembles the cascade model (Haines-Young and Potschin, 2010) with the addition of the governance level affecting all the other stages.

Lopes and Videira (2013) proposed an iterative process which breaks down the process into three main stages. Firstly “Set the scene”, which involves a listing of all the institutions that could potentially affect the decision, including the identification of relevant stakeholders. Secondly, “Deepen understanding”, involving three steps: (1) identification of ES and their variation; (2) identification of long-term impacts on ES; and (3) the social, ecological, and economic values attributed to the different affected services. Lastly, “Articulate values,” meaning to reach the aggregation and weighting of the different values assigned to ES and their effect in the decision-making process. This framework aims to produce information on the values of ES based on the different visions and perspectives of multiple stakeholder groups. Moreover, it promotes social learning for stakeholders and the institutions in charge of the decision-making process.

A downside of this iterative science-policy process is that it is time consuming (Ruckelshaus et al., 2015; Hummel et al., 2019). Hence, it is critical to align the scope of activities with the objective pursued by the stakeholders, so that a greater impact is generated (López-Rodríguez et al., 2019). In this sense, we must understand that impact could be diverse because it varies according to the context in which the ESA is applied. Kingdon (1995) argued that the time span for new information to influence real decision could vary from months to even decades. Therefore, success should be measured at several points along the decision making-process (Beaumont et al., 2017). Ruckelshaus et al. (2015):(12) talked about a ‘diversity of successes’, meaning that there are different pathways constituting some form of success in incorporating ES information into decisions and outcomes. For explanation, they developed a framework of four pathways (Figure 6), in which deeper impact is achieved when the process evolves from top to bottom down each pathway, and from left to right between the four pathways (Ruckelshaus et al., 2015). Science-policy engagement will traverse these pathways to a different extent, and the stages can be used to track progress which is not always linear.

**Lesson 2 - Simplicity Is Mostly Preferred Over Complexity**

The use of modeling tools for the ESA has a strong component of subjectivity (Daily et al., 2009; Ruckelshaus et al., 2015; Costanza et al., 2017). This means that models, usually developed solely by scientists, are perceived as complex and difficult to use by decision-makers and other stakeholders (Guerry et al., 2012). The models created by scientists are seen from their perspective as simplified versions of biophysical and socio-economic processes. In spite of that, decision-makers tend to ask for simpler, fit-to-objective, and more understandable tools which can be incorporated into the science-policy processes (Ruckelshaus et al., 2015; Wam et al., 2016). Decision-makers are often best served by relatively simple models, as long as they

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**FIGURE 6** | The four pathways of impact for biodiversity and ecosystem services (BES) information on decision-making process. A pathway means a level of success achieved when science and policy act together to achieve more sustainable management of CMES. Pathway 1 represents the creation of new, high-quality and relevant information on CMES. Pathway 2 provides new understanding, thus, raising awareness or achieving mind-shift. Pathway 3 represents the influence of CMES information on specific actions and the behavior of decision-makers. Pathway 4 achieves specific outcomes in terms of developing new policy or finance mechanisms, and making measurable improvements in ES provision, biodiversity and human wellbeing. Reproduced from Ruckelshaus et al. (2015); open access under the Creative Commons Attribution-Non-commercial-No Derivatives License - CC BY NC ND).
are clearly documented, published, and validation tests reveal limitations (Kareiva et al., 2011; Kim et al., 2012; Tallis et al., 2013; Arkema et al., 2013).

Once the ESA process has been framed into a science-policy context, the engagement and involvement of stakeholders is vital to ensure that the models used are the best possible to achieve the objectives established. According to Guerry et al. (2012), for a model to be useful in most decision context, it needs to evaluate how changes in ecosystem structure and function will affect the flow of services. In other words, by the analysis of input variation (i.e., labor, materials, habitat), models should predict the production of outputs (i.e., fish, wood, aesthetic value, coastal protection). The valuation of such changes, whether in monetary or other terms, ultimately provides the common language for decision making (Barbier et al., 2011). A tool developed to provide this is the NatCap Project’s suite of models called InVEST.

The InVEST is flexible and scientifically grounded and possess six main characteristics: it derives ES flow from the underlying biophysical processes that produce them, it is spatially explicit, outputs are not limited to monetary values, it is scenario driven, it reveals relationships among multiple ES, and it has a tiered approach to accommodate a range of data availability and knowledge of the system (Guerry et al., 2012; Arkema et al., 2013). The tiered modeling approach ensures that the models are useful in various contexts, including in places with sparse data (Dailly et al., 2009). Tier 1 models, for example, have modest data requirements and can inform general planning. Tier 2 models could calculate ES levels and corresponding values, whereas Tier 3 models integrate complex models, developed in partnerships with other research institutions, which include analyses of ES change in fine temporal scales and analysis of feedbacks among different ES (Fulton et al., 2004a,b; Sanchirico and Mumby, 2009).

Although the three tiers of complexity initially planned for InVEST models seemed reasonable, their engagement in the real world has led to the development of even simpler Tier 0 models, that are especially useful in those places where stakeholders have discrepancies about which ES should receive more attention, or which management scenarios should be evaluated with priority (Guerry et al., 2012). For example, in the spatial coastal planning of the West Coast of Vancouver Island, what decision-makers needed at first was a simple screening tool to rank and highlight areas most vulnerable to coastal hazards under different scenarios of habitat presence (Arkema et al., 2013).

Initially developed for land-based ecosystems, the InVEST toolbox now includes models for marine and coastal systems for renewable energy (Tier 1), food from fisheries (Tiers 0 and 1) and aquaculture (Tier 1), coastal protection (Tiers 0 and 1), the provisioning of aesthetic views (Tier 0), recreation (Tiers 0 and 1), carbon storage and sequestration (Tier 1). Marine InVEST also includes two supporting service models that account for ecological linkages between the processes that generate changes in the ES listed above: water quality (Tiers 0 and 1) and habitat risk assessment (Tier 0).

One main advantage of InVEST is that it was explicitly designed to integrate stakeholder engagement through an iterative process that involves the identification of objectives, for which the model is needed; the development of scenarios, meaning the possible effect of management interventions on the flow of CMES; and the compilation of data necessary to run the model. Therefore, simple and transparent tools with a low entry barrier can enable stakeholders with limited data availability to actively engage in a deliberative decision-making process (Kubiszewski et al., 2017). Complexity should be an aim only when conditions of data availability and capacity allow it, while special attention should be given to the value and demand for robust, simple models to inform decisions in the ESA context (Pendleton et al., 2015; Ruckelshaus et al., 2015; Evans, 2019).

Lesson 3 - Beyond Stakeholder Participation: Empowering Local Experts With Technical Tools

The operationalization of the ESA for the management of CMES requires the involvement of a variety of stakeholders (policy-makers, decision-makers, practitioners, local scientists), whom Drakou et al. (2017) called ‘end-users’. However, stakeholders have usually different interests or priorities (Weichselgartner and Kasperton, 2010; Dick et al., 2018), which makes it especially challenging to successfully integrate their participation into the ESA. Therefore, a pervasive trend has unfortunately established in the science-policy interface, i.e., that policy-makers use little available research-based knowledge, and researchers produce insufficient knowledge which is directly usable for policy-makers (Posner et al., 2016; Milon and Alvarez, 2019).

To achieve a better collaboration and integration of the science-policy interface, Drakou et al. (2017) proposes five points to be considered:

1. Balance values, power relations, attitudes, and expectations of involved stakeholders, by using collaborative decision-making tools like workshops, focus groups, participatory mapping, and modeling (Voinov and Bousquet, 2010; Ruiz-Frau et al., 2011; Kenter et al., 2016b; Ranger et al., 2016);
2. Communication of the levels of confidence of scientific results, thus, favoring the build-up of trust by decision-makers in research (Gissi et al., 2017);
3. Convergent use of language during stakeholder participation, less discipline-specific jargon and employment of user-oriented terminology (Weichselgartner and Kasperton, 2010);
4. Communication of challenges to funding agencies, since funding sources for interdisciplinary research are still sporadic and vary between countries, consistency in such funding should be encouraged (Bremer et al., 2015; Beaumont et al., 2017); and
5. Persuasion of government institutions and research agencies about the need for data and information sharing (Drakou et al., 2015).

Some authors have also acknowledged the importance of carrying on the ESA at a local level (Crossman et al., 2013; Ruckelshaus et al., 2015; Beaumont et al., 2017; Lau et al., 2019), because it addresses specific management issues, hence, having the potential to generate higher impact on the decision-making process. This local application opens the opportunity...
to bring closer the tools of the EEA to local experts (scientists and decision-makers), like those using the InVEST platform. Ruckelshaus et al. (2015) found that in places where local partners received training in EEA tools, like in Latin America (Goldman-Benner et al., 2012), Belize (Coastal Zone Management Authority Institute [CZMAI], 2012), and China (Daily et al., 2013), an ESA spreads rapidly and CMES information is adopted in policy. The final outcome is higher buy-in and ownership of the results by stakeholders, allowing policy change to happen more broadly (Dick et al., 2018; Robinne et al., 2018).

Lesson 4 - Value Does Not Always Mean Money

Regarding the EEA, nothing has been more controversial than the issue of valuation (Farley, 2012; Parks and Gowdy, 2013; Norton and Hynes, 2014; Marre et al., 2016). Perhaps the quote from Braat and de Groot (2012): (12) can illustrate this well: “To value is to monetize in the eyes of many, some of which state this with enthusiasm, others with horror.”

The rise of the EEA concept in 1997 saw many scientists facing the dichotomy of accepting EEA valuation as useful for policy-making (Costanza, 1998; Daly, 1998) or rejecting it for considering it useless and morally unacceptable (Norgaard and Bode, 1998; Toman, 1998). Regardless of this, few focused on arguing why valuation had to be considered a synonym of monetization (Parks and Gowdy, 2013).

Understanding the meaning of value and the subsequent act of valuation is key in this debate. Box 2 contains more information about the meaning of value in the EEA. To clarify the idea of valuation, we need to make a distinction between the process of valuation and the content of valuation. Himes and Muraca (2018) refer to the process of valuation as how it occurs that something we encounter becomes important, significant, or worth our attention; while the content of valuation is the product of the process of valuation and it refers to what is valued and how the value is attributed and articulated. Therefore, when people talk about valuation simply as monetization, they are just seeing half of the picture (Figure 7). The use of unidimensional value framings, for example economic, social, cultural, or ecological (Figure 7, left panel), creates valuation gaps and a strong contrast, if compared with the application of a more integrated approach that aims at bridging different value dimensions (Figure 7, right panel), associated with value pluralism (Pascual et al., 2017).

Moving beyond monetary valuation has been the focus of many interesting studies (Farley, 2012; Kenter et al., 2015; Irvine et al., 2016; Kenter et al., 2016a,b; Marre et al., 2016; Mavrommati et al., 2017; Small et al., 2017) which dealt with the widespread misconception that the EEA always requires an economic valuation (Engel et al., 2008; Loomis et al., 2014; Kubiszewski et al., 2017). Issues emerging from the use of monetarization in EEA valuation lie, on the one side, on the economic model itself, which has led conventional economists pursue the maximization of monetary value, by integrating ES into the market framework (Farley, 2012). Gowdy (2004, 2005) called the basic economic model a “financial investment model” which serves only individuals and purely economic decisions. On the other hand, a persistent and critical issue in this debate is the idea of cultural ecosystem services, and those services that are often quite intangible and without obvious material benefits, which nevertheless, are closely linked with our perceptions of the world and our well-being (Small et al., 2017).

Likewise, values’ elicitation has recently been questioned (Kenter et al., 2015) drawing attention to the assumption that value to society is just the aggregation of individual valuations (Klamer, 2003). This is especially true when considering the new findings in behavioral psychology, neuroscience, and social anthropology, which have shown that human decision-making is also a social, and not only an individual process (Parks and Gowdy, 2013; Rose et al., 2020). Kenter et al. (2015) elaborates extensively on the social component of EEA valuation, identifying this area as one of the biggest challenges to be faced by the EEA. Their work outlines a framework of shared/social values across five dimensions: value concept, provider, intention, scale, and elicitation process. Along these dimensions, seven (non-mutually exclusive) types of shared values are identified: transcendental; cultural/societal; communal; group; deliberated and other-regarding values; and value to society (Kenter et al., 2015).

Costanza and Folke (1997) described three objectives that valuation of EEA should achieve: ecological sustainability (Farley, 2012); fair distribution (Barnes, 2006; Lau et al., 2020); and efficient allocation (Farley, 2012; Steger et al., 2018). Therefore, a valuation process which explicitly includes non-market values, mostly from regulating and cultural services (Arias-Arévalo et al., 2017; Steger et al., 2018), and which allows the emergence of “shared and social values” (Kenter et al., 2015), is now elaborated in many places under names such as ‘integrated valuation’, and ‘participatory valuation’, where combinations of valuation methods are used to address the full set of values (Braat et al., 2014; Jacobs et al., 2016; Kenter et al., 2016a; Villegas-Palacio et al., 2016; Small et al., 2017).

Although decision makers are familiar with economic valuation and even consider it useful, most of them never or rarely use it (Cornwall Maritime Strategy, 2012; Marre et al., 2016; Beaumont et al., 2017). Ruckelshaus et al. (2015) reported in their work that, in many cases, stakeholders explicitly refused the attachment of monetary value to key benefits they obtain from EEA. On the other hand, in places like Hawaii, both monetary and non-monetary metrics are important to decision makers (Kaiser and Roumasset, 2002). Thus, an essential lesson rescued from the application of the EEA is that the possibility to calculate monetary metrics, does not necessarily imply its relevance to inform decisions (Robinne et al., 2018; Milon and Alvarez, 2019). Ruckelshaus et al. (2015) acknowledged that persistence on economic valuation only represents a barrier for both the science development side (i.e., for those who imagine their work is not relevant if they are not interested in formal monetary valuation) and the practitioner side (i.e., for those who believe that an EEA excludes the value of biodiversity for its own sake).

Lesson 5 - Emphasize the Link to Human Well-Being

The MEA (2005) concluded that although many of the EEA provided by nature have experienced a significant decline, a steady gain in Human Well-Being (HWB) at the global scale was undeniable. However, ecologists have argued that
environmental degradation would be followed by a decline in the provision of ES, thus, leading to a decline in HWB (Dasgupta, 2001; Kremen, 2005; Villamagna et al., 2013). This apparent incongruency is called by Raudsepp-Hearne et al. (2010) as the ‘Environmentalist’s Paradox’. This paradox represents a starting point to explore the casual relationship between ES and HWB, and why, although there is overwhelming evidence of human-induced change of biosphere through ES degradation (MEA, 2005), climate change (IPCC, 2019) and land-cover change (Kareiva et al., 2007), the consequences of those changes for human well-being are far less clear (Raudsepp-Hearne et al., 2010; Mulder et al., 2015; Rivero and Villasante, 2016).

In the ESF, a variety of definitions of HWB has been used (Agarwala et al., 2014), each of them using a different measure emphasis, based either on objective changes, i.e., higher income, nutrition, health (Haughton and Khandker, 2009); or subjective changes, i.e., happiness (Kahneman and Deaton, 2010) surrounding the individual. But well-being is a complex, multi-dimensional, dynamic concept that cannot be easily defined and measured (Fry et al., 2017); thus, hybrid approaches to HWB in the ESF should be prioritized. In this sense, the definition of HWB by Gough and McGregor (2007) as “a state of being with others, where human needs are met, where one can act meaningfully to pursue one's goals, and where one enjoys a satisfactory quality of life”, is a hybridized definition which promotes a holistic approach for monitoring the impacts of interventions on people and ecosystems (Fry et al., 2017). This definition is related to the three-dimensional approach to human well-being, that includes material well-being, what people have and whether or not their needs are met; relational well-being, how social relationships enable an individual to pursue good living conditions; and subjective well-being, how individuals feel about what they have.

Although the relationships among HWB and ES are considered in a growing number of case-based research studies (Blythe et al., 2020), authors like Ruckelshaus et al. (2015); Drakou et al. (2017) and Beaumont et al. (2017) have found the link between the change in ES delivery and HWB, as one of the weakest points on many ESA experiences. Reasons for
this weakness are various, having as main drawbacks the poor research on CMES trade-offs (Howe et al., 2014; Villasante et al., 2016); the unidimensional valuation of many CMES (Markandya et al., 2008; Naber et al., 2008); the excessive attention of research and assessment on provisioning services (Liquete et al., 2013; Grêt-Regamey et al., 2017); and the lack of exploration of the link between CMES and poverty (Daw et al., 2011; Fisher et al., 2013; Reyers et al., 2013).

In a review of the literature on trade-off and synergies when the ES concept was used to determine HWB, Howe et al. (2014) found that a major gap was the observed lack of studies within coastal and marine areas. Trade-offs occur when the provision of one ecosystem service is reduced as a consequence of increased use of another, or when more of a particular ecosystem service is captured by one stakeholder at the expense of others (Rodríguez et al., 2006). It is a particular challenge to understand the factors that influence this trade-off dynamic, since they can occur both among stakeholders as well as among the ES being delivered in any location (Howe et al., 2014). However, this should be prioritized if we are to understand how the simultaneous uses of CMES interact, because the consequences of such interactions can be positive, e.g., ecotourism and biodiversity conservation (Mahajan and Daw, 2016; Rasheed, 2020), but in others, especially when there is resource consumption by humans, e.g., fisheries, there can be conflictive and negative (Worm et al., 2006).

As formerly elaborated in Lesson 4 (Section “Lesson 4 - Value Does Not Always Mean Money”), unidimensional valuation has led policy-making to focus attention mostly on the monetarization of ES. As a consequence, economic growth strategies, especially concerning the marine environment, have failed to acknowledge the reciprocal role that humans play both as driver of change and recipient of the impact of those changes (Daw et al., 2016; Drakou et al., 2017). At the European level though, some directives, like the Marine Strategy Framework Directive (Berg et al., 2015), have tried to shift attention toward the maintenance of a Good Ecological Status, and the assessment of coastal and marine ecosystem health (Halpern et al., 2012), shifting the attention from the purely economic aspect.

There has been a limited exploration of the link between cultural ES and well-being, if compared with the major attention received by provisioning ES (Liquete et al., 2013; Bullock et al., 2018; Wood et al., 2018). This has already been observed in the literature (Rodríguez et al., 2006; Tallis et al., 2008), and could be due to the fact that these services are not as well understood as provisioning or regulating services (Crossman et al., 2013), therefore, they are more difficult to study and measure. It could also be related to the fact that, since they are often not captured by monetary valuations, there is a general lack of interest in these types of services (Pleasant et al., 2014). Nevertheless, the relationship between ecosystems and well-being is still framed largely in terms of material benefits, i.e., harvesting fish to generate income (MEA, 2005). Although in some ESA experiences the trend is reversing, e.g., in Vancouver Island, where a key human wellbeing concern is to maintain access of First Nation communities to culturally important shellfish harvest areas (Ruckelshaus et al., 2015); or in Kenya, where ecotourism showed important cascading benefits to local people including income, as well as a stronger connection to place and opportunities to engage with outsiders (Mahajan and Daw, 2016).

Emerging from the MEA (2005), there is a new research agenda which explores how ES allocation could help on the alleviation of poverty. Links have been made between poverty and environment because poor rural people in developing countries often have higher dependence on livelihood resources directly from nature (Narayan et al., 2000). Also, because poor people are highly vulnerable to environmental change and stressors (MEA, 2005). The assessment of CMES at different spatial scales (Barbier et al., 2008) have uncovered growing inequalities within developing countries, that in some cases has led local or regional collapses, with associated forced human migration and higher resource competition (Warner et al., 2008; Daw et al., 2011). If the ESA aims to inform policy-making, a disaggregated analysis is needed that focuses on who derives which benefits from ecosystems, and how such benefits contribute to the well-being of the poor (Daw et al., 2011). This analysis should include social groupings such as gender (Fortnam et al., 2019), age and ethnicity (Cruz-Garcia et al., 2019), especially in places where inequality is greatest (Bizikova, 2012).

Fisher et al. (2013) highlighted the advantages of addressing the complexity of the Ecosystem Service - Human Well-being relation through a comprehensive approach instead of a static checklist, when using the ESA to inform decision making. In the literature, there are a variety of conceptual frameworks (e.g., Costanza et al., 2007; Bateman et al., 2011; Fisher et al., 2013; Reyers et al., 2013; Díaz et al., 2015) which, however, have focused on specification of the ecological generation of ES to the detriment of understanding how they actually contribute to well-being (Fisher et al., 2013). Perhaps the Elasticity Approach (Daw et al., 2016) represents a good approximation that seeks to incorporate multidimensional well-being (Gough and McGregor, 2007) rather than simple aggregate value as the end point. The concept of elasticity enables the capturing of important ES-HWB responses, such as non-linearity, i.e., thresholds can trigger abrupt responses in ES provisioning; hysteresis or path-dependency, i.e., highly different HWB levels according to the status of the ecosystem, either degraded or in recovery; context-dependency, i.e., availability of compensation alternatives for decline in local ES; and access-recognition (Barbier et al., 2008; Koch et al., 2009; Andersson et al., 2015; Mahajan and Daw, 2016).

**Lesson 6 - Communicate Uncertainties and Enhance Societal Literacy**

Based on their experience applying ESA on six cases studies across South West England and North West France, Beaumont et al. (2017) recommend to work transparently with data gaps and uncertainty. They stated that: "Decisions in marine management have to be made even if data are imperfect, missing and incomplete." Data gaps are limitations for the implementation of effective policies. However, they do not represent a barrier for the existing knowledge to generate impact.
in the policy-making process (Posner et al., 2016). Here, it is important to create a distinction between information, as a tangible, factual output of scientific research produced through specific ES analyses, and knowledge, as a body of information learned and conveyed through scientific and policy processes (Posner et al., 2016).

The impacts of the ESA-generated knowledge on policy-making will depend on the efficient collaboration, engagement, and trust between stakeholders along the process of the ESA application to ultimately generate salient, credible, and legitimate knowledge (Cash et al., 2003). The latter three characteristics are identified by Cash et al. (2003) as enabling conditions to link knowledge with action. Salience refers to the relevance of scientific knowledge to the needs of decision-makers; credibility comes from scientific and technical arguments being trustworthy and expert-based; and legitimacy refers to knowledge that is produced in the less possible unbiased way and that fairly considers stakeholders’ different points of view.

In a quantitative review of 15 cases about the impacts of ESA-generated knowledge on policy-making, Posner et al. (2016) found that impact tends to increase with higher levels of salience, credibility, and legitimacy, however, legitimacy emerged as a stronger predictor of impact. This reaffirms Lesson 1 (Section “Lesson 1 - Impacts Are Greater When the ESA Is Part of a Science-Policy Process”), and indicates the importance of ES knowledge perception in the policymaking process to help shape conversations and raise awareness. Although uncertainty concerning model outputs, methods used, or knowledge gaps rising along the way, the importance of acknowledging and clearly communicating such uncertainty to end-users is vital to ensure better outcomes. In this regard, Irvin and Stansbury (2004); Reed (2008); and Beaumont et al. (2017) have found that stakeholders are used to making decisions where uncertainty is high, and responded well to the outputs despite the limitations.

Furthermore, in many cases, communication of final outcomes was restricted to end-users at workshops, which, in spite of empowering the users, limited the spread of the results to the general public as well as the decision-makers (Beaumont et al., 2017; Drakou et al., 2017). To mainstream ES and to subsequently use the ESA is another lesson. Perhaps the integration of the ESA in helping to achieve the UN Sustainable Development Goals (SDGs) will contribute to this purpose. Wood et al. (2018) found that many experts acknowledged that a variety of individual ES could make important contributions to achieving 41 targets across 12 SDGs. Although the contribution of these services to SDG14.1 (Life Below Water) was qualified as weak, results could be biased by expert’s background and their representation in the research survey, or by focusing mostly on conservation targets (Wood et al., 2018).

Most of the reviews have also recognized the variety of informal settings in which ESA knowledge is generated, and which is not properly stored. Therefore, it is recommended that the ESA is clearly stored, with good accessibility of documentation of the methods, the results, and the implementation of the results, including mistakes made (Ruckelshaus et al., 2015; Beaumont et al., 2017). Platforms like Oppla4, Panorama5, SWEEP6, and ValuES7 may provide a valuable method of storing and sharing data and experiences.

**CONCLUSION**

This review summarizes the current knowledge status of CMES, their emerging position into the ES concept, the development of their research agenda into the ESF, and its integration into policy and management through the ESA. The ES concept has certainly become one of the most used arguments to protect nature (Gómez-Baggethun et al., 2010; Muradian, 2017), possibly leading to the raise of a new paradigm in conservation, an ecosystem services paradigm (Braat and de Groot, 2012). However, discrepancies around its definition (Boyd and Banzhaf, 2007; Polasky and Segerson, 2009), system classification (Fisher et al., 2009; Lique et al., 2013), as well as with its knowledge framework (Haines-Young and Potschin, 2010; Costanza et al., 2017) are still present in the scientific community. However, practical approaches such as guided pluralism applied to the ES definition (Hermelingmeier and Nicholas, 2017) and the use of ES concept as boundary object (Nahlik et al., 2012) have facilitated the incorporation of the ES concept into the policymaking agenda globally (Daily et al., 2009; Maes et al., 2012; Börger et al., 2014).

CMES are a relatively young theme in the ESP (Gómez-Baggethun et al., 2010; Lique et al., 2013) gaining mainstream attention after the MEA release in 2005. Nevertheless, CMES has experienced a slower knowledge build-up and underdevelopment of assessment methods, when compare to land ES (Guerry et al., 2012; Maes et al., 2012; Patterson and Glavovic, 2013; Townsend et al., 2018). Reasons for this gap are related to the inherit challenges of studying marine ecosystems (Carr et al., 2003; Townsend et al., 2018), leading to major research being done in coastal ecosystems services (Costanza et al., 2008; Chen et al., 2009; Hicks, 2011; Zhang and Smith, 2011) leaving behind other services provided by areas such as the open sea (Stocker, 2015) or the deep sea (Thurber et al., 2014). Furthermore, spatial data scarcity (Guerry et al., 2012;
Townsend et al., 2014; Nahuelhual et al., 2020) have highly limited the elaboration of CMES delivery maps, which are significantly relevant when assessing land ES at large scales. As a consequence, most of the research on CMES has been focused at local scales (Liquete et al., 2013; Townsend et al., 2018), which might be detrimental for assessing global services such as climate regulation, in which the scale of assessment is at the level of square kilometers or larger (Bouillon et al., 2008; Lovelock, 2008).

Coastal and marine areas are denominated to great current human pressure (Small and Nicholls, 2003; Diop and Scheren, 2016), with future status having even less odds of improving if current management and business-as-usual continue (Burkhard et al., 2012; IPCC, 2019). Therefore, CMES need to be assessed, valued and gradually integrated into the decision-making process (Daily et al., 2009; Beaumont et al., 2017). The ESA has facilitated this integration, although with different levels of impact (Ruckelshaus et al., 2015) and success (Laurans et al., 2013). Nevertheless, important lessons can be rescued from worldwide ESA application experiences, serving as motors of future improvement for the approach (Richardson et al., 2015; Beaumont et al., 2017; Steger et al., 2018). This review derived six lessons: (1) integration of the ESA in a science-policy process; (2) more simplicity for the CMES prediction models; (3) move from simple participation toward empowering of stakeholders; (4) integration of the value pluralism of CMES with less focus on money; (5) the link of ES to WHB must not be forgotten; and (6) communication of results and social literacy are key. These lessons draw attention toward parts of the ESA which either need more research (Drakou et al., 2017; Blythe et al., 2020) or a change of perspective (Coastal Zone Management Authority Institute [CZMAI], 2012; Small et al., 2017; Evans, 2019), if the aim is to integrate the knowledge generated from CMES assessment to sustainably manage and preserve the coastal and marine ecosystems of the world (Mahajan and Daw, 2016; Wood et al., 2018). There is a great body of literature from which ESA can keep the momentum growth, whereby the whole society can achieve an organized, science-based, participatory, and inclusive management of CMES.

TERMS AND ACRONYMS

Ecosystem services (ES): the end products of ecosystems utilized actively or passively to produce human well-being. Ecosystem Services Framework (ESF): it is discursive tool focused in two core topics, the classification of ES, and the evaluation of the connections that those services have to human well-being. Ecosystem Services Approach (ESA): a novel management tool which gathers views from the ecosystem-based management approach and the ideas of the ESF. Ecosystem Services Paradigm (ESP): the incorporation and apparent prevalence of the ES concept into the conservation agenda. Coastal and Marine Ecosystem Services (CMES): those services produced on the marine ecosystems (areas deeper than 50 meters), and coastal ecosystems (areas located between 50 meters below mean sea level and 50 meters above the high tide level). Ecological production function (EPF): it is the set of processes that occur in nature which enable the generation of ES. Economic demand function (EDF): it represents the connection between the ES and the benefit we humans obtained from them. Human well-being (HWB): a state of being with others, where human needs are met, where one can act meaningfully to pursue one’s goals, and where one enjoys a satisfactory quality of life.

AUTHOR’S NOTE

In 2005, the Millennium Ecosystem Assessment defined ecosystem services and proposed it as a tool to preserve and manage our ecosystems. Since then, a significant progress has been made on understanding what an ecosystem service is and how it can be used to support policy-making. However, several points of discrepancies are detectable in the literature, especially when touching those services produced and delivered by coastal and marine ecosystems. Focusing particularly on CMES this review condensed relevant points of discrepancies related to the ES concept as well as the reasons proposed to explain why a knowledge gap and an underdevelopment of assessment tools is present when comparing land ES and coastal and marine ecosystem services. Touching on these topics will help researchers to direct efforts toward yet unexplored research paths or work on the limitations of the methods and/or assessments tools for CMES. Likewise, the exploration of the applicability of CMES assessment in management is focused on rescuing those valuable lessons driven out of the application of the ecosystem service approach (ESA) to manage coastal ecosystems. These lessons are expected to call stakeholders’ attention toward ESA points which need to be improved, or corrected, for a more reliable ecosystem management.

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

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