REGIONAL TOPICS FROM EUROPE

Integrating ecosystem services and life cycle assessment: a framework accounting for local and global (socio-)environmental impacts

Sue Ellen Taelman[1] · Laura V. De Luca Peña[1] · Nils Préat[1] · Till M. Bachmann[2] · Katrien Van der Biest[3] · Joachim Maes[4] · Jo Dewulf[1]

Abstract

Purpose Human activities put pressure on our natural ecosystems in various ways, such as globally through the spread of emissions or locally through the degradation of species-rich landscapes. However, life cycle assessment (LCA) studies that integrate ecosystem services (ES) are still in the minority because of intrinsic differences in data, modelling, and interpretation. This study aims to overcome these challenges by developing and testing a framework that comprehensively evaluates the (socio-)environmental impacts of human activities.

Methods LCA and ecosystem services assessment (ESA) were integrated in two different ways: (1) both methodologies run in parallel and results are combined, and (2) LCA as a driving method where ES are integrated. Because local ESA studies contain the most accurate information but will not be available for all processes in the value chain, it was necessary to advance the life cycle impact assessment method ReCiPe 2016 including three new midpoint impact categories (terrestrial provision, regulation, and cultural ES) and site-generic CFs based on the Ecosystem Services Valuation Database to account for changes in regulating, cultural and provisioning ES due to land use, for the remaining processes in the value chain. Monetary valuation is used to aggregate at the areas of protection (AoP).

Results and discussion A comprehensive LCA_{+ES}-ESA sustainability assessment framework is developed to account for local and global impacts due to human activities on three AoPs (natural resources, ecosystem quality, and human health and well-being), of which the results are expressed in monetary terms. The framework is able to visualize all benefits and burdens accounted for through the handprint/footprint approach. A simplified terrestrial case study on Scots pinewood shows the applicability of the proposed framework, resulting in a handprint ($€_{2022}$ 9.81E+02) which is four times larger than the footprint ($€_{2022}$ 2.31E+02) for 1 kg of wood produced. Challenges related to the framework such as data availability and database shortcomings (i.e., beyond land use) and ES interrelations are discussed.

Conclusion While classical LCA studies focus more on burdens, this framework can also take into account benefits, such as the provision of ecosystem services (or the value of the functional unit of the study). Although the integration of both LCA and ESA has been increasingly explored recently, until now no framework has been available that can incorporate results from local ESA, site-specific ESA, and classical LCA studies, which is considered highly relevant to decision-making.

Keywords Life cycle assessment · Ecosystem services · Integration · Monetization · (Socio-)environmental impacts · Characterization factors · Pinewood

1 Introduction

With a growing global population, humanity is increasingly using ecosystems and its services for food, medicines, shelter, energy, clean air and water, or protection from natural

disasters. The supply of ecosystem services (ES) is impacted by rising anthropogenic pressures varying from, e.g., local habitat fragmentation or decreased pollination to, e.g., accelerated global climate change due to the continuous release of greenhouse gas emissions. At the same time, to alleviate pressure, there are also efforts to restore nature locally resulting in positive impacts.

A comprehensive quantitative (socio-)environmental impact assessment of both local and global, adverse and

beneficial impacts is, however, not straightforward as there is no established methodology available to tackle those aspects. On the one hand, life cycle assessment (LCA) is a standardized quantitative tool to calculate the potential environmental burden of products (i.e., goods and services) from a lifecycle perspective (ISO 2006), i.e., covering all the stages of a product's life, which is from raw material extraction through materials processing, manufacture, distribution, use, and end of life. Human activities are modelled as the cause of environmental pressures either due to the generation of releases to air, water or soil, waste generation or the extraction of natural resources, and/or the occupation and transformation of land- or seascape. The environmental impacts are modelled through cause-effect chains, whereby this method mainly tackles regional to global effects such as eutrophication, human toxicity, or ozone depletion (Huijbregts et al. 2017). Characterization factors (CFs) are used in LCA to translate inventory flows into these specific impact categories. Furthermore, LCA can also present the impact at the endpoint of a cause-effect chain. Endpoints are the entities that humankind wants to safeguard. These are also called areas of protection (AoPs). The most frequently used AoPs are human health, ecosystem quality, and natural resources. To evaluate the overall environmental adverse and beneficial impacts, the AoPs can be further aggregated into a single score by using aggregation techniques such as multicriteria decision analysis or monetary valuation (Pizzol et al. 2017). However, the traditional LCA methodology fails in many ways to include a quantitative assessment of local effects, adverse or beneficial, that are due to changes in ecosystem functioning caused by product systems and human activities. Basically, these local impacts are reflected in changes in the supply of ES, which affect human well-being. Costanza et al. (2017) defined ES as "the ecological features, functions or processes that contribute directly or indirectly to human well-being: that is, the benefits people derive from functioning ecosystems."

Since the concept of ES was introduced by Costanza et al. (1997), many classifications have been proposed such as the Final Ecosystem Goods and Services Classification System (U.S. EPA 2013), the National Ecosystem Services Classification System (U.S. EPA 2015), The Economics of Ecosystems and Biodiversity (TEEB 2010), and the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2003). However, the typology Common International Classification of Ecosystem Services (CICES), and more in particular CICES (v5.1), is the currently most widely used one. It attempts to make a distinction between services and benefits (Haines-Young and Potschin 2018). The ES are divided into three categories: provisioning, regulating and maintenance, and cultural services. Each of these categories is further sub-divided into "divisions", "groups", and "classes" (Czúcz et al. 2018).

To assess the magnitude and importance of different impacts on ecosystems and the various services and benefits they provide in different locations, it is crucial to use appropriate assessment tools (Rugani et al. 2019). As outlined in De Luca et al. (2022), ecosystem services assessment (ESA) is a methodology designed to improve the understanding of the ecosystems, their biophysical processes and functions and their interactions, how humans value them as services, and how they contribute to human well-being. A widely adopted ES cascade framework was proposed by Haines-Young and Potschin (2010). It provides a conceptual model to guide ES assessment and was also used as a backbone to develop the CICES classification system. Following the rationale of the ES cascade framework, ES are supplied through ecosystem functioning (e.g., denitrification), which is a result of the complex interactions between the biophysical structures and processes of the system (e.g., biota, filtration). When altering ecosystem functions, this leads to changes in the delivery of ES (e.g., regulating ES, mediation of waste). This in turn will change the benefits humans receive (e.g., improved or reduced water quality). While ES are supplied from the natural environment, their valuation is most often done from a socio-economic perspective. To accurately judge the state of the ecosystems (aquatic and terrestrial)—and the ES they supply—it is necessary to gain an understanding of the processes underlying ecosystem functioning and ES supply and quantitatively assess these structures and processes in a spatially resolved way (Van der Biest 2018). However, such assessments are data-intensive, e.g., field data, monitoring, satellite observations, mathematical modelling, and therefore also time-consuming.

To date, ES are insufficiently covered in traditional environmental decision-making tools, such as LCA (Hauschild et al. 2018; Alejandre et al. 2019; Hardaker et al. 2022). There are attempts made to integrate both methodologies, to better, and more holistically, account for sustainability impacts covering multiple geographical scales, and often considering a life cycle perspective. De Luca et al. (2022) reviewed the various integration approaches used to combine LCA and ESA. About 25 papers were selected, leaving out the merely conceptual papers, and the authors focused on integration approaches with proven applicability. Three main categories of ES-LCA integration approaches were identified (Fig. 1).

The first type, named "post-analysis", performs LCA and ESA independently and in parallel without any real integration of the methods or aggregation of the results. The second type is named "integration through combination of results", where LCA and ESA are performed independently and in parallel as well. However, the results are aggregated by introducing an additional quantitative or qualitative step (e.g., multicriteria decision analysis). The integration always occurs at the endpoint of the cause-effect chains. The last
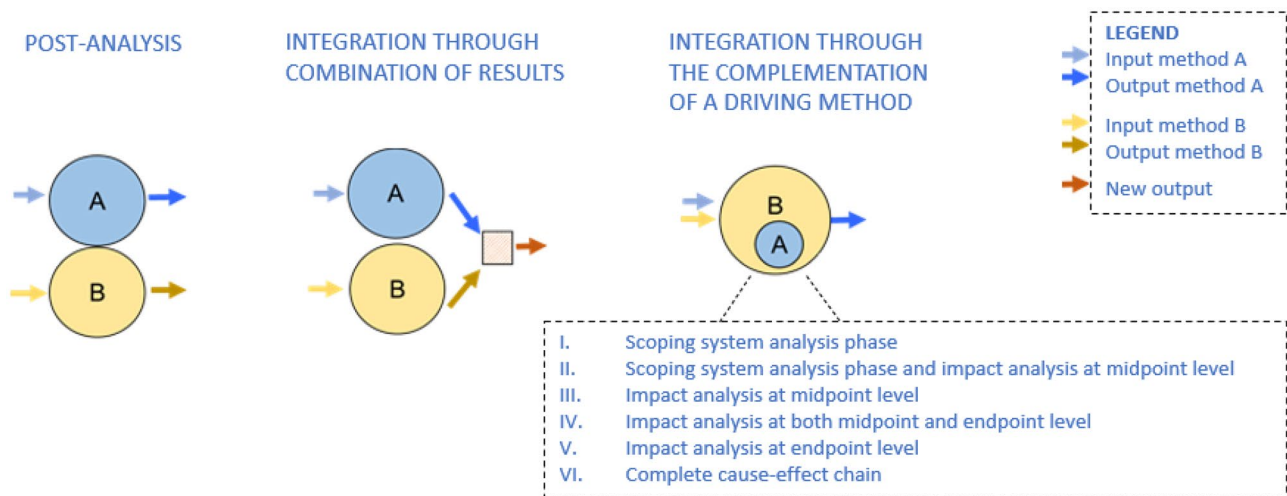
Fig. 1 Classification scheme based on the types of ESA and LCA integration. (1) Post-analysis, (2) integration through the combination of results, (3) integration through the complementation of a driving method (adapted from De Luca et al. 2022)

type is named "driving method", where ESA is integrated into LCA by including ES aspects in the goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and/or results interpretation. LCA is considered the driving method because it provides the main structure of the analysis. Table S1 (Supplementary Information) shows for each of the 25 studies to which integration category they belong. As a general result, it appears that the driving method is by far the most frequently used method for integrating ESA and LCA (22 papers). The way ES are integrated into LCA, however, differs strongly among the papers. Some studies use allocation methods to integrate ES into LCA, e.g., by allocating part of the burdens of the supply chain to regulating ES instead of a full allocation to provisioning ES (e.g., Boone et al. 2019; Bragaglio et al. 2020), a few studies adapt the LCI of agricultural processes with results from land use change models and ES quantification tools to optimize, e.g., CO_2 emissions and nitrate leaching due to changes in land use (e.g., Chaplin-Kramer et al. 2017) but most studies develop new ES midpoint impact categories in LCA (e.g., Blanco et al. 2018; Cao et al. 2015; Arbault et al. 2014). With regard to the latter approach, impacts on ES due to human activity are mostly calculated based on the use of land (occupation and transformation) which is considered the pressure.

According to De Luca et al. (2022), the three main categories of ES-LCA integration approaches have each their limitations (e.g., being time-consuming, data-intensive, risk for double counting, complex) and advantages (e.g., simple, straightforward, informative). The selection of a particular integration approach seems to be depending on the final aim and scope of the study. The choice has bearings on the collection of data, impact assessment calculations, number of

assumptions, system boundary and functional unit (FU) definition, and/or visualization and interpretation of results of a comprehensive study. So far, to the best of our knowledge, no sustainability assessment method combining ESA and LCA was developed that makes use of two or more integration approaches as shown in Fig. 1.

From the review study of De Luca et al. (2022), it became also clear that economic allocation or monetization are strategies that are often used to perform the aggregation, mainly at the endpoint level. According to Arendt et al. (2020), monetization can facilitate the creation of markets for non-marketed goods or services, such as for emission permits or the value of ES, but equally determine the economic value of socio-environmental impacts caused by releases or raw material extraction or adverse changes to ES, all leading to costs to society. A variety of different monetary valuation techniques are used nowadays to account for biophysical flows, non-marketed goods or services, or burdens to the environment and society. Frequently used techniques are the following: the market price approach, revealed preference approach (valuation on a surrogate market), or stated preference approach (valuation on a hypothetical market, i.e., through surveys). Despite the fact that the monetary valuation techniques differ, the advantage is that impacts can be unified and a single score can be derived from the results, which aids interpretation and communication (Vogtländer et al. 2001; Hardaker et al. 2022).

Furthermore, Alvarenga et al. (2020) point out that LCA mainly focusses on adverse impacts related to a product, while there are also potential beneficial impacts. These are however in most cases neither assessed nor visualized (apart from the functional unit). To better account for all beneficial impacts, beyond the functional unit, they developed a

framework to identify and represent both beneficial impacts (handprint) and adverse impacts (footprint) contributing to the results. It distinguishes between a direct and indirect handprint depending on the final beneficiary (intended or unintended users, the latter having no direct benefit of the functional unit) and between an absolute and relative handprint depending on the information required (single product versus comparative information). Human activities can contribute to unintended users positively (e.g., CO_2 sequestration in the value chain and/or in the final product) or negatively (e.g., high CO_2 releases) or to intended users positively (e.g., production of protein-rich food, as FU) or negatively (e.g., the protein-rich food containing also many saturated fats). All of these impacts are ideally captured by an integrated sustainability assessment method. We propose to integrate LCA and ESA to this end. Using this handprint/footprint concept can help in the interpretation of the impact results and support decision-making; however, from the state of the art, it is clear that such an approach has never been applied to unify and monetize results provided by LCA and ESA.

Therefore, the aim of this study is to develop a comprehensive sustainability impact assessment framework that is able to quantify beneficial and adverse, local to global, (socio-)environmental impacts of human activities and related products, from a life cycle perspective. For this purpose, several scientific developments were made, explained in Sect. 2 which addresses the definition of a new set of AoPs (Sect. 2.1), proposes a way to integrate LCA and local ESA studies (Sect. 2.2), and advances LCA by better accounting for ES changes (Sect. 2.3). Section 2.4 combines the results of the latter sections and discusses how to develop the final (socio-)environmental impact assessment framework, while Sect. 2.5 explains a case study on Scots pinewood production, used to test the feasibility of applying the framework. Section 3 ultimately shows the developed framework and provides the results of its application to the case study. Section 4 elaborates on the challenges faced during the development and implementation of the framework and Sect. 5 highlights the main findings of this paper.

2 Material and methods

To develop a comprehensive (socio-)environmental impact assessment framework that accounts for local and global impacts, beneficial and adverse, due to human activities, we propose to integrate LCA and ESA. The review of De Luca et al. (2022) revealed three ways of integrating LCA and ESA, while only two ways were truly quantitative integration options. Those two options are further discussed in this chapter, *both of which are applied in parallel*, i.e., "integration through combination of results" and "integration

through the complementation of a driving method" (cfr. Fig. 1). It is chosen to merge two integration options as they complement each other, as discussed later.

2.1 Defining the AoPs

Before addressing integration methods, the AoPs need to be redefined so as to cover not only the AoPs typically distinguished in LCA, i.e., "natural resources", "ecosystem quality", and "human health" (Verones et al. 2017), but also the impacts that human activities have on the three types of ES: provisioning, regulating, and cultural. We propose to redefine these AoPs for integrating LCA and ESA as follows: "natural resources" (NR) encompass impacts on natural resources from LCA and provisioning ES from ESA; "ecosystem quality" (EQ) includes the impacts on ecosystem quality in LCA and regulating ES in ESA; and "human health and well-being" (HH&WB) covers both human health impact pathways modelled in LCA and cultural ES in ESA (Fig. 2). These AoPs cover effects beyond the environmental impacts, by including social aspects linked to cultural ES and human health. Obviously, impacts should be factored in only once. This means that for land use in particular, impacts should not be taken into account simultaneously through midpoint impact categories related to land use in LCA and through ESA, to avoid double counting. Furthermore, to ease interpretation and communication, monetary valuation is used as an aggregation technique, i.e., the potential burdens (footprint) and benefits (handprint) that arise from the LCA and ESA results are monetized at the level of the AoPs.

2.2 Integration through combination of results (LCA-ESA)

Haines-Young and Potschin (2010) propose an ES cascade framework to connect human-induced pressures to changes in (the monetary value of) ES. The human-induced pressures can be manifold and do not necessarily start from land or sea occupation/transformation. For instance, Schallenberg et al. (2003) describe the impacts of global climate change and associated sea level rise, which is likely to intensify saline intrusions into, e.g., coastal lakes. Changes in salinity affect the ability of zooplankton to osmoregulate, which alters its taxonomic diversity and abundance. Zooplankton being the basis of our food web, this can affect our provisioning services. Performing an ESA study implies identifying relevant ES, gathering local data, and, depending on the ES, running ecosystem functioning models. As ES are quite site-specific, our sustainability assessment framework should be able to absorb information from local ESA studies. Quantitative information on the site-specific impacts on provisioning, regulating, and/or cultural ES needs to be considered in addition to LCA results to comprehensively

Fig. 2 Towards areas of protection (AoP) for LCA-ESA integration, starting from the classical AoPs in LCA and the three categories of ecosystem services (ES)
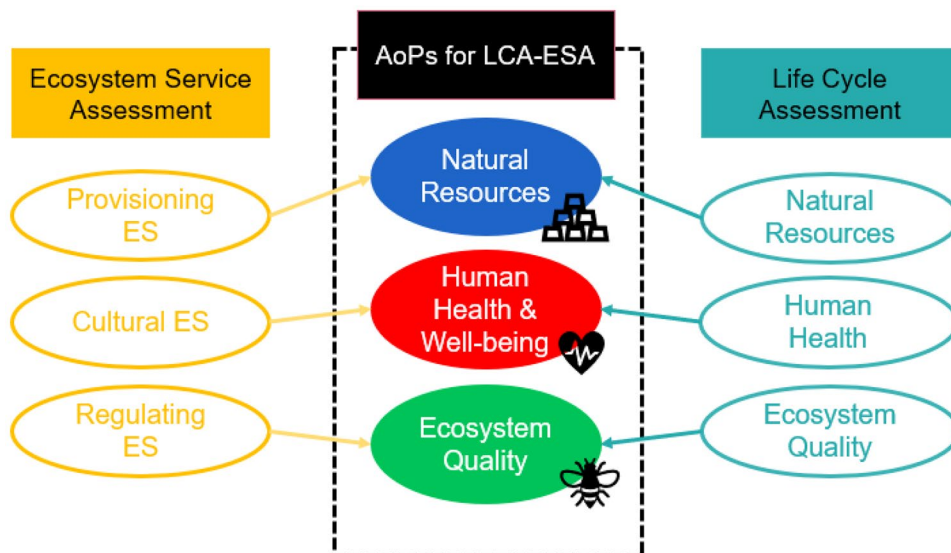
assess adverse and beneficial, local to global impacts associated to human activities and its related products. Based on these considerations, we propose a first approach to integrate both LCA and ESA, further referred to as "LCA-ESA," in which the site-specific impacts on local ES are combined with LCA results at the endpoint level, i.e., aggregation through monetization of the results of ESA and LCA at the level of the previously defined AoPs (see also Sect. 2.4). Besides the obvious advantage of including site-specific information on ES changes (taking place in the marine and/or terrestrial environment) in addition to conventional LCA impact categories, there is also a disadvantage to this way of integration. Acknowledging that ESA is data- and time-intensive, in most cases, the availability of ESA studies will

be insufficient to cover all processes of the product's life cycle as considered in LCA (Fig. 3a). Therefore, we propose to complement this LCA-ESA integration way with a second approach (cfr. driving method), integrating ES into the LCA method. The LCIA method to be used in the LCA-ESA integration method is similar in both approaches (see Sect. 2.3).

2.3 Integration through the complementation of a driving method (LCA$_{+ES}$)

The second integration approach proposed (LCA$_{+ES}$) takes LCA as its driving method, i.e., results of ESA are integrated into LCA. The LCIA method chosen is ReCiPe 2016 (H) version 1.05 (Huijbregts et al. 2017), further referred
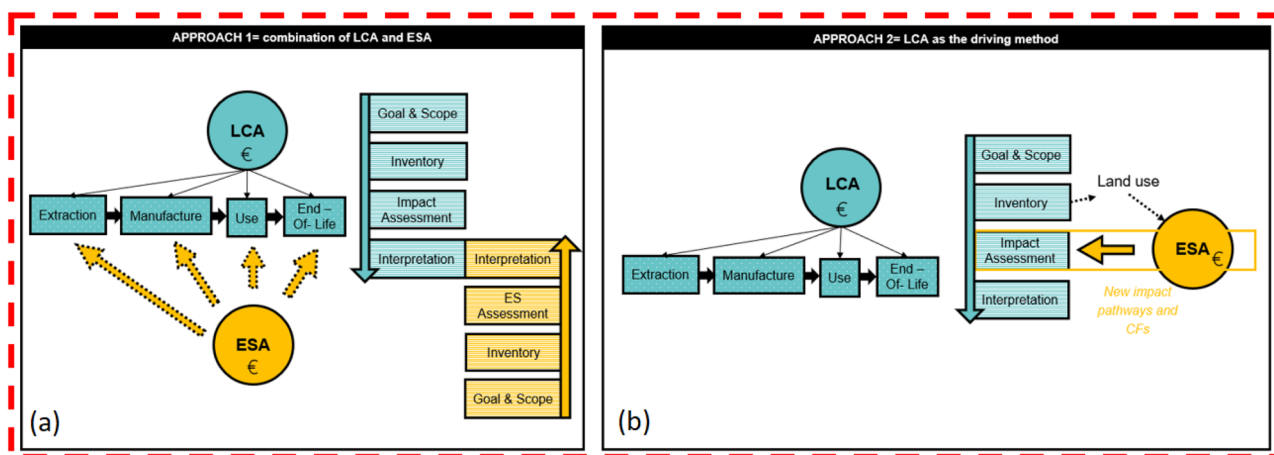

Fig. 3 Visualization of two approaches to integrate ESA and LCA. The first approach (**a**) shows that, in case local ESA studies are performed for particular life cycle processes, they can be combined with LCA at the AoP and interpretation stage. The second approach (**b**)

shows the development of new ESA impact categories in LCA and respective site-generic characterization factors for land use. The two approaches must be combined to have a comprehensive sustainability impact assessment (red dotted line)

to as ReCiPe 2016, containing 17 midpoint or intermediate impact categories and 3 AoPs "resource availability", "natural environment" and "human health", which are directly correlated in this study with the AoPs "natural resources", "ecosystem quality" and "human health and well-being", respectively.

2.3.1 New ES impact categories and CFs linked to land use

The review of De Luca et al. (2022) revealed that many attempts to integrate ES changes and life cycle impacts are based on the effect that land occupation and transformation have on the supply of ES. This can also be observed in the ReCiPe 2016 method, where the original "land use" midpoint impact category accounts for species losses due to land occupation and transformation (Huijbregts et al. 2017). However, addressing only species loss is limiting and does not comprehensively assess impacts on ES. Therefore, it is replaced by three new midpoint impact categories (Fig. 3b): "provisioning services, terrestrial", "regulating services, terrestrial" and "cultural services, terrestrial" (following the section level in CICES v5.1., Haines-Young and Potschin 2018), resulting overall in 19 midpoint impact categories in our LCA$_{+ES}$ methodology.

For the LCA$_{+ES}$ approach, we introduce a set of site-generic CFs (Fig. 3b) to assess changes in the value of the ES due to land use over the entire product's life cycle (i.e., CFs for the new midpoint impact categories). The CFs express quantitatively the potential impacts on ES and are designed in such a way that they can be combined with the life cycle data concerning "land occupation" (m^2 * year) and "land transformation" (m^2). Their development relies on the recommendations from Koellner et al. (2013) for assessing land use impacts on biodiversity and ES. Accordingly, CFs for land occupation are derived relative to a reference state (or benchmark): the land use leading to an impact relative to the potential natural vegetation, i.e., the ecological state that would develop if the area was left without further human influence. CFs for land transformation also rely on a benchmark but include regeneration time for reaching the potential natural vegetation state (Huijbregts et al. 2017). More information on the development of the CFs is provided in Box S1 (Supplementary Information). Including monetization (described next), the CFs for land occupation and for land transformation are finally expressed in € * m^{-2} * year^{-1} and € * m^{-2}, respectively.

2.3.2 The Ecosystem Services Valuation Database as a basis for CF development

Monetary valuation is conducted with the help of global databases reporting ES values per land use class such as The Economics of Ecosystems and Biodiversity (TEEB 2010)

and the Ecosystem Services Valuation Database (ESVD) (de Groot et al. 2020), of which the latter is a follow-up to the TEEB database. The ESVD provides robust and accessible global information on the economic benefits or costs associated with 23 different ES. More specifically, monetary values for provisioning, regulating, and cultural ES associated with 15 "biomes" (they are typically a mix of land cover and land use classes) are provided (de Groot et al. 2021). The database consists of 4,800 datapoints (version December 2020) from multiple studies assessing the value of ES in a specific location. The ESVD mostly contains value estimates from studies located in Europe and, more specifically, in the UK and The Netherlands. Furthermore, the values are associated with a particular biome. Value observations reported in the literature use a wide variety of units; therefore, standardization is needed. The ESVD is built upon a standardization approach which involves five steps: price level, currency, spatial unit, temporal unit, beneficiary unit. Finally, the common unit expressing the flow of ES is $\$_{2020}$ ha^{-1} year^{-1}. The values are aggregated into a summary statistics table, reporting the averages of all standardized values per ES (classified at the higher level according to CICES v5.1) and biome (de Groot et al. 2021). The summary statistics table from the ESVD excludes ES values derived using value transfer or ES values that could not be disaggregated by ESs or biomes, and the highest and lowest 2.5% of value estimates to remove outliers. After applying those criteria, the ES-biome values with less than 5 datapoints were excluded. This substantially reduces the amount of monetary values available per combination of ES and biome. As we consider completeness more important than addressing uncertainty in this way, we adapt the summary table from the ESVD by including ES—biome values having less than 5 datapoints (Table S2, Supplementary Information, italic numbers). For these ES, the highest and lowest datapoints (2.5%) are retained in the evaluation of the reported average ES value because of the very small number of datapoints that was often provided (from 1 to 4).

2.3.3 (Mis)matches between ESVD and ecoinvent and its effect on the CF development

Furthermore, the biomes of the ESVD need to be linked to land occupation and transformation flows available in LCA background databases (containing life cycle data on the value chain of product). A commonly used and comprehensive database is Ecoinvent (v3.8), which covers 42 land use classes. The ESVD biomes "open sea/ocean", "coral reefs", "coastal systems (incl. wetlands)", "desert", "tundra" and "high mountain and polar systems" are not further considered in this study because of their lack of compatibility with Ecoinvent land classes. However, we could assign the other nine biomes from the ESVD to a specific

land use class as considered in Ecoinvent v3.8 (see Table S3, Supplementary Information). Ecoinvent flows being not differentiated between tropical forests and temperate forests, the land classes "tropical forests" and "temperate forests" from the ESVD are merged into a single land use class (i.e., "forests"). The ES value of this land use class is obtained by weighting the ES values of "tropical forests" and "temperate forests" according to their relative area sizes according to World Wildlife Fund (2006). We assign the land use class "unspecified" from Ecoinvent to the average of all terrestrial land use classes. The monetary values from the ESVD are converted from $_{2020}$ to €$_{2022}$ considering the purchasing power parity (PPP) in 2020 and the Harmonized Index of Consumer Prices (HICP) in EU-27 in 2020 and 2021 (Table S4, Supplementary Information). Table 1 summarizes the CFs per type of ES in €$_{2022}$ * ha^{-1} * year^{-1} for the 8 land use classes (or biomes).

The CFs for land occupation relate the ES of a given land use class from Table 1 to the value of the benchmark, i.e., the potential natural vegetation state (cfr. Box S1, Supplementary Information). The weighted average of ES values for the natural land use classes "inland wetland", "tropical forests", "temperate forests", "woodland and shrubland", "grass-/rangeland" and "inland un- or sparsely vegetated" is used as a benchmark (i.e., "average terrestrial natural land classes" in Table 1). The benchmark does not consider "rivers and lakes" as it is unlikely that terrestrial land use will naturally change back to freshwater areas. Furthermore, the benchmark value is a weighted average of forests (40%) and open biomes (60%) (Olson et al. 2001), the latter including the land use classes "inland wetland", "woodland and shrubland", "grass-/rangeland" and "inland un- or sparsely vegetated."

Looking more closely at the way Ecoinvent v3.8 models land transformation, we see that "annual crop" is used

as a reference without being explicit about it. For instance, "Transformation, from wetland, inland (non-use)" means that a land is transformed from wetland to annual crop. Therefore, our CFs for land transformation take the land use class "cultivated areas" as a benchmark. Furthermore, land transformation impacts are only considered for natural land uses and not for anthropogenic land uses. This is because the transformation of any anthropogenic land use class implies the use of varying and often very limited regeneration times compared to natural regeneration times. This approach is identical to the one taken in the ReCiPe 2016 method for land transformation impacts on biodiversity (Huijbregts et al. 2017). For example, the impacts of the transformation of natural grassland to land covered by annual crops on ES are assessed by applying the CFs to the inventory flows, while the transformation between two anthropogenic land uses such as urban area and annual crops cannot be accounted for. The regeneration times for the "forest" land use class and for the other land use classes (being considered as open biomes) are respectively 73.5 and 7.5 years, corresponding to the median values estimated by Curran et al. (2014). A comprehensive list of developed CFs for land occupation and land transformation is provided in Tables S5 and S6 (Supplementary Information).

As a result of this LCA$_{+ES}$ integration method, the newly defined midpoint impact categories include impacts assessed through the site-generic CFs, expressed in €$_{2022}$ * m^{-2} * year^{-1} for occupation and €$_{2022}$ * m^{-2} for transformation. These impact categories are aggregated with the other ReCiPe 2016 midpoint impact categories (e.g., climate change, ozone depletion) at the AoP level. This LCA$_{+ES}$ integration approach is especially relevant if there is no information available on impacts on specific ecosystems (i.e., no ESA studies are available for certain supply chain processes); it

Table 1 Ecosystem services values (provisioning, regulating, and cultural) for different land use classes, original data are retrieved from the Ecosystem Services Valuation Database (de Groot et al. 2021) and adapted from $_{2020}$ to €$_{2022}$. Values per ecosystem services for each biome are shown in Table S3, Supplementary Information

Land use classes (or biomes in ESVD)	Ecosystem services values (€$_{2022}$ * ha^{-1} * year^{-1})		
	Total provisioning	Total regulating	Total cultural
Inland wetlands	5,603	4,305	503
Rivers and lakes	2,445	22,021	5,807
Forests[a]	210	1,957	988
Woodland and shrubland	397	879	177
Grass-/rangeland	470	877	207
Inland un- or sparsely vegetated	8	12	122
Cultivated areas	1,258	3,250	169
Urban green and blue infrastructures	0	9,077	22,814
Average terrestrial natural land use classes[b]	1,524	2,126	391
Average all terrestrial land classes	1,004	2,754	2,613

[a]Includes tropical forests and temperate forests

[b]includes inland wetland, tropical forests, temperate forests, woodland and shrubland, grass-/rangeland, inland un- or sparsely vegetated

does facilitate to complement LCA with ES impacts in a site-generic way.

2.4 AoPs as a metric to combine the LCA-ESA and LCA$_{+ES}$ approaches

The overall (socio-)environmental impact assessment framework (further referred to as LCA$_{+ES}$-ESA) developed in this paper combines the LCA-ESA and LCA$_{+ES}$ integration ways. This approach was chosen because both integration ways have their limitations and advantages, and at the same time complement each other well (see Table S7 in the Supplementary Information for more details).

The final framework comprises impacts related to (1) documented information on ES changes at a site-specific level, often available for a few processes of the value chain, (2) classical midpoint impact categories from the LCIA ReCiPe 2016 method such as climate change and human toxicity (except for land use), and (3) site-generic and life cycle–based provisioning, cultural and regulating ES impacts due to land use of processes in the value chain for which no local ESA studies are available. In the end, all of these are monetized and aggregated at the newly defined AoPs (NR, EQ, and HH&WB) through monetization.

The units of the three AoP are converted to €$_{2022}$ (i.e., value of euros on 31st of December 2021) based on existing monetization factors (adapted with the help of PPP and HICP to respectively convert currencies and account for inflation; Table S4, Supplementary Information). Impacts related to the AoP "natural resources" are expressed in $$_{2013}$ in ReCiPe 2016, representing extra costs for future extraction of fossil and mineral resources due to higher scarcity (Huijbregts et al. 2017). These are converted to €$_{2022}$ according to PPP for USA and EU-27 in 2013 (OECD 2022) and the HICP for the EU-27 between 2013 and 2021 (Eurostat 2022). Impacts on the AoP "human health" are measured in terms of disability-adjusted life years (DALY) in ReCiPe 2016, representing the amount of year equivalents lost or disabled per person due to an accident or a disease. DALYs are valued at 70,000 €$_{2015}$/DALY as a central value

according to de Bruyn et al. (2018). The value is converted to €$_{2022}$ by means of the HICP for the EU-27 between 2015 and 2021 (Eurostat 2022). The monetization of damages on natural ecosystems is more complex compared to damages on human health. The AoP "natural environment" (or ecosystem quality), as developed in ReCiPe 2016, covers impacts on terrestrial, freshwater, and marine species and is expressed in potentially disappeared fraction of species (PDF * year). Because of insufficient data on marine and freshwater biodiversity (e.g., species density and economic valuation of species loss), the monetization at the AoP "ecosystem quality" is only based on the valuation of terrestrial biodiversity. The monetary value of PDF * year has been estimated from the meta-analysis of economic studies related to the valuation of land use change and biodiversity loss performed by Kuik et al. (2008), i.e., having a focus on terrestrial biodiversity in a European context. Within their study, they used the Ecosystem Damage Potential (EDP) per hectare per year to account for PDF (as it is considered to be "basically the same") and they provided a value range, of which de Bruyn et al. (2010) adopted the average value of €$_{2004}$ 0.4706 PDF per m^2 per year. However, the median value is €$_{2004}$ 0.06 PDF per m^2 per year (eight times lower) and is more meaningful in a meta-analysis, according to de Bruyn et al. (2018). To fit with the units of the AoP in ReCiPe 2016 (species * year), they divide the median value by the average terrestrial species density (Box S2, Supplementary Information). The monetization factor, in €$_{2004}$/species * year, is converted to €$_{2022}$/species * year according to European HICP values between 2004 and 2021. The final monetization factors used for each AoP are shown in Table 2.

By monetizing the AoPs, the LCA-ESA and LCA$_{+ES}$ integration approaches can be combined. Furthermore, the handprint-footprint concept (cfr. Alvarenga et al. 2020) is used to distinguish between beneficial and adverse effects. The footprint results for each AoP include the burdens; thus, the LCA results of the product under study complemented with impacts due to a potentially reduced supply of ES (either from local ESA and/or LCA$_{+ES}$). The handprint can

Table 2 Monetization factors for the areas of protection as defined in the ReCiPe 2016 method (Huijbregts et al. 2017)

Areas of protection	Units in ReCiPe	Adapted monetization factors	References
Resource availability[a]	$$_{2013}$	0.793 €$_{2022}$/$$_{2013}$	OECD (2022); Eurostat (2022)
Human health[b]	DALY[a]	76,160 €$_{2022}$/DALY	Eurostat (2022); de Bruyn et al. (2013)
Natural environment[c]	Species * year	5.410*10^6 €$_{2022}$/species * year	Kuik et al. (2008); Goedkoop et al. (2009); de Bruyn et al. (2010)

DALY disability-adjusted life years

[a]Links to the AoP "natural resources"

[b]links to the AoP "human health and well-being"

[c]links to the AoP "ecosystem quality", see Sect. 2.3

consist of the functional unit of the LCA study (e.g., wood production) and possibly avoided products (e.g., benefits of recycling), in combination with a potential increase in the supply of ES (derived from local ESA and/or LCA$_{+ES}$).

2.5 Application to a case study: Scots pinewood production in Belgium

We will illustrate the approaches outlined in Sects. 2.2–2.4 using the production of Scots pine (*Pinus sylvestris* L.) in the forest "De Inslag" in Brasschaat, Belgium, as presented by Schaubroeck et al. (2016). For that part of the case study for which Schaubroeck et al. (2016) provide detailed ESA results, we will use the LCA-ESA integration approach, which is complemented with results from the LCA$_{+ES}$ approach to cover potential ES impacts due to land use in the rest of the system's life cycle.

Schaubroeck et al. (2016) quantified the environmental impact on human health, natural systems, and natural resources in physical units and used an ecosystem service valuation based on monetary valuation techniques to account for ES (including disservices with associated negative monetary values). More specifically, they accounted for wood production as a provisioning service, four regulating services (water purification/pollution via N-removal/emission, enhanced removal of particulate matter (PM), processing of NH_3, and sequestration of CO_2) and two disservices (net loss of fresh water and deposition of NO_x from air which can end up in the water causing net eutrophication). The ES supply for a Scots pine stand from 2010 to 2089 was quantified under different scenarios of environmental change (current, moderate, and severe) and management (low, mid, and high). For more details about the ES and disservices and scenarios, the reader is referred to Schaubroeck et al. (2016).

Part of the results of the study are taken to exemplify the LCA-ESA approach. More specifically, monetized results of the PM$_{2.5}$ removal, CO_2 sequestration, and N pollution/removal (dis)services are assessed for the mid management and moderate climate conditions scenario. Wood production during 80 years under this scenario is the provisioning service considered, and taken into account as the functional unit of the LCA study (see further). Schaubroeck et al. (2016) provided scenarios to test the sensitivity of the results to the inflation rate and the 2% inflation rate was considered for our case study. Detailed calculations regarding the local site-specific ESA results are provided in Box S3 (Supplementary Information).

As the ESA study of Schaubroeck et al. (2016) only covers part of the value chain of wood production, we also need the LCA$_{+ES}$ approach to understand the impact on ES due to, e.g., diesel production and sawing machine production. For this reason, we used the Simapro V9.2.0.2 software and the database Ecoinvent v3.8 (cut-off allocation—unit) to model

1 kg of the product "Cleft timber, measured as dry mass {DE}| softwood forestry, pine, sustainable forest management," which is the FU. First, the ReCiPe 2016 method (all categories except for land use) was applied and the results were aggregated to the respective AoPs. These were monetized with the conversion factors from Table 2. Second, the LCI of 1 kg of pine timber was extracted from the software, and the land occupation and transformation flows were multiplied with the respective newly developed site-generic CFs (already monetized) to provide results at the midpoint level in the provisioning regulating and/or cultural impact categories, and furthermore categorized within the particular AoPs (see Fig. 2 and Table S10, Supplementary Information).

The LCA-ESA approach and the LCA$_{+ES}$ approach are furthermore aggregated following the overall LCA$_{+ES}$-ESA framework (see Sect. 3.1), resulting in monetized values and leading to handprint and footprint results. While positive ES values in the LCA-ESA approach mean positive effects (handprint) and negative ES values a negative effect (footprint), it is the opposite for the LCA$_{+ES}$ approach where LCA midpoint and endpoint impact categories provide positive values for burdens (footprint) and negative values for benefits (handprint). Therefore, a sign reversal is needed for the latter method to be able to clearly visualize the results (footprint on a negative *y*-axis and handprint on a positive *y*-axis), which can avoid any miscommunication or misinterpretation.

3 Results

3.1 The LCA$_{+ES}$-ESA framework

Figure 4 shows the LCA$_{+ES}$-ESA framework, covering the two approaches as previously described in Sects. 2.2 and 2.3. The first approach (LCA-ESA) combines results of both LCA and ESA being applied to the same case study, focusing on a specific human activity, and the results are both monetized and unified and can be combined in this way and presented according to the handprint/footprint concept for the three AoPs NR, EQ, and HH&WB (cfr. Fig. 1). However, in most cases, ESA studies will not be available or performed for all foreground processes (under the scope of the study) and background processes (related to the value chain but covered by LCI databases), but only for a few of them as shown in Fig. 4. In case not a single ESA study is available, the LCA-ESA integration approach cannot be applied. For these reasons, the LCA$_{+ES}$ approach is needed to integrate elements of ESA into LCA so it is able to account for changes in local/regional ES due to all (other) life cycle processes. Figure 4 shows different LCA steps, from identifying the life cycle stages and processes in scope to collecting the inventory and calculating the (socio-)
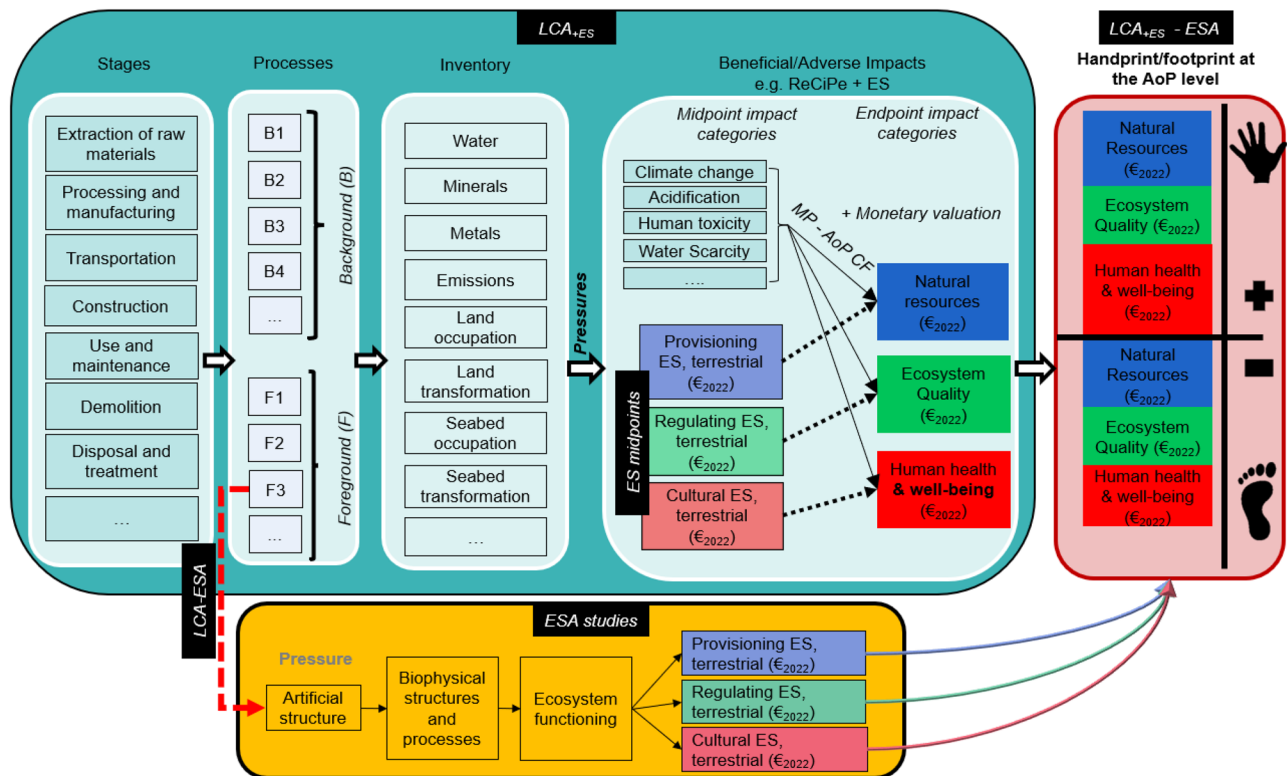
Fig. 4 The developed LCA$_{+ES}$-ESA framework; a way to quantitatively analyze the results of LCA and ESA in a combined manner for a specific human activity and related product (functional unit). Monetized results for three areas of protection (natural resources, ecosystem quality, and human health and well-being) are expressed in €$_{2022}$. Benefits are shown as handprint and burdens as footprint

environmental impacts/benefits. Apart from the more traditional life cycle impact categories from the ReCiPe 2016 method, three new midpoint impact categories are considered: provision, regulation, and cultural ES, currently limited to terrestrial ecosystems. These categories replace the former land use category of ReCiPe 2016: they account for the impact of land occupation and transformation on ES and are expressed in monetary terms as the site-generic (but land use and ES specific) CFs were built making use of the ESVD (see Sect. 2.3). Those three new midpoint impact categories are categorized into the earlier introduced endpoints or AoPs they affect (Figs. 1 and 4). Furthermore, the conventional midpoint impact categories are multiplied with the endpoint CF of ReCiPe 2016 to translate the results to burdens at the AoP level. Consequently, monetary valuation and conversion techniques are needed to be applied at this level to monetize all three AoP (in €$_{2022}$). Again, the handprint/footprint concept is then applied to show clearly the benefits and burdens for each AoP.

Practical guidelines for applying the LCA$_{+ES}$-ESA framework to any case study are depicted in Fig. 5. It comprises a total of 12 steps, in which the two integration approaches intermingle. In case no ESA study is available to account for local ES changes of some processes in the value chain, steps

2–5 are no longer valid. In this case, it is recommended to only follow the steps associated with the LCA$_{+ES}$ approach.

3.2 Results of the case study: pinewood production in Belgium

3.2.1 Results from local ESA (based on the LCA-ESA approach)

The results (expressed in € per kg Scots pine timber) for the three regulating ecosystem (dis)services PM$_{2.5}$ removal, CO$_2$ sequestration, and N pollution/removal, considered for the mid management and moderate climate conditions scenario of Schaubroeck et al. (2016), are shown in Table S8 (Supplementary Information) next to the provisioning service. The background calculations can be found in Box S3 (Supplementary Information). As a result, the ES wood provisioning and PM$_{2.5}$ removal bring the most added value to society, followed by CO$_2$ sequestration. By contrast, N pollution has a negative value per kilogram of wood and is therefore considered a disservice. According to Schaubroeck et al. (2016), this disservice is due to the extra presence of N in water flows resulting from the depletion of the nitrogen stocks in the soil. In addition, there is a damaging effect because of a change in the form of N, e.g., to nitrate, in which the amount
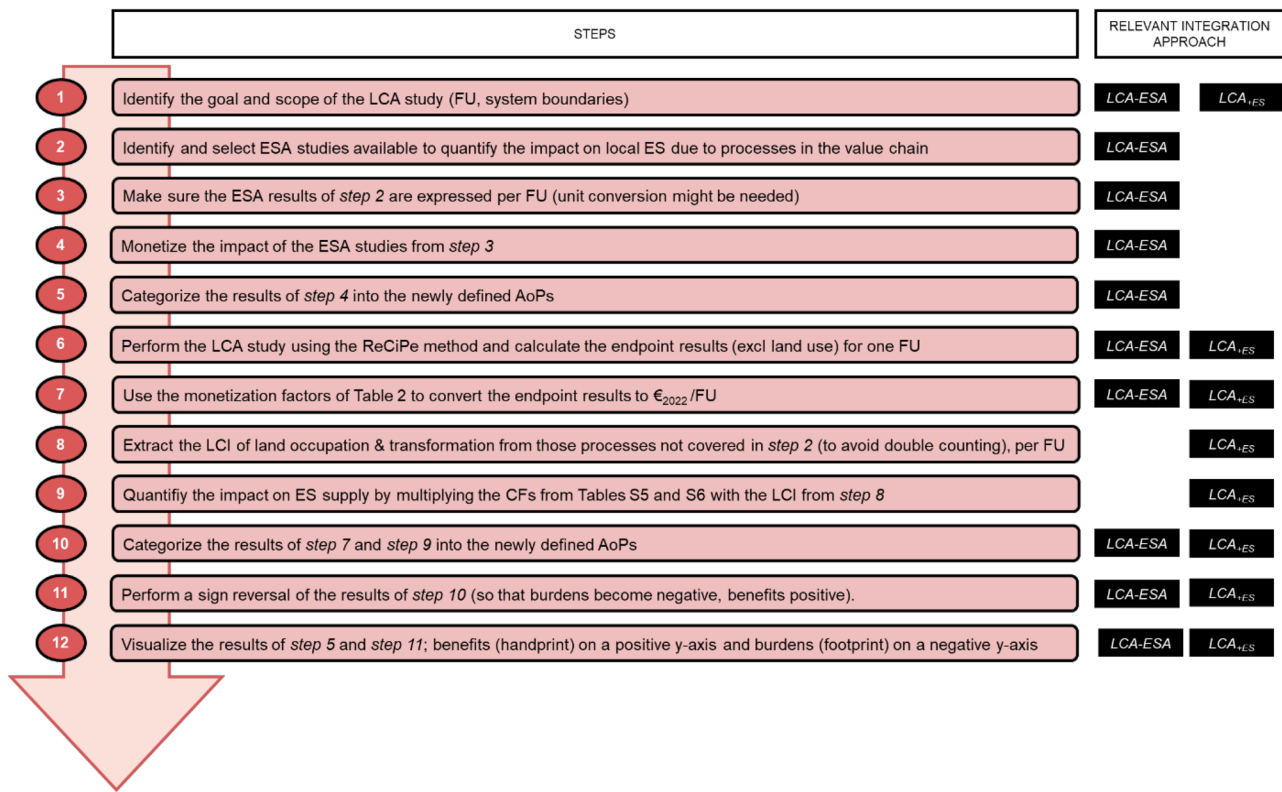
Fig. 5 A 12-step guideline to apply the LCA$_{+ES}$-ESA framework

is present. More details on the calculations can be found in Schaubroeck et al. (2016).

3.2.2 Results from LCA$_{+ES}$

The endpoint LCA impact results derived from applying ReCiPe 2016 expressed per AoP of 1 kg cleft timber of pine softwood are shown in Table S9 (Supplementary Information), except for "land use" as those impacts will be treated separately (see further). For the AoP HH&WB, the categories fine particular matter formation and stratospheric ozone depletion contributed most due to the tractor-powered clefting device used which runs on diesel and the process of skidding (logging operation to pull out cut trees from the forest). For the AoP EQ, global warming (terrestrial ecosystems) is mainly affected by the power sawing process, having a high share of direct greenhouse gas emissions. From the AoP NR, the fossil resource category is contributing most because of the high amount of petroleum being used in many cutting, clearing machineries/ devices, etc. In absolute terms, the impact per FU is 6.71E^{-08} DALY, 1.83E^{-10} species * year, and 5.22E^{-03} USD$_{2013}$, and after applying the monetization factors of Table 2, we end up with a total of €$_{2022}$ 1.02E^{-02} per FU of which the AoP HH&WB contributes 49.9% (€$_{2022}$ 5.11E^{-03}), the AoP NR 40.4% (€$_{2022}$ 4.14E^{-03}), and the AoP EQ 9.7% (€$_{2022}$ 9.89E^{-04}).

Furthermore, to account for ES changes due to land use, the LCI regarding land occupation/transformation flows is extracted from the SimaPro software, and the flows are multiplied with the respective provisioning, regulating, and cultural CFs from Tables S5 and S6 (Supplementary Information). For detailed calculations, see Table S10 (Supplementary Information). Results of ES changes due to land use are shown in Table S11 (Supplementary Information), summed per AoP. Only the AoP HH&WB undergoes a burden (positive values in LCA), while the other AoPs receive overall ES benefits (negative values).

3.2.3 Results of LCA$_{+ES}$-ESA

As can be seen from Table 3, the handprint is four times larger than the footprint to produce 1 kg of cleft timber pinewood (factor 4.2). The dominating contributions (both at the handprint and footprint side) are from the land use category where ES changes are quantified per m^2 of land transformed or m^2 year of land occupied for each of the processes in the life cycle of wood production. Table S10 (Supplementary Information) shows the high amount of specifically land transformed from the primary and secondary forest (non-use) which is accounted for with the use of the newly developed CFs. There is no handprint associated to

Table 3 Handprint and footprint results related to Scots pine production. Differentiation is made between the type of contribution and the AoP it connects to. The results are expressed in €$_{2022}$ per kg of wood

Handprint or footprint	AoP	Type of contribution	Result (€$_{2022}$/kg wood)
Handprint	AoP NR	Local ES—wood provisioning (FU)	6.61E−02
		Provisioning ES—MP	3.71E+02
	AoP EQ	Regulating ES—MP	6.09E+02
		Local ES—PM$_{2.5}$ removal	5.41E−02
		Local ES—CO$_2$ sequestration	1.64E−02
	Total handprint		*9.81E+02*
Footprint	AoP NR	ReCiPe 2016 MP's (except land use)	−4.14E−03
	AoP EQ	ReCiPe 2016 MP's (except land use)	−9.89E−04
		Local ES—*N* pollution	−1.74E−02
	AoP HH&WB	ReCiPe 2016 MP's (except land use)	−5.11E−03
		Cultural ES—MP	−2.31E+02
	Total footprint		*−2.31E+02*

MP midpoint, *FU* functional unit, *NR* natural resources, *EQ* ecosystem quality, *HH&WB* human health and well-being

the AoP HH&WB. Within the AoP NR, the handprint partly consists of the beneficial impact of the functional unit, i.e., the intended user is the one that buys the wood. On top, the other part of the handprint within the AoP NR is associated with benefits received along the value chain, i.e., received by unintended users (e.g., those living close to the forest under study or in the proximity of the location where the other processes of the value chain take place). The handprint in the AoP EQ includes beneficial impacts both from the local ESA studies carried out by Schaubroeck et al. (2016) (although minor) and from the ES supply linked to the other processes in the value chain. Avoided products in LCA were not considered in this simplified case study.

4 Discussion: challenges during the development and implementation of the LCA$_{+ES}$-ESA framework

When combining different established methods such as ESA and LCA, some challenges can be pointed out. To start with, there might be some issues regarding data availability/accessibility. The first approach (LCA-ESA) relies on local ESA studies that can be directly or indirectly linked to processes in the supply chain, and the functional unit as used in the LCA. However, sometimes ESA studies are not available, are deemed incomplete (cfr. not accounting for all relevant ES), or are only presented qualitatively. Mainly cultural and supporting ES face this issue (Blanco et al. 2018).

Because of an intrinsic limitation of most available LCA and ESA methods, our LCA$_{+ES}$-ESA framework has the inability to account for feedbacks and interrelations in the impact pathways, i.e., how the change in one ES can affect the supply of others. Methods have been developed

to solve this, such as Petri nets (e.g., Rova et al. 2019) or integrated earth system dynamic models (e.g., Arbault et al. 2014); however, these are generally more complex and may require even more data. Furthermore, within the LCA$_{+ES}$ approach, the focus is put on the cause-and-effect chain having land use as a stressor to ES change. Although this way of integrating ES in LCA is indeed frequently applied in the state of the art (De Luca et al. 2022), it does not account for potentially other relevant stressors (such as locally concentrated emissions or the use of freshwater resources) that can influence the supply of ES (Hardaker et al. 2022). For future work, it is therefore recommended to explore the cause-effect chains of other conventional LCA impact categories (e.g., water use and ecotoxicity) with respect to the supply of ES. Also, attention should be paid to the risk of double counting between ESA and LCA, such as overestimating the effect of a stressor (human activity) at the midpoint or endpoint damage level by accounting for emissions or resources use in similar cause-and-effect pathways considered in both methods, e.g., when freshwater eutrophication and ecotoxicity in LCA are combined with the nutrient cycling ES (Biones-Hidrovo et al. 2020). More research is also needed regarding the link between ES and the AoPs in LCA. In this study, we assumed that a change in regulating ES affects the AoP EQ, cultural ES the AoP HH&WB, and provisioning ES the AoP NR. Nevertheless, it is certainly more complex in reality due to the many interrelations among ES and LCA impact categories, which needs a thorough investigation. In addition, LCA tends to mainly model linear relationships between causes and effects, while ESA might be considered more advanced in covering the dynamics and complex interactions of natural processes, which are frequently non-linear and variable in

time and space but these changes are in most cases not yet captured in LCA impact modelling (de Souza et al. 2018).

Monetary valuation definitely has its advantages; e.g., it allows the conversion of several types of impacts expressed in different units into monetary units. That way, an easier interpretation of impacts derived through comprehensive assessment tools such as life cycle sustainability assessment (LCSA) which covers the three pillars (prosperity, environment, society) is possible. This may be a preferred solution compared to alternative weighting methods such as panel-based or distance-to-target methods as monetized impacts can be related to the economy, in this way facilitating decision-making (Amadei et al. 2021). However, monetization of impacts/benefits also brings along some challenges. For instance, there has been recurrent criticism about the appropriateness of using monetary valuation methods to measure the instrumental value of biodiversity or ES changes and/or the value to society, because valuation methods may not be effective to measure the scale, uniqueness, and threat of ecosystems. It can be seen controversial in case species, habitats and ES would be commodified, like any other, and would be monetized at the daily rate to be traded between different actors speculating for profit (Darbi 2020; Temel et al. 2018). Furthermore, monetization has a continuous need to adjust for inflation and/or to allow a fair comparison of economic productivity and standards of living between countries through the PPP. Apart from these potential constraints, there are inconsistencies to monetize (in €$_{2022}$) at the level of the AoP in LCA.

For the AoP HH&WB expressed in DALY, a contingent monetary valuation method was used, following the approach of the NEEDS project where the value of a life year was quantified by asking people for their willingness to pay for a three or 6-month longer life span as a result of improved air quality (de Bruyn et al. 2018). For the AoP EQ expressed in species * year, results from a meta-analysis of economic studies on biodiversity loss in a European context are used for terrestrial biodiversity alone (as there is no data for aquatic ecosystems), based on values provided by Kuik et al. (2008) and de Bruyn et al. (2010; 2018). Furthermore, they considered the Ecosystem Damage Potential (EDP) as equal to PDF and the calculated mean and median EDP values also differ by a factor of eight, indicating that the distribution of values is skewed with a long tail of high values and which may reduce the reliability of monetary values used in this study. For the AoP NR expressed in \$$_{2013}$, which refers to the extra costs for further natural resources extraction due to increased scarcity (Huijbregts et al. 2017), a conversion factor was used to convert to €$_{2022}$, taking into account the PPP differences and HICP.

To date, there is no standardized way of using monetary valuation techniques at the LCA AoP level. The Environmental Prices Handbook (de Bruyn et al. 2018) also shows a broad range of environmental prices at the pollutant, midpoint, and endpoint levels. In this study, central values were used as the upper and lower values are mainly relevant for social cost–benefit analysis studies. The central values obtained are highly dependent on the valuation method behind it, and could easily change when opting for another monetary valuation approach. A similar conclusion can be drawn from Amadei et al. (2021) who provided an overview of monetary valuation methods used across and within certain impact categories and who concluded that there is a clear lack of consensus. Furthermore, at the level of ES, there is a large variation regarding the monetary valuation methods used to calculate local ESA study results or the values from the ESVD. For instance, in the ESVD, in about 23% of the values, market prices are dominating dealing mainly with marketable ES such as provisioning and cultural ES, followed by contingent valuation (17%), damage costs (14%), choice experiment (12%), and travel costs (8%) which are especially used for recreational and cultural ES except for damage costs which are most suitable for regulating and provisioning ES (extracted from Fig. 7 in de Groot et al. 2020). On top, the ESVD has an important difference in the reliability of the ES valuation with the highest number of datapoints for cultural ES and the lowest for regulating and maintaining ES. The geographical scope of this database is mainly Europe and especially the UK and The Netherlands, and therefore, the average value for a biome (e.g., temperate forests) is rather a European-influenced average (de Groot et al. 2021).

When exploring the case study results, it can be observed that the monetized ES results through LCA (from the LCA$_{+ES}$ approach) are substantially larger than the monetized local ESA results and monetized LCA (ReCiPe 2016) impact categories. At first glance, this can be explained by the fact that monetization of impacts relies on a bunch of different valuation methods which can raise some issues when comparing the results obtained. For example, the monetary valuation methods used for the LCA AoPs HH&WB and NE are relying on contingent valuation methods (i.e., willingness to pay) while this method is rather limited for local ESA.

Monetizing ES changes and LCA impact results provides an intrinsic weighting that enables the handling of trade-offs between the midpoint categories and AoPs, i.e., they are not equally weighted but weighted based upon their monetary values. This is not necessarily a drawback as equal weighting would mean that they would be equally important and substitutable and that the effects on different categories or AoP could easily commensurate with each other (Hardaker et al. 2022).

Another challenge faced in the development of the LCA$_{+ES}$-ESA framework is the limited consistency between the ESVD and life cycle inventory databases such

as Ecoinvent v3.8. As the ESVD only deals with a limited amount of biomes (which we connected to land use classes), it appeared that for some land occupation or land transformation flows, no data was available from the ESVD, and therefore, it has not been possible to calculate CFs for all land use flows in Ecoinvent (Table S2, Supplementary Information). Furthermore, the biomes for the aquatic ecosystems (coral reefs, open sea/ocean, coastal systems, etc.) are not found compatible with sea occupation and transformation flows available in Ecoinvent (which mainly focusses on the seabed). In case the Ecoinvent database would expand its seabed/sea-surface use flows and the ESDV database would differentiate more in the aquatic biomes, then a similar calculation could be performed to obtain CFs that account for the effect on marine ES. Also, we used an area-weighted average of all biomes for the unspecified land use flows of Ecoinvent as there was not really an "unspecified biome". Although the ESVD provides ES values for biomes distributed globally, the datapoints are mainly European and interpolation was used, which may hamper its accuracy. The ESVD also lacks datapoints for some ES—biomes (or land use classes) combinations; for example, for the ES medicinal resources and biome temperate forest, there is not a single datapoint (Table S2, Supplementary Information). This might be because there is no link between the ES and biome, or because there are no local studies done so far. To be able to calculate as many CFs as possible, in this study, we have also taken into account the ES-biome combinations that have less than 5 datapoints (Table S2, Supplementary Information, italic numbers), which gives a higher uncertainty to the results.

The LCA$_{+ES}$-ESA framework proposed here is based on the ReCiPe 2016 LCA method with its respective categories and available CFs to link its midpoint impact categories to the three AoPs. Monetization of the impact results is done at the level of the AoP, although environmental prices are available as well at the midpoint level. However, if results at the endpoint level are preferable (e.g., to ease communication), it is deemed more relevant to monetize at this level instead of the midpoint level because midpoint-to-endpoint characterization is often missing in the Environmental Prices Handbook; therefore, allocating the monetized midpoint results to a certain AoP is dangerous as it might not account for the intrinsic burdens in the respective AoP, e.g., species loss in the AoP "natural environment" (cfr. ecosystem quality).

It might also be interesting to investigate the potential to adapt the LCA$_{+ES}$-ESA framework using another LCIA method, such as the product environmental footprint (PEF) method recommended by the EC (Manfredi et al. 2012; EC-JRC 2011). Amadei et al. (2021) provided an overview of monetary valuation coefficients (in €$_{2019}$) applicable for the PEF midpoint impact categories, which would allow the computation of a monetized PEF single score (e.g., by using weighting factors of Sala et al. 2018), but you cannot express the results at the level of AoP for instance. On top, additional research would be needed to integrate the monetized ES midpoint impact results calculated based on the CFs developed in this study in the single score, and to combine the local ESA results with it as well, as there is not a weighting set develop for integrating ES in the PEF method.

5 Conclusions

We propose to more adequately account for the pressures of human activities on our natural ecosystems, often generating local and global impacts, in order to be able to improve their net sustainability impact. To this end, we present a comprehensive LCA$_{+ES}$-ESA framework which can account for (socio-)environmental impacts of human activities and product systems by combining LCA and ESA by means of two integration approaches, namely "combination of results" (or LCA-ESA) and "complementation of a driving method" (or LCA$_{+ES}$). The advantage of the LCA-ESA approach is the possibility to include site-specific marine or terrestrial ES results in the overall assessment, next to conventional LCA impact categories such as climate change. Being time-consuming and data-intensive, however, the availability of location-specific ESA studies might be low. As a result, some processes in the value chain under study may lack specific ESA information. Therefore, the LCA$_{+ES}$ approach is developed to cover the remaining processes in the value chain: the LCIA method ReCiPe 2016 is updated to include new ES midpoint impact categories and associated site-generic CFs based on the ESVD to calculate changes in provisioning, regulating, and cultural ES due to land use. Monetary valuation techniques are used to aggregate the results of both approaches at the level of a newly defined set of AoPs: natural resources, human health and well-being, and ecosystem quality. Benefits such as an increased supply of ES, avoided products, and the functional unit in LCA can be visualized as a handprint while burdens such as results from the conventional LCA impact categories, or a reduction of ES, are shown as a footprint. To facilitate the applicability of the developed LCA$_{+ES}$-ESA framework, a 12-step plan is presented as a guide for the practitioner. While the framework proposed is ready to apply, as demonstrated through the pinewood case study, it still faces some challenges and limitations as outlined in the paper. Further research is therefore recommended, as described in Sect. 4. In general, this paper lays out a LCA$_{+ES}$-ESA framework to be applied to product systems in order to identify, measure, and address the key factors that contribute to (socio-)environmental burdens and potential gains, in an effort to optimize the sustainability of human activities around the globe.

Declarations

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Authors and Affiliations

Sue Ellen Taelman[1] ⬤ **· Laura V. De Luca Peña[1] · Nils Préat[1] · Till M. Bachmann[2] · Katrien Van der Biest[3] · Joachim Maes[4] · Jo Dewulf[1]**

✉ Sue Ellen Taelman
sueellen.taelman@ugent.be

[1] Ghent University, Green Chemistry and Technology, Coupure Links 653, 9000 Ghent, Belgium

[2] European Institute for Energy Research (EIFER) EDF-KIT EEIG, Emmy-Noether-Str. 11, 76131 Karlsruhe, Germany

[3] University of Antwerp, ECOSPHERE Research Group, Universiteitsplein 1, 2610 Wilrijk, Belgium

[4] European Commission, Brussels, Belgium