CIRED Working Paper
N° 2020-76 - Janvier 2020

Ecosystem accounting in support of the transition to sustainable societies – *the case for a parsimonious and inclusive measurement of ecosystem condition*

Adrien Comte[1,2], Yann Kervinio[1], Harold Levrel[1,2]
Corresponding author: yann.kervinio@m4x.org

[1]CIRED, AgroParisTech, CIRAD, CNRS, EHESS, Ecole des Ponts ParisTech, Université de Paris-Saclay, Campus du Jardin Tropical, Nogent-sur-Marne, France
[2]Université de Brest, Ifremer, CNRS, UMR6308 AMURE, IUEM, Plouzané, France

Centre international de recherche sur l'environnement et le développement
Unité mixte de recherche CNRS - Ecole des Ponts ParisTech - Cirad - EHESS - AgroParisTech
Site web: www.centre-cired.fr Twitter: @cired8568
Jardin Tropical de Paris -- 45bis, avenue de la Belle Gabrielle 94736 Nogent-sur-Marne Cedex

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Abstract

The development of ecosystem accounting systems at national levels to complete current wealth indicators with robust information on ecosystem degradation or enhancement is a crucial challenge, recognized in international strategies. However, the methodologies remain under development building, at the global level, on an *experimental ecosystem accounting* framework (the SEEA-EEA). Building on this framework and current academic discussions, this article aims at proposing a methodological advance for aligning the SEEA-EEA with the needs of ecosystem management and the principles of strong sustainability. It consists in structuring ecosystem condition measurement into a parsimonious and inclusive set of characteristics, indicators and reference levels with an explicit and inclusive value basis. This sets the grounds for the development of sound and policy-relevant ecosystem monitoring systems and the production of meaningful macro-aggregate indicators of ecosystem degradation at national levels.

Authors

Adrien Comte[1, 2], Yann Kervinio[1], Harold Levrel[1]

[1] CIRED, AgroParisTech, CIRAD, CNRS, EHESS, Ecole des Ponts ParisTech, Université de Paris-Saclay, Campus du Jardin Tropical, 45 bis, avenue de la Belle Gabrielle, 94736 Nogent-sur-Marne, France
[2] Université de Brest, Ifremer, CNRS, UMR6308 AMURE, IUEM, Plouzané, France

Corresponding author: yann.kervinio@m4x.org

Keywords

Environmental accounting, ecosystem accounting, SEEA-EEA, sustainability, ecosystem condition, environmental standards, biodiversity values.

Outline

Introduction

The development of national economic accounts made it possible to monitor a country's economic activity and compare it with other countries. Macro-aggregate indicators that are produced using national accounting, including Gross Domestic Product (GDP), provide influential measures of economic activity and side information that are useful in guiding economic policies.

Accounting systems provide a framework for collecting and organizing statistical information, particularly economic information, in order to build indicators that are useful to a multiplicity of decision-makers (Figure). They structure and integrate existing data to facilitate their access, foster their use and increase their impact. On top of increasing the value of existing data, they also potentially reduce data production costs by exploiting synergies. When defined according to international standards, they also allow for international comparisons.

It is now widely recognized that the national economic accounting framework has to be completed in order to account for social and environmental dimensions of progress (Stiglitz, Sen and Fitoussi, 2009; IPBES, 2019). In particular, the need to integrate the values of ecosystems into national accounting is recognized in the current international strategic biodiversity framework (Aïchi target 2).

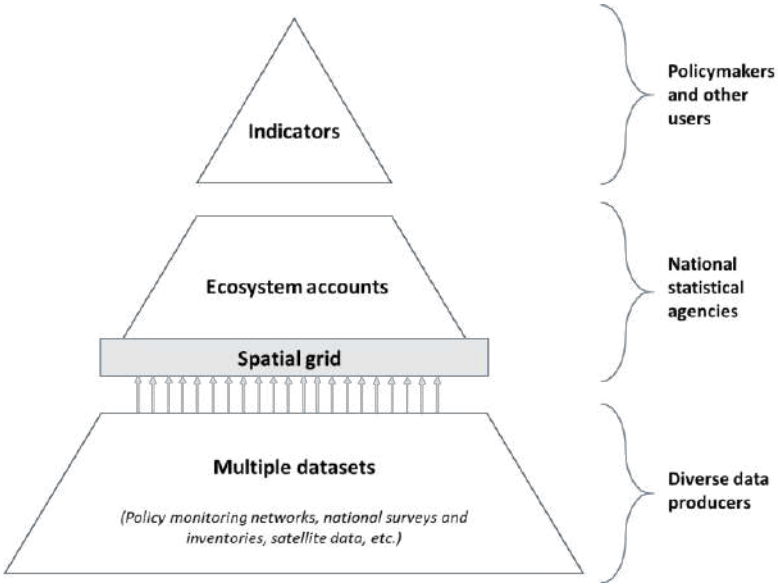

FIGURE 1: ACCOUNTING SYSTEMS, AN INTERFACE BETWEEN DATA AND ITS MULTIPLE USES. *The production of ecosystem accounts is a transition from a situation in which raw data are disparate (poorly interoperable, reflecting different scales) and scattered into an organized and user-oriented information system.*

The System of Environmental and Economic Accounting (SEEA) sets the ground for the development of a comprehensive, comparable, and reliable statistical framework on the environment and its relationship with human activities. It was first released in 1993 to respond to the societal demand for sustainable development. The rationale guiding this effort is that "*[...] individual and societal decisions concerning the use of the*

environment will be better informed through the use of information sets that are developed based on a recognition of the relationship between ecosystems and economic and other human activity." (UNSD, 2014b). After several iterations, the Central Framework of the SEEA (SEEA-CF) was adopted as a statistical standard in its 2012 version by the United Nations Statistics Division (UNSD, 2014a). This statistical framework is aligned with the System of National Accounts 2008[1]. It includes a series of accounts recording stocks and flows of environmental assets and is already implemented in 80 countries (UNCEEEA, 2019).

In the SEEA-CF, ecosystems and biodiversity were simply considered as resources (timber, fisheries, etc.) and not as functional entities. Recognizing the limits of the SEEA central framework with this regard, the UN developed a second framework, the SEEA Experimental Ecosystem Accounts (SEEA-EEA) (UNSD, 2014b). In this framework, a forest is not only assimilated to a resource of wood but also as the provider of multiple services and, potentially as well, with non-use values.

The SEEA-EEA remains experimental and does not holds the status of a statistical standard. Because of the complexity of ecosystem functioning and valuation, a number of theoretical and methodological issues are being discussed by a global community of practice and researchers (Edens and Hein, 2013; Obst et al., 2016). A revision of the SEEA-EEA is scheduled for 2020.

Practically speaking, the SEEA-EEA integrates spatialized data on ecosystems structured into *ecosystem accounting units*. For each of these units, accounts are produced, dealing with:

- the ***extent*** of ecosystem types (e. g. forests, grasslands, etc.)
- the ***condition*** of each ecosystem (e.g. soil carbon levels, etc.)
- ***ecosystem services*** through:
    - ecosystem services supplied by ecosystem type,
    - ecosystem uses and benefits, and
    - monetary accounts reflecting ecosystem services supply and use values
- the value of ***ecosystem assets***
- ***thematic issues*** (biodiversity, soil, carbon and water).

Despite the recognition of the need for ecosystem accounts, it must be noted that the development of ecosystem accounting is skidding. Recuero Virto, Weber and Jeantil (2018) recently showed that natural capital accounts were hardly used in public policy decisions. In the IPBES global assessment of ecosystems, Razzaque, Visseren-Hamakers et al. (2019) also note the great diversity of approaches taken in environmental accounts and conclude that "*there is as yet no evidence of the effectiveness of the use of environmental accounting approaches [and that], as an information instrument, its effectiveness is based on the premise that more information will result in better decision-making* […] *– a premise that is largely unsupported*". Yet, these authors also recognize the potential of well-designed accounts for sustainability.

This current state of affairs may mainly be due to the experimental character of existing environmental accounts and the remaining need to refine the framework and develop convincing case studies. Razzaque, Visseren-Hamakers et al. (2019)

[1] European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, United Nations and World Bank (2009).

emphasize that "*environmental accounting may be helpful as a tool for the facilitation of dialogue on the diverse values of nature and biodiversity*" and that, "*in order to enable this role, it is important that [environmental accounting] uses a broad perspective that includes non-economic values and that it employs a participatory approach so that relevant stakeholders can contribute to the definition and identification of indicators for nature, ecosystem services, environmental assets, and natural capital*". Recuero Virto, Weber and Jeantil (2018) also emphasize that developing robust and policy relevant ecosystem accounts would require to strengthen political support, secure the resources needed, tackle data scarcity, and refine concepts.

In connection with undergoing developments of the SEEA-EEA[2] and current academic discussions, this article intends to contribute to the development of methodologies along these lines. It makes the case for a measurement of ecosystem condition structured into a parsimonious and inclusive set of indicators focused on categories of management issues (i.e. managing conservation, uses and risks) and on the motivation of reference levels for the assessment of ecosystem degradation.

The structure of this article follows the SEEA-EEA approach for measuring ecosystem condition[3]: the first section of this paper is devoted to the construction of a set of relevant key characteristics and indicators; the second section is devoted to the motivation of reference condition levels on these characteristics and indicators. Each section is concluded by a discussion of these approaches.

1 – The selection of characteristics and indicators

In the SEEA-EEA, ecosystem condition is defined as "*the overall quality of an ecosystem asset in terms of its characteristics*" (UNSD, 2014b, § 2.35). It is recognized that "*recording the changes in condition of multiple ecosystem assets within a country (or sub-national region) is a fundamental ambition of ecosystem accounting*" (UNSD, 2017). However, determining an appropriate set of characteristics and associated indicators remains an open issue, recognized as "*a particularly important task for testing in ecosystem accounting*" (UNSD, 2017). This section focuses on this specific issue[4].

Defining ecosystem condition amounts to define a set of relevant characteristics and associated indicators[5]. Given the complex character of ecosystems, this resulted in many proposals[6] with most of them relying on an implicit selection process lacking a clear underlying value-basis (Failing and Gregory, 2003[7]; Bal et al., 2018).

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[2] This includes the SEEA-EEA technical recommendations (UNSD, 2017) but also the many contributions to the SEEA-EEA revision process.

[3] SEEA-EEA (UNSD, 2014b, §4.10): *"Measures of ecosystem condition are compiled in two stages. In the first stage, a set of relevant key characteristics such as water, soil, vegetation, biodiversity, carbon, nutrient flows are selected and various indicators concerning these characteristics are chosen. In the second stage, the indicators are related to a reference condition."*

[4] The issues of aggregation and reference level are two other issues related to the measurement of ecosystem condition (UNSD, 2017). We do not discuss the former. The latter is discussed in the next subsection.

[5] UNSD, 2014b, §4.66.

[6] See Maes et al. (2019) for a recent review.

[7] *"The mistakes relate to a failure to clarify the values-basis for indicator selection and a failure to integrate science and values to design indicators that are concise, relevant and meaningful to decision makers. The combined effects of these ten mistakes include inconsistent and indefensible on-ground management strategies and hidden trade-offs at a policy level."*

In this section, a selection process following a systematic approach using key concepts from structured decision making is developed (Bal et al., 2018). An explicit value-basis[8] which allows to justify the selection of an inclusive and parsimonious set of characteristics of interest for ecosystem monitoring and accounting at the national level is needed. Such a basis is essential to steer progress as it clarifies the relevant arguments supporting the consideration of a given characteristic. It opens up perspectives for consistent argument and steady progress in the definition and measurement of ecosystem condition on transparent and explicit grounds.

1.1 – Motivation and overall structure

Approaches for the selection of indicators of biodiversity and ecosystem condition are abundant in the academic literature (Hayes et al., 2015; Maes et al., 2014; Maes et al., 2019). All these approaches have limitations. Heink and Kowarik (2010) emphasize the diversity of criteria used to assess the quality of the selection process for biodiversity indicators and further remark that, in practice, the suitability of the selected indicators remains sparsely tested in light of these criteria. In a review of existing indicator selection processes, Niemeijer and De Groot (2008) also emphasize limitations regarding the transparency of the processes underlying indicator selection. In particular, they notice that existing criteria are focused on individual indicators while criteria to assess the relevance of a set of indicators are still lacking.

One common limitation deals with the elucidation of the values underlying the selection process. In many cases, this underlying value basis is absent or remain implicit. For instance, essential biodiversity variables have been established on the basis of statistical methods (Pereira et al., 2013; Geijzendorffer et al., 2016). The process proposed by Niemeijer and De Groot (2008) mostly relate to the DPSIR framework, which also leaves the value-basis underlying the selection of characteristics and indicators largely implicit. In other cases, the underlying value basis is limited. For instance, Vaissière et al. (2014) select a parsimonious set of ecological indicators tied to a restricted set of ecosystem services, hence overlooking some non-use and other use values.

In the context of the SEEA-EEA, an emphasis is put on the need for *scientific validity*, broadly understood as meeting usual quality standards for statistical information[9]. Aside from *timeliness*, *accuracy*, *accessibility* and the *quality of the institutional environment* in which the data are compiled, criteria of *relevance*, *coherence* and *interpretability* of the information collected are deemed essential in the context of ecosystem accounting[10]. However, these criteria are left to a peer review and accreditation process without further precisions.

In a recent discussion paper in the context of the SEEA-EEA revision process, Czùcz et al. (2019) propose criteria for the selection of indicators. Following Niemeijer and De Groot (2008), they distinguish criteria related to individual indicators. They propose to assess indicators in isolation on the basis of their *relevance*, *state orientation*,

[8] Here we mean value as the relevant reasons and information for guiding action, as the rest of the section will illustrate. This seeks to be consistent with the conclusions arising from the IPBES (2016) and subsequent work.
[9] UNSD, 2014b, § 4.68.
[10] UNSD, 2014b, § 2.210.

framework conformity, *spatial consistency*, *temporal consistency*, *feasibility*, *quantitativeness*, *reliability*, *normativity* and *simplicity*. They also propose to assess the set of indicators on the basis of its *parsimony* and *data gaps*[11]. This set of requirement is not comprehensive. Again, this attempt at clarifying criteria for indicator selection does not explicit the set of values that underlie this selection, its underlying value-basis. This limits the relevance of the set of indicators for ecosystem management.

On top of the *credibility*[12] and the *salience*[13] of information and knowledge, Cash et al. (2003) emphasize the need to ensure their *legitimacy* for an effective transfer into decision. They define legitimacy as "*the perception that the production of information has been respectful of stakeholders' divergent values and beliefs, unbiased in its conduct, and fair in its treatment of opposing views and interests*". In the context of ecosystem accounting, we suggest that this criterion implies that the set of characteristics retained is *inclusive* in the sense that it emerged from the exhaustive and respectful consideration of existing concerns related to ecosystem management. It also implies that the selection of indicators – which may be necessary to maintain a parsimonious and coherent set of characteristics and a reasonable cost of ecosystem monitoring – rely on a fair and transparent arbitration process.

Building on this diagnosis, a list of criteria which motivates our proposal is described in Table 1. The criteria are organized in two groups (depending on whether they apply to indicators individually or to the set of indicators) as well as along the three aspects of credibility, relevance and legitimacy (Cash et al., 2003).

As Cash et al. (2003) emphasized tensions between these three aspects, classifying the set of requirements in these three broad categories necessitates arbitration among criteria. In particular, *feasibility*, which depends on data availability, can be in tension with the need to cover *relevant* information. An excessive focus on the former would harm the other (the so-called *« streetlight effect »*)[14]. In order to emphasize the essential character of *relevance* affirmed in the SEEA-EEA and avoid the dismissal of essential information, we choose to drop *feasibility* as a core requirement for ecosystem condition indicators and to add a requirement of *completeness*. As a result, and as suggested by Czùcz et al. (2019) with their "*data gap*" criterion, maintaining relevant characteristics with missing data would point at data gaps and foster strategic data acquisition[15].

Family	Criterion	Description	Related source
Criteria on individual characteristics and indicators			
Credibility	Validity	The extent to which an indicator represents the issue to be indicated.	Heink et al., 2015
Salience	Policy relevance	The extent to which the information conveyed is aligned with the information needed for ecosystem management.	UNSD, 2014b, Cash et al, 2003 ("salience")
Legitimacy	Explicit value basis	The values underlying the choice of an indicator are explicit and related to the needs of ecosystem management.	Failing and Gregory, 2003; Bal et al., 2018

[11] Understood as conducting aside the identification of missing data among indicators of interest.
[12] "Credibility involves the scientific adequacy of the technical evidence and arguments" (Cash et al., 2003).
[13] "Salience deals with the relevance of the assessment to the needs of decision makers" (Cash et al., 2003).
[14] Cash et al (2003) further illustrates existing tensions between these criteria on specific cases.
[15] A point emphasized by the SEEA-EEA (UNSD 2017, §4.66).

Criteria on the set of characteristics and indicators			
Salience	Parsimony	The set contains a manageable and readable number of indicators.	UNSD, 2017[16] ; Stiglitz, Sen and Fitoussi, 2009
	Completeness	The set covers all the main dimensions of interests, including when data is missing.	Czùc et al., 2019 ("data gaps")
Legitimacy	Inclusiveness	The selection process has been inclusive in its considerations of existing concerns related to ecosystem management and fair in its treatment of opposing views and interests.	(derived from) Cash et al., 2003
	Explicit value basis	Both the process underlying the identification and selection of characteristics and indicators and its justification are explicit.	(derived from) Cash et al., 2003

Table 1 – Proposed core criteria for assessing the quality of the selection of ecosystem condition characteristics and indicators in the context of ecosystem accounting*. This list of criteria is not exhaustive and focuses on the main gaps of current approaches.*

In their technical recommendations about the implementation of the SEEA-EEA, the UNSD sketches three different approaches to ecosystem condition measurement[17]. They differ according to the extent that they leave room to other inputs than the ones provided by ecological science. Although ecosystem condition shall exclusively be composed of biophysical measures, the focus of what is to be monitored in ecosystems cannot adequately be framed without involving additional considerations about the underlying reasons (the "*value basis*").

Given that, "*ultimately, it is the aim of SEEA Experimental Ecosystem Accounting to present a systems-based approach to recording the relationships among ecosystems, the economy and society that is useful for public policymaking and environmental management*"[18], we argue that relevant criteria shall be derived from the needs of ecosystem management. This perspective calls for an approach that clarifies the objectives of ecosystem management and underlying values. Departing from the proposal by Czùcz et al. (2019) to structure ecosystem condition into categories derived from ecological science and elaborating on the third approach proposed in the SEEA-EEA technical recommendations[19], we propose to measure ecosystem condition through a set of biophysical indicators organized in three categories reflecting the distinct values underlying ecosystem management objectives[20]:

- the maintenance of ecosystem overall functionality ("functionality"),
- the conservation of specific features or elements of ecosystems ("heritage"),
- the capacity of ecosystems to sustainably provide goods and services ("capacity").

[16] UNSD, 2017, §4.30.
[17] UNSD, 2017, section 4.2.
[18] UNSD, 2014b, §1.71.
[19] UNSD, 2017, §4.15.
[20] These three sets can be found in multiple policy document. For instance, the EU 2050 biodiversity vision states that *« By 2050, European Union biodiversity and the ecosystem services it provides — its natural capital — are protected, valued and appropriately restored for biodiversity's intrinsic value and for their essential contribution to human wellbeing and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided. »* (our emphasis). See also UNSD (2017, §4.5) or Kervinio and Vergez (2018, in French).

Table 2 below provides an example of how these categories relate to existing management objectives for marine ecosystems as captured by the descriptor of the EU Marine Strategy Framework Directive (MSFD).

Category of objectives	Related MSFD Descriptors
Functionality	Descriptor 2. non-indigenous species Descriptor 4. food webs Descriptor 5. eutrophication Descriptor 6. sea floor integrity[21] Descriptor 7. hydrographical conditions Descriptor 8. contaminants Descriptor 10. marine litter Descriptor 11. energy (including underwater noise)
Heritage	Descriptor 1. biodiversity
Capacity	Descriptor 3. commercial fish species Descriptor 9. contaminants in seafood

Table 2 – Relation of ecosystem characteristics monitored in the context of the MSFD and the three classes proposed.

1.2 – Detailed presentation of the three families of management issues

In this subsection, each of these categories are detailed and both discuss their value-basis and their relation to ecosystem management.

1.2.1 - Ecosystem functionality

Ecosystems are complex and subject to risks of collapse or to irreversible and widespread degradation. As a result, ecosystem management resorts to objectives for controlling such risks and preserving the overall functionality of ecosystems. Ecosystem condition shall contain a set of characteristics and indicators related to this first set of objectives.

The values underlying such holistic objectives call for preserving the functionality of ecosystems *per se*, without a direct and explicit link to a particular set of ecosystem services or non-use values. They reflect our collective attitudes toward risks and uncertainty (Perrings and Pearce, 1994[22]; Rockström et al., 2009[23]; Steffen et al., 2015[24]). Justification thus refers to principles of decision in situations of risk and uncertainty, which emphasize precaution, proportion, diversification, robustness, learning and flexibility. In order to ensure the legitimacy of the underlying values, the choice to

[21] Several aspects of sea floor integrity relate to particular habitats that possess a heritage dimension as well

[22] *"economic instruments required to protect thresholds or discontinuities cannot be motivated by conventional economic objectives, such as the maximization of expected utility or welfare, but must rely on non-economic criteria […] They must be motivated by a judgement about the socially acceptable margin of safety in the exploitation of the natural environment. This is essentially an ethical judgement."*

[23] *"Determining a safe distance involves normative judgements of how societies choose to deal with risk and uncertainty."*

[24] *"Application of the precautionary principle dictates that the planetary boundary is set at the "safe" end of the zone of uncertainty."*

include indicators on risk must be justified on the basis of natural science knowledge but also on the basis of a relevant normative framework. This normative framework can be derived from relevant social science knowledge or directly revealed through an informed and legitimate political process as explained in section 2.1 of this article.

In practice, the management of these risks consists in defining thresholds ("*safe minimum standards*" ; Perrings et al., 1995) on pressure levels or impacts associated with one or cumulated pressures ("control variable") that make it possible to define a "*safe operating space*" for ecosystem management. In the definition of ecosystem condition, the related dimensions of interest are such control variables but also specific intrinsic characteristics of ecosystems reflecting their resilience[25], thus allowing for a lower safe minimum standard.

Examples of ecosystem characteristics that could be derived from such considerations are pressures[26] or stressors such as nutrient concentration in aquatic ecosystems, but also "*spatial features, such as connectivity and landscape configuration*"[27], which underpins ecosystem adaptive capacity and resilience in a context of climate change[28] (Tittensor et al., 2019). Interestingly, it covers most characteristics of interest in the context of marine ecosystem management in Europe (eight in eleven descriptors, see Table 2).

On these grounds, relevant arguments to add or withdraw indicators from the set would be informed by the absolute and relative critical character of risks. Each individual indicator could be improved iteratively in order to get closer to the precise control variable that underpins these risks or to the precise characteristics that underpin ecosystem resilience to cumulated pressures and global change.

1.2.2 – Ecosystem heritage

The SEEA-EEA recognizes that "*an important part of the value of ecosystems from a societal perspective can lie in the non-use values that, in principle, are captured in various cultural services provided by ecosystem assets*" (UNSD, 2017, §6.48). This idea that the ecosystem service framework could adequately reflect these values has recently been challenged in the context of the IPBES (Pascual et al., 2017; Diaz et al., 2018). In the French national ecosystem assessment, the notion of ecosystem service has also been restricted to *instrumental values* which leaves room for an additional category capturing *non-instrumental values*, close to the notion of *natural heritage* (CGDD, 2017).

The definition of such *natural heritage* requires to demonstrate the remarkable character of an element of an ecosystem and to justify a need for its conservation. This potentially requires more detailed knowledge than what non-use values are able to convey and can be motivated by aesthetic, relational, identity, socio-cultural, ethical or deontological values. What matters in this process is to get a clear

[25] UNSD, 2017, §4.27. Carpenter, S.R. et al. (2010) provide examples of such indicators.
[26] Note that this brings a conceptual clarification about why *"the types of indicators to be considered in the measurement of ecosystem condition may include indicators that reflect pressures being exerted on ecosystems"* (UNSD, 2017, §4.9), which is called for by the SEEA-EEA community (UNSD, 2017, §4.69 and 70).
[27] SEEA-EEA (p.72) mentions the interest of monitoring these features.
[28] Krosby et al (2010)

understanding of what makes the values of the characteristics that are to be conserved, which cannot be reduced to an economic issue[29].

It needs to be recognized that some particular characteristics of ecosystems, their functioning and our relationship to them, may require conservation effort despite the absence of a fully characterized ecosystem service[30]. Such consideration could be integrated within the SEEA-EEA by recognizing that the conservation status of such characteristics are of interest and including them in the definition of ecosystem condition.

This would get the definition of ecosystem condition even closer to the practice of environmental management. In the context of marine ecosystem management, examples of such characteristics fall under the first descriptor, called "biodiversity", and also, to some extent, under the sixth descriptor (see Table 2).

1.2.3 - Ecosystem service capacity

Ecosystems support human welfare through the provision of ecosystem goods and services. As a result, ecosystem management also seeks to maintain and foster the capacity of ecosystems to sustainably provide specific goods and services. The need for ecosystem condition to contain a set of characteristics and indicators related to this set of motivations is deemed central in the SEEA-EEA (UNSD, 2014b).

Use values capture the motivations underlying an interest in the capacity of ecosystems to provide goods and services. Use can be direct (for instance through fishing) or indirect (for instance through benefiting from the protection against coastal flooding provided by coral reefs). Ecosystem service are the boundary concept between ecosystems and human welfare.

Following the seminal definition of the Millennium ecosystem assessment (2005), the concept of ecosystem services has been reframed according to different perspectives, emphasizing either the *benefits* resulting from ecosystem use, or the *ecosystem functions*, understood as the biophysical features of ecosystems being used. Some conceptualizations explicitly locate the ecosystem service as an intermediate object. Among them, the "cascade model" proposed by Potschin and Haines-Young (2017) and the associated *Common International Classification of Ecosystem Services* (CICES)[31] provide the framework used in the SEEA-EEA (UNSD, 2014, §3.41). In the SEEA-EEA, ecosystem services are defined in this perspective as "*the* contributions *of ecosystems to benefits used in economic and other human activity*" (our emphasis), where *contributions* are further operationalized as a fraction of a benefit which can be attributed to ecosystems. In the presence of non-linear relationships between natural and other forms of capital, this attribution is left to conventional allocation rules[32].

[29] This does not mean that these dimensions are irrelevant to the SEEA-EEA. Indeed, the SEEA-EEA acknowledges the need to account for them. Besides, once a need for conservation recognized the implementation of conservation does have economic consequences which can be monitored.
[30] See next sub-section for a clarification of what is meant by « a fully characterized ecosystem service ».
[31] Haines-Young and Potschin (2018).
[32] Consider, for instance, the case of perfect complementarity. Natural capital N and man-made capital H can be combined to produce a benefit B. Perfect complementarity is represented by the relation B = min(N,H). How is the benefit to be shared in this

Interestingly, Costanza et al. (2017) suggest that such a cascade model may be "*an oversimplification of a complex reality and an unnecessary complication of what is essentially a very straightforward definition*". They call for a simpler definition of the service focused on the *benefit* resulting from the complex interactions and feedbacks required among built, human, social, and natural capital. With a similar intention, the French national ecosystem assessment (the EFESE program), defines ecosystem services as a *relationship* between a benefit that results from the use of an ecosystem function[33]. A *benefit* can be measured by an indicator reflecting an increase of human well-being on one of its dimensions[34]. An *ecosystem function* can be measured by a single or a combination of indicators reflecting the state of biophysical components and processes, where this combination eventually captures the *capacity* of the ecosystem to sustainably provide the benefit[35]. This latter definition of an ecosystem service captures in essence the boundary character of the ecosystem service concept and, hence, the possibility to describe it from different perspectives. More importantly, it follows Costanza et al. (2017) in discarding the ambition to measure ecosystem services as an additional object and in driving the focus on ecosystem functions, use, benefits, and their relationships (see Figure 2).

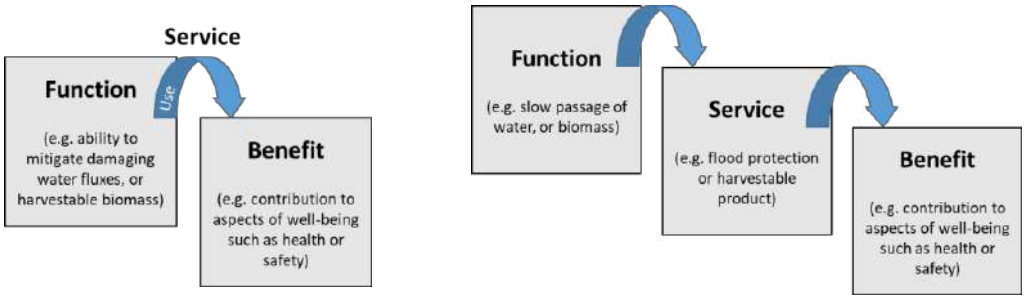

***Figure 2 – Two different conceptualizations of an ecosystem service**: on the left, the conceptualization of the EFESE program (CGDD, 2017), on the right the cascade model (adapted from Costanza et al., 2017). Boxes are objects to be defined and measured, while arrows represent dependencies between objects.*

As a boundary concept, the notion of *ecosystem service capacity* also leads to different perspectives. In a "stock" perspective, some define it on the basis of the biophysical properties of ecosystem assets (e.g. Villamagna, Angermeier and Bennett, 2013). In this perspective, the units of measurement are biophysical. In a "flow" perspective, others focus on the potential flows of benefits to be derived from

situation? If natural capital is absent, B is null, calling for the allocation of the whole benefit to natural capital. But the same occurs in the absence of man-made capital, calling for the whole allocation to this form of capital. Different fractions could be proposed in between, all relying at least implicitly on conventional rules with more or less relevant properties (cooperative game theory provides characterizations of such allocation rules motivated on the basis of their properties). Interestingly, any rule which would not allocate the full benefit to nature at this stage would fail to capture the full cost of ecosystem degradation. This meets the distinction between attributional and consequential approaches in life-cycle analysis where only the latter are more suited to support decision (see e.g. EC-JRC, 2010, p. 70). This question the relevance of such an approach based on contributions and the allocation of shares of a benefit to different forms of capital.

[33] Commissariat général au développement durable, 2017.

[34] In the EFESE, these dimensions are material living standards, health, safety, quality of the living environment, quality of social relationships and socio-economic inequalities. These were inspired by the dimensions identified in a report on social progress ordered by the French government, which considered: material living standards (income, consumption and wealth); health; education; personal activities including work, political voice and governance, social connections and relationships, environment (present and future conditions) and insecurity, of an economic as well as a physical nature (Stiglitz, Sen and Fitoussi, 2009, p.16).

[35] Considered, at this level of generality, as tantamount to *potential ecosystem services* or *ecosystem service supply*.

ecosystem assets (e.g. La Notte, Vallecillo and Maes, 2019). In this perspective the units of measurement are related to dimensions of human welfare.

In the SEEA-EEA, *capacity* is distinguished from the notion of the *condition* and defined in general terms as "*the ability of a given ecosystem asset to sustainably generate a set of ecosystem services into the future*" (UNSD, 2014b, glossary). It is not defined from a measurement perspective but rather as a tool to link ecosystem assets (extent and condition) with ecosystem service provision. Therefore, there is no intention to build specific ecosystem capacity accounts (UNSD, 2017). It is recognized that "*there remain significant challenges in understanding the links between measures of capacity for individual services and overall ecosystem condition*" (UNSD, 2017, §7.42).

Considering this, we propose to include the notion of *ecosystem service capacity*, understood as a "stock", in the definition of ecosystem condition as summarized in Figure 3. In such a perspective, the set of condition characteristics and indicators would explicitly comprise a subset of characteristics and indicators that reflect the capacity of the ecosystem to sustainably provide specific services.

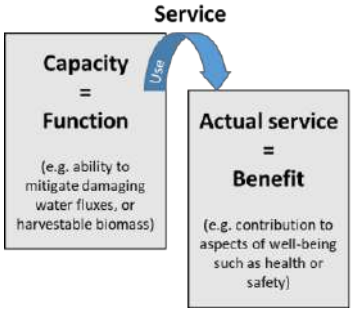

Figure 3 – The capacity concept and its relationship with sustainable provision of ecosystem services

In the context of marine ecosystem management, examples of such characteristics are the state of sustainably harvested commercial fish stocks or contaminants in seafood (see Table 2). While currently focused on productive activities, this set could be completed with the characteristics that underpins other major ecosystem services, such as protection against natural hazard or recreational activities.

Such a proposal is pragmatic. It better matches the practice of environmental management as the description of the ecological condition of marine ecosystem already include such indicators (Table 2). It also avoids unnecessary layers of complexity and resolves the pending issue of linking ecosystem capacity to degradation (UNSD, 2017, §7.3.2). It also matches the recommendation of considering "*the degree to which the indicator can be linked to measures of potential ecosystem services supply*"[36] in the selection of condition indicators while recognizing the need to complement this perspective with the functionality and heritage dimensions of ecosystems, therefore avoiding the pitfall of a reductive value basis (Ang and Van Passel, 2012).

[36] UNSD, 2017, §4.31.

1.3 - Discussion

A measure of ecosystem condition built on this framework is likely to meet all the quality criteria for the selection of indicators identified in Table 1.

Regarding the selection of individual ecosystem condition characteristics and indicators, first, by making the underlying value explicit, this framework enables a discussion about how a given indicator adequately reflects this value and lays the ground for reinforcing *its policy relevance*. The values underlying the choice of a given indicator are discussed along the lines relevant to each category, involving knowledge and expertise from social science. This clarification of the reasons that drive the choice of a given characteristic also strengthens the validity of indicators as the issue to be indicated is explicit.

Regarding the design of the set of characteristics, the framework facilitates the achievement of the quality criteria identified, although their satisfaction depend on the selection process. For instance, *inclusiveness* and *completeness* are facilitated by the capacity of this framework to reflect the diverse concerns observed in the context of ecosystem management. The selection of characteristics also requires a fair and transparent selection process with sufficient legitimacy for establishing relative priorities at the national scale. Such a process may explain collective and shared values through stakeholder engagement and collective valuation methods, including innovative methods intended to capture emerging societal norms and values (see e.g. Kenter et al, 2015; Everard, Reed and Kenter, 2016).

Although we did not include it in the set of core criteria, the *feasibility* of the measurement would also be eased by the explicit relationship to ecosystem management as it requires data that already exists or for which policy makers have a keen interest. This way, a virtuous cycle would emerge in which the statistical system would improve the informational basis of ecosystem management while policymakers would have an interest to take an active role in the production of data suited to fit in the accounts.

The SEEA-EEA insists that "*over time the accounts can be broadened in scope and filled with a larger range of indicators*". This framework enables such a process, articulating inputs from policymakers and experts in diverse disciplines including human and social sciences, each on their domain of legitimacy. Experiments along these lines would allow to strengthen a set of common indicators and identify additional indicators relevant at national or sub-national scales.

Based on such a framework, the SEEA-EEA community could propose a definition of ecosystem condition based on a list of *required* and *optional* characteristics and indicators. *Required* characteristics would allow the derivation of indicators related with the achievement of international objectives and be subject of international comparisons (e.g. soil degradation for the SDGs). *Optional* characteristics would allow the collection of relevant information in national or sub-national contexts.

As this framework is related to explicit values and stakes, it provides firm grounds to two challenging tasks:

- the setting of *reference levels* on each of the characteristics retained;
- the *integration*[37] of indicators across issues to produce aggregate measures of ecosystem condition.

The next section deals with an approach for a meaningful and policy-relevant setting of reference levels.

2 – The definition of reference levels

In the SEEA-EEA, reference condition is defined as a baseline against which indicators are to be assessed to derive a measure of condition that can be scientifically compared across ecosystem characteristics and types over time. Beyond this, the definition is ambiguous. It is recognized that "*selecting a reference condition applicable to all ecosystems in a country […] is a major challenge both from a conceptual and from a data perspective*" (SEEA-EEA, 2017, §4.40.v) and that "*investigating different approaches to determining reference conditions for the assessment of ecosystem condition, based on practical experience in countries*" is needed (SEEA-EEA, 2014b, §A.5).

In the current SEEA-EEA approach to reference condition, the main challenges are:

- whether reference condition apply to individual or aggregate measures of ecosystem condition, which was clarified in Keith et al. (2019) by distinguishing *reference levels* (applying to individual indicators) from *reference condition* (applying to aggregate condition indicator);
- whether reference levels are to be derived exclusively from natural science knowledge or whether they should account for social preferences;
- what values and objectives shall guide the setting of reference levels.

Resulting tensions can be found in the 2017 technical recommendation guide where it is said that "*a clear distinction has to be made between reference and target condition*" (§4.53) while acknowledging that "*[…] generally, it will be necessary to establish non-natural reference conditions perhaps based on a historical baseline or a condition prescribed in policies*" (§4.71). Another tension appears between the suggestion to adopt a fictitious natural state[38] or the condition at an arbitrary point in time as a reference, despite limited policy relevance, while acknowledging elsewhere that "*it would be expected that information on the actual and reference condition […] would be useful input to a discussion of target conditions*" (§4.53).

In the SEEA-EEA, it is recognized that "*as regards the [definition of reference levels], there are a number of options available for determining a reference condition, each with different conceptual underpinnings*"[39]. In their recent contribution the SEEA-EEA revision process, Keith et al. (2019) note that "*reference level and condition can refer to a natural state, a desired state, a prescribed or standard state, a historical state or*

[37] In the sense defined in Borja et al., 2014.

[38] *"The reference condition of species can refer to any time period, but ideally it should refer to an ecosystem subject to minimal human influence. While such a baseline can be difficult to establish, it does have the distinct advantage of allowing the relative abundances of different species, and of species within different ecosystems within one country and in different countries, to be compared."* (SEEA-EEA, 2017, §4.128)

[39] UNSD, 2014b, §4.15.

a point in time. These serve different purposes and both purpose and choice of reference level must be stated explicitly".

This section proposes and motivates an approach for the specification of *reference levels* focused on prescribed or desired levels consistent with the overall objectives of informing policy. The application of this approach to the categories of ecosystem condition identified in the former section is then discussed.

2.1 – The case for a policy relevant definition of reference levels

The SEEA-EEA states that "*ecosystem accounting does not include the use of target conditions*". However, Keith et al. (2019) suggest that "*[employing target or desired condition as a reference in ecosystem accounting] may be beneficial for policy applications of ecosystem accounting, but [that] the scientific objectivity of the process would need careful consideration*". Yet, they carefully conclude that "*there are potential problems of desired states being influenced by policy objectives, and themselves changing over time [and recommend] that this role be reconsidered and possibly that desired states be used outside the condition accounts in the process of analysis of the condition metrics as part of applications*". In this subsection, employing target or desired levels for specifying reference levels is detailed. Such an approach could conciliate policy-relevance with scientific objectivity and stability.

Adopting a natural state, a historical state or a point in time would disregard social preferences and feature limited relevance. Setting a reference level at the beginning of the accounting period would lead to a shifting baseline, that would hide changes in ecosystem conditions across accounting periods and prevent comparisons between ecosystem assets (an ambition of ecosystem accounting). The same would apply to comparisons carried out at a fixed point in time. Setting a pre-industrial benchmarks as a reference would match some scientific[40] and policy[41] practices. However, it would raise feasibility issues for ecosystems that have been shaped by human interventions for a very long time and would not acknowledge the need to include relevant social factors – a need reflected in latest policy practices[42]. Besides, such an approach would have limited relevance in the context of a rapidly changing climate.

While the SEEA-EEA (UNSD, 2014b) and Keith et al. (2019) discard the idea of adopting policy targets as reference levels, a careful integration of such targets is feasible and meaningful. In this perspective, Ekins and Usubiaga (2019) recently introduced a useful gradation between environmental *limits*, *standards* and *targets* according to the extent of their normative content. This distinction is promising as they allow to sort out potential reference levels according to the relative weight of the normative and scientific considerations that govern their design. The idea is to strike a balance between a stable science-based reference and a reference which is informed by collective values. This latter side requires seeking a sufficient level of political legitimacy to interpret the observed target as a reflection of collective preference (e.g. public spending is interpreted as such in national economic accounting). Building on the

[40] Instances are the definitions of threatened species or the biodiversity intactness index (Mace, 2005).
[41] Instances are the definition of *Good ecological status* in the European Union Water Framework Directive.
[42] For instance, the EU Marine strategy framework directive does not adopt historical or unperturbed reference levels as a baseline.

work of Ekins and Usbiaga (2019), a slight redefinition of these terms is proposed which is pragmatic and specific to the framework developed here.

Environmental limits are thresholds which indicate the likely occurrence of an undesirable phenomenon (regime shifts, species extinction, slow but irreversible degradation, etc.). Although they may require some degree of normative judgment[43], environmental limits are essentially informed by scientific knowledge. Using such limits as reference levels in ecosystem accounting would secure scientific objectivity but raises several issues. First, it would drastically restrict the possibilities to define policy relevant reference levels. For instance, ecological science produces information on the probabilities of regime shift, sometimes with probability distributions, but the design of policy relevant "safe minimum standards" involves additional considerations, related to our collective attitudes toward risks and uncertainty, the costs of mitigating these risks and trade-offs with other objectives (Farmer and Randall, 1998; Rockström et al, 2009). In a nutshell, environmental limits need complement.

Environmental standards complement environmental limits with additional normatively-relevant considerations[44] and knowledge about our collective values. Collective values cover collective attitudes toward risks and uncertainty, collective attachments, or shared values about what a legitimate political process is in a given context.

Environmental targets are the observed policy objectives resulting from the political process. They can be observed in existing regulations (law, political strategies, etc.). They are a keystone of current environmental policies[45] and may, to some extent, represent institutionalized shared values. Yet, their use as reference levels in ecosystem accounting requires careful attention. First, existing targets may result from irrelevant political factors (corruption, client politics, etc.), which would compromise their legitimacy[46]. They may also feature inconsistencies[47], requiring to decide which of two contradicting objectives better reflects collective preferences, or dismiss existing scientific knowledge. This enters in sharp tension with the ambition of independence and scientific objectivity of ecosystem accounting[48]. Second, existing targets may be fluctuating or rapidly changing. Changes that feature genuine evolutions in collective preferences could be integrated in ecosystem accounting just as public spending or changes in prices are in economic accounting. However, some smoothing could be proposed, for instance by proposing some delay intended to test for the reality of a change in underlying collective preferences. In a nutshell, environmental targets need laundering. This would involve a discussion about the legitimacy of the underlying political process and the impacts of irrelevant political factors on observed targets.

Environmental standards are a good candidate for the design of reference levels in the context of ecosystem accounting. While *environmental targets* are the exclusive outcome of a political process (which may or may not involve scientific communities)

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43 Ekins and Usubiaga (2019) emphasize the normative judgement involved in defining what constitutes an undesired consequence.

44 For instance, the costs and trade-offs involved in the choice of given level.

45 Climate mitigation objectives, good environmental status in the context of the Water Framework and Marine Strategy Framework Directives, no net loss objectives in the context of conservation policies, etc.

46 Ekins and Usubiaga (2019) emphasize that *"targets […] reflect people's desires to the extent to which policies are aligned with social preferences"*.

47 One example of this is the persistent widespread existence of biodiversity harmful subsidies.

48 As Keith et al. (2019) warns, *"if a target or desired condition were employed as a reference in ecosystem accounting, […] the scientific objectivity of the process would need careful consideration"*.

and *environmental limit* are primarily based on scientific considerations (essentially from natural sciences), *environmental standards* integrate relevant social science knowledge[49] and articulate the involvement of the scientific and political communities each on their domain of legitimacy (Figure 4).

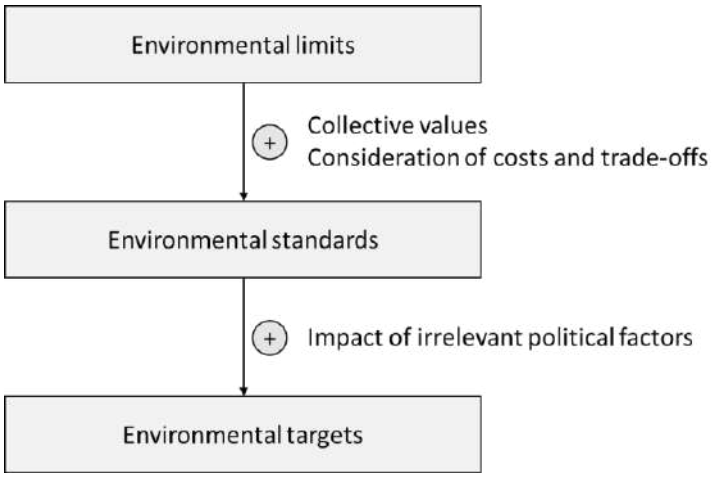

Figure 4 - Relationships between environmental limits, standards and targets, adapted from Ekins and Usubiaga (2019). *The figure enlarges the consideration leading from environmental limits to standard from "risk and uncertainties" to "collective values" (that include our collective attitudes toward risks and uncertainties, but also other shared values as discussed in section 2.2.2 hereafter). It also requires environmental standards to take into consideration costs and trade-offs. The realization of disproportionate costs may indeed be a relevant basis to revise a standard, while antagonisms between dimensions can require to mutually adjust the standard. As such adjustments are required to ensure a set of mutually consistent standards, they are retained in the definition of the standard. Other irrelevant political factors are what distinguishes the environmental standard from observed political objectives.*

Such reference levels may apply to ecosystem extent or ecosystem conditions (or both)[50]. One example of a reference level set on ecosystem extent is the no-net-loss of wetlands policy in the United States, where the extent of wetlands is not allowed to decrease, and compensation (often in the form of restoration) has to take place if projects impact the extent of a wetland (Bendor, 2009).

To conclude, reference levels are not interpreted here as a theoretically or empirically-determined historical state of a pristine environment, but as the scientifically and politically motivated collectively desired state of the environment. The next subsection shows how to operationalize this perspective for the three categories of ecosystem condition formerly identified.

[49] The SEEA-EEA suffers from too sharp a separation between natural and social sciences, a flaw already identified as an impediment for progress in human-nature relationship (see e.g. Jetzkowitz et al., 2017). A better integration of both source of knowledge suggests a way toward the development of a promising and policy-relevant set of ecosystem accounts.

[50] « There may also be some overlap between measures of ecosystem extent and ecosystem condition in the sense that, at certain scales of analysis, changes in extent may also be considered to be encompassed by the measurement of overall changes in ecosystem condition. At the same time, it is not considered that measures of changes in ecosystem extent can be used as a substitute for measuring changes in ecosystem condition" (UNSD, 2017, §4.23).

2.2 – Application to the three families of management issues

The following discussion applies reference levels to the three categories of ecosystem condition indicators formerly identified. For each of these families, building environmental standards by complementing environmental limits with knowledge on collective values, costs and trade-offs with other objectives is discussed.

2.2.1 – Setting reference levels for ecosystem functionality

The existence of thresholds of regime shifts in ecosystem functioning and related risks of massive perturbation (e.g. eutrophication) or slow irreversible changes in ecosystem functioning (e.g. land degradation; Lenton et al., 2008; 2019) form the basis of *environmental limits*.

Facing these risks, "*safe minimum standards*" complement this knowledge with our collective attitudes toward risks and uncertainties (Perrings and Pearce, 1994; Rockström, 2009), the involvement of non-disproportionate costs in managing them (Crowards, 1998; Farmer and Randall, 1998) and potential trade-offs with competing objectives (for instance, the willingness to conserve native tree species and traditional landscapes in cases where adaptation to climate change calls for a shift to non-native tree species).

In practice, reference levels correspond to limits on cumulated pressures or to some intrinsic characteristics of an ecosystem which guarantee that its functioning is kept within safe boundaries. They define a set of conditions to be interpreted as a set of necessary and sufficient conditions for the maintenance of the overall functionality of an ecosystem.

2.2.2 – Setting reference levels for ecosystem heritage

The heritage dimension of ecosystems relies on non-use values, involving specific features of ecosystems and societies' relationships to them (e.g. traditional landscapes). The motives underlying societies' attachment to these features and relationships stems from complex considerations which are not appropriately reflected by the usual non-use categories of the total economic value framework (altruistic, bequest and existence values; Johansson-Stenman, 1998). As a consequence, Crowards (1997) also suggests the use of the "safe minimum standard" approach.

In this framework, *environmental limits* reflect the levels on a set of conditions under which the risk of extinction or denaturation of a feature of interest exceeds some level. *Environmental standards* reflect the conservation status of a specific feature of interest. It stems both from a collective willingness to preserve a specific feature (e.g. as reflected in the EU lists of species of Community interest) and an evaluation of the risk of extinction or denaturation (e.g. through the regulatory assessment of conservation status or the Red list approach). In principle, environmental standards would also involve our collective attitudes toward risks and uncertainties, the involvement of non-disproportionate costs in managing them and potential trade-offs with competing objectives.

In practice, reference levels could refer to existing assessments of species extinction risks and the objectives of conservation policies. It is interesting to note that, according to some frameworks, the monitoring required would exceed the mere monitoring of a population but would encompass measures of the threats involved, therefore extending the measures of interest to pressures or other relevant information.

2.2.3 – Setting reference levels for ecosystem capacity

The rationale for setting reference levels in ecosystem service capacity is questionable. In principle, the greater the capacity of an ecosystem to provide a service, the better. Yet, conflicting uses are pervasive in ecosystem management (How et al., 2014; Rodriguez et al., 2006). In this perspective, *environmental limits* would be the production boundaries of ecosystem (i.e. sustainability) and *environmental standards* would reflect trade-offs between uses, involving diverse economic and social considerations.

One example of such reference levels is the *Maximum sustainable yield* (MSY) norm. Such a reference level does not reflect a risk of collapse of the fishery but the maximum level of yearly catches which can be sustained in a fishery therefore trading-off present and future uses. Another example is the sanitary levels required for the suitability of marine waters for aquaculture or bathing activities. Up to some limit, these norms can be interpreted as a reversible arbitrage between these activities and the sink function of ecosystems used by wastewater infrastructures.

2.3 – Application to the marine environment in France

In the context of the marine environment in France, it is suggested to use *environmental targets* resulting from the second implementation cycle of the Marine strategy framework directive. Given the conduct of the policy, these targets can reasonably be deemed immune to the impact of irrelevant political factors. Furthermore, this process did involve the scientific community and an extensive stakeholder consultation.

In the absence of contradiction, this set of targets will be considered as a relevant set of *environmental standards* and retained as references levels for the accounts. Further discussion could lead to alter these levels or complement the set of dimensions to include additional significant stakes.

2.4 – Discussion

The former discussion shows that setting reference levels rely on distinct rationales depending on the family of issues covered. For *ecosystem functionality* and *heritage*, these levels draw the contours of *critical natural capital* (Ekins et al., 2003). For *ecosystem service capacity*, they can be rationalized by trade-offs between competing uses and an optimization of ecosystem service supply, conciliating conflicting uses, within safe and sustainable boundaries.

Note that the answer to the question "*are we sustainable in our relation to ecosystems?*" can already be answered at this stage. A simple count of the number of sustainability conditions met in the functionality and heritage categories would already be informative. The SGAP developed by Ekins et al. (2003) further proposes to report the time to sustainability based on an extrapolation of past trends for each indicator.

Complications arise when seeking an aggregate sustainability indicator. As emphasized in the SEEA-EEA (2017), "*forming the overall measures requires the use of assumptions on the relative importance of each characteristic and correlations among them*". Examples are the Ecological condition index (Nel and Driver, 2015, SEEA-EEA, 2017, box 4.1), the Norwegian Nature Index (Certain et al, 2011), the Ecosystem capability unit (Weber, 2014), or notion of *Good ecological status* of the EU Water framework directive for freshwater ecosystems (this last example relying on a one-out-all-out aggregation scheme). However, most of these composite indicators fail to justify the aggregation process on the basis of transparent and explicit objectives and underlying values. As a result, the index produced may have limited decision relevance or, worse, convey misleading information with regards to collective preferences. This may for instance be the case when relying on an implicit or poorly justified weighting scheme[51]. At present, few attempts have built composite indicators on the basis of a transparent and policy relevant normative framework. The framework developed in this article would allow developments along these lines.

The design of a monetary macro-aggregate indicator of the maintenance and restoration costs involved in bridging the observed gaps is a way forward (Vanoli 1995; Bartelmus, 2009; Ekins and Usubiaga, 2019). The effort taken to structure condition accounts and set reference levels based on environmental standards provide a firm basis for accounting for the costs of ecosystem degradation on the basis of maintenance and restoration costs.

Conclusion

The structuration of the ecosystem condition account builds an inclusive set of objectives that captures the two main underlying rationales for strong sustainability approaches (Stern, 1997): the physical non-substitutability between different forms of capital (which motivates a careful monitoring of functionality) and the incommensurability of certain non-use values (which motivates a careful monitoring of the conservation of natural heritage). Figure 5 shows how this proposal structures an inclusive monitoring of ecosystem condition.

[51] See for instance Decancq and Lugo's (2013, section 4.2) discussion of the *Human development index*.

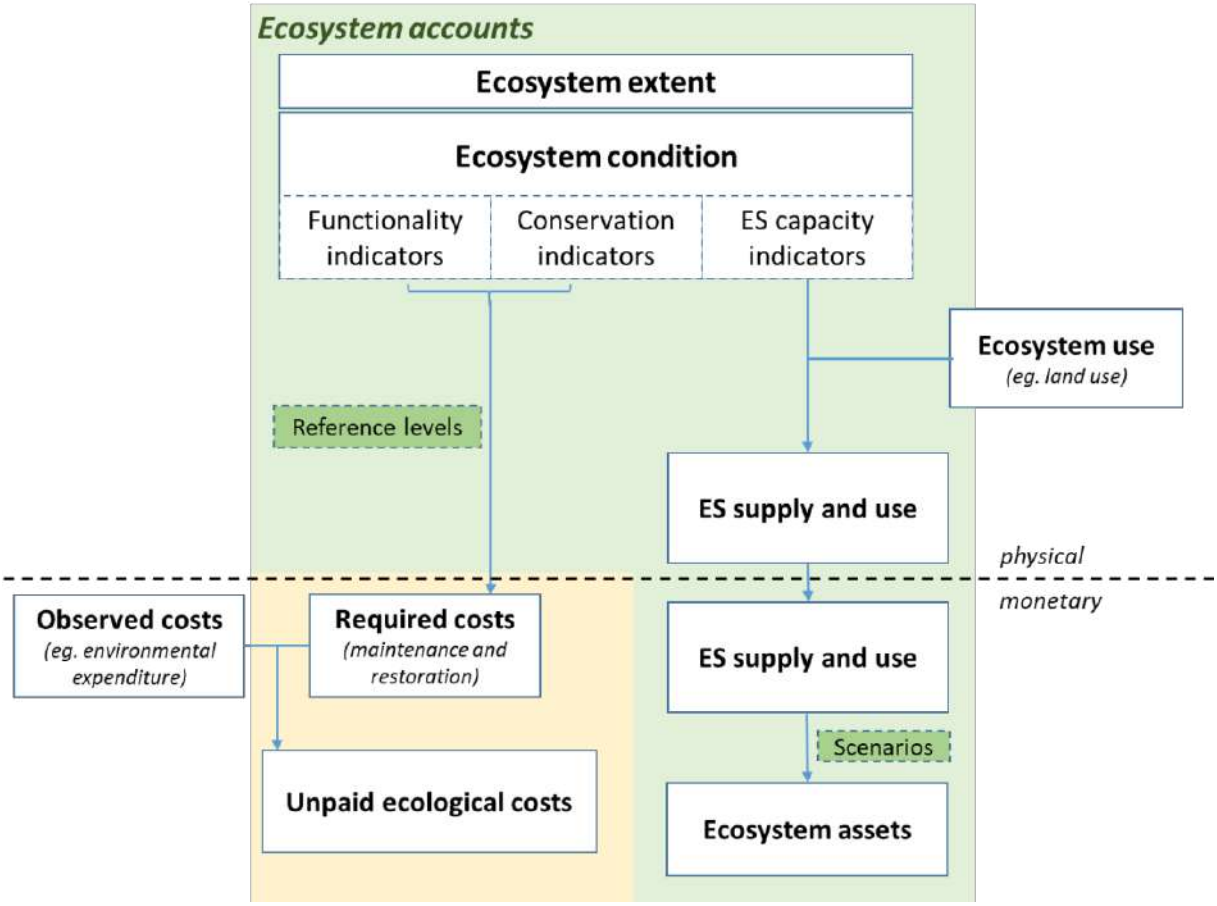

Figure 5 - Structure of ecosystem accounts discussed in this paper. *White boxes represent accounts, distinguishing between physical accounts on the top and monetary accounts below. Within the ecosystem accounts, the standard components of the SEEA-EEA are presented on the green area. Some potentially useful side accounts are represented outside the boundary: ecosystem use accounts may include land use accounts; observed costs accounts may include environmental expenditures accounts as prescribed by the SEEA-CF. The comparison of functionality and conservation indicators with reference levels allows to derive a meaningful measure of the costs required for the maintenance and restoration of natural capital. Net of incurred cost they allow to derive a measure of unpaid ecological costs (Vanoli, 1995, 2015), which may be structured in dedicated accounts.*

The design of an inclusive ecosystem accounting system needs to overcome the opposition between the weak and strong sustainability paradigms organized around the question of whether natural capital can be substituted with other forms of capital[52], which makes little sense at this level of generality. There is also a need to overcome "*the reductive nature of focusing only on a stock–flow framework in which a natural-capital stock produces ecosystem services*" (Ang and Van Passel, 2012). However, a pure biophysical perspective disconnected from underlying values would fail to provide the critical information needed for the transition to sustainable societies.

[52] Although, the impossibility of getting a general answer to this question is generally acknowledged, debates often frame the problem as follows: *"In taking an economic approach to the problem, the key choice is whether one believes that natural capital […] should be afforded special protection, or whether it can be substituted by other forms of capital, especially produced capital. This is the* choice *between weak sustainability and strong sustainability"* (Dietz and Neumayer, 2007, our emphasis)

Instead of opposing the paradigms, the dual approach to ecosystem management proposed in this article articulates the weak and strong sustainability paradigms on their respective domains of relevance. Such a framework complements the current structure of the SEEA-EEA and increases its inclusive monitoring of ecosystems and their interactions with society and the economy. In particular, it allows to derive meaningful monetary values according to the two main paradigms for the measurement of ecosystem degradation (OECD, 2018).

In a first approach related to the strong sustainability paradigm, the value of ecosystem asset is inferred implicitly by the shadow value of the reference levels imposed on the condition of ecosystems. This approach is most relevant to functionality and heritage dimensions of ecosystem conditions. When these constraints are taken as the outcome of an informed and legitimate collective decision process, they can be interpreted as the value collectively and implicitly granted to each of these functionality and heritage characteristics. Reference levels provide relevant information to this aim.

In a second approach related to the weak sustainability paradigm, the value of the ecosystems is inferred through the explicit valuation of the benefits (or damages) arising from the state of specific components. This approach is the approach recommended in the SEEA-EEA (UNSD, 2014b). Such an approach allows to monitor the – often conflicting – use of ecosystems by humans and manage their overall contribution to human welfare.

Such an accounting system is particularly suited to integrated ecosystem management, where the capacity of ecosystems to support human welfare is managed on a safe operating zone, where the maintenance of options and irreplaceable features are guaranteed. Noticeably, it matches and rationalizes the monitoring system currently used in the context of the integrated management of marine ecosystems in Europe, and clarifies the basis for discussing and complementing it.

Beyond the uses emphasized by the SEEA-EEA in its current version, some potential uses of the ecosystem extent and condition accounts include:

- designing meaningful aggregate indicators at national levels fitted for sustainability dashboards[53];
- reinforcing policymaker accountability through the provision of objective and relevant information on distance to targets and inconsistencies in policy objectives;
- supporting the discussion and iterative refinement of SMART[54] policy targets;
- providing robust information tailored to the design of innovative policy instruments[55];
- increasing the efficiency of ecosystem monitoring and fostering strategic data collection tailored to the needs of integrated ecosystem management.

While this article mainly discusses the conditions for a meaningful monitoring system of ecosystems and the related setting of reference condition levels, the practical details for the implementation of these account and the realism of each of these potential

[53] for instance about the costs of ecosystem degradation based on maintenance and restoration costs.
[54] specific, measurable, achievable, relevant and time-bound.
[55] For instance, by linking the pressures and impacts to specific sectors or economic entities.

use remain to be investigated. This approach will be implemented for the development of marine ecosystem accounts in France. We also hope that this proposal will advance the revision of the SEEA.

Besides, the detailed methods leading to unpaid ecological costs accounts (Vanoli, 1995, 2015) as presented on Figure remain to be further specified and discussed. This is left to further research.

Acknowledgements

This work is supported by funding from the European Union's H2020 MAIA project. We also thank the "Comissariat Général au Développement Durable", the members of the "Chaire de Comptabilité Ecologique", as well as Rémi Mongruel and Pierre Scemama for their inputs.

References

Ang, F., & Van Passel, S. (2012). Beyond the environmentalist's paradox and the debate on weak versus strong sustainability. BioScience, 62(3), 251-259.

Bal, P., Tulloch, A. I., Addison, P. F., McDonald-Madden, E., & Rhodes, J. R. (2018). Selecting indicator species for biodiversity management. Frontiers in Ecology and the Environment, 16(10), 589-598.

Bartelmus, P. (2009). The cost of natural capital consumption: Accounting for a sustainable world economy. Ecological Economics, 68(6), 1850-1857.

Bendor, T. (2009). A dynamic analysis of the wetland mitigation process and its effects on no net loss policy. Landscape and Urban Planning, 89(1-2), 17-27.

Borja, A., Prins, T. C., Simboura, N., Andersen, J. H., Berg, T., Marques, J. C., ... & Uusitalo, L. (2014). Tales from a thousand and one ways to integrate marine ecosystem components when assessing the environmental status. Frontiers in Marine Science, 1, 72.

Carpenter, S.R., et al. (2010) General Resilience to Cope with Extreme Events, Sustainability, 4(12), 3248-3259; doi:10.3390/su4123248

Cash, D. W., Clark, W. C., Alcock, F., Dickson, N. M., Eckley, N., Guston, D. H., ... & Mitchell, R. B. (2003). Knowledge systems for sustainable development. Proceedings of the national academy of sciences, 100(14), 8086-8091.

Certain, G., Skarpaas, O., Bjerke, J. W., Framstad, E., Lindholm, M., Nilsen, J. E., ... & van der Meeren, G. I. (2011). The Nature Index: A general framework for synthesizing knowledge on the state of biodiversity. Plos One, 6(4).

Commissariat général au développement durable (2017). EFESE – Cadre conceptuel. Collection Théma Références.

Costanza, R., De Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., ... & Grasso, M. (2017). Twenty years of ecosystem services: how far have we come and how far do we still need to go?. Ecosystem services, 28, 1-16.

Crowards, T. (1997). Nonuse values and the environment: Economic and ethical motivations. Environmental Values, 143-167.

Crowards, T. M. (1998). Safe minimum standards: costs and opportunities. Ecological Economics, 25(3), 303-314.

Czúcz, B., Keith, H., Jackson, B., Maes, J., Driver, A., Nicholson, E., Bland, L. (2019) Discussion paper 2.3: Proposed typology of condition variables for ecosystem accounting and criteria for selection of condition variables. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 13 March 2019. 23 pp.

Decancq, K., & Lugo, M. A. (2013). Weights in multidimensional indices of wellbeing: An overview. Econometric Reviews, 32(1), 7-34.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., ... & Polasky, S. (2018). Assessing nature's contributions to people. Science, 359(6373), 270-272.

Dietz, S., & Neumayer, E. (2007). Weak and strong sustainability in the SEEA: Concepts and measurement. Ecological economics, 61(4), 617-626.

Edens, B., & Hein, L. (2013). Towards a consistent approach for ecosystem accounting. Ecological Economics, 90, 41-52.

Ekins, P., Simon, S., Deutsch, L., Folke, C., & De Groot, R. (2003). A framework for the practical application of the concepts of critical natural capital and strong sustainability. Ecological economics, 44(2-3), 165-185.

Ekins, P., & Usubiaga, A. (2019). "Brundtland+30: the continuing need for an indicator of environmental sustainability". In What Next for Sustainable Development?. Cheltenham, UK: Edward Elgar Publishing.

European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, United Nations and World Bank (2009). System of National Accounts 2008. Sales No. E.08.XVII.29.

European Commission - Joint Research Centre - Institute for Environment and Sustainability (2010). International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union; 2010

Everard, M., Reed, M. S., & Kenter, J. O. (2016). The ripple effect: Institutionalising pro-environmental values to shift societal norms and behaviours. *Ecosystem services*, *21*, 230-240.

Failing, L., & Gregory, R. (2003). Ten common mistakes in designing biodiversity indicators for forest policy. Journal of environmental management, 68(2), 121-132.

Farmer, M. C., & Randall, A. (1998). The rationality of a safe minimum standard. Land Economics, 287-302.

Geijzendorffer, Ilse R., et al. "Bridging the gap between biodiversity data and policy reporting needs: An Essential Biodiversity Variables perspective." Journal of Applied Ecology 53.5 (2016): 1341-1350.

Haines-Young, R. and M.B. Potschin (2018): Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure.

Hayes, K. R., et al. "Identifying indicators and essential variables for marine ecosystems." Ecological Indicators 57 (2015): 409-419.

Hein, L., Bagstad, K., Edens, B., Obst, C., de Jong, R., & Lesschen, J. P. (2016). Defining ecosystem assets for natural capital accounting. PloS one, 11(11), e0164460.

Heink, U., & Kowarik, I. (2010). What criteria should be used to select biodiversity indicators?. Biodiversity and Conservation, 19(13), 3769-3797.

Heink, U., Marquard, E., Heubach, K., Jax, K., Kugel, C., Nesshoever, C., ... & Vandewalle, M. (2015). Conceptualizing credibility, relevance and legitimacy for evaluating the effectiveness of science–policy interfaces: challenges and opportunities. Science and Public Policy, 42(5), 676-689.

Heink, U., Hauck, J., Jax, K., & Sukopp, U. (2016). Requirements for the selection of ecosystem service indicators–The case of MAES indicators. Ecological Indicators, 61, 18-26.

Howe, C., Suich, H., Vira, B., & Mace, G. M. (2014). Creating win-wins from trade-offs? Ecosystem services for human well-being: a meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change*, *28*, 263-275.

IPBES, (2016). Preliminary guide regarding diverse conceptualization of multiple values of nature and its benefits, including biodiversity and ecosystem functions and services (deliverable 3 (d)). IPBES/4/INF/13.

IPBES (2019): Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. S. Díaz, J. Settele, E. S. Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A. Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J. Willis, and C. N. Zayas (eds.). IPBES secretariat, Bonn, Germany. 56 pages.

Jetzkowitz, J., Van Koppen, C. S. A., Lidskog, R., Ott, K., Voget-Kleschin, L., & Wong, C. M. L. (2018). The significance of meaning. Why IPBES needs the social sciences and humanities. Innovation: The European Journal of Social Science Research, 31(sup1), S38-S60.

Johansson-Stenman, O. (1998). The importance of ethics in environmental economics with a focus on existence values. Environmental and Resource Economics, 11(3-4), 429-442.

Keith, H., Maes, J., Czúcz, B., Jackson, B., Driver, A., Bland, L., Nicholson, E. (2019). Discussion paper 2.1: Purpose and role of ecosystem condition accounts. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 13 March 2019. 34 pp.

Kenter, J. O., O'Brien, L., Hockley, N., Ravenscroft, N., Fazey, I., Irvine, K. N., ... & Church, A. (2015). What are shared and social values of ecosystems?. Ecological Economics, 111, 86-99.

Kervinio, Y. and Vergez, A. (2018). Agir face à la complexité des valeurs de la biodiversité – Joindre les approches normatives et « coût-efficacité », in : Dron, D. et Espagne, E. (coord.) Les communs environnementaux : gérer autrement la rareté. Annales des Mines - Responsabilité et environnement 2018/4 (N° 92), p. 30-35.

Krosby, M., Tewksbury, J., Haddad, N. M., & Hoekstra, J. (2010). Ecological connectivity for a changing climate. Conservation Biology, 24(6), 1686-1689.

La Notte, A., Vallecillo, S., & Maes, J. (2019). Capacity as "virtual stock" in ecosystem services accounting. Ecological indicators, 98, 158-163.

Lai, T. Y., Salminen, J., Jäppinen, J. P., Koljonen, S., Mononen, L., Nieminen, E., ... & Oinonen, S. (2018). Bridging the gap between ecosystem service indicators and ecosystem accounting in Finland. *Ecological Modelling*, *377*, 51-65.

Lenton, T. M., Held, H., Kriegler, E., Hall, J. W., Lucht, W., Rahmstorf, S., & Schellnhuber, H. J. (2008). Tipping elements in the Earth's climate system. Proceedings of the national Academy of Sciences, 105(6), 1786-1793.

Lenton, T. M., Rockström, J., Gaffney, O., Rahmstorf, S., Richardson, K., Steffen, W., & Schellnhuber, H. J. (2019). Climate tipping points—too risky to bet against.

Mace, G. M. (2005). Biodiversity: an index of intactness. Nature, 434(7029), 32.

Maes J. et al., (2014). Mapping and Assessment of Ecosystems and their Services: Indicators for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. 2nd Report. Final, Technical Report 080.

Maes, J., Driver, A., Czúcz, B., Keith, H., Jackson, B., Bland, L., Nicholson, E., Dasoo, M. (2019). Discussion paper 2.2: Review of ecosystem condition accounting case studies: Lessons learned and options for developing condition accounts. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 13 March 2019. 25 pp.

Millennium Ecosystem Assessment (2005). Ecosystems and human well-being. (Island Press, Washington, D.C.).

Nel, J. L., & Driver, A. (2015). National River Ecosystem Accounts for South Africa. Discussion document for Advancing SEEA Experimental Ecosystem Accounting Project.

Niemeijer, D., & de Groot, R. S. (2008). A conceptual framework for selecting environmental indicator sets. Ecological indicators, 8(1), 14-25.

Obst, C., Hein, L., & Edens, B. (2016). National accounting and the valuation of ecosystem assets and their services. Environmental and Resource Economics, 64(1), 1-23.

OECD (2018), Cost-Benefit Analysis and the Environment: Further Developments and Policy Use, Éditions OCDE, Paris, https://doi.org/10.1787/9789264085169-en.

Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., ... & Maris, V. (2017). Valuing nature's contributions to people: the IPBES approach. Current Opinion in Environmental Sustainability, 26, 7-16.

Pereira, Henrique Miguel, et al. "Essential biodiversity variables." Science 339.6117 (2013): 277-278.

Perrings, C., & Pearce, D. (1994). Threshold effects and incentives for the conservation of biodiversity. Environmental and Resource Economics, 4(1), 13-28.

Perrings, C. et al., 1995. The Economic Value of Biodiversity. In: V.H. Heywood (Ed.), Global Biodiversity Assessment. Cambridge University Press, for UNEP, Cambridge.

Potschin, M., Haines-Young, R., 2017. Linking people and nature: Socio-ecological systems. In: Burkhard, B., Maes, J. (Eds.), Ecosystem Services Mapping. Pensoft Publishers, Bulgaria, pp. 41–43.

Razzaque, Visseren-Hamakers et al (2019). Chapter 6. Options for Decision Makers. In: IPBES Global Assessment on Biodiversity and Ecosystem Services. Unedited draft chapters, 31 May 2019.

Recuero Virto, L., Weber, J. L., & Jeantil, M. (2018). Natural capital accounts and public policy decisions: Findings from a survey. Ecological Economics, 144, 244-259.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E. F., ... & Nykvist, B. (2009). A safe operating space for humanity. nature, 461(7263), 472.

Rodríguez, J. P., Beard Jr, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., ... & Peterson, G. D. (2006). Trade-offs across space, time, and ecosystem services. *Ecology and society*, *11*(1).

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., & Folke, C. (2015). Planetary boundaries: Guiding human development on a changing planet. Science, 347(6223), 1259855.

Stern, D. I. (1997). Limits to substitution and irreversibility in production and consumption: a neoclassical interpretation of ecological economics. Ecological economics, 21(3), 197-215.

Stiglitz, J. E., Sen, A. and Fitoussi, J.-P. (2009). Report by the Commission on the measurement of economic performance and social progress.

Tittensor, Derek P., et al. "Integrating climate adaptation and biodiversity conservation in the global ocean." Science Advances 5.11 (2019): eaay9969.

UNCEEEA, 2019. The Role of the System of Environmental-Economic Accounting as a Measurement Framework in Support of the post-2020 Agenda. 14 December 2018

United Nations Statistical Division, 2014a. System of Environmental-Economic Accounting: Central Framework, Official publication.

United Nations Statistical Division, 2014b. System of Environmental-Economic Accounting: Experimental Ecosystem Accounting – Official publication.

United Nations Statistical Division, 2017. Technical Recommendations in support of the System of Environmental-Economic Accounting 2012–Experimental Ecosystem Accounting. White cover publication, pre-edited text subject to official editing.

Vaissière, A. C., Levrel, H., Hily, C., & Le Guyader, D. (2013). Selecting ecological indicators to compare maintenance costs related to the compensation of damaged ecosystem services. Ecological indicators, 29, 255-269.

Vanoli, A. (1995). Reflections on environmental accounting issues. Review of income and wealth, 41(2), 113-137.

Vanoli A., 2015. National accounting and consideration of the Natural heritage. in CGDD (ed.) Nature and the Wealth of Nations – la Revue du CGDD, décembre 2015, pp. 75-84.

Villamagna, A. M., Angermeier, P. L., & Bennett, E. M. (2013). Capacity, pressure, demand, and flow: a conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity, 15, 114-121.

Weber, J.-L. (2014). Ecosystem Natural Capital Accounts: A Quick Start Package, Montreal, Technical Series No. 77, Secretariat of the Convention on Biological Diversity, 248 pages.