Synergies and complementarities between ecosystem risk assessment and ecosystem accounting

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Abstract

Safeguarding biodiversity and human well-being depends on sustaining ecosystems. Global agreements, such as the Post-2020 Global Biodiversity Framework and UN Sustainable Development Goals, aim to halt and reverse loss and degradation of ecosystems, associated biodiversity and ecosystem services. They require standardised information for quantifying where ecosystem loss and degradation is occurring, and where action can most effectively mitigate its impacts. Two global standards developed to quantify ecosystem changes are: 1) the IUCN Red List of Ecosystems (RLE) protocol for assessing the risk of ecosystem collapse; and 2) the United Nations System for Environmental Economic Accounting Ecosystem Accounting (SEEA EA), which tracks change in ecosystem assets and their contributions to the economy and human well-being. In this paper, we explore the similarities between the frameworks, identifying common concepts, models and data, and highlight differences in their conceptual framings and practical implementation that make them complementary in environmental policy and decision-making. We illustrate how information on ecosystem classifications, maps, extent and condition can be shared between RLE assessments and SEEA EA, through three case studies: South Africa, Colombia, and Meso-American Coral Reef. Key differences include the RLE's focus on ecosystem dynamics and risk of collapse, consideration of future projections, and its explicit treatment of uncertainty. We recommend that ecosystem risk assessment and accounting should not be treated as unrelated processes nor undertaken in isolation of each other, or they risk producing outputs, in particular ecosystem classifications, maps and condition indicators, that are needlessly inconsistent or even contradictory. Finding pathways for sharing data and knowledge between frameworks, purposes and user-groups, as well as their complementary roles in environmental reporting and policy, will support more efficient and effective action to safeguard biodiversity, ecosystems and their contributions to people.

Introduction

Ecosystems are vital to sustaining biodiversity and human well-being through ecosystem services provision such as food, water and disaster risk reduction[1,2]. Ecosystem loss and degradation present a major challenges of the 21st century, with concomittant declines in biodiversity and ecosystem functions and services [3-5] posing significant threats to the economy and human well-being [6,7]. To address ecosystem loss and degradation, multiple policy agreements have been developed, including the UN Sustainable Development Goals and the Post-2020 Global Biodiversity Framework under the Convention on Biological Diversity. These require standardised approaches to measuring change in biodiversity and the benefits it provides, to understand current states and trajectories to support decisions, and to monitor change through time [8-10]. For example, biodiversity risk assessment approaches, such as the IUCN (International Union for Conservation of Nature) Red List of Threatened Species [11] and Red List of Ecosystems (RLE) [12], are used in national and in global policy to assess the state of biodiversity and trends [13,14], and guide decisions [15,16]. Social-ecological indicators, such as the Genuine Progress Indicator (GPI) [17], Inclusive Wealth Index (IWI) [18], and Gross Ecosystem Product (GEP) [19], capture the value of the nature and natural capital to people and the economy [20]. Global standards and initiatives have been established to support countries in recognising their natural capital, such as the United Nation's (UN) System for Environmental Economic Accounting (SEEA) [21], and the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) partnership initiated by the World Bank [22]. While these approaches can be viewed as alternative ways of measuring change for different purposes, understanding how they relate to one another, including synergies such as shared concepts and data, and differences, such as perspective or objective that make them complementary, can provide insight into different dimensions of environmental change and efficiencies in monitoring and data compilation.

Two relatively new approaches that measure ecosystem change – ecosystem risk assessment and natural capital accounting – are increasingly applied to inform policy at global, national and sub-national levels. The IUCN Red List of Ecosystems (RLE) was adopted by IUCN as the global standard for ecosystem risk assessment in 2014. It evaluates the risk of ecosystem collapse, using quantitative criteria relating to change in ecosystem extent and degradation to place ecosystems in ordinal categories of risk [23]. RLE has seen broad application and uptake, including assessment of over 3000 ecosystem types in more than 100 countries, influence on national legislation, impact on ecosystem management [2,15,24], and applications in national and global monitoring (e.g. the Post-2020 Global Biodiversity Framework and SDGs) [2,13].

Natural capital accounting aims to provide a consistent means of reporting on stocks and flows of natural resources. The global standard for natural capital accounting is the UN's System for Environmental Economic Accounting (SEEA) [21], which is an internationally agreed statistical standard for combining environmental and economic information presented in the form of different types of accounts. The SEEA applies accounting principles that are consistent with the System of National Accounts (SNA), which provides the global standard for national accounts including macroeconomic aggregates such as gross domestic product (GDP). The SEEA Central Framework was adopted by the UN Statistical Commission in 2012, and in March 2021 the most recent volume of the SEEA—SEEA Ecosystem Accounting (SEEA EA) was adopted as a statistical standard by the UN Statistical Commission, which tracks changes in ecosystems and their contributions to people and the economy

[25]. At least 89 countries are implementing the SEEA, with most of the end users being governments [26], while SEEA EA has been applied in varying degrees of detail in 34 countries [27]. Indicators derived from SEEA EA have been proposed to support the Post-2020 Global Biodiversity Framework [28,29] and the SDGs [30,31].

In this study, we sought to improve understanding of the relationships between RLE and SEEA EA, including how knowledge and data from one can inform the other. The implementation of each requires extensive time, effort and data analysis, thus we analysed which concepts and data are common to both frameworks, and how the frameworks are complementary, rather than competing, in terms of their capacity to support environmental policy. We examine similarities and differences in six core areas:

1. Objectives and approaches
2. Definition and conceptualisation of ecosystems and ecosystem types
3. Practical implementation, including spatial scope and mapping of ecosystems
4. Measuring change in ecosystem extent
5. Measuring change in ecosystem condition
6. Consideration of ecosystem dynamics and uncertainties

Potential relationships between RLE and SEEA EA are summarised in Figure 1, and we present three case studies (Boxes 1-3) that illustrate these links . Because the RLE standard was adopted several years before SEEA EA, more RLE assessments have been completed than ecosystem accounts. Our case studies reflect this, with information developed for RLE assessments being used to compile ecosystem accounts. However, in principle the relationship could be the reverse, with information developed for ecosystem accounts being applied in RLE assessments, as well as other applications. We conclude with a discussion of recommendations on how to best use the two frameworks together, and on how each can enhance the other.

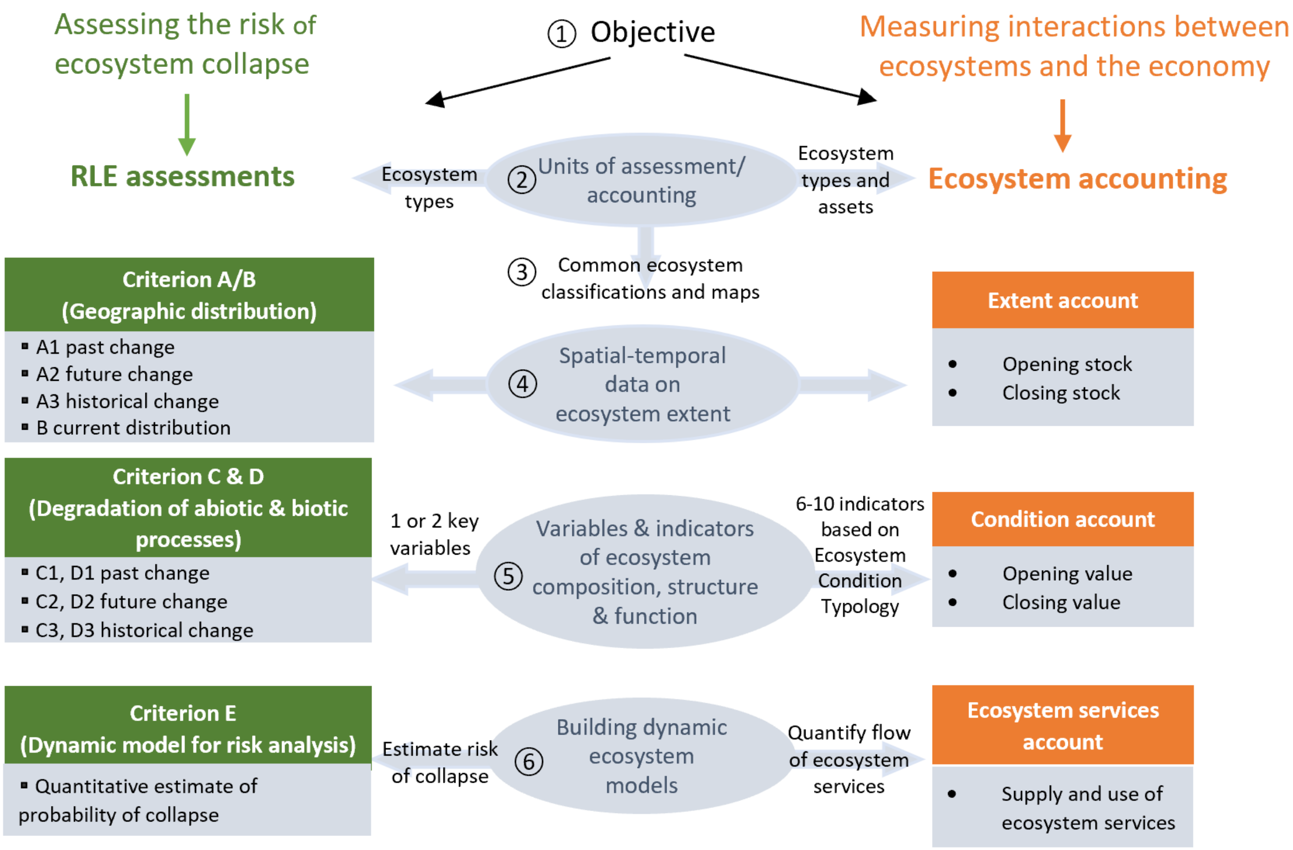

Figure 1. Potential commonalities in information (grey ovals) required for RLE assessments (green) and ecosystem accounts (orange), from ecosystem conceptualization and classification, maps of ecosystem types, estimates of change in extent and condition. *①-⑥ corresponds to our six core areas of discussion between RLE and SEEA EA.*

1. Objectives and approaches for RLE and SEEA EA.

While RLE and SEEA EA aim to provide comprehensive assessment of ecosystem change through the collation and synthesis of data and evidence, their perspectives differ: RLE focusses primarily on risks to biodiversity at the ecosystem level, while the SEEA EA has as its primary aim to quantify links between ecosystems and the economy. These differences have some implications for synergies between the frameworks, as well as their complementarity.

The overall objective of RLE is to support conservation and management of biodiversity by identifying ecosystems most at risk of loss of biodiversity and ecological processes [23,32], which underpin ecosystem services. Because ecosystem changes that promote some ecosystem services can be detrimental to biodiversity, RLE does not focus on risk of ecosystem service loss, though it can support such assessments (see Box 3). RLE assesses the risk of ecosystem collapse, where collapse is defined as a transformation of identity through the loss of defining biotic and abiotic features (see SI.1 Glossary) [23,32]. RLE comprises five quantitative criteria [23], each with sub-criteria and quantitative thresholds: A) change in ecosystem distribution or extent, B) restricted distribution with ongoing threats, C) degradation of the abiotic environment and processes, D) change in biotic features and processes, and E) the probability of ecosystem collapse estimated with an ecosystem model. Criterion A, C and D can be assessed over three timeframes (while criterion B focuses on the current distribution), under different sub-criteria: 1) the last 50 years; 2) any 50 year window including the future and recent past; and 3) historical change, typically since 1750. Through quantitative assessment of state and trends under each criterion, an ecosystem is allocated to an ordinal risk category that indicates a relative risk of collapse, from Critically Endangered, Endangered, Vulnerable, Near Threatened, to Least Concern. All criteria should preferably be assessed, including at least one of the functional criteria (C and D), but an assessment can be considered complete even if one or more criteria are not assessed, making the RLE a flexible tool even in data constrained contexts. Ideally the RLE assessment is updated every 5-10 years, though this may vary across countries.

The aim of SEEA EA is to provide a consistent means of reporting on stocks of ecosystem assets, flows of ecosystem services and their contributions to people and the economy [25], applying the same accounting principles as the SNA. SEEA EA collates and organises information on ecosystems and their changes [25], and comprises five core accounts: 1) ecosystem extent account, which measures ecosystem area and changes in area across ecosystem types; 2) ecosystem condition account, which records the “quality” of an ecosystem in terms of its abiotic and biotic characteristics; 3) ecosystem services supply and use account, measured in biophysical terms (e.g. cubic metres or tonnes), which records the flows of different ecosystem services, identifies the users of those services, and assesses how those patterns of supply and use change over time; 4) ecosystem services supply and use account in monetary terms, which provides a consistent way to compare different ecosystem services and assets with standard measures of products in the SNA; and 5) ecosystem monetary asset account, which records the net present value of the ecosystem services provided by the ecosystem assets. A modular approach can be taken, where a country can compile those accounts that are most useful and for which data are available. SEEA ecosystem accounts are intended to be updated regularly, ideally annually but with the option of compiling ecosystem accounts less frequently depending on the data availability and factors such as available capacity to do so.

2. Conceptualizing the ecosystem and ecosystem types.

The definitions of ecosystems used by RLE and SEEA EA are consistent with the Convention on Biological Diversity definition of an ecosystem as a dynamic complex of plant, animal and micro-organism communities and their non-living environment [23,25] (see SI.1 Glossary). Ecosystem types, the units of assessment for both RLE and SEEA EA, represent distinct sets of biota, abiotic environment, key processes and interactions between and amongst them. The concept of ecosystem type is scalable (e.g. from local to global), depending on the scope and purpose of an ecosystem assessment. Both standards use the IUCN Global Ecosystem Typology (GET) as their reference classification system, aligning them conceptually, in terms of the core definitions of ecosystem types, and practically, by allowing comparision and aggregation across countries, integration of existing national classifications and maps and development of new ecosystem classifications at the country level (e.g. in Myanmar [33]). The GET is comprehensive, including all ecosystem types across marine, terrestrial, subterranean, and freshwater environments, in a hierarchical structure that classifies ecosystems based on their functional characteristics at the upper levels, and compositional features at lower levels [34].

RLE assessment requires a qualitative understanding of ecosystem dynamics, typically represented in a diagrammatic conceptual model that captures key biotic and abiotic features and processes of a particular ecosystem type. The conceptual model forms the foundation of the assessment, describing a shared understanding and diagnosis of the key features of the ecosystem and how they are threatened by drivers of biodiversity loss (an example conceptual model for a coral reef ecosystems is shown in Box 3), and helps to identify indicators for measuring important changes [12,23,35]. Such models can also be used to support the development of ecosystem accounts, although not prescribed in the SEEA EA standard. For example, in South Africa, a driver-response model was used as the conceptual model for river ecosystems to directly inform the indicators for the river condition account [36].

Ecosystems exist along a spectrum from relatively natural and undisturbed, to anthropogenic, where human intervention is the dominating influence on the biota and abiotic processes (also known as anthromes) [34,37]. RLE focusses primarily on risks to more natural ecosystems where anthropogenic changes may threaten indigenous biodiversity (although it has been used for some semi-natural ecosystems such as managed woodlands and arable lands in European Red List of Habitats [38]) [39,40]. In constrast, SEEA EA explicitly aims to include all ecosystems, including natural, semi-natural and anthropogenic ecosystems, in part to ensure spatially comprehensive reporting of changes in ecosystems over time, which is a requirement of accounting.

3. Practical implementation, including spatial scope and mapping of ecosystems

The fundamental starting point for RLE and SEEA EA is the need to classify and map ecosystem types, which in SEEA EA can be further broken down to ecosystem assets (individual occurrences of a particular ecosystem type [41]). The same ecosystem classification and maps can form the foundation of both RLE assessments and SEEA EA, as demonstrated in Norway [42] and South Africa [43,44] (Box 1), providing both efficiencies, given the investment required to produce accurate maps of ecosystem types, and consistency across sectors. The need to update SEEA ecosystem accounts frequently can provide impetus for increased production of regular updates on changes in ecosystem extent and condition, supporting more regular and timely updates to RLE assessments.

The scope of assessment for RLE and SEEA EA is flexible. Both are most typically applied within national jurisdictional and administrative boundaries, where all ecosystem types are systematically assessed within a country or region. Alternatively, both RLE and SEEA EA can be conducted for one specific ecosystem or a group of ecosystems, at regional, national, or global scales (for example, all coral reef ecosystems in a region) [15], although small assessment areas can present methodological challenges for RLE, including where the ecosystem type extends beyond the scope of the assessment (e.g. into a neighbouring jurisdiction) [12]. SEEA EA uses the term 'ecosystem accounting area' to define the spatial scope for a set of accounts, which can can range from whole countries [45] to highly placed-based accounts, such as an account for a particular national park or bay [46,47], and generally includes multiple ecosystem types. This may result in mismatches in scope between RLE assessments and SEEA ecosystem accounts. Conversely, RLE assessments may not be compatible with providing a complete assessment of ecosystems within an ecosystem accounting area by excluding anthropogenic ecosystems.

4. Measuring change in ecosystem extent

Both RLE and SEEA EA assess change in ecosystem extent: as the basis for criterion A in RLE, and in ecosystem extent accounts in SEEA EA. However, they differ in two ways: they measure change over differing timeframes; and RLE evaluates the magnitude of change against specified thresholds to allocate risk categories.

RLE assesses change in extent over three timeframes (forming sub-criteria for criterion A): change over 50 years (past or future), and/or relative to a historical reference point, typically since industrialisation (ca. 1750). A minimum of two time points is required to assess the criteria, but denser time series are encouraged to understand rates of change and trajectories [12]. In constrast, SEEA EA can accommodate variable timeframes for measuring change in extent, including annual change (ideal) or less frequent change (e.g. five yearly intervals). Ecosystem extent accounts may also include a historical reference extent (similar to the pre-industrialisation extent used in RLE) or other fixed baseline extent (e.g. in South Africa, Box 1). Although an annual accounting period may be considered ideal, longer accounting periods are quite common practice in SEEA EA [25]. In translating data from RLE to EA, the assessment time points in RLE could be used as opening and closing stock in extent accounts (see Figure 1), while conversely data from extent accounts can be used in RLE criterion A, provided time-series are of sufficient length. Thus data on ecosystem extent can be readily shared between the two frameworks, even where the purpose, format and associated analyses and indicators differ (see case studies in Boxes 1-3).

The intent of RLE is not simply to record change, but to provide insight into how the magnitude of change increases risk of biodiversity loss. Therefore change in area is assessed against threholds that allocate ecosystems to a risk category, allowing consistent comparison between ecosystem assesments; this provides a simple warning system to flag threatened ecosystems (see supplementary material S2.1 for thresholds). For example, if an ecosystem's distribution has declined by between 50% and 80% in the last 50 years, its risk category is *Endangered*, i.e. it is considered to be at a very high risk of collapse [12]. Comparable methods are used to flag ecosystems at risk due to degradation (see below). RLE also assess an ecosystems risk based on restricted distribution with ongoing threats (criterion B). Here metrics of current extent are assessed against fixed thresholds, enabling an evaluation of whether the ecosystem is intrinsically at risk due to stochastic threats [48].

[Box 1 here]

5. Measuring change in ecosystem condition

Both RLE and SEEA EA frameworks are designed to support quantitive assessment of change in ecosystem condition (in SEEA EA) or degradation (in RLE). In RLE, criterion C quantifies degradation of the abiotic environment, while criterion D assesses disruption to biotic processes and interactions. In SEEA EA condition accounts provide a structured approach to recording data on the state and functioning of ecosystem assets, using a combination of relevant variables that are converted into normalised indicators and can be aggregated to a condition index. While there is potential for shared data (Figure 1), in practice, there are differences between the EA condition account and RLE criteria C and D that require additional diagnosis, analysis and collation of evidence to allow the two frameworks to support each other, in particular, the selection and number of variables, and approach to converting the variables into indicators of ecosystem change.

The selection of condition variables in SEEA EA and RLE follow similar principles — select variables that reflect a key role in ecosystem processes, and are sensitive to change [23,25,49,50]. Differences in the set of variables selected may arise due to differences in the aims of the frameworks. RLE aims to select variables that are most relevant to risk from a biodiversity perspective and where a quantitative threshold for collapse can be specified (typically 1-4 indicators); this allows a meaningful analysis of proximity to ecosystem collapse. Indicator selection is supported by the development of conceptual models specific to different ecosystem types, which help identify key processes, features, threats and pathways to ecosystem collapse. In contrast, SEEA EA aims to cover a greater range of relevant ecological information, because these data may be used in a variety of ways, for example to analyse change in ecosystem capacity to supply a particular ecosystem service. SEEA EA calls for a parsimonious set of variables and indicators (approximately 6-10), guided by an Ecosystem Condition Typology, which includes biotic, abiotic and landscape- or seascape-level characteristics [25]; these accounts could provide an expanded set of indicators for RLE assessments, if other RLE indicator selection criteria are met. Because a conceptual model of the ecosystem type is not required as part of SEEA EA, the relationships between the different variables for a particular ecosystem type may not be explicitly considered – we discuss this further below (section 6).

Both frameworks convert variables into indicators of ecosystem change that are scaled from 0 to 1. This requires values to be nominated that enable scaling, at both 0 and 1, which are often termed reference levels [25,51]. There are three potential differences in terms of the reference levels and normalisation of variables for RLE and SEEA EA. First, in RLE, indicators are normalised to measure *relative severity* of degradation (scaled between the initial state and the collapsed state). RLE requires that collapse thresholds are defined for each indicator; for example, in the Meso-American Reef [52], live coral cover in a coral reef is selected as the most important variable, with collapse occurring between 1-5% live coral cover. In contrast, SEEA EA does not require an explicitly defined threshold for ecosystem collapse or for the lower bound of condition for a given indicator.

Second, the *reference condition* (or reference state) in SEEA EA can take a range of forms, including '*undisturbed condition/natural state'*, which may be a historical state, or a *contemporary condition* (e.g., best of what's left, or best-attainable condition), depending on the ecosystem considered and data availability [25]. In contrast, degradation in RLE is assessed over specific timeframes (50 years or since 1750. The historical baseline (pre-industrial at circa 1750) in RLE (criterion A3, C3 and D3) can be used for SEEA EA as a proxy for natural condition (or termed the initial state), prior to major modification of the landscape or seascape (for example, in South Africa, Box 1). The 50 year timeframe of the RLE can also be used in accounts where data availability is limited (e.g., Colombia in Box 2). This does not assume that the starting point of the 50 years was in 'natural' or 'good' condition, but rather measures how rapidly an ecosystem is changing.

Third, changes in condition are summarised in different ways. RLE measures degradation of an ecosystem over two dimensions: the relative severity of degradation, and the extent of area degraded over the assessment timeframes; these are combined to assign the risk categories, allowing multiple pathways of degradation to be recognised, for example high levels of degradation over a part of the distribution, or moderate levels of degradation over the whole extent (Figure S1, also see Table S1, and Figure S18 from [35]). In contrast, SEEA EA combines both into one condition account by measuring condition for each ecosystem asset within an ecosystem type, and aggregating (e.g. averaging) condition across assets (section 5.4 in [25], Table S2,S3).

[Box 2 here]

6. Incorporating ecosystem dynamics and uncertainties

Three points of difference between RLE and SEEA EA lie in how they address ecosystem dynamics, future trends and uncertainties. RLE asks assessors to develop a qualitative conceptual model of the ecosystem type to underpin its assessment [12,23], that depicts understanding of dynamics and relationships of defining ecosystem features and causal pathways for threatening processes (e.g. Figure 4 [35,53,54]). SEEA EA does not specify the use of such models for the compilation of accounts but accounting practice would benefit from their use in a number of ways. First, an ecosystem-specific conceptual model could facilitate the selection of the most relevant biotic and abiotic variables for different ecosystem types, similar to its role in RLE. This would improve complementarity and reduce redundancy of indicators [25,55], and identify data gaps, for example, aspects of an ecosystem type that are not captured with currently selected indicators. Second, a model can reveal potential relationships within and between accounts, for example how changes in one condition indicator may correlate with changes in other condition indicators, and how changes in condition indicators could influence the flow of ecosystem services, which is quantified in ecosystem service accounts (see Box 3 and Table 3). Third, a conceptual model can help managers to diagnose threats to the ecosystem concerned and identify the best actions to mitigate them (e.g., [56-58]). Finally, conceptual models can be efficient communication tools that summarize key features of an ecosystem type for different stakeholders and the wider community.

Conceptualisation of ecosystem dyamics can be extended to develop quantitative, process-based ecosystem models [59]. The RLE framework has provision for the application of such process-based ecosystem models (e.g. under criterion E to estimate a probability of collapse, e.g. [35,60-62], although in practice this criterion is rarely used due to the lack of appropriate ecosystem models for most ecosystems. Such models can also be used for hindcasting, forecasting and scenario analysis (see below), examining emergent properties of ecosystems, including interactions between multiple threatening processes, and modelling ecosystem services (for ecosystem service accounts), through modelling human use or via production functions (see Box 3).

Assessment of projected or forecast future trends is explicitly included in the RLE framework (under subcriterion 2 in criterion A, C and D). This can be achieved through a range of modelling approaches, including correlative models, such as bioclimatic envelope models (e.g. [23,60] and Box 2), by extrapolating current trends in key variables [63-65], and process-based ecosystem models (Box 3). Such models can allow for the exploration of alternative scenarios, for example of climate change, resource use or socio-economic pathways, to estimate the most likely trajectories of future risk. They have strong potential for application in testing alternative policies or management actions. Although accounting is focused on recording information *ex-post*, i.e. for past periods, accounting information, especially in the economic domain, is used to underpin a vast array of scenario analysis and modelling work. Using SEEA EA accounts in comparable scenario analysis or forecasting is in its infancy [66], and there is substantial potential to inform decision-making by exploring alternative management or policy scenarios. Future projections can also contribute to calculating Net Present Value (NPV) in the ecosystem monetary asset account that forms part of the SEEA EA framework, based on the projected future supply and use of ecosystem services from an ecosystem asset or type [25].

Uncertainties exist in many aspects of conservation decision-making, that can lead to unexpected or undesired outcomes [67-70]. There are many types and sources of uncertainty in RLE assessment and accounting [12,71], including model uncertainty associated with ecosystem classifications and indicator selection, and epistemic uncertainty from mapping error, imperfect measurement of indicators, and (inevitable) incomplete spatial and temporal data [72,73]. RLE addresses some of these uncertainties in the assessment process, mostly epistemic uncertainties in estimated values (e.g. degree of decline in area or degradation) and collapse thresholds, by transparently describing sources of uncertainty, and presenting bounded values for collapse thresholds and estimates of change (analogous to reporting error estimates around mean values). These are propagated into risk assessment outcomes by reporting the most likely risk category as well as plausible upper and lower bounds of the assessment outcomes [12,74]. Uncertainties around ecosystem classification remain largely implicit, though the development of the IUCN Global Ecosystem Typology is likely to increase the consistency of classification between individual scientists and jurisdictions, and hence reduce uncertainty related to subjective bias. SEEA EA, like the SEEA Central Framework and the SNA [75], focuses on identifying and presenting the best estimate of each variable, and measures of uncertainty are generally not presented. Drawing on the experience in dealing with uncertainty in conservation and ecology could strengthen the rigour and transparency of ecosystem accounting, for example, presenting ecosystem accounts with bounded values and error terms to reflect potential confidence in the estimates, making the accounts more informative and reliable. Further works on practical ways to include those uncertainty measurements in accounting tables would need to be explored.

[Box 3 here]

Conclusions

Global standards for measuring ecosystem change are critical for understanding biodiversity loss and degradation, its impacts on human well-being, and designing policies and management actions to halt or reverse it [2,32]. Understanding the relationships between global standards developed for different purposes is key to efficient and meaningful reporting, reducing redundancies and improving complementarities between frameworks [76-78]. Our comparative analysis of two global standards for measuring ecosystem change, RLE and SEEA EA, found important similarities and differences between them. Both frameworks are essential to the sustainable development toolkit because they are means to different but often complementary ends: SEEA EA aims to collate consistent data on ecosytems and their interactions with people and the economy, with the potential to provide information for a range of purposes; whereas RLE has a clear focus on assessing risks of ecosystem collapse and biodiversity loss, with information that can be used in prioritising conservation actions, including protected area expansion, ecosystem management, and restoration [15,79-81]. Nothwithstanding the different purposes for which they were developed, there are multiple potential synergies between the frameworks that emerge from our analysis.

The most obvious and fundamental synergy is shared knowledge and data, including ecosystem classifications and maps, change in extent and trends in indicators of condition or degradation that are essential for both RLE and SEEA EA; these are illustrated in the examples we provided for South Africa, Colombia, and Meso-America Reef. One can start with either RLE or SEEA EA, or undertake them in parallel. However, we recommend the two frameworks should not be treated as unrelated processes nor undertaken in isolation: at best it risks duplicating work; at worst it risks producing contradictory outputs that could confuse users. This is particularly the case for ecosystem classifications and maps and for condition indicators [2]). In South Africa (Box 1), the large overlap in the people involved in RLE and ecosystem accounting, who are part of the same community of practice, helps ensure that concepts such as ecosystem type and ecosystem condition are operationalised in an equivalent way. An important lesson from South Africa is the importance of having active communication and collaboration between the core teams involved in RLE and SEEA EA. In some countries, this collaboration may require new relationships to be formed between the statistical and accounting communities and ecologists.

The greatest unrealised potential for synergies between the frameworks lies in tapping into the understanding of ecosystem dynamics, using quantitative and qualitative models. The application of conceptual models of ecosystem dynamics for different ecosystem types would support the design of accounts that capture all relevant parts of the ecosystem, improve interpretation of changes in these components, and inform sensible selection of indicators and their aggregation into indices of ecosystem condition. The extension of RLE conceptual models to explicitly consider relationships between ecosystems and human well-being (as shown in Figure 4) provides an important basis for reinforcing the importance of ecosystem management and conservation, and mainstreaming RLE beyond conservation. Thus bringing together RLE and SEEA EA can benefit RLE by providing greater context and introducing different audiences to the multiple values of ecosystems as social-ecological systems [82-85]. The application of quantitative models for forecasting would improve the capacity of

SEEA EA to support decisions for policy and management, through evaluation of the impacts of alternative decisions on ecosytsems and human well-being under a range of future scenarios (e.g. Box 3; [66,86,87])

The more frequent reporting needs of SEEA EA (ideally annual, versus 5-10 years in RLE) will require more consistent and repeatable methods for updating data to be developed and funded. Establishing governance and institutional mechanisms to support reliable data streams and work flows, allowing efficient re-analysis and collation of data will allow more frequent, timely and consistent RLE re-assessments. These will be essential if SEEA EA and RLE are to support implementation, monitoring and reporting for the post-2020 Global Biodiversity framework and SDGs [28,29]. Indicators have been developed to summarise RLE data at national or global scales [13] could be aligned with indicators to summarise ecosystem accounts [44]. The intended greater frequency of SEEA EA reporting can also enable the effectiveness of actions aimed at reducing risk of ecosystem collapse (such as protected areas or ecosystem restoration or management) to be evaluated. Available national RLE assessments, including Finland, Norway, the USA and Myanmar, could serve as an important starting point for developing ecosystem accounts [33,74,88]. Although the development of ecosystem accounts is progressing, ecosystem condition accounts are currently the least advanced of the suite of ecosystem accounts (e.g. [19,89-92]); similarly, the majority of published terriestial RLE assessments only assess distribution of ecosystems (criteria A and B)[93]. This reflects the paucity of consistent data on the condition of ecosystems that could underpin ecosystem condition accounts and assessments of degradation under RLE criteria C and D, a gap that should motivate greater investment by governments in data collection, collation and analyses related to ecological condition.

Among the evidence that the connections between the biodiversity, the state of the environment, and human well-being have not been effectively addressed is the recent failure to reach the Aichi targets of the Strategic Framework for Biodiversity 2011-2020 [94,95]. To ensure effective implementation of the post-2020 Global Biodiversity Framework, better measurement of the importance of nature is needed, reflecting its multiple values [96]. Although stakeholders may focus primarily on ecosystem account summaries or RLE risk categories, the assessment processes of both RLE and EA provide of a wealth of information, including highlighting where there are data or knowledge gaps. This underlying information can be as important as the final results, allowing in-depth diagnosis for local management, and providing data streams for further analysis [44], including reporting on national and global goals [2,13]. The complementary roles of RLE and ecosystem accounting in providing credible measurement of both ecosystem risk status, ecosystem assets and ecosystem services can shape improved policies for sustaining nature and human wellbeing.

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Author contributions

HX, EN, and AD conceptualized and led the writing of the paper; all authors contributed to discussion, drafting and writing.

Competing interests

The authors declare no competing interests.

Boxes

Box 1: South Africa—common ground between RLE assessment and ecosystem accounts
South Africa has exceptional biodiversity, characterised by a wide variety of ecosystem types, high species richness and high levels of endemism. South Africa was also a pioneer in undertaking both national RLE assessments and SEEA ecosystem accounts, which makes it a useful case study to explore differences and synergies between RLE and SEEA EA. A national assessment of threat status of ecosystems was undertaken for the first time in 2004 as part of the country's first National Biodiversity Assessment (NBA), covering all realms (terrestrial, freshwater and marine) [97], and repeated in NBA 2011 and NBA 2018, including full alignment with the IUCN RLE criteria in the 2018 assessment [98,99]. The RLE has been mainstreamed into policy and legislation to inform conservation action and decision-making in several sectors [24]. South Africa's first national ecosystem accounts (extent and condition) were developed for river ecosystems in 2014 [36], followed in 2020 by terrestrial ecosystem extent accounts [100] and estuary ecosystem accounts (including extent, condition, and some ecosystem services accounts in physical terms) [101], with marine ecosystem accounts (extent and condition) underway. Ecosystem accounts take information about ecosystems to a wide audience as they are published by the national statistical office.

Both RLE assessments and ecosystem accounts use as their basis the South African National Ecosystem Classification System (SA-NECS), which maps and classifies ecosystems in a nested hierarchy with approximately 1000 ecosystem types across all realms (Figure 2), and aligns well with the IUCN's Global Ecosystem Typology [102]. The successive NBAs provided the major impetus to formalise and standardise the SA-NECS, which provides a nationally endorsed set of spatial data layers of ecosystem types that can be applied for a range of purposes. The existence of the NBA and SA-NECS provided foundational datasets that made it possible for South Africa to develop ecosystem accounts relatively rapidly. The terrestrial ecosystem accounts relied on the same map used in the RLE assessment, in the form of the National Vegetation Map, which delineates the historical extent of ecosystem types. This historical extent provides a stable baseline extent prior to major human modification of the landscape, used in criterion A in the RLE and as the opening stock in the ecosystem extent account (Table 1, Table S2). The current extent of terrestrial ecosystems is measured in a similar way in RLE and accounts, using land cover data to identify areas that have been intensively modified by human activity, but with some differences in how the data is pre-processed and used and differences in the treatment of semi-natural areas. A further essential data layer for both RLE and ecosystem accounting is a spatial assessment of ecosystem condition, which was collated and consolidated from a wide range of sources for the NBA, and used in RLE criteria C and D. The same information can be drawn on to compile ecosystem condition accounts (see example of South Africa's river ecosystem condition accounts in Table S2-S4).

A key lesson is that a national map and classification system for ecosystem types is an essential foundation for both the RLE and SEEA EA. Whether developed to support RLE or SEEA EA is not of special significance. However, if the map and classification is first developed to support ecosystem accounting, it is critical that ecologists and other specialists in natural sciences are centrally involved, so that the maps and classification are suitable not only for ecosystem accounting and associated analysis (e.g. integrated or extended economic modelling) but also for RLE and related purposes. Development of separate ecosystem classifications and maps for RLE and ecosystem accounting should be avoided. Apart from duplicating effort and increasing costs, this could result in confusing findings and inconsistent messages to policymakers and decision makers.

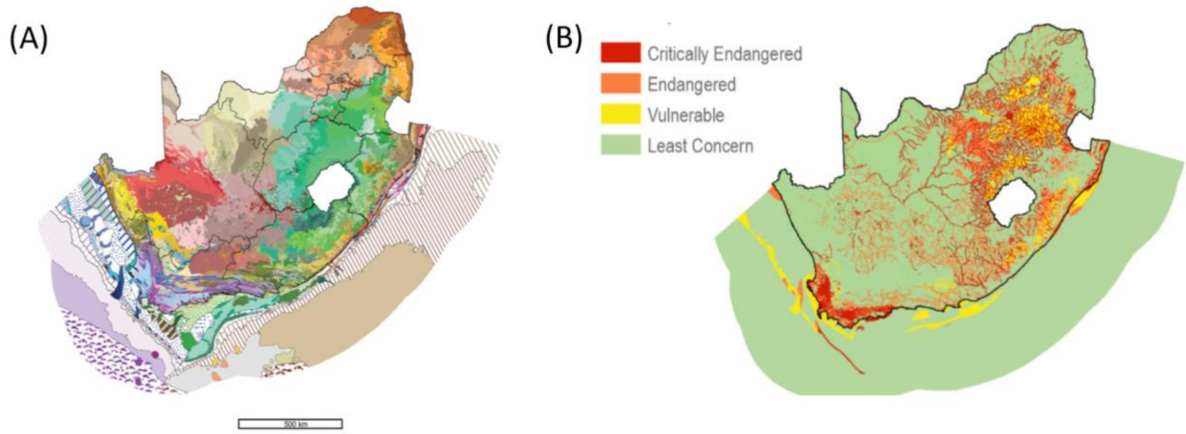

Figure 2. The South African National Ecosystem Classification System (SA-NECS) (a) provides the foundation for RLE assessments (showing risk status, B) and ecosystem accounts: *SA-NECS includes maps and hierarchical classifications of ecosystem types for all realms (seamless map of terrestrial, marine, and estuarine ecosystem types (an additional map of freshwater ecosystem type is not shown here); the RLE map (B) shows overall risk categories for terrestrial, freshwater, estuarine and marine ecosystems in South Africa; maps are adapted from* [24,98].

Table 1. Data on ecosystem extent in South Africa were shared between RLE and the ecosystem account, shown here in the extent account for terrestrial ecosystems, summarised by biome (following South Africa's own biome categories). *The Ecosystem Extent Index is calculated as the remaining natural or semi-natural extent of each biome as a proportion of its historical extent. The historical extent in the ecosystem extent account corresponds with the pre-industrialisation extent used in criterion A3 in RLE assessments. Table adapted from Table 15 in* [100].

Biome	Historical extent (ha)	EEI historical	Remaining natural or semi-natural extent 1990 (ha)	EEI 1990	Remaning natural or semi-natural extent 2014 (ha)	EEI 2014
Albany Thicket	3,531,231	100%	3,301,140	93%	3,309,564	94%
Desert	626,207	100%	617,970	99%	617,852	99%
Forest	462,518	100%	391,845	85%	409,056	88%
Fynbos	8,165,366	100%	5,911,991	72%	5,957,140	73%
Grassland	33,090,325	100%	21,759,719	66%	22,023,982	67%
Indian Ocean Coastal Belt	1,171,284	100%	551,628	47%	562,959	48%
Nama-Karoo	24,936,548	100%	24,515,553	98%	24,584,425	99%
Savanna	39,418,522	100%	34,022,403	86%	34,297,155	87%
Succulent Karoo	7,812,579	100%	7,570,206	97%	7,574,997	97%

Box 2: Colombia (national)—moving from RLE assessments to extent and condition accounts

Systematic RLE assessments on national scale hold potential for supporting ecosystem accounting, in particular the ecosystem classifications and maps, as demonstrated in South Africa [98] (Box 1) and Norway [42]. Here we use Colombia, a country with outstanding environmental variability in relation to its geographical size, as a case study to illustrate how information from an existing national-level RLE assessment could be used in building national ecosystem accounts under the SEEA EA framework. In 2017, Colombia published its first national level ecosystem assessment using the RLE protocol [103], providing insight into ecosystem status across all terrestrial biomes within the country, and informing protected area expansion and restoration planning [15,103]. Among the 81 recognised ecosystem types, 22 (27%) were assessed as 'Critically Endangered' and 14 (17%) were 'Endangered' based on four criteria A-D over three major periods: the recent past (1970-2014), the future (2014-2040), and the historical past (1750-2014) (Figure 3). The Colombian RLE addresses degradation by analyzing change in two important ecological processes that are closely linked to ecosystem services (pollination and seed dispersal), which could be used within the SEEA EA framework for condition accounting. Colombia was a focus of the WAVES natural capital accounting project [22], producing thematic accounts of land, forest, and water. Although this initiative provided an overview of the extent and stock of major environmental assets, it did not address ecosystem condition nor provide a comprehensive study across all ecosystem types within the country.

We used the existing RLE assessment for Colombia as a starting point to create corresponding ecosystem extent and condition accounts for all terrestrial ecosystem types. Information on ecosystem area, percentage change, and the Ecosystem Area Index [13] can be used in the ecosystem extent account (SI 2.3). For the ecosystem condition account, we combined information on the relative severity and the spatial extent of degradation of seed dispersal and pollination, using the Ecosystem Health Index [13] as a summary condition index (Table 2; for more detail see SI 2.3). The multiple assessment timeframes in RLE provided values for condition and extent accounts at the historical states (opening value at 1750), state at 2014, and several intermediate states in between (i.e. 1970, 1990), as well as projected future condition under a changing climate (for more detail see SI 2.3). Table 2 provides a summary showing the historical and 2014 values.

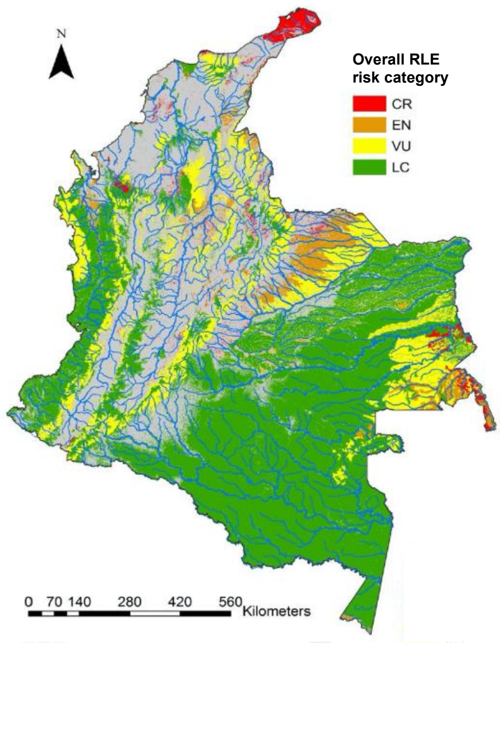

Figure 3. *RLE risk categories of Colombian terrestrial ecosystems, adapted from [103], where CR is critically Endangered, EN is Endangered, VU is vulnerable and LC is Least Concern.*

Table 2. *Ecosystem condition account for terrestrial ecosystems in Colombia by Ecosystem Functional Groups (EFG). We use Ecosystem Health Index (EHI) from [93], which combines the spatial extent and relative severity of degradation, using the information on seed dispersal and pollination processes from RLE assessment criterion D.*

Ecosystem Functional Group	Historial condition 1750 (opening)	Conditon in 2014 (closing)	Change between 1750-2014
MFT 1.1 Coastal river deltas	1	0.84	-0.16
MFT1.2 Intertidal forests and shrublands	1	0.79	-0.21
T1.1 Tropical-subtropical lowland rainforests	1	0.57	-0.43
T1.2 Tropical-subtropical dry forests and thickets	1	0.28	-0.72
T1.3 Tropical-subtropical montane rainforests	1	0.47	-0.53
T1.4 Tropical heath forests	1	1.00	0.00
T3.1 Seasonally dry tropical shrublands	1	0.98	-0.02
T6.5 Tropical alpine grasslands and shrublands	1	0.92	-0.08
TF1.1 Tropical flooded forests and peat forests	1	0.72	-0.28
TF1.4 Seasonal floodplain marshes	1	0.12	-0.88
TF1.5 Episodic arid floodplains	1	0.15	-0.85
All ecosystems	1	0.51	-0.49

Box 3: Meso-American reef (regional)—bringing RLE assessment models into ecosystem accounting

Strategic RLE assessments that focus on particular ecosystems provide detailed and important knowledge, which could be used in compiling accounts, which we illustrate here using a regional and relatively data-rich case study—the Meso-American Reef (MAR) [35]. The MAR contains the second longest barrier reef in the world, extending over 1000km from the northern tip of Mexico's Yucatan Peninsula to the Bay Islands in Honduras (Figure 4). It is of particular conservation interest due to multiple threats over the last 50 years, such as overfishing, pollution, mass bleaching, ocean acidification, and hurricanes (Figure 4). The MAR was assessed as 'Critically Endangered' based on five criteria over three time periods (1966-2015, 2016-2065, and pre-industrial-2015), while a dynamic simulation model applied under criterion E also provided 50 year forecasts under different future scenarios [35].

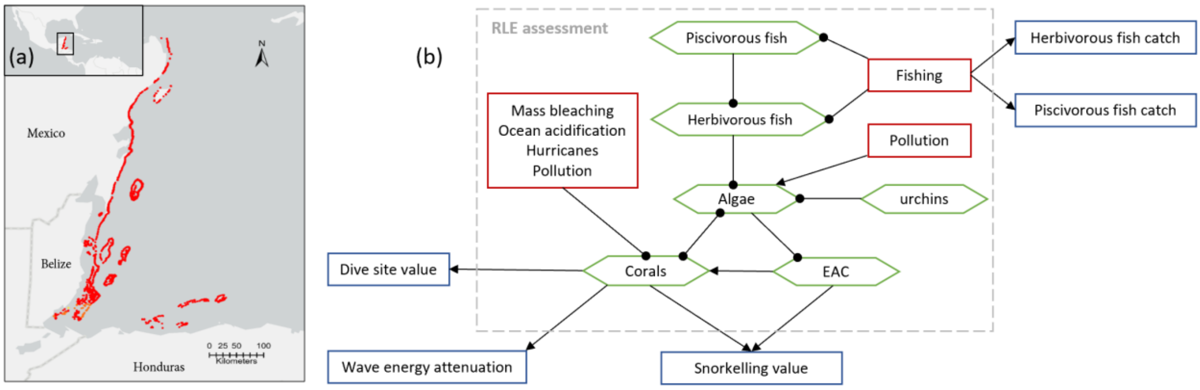

Figure 4. (A) The location and extent of the Meso-American Reef (from [35]) and (B) the conceptual model of key ecosystem features, ecological processes, threats, and dependent ecosystem services that underpinned the RLE assessment and SEEA EA accounts in the Meso-American Reef. *A simplified version of the conceptual model that supported RLE assessment [35] is shown within the grey-dashed box; this model was extended for SEEA EA accounting purposes, by including ecosystem services such as recreational values (snorkling and diving), coastal protection (wave attenuation), and fisheries, shown outside the grey-dashed box. Biotic feaures are shown in green, threats in red, and ecosystem services in blue. Solid arrows indicate positive effects whereas rounded arrows indicate negative effect . EAC: epilithic algal communities.*

We used information from RLE assessment to construct projections of ecosystem extent and condition (Table S5, S6). Data on change in the extent under criterion A was used directly in extent accounts. Projected spatial distribution declines for coral cover in the next 50 years (2016-2065) under different scenarios enabled projected accounting entries to be included for 2065 (Table S5). To compile the future condition measures in an account format, we chose three variables: coral cover, herbivorous fish biomass, and piscivorous fish biomass from criterion D, using empirical data from 1970 to 2013, together with the ecosystem model projections from 2016-2065 from Bland, et al. [104] (Table S6). Comparable projections could be compiled from other RLE assessments of coral reef ecosystems in Colombia [105] and the Western Indian Ocean [106].

For ecosystem services, we projected four ecosystem services (2016-2065) under different future scenarios using the modelled outcomes from the RLE assessment (See detailed methods in S2.4 and [107]). The projected ecosystem service variables were estimated using the dynamic simulation model used to assess criterion E [35] for three major ecosystem services categories: provisioning (herbivorous fish catch, and piscivorous fish catch, both emergant variables from the ecosystem model, which included fishing), recreational (dive site value, a production function of live coral cover and fish diversity and density derived from the model), and regulating (wave energy attenuation, projected using biophysical process models and predominantly dependent on the condition of the live reef) (Figure 5, S2.4, Table S7).

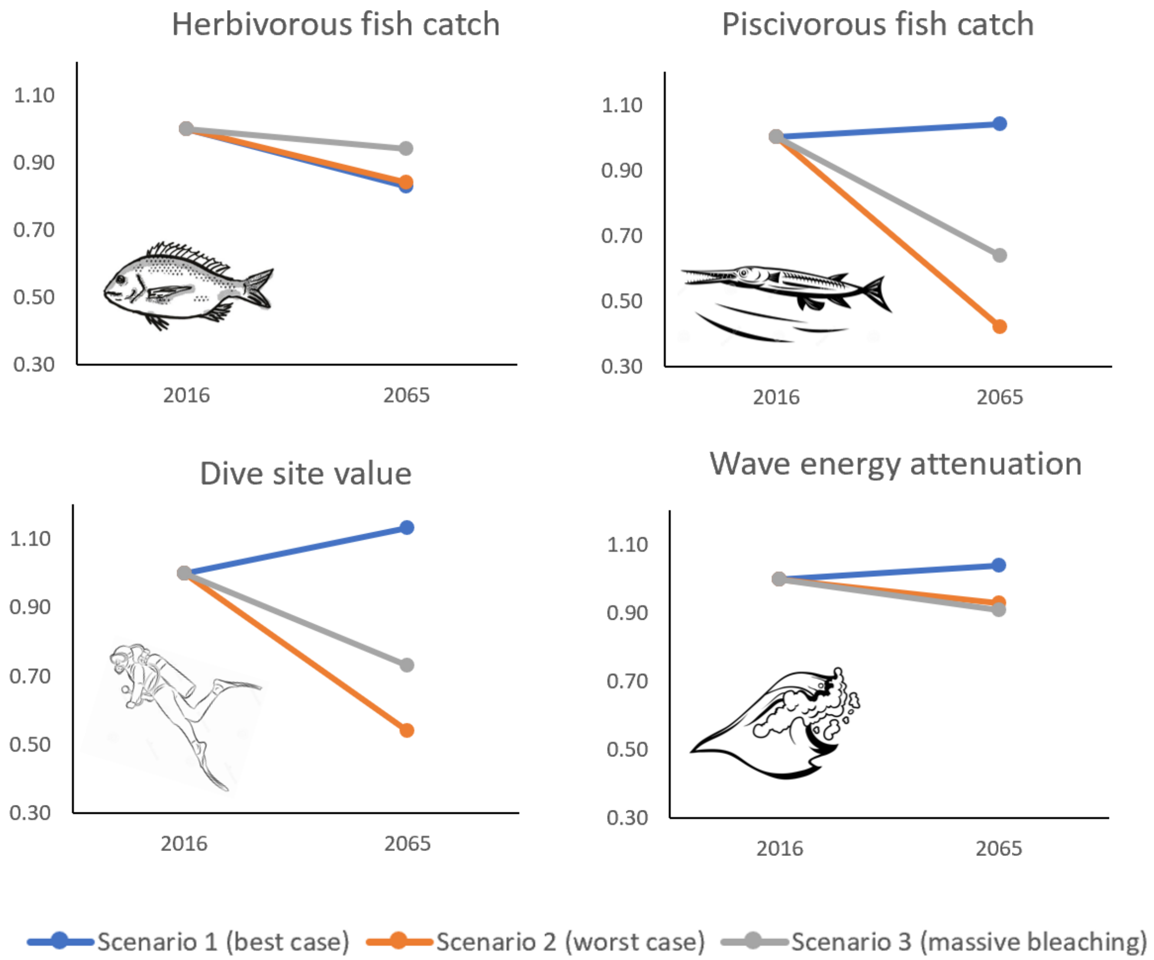

Figure 5. Ecosystem services indicators under future scenarios. *When combining with ecological production function, we could calculate ecosystem services indicators under future scenarios to evaluate the predicted change in services. Indicator values are normalized and set to 1 at the closing state at 2016 from the accounts. Future projections at 2065 are presented under three scenarios: the best case (S1) where all threats are low; the worst case (S2) where all threats are high, and massive bleaching (S3) where mass bleaching is high and other threats are low. Scenarios 2 and 3 are more likely than scenario 1, with current trajectories of climate change and bleaching. The corresponding projected ES account table is presented in Table S7.*

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