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Methods

Defining ecological liabilities and structuring ecosystem accounts to support the transition to sustainable societies

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Abstract

Harnessing reliable and relevant information on ecosystems requires focusing and prioritising information acquisition on dimensions of interest. As a boundary object between ecosystem monitoring, research and public decision-making, ecosystem accounting can serve this purpose. We develop an argument in favour of a set of accounts, consistent with the statistical standard part of the System of Environmental-Economic Accounting (SEEA-EA), that explicitly links monetary accounts to ecosystem extent and condition accounts. The ecosystem condition account is structured in three categories reflecting the main values motivating integrated ecosystem management targets and notions of «good ecological status». These categories are: (i) the maintenance of their heritage dimensions, (ii) their capacity to sustainably provide ecosystem services and (iii) the maintenance of their overall functionality. We discuss how such ecosystem accounts and associated monitoring can form the basis both for assessing an ecological debt by using a cost-based approach and for designing an action-orientated information system suitable to support the transition towards sustainable societies.

Keywords

ecosystem accounting, ecosystem condition, ecological debt, monetary valuation, cost-based approach

Introduction

The need to complement traditional economic indicators, such as gross domestic product (GDP), employment or public debt with a list of indicators able to account for a long-term and broader vision of social progress is widely recognised as a key component of sustainable pathways*[1]. In order to meet this need, the Ecosystem Accounting framework of the System of Environmental-Economic Accounting (SEEA-EA) was adopted in March 2021 by the United Nations Statistical Commission (UNSC) as a statistical standard in its biophysical accounts (chapters 1-7) and as a recommendation in its monetary accounts (chapters 8-11). This system organises an integrated and spatialised framework to monitor and account for ecosystem extent, condition, services and assets. This new standard leaves much leeway for implementation by regional and national statistical offices. At the European level, harmonising methods within a reporting framework will be supervised by Eurostat.

Ecosystem accounts gather multiple data in a structured framework and derive standardised tables and indicators on this basis (Fig. 1). It is useful to distinguish the accounts from the underlying integrated information system as both could be useful in different ways depending on their design. The information system required to build the accounts is spatialised and organised around basic spatial units (BSU) to which diverse information can be attributed. Through a set of categories and conventions*[2], ecosystem extent, condition, supply and use services accounts are derived from this spatial grid. They form a set of biophysical ecosystem accounts that can be used to derive standardised indicators. These accounts, in turn, shape the information system by drawing attention to specific ecosystem features.

As a boundary object*[3] between ecosystem monitoring, research and public decision-making, this framework offers great potential for multiple uses. However and despite a diversity of exploratory implementation for ecosystem accounting, the effectiveness of these approaches to improve decisions remains unsupported (Razzaque et al. 2019, Comte et al. 2022, IPBES 2022). Further implementation requires pursuing challenging research on measurement and valuation, with greater linkages with the discussions on diverse value perspectives and on the actual uses and impacts of accounting (IPBES 2022, message B7).

In this article, we propose answers to such challenges. We specify the features of an ecosystem accounting system suitable for supporting integrated ecosystem management and monitoring ecosystem degradation at different scales. Thereby, we emphasise how the estimation of the costs required to reach ecological targets can induce the production of useful information involving diverse communities and value perspectives. We also stress how some features of this system (definitions, categories and valuation methods) crucially depend on the explicit identification of its intended uses.

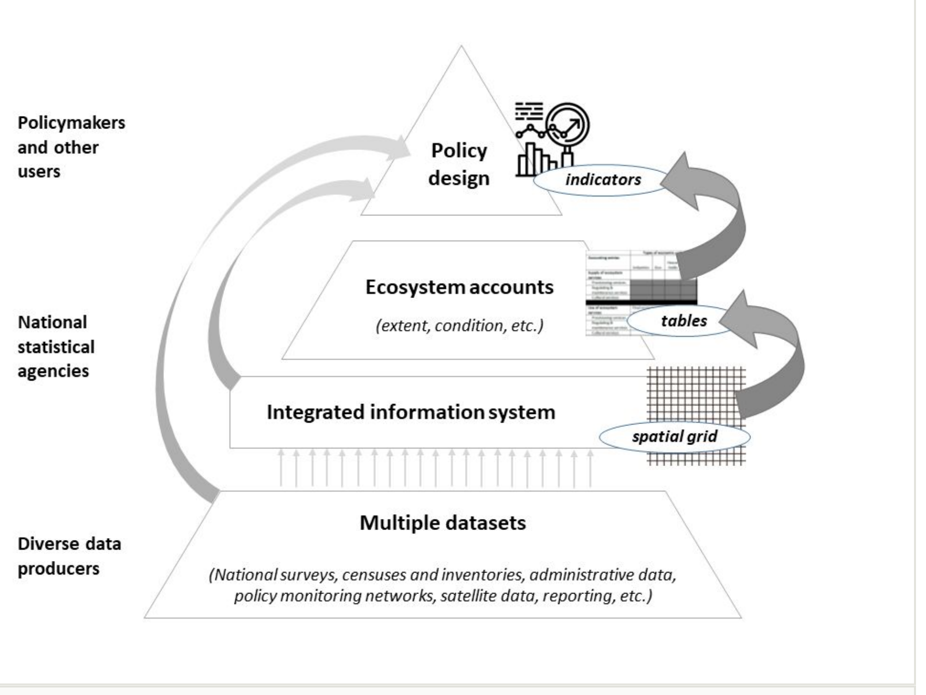

Figure 1.

Ecosystem accounts as a boundary object between data producers and policy-makers.

Comment: indicators supporting policies can both be derived from a structured accounting system where elements are gradually built through a set of rules and conventions motivated by the conceptual framework (arrows on the right) or directly from the underlying information system (arrows on the left). In this perspective, the accounting system not only serves the production of indicators, but can also shape the information system on Nature by emphasising data gaps.

In the first section, we describe how liabilities can be defined on the basis of commitments to maintain and restore ecosystems, and expressed both in biophysical and monetary terms. In the second section, we then discuss how such an approach could fit into a dynamic socio-political process of target setting and implementation. In the third section, we discuss how the dimensions of interest and reference levels used in biophysical accounts could be inferred from existing management targets with diverse underlying rationales. To conclude, we discuss the main avenues of research required for such a system of accounts to be used to a degree comparable to economic accounts and provide substantial support for the transition to sustainable societies.

Valuing the liabilities due to ecosystem degradation

Amongst existing proposals to complement GDP, a monetary indicator of the costs of ecosystem degradation could offer a measurement comparable with economic outcomes. First, we present and discuss the conceptual framing of the cost-based approach for estimating such an indicator. Second, we consider the practical options regarding its implementation.

Valuing the costs of ecosystem degradation - conceptual framing

Two distinct framings have been discussed through time in SEEA manuals regarding the monetary valuation of the costs of ecosystem degradation (United Nations 1993, United Nations et al. 2003, United Nations 2021).

In the first framing, the focus is on the *costs borne by humans* due to environmental degradation, where such *costs* are understood as *damage* or *(negative) benefits*. Two approaches developed from this originally called "cost-borne" perspective (United Nations 1993) are the *damage-based methods* (United Nations et al. 2003, chap. 9 and 10) and current recommendations for valuing ecosystem degradation in the SEEA-EA (United Nations 2021, ch. 10). This latter approach mainly differs from the former by moving away from "welfare" towards "exchange" values to assess losses in ecosystem services. In this latter approach, ecosystem degradation and its costs are identically defined as the fall in the net present value of expected future returns of the ecosystem services supplied by "ecosystem assets"*[4].

In the second framing, the focus is on the *costs caused by (human-induced)**[5] *environmental degradation*, where such costs are understood as the *expenses* or *efforts* required to avoid ecosystem degradation or to restore ecosystems. Approaches developed from this originally called "cost-caused" perspective are currently called *restoration cost-based approaches**[6] in the SEEA-EA (United Nations 2021, §12.32). Examples are estimates resulting from greened-economy modelling (United Nations et al. 2003, §10.199, Hueting 2013) or the *unpaid ecological costs* proposed by Vanoli (1995). In all these approaches, *ecosystem degradation* remains defined in biophysical terms as the discrepancy between current and some *reference condition*, while its costs are the costs of the measures needed to maintain or to restore ecosystems from their current to this reference condition. The proposal discussed in this paper also follows this framing.

As will be argued in the second section, both valuation approaches could inform ecosystem management at different stages of the policy cycle. In a nutshell, estimates based on the former framing could be useful to provide rationales for policy and action, while, based on the latter, estimates can serve organising action to prevent or remedy ecosystem degradation (United Nations et al. 2003, §11.124).

However, we may stress several key difficulties when relying on the first framing for valuing the cost of ecosystem degradation at the national level. A first difficulty is related to indeterminants in the preferences required to conduct such an assessment. Valuing future flows of ecosystem services requires predicting future conditions, but also quantifying and valuing related uncertainties consistently with existing attitudes regarding risks and uncertainties, inferring the fundamentally unknown future generations' preferences and including non-use values. All of these issues pose considerable conceptual and practical difficulties. For example, we can note that, despite their recognition, non-use values are currently excluded from the ecosystem asset value in the SEEA-EA (United Nations 2021; §6.72). We may also emphasise that the values associated with ecosystems and biodiversity may not always pre-exist, but that they could emerge from individual reflection and public discussion and require considering valuation in the context of socio-political value formation processes (Sen 1995, Spash and Hanley 1995, Kenter et al. 2015).

Another difficulty relates to the different nature of what is valuable when facing complexity and uncertainty. The SEEA-EA approach to valuing ecosystem degradation relies on simple DPSIR-like*[7] local causality representations, first from pressures to condition change and then from condition change to impacts on benefits. However, in complex systems, where non-linearity and uncertainty prevail, local causal chains can no longer be assumed (Chavalarias 2020). One consequence of this is that targets on pressures have to be defined as they cannot be unambiguously inferred from targets on ecosystem condition. This is what the French Cour des Compte recognises, for instance, when it recommends to set targets directly on agricultural practices due to the delay and many sources of variability from them to ecosystem eutrophication (Cour des Comptes 2021). In those examples, we see how complexity leads to recognise that reductions in pressures shall be valuable *as such*. Here, the value of a reduction in pressure cannot be estimated through a difference in damage, but through an implicit valuation approach, based on what is collectively valued as revealed by existing targets.

A third difficulty is related to the diverse meanings of the monetary values covered by existing valuation methods. In practice, ecosystem services are assessed using diverse interrelated, but distinct concepts (for example, market price, opportunity cost, real cost and willingness to pay). This limits the possibilities to commensurate or sum resulting figures, though many of these figures would prove useful in different contexts. The impossibility to commensurate *use* and *exchange* values and the drastic reduction of the scope of valuation is one prominent example of such concerns. In this line, Femia and Capriolo (2022) advocate that each of the numerous monetary values proposed may not be additive, but that they can all be useful separate pieces of a rich and inclusive information system able to fit a diversity of policy needs.

Most of these discussions are not new and we shall note that the London Group's research agenda in the 1990's suggested investigating the restoration cost-based approach before relegating it to the background (United States Bureau of Economic Analysis 1995, Brouwer et al. 1999, Radermacher et al. 1999, Bartelmus 2013). Amongst the most advanced proposals, André Vanoli developed an argument in favour of a *cost-based approach* to measure, in exchange value, the costs of observed ecosystem degradation (Vanoli 1995, Vanoli 2017). In Vanoli's perspective, the "Economy" would remain equipped with its own information system structured according to the SNA*[8] and a specific institutional sector called "Nature" would be considered as an entity distinct from the "Economy" and equipped with its specific information system. Relationships between "Nature" and the "Economy" could then be monitored according to liabilities between them. *Unpaid ecological costs* can then be defined as “[*representing] the value, in terms of avoidance or restoration costs, of the degradation of ecosystem assets in a given period due to economic activities*” (Vanoli 2017). In contrast with the imputed maintenance costs of the 1993 version of the Integrated Environmental and Economic Accounting (United Nations 1993), these costs are considered as a *liability*, meaning a commitment to pay in the future*[9]. More recently, Germain and Lellouch (2020) have defined a "*prospective debt*" in an economic perspective as " *the discounted equivalent of the future expenditure stream required to meet a given liability*", thereby defining a "*notion of implicit liabilities used for other types of public expenditure such as pensions*".

Building on these notions of “*unpaid ecological costs*”*[10] and "*prospective debt*", we define a monetary *ecological debt* as the costs that would have to be incurred in order to attain some reference levels on *dimensions of interest* of ecosystem extent and condition*[11]. We now focus on the practical implementation of an aggregate measure of the costs of ecosystem degradation at the national scale, starting with the practical valuation of the costs required to attain some reference levels.

Valuing in practice - modelling or observing?

In the second framing (cost-caused), two distinct perspectives are possible for the practical estimation of the cost of ecosystem degradation, which we may, respectively, call the *economic* and the *accounting* perspectives.

In an **economic perspective**, the estimation is carried out through different kinds of modelling. Technico-economic models are the first kind. They rely on a database of possible measures to reach defined targets, along with their costs and their impacts. From this, the estimation generally consists in adding the required cost for implementing measures from the least to the most costly until the reference level is reached*[12]. Macroeconomic models are a second kind (Brouwer et al. 1999). They directly estimate the cost required to achieve the targets from a representation of how financial efforts translate into impact (dose-response models).

Resulting estimates are uncertain, in particular, due to limited knowledge on future market conditions and technical progress. Given the limits and uncertainties associated with modelling, intercomparisons of model outcomes or regular model updates are necessary to ensure reasonable estimates. For example, Germain and Lellouch (2020) assess what they call a "*prospective debt*" related to climate change mitigation targets for France. They develop a simple macroeconomic model which, they show, yields consistent results with another pre-existing technico-economic model. They also show that the carbon value derived from their macroeconomic model is roughly consistent with the tutelary carbon value used in France for the evaluation of public investments, suggesting their estimates are reasonably robust. We can further stress that the estimation of this latter tutelary carbon value is itself based on such intercomparison between the results of technico-economic and macroeconomic models (Quinet et al. 2019, Bureau et al. 2021).

For biodiversity and ecosystems, numerous models relating responses with outcomes on dimensions of interest could be used to carry out similar estimations. For example, the scientific literature on wetland restoration costs can provide sound estimates of the required budget to reach specific ecological outcomes (Szałkiewicz et al. 2018). Recent reports also estimate how much it would cost to stop species and natural habitat erosion by 2030 (Deutz et al. 2020). However, greater complexity may arise due to the need to account for the multiplicity of drivers of change (and their non-linear combination) and for the multiplicity of dimensions of interest (and their interactions)*[13]. Sketching the contours of efficient trajectories, as well as their overall costs, will require integrating multiple models and explicit spatial modelling. The resulting estimates will involve high uncertainty. Thus, as for climate, credible values will require intercomparisons of models and regular updates (Guivarch et al. 2017, Riahi et al. 2022).

In an **accounting perspective**, drawing from a comparison with national economic accounting, the indicator results from the aggregation of costs estimated and reported at the level of economic units (corporation, household, government etc.). This approach requires widening organisational accounting in order to monitor impacts and liabilities at this level. Some legal procedures, although still partial, already exist, for example, through impact assessments for development projects supported by corporations or governments. Non-financial reporting is also increasingly standardised, for example, through initiatives such as the European Union's Corporate Sustainability Reporting Directive (CSRD).

More ambitiously, extended and normalised accounting models could ensure the production of suitable information. The Comprehensive Accounting in Respect of Ecology (CARE) is an example of such a model. With it, organisations would be required to monitor and account for their impacts or pressures on ecosystems, in relation to ecological reference levels*[14] (Rambaud and Richard 2015, Rambaud and Chenet 2021). In this framework, organisations report as liabilities in their own accounts the cost of the prevention and restoration measures required to ensure these reference levels are achieved*[15]. With a similar auditing procedure as financial reporting, reliable data would be produced at organisational level. Should the reference levels set at organisations level be consistent, on larger scales, with the environmental norms of the territories where they operate, the bottom-up aggregation of liabilities would provide an estimation of the costs of ecosystem degradation.

The economic and accounting approaches both cover investment and recurrent costs, in exchange value, for achieving existing targets. However, they differ regarding the relevant information on costs, possibly reflecting differences in objects and concerns of the economic and accounting disciplines. While the accounting discipline seeks to assess reliable, tangible as well as opposable commitments to pay, which are unambiguously attributed to an entity, the economic discipline seeks to assess values that are relevant to decision-making, even though they are more hypothetical or not clearly attributed. As a result, the scope and nature of the costs covered in the ecological debt indicator could differ between the two perspectives, leading to different interpretations and relevant uses. Regarding the scope, the reported costs cover reduction and restoration measures in the accounting perspective*[16]. In contrast, the economic perspective adopts a wider scope, possibly including avoidance measures, which are the opportunity costs of renounced projects. Regarding the nature of the information on costs covered, the economic perspective departs from the mere observation of actual transactions and other facts. It involves models, hypotheses, normative inputs and requires interpretation (e.g. intercomparisons of model results). Many authors have argued against such “hard modelling” approaches in official statistics as they are alleged to undermine accuracy and trust in the information produced (Desrosieres 2009, Vanoli 2017, Radermacher 2020). Therefore, the economic perspective may be best carried out outside official statistics, in a specific institutional context involving research and other public institutions in close articulation with official statistics. The accounting perspective, on the contrary, relies on the observation of approved accounting information. This is more in line with the traditional role of public statistics and national economic accounting.

Fig. 2 presents how an ecological debt account could be built at the margins of national economic and ecosystem accounting. Whether conducted within or outside official statistics, interpretations of the resulting aggregate indicator and its variations may depend on the perspective taken towards implementing it.

In the accounting perspective, for instance, variations of such an aggregate indicator from an accounting period to the next could be further broken down and interpreted in a similar way as SNA categories for variation of assets and liabilities (United Nations et al. 2009, §§12.1-3). They would be:

1. changes in ecosystem condition attributable to an economic unit (e.g. the destruction of a hedge by a farmer or the revegetation of a degraded area by a manager); treated as "economic transactions",
2. changes in ecosystem condition resulting from exogenous causes (e.g. hurricane damage to a coral reef); treated as "other volume change",
3. changes in the reference levels reflecting changes in collective preferences*[17]; treated as "other volume change",
4. changes resulting from improved data and assessment methods; treated as "other volume change",
5. technical progress and the evolution of prices conditioning the costs of maintenance and restoration actions required to reach related reference levels; treated as a "revaluation" (change in prices level or structure).

Regardless of the perspective taken, interpreting these costs as an ecological debt, i.e. a liability that would have to be paid at some point, requires a relevant choice of reference levels. This point is discussed in the next section.

Good ecological status as a boundary object for the strategic discussion of environmental targets and reference levels

While recognising the existence of a large diversity of approaches, the SEEA-EA recommends defining reference condition levels “*using the natural state as the reference* *condition*" (United Nations 2021, § 5.72). The rationale is that the methodology "*should allow accounts to be developed devoid of value judgements and which do not imply a policy goal or a desired condition*" (Keith et al. 2020). However, in order to be interpreted as an ecological debt, the costs required for reaching some reference levels, both on ecosystem extent and condition, would have to be somehow related to our collective willingness to pay for maintaining and restoring ecosystems*[18]. That is why Vanoli (2017) estimates the ecological debt on the basis of *environmental norms* determined "*in the form of societal standards*", as revealed by policy goals. While simple in principle, such a perspective raises challenging practical questions. It also requires considering valuation as part of a larger sociopolitical process such as argued, for instance, by Godard and Laurans (2004). In this section, we discuss how a collective willingness to pay for maintaining and restoring ecosystems can emerge from a dynamic political process involving different communities around the discussion and collective legitimation of the targets defining good ecological status.

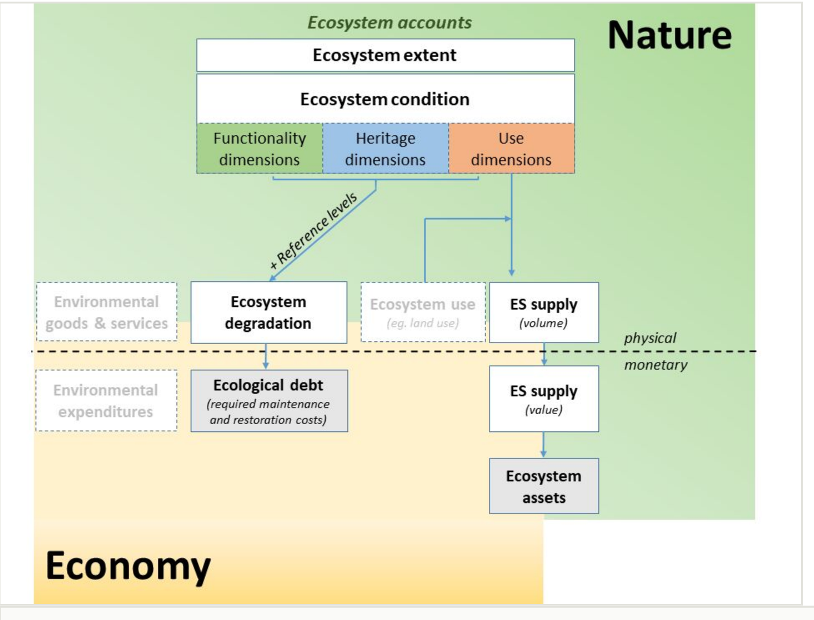

Figure 2.

Structure of the accounts derived from an ecosystem monitoring framework.

Comment: Boxes reflect different accounts. Some of these components are already required in statistics as the biophysical side of the SEEA-EA. Other components are not explicit in the SEEA-EA. Accounts specific to the Nature information system are in the green area, accounts common to the Economy and Nature are in the yellow area. Accounts in grey may need to be produced in specific institutional contexts, as they may not meet certain quality criteria for official statistics. Under some conditions, existing accounts represented with dashed borders could provide useful information to monitor pressures (e.g. ecosystem use) or monitor the actions taken and their effectiveness, thereby fostering learning regarding solutions (for example, environmental goods and services accounts).

Source: adapted from Comte et al. (2020).

The detailed description of such a process first requires careful consideration of how the scientific, political and administrative spheres can be involved in the dynamic processes of target setting and implementation at different scales. Thus, we shall also describe good ecological status as a boundary object, involving different communities without getting them to drastically change their referentials. To define such a process, we will elaborate on the useful distinction between environmental limits, norms and targets, initially proposed by Usubiaga-Lião and Ekins (2021), emphasising the processes and communities involved in their construction. *Environmental limits* come from scientific arenas. They warn about the risks of crossing specific thresholds, just as, for example, Steffen et al. (2015) did at the global level with planetary boundaries. Although they may encompass some normative content, they leave the most crucial trade-offs to public discussion. *Environmental targets* are elaborated through the political process, whose role is notably to ensure the legitimacy of collective choices and building political preferences weighing economic and social considerations. At this level, environmental targets may be expressed in laws, regulations, plans and strategies. However, they may still be insufficiently consistent, specific, measurable, ambitious or realistic for implementation*[19]. Therefore, we may define *environmental norms* as specifications of environmental targets performed by the administration in order to enforce them.

With these distinctions in mind, an ideal socio-political process can be described that articulates – rather than opposes – existing valuation approaches for sustainable ecosystem management. First, the warnings given by scientists lead, for example, to identify limits beyond which the population is exposed to risks or other considerations. Within the political arenas, the discussion and interactions amongst politicians, scientists and the public lead to setting environmental targets informed by science. These objectives are then translated into operational norms at relevant levels for implementation (Commons 1970). Of course, this process is fundamentally iterative and dynamic. Scientific advances can lead to updates of environmental limits just as societal changes can induce evolutions in targets and associated norms. Inconsistencies between different environmental norms can also lead to adjust targets.

Alongside other scientific inputs, ecosystem accounts can provide information for this process at different levels, as illustrated in Fig. 3. In political arenas, knowledge on ecosystem services could complement knowledge on environmental limits with economic and social considerations to define targets. In turn, norms clarify residual trade-offs or the realism of existing targets by making explicit the required measures and associated costs, possibly leading to their revision in an iterative dialogue. Such processes are already underway, for example, in the European Union's management of continental aquatic ecosystems, where disproportionate costs may justify relaxing targets, through postponing the deadline for achieving good status of water bodies (see, for example, Boeuf et al. (2018)). The discrepancy between existing targets and actual policies as documented by the ecological debt and its evolution could also reinforce accountability in the political sphere and call for adjustments in ambition or actions.

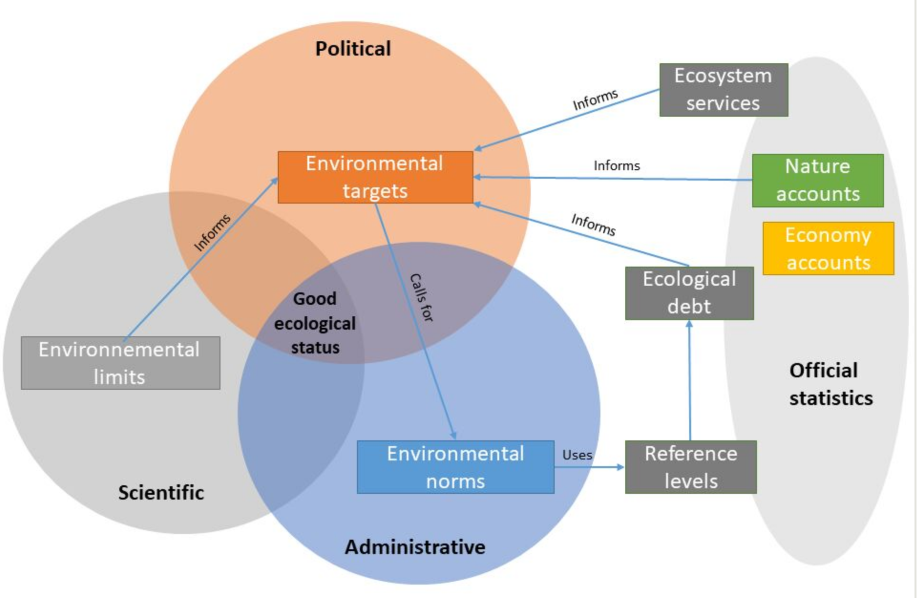

Figure 3.

Good ecological status and ecosystem accounts as boundary objects at the level of strategic discussion around environmental targets.

Comment: Each of the spheres gathers a diversity of actors with adequate governance systems. Arrows represent some possible interactions between spheres around the objects introduced in this article. They are not exhaustive of the complex and moving interactions between these spheres. For instance, Environmental targets established in the political sphere can be informed by scientific inputs regarding environmental limits, but also with the social and economic information built around Nature and Economy accounts, for instance, regarding losses in ecosystem services or the cost along the pathways towards the targets. In return, the targets can be used to establish environmental norms most suitable to provide sound reference levels usable to provide robust information on the effort needed towards *good ecological status*.

What is worth noting at this stage is that such an indirect approach to valuation allows for the production of a meaningful macro-aggregate indicator without constraining the expression of values presiding over the formulation of targets. Such values may be expressed by a diversity of actors with different interests, concerns and world-views, including ethical, symbolic or identity-related considerations. They may even not pre-exist, but emerge from individual reflections and public discussions. Such an indirect approach to valuation can be particularly relevant, as managing ecosystems and their biodiversity requires embracing their complex functioning (OCDE 2018), value formation (Sen 1995) and pluralism (Pascual et al. 2021).

Focusing on what matters: ecosystem extent and condition as the component of a broad information system on Nature

At this stage of the argument, it shall be clear that no meaningful monetary indicator of ecosystem degradation could be obtained in the absence of sound and relevant biophysical monitoring of ecosystem extent and condition*[20]. This requires identifying *dimensions of interest*, i.e. dimensions related to specific *values*[*21], either explicitly or implicitly based on existing targets. In particular, the estimation of an ecological debt through the cost-based approach requires careful attention to variable selection so that all dimensions subject to environmental norms as previously defined are monitored. Therefore, we discuss here how such an intention in terms of monetary valuation could, in return, shape the ecosystem extent and condition accounts.

Currently, the measurement of ecosystem *extent* is organised around the SEEA ecosystem type reference classification types, based on the IUCN Global Ecosystem Typology (GET) and the measurement of ecosystem *condition* is organised around the SEEA-EA ecosystem condition typology (ECT). This latter typology remains primarily orientated according to natural science categories, where links to values are made through the use of two indicator selection criteria: instrumental or intrinsic relevance (Czùcz et al. 2021, United Nations 2021). With the intention to strengthen ecosystem accounting as a boundary object, Comte et al. (2020) propose to structure an ecosystem condition dashboard according to three categories motivated by an explicit relationship with existing policy targets and underlying values, so that all dimensions of interest can be specified along with the underlying rationale for their monitoring. Each of these categories are related to the value concepts invoked to justify management targets, which are of three contrasting types.

- *Heritage* dimensions comprise the features of ecosystems that are deemed remarkable for some reason. The value of monitoring such dimensions comes from the non-use or intrinsic values tied to such specific elements and that are invoked to justify conservation targets.
- *Use* dimensions*[22] comprise the features of ecosystems that determine their capacity to sustainably provide specific ecosystem services*[23]. The advantage of monitoring such dimensions comes from the benefits (use values) invoked to justify some optimal management targets (e.g. the *maximum sustainable yield* (MSY) for fisheries), possibly tempered by existing trade-offs with other dimensions.
- *Functionality* dimensions finally comprise the features of ecosystems ensuring the maintenance of their overall resilience and functionality. The benefit of monitoring such dimensions comes from the values invoked to justify fostering ecosystem resilience and maintaining pressures within a safe operating space, without reference to specific ecosystem services or remarkable elements*[24].

These three categories reveal three major rationales that are often opposed, overlooked or hierarchised in ecosystem monitoring and management. For example, the *functionality* rationale dominates in the *planetary boundary* (Rockström et al. 2009, Steffen et al. 2015) or in the *environmental sustainability gap* (Usubiaga-Lião and Ekins 2021) frameworks. The domination of the *use* rationale in the SEEA-EA is also worth noting, at least when it comes to choosing what will count in monetary accounts (United Nations 2021).

This categorisation is useful for ensuring an inclusive selection of variables of interest. It recognises that each of these rationales has its own logics and legitimacy for ecosystem monitoring and management, without prioritising them. It allows the construction of an ecosystem monitoring system that does not create *a priori* power asymmetries and is suitable to support a diversity of political projects. Such a framework thus embraces a broad and inclusive scope, both consistently with the SEEA-EA approach to condition (Keith et al. 2020) and with the increasingly recognised need to embrace diverse valuation perspectives in accounting and policy-making (see, for example, Pascual et al. (2021) or IPBES (2022), messages KM7 and B7).

Through a quick correspondence with the descriptors defining good ecological status (GES) for marine integrated ecosystem management, Comte et al. (2020) also show that these categories can easily be matched with the descriptors defining *good ecological status* in the Marine Strategy Framework Directive (MSFD) (Fig. 4). They further note that the *functionality* category provides a rationale for most of the dimensions defining *good ecological status* for the marine environment (8 out of 11 descriptors), thereby emphasising the importance of a holistic perspective on ecosystem functionality for actual management.

Such a typology helps to bridge a gap between the communities and also helps overcome some of the issues identified in the SEEA-EA research and development agenda (United Nations 2021, p. 348). In particular, this typology ensures considering all dimensions of non-instrumental relevance (heritage and functionality) by establishing explicit linkages with underlying values, thereby allowing the broad and inclusive perspective on values to be reflected in valuation and monetary accounts through the indirect and dynamic valuation approach discussed in the former sections*[25]. The functionality category, motivated by a recognition of the complexity of ecosystem functioning, also provides a compelling rationale for monitoring specific dimensions of ecosystem resilience and pressures as valuable in themselves, even in the absence of an explicit link with some ecosystem services or the intrinsic value tied to specific elements. We insist that the accounting system needs ensuring such monitoring so as to support current policies and complementary valuation approaches, such as those discussed in this article. As revealed by existing targets, required dimensions cover specific dimensions of ecosystem state (e.g. plastic wastes in the oceans), but also the attribution of specific change in ecosystem condition to human activities (e.g. fishing mortality) or pressures which could not easily be connected to change in ecosystem condition due to, amongst other difficulties, delayed impacts or natural variability (e.g. pesticide use or excess nitrogen in agriculture). At present, the SEEA-EA only accommodates pressure indicators in condition accounts as surrogate for state indicators (United Nations 2021, §5.103). This requires particular attention, as it puts the overall ecosystem account at risk of dismissing critical information regarding functionality concerns.

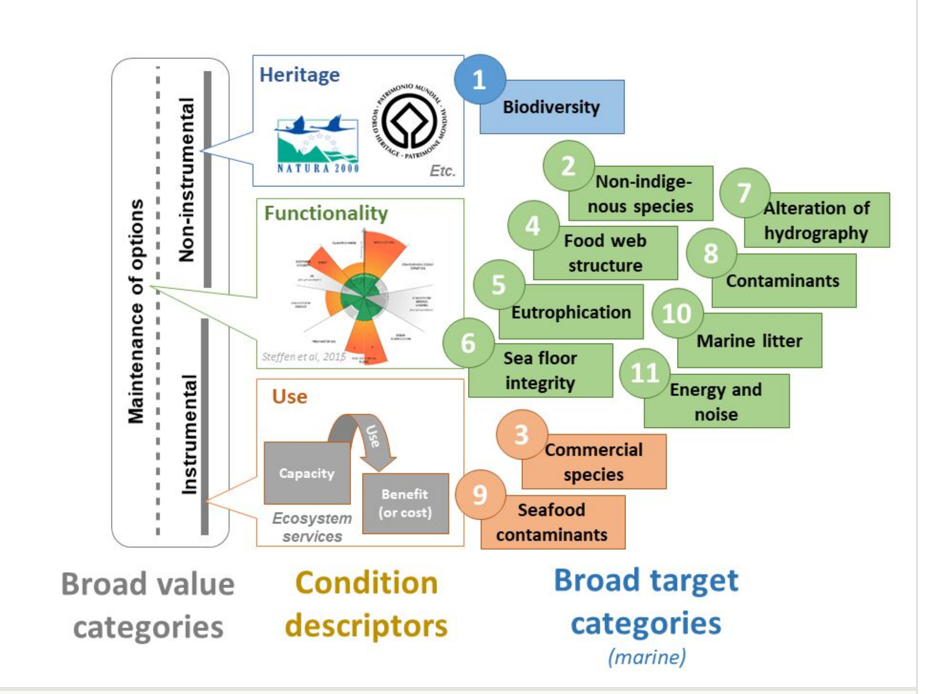

Figure 4.

Linkages between categories of ecosystem condition indicators, broad value concepts and broad categories of integrated management targets using the EU Marine Strategy Framework Directive as an example.

Comment: The *Heritage* category includes the conservation status of all ecosystem elements with intrinsic or non-instrumental worth as recognised through labels of diverse sorts (species of Community interest in the EU, World Heritage etc.). The *Functionality* category refers to the dimensions which need to be monitored in order to ensure that the overall functionality of the ecosystem is not threatened, as indicated by resilience indicators or when some pressures are above specific safe thresholds. Finally, the *Use* category monitors all direct determinants of ecosystems' capacity to contribute to specific dimensions of human welfare (ecosystem services). Targets categories are the 11 descriptors of the good ecological status of the EU Marine Strategy Framework Directive.

Credit: Planetary boundaries are designed by Azote for Stockholm Resilience Centre, based on analysis in Steffen et al. (2015) and Persson et al. (2022).

Conclusion - An agenda for research and action

In this article, ecosystem accounting and related data are framed so as to be part of a broad information system on nature tailored to support policy-making and other uses. This leads us to explicitly relate dimensions of interest with the multiple underlying values that motivate their monitoring. This provides a sound basis to discuss, expand and prioritise monitoring efforts. This also leads us to stress the need to complement SEEA-EA guidelines with a stronger focus on sustainable management issues, pressures, solutions and their costs. Such an information system would expand and benefit from the rich data that already exist in support of integrated ecosystem management policies.

An ecological debt indicator could be derived from such a system and included in sustainability dashboards (Stiglitz et al. 2009). Making visible ecosystem degradation alongside other dimensions of national progress may prove performative. Being monetary, comparisons with wealth creation as measured by GDP may be eased and ecosystem concerns may gain weight in budgetary discussions. More importantly and as for GDP, such an aggregate indicator (the destination) may not only be useful in itself, but also because of the whole information system associated with it and the processes that go with its production (the journey). One major benefit of this approach is to channel economic valuation efforts from those focused on the rationale for action (for example, the cost of inaction) to those focused on action and solutions, more in tune with policy- and decision-makers' needs. From the support of integrated ecosystem management*[26] to the design of policy instruments*[27], many potential uses could emerge. With a related ecological debt as a flagship indicator, such an accounting system may be used to an extent comparable to GDP and associated economic accounts.

The potential uses of ecosystem accounts are thus numerous, but we need more proof that such uses can be more than speculative or anecdotal (IPBES 2022, messages KM7 and B7). Further discussion and investigation are needed to identify and specify these potential uses – along with underlying theories of change*[28] –, validate them and discuss their relative relevance for supporting the transition to sustainable societies (IPBES 2022, message B7). Work is also needed to reinforce a co-construction framework complementing the “technical-push” orientation that largely dominated the development of accounts until now (Vardon et al. 2016). It shall also be clear, from this discussion, that many technical questions would require further investigations, for instance, regarding the relative merits of typologies for ecosystem types or condition, the different ways to define reference levels or the relevant categories for accounting treatments. Yet, these important discussions could not be solved rigorously without an explicit and specific identification of the main intended uses of the accounts and their context.

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Author contributions

Yann Kervinio and Clément Surun are co-first authors.

Conflicts of interest

The authors declare no conflict of interest.

References

- Bartelmus P (2013) Environmental-Economic Accounting: Progress and Digression in the SEEA Revisions. Review of Income and Wealth https://doi.org/10.1111/roiw.12056
- Boeuf B, Fritsch O, Martin-Ortega J (2018) Justifying exemptions through policy appraisal: Ecological ambitions and water policy in France and the United Kingdom. Water Policy 20 (3): 647-666. https://doi.org/10.2166/wp.2018.108
- Bowker G, Star SL (1999) Sorting things out. Classification and its consequences. The MIT Press
- Brouwer R, O'Connor M, Radermacher W (1999) GREEned National STAtistical and Modelling Procedures: the GREENSTAMP approach to the calculation of environmentally adjusted national income figures. International Journal of Sustainable Development 2 (1): 7-31. https://doi.org/10.1504/IJSD.1999.004304
- Bureau D, Quinet A, Schubert K (2021) Benefit-cost analysis for climate action. Journal of Benefit-Cost Analysis 12 (3): 494-517. https://doi.org/10.1017/bca.2021.11
- Chavalarias D (2020) From inert matter to the global society life as multi-level networks of processes. Philosophical Transactions of the Royal Society B 375 (1796): 20190329. https://doi.org/10.1098/rstb.2019.0329
- Commons JR (1970) The economics of collective action. University of Wisconsin Press
- Comte A, Kervinio Y, Levrel H (2020) Ecosystem accounting in support of the transition to sustainable societies: The case for a parsimonious and inclusive measurement of ecosystem condition. CIRED Working Paper 76.
- Comte A, Campagne SC, Bruzon AG, Santos-Martin F, Hein L, Levrel H (2022) Ecosystem accounting: past scientific developments and future challenges. Ecosystem Services 58: 101486. https://doi.org/10.1016/j.ecoser.2022.101486
- Cour des Comptes (Ed.) (2021) Evaluation de la politique publique de lutte contre la prolifération des algues vertes en Bretagne (2010-2019). Rapport public thématique. Juillet 2021..
- Czùcz B, Keith H, Maes J, Driver A, Jackson B, Nicholson E, Obst C (2021) Selection criteria for ecosystem condition indicators. Ecological Indicators 133: 108376. https://doi.org/10.1016/j.ecolind.2021.108376
- Desrosieres A (2009) How to be real and conventional: a discussion of the quality criteria of official statistics. Minerva 47: 307-322. https://doi.org/10.1007/s11024-009-9125-3
- Deutz A, Heal GM, Niu R, Swanson E, Townshend T, Zhu L, Delmar A, Meghji A, Sethi SA, Puente J (2020) Financing nature: Closing the global biodiversity financing gap. The Paulson Institute,The Nature Conservancy.

- Ellis E, Gauthier N, Klein Goldewijk K, Bliege Bird R, Boivin N, Díaz S, Fuller D, Gill J, Kaplan J, Kingston N, Locke H, McMichael CH, Ranco D, Rick T, Shaw MR, Stephens L, Svenning J, Watson JM (2021) People have shaped most of terrestrial nature for at least 12,000 years. Proceedings of the National Academy of Sciences 118 (17). https://doi.org/10.1073/pnas.2023483118
- European Environment Agency (1999) Environmental indicators: Typology and overview. Technical report No 25/1999.
- Feger C, Mermet L, Vira B, Addison PF, Barker R, Birkin F, Burns J, Cooper S, Couvet D, Cuckston T, Daily GC, Dey C, Gallagher L, Hails R, Jollands S, Mace G, Mckenzie E, Milne M, Quattrone P, Rambaud A, Russell S, Santamaria M, Sutherland WJ (2019) Four priorities for new links between conservation science and accounting research. Conservation Biology 33 (4): 972-975. https://doi.org/10.1111/cobi.13254
- Femia A, Capriolo A (2022) Beyond valuation. Monetary aggregates for the SEEA-EA. The Italian proposal. One Ecosystem 7 https://doi.org/10.3897/oneeco.7.e84689
- Germain J-M, Lellouch T (2020) The social cost of global warming and sustainability indicators: lessons from an application to France. Economie et Statistique / Economics and Statistics 517-518-519: 81-102. https://doi.org/10.24187/ecostat.2020.517t.2024
- Godard O, Laurans Y (2004) Evaluating environmental issue-Valuation as co-ordination in a pluralistic world. hal-00242936f.
- Guivarch C, Lempert R, Trutnevyte E (2017) Scenario techniques for energy and environmental research: An overview of recent developments to broaden the capacity to deal with complexity and uncertainty. Environmental Modelling & Software 97: 201-210. https://doi.org/10.1016/j.envsoft.2017.07.017
- Hueting R (2013) Environmentally sustainable national income: indispensable information for attaining environmental sustainability. Environmental Values 22 (1): 81-100. https://doi.org/10.3197/096327113X13528328798318
- 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. IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.3553579
- IPBES (2022) Methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Zenodo https://doi.org/10.5281/zenodo.6522523
- Keith H, Czúcz B, Jackson B, Driver A, Nicholson E, Maes J (2020) A conceptual framework and practical structure for implementing ecosystem condition accounts. One Ecosystem 5: 58216. https://doi.org/10.3897/oneeco.5.e58216
- Kenter JO, O'Brien L, Hockley N, Ravenscroft N, Fazey I, Irvine KN, Williams S, et al. (2015) What are shared and social values of ecosystems? Ecological economics 111: 86-99. https://doi.org/10.1016/j.ecolecon.2015.01.006
- Levrel H, Jacob C, Bailly D, Charles M, Guyader O, Aoubid S, Bas A, Cujus A, et al. (2014) The maintenance costs of marine natural capital: A case study from the initial assessment of the Marine Strategy Framework Directive in France. Marine Policy 49: 37-47. https://doi.org/10.1016/j.marpol.2014.03.028
- Mermet L, Homewood K, Dobson A, Billé R (2013) Five paradigms of collective action underlying the human dimension of conservation. In: W.cdonald D, Willis KJ (Eds) Key Topics in Conservation Biology 2. John Wiley & Sons URL: http://onlinelibrary.wiley.com/doi/10.1002/9781118520178.ch3/summary

- Millennium Ecosystem Assessment (2005) Ecosystems and Human Well-Being: Synthesis. Island Press, Washington, DC.
- OCDE (2018) Cost-benefit analysis and the environment: Further developments and policy use. Éditions OCDE, Paris, France. https://doi.org/10.1787/9789264085169-en.
- Pascual U, Adams WM, Diàz S, Lele S, Mace GM, Turnhout E (2021) Biodiversity and the challenge of pluralism. Nature Sustainability 4: 567-572. https://doi.org/10.1038/s41893-021-00694-7
- Persson L, Carney Almroth BM, Collins CD, Cornell S, Wit CA, Diamond ML, Hauschild MZ, et al. (2022) Outside the safe operating space of the planetary boundary for novel entities. Environmental science & technology https://doi.org/10.1021/acs.est.1c04158
- Pittini M (2011) Monetary valuation for ecosystem accounting. Issue paper. UN/World Bank/EEA Expert Meeting on Ecosystem Accounts, London, 5-7 December 2011. 30 pp.
- Quinet A, Bueb J, Le Hir B, Mesqui B, Pommeret A, Combaud M (2019) La valeur de l'action pour le climat. France stratégie
- Radermacher W, Riege-Wcislo W, Heinze A (1999) A statistical?analytical methodology for the construction of abatement cost curves. International Journal of Sustainable Development 2 (1). https://doi.org/10.1504/ijsd.1999.004309
- Radermacher WJ (2020) Official statistics 4.0: verified facts for people in the 21st century. Springer International Publishing https://doi.org/10.1007/978-3-030-31492-7
- Rambaud A, Richard J (2015) The "Triple depreciation line" instead of the "Triple bottom line": towards a genuine integrated reporting. Critical Perspectives on Accounting 33: 92-116. https://doi.org/10.1016/j.cpa.2015.01.012
- Rambaud A, Chenet H (2021) How to re-conceptualise and re-integrate climate-related finance into society through ecological accounting? Bankers, Markets & Investors 166: 20-43. https://doi.org/10.54695/bmi.166.3366
- Razzaque J, Visseren-Hamakers I, Gautam AP, Gerber L, Islar M, Karim MS, Williams M, et al. (2019) Options for decision makers. In: IPBES Secretariat, Bonn, Germany (Ed.) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.
- Riahi K, Schaeffer R, Arango J, Calvin K, Hasegawa T, Jiang K, Kriegler E, Matthews R, Robertson S, Sebbit AM, Steinberger J, Khourdajie AA, Brutschin E, Byers E, Carleton T, Clarke L, Chaturvedi V, Chen W, Martínez JT (2022) Mitigation Pathways Compatible with Long-term Goals. In: Cambridge University Press (Ed.) Climate Change 2022: Mitigation of Climate Change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK and New York, NY, USA. https://doi.org/10.1017/9781009157926.001
- Rockström J, Steffen W, Noone K, et al. (2009) A safe operating space for humanity. Nature 461 (7263): 472-475. https://doi.org/10.1038/461472a
- Sen A (1995) Rationality and social choice. American Economic Review 85 (1): 1-24.
- Spash CL, Hanley N (1995) Preferences, information and biodiversity preservation. Ecological economics 12 (3): 191-208. https://doi.org/10.1016/0921-8009(94)00056-2
- Steffen W, Richardson K, Rockström J, Cornell ES, Fetzer I, Bennett EM, Sörlin S (2015) Planetary boundaries: Guiding human development on a changing planet. Science 347 (6223): 1259855. https://doi.org/10.1126/science.1259855
- Stiglitz JE, Sen A, Fitoussi J-P, et al. (2009) Report by the commission on the measurement of economic performance and social progress.

- Szałkiewicz E, Jusik S, Grygoruk M (2018) Status of and perspectives on river restoration in Europe: 310,000 Euros per hectare of restored river. Sustainability 10 (1): 129. https://doi.org/10.3390/su10010129
- United Nations (Ed.) (1993) Integrated environmental and economic accounting, Studies in methods, Handbook of National Accounting. United Nations, New York.
- United Nations, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank (2003) Integrated environmental and economic accounting 2003, Serie F.
- United Nations, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank (Eds) (2009) System of national accounts 2008. United Nations, New York.
- United Nations (Ed.) (2021) System of Environmental-Economic Accounting — Ecosystem Accounting (SEEA EA). White cover publication, pre-edited text subject to official editing. URL: https://seea.un.org/ecosystem-accounting
- United States Bureau of Economic Analysis (1995) Conference papers. Second meeting of the London Group on natural resource and environmental accounting, Washington D.C., 1995. 432 pp.
- Usubiaga-Lião A, Ekins P (2021) Time for science-based national targets for environmental sustainability: An assessment of existing metrics and the ESGAP framework. Frontiers in Environmental Science 524.
- Vanoli A (1995) Reflections on environmental accounting issues. Rev Income Wealth 41: 113-137. https://doi.org/10.1111/j.1475-4991.1995.tb00104.x
- Vanoli A (2017) The Future of the SNA in a broad information system perspective. Review of Income and Wealth 63: 238-265. https://doi.org/10.1111/roiw.12332
- Vardon M, Burnett P, Dovers S (2016) The accounting push and the policy pull: balancing environment and economic decisions. Ecological Economics 124: 145-152. https://doi.org/10.1016/j.ecolecon.2016.01.021

Endnotes

*1 See, for example, IPBES (2019), message D10.

*2 Examples in the SEEA-EA include the delimitation of "ecosystem assets" or "ecosystem accounting areas".

*3 “*Boundary objects are those objects that both inhabit several communities of practice and satisfy the informational requirements of each of them. [...] Such objects have different meanings in different social worlds but their structure is common enough to more than one world to make them recognizable, a means of translation. The creation and management of boundary objects is a key process in developing and maintaining coherence across intersecting communities.*” (Bowker and Star 1999)

*4 In this article, we simply use the term *ecosystem* and not "*ecosystem asset*" as this term evokes a narrow notion of value restricting the total economic value to flows of ecosystem services as well as an inclusion of ecosystems within the economy.

*5 We can note that, although the causal attribution of degradation to economic entities was central in the original framing (costs were caused *by economic entities*), such an attribution has later proved not central to this notion with the apparitions of proposals of macro-aggregates with no explicit treatment of who caused the degradation. This is why we present this framing in a way that does not require such attribution.

*6 Note that we use here the latest denomination (United Nations 2021, §12.32-42), though it is interesting to comment on the progressive evolution of their denomination and related source of confusion. Starting from the original – more accounting-driven – distinction between *cost-borne* and *cost-caused* approaches, these contrasted framings of ecosystem degradation valuation where carry over the 2003 version into the – more economic-driven – distinction between *damage-based* and *cost-based* approaches. Both approaches were later understood to focus on valuation methods in welfare values, though this is not the case. The recent shift away from welfare value leads to explicitly move away from damage-based methods (United Nations 2021, §12.6) though maintaining the main focus (on lost benefits). In parallel, the meaning of "cost-based approaches" also changed. After the presentation of a paper at the 2011 expert meeting on ecosystem accounting (Pittini 2011), cost-based methods are no longer opposed to damage-based methods as two contrasting ways to frame the valuation of ecosystem degradation, but they more widely describe all valuation methods that use some "cost" notion to value either *ecosystem degradation* (maintenance costs, avoidance costs, restoration costs, abatement costs etc.) or *ecosystem services* (replacement costs, mitigation costs, travel costs etc.). What is further confusing is that, when used to estimate ecosystem services, "cost-based" methods for valuing ecosystem services can actually be used in a "damage-based" framing for valuing ecosystem degradation. This explains why “cost-based methods” has now to be specified as “restoration (or maintenance) cost-based approaches” (rather than simply “cost-based approaches to valuing ecosystem degradation”).

*7 The "*Driving force" - "Pressure" - "State" - "Impact" - "Response*" (DPSIR) framework (European Environment Agency 1999).

*8 Which he proposes to rename System of National *Economic* Accounts (SNEA) instead of SNA to make explicit its restricted scope to the economic sphere.

*9 In this sense, the ecological debt refers to the the debt defined in a "prospective" way in Germain and Lellouch (2020).

*10 We shall note from now on that our proposal could differ from the initial one in several regards. In particular, from an accounting technical point of view, Vanoli (2017) proposed to increase the final consumption (and thus the savings) of the institutional sectors that impacted the environment in order to balance the accounts with the new debt. We may consider a different treatment of the ecological debt more in line with the CARE model (Rambaud and Richard 2015) and follow a more classic recording, in the manner of what is done for a financial debt: an entry is made on the liabilities side and another, of an equal amount, is made on the assets side. This requires to create a particular category of natural assets, which corresponds to the way in which debts are used by sectors. For example, for a climate debt, the corresponding asset is "CO_2 warehousing". This approach also guarantees the double entry and the balance of the accounts (Rambaud and Chenet 2021). Although theoretically sound from an accounting perspective, its actual content and quantification still requires further work.

*11 We will focus on how to build the biophysical information system and the physical and monetary debts accounts. We will not discuss the accounting treatments in the sequence of accounts which would deserve a more extensive discussion.

*12 Note that such a focus on (economically) efficient trajectories could be discussed due to non-market side effects, complex dynamic effects (lock-in, uncertainties, knowledge gaps etc.) or difficult trade-offs. As a result, efficient trajectories coming from these models may not be socially desirable and the resulting estimation may indicate a lower bound of required costs, whose practical reach, for instance, as an indication of funding needs, requires careful consideration.

*13 This would be all the more critical as the benefits of some measures, such as nature-based solutions, is their *multifunctionality*, i.e. their potential to address multiple conservation and societal issues at the same time. Failure to account for these synergies may induce a systematic bias against such measures.

*14 Note that such reference levels may - but need not - reflect actual legal obligations. They may be defined and distributed amongst economic units (corporations, government etc.) according to conventional rules for reporting purposes.

*15 Note that our approach is only possible if there exists at least one reasonable way to prevent (*ex ante*) or mitigate (*ex post*) ecosystem degradation. Otherwise, we lie outside the scope of ecological liability accounting. The need to acknowledge that the impacts are irreversible or their cost "disproportionate" would lead to redefine new, more realistic, targets and related condition indicators. This limitation can also be a strength from a practical perspective as it allows identifying realistic (and thus more likely to actually take place) actions for sustainably managing ecosystems. In addition, it goes along with the need to define precautionary targets on pressures, especially when facing irreversible risks as argued elsewhere in this article.

*16 In the CARE accounting model, preservation costs are the expenditure that do not change an organisation's business model and whose primary function is to preserve the environment. They include reduction costs (prevention actions) and restoration costs (repairing actions). In contrast, avoidance costs are related to actions that change the business model with the secondary objective of having less impact on the environment (e.g. electric cars). To prevent double-counting and “hard modelling”, they are not included in the calculation of ecological debts.

*17 As change in prices can be interpreted as resulting from changes in individual preferences and change in public spending from changes in collective preferences in national accounting.

*18 The idea as to the possibility to define a pristine, natural state on objective bases is also a controversial idea since humans have shaped ecosystems for a very long period (Ellis et al. 2021).

*19 These requirements are the widely used SMART criteria: “Specificity”, “Measurablity”, “Ambition”, “Realism” and “Time-bound”. More precisely, “Specificity” requires targets are set at levels suitable for implementation (scales, time horizons, sectors), “Measurablity”, that indicators are specified, so that progress towards the targets can be evaluated, “Ambition”, that the norm used is fully consistent with the existing targets' ambition and “Realism” that a credible action plan consistent with reaching related targets is made explicit.

*20 *Dimensions of interest* can be related to ecosystem *extent* or *condition* as some forms of ecosystem degradation (e.g. drainage of wetlands, deforestation for agricultural purpose, conversion of pasture to cropland, desertification, land-take or coral reef bleaching) may be reflected in the accounts as changes in types depending on the retained ecosystem typology.

*21 Here, we mean *value* in a broad and inclusive sense consistently with IPBES (2022) recent key messages to policy-makers. They can be *use values*, but also, for instance, *values as principles* such as the precautionary principles which motivate safe minimum standards in the spirit of the planetary boundaries framework (Rockström et al. 2009, Steffen et al. 2015).

*22 Though we rephrased the term *capacity*, initially used by Comte et al. (2020), we maintain an "upstream" perspective by considering under this category the dimensions included in the measurement of ecological condition that inform us about the capacity of ecosystems to deliver specific services in a sustainable manner, as are, for example, the indicators associated with descriptors 3 and 9 in the Marine Strategy Framework Directive. These are dimensions of interest because they can be directly related to the ability of the ecosystem to deliver a given service sustainably and at an optimal or satisfactory level (e.g. distance to MSY for the different dimensions of a fish stock). It is not necessarily one-dimensional and it is not required that these dimensions be expressed in the same unit as the ecosystem service concerned as recommended by the SEEA-EA (United Nations 2021, § 6.149) for *capacity*.

*23 Ecosystem services are defined, in the original and broad sense, as "the benefits people obtain from ecosystems" (Millennium Ecosystem Assessment 2005). We define *benefits* as an increase in some explicit dimension of individual or collective welfare, such as, for instance, those identified by the Stiglitz, Sen and Fitoussi Commission (Stiglitz et al. 2009).

*24 Such values, such as the precautionary principle, are closely related to our collective attitudes towards risks and uncertainties.

*25 Regarding non-use values, we find in the SEEA-EA that “it is not considered, from an accounting perspective, that a transaction has taken place consistent with the framing used for recording ecosystem services in the SEEA EA” so “these values can [only] be presented in complementary valuations” (United Nations 2021, §6.72-73).

*26 We may first note that measures of the costs of degradation at the national scale are already explicitly required, for example, in the initial assessment of the MSFD (Levrel et al. 2014).

*27 For example, reducing environmentally harmful subsidies and increasing payments for environmental services, environmental taxation, green public procurement or environmental disclosure.

*28 A *theory of change* is an explicit account of how an intervention (for example, here the development of a specific ecosystem accounts or indicator) would entail specific outcomes at different time horizons. By requiring an explicit representation of the causal linkages that connect an intervention with its impacts, it provides the basis of a rational discussion. For examples linking accounting to conservation outcomes, see, for example, Mermet et al. (2013) or Feger et al. (2019).