Science of the Total Environment 808 (2022) 152125

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Towards a comprehensive sustainability methodology to assess anthropogenic impacts on ecosystems: Review of the integration of Life Cycle Assessment, Environmental Risk Assessment and Ecosystem Services Assessment

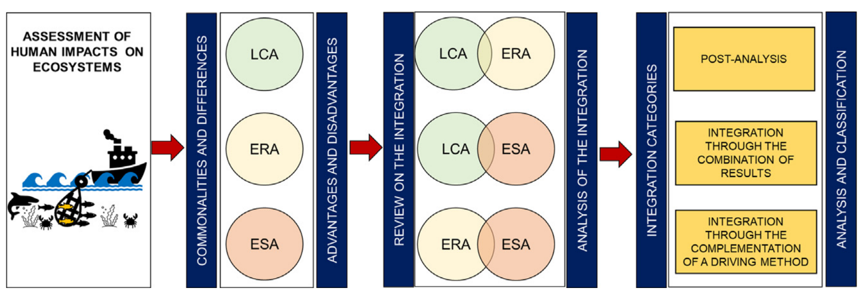
Laura Vittoria De Luca Peña [a,*], Sue Ellen Taelman [a], Nils Préat [a], Lieselot Boone [a], Katrien Van der Biest [b], Marco Custódio [c], Simon Hernandez Lucas [d,e], Gert Everaert [c], Jo Dewulf [a]

[a] Department of Green Chemistry and Technology, Faculty of Bioscience Engineering, Ghent University, Coupure Links 653, 9000 Gent, Belgium
[b] Ecosystem Management Research Group, University of Antwerp, Universiteitsplein 1, 2610 Wilrijk, Belgium
[c] Flanders Marine Institute, Wandelaarkaai 7, B8400 Ostend, Belgium
[d] Ghent University, Laboratory of Environmental Toxicology and Aquatic Ecology, Faculty of Bioscience Engineering, 9000, Ghent, Belgium
[e] Ghent University, BLUEGent Business Development Center in Aquaculture and Blue Life Sciences, 9000 Ghent, Belgium

HIGHLIGHTS

- A review on the integration of LCA, ERA and ESA is presented.
- The integration approaches of the selected papers are analysed.
- A classification scheme is developed based on the types of integration.
- There appears no full integration of the methodologies.
- A harmonized framework for integration needs to be developed.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 17 June 2021
Received in revised form 22 November 2021
Accepted 28 November 2021
Available online 3 December 2021

Editor: Paulo Pereira

Keywords:
Integration
Sustainability
Life cycle assessment
Ecosystem services

ABSTRACT

Nowadays, a variety of methodologies are available to assess local, regional and global impacts of human activities on ecosystems, which include Life Cycle Assessment (LCA), Environmental Risk Assessment (ERA) and Ecosystem Services Assessment (ESA). However, none can individually assess both the positive and negative impacts of human activities at different geographical scales in a comprehensive manner. In order to overcome the shortcomings of each methodology and develop more holistic assessments, the integration of these methodologies is essential. Several studies have attempted to integrate these methodologies either conceptually or through applied case studies. To understand why, how and to what extent these methodologies have been integrated, a total of 110 relevant publications were reviewed. The analysis of the case studies showed that the integration can occur at different positions along the cause-effect chain and from this, a classification scheme was proposed to characterize the different integration approaches. Three categories of integration are distinguished: post-analysis, integration through the combination of results, and integration through the complementation of a driving method. The literature review highlights that the most recurrent type of integration is the latter. While the integration through the complementation of a driving method is more realistic and accurate compared to the other two categories, its development is more complex and a higher

Abbreviations: AoP, Area of Protection; CF, characterization factor; ERA, Environmental Risk Assessment; ES, ecosystem services; ESA, Ecosystem Services Assessment; HRA, habitat risk assessment; InVEST, Integrated Valuation of Ecosystem Services and Tradeoffs; LCA, Life Cycle Assessment; LCI, life cycle inventory; LCIA, life cycle impact assessment; MCDA, Multi-Criteria Decision Analysis; MEA, Millennium Ecosystem Assessment; RA, Risk Assessment; SM-A, Supplementary Material – Annex A; SM-B, Supplementary Material – Annex B; SPGs, specific protection goals; SPUs, service providing units; UN, United Nations; UNEP-SETAC, UN Environment Programme and Society of Environmental Toxicology and Chemistry.

* Corresponding author.
E-mail address: laura.delucapena@ugent.be (L.V. De Luca Peña).

http://dx.doi.org/10.1016/j.scitotenv.2021.152125

Risk assessment
Multifunctional use

data requirement could be needed. In addition to this, there is always the risk of double-counting for all the approaches. None of the integration approaches can be categorized as a full integration, but this is not necessarily needed to have a comprehensive assessment. The most essential aspect is to select the appropriate components from each methodology that can cover both the environmental and socioeconomic costs and benefits of human activities on the ecosystems.

Contents

1. Introduction

Today, human activities have been increasing and diversifying across the world, which has led to a multifunctional use of our landscapes and seascapes. This multifunctionality comes hand in hand with the challenge to guarantee a sustainable use and management of our natural resources for economic growth while preserving healthy terrestrial and aquatic ecosystems (Lee et al., 2020; Wu, 2013). The United Nations (UN) estimates that 20% of the Earth's total land area has been degraded just between 2000 and 2015 (United Nations, 2020). Meanwhile, about 59% of our oceans are experiencing cumulative impacts from a variety of stressors such as climate change, overexploitation of resources, pollution and shipping (Halpern et al., 2019). The degradation of both terrestrial and aquatic ecosystems threatens the well-being of 3.2 billion people (United Nations, 2020).

Different decision-support tools have been developed to assess the magnitude and importance of the impacts caused by human activities on ecosystems, such as Life Cycle Assessment (LCA), Risk Assessment (RA) and Ecosystem Services Assessment (ESA). LCA is a process-oriented methodology that assesses the potential environmental impacts that a good or service has on natural resources, ecosystem health and human health over its entire or partial life cycle (i.e. raw material extraction, manufacturing and processing, transportation, usage and end of life). The first studies of LCA date back to the late 1960s and 1970s (Guinée et al., 2011) but it was not only until the decade 1990–2000 that the methodology was framed by international standards, leading to a higher acceptance by the scientific community (Guinée et al., 2011). On the other hand, RA is a broad procedure studying and managing the risks of specific activities (e.g. fossil fuel extraction, financial investment) or events (e.g. natural disaster). The ideas and principles of RA, as known today, have been developed in the 1970s and 1980s (Aven, 2016). Environmental Risk Assessment (ERA) is a more specific process within RA that aims to identify, analyse and evaluate potential environmental risks resulting from human activities that can cause harm to humans (Human Health ERA) and/or ecological receptors (Ecological ERA).

ESA is an ecosystem-oriented methodology that assesses the contributions of ecosystems to human well-being through the delivery of ecosystem services (ES). Moreover, the methodology highlights the trade-offs and synergies that can occur between ES when a human activity takes place within an ecosystem. The concept of ES has its origins in the 1970s (Gómez-Baggethun et al., 2010). Later, in the 1990s, ES have been defined as *"flows of materials, energy, and information from natural capital stocks which combine with manufactured and human capital services to produce human welfare"* (Costanza et al., 1997). The mainstreaming of ES into literature began in the 1990s (Costanza et al., 1997; Costanza and Daly, 1992; de Groot, 1992; Daily, 1997) and gained recognition in the policy-making context with the publication of the UN's Millennium Ecosystem Assessment (MEA) in 2005 (MEA, 2005). The MEA encouraged the development of some of the first frameworks where ES are assessed to support decision-making in the context of land and water use by humans (Cowling et al., 2008; Daily et al., 2009; de Groot et al., 2010a). Despite the vast research and knowledge about ES, compared to LCA and ERA, ESA is a rather young field that requires further standardization and guidance for planning and decision-making (Hauck et al., 2013; Häyhä and Franzese, 2014; Rosenthal et al., 2015).

The LCA, ERA and ESA methodologies have their own individual advantages but also present shortcomings when it comes to assessing environmental impacts on ecosystems. So far, a methodology that comprehensively assesses both positive and negative impacts from human activities at different geographical scales (from local to global) is still lacking. More specifically, such a methodology would be beneficial to address the impacts resulting from the multifunctional use of terrestrial and aquatic ecosystems. However, efforts have been made to integrate the above-mentioned concepts (i.e. LCA, ERA, ESA), mainly with the purpose to broaden the scope of a study (e.g. Liu et al., 2012; Liu and Bakshi, 2019), to assess impacts which are usually not taken into account in one of the methodologies (e.g. Blanco et al., 2018; Crenna et al., 2020; Galic et al., 2019), to combine and/or interpret the results using different perspectives (e.g. Barberio et al., 2014; Briones-Hidrovo et al., 2020; Culhane et al., 2019) and to improve or optimize a methodology by adding and/or complementing it with features of another one (e.g. Csiszar et al., 2016; Deacon et al., 2015; Othoniel et al., 2019).

This paper aims to provide insight into the state-of-the-art of LCA, ERA and ES integration (i.e. the way it has been conceptualized and/or operationalized through case studies) in order to contribute to the development of a holistic methodology assessing the sustainability of human activities. We aim to understand how and to what extent these methodologies have been integrated and what are the existing gaps limiting the integration. The identification of these aspects are necessary to conduct further research on comprehensive sustainability assessments. Based on the literature review carried out, we aim to develop a classification system that distinguishes between types of integration in order to provide a clear overview of integration possibilities. Several papers on integration of LCA and ERA have identified and described different classification categories (Flemström et al., 2004; Harder et al., 2015; Kobayashi et al., 2015b; Muazu et al., 2021).), however, this is entirely lacking for LCA-ES and ES-ERA publications. The aim of this paper is not to prioritize a certain integration approach, but rather to be transparent about the pros and cons of each way of integration and to describe the different levels of integration of steps/components of LCA, ESA and ERA methodologies holistically. The proposed classification system could be used in further studies to help identify integration approaches and develop new frameworks for comprehensive sustainability assessment studies.

2. Overview of the general methodological frameworks

2.1. The methodological approaches: commonalities and differences

Fig. 1 illustrates the frameworks of LCA, ERA and ESA to conduct an impact assessment. All of the frameworks start by defining the goal and scope of the assessment and then followed by the data collection step (i.e., data inventory in LCA, problem formulation in ERA and data collection and scenario development in ESA)(Fig. 1). Subsequently, the impacts are evaluated using quantitative and qualitative methods, the results interpreted and then communicated to stakeholders, decision-makers and/or project managers. Conducting an LCA, ERA or ESA is an iterative process, which means that phases in the framework can be revised (Fig. 1). A more exhaustive description of other features of LCA, ERA and ESA (i.e. description of their frameworks and the quantitative/qualitative methods used) are found in the Supplementary Material - Annex B (SM-B).

2.1.1. Life cycle assessment

The modelling of the environmental impacts in LCA relies on cause-effect pathways that link specific environmental stressors caused by human activities (such as the generation of emissions in the air, water or soil, waste generation, the extraction of a natural resources and/or the occupation of a landscape or seascape) with one or multiple potential effects on the environment (Fig. 2a). These effects are classified into defined impact categories at the midpoint and/or endpoint (damage) level along the cause-effect chain (Fig. 2a). Several quantitative life cycle impact assessment methods (LCIA) methods exist to quantify a broad number of impact categories. For example, the ReCiPe 2016 method (Huijbregts et al., 2017) has 17 midpoint categories in total (e.g. global warming, land use, freshwater ecotoxicity). The LCIA methods use characterization factors (CFs) as unit convertors, i.e. to transform inventory flows into the common unit of the impact category indicator (ISO, 2006). These impact categories are related to overarching endpoints, termed Areas of Protection (AoPs); these are the entities that need to be safeguarded. The most used AoPs in LCA are human health, ecosystem health and natural resources. However, other AoPs have been identified such as the man-made environment (de Haes et al., 1999) or human prosperity/welfare and human well-being (Dewulf et al., 2015; Taelman et al., 2020). On top, in some cases, it is

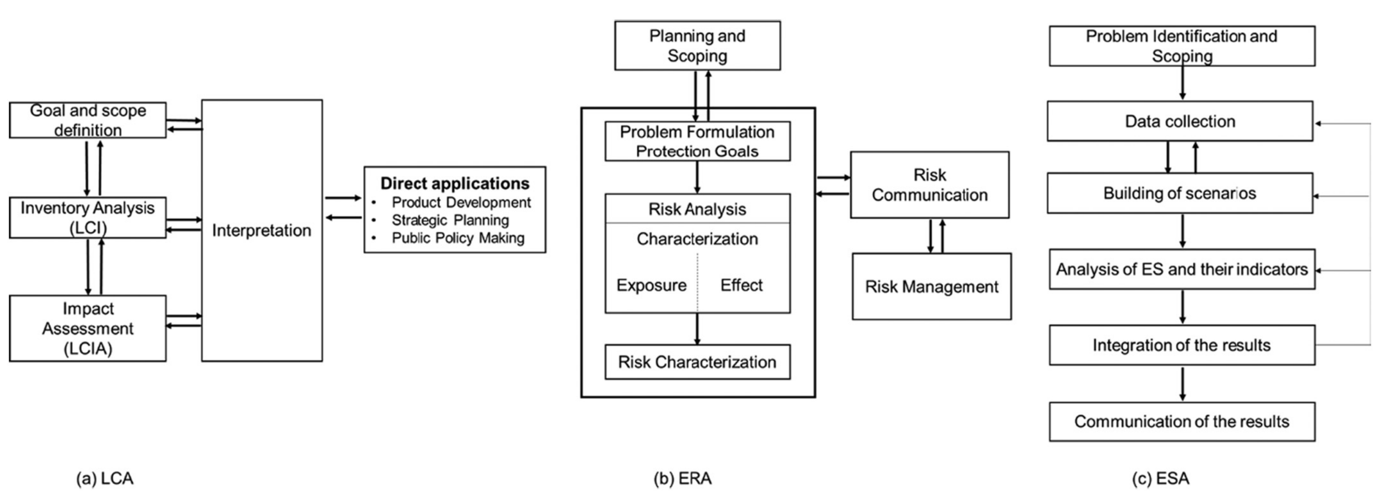


Fig. 1. Frameworks for LCA, RA and ES. (a) LCA framework based on ISO (2006), (b) ERA framework based on EPA (1998), (c) ESA framework based on Rosenthal et al. (2015) and Burkhard et al. (2018). .

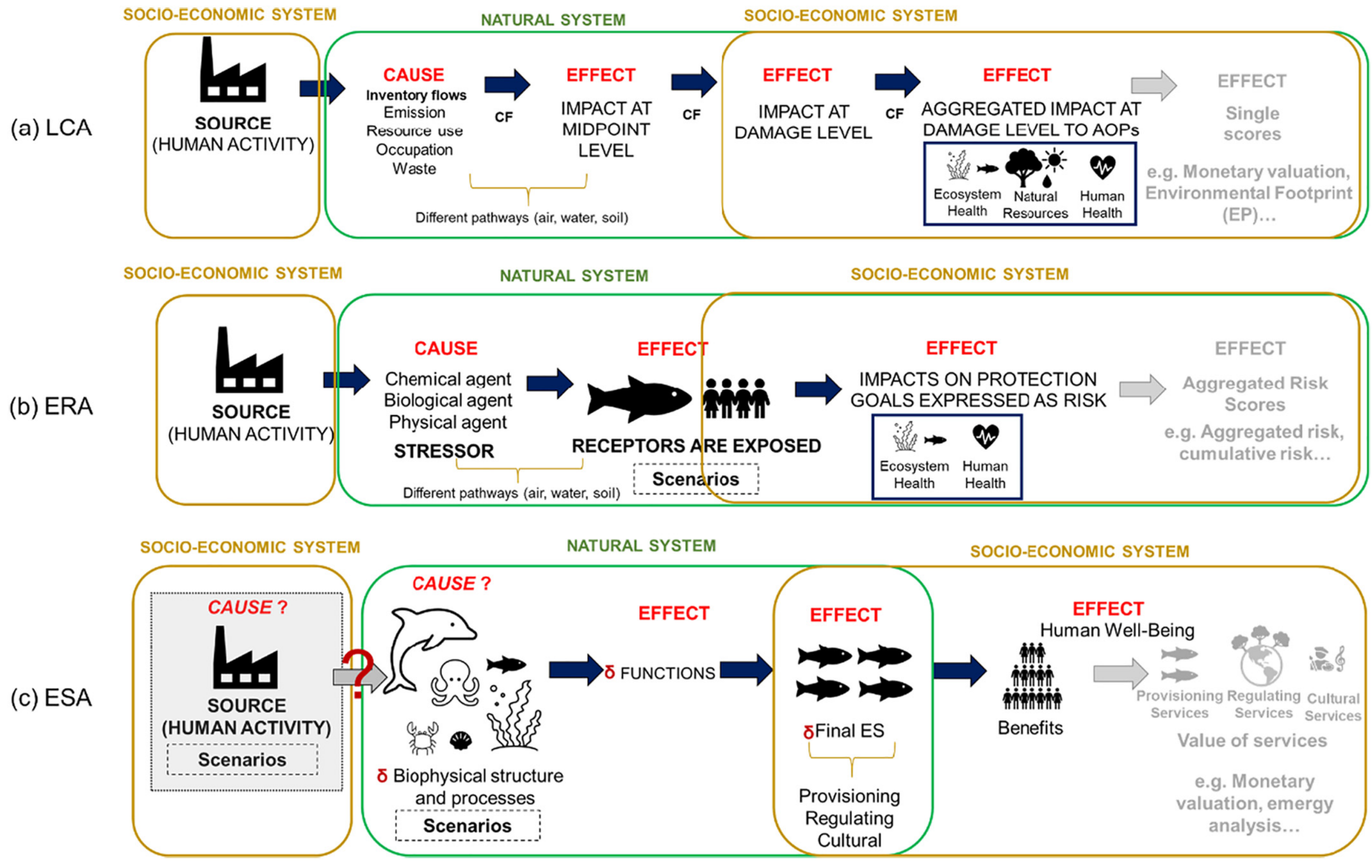


Fig. 2. Representation of the cause-effect chain modelling in (a) LCA, (b) ERA and (c) ESA. The yellow, green and blue boxes encompass the components of a socio-economic system, the components of a natural system and the entities that need to be protected respectively. The grey shadings at the end of the cause-effect chains represent the elements that are not always assessed in LCA, ERA and ESA. The grey box at the start of the cause-effect chain of ESA depicts its heterogeneity. The *causes* in ESA do not always start with human activity's effect on biophysical structures and processes, but it can also start from a change within a biophysical structure and process non-related with a human intervention.

possible to aggregate the impacts into a single score to evaluate the overall environmental costs and benefits (e.g. in terms of monetary values) (Manfredi et al., 2012).

2.1.2. Environmental risk assessment

Environmental Risk Assessment (ERA) is a spatially-explicit methodology that examines the risks posed by human activities or interventions on human health and ecological receptors such as animals, plants or an entire ecosystem. While Human Health ERA estimates the likelihood of adverse health effects when humans are exposed to harmful agents present in the air, soil or water (EPA, 2016a), Ecological ERA focuses on the probability that adverse ecological effects may occur when ecological receptors are exposed to one or more chemical (e.g. toxic metals, pesticides), biological (e.g. invasive species) or physical (e.g. climate change, land-use change) stressors (Fairman et al., 1998; EPA, 2016b).

In Fig. 2b, the cause-effect pathway in ERA is illustrated. It starts by identifying a specific physical, biological and/or chemical stressor caused by one or more human activities. This stressor can reach the receptor (human or ecological entity) via different pathways such as air, water and/or soil. After this, the receptor will be exposed to the stressor and the consequence of this exposure will depend on its duration, frequency and magnitude. The impact(s) of the stressor(s) are expressed in terms of the risk(s) to the receptors and their attributes. In ERA, the concept of Protection Goals is used instead of AoP, as considered in LCA. These are usually broad and determined by a specific regulatory system (e.g. at country level) (Garcia-Alonso and Raybould, 2014; Romeis et al., 2011). On the other hand, the Specific Protection Goals (SPGs) are the 'endpoints' in an ERA (EFSA Scientific Committee, 2016). However, there is a noticeable difference in the concept of 'endpoint' in LCA and ERA. In the former, an endpoint is a group of indicators that express the impact of a product/service at damage level (end of the cause-effect chain), while in ERA, as mentioned previously, the endpoint is the receptor(s) to be protected (i.e. ecological entities and their attributes). The impacts could also be aggregated into a single score to evaluate the aggregated or cumulative risk.

2.1.3. Ecosystem services assessment

ESA is a methodology designed to give a better understanding of the ecosystems and their functions, how we value them and how they contribute to human well-being. Various frameworks have been developed to understand the links and interactions between the ecosystems and human well-being (Costanza et al., 2017; de Groot et al., 2010b; Haines-Young and Potschin, 2010; Potschin-Young et al., 2018). The ES cascade framework proposed by Haines-Young and Potschin (2010) has been widely adopted as a conceptual model that guides the way on how to assess ES (Potschin-Young et al., 2018). However, the cause-effect pathway in an ESA is heterogeneous compared with LCA and ES, when it comes to defining the cause of the impact. On the one hand, there is a human activity or intervention that will directly affect biophysical structures and processes (Fig. 2c). An example of this are studies that assess the effect of land-use changes on ES under different scenarios (e.g. Förster et al., 2015). On the other hand, the pathway can start only with changes (δ) on a biophysical structure (Fig. 2c). For example, a study assessed how the recovery of a sea otter population could trigger changes in a kelp ecosystem and the production of ES (Gregr et al., 2020).

Fig. 2c illustrates how the changes of these structures or processes will lead to changes in the ecosystem functions and hence, also on the supply of ES. The latter has a direct impact on human well-being and the ES is valued at the end of the cause-effect chain. In ESA, the main endpoints of the assessment are the ES values, which will determine the level of protection needed for one or more ES. These ES values can be expressed and measured in monetary terms (Christie et al., 2012). Nonetheless, this value not always can be readily expressed through monetary valuation due to the complexity of ES and their interactions with humans. In such cases, the use of non-monetary valuation (e.g. ecological valuation by emergy) is required (Christie et al., 2012; Yang et al., 2018). It is important to mention that the change in ES values are usually obtained by comparing the ES values between different development scenarios in a study area. For example, comparing a past *"pristine"* environment scenario or a present reference scenario with future alternative scenarios using different management strategies related to one or more human activities (Carpenter et al., 2006; Rosenthal et al., 2015).

2.1.4. Towards a harmonized representation of the phases of an impact assessment for ecosystems

We analysed the frameworks and cause-effects chains of each methodology and from this developed a harmonized representation of the phases and cause-effect chains required to assess the impacts on ecosystems due to human activities (Fig. 3). This representation considers the commonalities and differences of the cause-effect chain between LCA, ERA and ESA.

2.1.4.1. Scoping-System Analysis phases. The first phase consists of defining the goals and scope of the assessment. In this step, three key aspects are clarified: (i) what is (are) the problem (s) with human activities and the purpose of the study (e.g. human interventions, pressures to the natural environment), (ii) which components of the socio-economic and natural system are relevant and how will the impact on these be assessed and (iii) which are the system boundaries of the system in terms of the spatial and temporal scale.

The second phase focuses on analysing and describing the system and its components (this depends on the boundaries previously established). Also, the direct and indirect interactions/relationships of components between and within the socio-economic (e.g. emission from a human activity) and the natural system (e.g. a wetland) in past, current or future scenarios are identified and described. This phase also involves the collection of primary and secondary data and the analysis of the data quality.

2.1.4.2. Impact Analysis phase. The third phase analyses the changes (δ) in the natural system (due to its interaction with the socio-economic system) and how this impacts the protection goals. For this purpose, the cause-effect chain needs to be described and modelled using qualitative and/or quantitative methods. Once the changes in the system are identified, an analysis of their total impact (Δ) on the protection goals (which can be socio-economically or naturally oriented) (previously defined) is done. Specific examples of such goals are the protection of natural resources, ecosystem health, human health and well-being, prosperity and welfare. These total impacts (Δ) can be further aggregated into single scores in order to determine the environmental costs and benefits of human activities.

2.1.4.3. Interpretation and Communication phases. The aim of the fourth phase is to interpret and synthesize the results in a way that they can be easily communicated and understood by the decision-makers and stakeholders. The main questions to be answered in this step are: (i) which are the most relevant impacts and what components are mostly affected, (ii) which are the trade-offs and synergies, (iii) how sensitive is the study to changes in variables and/or parameters and what are the uncertainties, (iv) how complete and transparent is the study performed and (v) what are the main conclusions and limitations of the assessment. While answering these questions, previous phases might need to be revisited and modified as new information and additional questions emerge.

Finally, the fifth phase is where results are communicated to the parties involved in the assessment (e.g. project managers, stakeholders, companies' representatives, government authorities). These parties will participate in a discussion to evaluate the results and to provide feedback on how the assessment was conducted in order to make changes in the previous phases (only if needed). Based on the results of the assessment, they also take actions and implement their decisions.

2.2. The strengths and weaknesses of the methodological approaches

In this section we highlight and describe some of the most pronounced strengths and weaknesses of LCA, ERA and ESA that may help answer the question of why there should be an integration. Table 1 summarizes this section and a detailed description of the main advantages and limitations for each methodology can be found in SM-B.

LCA is a quantitative methodology characterized by having a high resolution when looking at the human activity (i.e. products, processes and technologies) (Bjørn et al., 2017; Taelman et al., 2018) (Table 1). Nonetheless, LCA can assess multiple effects of several stressors at a global scale and throughout an entire life cycle of a product or process (Muazu et al., 2021; Taelman et al., 2018; Tsang et al., 2017) (Fig. 2a). Though LCA considers

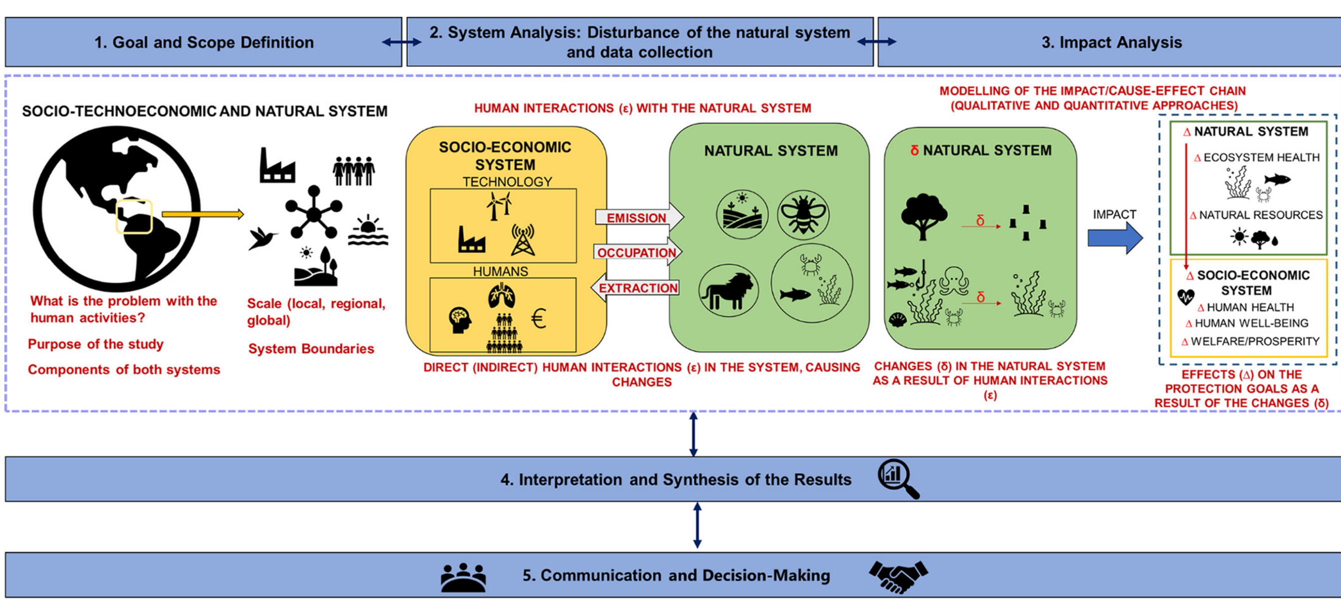


Fig. 3. Harmonized representation of the phases and cause-effect chain that are required to assess the impacts on ecosystems due to human activities.

Table 1
Characteristics of the methodologies: Life Cycle Assessment (LCA), Environmental Risk Assessment (ERA) and Ecosystem Services Assessment (ESA) + (Strength) +/− (Needs further development) − (Weakness); NP (Not Present).

Characteristic	LCA	ERA	ESA
Spatial scale			
Human activity	Site-specific	Site-specific	Site-specific/Sometimes the human activity is not defined
Cause-effect chain (Impacts)			
Global	+	−	−
Regional	+/−	+/−	+
Local	+/−	+	+
Temporal Scale			
Time-frames (baseline vs. future)	+/−	+	+
Static models	+	+	+
Dynamic models	−	+/−	+/−
Cause-effect pathway			
Type	Process-oriented	Receptor-oriented	Ecosystem-oriented
Well-defined human activity	+	+	+/−
Multiple stressors	+	+/−	+/−
Multiple cause-effect impact pathways	+	−	−
Linear causality	+	+	+/−
Positive effects of human activities	+/−	+/−	+/−
Type of approaches			
Qualitative	NP	+	+
Semi-quantitative	NP	+	+
Quantitative	+	+/−	+/− (not all ES can be quantified)
Aggregation			
Midpoints	+	NP	NP
Midpoints → Endpoints	+	+/−	+/−
Endpoints → Single scores	+/−	+/−	+/−

multiple impact pathways, it can only quantify potential impacts and not actual ones because CFs are limited in spatial resolution (Hu, 2018; Lueddeckens et al., 2020; Muazu et al., 2021; Tsang et al., 2017).

Meanwhile, ERA and ESA can look at effects at a finer resolution. By ERA being a receptor-based methodology and focusing on local impacts, it makes the methodology have limitations when assessing global impacts but also, assessing cumulative impacts due from multiple stressors (given that, in some cases, it cannot link the stressor with the human or ecological receptor and that it rather relies on semi-quantitative methods) (Goussen et al., 2016; Muazu et al., 2021; Simmons et al., 2017;Taelman et al., 2018).

ESA relies on both qualitative and quantitative methods to capture the changes within an ecosystem across space and time, as well as the supply and demand of ES; the latter depending on socio-economic systems (e.g. from local communities to nations) (Grêt-Regamey et al., 2017; Harrison et al., 2018; Hein et al., 2006). By being a benefit-oriented methodology (Baker et al., 2013; Van der Biest et al., 2020), the cause-effect chain of ESA links the transformation of the components of an ecosystem to changes in ES supply, flows, demand, benefits and value (i.e. provisioning, regulating and cultural ES) (Potschin-Young et al., 2018). This is something that LCA and ERA cannot assess by themselves because they are mainly impact-oriented methodologies. However, given that ESA is an ecosystem-based approach and that the *cause* is not necessarily a human activity (Potschin-Young et al., 2018) (Fig. 2c and Table 1), this methodology alone cannot quantify the impacts of human activities on ES and their benefits to human well-being at different scales without relying on impact assessment methodologies such as LCA and ERA (Häyhä and Franzese, 2014). Besides this, there are still existing quantification gaps on certain ES (Boerema et al., 2017).

Despite aggregation being useful in LCA and ESA to help simplify the communication of the results, to identify trade-offs among different alternatives and to inform policy makers, it has been considered a controversial step in both methodologies (Buckwell et al., 2018; Kalbar et al., 2017; Prado et al., 2020). LCA has been criticized because of its subjectivity in the normalization or its weighting methods and the risk of misinterpretation or overinterpretation of results (Kalbar et al., 2017; Prado et al., 2020; Reap et al., 2008), while aggregation through monetary valuation and commodification of ecosystems in ESA are controversial and heavily criticized for ethical reasons (Sullivan, 2009; Sullivan and Hannis, 2017). Also, the aggregation of ES into a single value is challenging due to the variation of ES across multiple scales (Meyerson et al., 2005; Small et al., 2017). In ERA, impacts caused by multiple stressors (via various pathways) to one or more receptors can be assessed by aggregated or cumulative exposure assessments in ERA (EPA, 2003) (Table 1). Though aggregate or cumulative risk assessments allow one to see the relative contribution of the stressors, exposure pathways and sources in the overall impact (Lentz et al., 2015), these type of assessments can be limited due to their complexity and by assuming that the effect of the stressors can be additive (Holsman et al., 2017; Stelzenmüller et al., 2018).

3. State-of-the-art of the level of integration of LCA, ESA and ERA

3.1. Literature research approach

A review was conducted to identify the various approaches in which LCA, ESA and ERA have been integrated in previous studies. Fig. 4 outlines the methodology followed for searching, selecting and reviewing the literature studies.

The Web of Science was the main search engine for selecting the literature and Google Scholar was used to look for additional scientific papers or grey literature that were not captured by Web of Science. Literature published between 2010 and 2021 was searched and selected. In total, the number of publications obtained from both search engines was 1204. A paper was considered relevant if its title explicitly mentioned two or three methodologies with words implying an integration (e.g. "integrated framework", "accounting", "combining", "coupling", "linking", "introducing", "comprehensive assessment"). For this reason, a first screening of the titles was conducted reducing the publications to 138. We did not find relevant papers that considered or attempted the integration of all three concepts simultaneously (LCA, ERA and ES). After this, a second screening was done to guarantee that the aim of the selected literature truly focused on the integration of LCA, ES and ERA. From those papers, 94 were retained and we searched in some of their reference lists for papers that did not appear using our literature search approach. In total, 25 new papers were obtained from the screening of the reference lists, from which 16 papers were further considered. In total 110 relevant scientific papers (42 of LCA-ES, 34 of LCA-ERA and 34 of ES-ERA) were selected to analyse their main characteristics and content.

For the analysis, six criteria were selected (Fig. 4), which represent aspects that were considered most important for understanding the goal and scope of the publications, how the integration was carried out and also the shortcomings of the approaches (Section 3.2). Finally, an in-depth analysis was performed, focused only on the types of integration. This analysis only considered the peer-reviewed papers with case studies. The publications that only considered the integration at a conceptual level were not included because there is no actual application of the integration methodology and therefore the practical integration potential could be questioned. In total 59 publications were considered for the in-depth analysis (Fig. 4).

A detailed description of the literature review procedure and the list of selected literature with its content description can be found in the Supplementary Material - Annex A (SM-A).

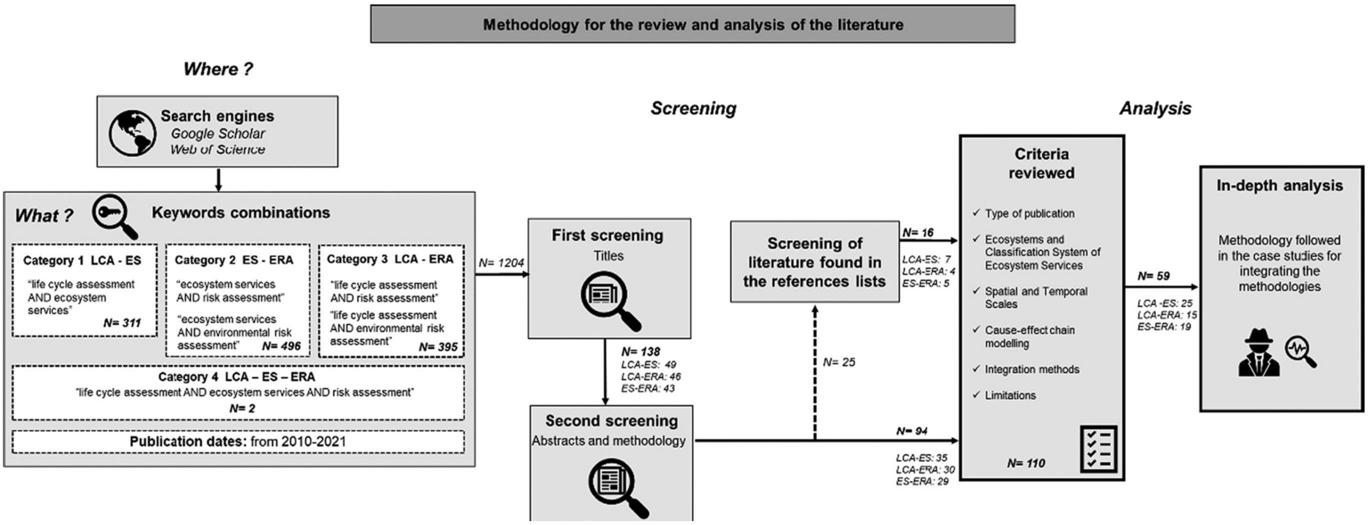


Fig. 4. Approach for conducting the search and analysis of the literature.

3.2. Characteristics of the papers

The following section summarizes the main findings from the analysis of the 110 papers reviewed; based on the criteria presented in Fig. 4. These findings were clustered in three groups: LCA-ES, ES-ERA and LCA-ERA. This section will mainly focus on describing the integration approaches and its limitations for each cluster, however a detailed description of the findings for the other criteria can be found in SM-A and SM-B. We describe in a generic way the methods conceptualized and/or used to integrate the methodologies. More details and specific examples of the integration methodologies used in the papers can be found in the SM-A and SM-B.

3.2.1. Group 1: LCA -ES

3.2.1.1. Integration methods. Several integration approaches were observed in the literature belonging to the LCA-ES group. Most of the papers focused on incorporating ES as an additional impact pathway to traditional pathways in LCA and develop spatially-differentiated midpoint and endpoint CFs following the UN Environment Programme and Society of Environmental Toxicology and Chemistry (UNEP-SETAC) guidelines (Brandão and Milà i Canals, 2013; Cao et al., 2015; Koellner et al., 2013; Müller-Wenk and Brandão, 2010; Muñoz et al., 2014; Saad et al., 2013). Other papers did not follow these guidelines (e.g. Arbault et al., 2014; Blanco et al., 2018; Bos et al., 2016; Jeswani et al., 2018; Liu et al., 2017; Núñez et al., 2013; Othoniel et al., 2019; Rugani et al., 2013; van Zelm et al., 2018; Zhang et al., 2010a, 2010b).

At a conceptual level, different papers suggested rethinking the AoPs in LCA to include ES as a distinct AoP (Callesen, 2016; Dewulf et al., 2015; Koellner et al., 2013), as well as human well-being (Schaubroeck and Rugani, 2017). Other approaches focused on modifying the LCI (life cycle inventory) by using allocation as a way to integrate ES into LCA (Boone et al., 2019; Bragaglio et al., 2020) or by using ES tools to replace or add new data to the LCI (Chaplin-Kramer et al., 2017). Some papers expanded the system boundaries of LCA to account for the supply and demand of ES (Bakshi et al., 2015; Liu et al., 2017, 2018a, 2018b, 2019, 2020; Liu and Bakshi, 2019) or to integrate the industrial environment and natural environment (Schaubroeck et al., 2013). Moreover, some approaches suggested the integration of the cascade framework into LCA both conceptually (Maia de Souza et al., 2018; Pavan and Ometto, 2018; Rugani et al., 2019) and with applied case studies (Liu et al., 2020; Othoniel et al., 2019).

Other approaches proposed (1) using the LCI as an input for a bio-economic model of ES (Bruel et al., 2016), (2) including ES in the ReCiPe 2016 based on an ES monetary valuation analysis (Alejandre et al., 2019), (3) using the LCA outputs to obtain the value of an ES (Xue et al., 2014), (4) analysing the ecological fund flows based on the results from the LCA (Morales et al., 2020) and, (5) using integrated earth system models to conduct comprehensive sustainability assessments (Schaubroeck, 2018). While most papers proposed or attempted to integrate parts of both LCA and ERA methodologies, two papers conducted LCA and ES assessments independently (Briones-Hidrovo et al., 2020; Viglia et al., 2013).

3.2.1.2. Limitations. One recurrent limitation in the LCA-ES group is the difficulty in harmonizing the different spatial and temporal resolutions of ES into LCA (Maia de Souza et al., 2018; Othoniel et al., 2016; Pavan and Ometto, 2018; Rugani et al., 2019). Also the accounting of multiple scales of the socio-economic system that benefits from these services remains a challenge (Liu et al., 2020; Othoniel et al., 2016). In addition to this, the interconnections and feedback between different impact pathways and ES cannot be fully captured in LCA (Blanco et al., 2018; Briones-Hidrovo et al., 2020; Pavan and Ometto, 2018). Moreover, Bakshi et al. (2015) and Zhang et al. (2010a) point out that there could be bias towards the inclusions of provisioning ES in LCA because the quantification of the supply and demand of regulating and cultural ES is less straightforward and more challenging. Zhang et al. (2010a) and Bakshi and Small (2011) also pinpointed the difficulty to find appropriate aggregation metrics for ES, which can lead to services being ignored because they cannot be aggregated under the same unit. The commonly used aggregation metric for ES is their value expressed in monetary terms, yet several papers did not monetize ES due to concerns about under- or over-estimations of value (Cao et al., 2015).

Another important drawback is that some of the methods proposed cannot be operationalized in available LCA software (Arbault et al., 2014; Liu et al., 2017, 2018a, 2018b; Núñez et al., 2013; Othoniel et al., 2019; Rugani et al., 2019). Besides these challenges, some authors identified the risk of overlap and double-counting when establishing ES as impact categories (Blanco et al., 2018; Briones-Hidrovo et al., 2020; Callesen, 2016; Rugani et al., 2019). Some papers encountered problems with the reliability of ES indicators (Jeswani et al., 2018), the availability of data (Alejandre et al., 2019; Liu et al., 2017, 2018a, 2018b; Zhang et al., 2010a) and the uncertainty of the methods (Bruel et al., 2016; Jeswani et al., 2018; Liu et al., 2020; Núñez et al., 2013; Othoniel et al., 2019; Rugani et al., 2013).

3.2.2. Group 2: ES-ERA

3.2.2.1. Integration methods. The approach followed by most of the reviewed papers for the ES-ERA group consisted of connecting conventional ERA endpoints with ES. Some papers conceptually describe the potential of this approach, including its advantages and disadvantages (EFSA Scientific Committee, 2016; Faber and van Wensem, 2012; Galic et al., 2012; Munns et al., 2016, 2017). One way to achieve this connection is through the development of EPFs, which link a biophysical structure or service-providing units (SPUs) with the supply of ES. Several papers suggest the use of statistical or mechanistic models to obtain these EPFs (Devos et al., 2019; Faber et al., 2019; Forbes et al., 2017; Forbes and Calow, 2013; Galic et al., 2012). Few papers applied these types of models to a case study (Awuah et al., 2020; Forbes et al., 2019; Galic et al., 2019). There are other approaches where ES are essential to define the scope (boundaries and scenarios) of the ERA (Apitz, 2012; Gilioli et al., 2017; Nienstedt et al., 2012; Sample et al., 2016; Syberg et al., 2017).

There is a group of papers that are characterized by conducting individual ES and habitat RA assessments, using the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) tool and following semi-quantitative approaches to obtain risk scores and scores of ES supply. A habitat risk assessment (HRA) model can be found in InVEST, which was frequently used in this group of papers (e.g. Arkema et al., 2015). Few papers combined the cumulative risk scores (obtained from the HRA) and ES supply scores to calculate a vulnerability index determining how susceptible an ecosystem is to damage. The changes in vulnerability are used as a proxy of a habitat's potential to deliver ES (Cabral et al., 2015; Caro et al., 2020; Willaert et al., 2019). Some papers did not directly use InVEST but, instead, they adapted their method based on this tool and other approaches in order to calculate the cumulative risk on ecosystems due to multiple stressors, which can be used to determine the risk to ES provision (e.g. Battista et al., 2017; Culhane et al., 2019). Another approach consisted in describing conceptually how to link ecological vulnerability with social systems vulnerability and established the changes in ES provision as threat(s) to the social systems (Berrouet et al., 2018).

Other methods obtained ES indicators (Xu et al., 2016; Zhao and Zhang, 2013), changes in ES supply and demand (Zhang et al., 2020), and ES monetary values (Kang et al., 2018; Pártl et al., 2017; Xing et al., 2020) to define risk scores. One paper used an opposite approach by obtaining risk scores to calculate the ES value using monetary and non-monetary techniques (Deacon et al., 2015). Finally, one paper suggested, at a conceptual level, a framework where decision-analytic strategies could be used for decision-making when combining the results from Ecological RA, Human health RA, and ES endpoints (von Stackelberg, 2013).

3.2.2.2. Limitations. The drawbacks from the ES-ERA group are similar to those of the LCA-ES. For example, the interconnections, dynamics, and feedback between ES were sometimes difficult to establish or understand, and some assessments were restricted to a few ES (Arkema et al., 2015; Cabral et al., 2015; Faber et al., 2019; Galic et al., 2012; Guerry et al., 2012; Zhang et al., 2020; Zhao and Zhang, 2013). It was also challenging to link multiple stressors to a single ES (Syberg et al., 2017), biodiversity to ES (Devos et al., 2019; EFSA Scientific Committee, 2016), and biophysical structures/processes and/or functions to ES supply (Galic et al., 2012; Munns et al., 2017). Besides this, the variations in the definition of ES and its components can be challenging when trying to conduct an ERA based on ES (Devos et al., 2019; Pártl et al., 2017). Also, there was a case where conflicts of interest emerged between stakeholders when determining the most relevant ES (Syberg et al., 2017). In addition to this, several papers identified the linkage of ERA measurement endpoints to ES as one of the most recurrent limitations between ES and ERA (Forbes and Calow, 2013; Maltby et al., 2018; Munns et al., 2016, 2017). This is partly because of the difficulty in extrapolating organism's responses to higher levels of biological organization or from tested to untested species (Devos et al., 2019). Moreover, there is also the risk to choose ERA endpoints that are not representative of a particular ES (Syberg et al., 2017). The valuation methods (monetary and non-monetary) of ES also presented downsides due to their intrinsic limitations (Deacon et al., 2015) and their controversial aspect (Devos et al., 2019; Forbes et al., 2017; Forbes and Calow, 2013; Galic et al., 2012).

The spatial and temporal scales of the assessments also presented drawbacks. For example, the scale problem arises when trying to account for the effect of stressors on the SPU and the delivery of ES across multiple spatial and temporal scales (Devos et al., 2019; EFSA Scientific Committee, 2016). Besides the scale, some papers relied on semi-quantitative approaches that require expert judgement, which can be subject to a certain degree of bias (Munns et al., 2016). Also, in most cases, weighting is applied in these semi-quantitative studies to determine the relevance of the risks and/or ES. Some papers (e.g. Cabral et al., 2015) assumed an equal relevance of ES (i.e. equal weights), which is not always representative given that their importance can vary depending on the context (Caro et al., 2020; Xu et al., 2016). Furthermore, Cabral et al. (2015) also conclude that the addition of risk scores is not always the best method to determine the cumulative impacts on a habitat because it can ignore the synergistic and antagonistic interactions between stressors.

Other problems encountered in the ES-ERA papers were related to the lack of data availability for conducting the assessments (Arkema et al., 2015; Berrouet et al., 2018; Cabral et al., 2015; Caro et al., 2020; Culhane et al., 2019; Galic et al., 2019; Willaert et al., 2019), level of complexity and high data requirement (Apitz, 2012; Maltby et al., 2018; Sample et al., 2016; Xing et al., 2020) and uncertainties (Galic et al., 2012, 2019; Gilioli et al., 2017; Kang et al., 2018; Xu et al., 2016). These uncertainties could be due to data used in the study (Galic et al., 2012; Pártl et al., 2017), assumptions and models used (Galic et al., 2012). Both quantitative and conceptual models describing the relationships between human activities, pressures and ES require validation (Culhane et al., 2019; Forbes et al., 2017).

3.2.3. Group 3: LCA-ERA

3.2.3.1. Integration methods. Several review papers (Guinée et al., 2017; Harder et al., 2015; Herva and Roca, 2013; Kobayashi et al., 2015b; Muazu et al., 2021; Tsang et al., 2017) have addressed the different approaches on how LCA and ERA can be integrated over the years. For example, the most recent review from Muazu et al. (2021) described different integration methods (i.e. parallel integration, subset integrations and complementary integration).

Almost half of the reviewed papers included ERA into LCA by developing new spatially-differentiated CFs (Csiszar et al., 2016; Lin et al., 2018; Tian and Bilec, 2018) and impact pathways (Breedveld, 2013; Crenna et al., 2020; Fransman et al., 2017; Gust et al., 2016; Harder et al., 2016; Sala and Goralczyk, 2013), or by modifying or suggesting changes in already existing LCA impacts categories (Milazzo and Spina, 2015; Müller et al., 2017; Pizzol et al., 2011).

Another majority of the reviewed papers followed an approach where LCA and ERA were conducted independently, and their results were combined later on qualitatively or quantitatively. Among these papers, some of them aggregated LCA and ERA results to single score values (Hou et al., 2017; Kobayashi et al., 2015a). Some papers suggested conceptually the use of decision analysis methods, such as Multiple-Criteria Decision Analysis (MCDA), as a way to quantitatively combine the results from individually performed LCA and ERA studies on the same case study (Linkov et al., 2017; Linkov and Seager, 2011). Contrarily, one paper used a qualitative approach (i.e. fuzzy logic reasoning) to combine the results from LCA and ERA (Herva et al., 2012).

Other papers just qualitatively combined their results from LCA and ERA by plotting them together, i.e. using a displaying integration approach (Ribera et al., 2014; Walser et al., 2017) or, by normalizing the LCA results and compare them with the ERA results (Lemming et al., 2012). A paper used the outputs of LCA to identify high environmental impact processes in the life cycle of a product and then conduct an ERA based on those specific processes (Ayoub et al., 2015). Meanwhile, other papers conducted LCA and ERA assessment separately (Barberio et al., 2014; Walser et al., 2014; Weyell et al., 2020).

Finally, few papers relied on the incorporation of the life-cycle thinking concept in ERA (Aissani et al., 2012; Kuczenski et al., 2011; Liu et al., 2012; Shatkin and Kim, 2015). This mainly consists of using life cycle thinking to identify life cycle stages with the highest potential impacts on ecological processes, although there has been no actual application of the LCA methodology.

3.2.3.2. Limitations. The limitations in the approaches for integrating LCA and ERA varied across the papers. A recurrent limitation was the lack of available data (Barberio et al., 2014; Crenna et al., 2020; Gust et al., 2016; Liu et al., 2012; Muazu et al., 2021) and its low quality (Müller et al., 2017), but also high data requirement was a problem (Walser et al., 2017). Besides the data, the differences in the model structure and scopes from LCA and ERA are an obstacle when integrating both approaches (Harder et al., 2015; Muazu et al., 2021; Tsang et al., 2017). Moreover, proper exposure pathways need to be selected in order to reduce bias in the assessment (Crenna et al., 2020; Csiszar et al., 2016; Kobayashi et al., 2015a; Tian and Bilec, 2018).

The uncertainties due to model assumptions (Harder et al., 2016; Lemming et al., 2012; Sala and Goralczyk, 2013) and the parameters used (Kobayashi et al., 2015a; Lin et al., 2018) were also recognized. Some papers described the problems when integrating different spatial and temporal scales (Barberio et al., 2014; Guinée et al., 2017; Hou et al., 2017; Tsang et al., 2017; Walser et al., 2014) and when implementing the approaches due to their complexity (Herva and Roca, 2013; Hou et al., 2017; Liu et al., 2012; Sala and Goralczyk, 2013). For example, Hou et al. (2017) highlight that their method could become very complex when assessing multiple stressors instead of one. Other papers recognize that their approach could be biased when using MCDA methods, for combining the results of ERA and LCA, due to their subjective characteristics (Herva et al., 2012; Herva and Roca, 2013; Linkov et al., 2017; Linkov and Seager, 2011). Few papers reflected that a limitation of their approaches was because they did not consider the whole life cycle in their assessments (Ayoub et al., 2015; Barberio et al., 2014; Tian and Bilec, 2018).

Review papers also highlighted some limitations when integrating LCA and ERA. Among these drawbacks it was mentioned the risk of double-counting (Harder et al., 2015; Muazu et al., 2021), inconsistencies when choosing parameters and within the modelling (Harder et al., 2015; Muazu et al., 2021), omissions of crucial components from LCA and ERA (Harder et al., 2015; Muazu et al., 2021) and the lack of guidance and high ambiguity when establishing a procedure to integrate LCA and ERA (Harder et al., 2015; Muazu et al., 2021).

4. Classification of the case studies based on the type of integration

A first proposal was done to develop a classification scheme for categorising the case studies under a certain type of integration. A total of 59 case studies were selected for the in-depth analysis of which 25 belong to the LCA-ES group, 19 to the ES-ERA group and 15 to the LCA-ER group. In this section, we describe the different categories integration and we apply the scheme to the selected case studies.

A detailed list of the case studies assigned to a particular category can be found in the SM-A.

4.1. Description of the categories

4.1.1. Post-analysis: Qualitative interpretation

In the post-analysis category (Figs. 5, 1), methodology A and methodology B are conducted independently from each other and there is no actual integration between the different phases of the methodologies nor a quantitative integration of both results. There is simply a combined qualitative interpretation of the results. An example of this category is the research from Battista et al. (2017). The authors conducted an ERA and ESA for the same case study. The ERA determined the impacts of multiple stressors on a coastal and marine ecosystem, whereas the ESA assessed the capacity of those ecosystems to deliver ES.

4.1.2. Integration through the combination of results

In this case, methodology A and methodology B are also conducted independently from each other but their results are combined/aggregated by introducing an additional quantitative or qualitative step (Figs. 5, 2). The integration always occurs at the endpoint of the cause-effect chain. An illustrative example of this category is the paper carried out by Briones-Hidrovo et al. (2020), where an ESA and a LCA were conducted independently for the same case study. The results at the endpoint level from the ESA and LCA were aggregated quantitatively to a single score of environmental performance expressed in monetary terms. Qualitative aggregation of the results can be illustrated with the research of Herva et al. (2012), where LCA and ERA were conducted independently and from which three aggregated indicators were obtained (i.e. a cancer risk index, a hazard quotient and variation in the ecological footprint). A fuzzy logic approach (i.e. if-then rules) was applied to specify qualitative criteria in order to determine the level of acceptance of an eco-design, which was expressed through an indicator named Fuzzy Eco-Design Index.

4.1.3. Integration through the complementation of a driving method

In this category, methodology B drives the assessment and incorporates parts of the cause-effect chain of methodology A within its own cause-effect chain (Figs. 5, 3). The integration of methodology A in the cause-effect chain of the driving methodology B can be done in some specific parts (partial) (Fig. 5, 3a) or along the complete chain (Fig. 5, 3b).

- Partial chain of the driving methodology
- Level 1: Scoping-System Analysis phase.

In level 1, the integration of A in the driving methodology B occurs at the beginning of the cause-effect chain, where the inputs for methodology B are being complemented (Fig. 5, 3a-i). This level can be further subdivided into two types. In type I, the outputs of methodology A are used to complement the inputs of the driving methodology B. For example, the paper of Chaplin-Kramer et al. (2017) used the outputs from an ESA tool to modify and complement the flows in the LCI. This complemented LCI was used to models the impacts with LCIA methods. On the other hand, in type II, the inputs from methodology A are being used to complement the inputs of the driving methodology B. For instance, flows from a LCI were used in the paper of Bruel et al. (2016) to complement the inputs of a bio-economic model of ES. In this case, the driving method was ESA.

- Level 2: Scoping-System Analysis phase and Impact Analysis phase at midpoint level

Level 2 occurs when the integration of A in B happens both at the beginning and middle of the cause-effect chain of B (Fig. 5, 3a-ii). This level can be illustrated with the paper of Galic et al. (2019), where the outputs from an ERA model (i.e. a model that integrates the environmental fate of pollutants and their impacts on the food webs in aquatic environments) were used in the driving method ESA. The selection of relevant ES depended on the Ecological ERA receptors so there could be a direct linkage between ESA and ERA. In addition to this, the modelled effect of the stressor on the receptors represents the changes in the biophysical structures and processes that are assessed in ESA. As it can be seen, the ERA model influences the beginning and middle part of the cause-effect chain of ESA. The modelling in ESA continues by quantifying the impacts on ES supply and their value.

- Level 3: Impact Analysis phase only at midpoint level

Level 3 occurs when the integration of A in B happens only at the middle of the cause-effect chain of B (Fig. 5, 3a-iii). An example of this level is the paper of Brandão and Milà i Canals (2013). To quantify the impact of land-use change on the biotic productivity potential, spatially-differentiated CFs for LCA were calculated at a midpoint level based on an indicator for this specific ES.

- Level 4: Impact Analysis phase both at midpoint and endpoint level

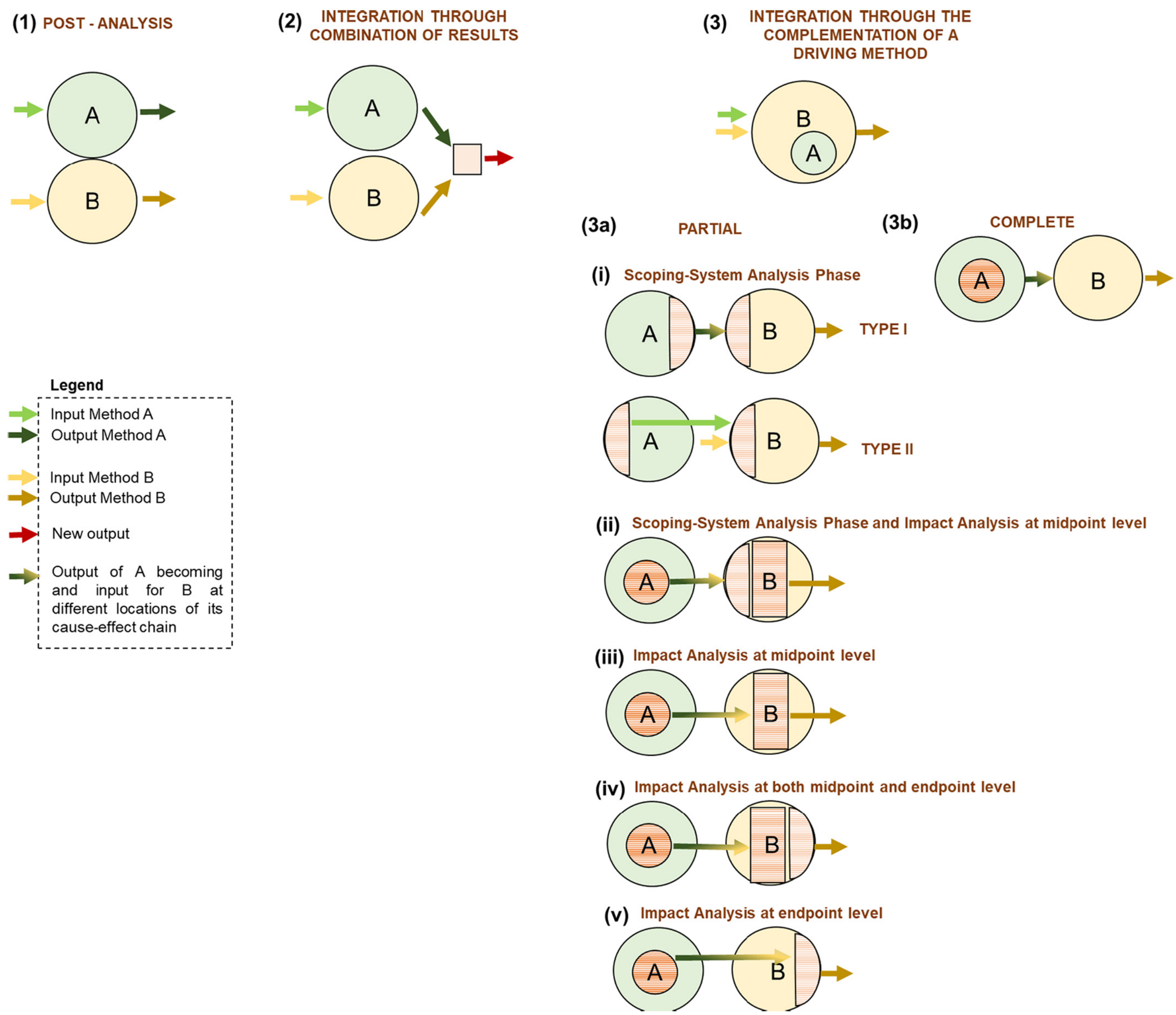


Fig. 5. Classification scheme based on the types of integration. (1) Post-analysis, (2) Integration through the combination of results, (3) Integration through the complementation of a driving method. Division of category 3 (Complementation of a driving method) into two subs-categories 3a) Partial: Level 1 (Type I and Type II), Level 2, Level 3, Level, Level 5 and 3b).

Level 4 occurs when the integration of A in B happens in the middle and end of the cause-effect chain of B (Fig. 5, 3a-iv). An example is the paper of Othoniel et al. (2019) where CFs were calculated both at midpoint and endpoint level for LCA using ES models in order to assess impacts of land use in Luxembourg. The midpoint CFs are calculated by integrating different models and tools (e.g. InVEST, ecosystem processes models, land-use models) in an integrated earth system model. Meanwhile, the endpoint CFs are calculated using ES monetary valuation methods.

- Level 5: Impact Analysis phase only at endpoint level

Level 5 occurs when the integration of A in B happens only at the end of the cause-effect chain of B (Fig. 5, 3a-v). This level can be illustrated by the paper of Xing et al. (2020), where ES values were used to define and calculate risk scores in an ERA. The risk scores are considered to be located at the end of the cause-effect chain in the impact analysis of ERA (Table D1 in SM-B). For this reason, the integration of ESA and ERA is only occurring at this level when analysing the research of Xing et al. (2020).

- Complete chain of the driving methodology

In this subcategory, parts of the non-driving methodology (A) are integrated along the entire cause-effect chain of the driving methodology (B). This means that both the scoping-system analysis and impact phase of methodology B are being complemented by methodology A (Fig. 5). To illustrate this sub-category, the paper of Gilioli et al. (2017) is taken as a reference. In this case study, ERA and ESA are the driving and non-driving methodologies respectively. The first phase of the assessment corresponds to the scoping and selection of relevant SPUs and the ES they provide. Also, the main stressor is identified, which is an invasive snail species. In the impact analysis phase, the variation over time of the stressors (i.e. invasive species) is evaluated by using population models. The results were considered by a panel of experts, who assessed how changes in the biomass of the invasive species could potentially affect the SPUs and the supply of ES. They followed a semi-quantitative approach to determine and assign risk scores to the ES. This case study is a clear example of an ERA based on

ES, where the latter is considered throughout the entire cause-effect chain of ERA.

4.2. Classification of the case studies

The selected case studies were classified according to the categories introduced in Section 4.1. We summarize the main findings in Fig. 6, which highlights the distribution of the papers according to the type of integration. A more detailed description of the characteristics of case studies and how they were classified can be found in the SM-A and SM-B.

In general, our results show that most of the case studies (42 papers ~71%) used an approach where a driving methodology was being complemented by parts of a non-driving one (Section 4.1 and Fig.E1 in SM-B). Within this category, 35 papers belong to the partial cause-effect chain sub-category and 7 to the complete sub-category (Section 4.1). Nine papers were classified under the integration through the combination of results category and eight under the post-analysis category.

4.2.1. Classification of LCA-ES case studies

For the LCA-ES group, most of the case studies complemented the cause-effect chain of a driving methodology (22 papers). In most cases, LCA was the driving methodology and parts of ESA were incorporated. Within this category, nineteen papers complemented partially the cause-effect chain of LCA and only three complemented it completely. The remaining case studies were categorized as integration through the combination of results (two papers) and post-analysis (one paper) (Fig. 6 and Fig. E2 in the SM-B).

4.2.2. Classification of ES-ERA case studies

For the group of papers of the ES-ERA group, also most of the case studies (fourteen papers) complemented a driving methodology. In this case, the most frequent driving methodology was ERA. From these fourteen papers, only three papers complemented the complete cause-effect chain of ERA and the remaining eleven papers complemented partially the cause-effect chain. Also, the case studies from this group represent the largest contribution (~44% of the papers) to the integration approach based on the combination of the results (Fig. 6 and Fig. E2 in the SM-B).

4.2.3. Classification of LCA-ERA case studies

Most of the case studies from the LCA-ERA group (six papers) contribute to the category of post-analysis. The next dominant category is the complementation of a driving methodology with six papers in total. LCA was again the driving methodology. This complementation can be partial (five papers) or complete (one paper) (Fig. 6 and Fig. E2 in the SM-B).

4.3. Potential advantages and limitations of the main integration categories

Each of the integration approaches can have potential advantages and limitations. The post-analysis is an option to capture both local and global effects without having to connect quantitatively the results from two different methodologies. It also avoids the problems caused by having two methodologies with different structures and, in some cases, reduce the amount of data required for the assessments. Nonetheless, this approach is not a real integration and there is a risk of double-counting impacts (Muazu et al., 2021). In addition, the results could be non-comparable and difficult to understand by decision-makers. Besides this, performing two independent assessments can be time-consuming, which is something that could make this approach non-appealing to decision-makers (Muazu et al., 2021). Most of the case studies belonging to the LCA-ERA group used this approach, probably to tackle the differences of the methodologies' structures and because it is not easy to develop spatially-differentiated cause-effect pathways.

The integration through the combination of results is also an approach capturing local and global effects by combining the results from two different methodologies. This approach also avoids the problems resulting from the integration of methodologies with different structures (Briones-Hidrovo et al., 2020; Linkov et al., 2017). However, like the post-analysis approach, there is a risk of double-counting impacts and it can be time-consuming (Briones-Hidrovo et al., 2020; Muazu et al., 2021). The issue of having non-comparable results is often fixed by aggregating and/or combining the results (Linkov et al., 2017). Nonetheless, aggregation has several problems. First, there are inherent biases associated with this process that need to be considered carefully and second, sometimes it is not easy to find the appropriate aggregation metrics (Bakshi and Small, 2011; Zhang et al., 2010a). The majority of the case studies that used this

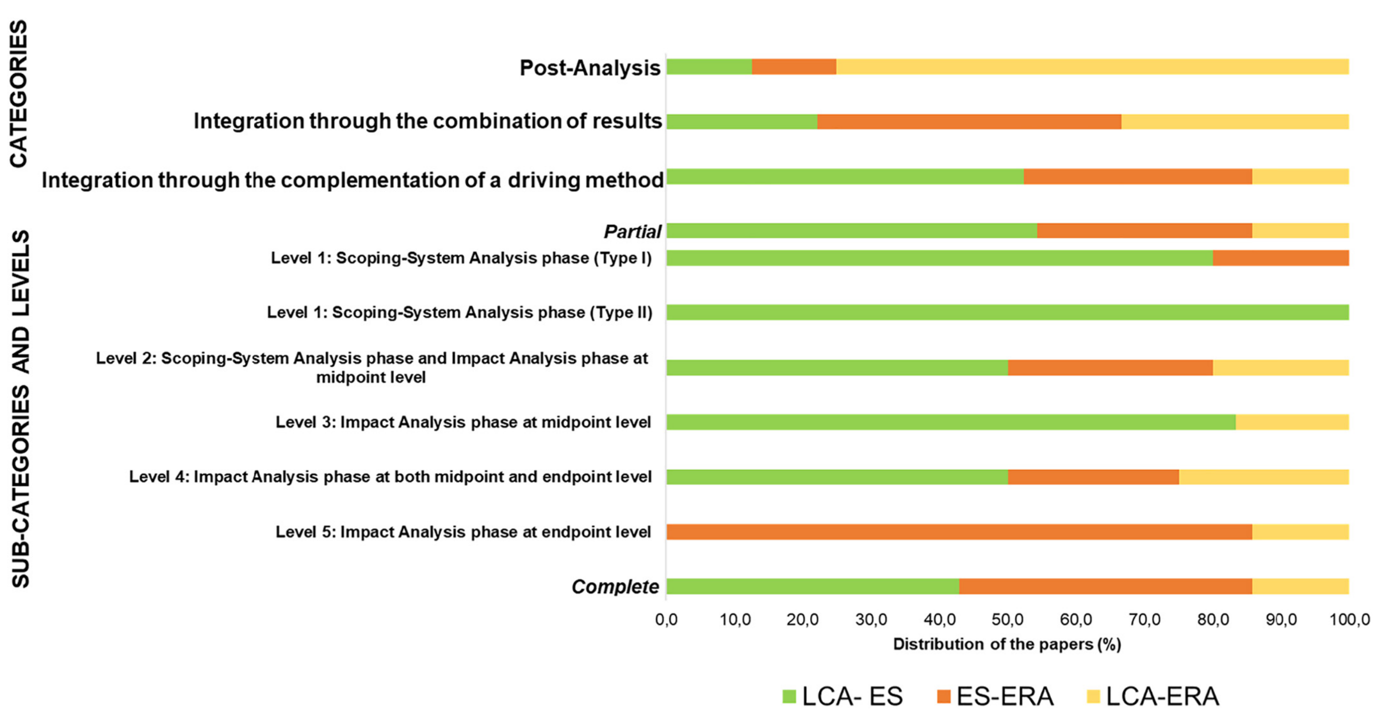


Fig. 6. Classification of the selected case studies under an integration approach and level (for partial integration). The figure shows the distribution of the group of papers (LCA-ES, ES-ERA, LCA-ERA) according to a category and sub-category.

approach belong to the ES-ERA group. By applying this approach, the papers were able to connect multiple stressors to one or several ES in a marine context. To do this, they relied on semi-quantitative methods due to a lack of data availability (Sections 3.2.1.2, 3.2.2.2, 3.2.3.2).

In the case of the integration through the complementation of a driving method, the local and global effects are captured by incorporating elements from one methodology into another. There is a deeper linkage between the two methodologies by using this approach because a methodology's structure is being modified. Though the assessments are more realistic and less time-consuming by using this approach, its development is more complex (i.e. identifying indicators and models, establishing cause-effect chains). This approach has a high data requirement and it would be affected in case of data gaps. Besides this, there is also the risk of double-counting impacts (Muazu et al., 2021). As mentioned beforehand, most of the case studies used this approach, especially the case studies belonging to the LCA-ES and ES-ERA groups. In the case of the LCA-ES group, this approach was used to create an impact pathway for ES within LCA, and this way it captures the local effects and develop impact categories for ES. LCA allows the connection of a human activity with ES but mainly in the context of land use. Other stressors besides land occupation have not been linked completely to an ES (Blanco et al., 2018). For the ES-ERA group, this approach was used to also connect a stressor to an ES, mainly through a representative receptor. The main question that this integration approach answers is which are the ES with a higher risk to be severely impacted by human activities.

5. Conclusions and perspectives

5.1. Concluding remarks

Several methodological approaches have been developed to assess more comprehensively the impacts of human activities on terrestrial and aquatic ecosystems through the integration of frameworks such as LCA, ERA and ESA. To the best of our knowledge, the integration of these three methodologies has been not been attempted and the studies focus on the integration of two methodologies (i.e. LCA-ESA, ERA-ESA and LCA-ERA). By conducting a literature review, we identified different integration approaches, which were used a baseline to develop a first attempt of a classification system based on the type of integration. The main categories identified were integration post-analysis, integration through the combination of results and integration through the complementation of a driving method. The latter one is more comprehensive and accurate compared to the other categories, however its development is more complex and it could lead to high data requirement. Besides this, all the categories have the risk of double-counting impacts.

After analysing the different integration approaches, none of them can be categorized as a *full* integration. To have a *fully* integrated environmental impact assessment approach, all the components of one methodology need to be integrated with all the components from the other methodology along the whole cause-effect chain, as depicted in Fig. 3. These case studies merely incorporated certain components or combined their results. One of the most recurrent constraints in a *full integration* is the differences in the structure of the assessments and models of LCA, ERA and ESA. This is something already pointed out for LCA-ERA integration (Muazu et al., 2021), but that is also evident for LCA-ES and ES-ERA integration. These differences can become more challenging when attempting the integration of the three methodologies. The integration of all components of the methodologies is perhaps not required to conduct a comprehensive assessment. Such assessment could be limited by the complexity of integrating the three methodologies and hence, leading to highly uncertain results in case of limited data availability. The key aspect is to select the appropriate components from each methodology to make a holistic assessment, where both the burdens on ecosystems and the benefits of the ecosystems are considered.

Currently, the integration approaches are not really standardized. The challenge is higher when integrating a methodology, such as ESA, where there is a lack of a consensus on a homogenous terminology and steps to conduct the assessment (Heink and Jax, 2019; La Notte et al., 2017). This classification system provides a standardized picture of the integration possibilities and further studies could rely on this manuscript to identify specific integration approaches and adopt a consistent terminology or further improve the classification system. Also, it could potentially be used to conceptualize a framework that integrates all three methodologies for conducting a holistic sustainability assessment.

5.2. Perspectives

5.2.1. Perspectives on the classification system

The classification scheme proposed is a first attempt to structure the different approaches of integrating methodologies. However, this is not the only way to understand integration and classify the case studies (e.g. Harder et al., 2015; Kobayashi et al., 2015b; Muazu et al., 2021). For example, more subcategories (or levels) could be added to the category *integration through the complementation of a driving method* based on the parts of the cause-effect chain of the non-driving methodology that were incorporated into the driving one.

5.2.2. Perspectives on the integration of LCA, ERA and ESA

LCA, ERA and ESA cannot by themselves provide a full assessment of both positive and negative impacts from human activities at different geographical scales because of their structure and scope (i.e. ESA focuses on the benefits obtained from ecosystems, ERA on impacts at a local scale and LCA on multiple impacts at a global scale). An integration would make it possible to evaluate not only the impacts of humans on ecosystems and its services, but also how natural systems affect humans, i.e. human well-being, by considering multiple interactions between the human/industrial systems and the natural systems. By capturing all these interactions, a thorough coverage of these multiscale positive and negative effects of human activities could be achieved.

Moreover, integrating these methodologies would also provide a better view of potential trade-offs of environmental burdens versus benefits (i.e. trade-offs between ecosystem functioning, ecosystem services an socio-economic activities) which would lead to a more transparent assessment of human activities on ecosystems and facilitate decision-making processes in different contexts.

One of these contexts are aquatic and terrestrial ecosystems where numerous human activities take place. For instance, a wide range of activities take place in the Belgian part of the North Sea, such as fishing, shipping, power generation, dredging, sand extraction, cables and pipelines, military exercises, scientific research, tourism and cultural heritage (Belgian Royal Decree, 2014). This becomes a challenge when it comes to ensuring a sustainable use and management of the marine resources for economic growth and human well-being, while still preserving healthy marine ecosystems. For this reason, it has become extremely valuable for stakeholders to understand, measure and quantify the impacts caused by human activities on the marine environment. A holistic sustainability impact assessment, which considers the integration of LCA, ERA and ESA, could serve as a supporting tool for the different stakeholders by giving them a better understanding of the burdens and benefits of all these human activities.

Another example for the application of an integrated framework could be on ecosystems that have been converted by humans (e.g. cultivated land, artificial lakes, artificial forests, artificial reefs, etc.). Barot et al. (2017) argue that man-made ecosystems, such as agro-ecosystems, can provide services (e.g. food provisioning) but also non-desirable effects (e.g. nutrient leaching). Therefore, these services cannot really be considered as ‘ecosystem services’ because of their negative effects on other ecosystems. An ESA would not be appropriate to apply (as single use) in this context and would require the support of other methodologies, such as LCA and ERA, to exhaustively assess the impacts caused by these anthropized ecosystems.

Several challenges could be faced during the integration of LCA, ERA and ESA. Some of these limitations are the identification and selection of relevant parameters and indicators for all the methodologies. This needs to be done carefully to avoid double-counting and to find appropriate aggregation metrics. The next limitation is the harmonization of different spatial and temporal resolutions and the structures of the methodologies. Additionally, it is crucial to assess which data is available to identify potential integration approaches. This means that the integration should be partly data-driven. The availability of this data will also depend on the ecosystem targeted, which at the end influences the selected integration approach. For instance, the development of impact pathways and CFs for LCA in a marine context needs further research compared to terrestrial ecosystems due to lack of data (Woods et al., 2016). This could be one reason why most of the LCA-ES papers assessed mainly impacts on terrestrial ecosystems compared to aquatic ecosystems. Another example is found in several ES-ERA papers, were some case studies assessed impacts on marine ecosystems by relying on semi-quantitative integration approaches also because of low data availability (Cabral et al., 2015; Caro et al., 2020; Culhane et al., 2019; Willaert et al., 2019).

Another challenge is the assessment of the positive effects of human activities. Most of the papers focused on the negative impacts and the positive ones were not properly addressed. To have a holistic sustainability assessment, this aspect should not be overlooked. There are different contexts where humans can contribute positively to ecosystem-functioning (Alvarenga et al., 2020). These positive aspects should be considered in future research.

Finally, one potential integration approach consists of aggregation of results. However, there are some risks associated with aggregating indicators from different methodologies. Among them, double-counting is mostly mentioned and this is introduced when indicators are overlapping. This risk should be carefully considered when conducting an assessment with integrated methodologies by avoiding the aggregation of redundant impact category indicators between LCA, ERA and ESA, by clearly indicating the redundancy or by quantifying the redundancy of indicators using correlation matrices (Genovese et al., 2017; ISO, 2006). In addition to double-counting risk, aggregation consolidates multivariate data. If not done carefully, aggregation can lead to the loss of information, misinterpretation of results and the communication of results in not a transparent and comprehensive way (Prado et al., 2020).

CRediT authorship contribution statement

Laura Vittoria De Luca Peña: Conceptualization, Formal analysis, Writing – original draft. **Sue Ellen Taelman:** Conceptualization, Formal analysis, Writing – review & editing, Funding acquisition. **Nils Préat:** Conceptualization, Formal analysis, Writing – review & editing. **Lieselot Boone:** Conceptualization, Formal analysis, Writing – review & editing. **Katrien Van der Biest:** Writing – review & editing. **Marco Custódio:** Writing – review & editing. **Simon Hernandez Lucas:** Writing – review & editing. **Gert Everaert:** Funding acquisition. **Jo Dewulf:** Conceptualization, Formal analysis, Writing – review & editing, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work was financially supported by the Agentschap Innoveren & Ondernemen (Flanders Innovation and Entrepreneurship Agency) within the framework of the Sustainable Marine Ecosystem Services (SUMES) project [VLAIO grant number: HBC.2019.2903].

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.152125.

References

Aissani, L., Jabouille, F., Bourgois, J., Rousseaux, P., 2012. A new methodology for risk evaluation taking into account the whole life cycle (LCRA): validation with case study. Process Saf. Environ. Prot. 90 (4), 295–303. https://doi.org/10.1016/j.psep.2011.10.003.

Alejandre, E.M., van Bodegom, P.M., Guinée, J.B., 2019. Towards an optimal coverage of ecosystem services in LCA. J. Clean. Prod. 231, 714–722. https://doi.org/10.1016/j.jclepro.2019.05.284.

Alvarenga, R.A.F., Huysveld, S., Taelman, S.E., Sfez, S., Préat, N., Cooreman-Algoed, M., Sanjuan-Delmás, D., Dewulf, J., 2020. A framework for using the handprint concept in attributional life cycle (sustainability) assessment. J. Clean. Prod. 265, 1–9. https://doi.org/10.1016/j.jclepro.2020.121743.

Apitz, S.E., 2012. Conceptualizing the role of sediment in sustaining ecosystem services: sediment-ecosystem regional assessment (SEcoRA). Sci. Total Environ. 415, 9–30. https://doi.org/10.1016/j.scitotenv.2011.05.060.

Arbault, D., Rivière, M., Rugani, B., Benetto, E., Tiruta-Barna, L., 2014. Integrated earth system dynamic modeling for life cycle impact assessment of ecosystem services. Sci. Total Environ. 472, 262–272. https://doi.org/10.1016/j.scitotenv.2013.10.099.

Arkema, K.K., Verutes, G.M., Wood, S.A., Clarke-Samuels, C., Rosado, S., Canto, M., Rosenthal, A., Ruckelshaus, M., Guannel, G., Toft, J., Faries, J., Silver, J.M., Griffin, R., Guerry, A.D., 2015. Embedding ecosystem services in coastal planning leads to better outcomes for people and nature. Proc. Natl. Acad. Sci. U. S. A. 112 (24), 7390–7395. https://doi.org/10.1073/pnas.1406483112.

Aven, T., 2016. Risk assessment and risk management: review of recent advances on their foundation. Eur. J. Oper. Res. 253 (1), 1–13. https://doi.org/10.1016/j.ejor.2015.12.023.

Awuah, K.F., Jegede, O., Hale, B., Siciliano, S.D., 2020. Introducing the adverse ecosystem service pathway as a tool in ecological risk assessment. Environ. Sci. Technol. 54 (13), 8144–8157. https://doi.org/10.1021/acs.est.9b06851.

Ayoub, N., Musharavati, F., Pokharel, S., Gabbar, H.A., 2015. Risk based life cycle assessment conceptual framework for energy supply systems in large buildings. J. Clean. Prod. 107, 291–309. https://doi.org/10.1016/j.jclepro.2015.04.075.

Baker, J., Sheate, W.R., Phillips, P., Eales, R., 2013. Ecosystem services in environmental assessment - help or hindrance? Environ. Impact Assess. Rev. 40 (1), 3–13. https://doi.org/10.1016/j.eiar.2012.11.004.

Bakshi, B., Small, M.J., 2011. Incorporating ecosystem services into life cycle assessment. J. Ind. Ecol. 15 (4), 477–478. https://doi.org/10.1111/j.1530-9290.2011.00364.x.

Bakshi, B.R., Ziv, G., Lepech, M.D., 2015. Techno-ecological synergy: a framework for sustainable engineering. Environ. Sci. Technol. 49 (3), 1752–1760. https://doi.org/10.1021/es5041442.

Barberio, G., Scalbi, S., Buttol, P., Masoni, P., Righi, S., 2014. Combining life cycle assessment and qualitative risk assessment: the case study of alumina nanofluid production. Sci. Total Environ. 496, 122–131. https://doi.org/10.1016/j.scitotenv.2014.06.135.

Barot, S., Yé, L., Abbadie, L., Blouin, M., Frascaria, N., 2017. Ecosystem services must tackle anthropized ecosystems and ecological engineering. Ecol. Eng. 99, 486–495. https://doi.org/10.1016/j.ecoleng.2016.11.071.

Battista, W., Karr, K., Sarto, N., Fujita, R., 2017. Comprehensive assessment of risk to ecosystems (CARE): a cumulative ecosystem risk assessment tool. Fish. Res. 185, 115–129. https://doi.org/10.1016/j.fishres.2016.09.017.

Belgian Royal Decree, 2014. Marine Spatial Plan for the Belgian Part of the North Sea.

Berrouet, L.M., Machado, J., Villegas-Palacio, C., 2018. Vulnerability of socio—ecological systems: a conceptual framework. Ecol. Indic. 84, 632–647. https://doi.org/10.1016/j.ecolind.2017.07.051 (February 2017).

Bjørn, M.O., Molin, C., Laurent, A., 2017. Main characteristics of LCA. In: Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I. (Eds.), Life Cycle Assessment: Theory and Practice. Springer, Cham, pp. 9–16 https://doi.org/10.1007/978-3-319-56475-3.

Blanco, C.F., Marques, A., van Bodegom, P.M., 2018. An integrated framework to assess impacts on ecosystem services in LCA demonstrated by a case study of mining in Chile. Ecosyst. Serv. 30, 211–219. https://doi.org/10.1016/j.ecoser.2017.11.011.

Boerema, A., Rebelo, A.J., Bodi, M.B., Esler, K.J., Meire, P., 2017. Are ecosystem services adequately quantified? J. Appl. Ecol. 54 (2), 358–370. https://doi.org/10.1111/1365-2664.12696.

Boone, L., Roldán-Ruiz, I., Van Linden, V., Muylle, H., Dewulf, J., 2019. Environmental sustainability of conventional and organic farming: accounting for ecosystem services in life cycle assessment. Sci. Total Environ. 695, 133841. https://doi.org/10.1016/j.scitotenv.2019.133841.

Bos, U., Horn, R., Beck, T., Lindner, Jan P., Fischer, M., 2016. LANCA - characterization factors for life cycle impact assessment (version 2.). Retrieved December 2, 2020 fromFraunhofer-Institut fur Bauphysik IBP. http://publica.fraunhofer.de/eprints/urn_nbn_de_0011-n-3793106.pdf.

Bragaglio, A., Braghieri, A., Pacelli, C., Napolitano, F., 2020. Environmental impacts of beef as corrected for the provision of ecosystem services. Sustainability (Switzerland) 12 (9), 1–15. https://doi.org/10.3390/su12093828.

Brandão, M., Milà i Canals, L.M., 2013. Global characterisation factors to assess land use impacts on biotic production. Int. J. Life Cycle Assess. 18 (6), 1243–1252. https://doi.org/10.1007/s11367-012-0381-3.

Breedveld, L., 2013. Combining LCA and RA for the integrated risk management of emerging technologies. J. Risk Res. 16 (3–4), 459–468. https://doi.org/10.1080/13669877.2012.729526.

Briones-Hidrovo, A., Uche, J., Martínez-gracia, A., 2020. Determining the net environmental performance of hydropower: a new methodological approach by combining life cycle and ecosystem services assessment. Sci. Total Environ. 712, 136369. https://doi.org/10.1016/j.scitotenv.2019.136369.

Bruel, A., Troussier, N., Guillaume, B., Sirina, N., 2016. Considering ecosystem Services in Life Cycle Assessment to evaluate environmental externalities. Procedia CIRP 48, 382–387. https://doi.org/10.1016/j.procir.2016.03.143.
Buckwell, A., Fleming, C., Smart, J., Mackey, B., Ware, D., Hallgren, W., Sahin, O., Nalau, J., 2018. Valuing aggregated ecosystem services at a national and regional scale for Vanuatu using a remotely operable, rapid assessment methodology. No. 2057-2018–4173, 22. Retrieved May 4, 2021 from http://ageconsearch.umn.edu/record/273524.
Burkhard, B., Santos-Martin, F., Nedkov, S., Maes, J., 2018. An operational framework for integrated mapping and assessment of ecosystems and their services (MAES). One Ecosyst. 3. https://doi.org/10.3897/oneeco.3.e22831.
Cabral, P., Levrel, H., Schoenn, J., Thiébaut, E., Le Mao, P., Mongruel, R., Rollet, C., Dedieu, K., Carrier, S., Morisseau, F., Daures, F., 2015. Marine habitats ecosystem service potential: a vulnerability approach in the Normand-breton (Saint Malo) gulf, France. Ecosyst. Serv. 16, 306–318. https://doi.org/10.1016/j.ecoser.2014.09.007.
Callesen, I., 2016. Biodiversity and ecosystem services in life cycle impact assessment – inventory objects or impact categories? Ecosyst. Serv. 22 (March), 94–103. https://doi.org/10.1016/j.ecoser.2016.09.021.
Cao, V., Margni, M., Favis, B.D., Deschênes, L., 2015. Aggregated indicator to assess land use impacts in life cycle assessment (LCA) based on the economic value of ecosystem services. J. Clean. Prod. 94, 56–66. https://doi.org/10.1016/j.jclepro.2015.01.041.
Caro, C., Carlos, J., Cunha, P.P., Teixeira, Z., 2020. Ecosystem services as a resilience descriptor in habitat risk assessment using the InVEST model. Ecol. Indic. 115 (April), 106426. https://doi.org/10.1016/j.ecolind.2020.106426.
Carpenter, S.R., Bennettt, E.M., Peterson, G.D., 2006. Scenarios for ecosystem services: an overview. Ecol. Soc. 11 (1).. http://www.ecologyandsociety.org/vol11/iss1/art29/.
Chaplin-Kramer, R., Sim, S., Hamel, P., Bryant, B., Noe, R., Mueller, C., Rigarlsford, G., Kulak, M., Kowal, V., Sharp, R., Clavreul, J., Price, E., Polasky, S., Ruckelshaus, M., Daily, G., 2017. Life cycle assessment needs predictive spatial modelling for biodiversity and ecosystem services. Nat. Commun. 8. https://doi.org/10.1038/ncomms15065.
Christie, M., Fazey, I., Cooper, R., Hyde, T., Kenter, J.O., 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. Ecol. Econ. 83 (2012), 67–78. https://doi.org/10.1016/j.ecolecon.2012.08.012.
Costanza, R., Daly, H.E., 1992. Natural capital and sustainable development. Conserv. Biol. 6 (1), 37–46. https://www.jstor.org/stable/2385849?seq=1#metadata_info_tab_contents.
Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387 (6630), 253–260. https://www.nature.com/articles/387253a0.
Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: how far have we come and how far do we still need to go? Ecosyst. Serv. 28, 1–16. https://doi.org/10.1016/j.ecoser.2017.09.008.
Cowling, R.M., Egoh, B., Knight, A.T., O'Farrell, P.J., Reyers, B., Rouget, M., Roux, D.J., Welz, A., Wilhelm-Rechman, A., 2008. An operational model for mainstreaming ecosystem services for implementation. Proc. Natl. Acad. Sci. U. S. A. 105 (28), 9483–9488. https://doi.org/10.1073/pnas.0706559105.
Crenna, E., Jolliet, O., Collina, E., Sala, S., Fantke, P., 2020. Characterizing honey bee exposure and effects from pesticides for chemical prioritization and life cycle assessment. Environ. Int. 138, 105642. https://doi.org/10.1016/j.envint.2020.105642 (September 2019).
Csiszar, S.A., Meyer, D.E., Dionisio, K.L., Egeghy, P., Isaacs, K.K., Price, P.S., Scanlon, K.A., Tan, Y.M., Thomas, K., Vallero, D., Bare, J.C., 2016. Conceptual framework to extend life cycle assessment using near-field human exposure modeling and high-throughput tools for chemicals. Environ. Sci. Technol. 50 (21), 11922–11934. https://doi.org/10.1021/acs.est.6b02277.
Culhane, F., Teixeira, H., Nogueira, A.J.A., Borgwardt, F., Trauner, D., Lillebø, A., Piet, G.J., Kuemmerlen, M., McDonald, H., O'Higgins, T., Barbosa, A.L., van der Wal, J.T., Iglesias-Campos, A., Arevalo-Torres, J., Barbière, J., Robinson, L.A., 2019. Risk to the supply of ecosystem services across aquatic ecosystems. Sci. Total Environ. 660, 611–621. https://doi.org/10.1016/j.scitotenv.2018.12.346.
Daily, G.C., 1997. Nature's Services: Societal Dependence on Natural Ecosystems.
Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. Front. Ecol. Environ. 7 (1), 21–28. https://doi.org/10.1890/080025.
de Groot, R.S., 1992. Functions of Nature: Evaluation of Nature in Environmental Planning, Management and Decision Making. Wolters-Noordhoff BV.
de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010a. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complex. 7 (3), 260–272. https://doi.org/10.1016/j.ecocom.2009.10.006.
de Groot, R., Fisher, B., Christie, M., Aronson, J., Braat, L., Gowdy, J., Haines-Young, R., Maltby, E., Neuville, A., Polasky, S., Portela, R., Ring, I., Blignaut, J., Brondízio, E., Costanza, R., Jax, K., Kadekodi, G.K., May, P.H., Mc Neely, J.A., Kadekodi, G.K., 2010b. Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations, pp. 9–40 https://doi.org/10.4324/9781849775489 March.
de Haes, U., Jolliet, O., Finnveden, G., Hauschild, M., Krewitt, W., Müller-Wenk, R., 1999. Best available practice regarding impact categories and category indicators in life cycle impact assessment. Int. J. Life Cycle Assess. 4 (2), 66–74. https://doi.org/10.1007/bf02979403.
Deacon, S., Norman, S., Nicolette, J., Reub, G., Greene, G., Osborn, R., Andrews, P., 2015. Integrating ecosystem services into risk management decisions: case study with spanish citrus and the insecticide chlorpyrifos. Sci. Total Environ. 505, 732–739. https://doi.org/10.1016/j.scitotenv.2014.10.034.
Devos, Y., Munns, W.R., Forbes, V.E., Maltby, L., Stenseke, M., Brussaard, L., Streissl, F., Hardy, A., 2019. Applying ecosystem services for pre-market environmental risk assessments of regulated stressors. EFSA J. 17 (S1), 1–24. https://doi.org/10.2903/j.efsa.2019.e170705.
Dewulf, J., Benini, L., Mancini, L., Sala, S., Blengini, G.A., Ardente, F., Recchioni, M., Maes, J., Pant, R., Pennington, D., 2015. Rethinking the area of protection "natural resources" in life cycle assessment. Environ. Sci. Technol. 49 (9), 5310–5317. https://doi.org/10.1021/acs.est.5b00734.
EFSA Scientific Committee, 2016. Guidance to develop specific protection goals options for environmental risk assessment at EFSA, in relation to biodiversity and ecosystem services. EFSA J. 14 (6). https://doi.org/10.2903/j.efsa.2016.4499.
EPA, 1998. Guidelines for ecological risk assessment. Risk assessment forum. Retrieved February 22, 2021 fromU.S. EPA. https://www.epa.gov/sites/production/files/2014-11/documents/eco_risk_assessment1998.pdf.
EPA, 2003. Framework for cumulative risk assessment. Risk assessment forum. Retrieved February 22, 2021 fromU.S. EPA. https://www.epa.gov/sites/production/files/2014-11/documents/frmwrk_cum_risk_assmnt.pdf.
EPA, 2016a. Human health risk assessment. Retrieved February 3, 2021 from https://www.epa.gov/risk/human-health-risk-assessment.
EPA, 2016b. Ecological risk assessment. Retrieved February 3, 2021 from https://www.epa.gov/risk/ecological-risk-assessment.
Faber, J.H., van Wensem, J., 2012. Elaborations on the use of the ecosystem services concept for application in ecological risk assessment for soils. Sci. Total Environ. 415, 3–8. https://doi.org/10.1016/j.scitotenv.2011.05.059.
Faber, J.H., Marshall, S., Van den Brink, P.J., Maltby, L., 2019. Priorities and opportunities in the application of the ecosystem services concept in risk assessment for chemicals in the environment. Sci. Total Environ. 651, 1067–1077. https://doi.org/10.1016/j.scitotenv.2018.09.209.
Fairman, R., Mead, C.D., Williams, W.P., 1998. Environmental risk assessment: approaches, experiences and information sources. Retrieved February 3, 2021 from https://www.eea.europa.eu/publications/GH-07-97-595-EN-C2/riskindex.html.
Flemström, K., Carlson, R., Erixon, M., 2004. Relationships between life cycle assessment and risk assessment. Retrieved February 3, 2021 fromNaturvårdsverket. https://www.naturvardsverket.se/Documents/publikationer/620-5379-5.pdf.
Forbes, V.E., Calow, P., 2013. Use of the ecosystem services concept in ecological risk assessment of chemicals. Integr. Environ. Assess. Manag. 9 (2), 269–275. https://doi.org/10.1002/ieam.1368.
Forbes, V.E., Salice, C.J., Birnir, B., Bruins, R.J.F., Calow, P., Ducrot, V., Galic, N., Garber, K., Harvey, B.C., Jager, H., Kanarek, A., Pastorok, R., Railsback, S.F., Rebarber, R., Thorbek, P., 2017. A framework for predicting impacts on ecosystem services from (sub)organismal responses to chemicals. Environ. Toxicol. Chem. 36 (4), 845–859. https://doi.org/10.1002/etc.3720.
Forbes, V.E., Railsback, S., Accolla, C., Birnir, B., Bruins, R.J.F., Ducrot, V., Galic, N., Garber, K., Harvey, B.C., Jager, H.I., Kanarek, A., Pastorok, R., Rebarber, R., Thorbek, P., Salice, C.J., 2019. Predicting impacts of chemicals from organisms to ecosystem service delivery: a case study of endocrine disruptor effects on trout. Sci. Total Environ. 649, 949–959. https://doi.org/10.1016/j.scitotenv.2018.08.344.
Förster, J., Barkmann, J., Fricke, R., Hotes, S., Kleyer, M., Kobbe, S., Kübler, D., Rumbaur, C., Siegmund-Schultze, M., Seppelt, R., Settele, J., Spangenberg, J.H., Tekken, V., Václavík, T., Wittmer, H., 2015. Assessing ecosystem services for informing land-use decisions: a problem-oriented approach. Ecol. Soc. 20 (3). https://doi.org/10.5751/ES-07804-200331.
Fransman, W., Buist, H., Kuijpers, E., Walser, T., Meyer, D., Zondervan-van den Beuken, E., Westerhout, J., Klein Entink, R.H., Brouwer, D.H., 2017. Comparative human health impact assessment of engineered nanomaterials in the framework of life cycle assessment. Risk Anal. 37 (7), 1358–1374. https://doi.org/10.1111/risa.12703.
Galic, N., Schmolke, A., Forbes, V., Baveco, H., van den Brink, P.J., 2012. The role of ecological models in linking ecological risk assessment to ecosystem services in agroecosystems. Sci. Total Environ. 415, 93–100. https://doi.org/10.1016/j.scitotenv.2011.05.065.
Galic, N., Salice, C.J., Birnir, B., Bruins, R.J.F., Ducrot, V., Jager, H.I., Kanarek, A., Pastorok, R., Rebarber, R., Thorbek, P., Forbes, V.E., 2019. Predicting impacts of chemicals from organisms to ecosystem service delivery: a case study of insecticide impacts on a freshwater lake. Sci. Total Environ. 682, 426–436. https://doi.org/10.1016/j.scitotenv.2019.05.187.
Garcia-Alonso, M., Raybould, A., 2014. Protection goals in environmental risk assessment: a practical approach. Transgenic Res. 23 (6), 945–956. https://doi.org/10.1007/s11248-013-9760-1.
Genovese, A., Morris, J., Piccolo, C., Koh, S.C.L., 2017. Assessing redundancies in environmental performance measures for supply chains. J. Clean. Prod. 167, 1290–1302. https://doi.org/10.1016/j.jclepro.2017.05.186.
Gilioli, G., Schrader, G., Carlsson, N., van Donk, E., van Leeuwen, C.H.A., Martín, P.R., Pasquali, S., Vilà, M., Vos, S., 2017. Environmental risk assessment for invasive alien species: a case study of apple snails affecting ecosystem services in Europe. Environ. Impact Assess. Rev. 65, 1–11. https://doi.org/10.1016/j.eiar.2017.03.008.
Gómez-Baggethun, E., de Groot, R., Lomas, P.L., Montes, C., 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecol. Econ. 69 (6), 1209–1218. https://doi.org/10.1016/j.ecolecon.2009.11.007.
Goussen, B., Price, O.R., Rendal, C., Ashauer, R., 2016. Integrated presentation of ecological risk from multiple stressors. Sci. Rep. 6 (September), 1–11. https://doi.org/10.1038/srep36004.
Gregr, E.J., Christensen, V., Nichol, L., Martone, R.G., Markel, R.W., Watson, J.C., Harley, C.D.G., Pakhomov, E.A., Shurin, J.B., Chan, K.M.A., 2020. Cascading social-ecological costs and benefits triggered by a recovering keystone predator. Science 368 (6496), 1243–1247. https://doi.org/10.1126/science.aay5342.

Grêt-Regamey, A., Sirén, E., Brunner, S.H., Weibel, B., 2017. Review of decision support tools to operationalize the ecosystem services concept. Ecosyst. Serv. 26, 306–315. https://doi.org/10.1016/j.ecoser.2016.10.012 (April 2016).

Guerry, A.D., Ruckelshaus, M.H., Arkema, K.K., Bernhardt, J.R., Guannel, G., Kim, C.K., Marsik, M., Papenfus, M., Toft, J.E., Verutes, G., Wood, S.A., Beck, M., Chan, F., Chan, K.M.A., Gelfenbaum, G., Gold, B.D., Halpern, B.S., Labiosa, W.B., Lester, S.E., Spencer, J., 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 8 (1–2), 107–121. https://doi.org/10.1080/21513732.2011.647835.

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: past, present, and future. Environ. Sci. Technol. 45 (1), 90–96. https://doi.org/10.1021/es101316v.

Guinée, J.B., Heijungs, R., Vijver, M.G., Peijnenburg, W.J.G.M., 2017. Setting the stage for debating the roles of risk assessment and life-cycle assessment of engineered nanomaterials. Nat. Nanotechnol. 12 (8), 727–733. https://doi.org/10.1038/NNANO.2017.135.

Gust, K.A., Collier, Z.A., Mayo, M.L., Stanley, J.K., Gong, P., Chappell, M.A., 2016. Limitations of toxicity characterization in life cycle assessment: can adverse outcome pathways provide a new foundation? Integr. Environ. Assess. Manag. 12 (3), 580–590. https://doi.org/10.1002/ieam.1708.

Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D.G., Frid, C.L.J. (Eds.), Ecosystem Ecology: A New Synthesis. Cambridge University Press, pp. 110–139 https://doi.org/10.1017/CBO9780511750458.007.

Halpern, B.S., Frazier, M., Afflerbach, J., Lowndes, J.S., Micheli, F., O'Hara, C., Scarborough, C., Selkoe, K.A., 2019. Recent pace of change in human impact on the world's ocean. Sci. Rep. 9 (1), 1–8. https://doi.org/10.1038/s41598-019-47201-9.

Harder, R., Holmquist, H., Molander, S., Svanström, M., Peters, G.M., 2015. Review of environmental assessment case studies blending elements of risk assessment and life cycle assessment. Environ. Sci. Technol. 49 (22), 13083–13093. https://doi.org/10.1021/acs.est.5b03302.

Harder, R., Peters, G.M., Molander, S., Ashbolt, N.J., Svanström, M., 2016. Including pathogen risk in life cycle assessment: the effect of modelling choices in the context of sewage sludge management. Int. J. Life Cycle Assess. 21 (1), 60–69. https://doi.org/10.1007/s11367-015-0996-2.

Harrison, P.A., Dunford, R., Barton, D.N., Kelemen, E., Martín-López, B., Norton, L., Termansen, M., Saarikoski, H., Hendriks, K., Gómez-Baggethun, E., Czúcz, B., García-Llorente, M., Howard, D., Jacobs, S., Karlsen, M., Kopperoinen, L., Madsen, A., Rusch, G., van Eupen, M., Zulian, G., 2018. Selecting methods for ecosystem service assessment: a decision tree approach. Ecosyst. Serv. 29, 481–498. https://doi.org/10.1016/j.ecoser.2017.09.016.

Hauck, J., Schweppe-Kraft, B., Albert, C., Görg, C., Jax, K., Jensen, R., Fürst, C., Maes, J., Ring, I., Hönigová, I., Burkhard, B., Mehring, M., Tiefenbach, M., Grunewald, K., Schwarzer, M., Meurer, J., Sommerhäuser, M., Priess, J.A., Schmidt, J., Grêt-Regamey, A., 2013. The promise of the ecosystem services concept for planning and decision-making. Gaia 22 (4), 232–236. https://doi.org/10.14512/gaia.22.4.6.

Häyhä, T., Franzese, P.P., 2014. Ecosystem services assessment: a review under an ecological-economic and systems perspective. Ecol. Model. 289, 124–132. https://doi.org/10.1016/j.ecolmodel.2014.07.002.

Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol. Econ. 57 (2), 209–228. https://doi.org/10.1016/j.ecolecon.2005.04.005.

Heink, U., Jax, K., 2019. Going upstream - how the purpose of a conceptual framework for ecosystem services determines its structure. Ecol. Econ. 156, 264–271. https://doi.org/10.1016/j.ecolecon.2018.10.009 (July 2018).

Herva, M., Franco-Uría, A., Carrasco, E.F., Roca, E., 2012. Application of fuzzy logic for the integration of environmental criteria in ecodesign. Expert Syst. Appl. 39 (4), 4427–4431. https://doi.org/10.1016/j.eswa.2011.09.148.

Herva, M., Roca, E., 2013. Review of combined approaches and multi-criteria analysis for corporate environmental evaluation. J. Clean. Prod. 39, 355–371. https://doi.org/10.1016/j.jclepro.2012.07.058.

Holsman, K., Samhouri, J., Cook, G., Hazen, E., Olsen, E., Dillard, M., Kasperski, S., Gaichas, S., Kelble, C.R., Fogarty, M., Andrews, K., 2017. An ecosystem- based approach to marine risk assessment. Ecosyst. Health Sustain. 3 (1). https://doi.org/10.1002/ehs2.1256.

Hou, D., Qi, S., Zhao, B., Rigby, M., O'Connor, D., 2017. Incorporating life cycle assessment with health risk assessment to select the 'greenest" cleanup level for pb contaminated soil'. J. Clean. Prod. 162, 1157–1168. https://doi.org/10.1016/j.jclepro.2017.06.135.

Hu, M., 2018. Dynamic life cycle assessment integrating value choice and temporal factors—a case study of an elementary school. Energ. Buildings 158, 1087–1096. https://doi.org/10.1016/j.enbuild.2017.10.043.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J. Life Cycle Assess. 22 (2), 138–147. https://doi.org/10.1007/s11367-016-1246-y.

ISO, 2006. Environmental management - life cycle assessment - principles and framework. The International Organization for Standardization. https://www.iso.org/standard/37456.html.

Jeswani, H.K., Hellweg, S., Azapagic, A., 2018. Accounting for land use, biodiversity and ecosystem services in life cycle assessment: impacts of breakfast cereals. Sci. Total Environ. 645, 51–59. https://doi.org/10.1016/j.scitotenv.2018.07.088.

Kalbar, P.P., Birkved, M., Nygaard, S.E., Hauschild, M., 2017. Weighting and aggregation in life cycle assessment: do present aggregated single scores provide correct decision support? J. Ind. Ecol. 21 (6), 1591–1600. https://doi.org/10.1111/jiec.12520.

Kang, P., Chen, W., Hou, Y., Li, Y., 2018. Linking ecosystem services and ecosystem health to ecological risk assessment: a case study of the Beijing-Tianjin-Hebei urban agglomeration. Sci. Total Environ. 636, 1442–1454. https://doi.org/10.1016/j.scitotenv.2018.04.427.

Kobayashi, Y., Peters, G.M., Ashbolt, N.J., Heimersson, S., Svanström, M., Khan, S.J., 2015a. Global and local health burden trade-off through the hybridisation of quantitative microbial risk assessment and life cycle assessment to aid water management. Water Res. 79, 26–38. https://doi.org/10.1016/j.watres.2015.03.015.

Kobayashi, Y., Peters, G.M., Khan, S.J., 2015b. Towards more holistic environmental impact assessment: hybridisation of life cycle assessment and quantitative risk assessment. Procedia CIRP 29, 378–383. https://doi.org/10.1016/j.procir.2015.01.064.

Koellner, T., Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Canals, L.M., Saad, R., Souza, D.M., Müller-Wenk, R., 2013. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int. J. Life Cycle Assess. 18 (6), 1188–1202. https://doi.org/10.1007/s11367-013-0579-z.

Kuczenski, B., Geyer, R., Boughton, B., 2011. Tracking toxicants: toward a life cycle aware risk assessment. Environ. Sci. Technol. 45 (1), 45–50. https://doi.org/10.1021/es101467z.

La Notte, A., D'Amato, D., Mäkinen, H., Paracchini, M.L., Liquete, C., Egoh, B., Geneletti, D., Crossman, N.D., 2017. Ecosystem services classification: a systems ecology perspective of the cascade framework. Ecol. Indic. 74, 392–402. https://doi.org/10.1016/j.ecolind.2016.11.030.

Lee, K.H., Noh, J., Khim, J.S., 2020. The blue economy and the United Nations' sustainable development goals: challenges and opportunities. Environ. Int. 137 (January), 105528. https://doi.org/10.1016/j.envint.2020.105528.

Lemming, G., Chambon, J.C., Binning, P.J., Bjerg, P.L., 2012. Is there an environmental benefit from remediation of a contaminated site? Combined assessments of the risk reduction and life cycle impact of remediation. J. Environ. Manag. 112, 392–403. https://doi.org/10.1016/j.jenvman.2012.08.002.

Lentz, T.J., Dotson, G.S., Williams, P.R.D., Maier, A., Gadagbui, B., Pandalai, S.P., Lamba, A., Hearl, F., Mumtaz, M., 2015. Aggregate exposure and cumulative risk assessment-integrating occupational and non-occupational risk factors. J. Occup. Environ. Hyg. 12, S112–S126. https://doi.org/10.1080/15459624.2015.1060326.

Lin, X., Yu, S., Ma, H., 2018. Integrative application of life cycle assessment and risk assessment to environmental impacts of anthropogenic pollutants at a watershed scale. Bull. Environ. Contam. Toxicol. 100 (1), 41–48. https://doi.org/10.1007/s00128-017-2257-5.

Linkov, I., Seager, T.P., 2011. Coupling multi-criteria decision analysis, life-cycle assessment, and risk assessment for emerging threats. Environ. Sci. Technol. 45 (12), 5068–5074. https://doi.org/10.1021/es100959q.

Linkov, I., Trump, B.D., Wender, B.A., Seager, T.P., Kennedy, A.J., Keisler, J.M., 2017. Integrate life-cycle assessment and risk analysis results, not methods. Nat. Nanotechnol. 12 (8), 740–743. https://doi.org/10.1038/nnano.2017.152.

Liu, K.F.R., Ko, C.Y., Fan, C., Chen, C.W., 2012. Combining risk assessment, life cycle assessment, and multi-criteria decision analysis to estimate environmental aspects in environmental management system. Int. J. Life Cycle Assess. 17 (7), 845–862. https://doi.org/10.1007/s11367-012-0407-x.

Liu, W., Yan, Y., Wang, D., Ma, W., 2017. Integrate carbon dynamics models for assessing the impact of land use intervention on carbon sequestration ecosystem service. Ecol. Indic. 91, 268–277. https://doi.org/10.1016/j.ecolind.2018.03.087 (November 2017).

Liu, X., Ziv, G., Bakshi, B.R., 2018a. Ecosystem services in life cycle assessment - part 1: a computational framework. J. Clean. Prod. 197, 314–322. https://doi.org/10.1016/j.jclepro.2018.06.164.

Liu, X., Ziv, G., Bakshi, B.R., 2018b. Ecosystem services in life cycle assessment - part 2: adaptations to regional and serviceshed information. J. Clean. Prod. 197, 772–780. https://doi.org/10.1016/j.jclepro.2018.05.283.

Liu, X., Bakshi, B.R., 2019. Ecosystem Services in Life Cycle Assessment while encouraging techno-ecological synergies. J. Ind. Ecol. 23 (2), 347–360. https://doi.org/10.1111/jiec.12755.

Liu, X., Charles, M., Bakshi, B.R., 2019. Including ecosystem services in life cycle assessment: methodology and application to urban farms. Procedia CIRP 80, 287–291. https://doi.org/10.1016/j.procir.2018.12.004.

Liu, X., Bakshi, B.R., Rugani, B., de Souza, D.M., Bare, J., Johnston, J.M., Laurent, A., Verones, F., 2020. Quantification and valuation of ecosystem services in life cycle assessment: application of the cascade framework to rice farming systems. Sci. Total Environ. 747, 141278. https://doi.org/10.1016/j.scitotenv.2020.141278.

Lueddeckens, S., Saling, P., Guenther, E., 2020. Temporal issues in life cycle assessment—a systematic review. Int. J. Life Cycle Assess. 25 (8), 1385–1401. https://doi.org/10.1007/s11367-020-01757-1.

Maia de Souza, D., Lopes, G.R., Hansson, J., Hansen, K., 2018. Ecosystem services in life cycle assessment: a synthesis of knowledge and recommendations for biofuels. Ecosyst. Serv. 30, 200–210. https://doi.org/10.1016/j.ecoser.2018.02.014.

Maltby, L., van den Brink, P.J., Faber, J.H., Marshall, S., 2018. Advantages and challenges associated with implementing an ecosystem services approach to ecological risk assessment for chemicals. Sci. Total Environ. 621, 1342–1351. https://doi.org/10.1016/j.scitotenv.2017.10.094.

Manfredi, S., Allacker, K., Chomkhamsri, K., Pelletier, N., Maia de Souza, D., 2012. Product environmental footprint (PEF) guide. Retrieved April 12, 2021 fromEuropean Commision Joint Research Centre. http://ec.europa.eu/environment/eussd/pdf/footprint/PEF methodologyfinaldraft.pdf.

MEA, 2005. Ecosystems and human well-being: synthesis. Retrieved February 3, 2021 fromIsland Press. http://www.millenniumassessment.org/en/Synthesis.aspx.

Meyerson, L.A., Baron, J., Melillo, J.M., Naiman, R.J., O'Malley, R.I., Orians, G., Palmer, M.A., Pfaff, A.S.P., Running, S.W., Sala, O.E., 2005. Aggregate measures of ecosystem services: can we take the pulse of nature? Front. Ecol. Environ. 3 (1 SPEC. ISS.), 56–59. https://doi.org/10.1890/1540-9295(2005)003[0056:amoesc]2.0.co;2.

Milazzo, M.F., Spina, F., 2015. The use of the risk assessment in the life cycle assessment framework: human health impacts of a soy-biodiesel production. Manag. Environ. Qual. 26 (3), 389–406. https://doi.org/10.1108/MEQ-03-2014-0045.

Morales, M.A.M., Bravo, R.D.M., Baril, C.F., Hernández, M.F., Delgadillo, S.A.M., 2020. An integrated approach to determining the capacity of ecosystems to supply ecosystem services into life cycle assessment for a carbon capture system. Appl. Sci. 10 (2). https://doi.org/10.3390/app10020622.

Muazu, R.I., Rothman, R., Maltby, L., 2021. Integrating life cycle assessment and environmental risk assessment: a critical review. J. Clean. Prod. 293, 126120. https://doi.org/10.1016/j.jclepro.2021.126120.

Müller, N., de Zwart, D., Hauschild, M., Kijko, G., Fantke, P., 2017. Exploring REACH as a potential data source for characterizing ecotoxicity in life cycle assessment. Environ. Toxicol. Chem. 36 (2), 492–500. https://doi.org/10.1002/etc.3542.

Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. Int. J. Life Cycle Assess. 15 (2), 172–182. https://doi.org/10.1007/s11367-009-0144-y.

Munns, W.R., Rea, A.W., Suter, G.W., Martin, L., Blake-Hedges, L., Crk, T., Davis, C., Ferreira, G., Jordan, S., Mahoney, M., Barron, M.G., 2016. Ecosystem services as assessment endpoints for ecological risk assessment. Integr. Environ. Assess. Manag. 12 (3), 522–528. https://doi.org/10.1002/ieam.1707.

Munns, W.R., Poulsen, V., Gala, W.R., Marshall, S.J., Rea, A.W., Sorensen, M.T., von Stackelberg, K., 2017. Ecosystem services in risk assessment and management. Integr. Environ. Assess. Manag. 13 (1), 62–73. https://doi.org/10.1002/ieam.1835.

Muñoz, I., Flury, K., Jungbluth, N., Rigarlsford, G., Canals, L.M., King, H., 2014. Life cycle assessment of bio-based ethanol produced from different agricultural feedstocks. Int. J. Life Cycle Assess. 19 (1), 109–119. https://doi.org/10.1007/s11367-013-0613-1.

Nienstedt, K.M., Brock, T.C.M., van Wensem, J., Montforts, M., Hart, A., Aagaard, A., Alix, A., Boesten, J., Bopp, S.K., Brown, C., Capri, E., Forbes, V., Köpp, H., Liess, M., Luttik, R., Maltby, L., Sousa, J.P., Streissl, F., Hardy, A.R., 2012. Development of a framework based on an ecosystem services approach for deriving specific protection goals for environmental risk assessment of pesticides. Sci. Total Environ. 415, 31–38. https://doi.org/10.1016/j.scitotenv.2011.05.057.

Núñez, M., Antón, A., Muñoz, P., Rieradevall, J., 2013. Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain. Int. J. Life Cycle Assess. 18 (4), 755–767. https://doi.org/10.1007/s11367-012-0525-5.

Othoniel, B., Rugani, B., Heijungs, R., Benetto, E., Withagen, C., 2016. Assessment of life cycle impacts on ecosystem services: promise, problems, and prospects. Environ. Sci. Technol. 50 (3), 1077–1092. https://doi.org/10.1021/acs.est.5b03706.

Othoniel, B., Rugani, B., Heijungs, R., Beyer, M., Machwitz, M., Post, P., 2019. An improved life cycle impact assessment principle for assessing the impact of land use on ecosystem services. Sci. Total Environ. 693, 133374. https://doi.org/10.1016/j.scitotenv.2019.07.180.

Pártl, A., Vačkář, D., Loučková, B., Lorencová, E.K., 2017. A spatial analysis of integrated risk: vulnerability of ecosystem services provisioning to different hazards in the Czech Republic. Nat. Hazards 89 (3), 1185–1204. https://doi.org/10.1007/s11069-017-3015-z.

Pavan, A.L.R., Ometto, A.R., 2018. Ecosystem Services in Life Cycle Assessment: a novel conceptual framework for soil. Sci. Total Environ. 643, 1337–1347. https://doi.org/10.1016/j.scitotenv.2018.06.191.

Pizzol, M., Christensen, P., Schmidt, J., Thomsen, M., 2011. Eco-toxicological impact of "metals" on the aquatic and terrestrial ecosystem: a comparison between eight different methodologies for life cycle impact assessment (LCIA). J. Clean. Prod. 19 (6–7), 687–698. https://doi.org/10.1016/j.jclepro.2010.12.008.

Potschin-Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., Schleyer, C., 2018. Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. Ecosyst. Serv. 29, 428–440. https://doi.org/10.1016/j.ecoser.2017.05.015.

Prado, V., Cinelli, M., Ter Haar, S.F., Ravikumar, D., Heijungs, R., Guinée, J., Seager, T.P., 2020. Sensitivity to weighting in life cycle impact assessment (LCIA). Int. J. Life Cycle Assess. 25, 2393–2406. https://doi.org/10.1007/s11367-019-01718-3.

Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation. Int. J. Life Cycle Assess. 13 (5), 374–388. https://doi.org/10.1007/s11367-008-0009-9.

Ribera, G., Clarens, F., Martínez-Lladó, X., Jubany, I., Martí, V., Rovira, M., 2014. Life cycle and human health risk assessments as tools for decision making in the design and implementation of nanofiltration in drinking water treatment plants. Sci. Total Environ. 466–467, 377–386. https://doi.org/10.1016/j.scitotenv.2013.06.085.

Romeis, J., Hellmich, R.L., Candolfi, M.P., Carstens, K., de Schrijver, A., Gatehouse, A.M.R., Herman, R.A., Huesing, J.E., McLean, M.A., Raybould, A., Shelton, A.M., Waggoner, A., 2011. Recommendations for the design of laboratory studies on non-target arthropods for risk assessment of genetically engineered plants. Transgenic Res. 20 (1), 1–22. https://doi.org/10.1007/s11248-010-9446-x.

Rosenthal, A., Verutes, G., McKenzie, E., Arkema, K.K., Bhagabati, N., Bremer, L.L., Olwero, N., Vogl, A.L., 2015. Process matters: a framework for conducting decision-relevant assessments of ecosystem services. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 11 (3), 190–204. https://doi.org/10.1080/21513732.2014.966149.

Rugani, B., Benetto, E., Arbault, D., Tiruta-Barna, L., 2013. Emergy-based mid-point valuation of ecosystem goods and services for life cycle impact assessment. Rev. Metall./Cah. d'Inf. Tech. 110 (4), 249–264. https://doi.org/10.1051/metal/2013067.

Rugani, B., Maia de Souza, D., Weidema, B.P., Bare, J., Bakshi, B., Grann, B., Johnston, J.M., Pavan, A.L.R., Liu, X., Laurent, A., Verones, F., 2019. Towards integrating the ecosystem services cascade framework within the life cycle assessment (LCA) cause-effect methodology. Sci. Total Environ. 690, 1284–1298. https://doi.org/10.1016/j.scitotenv.2019.07.023.

Saad, R., Koellner, T., Margni, M., 2013. Land use impacts on freshwater regulation, erosion regulation, and water purification: a spatial approach for a global scale level. Int. J. Life Cycle Assess. 18 (6), 1253–1264. https://doi.org/10.1007/s11367-013-0577-1.

Sala, S., Goralczyk, M., 2013. Chemical footprint: a methodological framework for bridging life cycle assessment and planetary boundaries for chemical pollution. Integr. Environ. Assess. Manag. 9 (4), 623–632. https://doi.org/10.1002/ieam.1471.

Sample, J.E., Baber, I., Badger, R., 2016. A spatially distributed risk screening tool to assess climate and land use change impacts on water-related ecosystem services. Environ. Model. Softw. 83, 12–26. https://doi.org/10.1016/j.envsoft.2016.05.011.

Schaubroeck, T., Alvarenga, R.A.F., Verheyen, K., Muys, B., Dewulf, J., 2013. Quantifying the environmental impact of an integrated human/industrial- natural system using life cycle assessment; a case study on a forest and wood processing chain. Environ. Sci. Technol. 47 (23), 13578–13586. https://doi.org/10.1021/es4046633.

Schaubroeck, T., Rugani, B., 2017. A revision of what life cycle sustainability assessment should entail: towards modeling the net impact on human well-being. J. Ind. Ecol. 21 (6), 1464–1477. https://doi.org/10.1111/jiec.12653.

Schaubroeck, T., 2018. Towards a general sustainability assessment of human/industrial and nature-based solutions. Sustain. Sci. 13 (4), 1185–1191. https://doi.org/10.1007/s11625-018-0559-0.

Shatkin, J.A., Kim, B., 2015. Cellulose nanomaterials: life cycle risk assessment, and environmental health and safety roadmap. Environ. Sci. Nano 2 (5), 477–499. https://doi.org/10.1039/c5en00059a.

Simmons, D.C., Dauwe, R., Gowland, R., Gyenes, Z., King, A.G., Riedstra, D., Schneiderbauer, S., 2017. Qualitative and quantitative approaches to risk assessment. In: Poljansek, K., Ferrer, M. Marin, Groeve, T. De, Clark, I. (Eds.), Science for Disaster Risk Management 2017: Knowing Better and Losing Less. Publications Office of the European Union, pp. 44–130 https://doi.org/10.2788/842809.

Small, N., Munday, M., Durance, I., 2017. The challenge of valuing ecosystem services that have no material benefits. Glob. Environ. Chang. 44, 57–67. https://doi.org/10.1016/j.gloenvcha.2017.03.005.

Stelzenmüller, V., Coll, M., Mazaris, A.D., Giakoumi, S., Katsanevakis, S., Portman, M.E., Degen, R., Mackelworth, P., Gimpel, A., Albano, P.G., Almpanidou, V., Claudet, J., Essl, F., Evagelopoulos, T., Heymans, J.J., Genov, T., Kark, S., Micheli, F., Grazia, M., Rumes, B., 2018. A risk-based approach to cumulative effect assessments for marine management. Sci. Total Environ. 612, 1132–1140. https://doi.org/10.1016/j.scitotenv.2017.08.289.

Sullivan, S., 2009. Green capitalism, and the cultural poverty of constructing nature as service provider. Radic. Anthropol. 3, 18–27. https://core.ac.uk/download/pdf/9630316.pdf.

Sullivan, S., Hannis, M., 2017. "Mathematics maybe, but not money": on balance sheets, numbers and nature in ecological accounting. Account. Audit. Account. J. 30 (7), 1459–1480. https://doi.org/10.1108/AAAJ-06-2017-2963.

Syberg, K., Backhaus, T., Banta, G., Bruce, P., Gustavsson, M., Munns, W.R., Rämö, R., Selck, H., Gunnarsson, J.S., 2017. Toward a conceptual approach for assessing risks from chemical mixtures and other stressors to coastal ecosystem services. Integr. Environ. Assess. Manag. 13 (2), 376–386. https://doi.org/10.1002/ieam.1849.

Taelman, S.E., Tonini, D., Wandl, A., Dewulf, J., 2018. A holistic sustainability framework for waste management in european cities: concept development. Sustainability (Switzerland) 10 (7). https://doi.org/10.3390/su10072184.

Taelman, S., Sanjuan-Delmás, D., Tonini, D., Dewulf, J., 2020. An operational framework for sustainability assessment including local to global impacts: focus on waste management systems. Resour. Conserv. Recycl. 162 (June), 104964. https://doi.org/10.1016/j.resconrec.2020.104964.

Tian, S., Bilec, M., 2018. Integrating site-specific dispersion modeling into life cycle assessment, with a focus on inhalation risks in chemical production. J. Air Waste Manag. Assoc. 68 (11), 1224–1238. https://doi.org/10.1080/10962247.2018.1496189.

Tsang, M.P., Kikuchi-Uehara, E., Sonnemann, G.W., Aymonier, C., Hirao, M., 2017. Evaluating nanotechnology opportunities and risks through integration of life-cycle and risk assessment. Nat. Nanotechnol. 12 (8), 734–739. https://doi.org/10.1038/nnano.2017.132.

United Nations, 2020. The sustainable development goals report 2020. Retrieved June 3, 2021 fromUnited Nations. https://unstats.un.org/sdgs/report/2020/The-Sustainable-Development-Goals-Report-2020.pdf.

Van der Biest, K., Meire, P., Schellekens, T., D'hondt, B., Bonte, D., Vanagt, T., Ysebaert, T., 2020. Aligning biodiversity conservation and ecosystem services in spatial planning: focus on ecosystem processes. Sci. Total Environ. 712, 136350. https://doi.org/10.1016/j.scitotenv.2019.136350.

van Zelm, R., van der Velde, M., Balkovic, J., Čengić, M., Elshout, P.M.F., Koellner, T., Núñez, M., Obersteiner, M., Schmid, E., Huijbregts, M.A.J., 2018. Spatially explicit life cycle impact assessment for soil erosion from global crop production. Ecosyst. Serv. 30, 220–227. https://doi.org/10.1016/j.ecoser.2017.08.015.

Viglia, S., Nienartowicz, A., Franzese, P.P., 2013. Integrating environmental accounting. Life cycle and ecosystem services assessment. J. Environ. Acc. Manag. 1 (4), 307–319. https://doi.org/10.5890/JEAM.2013.11.001.

von Stackelberg, K.E., 2013. Decision analytic strategies for integrating ecosystem services and risk assessment. Integr. Environ. Assess. Manag. 9 (2), 260–268. https://doi.org/10.1002/ieam.1393.

Walser, T., Juraske, R., Demou, E., Hellweg, S., 2014. Indoor exposure to toluene from printed matter matters: complementary views from life cycle assessment and risk assessment. Environ. Sci. Technol. 48 (1), 689–697. https://doi.org/10.1021/es403804z.

Walser, T., Bourqui, R.M., Studer, C., 2017. Combination of life cycle assessment, risk assessment and human biomonitoring to improve regulatory decisions and policy making for chemicals. Environ. Impact Assess. Rev. 65, 156–163. https://doi.org/10.1016/j.eiar.2017.05.004 (December 2016).

Weyell, P., Kurland, H.D., Hülser, T., Grabow, J., Müller, F.A., Kralisch, D., 2020. Risk and life cycle assessment of nanoparticles for medical applications prepared using safe- and benign-by-design gas-phase syntheses. Green Chem. 22 (3), 814–827. https://doi.org/10.1039/c9gc02436k.

Willaert, T., Garcia-Alegre, A.G., Queiroga, H., Lillebø, A.I., Cunha e Sá, M.A., 2019. Measuring vulnerability of marine and coastal habitats' potential to deliver ecosystem services: complex Atlantic region as case study. Front. Mar. Sci. 6 (MAR), 1–13. https://doi.org/10.3389/fmars.2019.00199.

Woods, J.S., Veltman, K., Huijbregts, M.A.J., Verones, F., Hertwich, E.G., 2016. Towards a meaningful assessment of marine ecological impacts in life cycle assessment (LCA). Environ. Int. 89–90, 48–61. https://doi.org/10.1016/j.envint.2015.12.033.

Wu, J., 2013. Landscape sustainability science: ecosystem services and human well-being in changing landscapes. Landsc. Ecol. 28 (6), 999–1023. https://doi.org/10.1007/s10980-013-9894-9.

Xing, L., Hu, M., Wang, Y., 2020. Integrating ecosystem services value and uncertainty into regional ecological risk assessment: a case study of Hubei Province, Central China. Sci. Total Environ. 740, 140126. https://doi.org/10.1016/j.scitotenv.2020.140126.

Xu, X., Yang, G., Tan, Y., Zhuang, Q., Li, H., Wan, R., Su, W., Zhang, J., 2016. Ecological risk assessment of ecosystem services in the taihu Lake Basin of China from 1985 to 2020. Sci. Total Environ. 554–555 (2016), 7–16. https://doi.org/10.1016/j.scitotenv.2016.02.120.

Xue, J.F., Liu, S.L., Chen, Z.Du., Chen, F., Lal, R., Tang, H.M., Zhang, H.L., 2014. Assessment of carbon sustainability under different tillage systems in a double rice cropping system in southern China. Int. J. Life Cycle Assess. 19 (9), 1581–1592. https://doi.org/10.1007/s11367-014-0768-4.

Yang, Q., Liu, G., Casazza, M., Campbell, E.T., Giannetti, B.F., Brown, M.T., 2018. Development of a new framework for non-monetary accounting on ecosystem services valuation. Ecosyst. Serv. 34 (October), 37–54. https://doi.org/10.1016/j.ecoser.2018.09.006.

Zhang, Y.I., Singh, S., Bakshi, B.R., 2010a. Accounting for ecosystem sewices in life cycle assessment part I: a critical review. Environ. Sci. Technol. 44 (7), 2232–2242. https://doi.org/10.1021/es9021156.

Zhang, Y.I., Anil, B., Bakshi, B.R., 2010b. Accounting for ecosystem services in life cycle assessment part II: toward an ecologically based LCA. Environ. Sci. Technol. 44 (7), 2624–2631. https://doi.org/10.1021/es900548a.

Zhang, H., Feng, J., Zhang, Z., Liu, K., Gao, X., Wang, Z., 2020. Regional spatial management based on supply–demand risk of ecosystem services—a case study of the Fenghe River watershed. Int. J. Environ. Res. Public Health 17 (11), 1–25. https://doi.org/10.3390/ijerph17114112.

Zhao, Z., Zhang, T., 2013. Integration of ecosystem services into ecological risk assessment for implementation in ecosystem-based river management: a case study of the Yellow River, China. Hum. Ecol. Risk. Assess. 19 (1), 80–97. https://doi.org/10.1080/10807039.2012.683744.