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Ecosystem service framework and typology for an ecosystem approach to aquaculture

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Abstract

The ecosystem approach to aquaculture (EAA) considers ecosystem services (ES) important, but does not provide a conceptual framework or a typology to integrate and assess them. To supplement the EAA, a literature review of the ES conceptual framework and ES typologies was combined with selected criteria from the EAA and ES literature. Eight criteria of transition from a conventional approach to aquaculture to the EAA were used as selection criteria to choose a conceptual framework of ES relevant with the EAA. To select a typology, we determined that ES must be distinguished from benefits, be a part of nature, be usable directly and indirectly, and not contain support or habitat ES. The conceptual framework of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is the most compatible with the EAA but does not provide an ES typology. The Common International Classification of Ecosystem Services (CICES) provides the ES typology most consistent with EAA criteria to supplement the conceptual framework. We identified 10 provisioning ES, 20 regulation and maintenance ES, and 11 cultural ES. Integration of the IPBES conceptual framework with the CICES typology preserves the generic approach of the EAA. This integration could highlight the main interactions among an aquaecosystem, its ES supply, its management, and its relevant stakeholders at multiple

spatial and temporal scales. Moreover, it fulfils the three main goals of the EAA by identifying them in a clear and common framework.

Keywords: ecosystem services, sustainable aquaculture, integration, aquaculture, ecosystem approach to aquaculture

1. Introduction

1.1. Issues and perspectives for aquaculture

Aquaculture has increased worldwide since the 1980s (FAO, 2010b), which corresponds to a “blue revolution” in response to the increasing demand for seafood. Seafood supplies large amounts of animal protein worldwide, especially for those living in poverty (FAO, 2000), but fisheries, which capture 70-80 million tons of seafood per year (FAO, 2016), cannot meet global demand. By 2030, aquaculture is expected to provide more than half of seafood supply (Stickney and McVey, 2002; FAO, 2016) and by 2050 might supply nearly two thirds of fish consumption (FAO, 2010b).

Apart from the demand for seafood, many external issues could influence the development of aquaculture. Climate change is the greatest external environmental influence on aquaculture production (De Silva and Soto, 2009). Water crises are another issue encountered by aquaculture. Water scarcity decreases water quality (Molden, 2007), as freshwater aquaculture can require large amounts of freshwater (Verdegem et al., 2006). Another issue is competition for land between freshwater aquaculture and agriculture (Bosma and Verdegem, 2011; OECD-FAO, 2013). Demand for seafood and external influences encourage intensification of aquaculture, i.e. to produce more with the same area and less water. The type and intensity of an aquaculture system influence its impacts (Bergheim and Asgard, 1996; Primavera, 2006; Diana, 2009). Aquaculture may affect wild fish populations indirectly, by using fish meal and fish oil as feed ingredients (Naylor and Burke, 2005; Naylor et al., 2005; Alder and Pauly, 2008), and directly, by providing fish for stocking natural areas (Khavtasi et al., 2010). Using these kinds of ingredients in fish feed can decrease food security, especially for low-earning fish farms (Tacon and Metian, 2009). An approach that mitigates external, environmental, and social issues and improves positive aspects of aquaculture might require considering aquaculture systems as ecosystems, which provide multiple levels of ecosystem services (ES).

The United Nations Convention on Biological Diversity (CBD (2004) developed some of the first guidelines for the ecosystem approach (EA), including 12 principles that can be applied to aquaculture (Supplementary Materials 1). One of the main principles is "conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach" (CBD, 2004). According to Soto et al. (2008), an ecosystem approach to aquaculture (EAA) is a plan for integrating aquaculture within broader ecosystems. The EAA should promote sustainable development, equity, and the resilience of interconnected social-ecological systems. The FAO (2010a) developed guidelines for sustainable aquaculture and the EAA based on three main principles: (i) include sustainable ecosystem functions and services; (ii) improve stakeholder well-being and equity; and (iii) be developed in the context of other sectors, policies, and goals. These guidelines consider the concept of ES relevant, but do not provide a suitable definition. The objective of this study is to provide a conceptual framework and typology of ES, based on the literature that is relevant for the EAA.

1.2. Ecosystem services definitions

ES can be defined as "nature's benefits to people" (Díaz et al., 2015). Two major works were published in 1997 that addressed two distinctive streams of thoughts around ES. Daily (1997) highlighted the human dependence on natural ecosystems (ecology-centred thinking), while Costanza et al. (1997) estimated the economic value of 17 ES for 16 biomes with the assertion that human capital (e.g., built, social) must be added to ES to obtain human benefits (economy-centred thinking). Eight years later, the Millennium Ecosystem Assessment (MEA, 2005) placed ES on the international agenda, developing a conceptual framework and typology of ES to assess and communicate about ES. The MEA improved research on ES and generated long-lasting enthusiasm about them. Since then, many conceptual frameworks and typologies have been developed based on the Daily school of thought (Boyd and Banzhaf, 2007; Sukhdev et al., 2010; Wallace, 2007) or the Costanza school of thought (Díaz et al., 2015; EPA, 2015; Haines-Young and Potschin, 2013; Landers and Nahlik, 2013; Staub et al., 2011). In addition to these two main philosophies, conceptual frameworks of ES can be classified into four main types:

- consequential, which describes quasi-mono-causal relations between components in the framework (Daily, 1997; Sukhdev et al., 2010; Haines-Young and Potschin, 2013)
- structural, which focuses on connections between components in the framework (MEA 2005; Díaz et al., 2015)
- operational, which focuses on ES provided by a specific site (e.g., ecosystem, region) (Wallace, 2007; Staub et al., 2011)
- beneficiary-oriented, which focuses on relations between beneficiaries and their use of ES (Landers and Nahlik, 2013; EPA, 2015)

Similarly, ES typologies can be classified into three types:

- functional, which focuses on what is obtained from ecosystems. In other words, they aggregate ecological functions and processes into ES (e.g., climate regulation) (De Groot et al., 2002; MEA 2005; Sukhdev et al., 2010; Haines-Young and Potschin, 2013).

- human-need-oriented, which focuses on human needs for ES (Boyd and Banzhaf, 2007; Wallace, 2007; Staub et al., 2011). For instance, Wallace (2007) classifies ES into four types of value to humans (e.g., a benign physical and chemical environment).
- beneficiary-oriented, which reflects the anthropocentric part of ES (Landers and Nahlik, 2013; EPA, 2015) and focuses on beneficiaries and how they use ES.

Different conceptualisations of ES frameworks and typologies are summarised in Table 1.

1.3. Aquaecosystem definition

The initial concept of ES was essentially focused on natural ecosystems. Some authors claim that natural ecosystems provide ES that support agriculture (Bommarco et al., 2013), while others argue that agroecosystems provide their own ES (Dore et al., 2011). An agroecosystem can be defined as an ecosystem that humans transform and manage to extract resources through agriculture. By analogy, an aquaecosystem can be defined as providing ES as "a human-managed aquatic ecosystem oriented toward the provision of ecosystem services" (Aubin et al., 2014); see Aubin et al. (2017) for an example of use). The transition from aquaculture to aquaecosystems and application of the EAA should involve valuating and assessing the ES provided. Despite this, the guidelines mentioned previously (CBD, 2004; FAO, 2010a) do not use a standard conceptual framework or typology, which restricts full application of the EAA guidelines. A conceptual framework and typology for ES of aquaculture adapted to the EAA could encourage including ES in future aquaculture systems. Moreover, connecting ES to the EAA is a future step in the use of the EAA (Brugère et al., 2018).

2. Materials and methods

2.1. Literature review

The ES literature is extensive. We used the Web of Knowledge and Google Scholar to list all relevant articles and reports about ES and their typologies and conceptual frameworks. We searched for the keywords “ecosystem services”, “typology” and “conceptual framework”. Only scientific reports, articles, and grey literature published since 2002 were reviewed, as de Groot et al. (2002) published the first conceptual framework that year, which greatly inspired the MEA (2005). Before 2002, the two works described above (Costanza et al., 1997; Daily, 1997) provided lists rather than typologies.

Several conceptual frameworks and typologies were excluded because they were too specific to be applied to aquaculture. For instance, conceptual frameworks for specific ecosystems were excluded, such as the social-ecological approach to agroecosystems framework (Lescourret et al., 2015). As valuations for individual countries were too specific to be extended to a global scale, we excluded the United Kingdom’s National Ecosystem Assessment (NEA, 2013) and Spain’s National Ecosystem Assessment (Santos-Martín et al., 2014), among others.

2.2. Selection criteria for the conceptual framework and typology of ecosystem services

To select the best conceptual framework for the EAA, we needed criteria. The 12 principles of the ecosystem approach guidelines (CBD, 2004) (Supplementary Materials 1) and of the EAA (FAO, 2010a) are the foundation for both approaches. To select criteria for the conceptual framework and typology of ES, we grouped the 12 principles of the ecosystem approach guidelines according to their relevance to the eight criteria of the EAA (Fig. 1). We used these eight criteria to select the conceptual framework:

- participatory (Principles 1, 2, 12)
- multiple objectives (Principles 1, 3, 4, 8)
- interaction with other sectors (Principles 1, 3)
- multiple (nested) scales (Principles 3, 5, 7, 8)
- adaptive (Principles 3, 9, 10)
- extended knowledge (Principles 1, 11, 12)
- incentives (Principles 1, 12)
- public/transparent (Principles 1, 2, 12)

ES have been defined many times since 1997. The different definitions refer to different conceptualisations of ES over time. We needed precise criteria to select the best typology (and concept behind the typology) of ES for the EAA. First, Fisher and Turner (2008) suggested that ES and benefits should be distinguished, as benefits have an explicit influence on human well-being, such as more or better food. Thus, we decided to distinguish ES from benefits because humans need to add human capital to obtain benefits from ES. Second, ES should be considered a part of nature (Fisher and Turner, 2008) because they are connected to and/or derived from ecosystem functions and processes. ES are not defined by the human investment or effort made to use the benefits from ES. Third, ES can be used both directly and indirectly (Fisher and Turner, 2008). Finally, the ES typology must exclude support ES and/or habitat ES to avoid double counting (Boyd and Banzhaf, 2007). For instance, carbon sequestration is an ES because it can provide net benefits from ecosystem functions and processes, especially from a climate-change perspective (Daily, 1997). In contrast, pollination may be counted twice during ES evaluation as (i) a pollination ES itself and (ii) indirectly in a provisioning ES (e.g., crop yield) or a cultural ES (e.g., an endemic/heritage plant maintained by pollination).

These four elements of ES conceptualisations became our four criteria for selecting the ES typologies:

- Are ES distinguished from benefits?
- Are ES a part of nature?
- Can ES be used both directly and indirectly?
- Are support and/or habitat services excluded?

2.3. Assessment method

Conceptual frameworks and typologies were selected independently, even if one publication presented both (e.g., the MEA). If a conceptual framework was relevant for the EAA, but not its typology (or vice-versa) we needed a back-up conceptualisation. For instance, the conceptual framework of IPBES (Díaz et al., 2015) has no typology, while the typology of (Boyd and Banzhaf, 2007) has no conceptual framework. We used the eight criteria from the EAA to select the best conceptual framework and the four criteria to select the best ES typologies. These criteria were not ranked, and the sum of the agreement between criteria and a given publication provided a grade for the conceptual framework or typology. We searched the conceptual frameworks and typologies selected for implicit and explicit information relevant for our criteria. The highest grade indicated the greatest consistency between the conceptual framework and the typology from the EAA.

3. Results

The publications selected correspond to both the published scientific literature and the grey literature on conceptual frameworks and typologies for ES (Tables 2 and 3). Few conceptual frameworks and typologies have been published. We selected conceptual frameworks in the following articles: De Groot et al. (2002), Díaz et al. (2015), EPA (2015), Haines-Young and Potschin (2013), Landers and Nahlik (2013), MEA (2005), Staub et al. (2011), Sukhdev et al. (2010), and Wallace (2007). We selected typologies in the following articles: Boyd and Banzhaf (2007), De Groot et al. (2002), EPA (2015), Haines-Young and Potschin (2013), Landers and Nahlik (2013), MEA (2005), Staub et al. (2011), Sukhdev et al. (2010), and Wallace, (2007).

The IPBES framework appeared to be the one most consistent with the EAA (Díaz et al., 2015) (Table 2). It met seven criteria derived from EAA principles and nearly all the criteria developed by the EAA (Table 2). As the IPBES does not provide an ES typology, however, we had to identify a typology that is consistent with the EAA approach and ideally with the IPBES framework.

The CICES (Haines-Young and Potschin, 2013) typology met all the selected ES criteria (Table 3) and was therefore the most suitable typology to assess ES using the EAA. It considers ES part of nature and does not contain habitat or support services, which are contained within other ES. Ultimately, we identified 10 provisioning ES (Table 4), 20 regulation and maintenance ES (Table 5), and 11 cultural ES (Table 6).

4. Discussion

4.1. Selected conceptual framework

The IPBES framework explicitly includes multiple nested scales from several spatial, temporal, and management scales. It also explicitly includes different sources of knowledge, from “Western science” to “indigenous knowledge” (expressions used by the IPBES). Experiential indigenous knowledge can supply information over a long time about a local area (Bohensky and Maru, 2011) and can fill gaps in scientific knowledge (Gadgil et al., 1993; Bohensky and Maru, 2011). According to Moller et al. (2004), scientific and indigenous knowledge are complementary in five respects:

- science tends to collect short-term data over large areas, while indigenous knowledge tends to collect long-term data over small areas
- science focuses on averages, while indigenous knowledge focuses on extremes

- science is more quantitative, while indigenous knowledge is more qualitative
- science improves tests of mechanisms, while indigenous knowledge improves hypotheses
- science seeks objectivity, while indigenous knowledge mobilises subjectivities into useful know-how

With these complementarities, scientific and indigenous knowledge can be used in collaborative approaches and can find some common ground (Bohensky and Maru, 2011). These complementarities are suitable for the EAA and its aim to integrate extended knowledge into management. Also, it “has been constructed in a transparent, inclusive and participatory manner” (Díaz et al., 2015). The IPBES’s framework has the ability to be adaptive and incentive considering its integration of “indigenous knowledge”.

Surprisingly, few ES frameworks appear consistent with the anthropocentric and utilitarian view of nature. The most suitable frameworks of ES are *a priori* the structural frameworks, which focus on interactions between ecosystems and human systems. To conceptualise ES, the human-need-oriented ES seem to correspond most to the anthropocentric and utilitarian philosophies behind the ES concept. Wallace (2007) focuses on “adequate resources” or a “benign physical and chemical environment”, not only on an aggregate of ecosystem functions and process. Consequently, human-need-oriented ES are more anthropocentric than others but could become more ecocentric if similar functions and processes are aggregated into a broader process, i.e., an ES.

4.2. Selected typology

One advantage of the CICES is its subdivision into categories of ES (Fig. 2), in which a section corresponds to the main ES categories of other typologies (e.g., provisioning, regulation, cultural). A class is defined as “a further sub-division of group categories into biological or material outputs and bio-physical and cultural processes that can be linked back to concrete identifiable service sources", while class types "break the class categories into further individual entities and suggest ways of measuring the associated ecosystem service output". The class level of CICES is the hierarchical level that we chose for our typology because it is the penultimate level of the hierarchy and allows for an accurate ES typology while remaining compatible with the diversity of aquaecosystems. The lowest level is a class type, such as the provision of X tons of common carp or the sequestration by sediments of Y tons of carbon. This ES subdivision complicates the comparison of studies; however, a tool is available that enables the CICES, MEA and The Economics of Ecosystems and Biodiversity (TEEB) typologies to be compared (http://openness.hugin.com/example/cices).

We identified 10 possible provisioning ES in aquaecosystems (Table 4). Animals (e.g., fish, molluscs, crustaceans) produced by "*in situ* aquaculture" and plants and algae from aquaculture (Hong and Lee, 1993) are the main provisioning ES. Nutrients (e.g., ammonia, phosphate) from the water in inland aquaculture systems (e.g., ponds) can be used to irrigate crops (FAO, 2005). This practice is performed in northern Vietnam, in which paddy fields and ponds are interconnected to exchange water (Steinbronn, 2010). Organic-rich sediment from aquaculture can be removed from ponds and used as fertiliser for adjacent crops (FAO, 2005). The other provisioning ES selected seem minor (producing biofuel with phytoplankton) or specific to a location (producing breeders as new genetic material). With the increasing demand for seafood, it seems pointless at present to use biomass to produce energy. The fibre and other material from biomass seem less important ES than the food provision ES in the provisioning category.

We identified 20 possible regulation ES (Table 5). Although the main function of aquaecosystems is to provide seafood, they could provide more regulation and maintenance ES at different spatial and temporal scales than provisioning ES. Algae can purify wastewater efficiently in algal ponds (Park and Craggs, 2010; Craggs et al., 2012); perhaps this potential ES could be extended to other pond systems. Fish ponds can recycle organic waste from adjacent agroecosystems, such as organic waste from pigs and poultry (Nhan et al., 2007). Sediments can accumulate and sequester pesticides in freshwater aquaecosystems (Thomas et al., 2012), which could lead to more rapid metabolisation of the pesticides or at least decrease their concentrations in water (Warren et al., 2003).

We identified 11 possible cultural ES (Table 6), which are also the least documented ES. In general, pond systems support a variety of cultural ES (Blayac et al., 2014). For instance, the Dombes region in France hosts many ponds due to monastic activities in the 13th century, when fish were produced mainly for the Catholic tradition of avoiding meat on Friday and during Lent (Hoffmann and Winiwarter, 2010). These ponds provide a heritage and cultural (or religious) ES to the region. A traditional system in China that has lasted more than 1000 years is based on the polyculture of Chinese carp, which were chosen because their trophic complementarity uses the many ecological compartments in ponds more effectively. These are considered heritage practices.

Methods to assess bundles of ES, not only one ES or a group of similar ES, are required to apply the ES concept to aquaecosystems in the EAA (de Groot et al., 2010; Raudsepp-Hearne et al., 2010). The CICES recommends assessing provisioning ES by type and amount (e.g., x tons of carp), similar to that for regulation and maintenance ES (e.g., y kilograms of pesticide accumulated by an ecosystem) and cultural ES. An inventory based on amount and type is similar to a life cycle inventory (ISO, 2006) extended to all components in an ecosystem. According to Zhang et al. (Zhang et al., 2010b; Zhang et al., 2010a), combining a life cycle method with an energy-based method (Odum and Odum, 2000) could be one way to assess a bundle of ES. By following the origins of matter and energy flows, these methods can separate the natural and human parts of ES. In line with EAA and according to Potschin, Haines-Young (2011) and Koschke et al. (2012), the valuation system of ES should be based on a diversity of criteria and units, and not only on the recommendation of the CICES. This diversity should underlie the diversity of stakeholders' perspectives (e.g., ecological, economic, social) and the diversity of relationships with nature.

If, like the selected criteria, ES cannot be benefits, cultural ES become problematic. According to Boyd, Banzhaf (2007), the main cultural ES require adding components that are not part of the ecosystem (functions and processes) (e.g., built capital, human capital) to obtain a service. Thus, they cannot be considered part of nature or an ecosystem function or process. Instead, cultural ES may be benefits obtained from the structure of an ecosystem or a landscape with additional capital. Ultimately, the issue with cultural ES is included in the CICES typology and in nearly all typologies that contain cultural ES (or an equivalent).

Other issues appear with the ES concept. Natural ecosystems provide ES, but anthropogenic ecosystems (such as aquaecosystems) also provide ES (Dore et al., 2011). The main distinction in anthropogenic ecosystems is the addition of human components that optimise or maximise one or several ES; for instance, feeding practices in aquaecosystems to maximise one provisioning ES. It seems relevant to distinguish the natural part from the human part involved in improving provisioning services. This would highlight ways humans can maximise ecosystem processes and functions to obtain ES and their resulting benefits.

4.3. Integration in the EAA

The CICES typology must be integrated into the conceptual framework of the IPBES. In the latter, the present typology is consistent with the "nature's benefits to people" dimension (ecosystem goods and services) (Figure 3). Among the three main goals of the EAA (FAO, 2010a) – (i) ensuring human well-being, (ii) ensuring ecological well-being, and (iii) facilitating the achievement of both – the first goal is embodied in the "good quality of life" dimension (human well-being) of the conceptual framework, while the second is embodied in the “nature” dimension (biodiversity and ecosystem). The third goal can be embodied as using ES sustainably by relating a good ecological state and human well-being to production of ES. Moreover, the conceptual framework of the IPBES highlights the main interactions among all components of an aquaecosystem at different spatial and temporal scales.

The CICES has no participatory component; however, it is possible to ask stakeholders to classify or distinguish the typology of the main ES of a specific aquaecosystem. Because the presence of an ES is due to its direct or indirect use, or the recognition of its value (Mathe and Rey-Valette, 2015), surveys can help to specify a suitable local ES typology from those in our typology (Blayac et al., 2014; Rey-Valette et al., 2017). The terminology used by Haines-Young, Potschin (2013) for the CICES may appear too technocratic for participatory approach with a wide diversity of stakeholders. Therefore, the name of each ES selected could be adapted to become more comprehensible to all stakeholders. For instance, CICES provides the example “in-situ farming of freshwater…and marine fish…also in floating cages; shellfish aquaculture” for the “animals from in-situ aquaculture” ES. If shellfish aquaculture is chosen as an example, “animals from in-situ aquaculture” ES corresponds to shellfish production. To apply the EAA, this ES could be translated simply into “production of a shellfish farm”. The present conceptual framework and typology of ES is not written in stone. For a specific aquaecosystem, stakeholders may identify an ES that is missing from the present typology. If so, they should describe the ES and then examine the entire CICES typology to find an ES corresponding to the specific description of the missing ES.

Like the EAA, a relevant conceptual framework and typology of ES must be general enough to represent the broad diversity of aquaecosystems around the world. Nevertheless, holistic thinking about the EAA (and ES) is limited by institutional and human capacity (Brugère et al., 2018). Although common understanding of the EAA is lacking (Brugère et al., 2018), having the same typology of ES for the EAA could be a good base from which to compare different aquaecosystems easily, including their regional and national variations. It is possible to assess ES in an EAA at different scales and for different aquaecosystems. Nevertheless, ES classes could be used at the farm scale, while ES divisions could be used at the regional scale. For instance, a shrimp farm and a rice-carp farm in the same region have different “animals from in-situ aquaculture” ES but the same “plants and algae from in-situ aquaculture” ES. At the farm scale, the ES can be “shrimp production”, “carp production”, and “rice production”. At a broader scale (e.g., catchment, region, country), the ES can be “aquaculture’s contribution to nutrition”.

In addition to the different qualities of biophysical ES valuations, management should consider social aspects of ES. It is important to know how different stakeholders perceive ES in a local context (social, economic, ecological) because ES perception is person-dependent (Hein et al., 2006). These different perceptions of ES should lead to some social trade-offs between stakeholders on the provisioning of ES (Barnaud and Antona, 2014). Management of an aquaecosystem could become complex if fish farmers or managers have to consider economic, ecological, and social aspects, their own perceptions of and benefits from ES, and the other beneficiaries of ES.

Policy makers and/or government agencies can provide advice or directives to coordinate management of aquaecosystems for the provision of ES (Leeuwis, 2013) as indirect drivers (Fig. 3). Aquaecosystem management can also be collaborative (Barnaud and Antona, 2014), involving multiple stakeholders. This kind of management should be performed at the landscape scale (Tscharntke et al., 2005; Brugère et al., 2018). According to Maris et al. (2017), however, stakeholders want more accurate ecological predictions, but these predictions remain uncertain. In addition, better knowledge does not automatically lead to better decisions (Maris et al., 2017). Nevertheless, despite the complexity of this kind of approach, introduction of the ES concept has the potential to highlight the multi-functionality of aquaculture and how aquaculture could positively influence the environment and human life.

Without ignoring controversies about ES valuation (Nunes and van den Bergh, 2001; Norgaard, 2010) or the diversity of relationships between humans and nature (Serres, 1990; Descola, 2015), some have criticised the recent work of the IPBES's conceptual group (Diaz et al., 2018), stating that it creates a term ("nature's contributions to people") nearly synonymous with "ecosystem services" (Kenter, 2018) and ignored much of the ES literature (Braat, 2018). This debate does not alter the concept of ES or its pertinence (e.g., valuing nature, human relation with it). ES's pertinence may be useful for management and policy in the context of environmental crises.

Indeed, ES could provide a variety of stakeholders (e.g., scientists, politicians, managers, advisers, citizens) with a "common language" for communication (Granek et al., 2010) despite the need for them to learn how to use the ES concept. Valuating bundles of ES could help identify trade-off situations (Howe et al., 2014) and put different technical matters on equal terms. These trade-offs could lead to win-win situations between stakeholders (Howe et al., 2014). Nevertheless, a bundle of ES cannot be managed as a function of only one ES (Chan et al., 2006; Egoh et al., 2008). Although the early literature (Costanza et al., 1997; Daily, 1997; MEA 2005) considered ES as a product of ecosystems and human activity as pressures on ecosystems, recent studies (Engel et al., 2008) highlight that humans can increase ES supply. From a management perspective, it would be interesting to consider both viewpoints. Indeed, human activities have both positive and negative impacts on aquaecosystems. This means that trade-offs would exist between possible bundles of ES in managed ecosystems (Foley et al., 2005), as would some resulting "bundles" of environmental impacts due to the management. Aquaecosystem management should consider the future provisioning of ES for the next generation (Norgaard, 2010).

5. Conclusion

The IPBES (Díaz et al., 2015) conceptual framework combined with the CICES (Haines-Young and Potschin, 2013) ES typology seem to be the most suitable combination for assessing ES in the EAA. We identified 41 potential ES (10 provisioning, 20 regulation and maintenance, 11 cultural) that aquaecosystems can provide. This combination of ES can be used to apply the EAA at a global scale. Applying the EAA and this typology could provide a more integrative and sustainable way to develop and maintain aquaculture. The demand for aquatic products will influence the future of aquaculture, and the EAA is a promising way to improve regulation and maintenance ES. ES help provide a safe environment for humans and therefore are conducive to human well-being. Cultural ES could be used to identify different ways (e.g., physical, spiritual, heritage) that aquaecosystems benefit different stakeholders (e.g., tourists, students, scientists).

The increasing worldwide demand for aquatic products should not overshadow the other ES. An ES perspective (conceptual framework and typology) integrated with the EAA could help aquaculture systems to be viewed as aquaecosystems providing multiple ES instead of systems seen mostly as negative or disruptive (Brugère et al., 2018). The EAA is one way to consider all of the ES that aquaecosystems provide.

Applying the EAA with an ES typology and conceptual framework could enhance integration of aquaecosystems in broader seascapes or landscapes at multiple spatial and temporal scales with different stakeholders' perspectives (Brugère et al., 2018). Although the combined IPBES framework and CICES typology can be used to assess ES in an EAA context, no consensus exists on which methods to use to assess ES. Using a method based on the life cycle concept seems a potential way to assess ES bundles (Zhang et al., 2010b; Zhang et al., 2010a). Consequently, future research could focus on developing methods to assess ES bundles, especially for aquaecosystems.

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Declaration of interest

The authors declare no conflicts of interest.

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ACCEPTED MANUSCRIPT

Figure captions

Figure 1. Transition from a conventional approach to aquaculture to an ecosystem approach to aquaculture (from (FAO, 2010a)).

Figure 2. Example of hierarchical levels of provisioning services (adapted from Haines-Young, Potschin (2013))

Figure 3. The IPBES conceptual framework (from (Díaz et al., 2015))

Table 1. Categories of conceptual frameworks and typologies for ecosystem services (ES). Non-bold and bold text indicate "ecologist" and "economist" viewpoints of ES, respectively.

Category		Conceptual Framework				
		Consequential	Structural	Operational	Beneficiary-oriented	None
Typology	Functional	de Groot et al. (2002) TEEB (2010) **CICES (2013)**	MEA (2005)			
	Human-need			Wallace (2007)		
	Beneficiary-oriented			**FOEN (2011)**	**FEGS (2011) NES-CS (2013)**	Boyd and Banzhaf (2007)
	None		**IPBES (2005)**			

Table 2. Ability of selected conceptual frameworks to meet criteria developed from the ecosystem approach to aquaculture (EAA). The symbol "•" signifies "present/yes", while "-" signifies "absent/no".

Selection criterion \ Publication	MEA (2005)	Wallace (2007)	Sukhdev et al. (2010)	EPA (2015)	Haines-Young and Potschin (2013)	Díaz et al. (2015)
Participatory	-	-	-	-	-	•
Multi-objective and includes other sectors	•	•	•	•	•	•
Multiple (nested) scales	•	-	-	-	•	•
Adaptive	-	•	-	-	-	•
Management and conservation of structures and functions	•	•	-	-	-	-
Extended knowledge	-	•	-	-	-	•
Incentives	-	-	-	-	-	•

Transparent	•	•	•	•	•	•
Total	4	5	2	2	3	7

Table 3. Ability of selected typologies to meet criteria developed from combining ecosystem services (ES). The symbol "•" signifies "present/yes", while "-" signifies "absent/no".

Publication / Selection criterion	MEA (2005)	Wallace (2007)	Boyd and Banzhaf (2007)	Sukhdev et al. (2010)	Staub et al. (2011)	Landers and Nahlik (2013)	EPA (2015)	Haines-Young and Potschin (2013)
Are ES distinguished from benefits?	-	-	•	-	-	•	•	•
Are ES a part of nature?	•	-	•	-	•	•	•	•
Can ES be used both directly and indirectly?	•	-	-	•	•	-	-	•
Are support and habitat services excluded?	-	-	•	-	-	•	•	•
Total	2	0	3	1	2	3	3	4

Table 4. Provisioning ecosystem services for the ecosystem approach to aquaculture

Section	Group	Class
Nutrition	Biomass	Wild animals and their outputs Animals from in-situ aquaculture Plants and algae from in-situ aquaculture
	Water	Surface water for drinking
Materials	Water	Surface water for non-drinking purposes
	Biomass	Materials from plants, algae and animals for agricultural use Genetic materials from all biota Fibre and other materials from plants, algae and animals for direct use or processing
Energy	Biomass-based energy sources	Plant-based resources Animal-based resources

Table 5. Regulation and maintenance ecosystem services for the ecosystem approach to aquaculture

Section	Group	Class
Mediation of waste, toxins and other nuisances	Mediation via biota	Bio-remediation via micro-organisms, algae, plants, and animals Filtration/sequestration/storage/accumulation via micro-organisms, algae, plants, and animals
	Mediation via ecosystems	Filtration/sequestration/storage/accumulation via ecosystems Dilution via the atmosphere, freshwater and marine ecosystems
Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates Buffering and attenuation of mass flows

	Liquid flows	Hydrological cycle and water flow maintenance Flood protection
	Gas/air flows	Storm protection Ventilation and transpiration
Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, protecting habitats and gene pools	Pollination and seed dispersal Maintaining nursery populations and habitats
	Pest and disease control	Pest control Disease control
	Soil formation and composition	Weathering processes Decomposition and fixing processes
	Water conditions	Chemical condition of freshwater Chemical condition of salt water
	Atmospheric composition and climate regulation	Global climate regulation by reducing greenhouse gas concentrations Micro and regional climate regulation

Table 6. Cultural ecosystem services for the ecosystem approach to aquaculture

Section	Group	Class
Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes in environmental settings Physical use of landscapes and seascapes in environmental settings
	Intellectual and representative interactions	Entertainment Scientific Educational Aesthetic Heritage, cultural
Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes	Spiritual and/or emblematic	Symbolic Sacred and/or religious
	Other cultural outputs	Existence Bequest

Highlights

- IPBES is the conceptual framework of ecosystem services most consistent with the EAA.
- CICES provides the typology of ecosystem services most consistent with the EAA.
- Aquaculture systems potentially provide 41 ecosystem services.