D5.1

Suitable aggregation techniques: overview

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Acronyms

AoP	Area of Protection
BBP	Biotic Production Potential
CF	Characterisation Factor
ES	Ecosystem Services
ERA	Ecological Risk Assessment
ESA	Ecosystem services assessment
HANPP	Human Appropriation of Net Primary Production
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LUC	Land Use Change
OWF	Offshore Wind Farm
SOC	Soil Organic Carbon
TES	Techno-Ecological Systems

1. Executive summary

Multiple methods have been developed to assess environmental and socio-economic impacts caused by human activities, and the selection of a specific method depends of the scope of the study. For example, Life Cycle Assessment (LCA) is widely used to assess a broad range of environmental impacts from a regional to global perspective. On the other hand, Ecosystem Services Assessment (ESA) is used to quantify the local to regional impacts of benefits of a human activity on provisioning, regulating and cultural ecosystem services. While both methods can be be applied in parallel to the same activity to assess its different environmental impacts, attempts have been made to combine them into a more exhaustive methodology to enhance the understanding of the overall environmental sustainability impact. This deliverable aims at reviewing the state-of-the art regarding integration of ESA and LCA, showing the benefits and drawbacks of each method separately and discussing the integration challenges, and classying the different types into integration categories. A total of 25 studies are identified and 3 major integration categories are distinguished: (1) the "post analysis" approach where both methods are ran in parallel without any integration of the results; (2) the "integration" approach where both methods are ran in parallel and their results are integrated by applying a combined metric; and (3) the "driving method" approach where one method is integrated into the other one to form a hybrid method. The last is by far the mostly applied approach to combine ESA and LCA with 22 papers out of the 25 following this approach. For most of the cases, LCA is the driving method and ESA is adapted to the LCA framework. Despite its higher complexity to adapt method-specific impact pathways (e.g. adapting different impact pathways on a particular ecosystem service into the LCA structure), this approach has potential to capture and balance both local, regional and global aspects in a single method. Furthermore, if a single score approach is preferred to combine local to global effects, then monetisation is seen as a commonly used aggregation method. Therefore, in the SUMES project, this type of integration looks also very promising and will be explored to use in practice (cfr. upcoming D4.3).

2. Goal and scope of the deliverable

This deliverable is linked to Task 5.1 (Aggregating adverse and beneficial impacts, locally and globally, in function of a decision support tool) and aims to identify potentially interesting/meaningful aggregation techniques for the methods used in SUMES project. The environmental and socio-economic effects of human activities on the marine environment may be local (e.g. reduction of marine fish biodiversity) or more global (e.g. impact on global warming through consumption of materials related to the activity). Important methods to assess these different impacts/benefits occurring at different geographical scales, are Life Cycle Assessment (LCA) and Ecosystem Services Assessment (ESA). Both methods are briefly discussed separately, highlighting their main principles and limitations (section 3). In addition, as these methods cannot comprehensively assess the impacts of marine activities, it is worth to explore how they can be integrated. A total of 42 publications from the state-of-the art have been reviewed to better understand how these methods (ESA and LCA) are currently integrated (either only conceptually or also applied). Section 4 is devoted to the description of the main purpose of the papers where some kind of integration between ESA and LCA methods is applied. Moreover, the challenges in terms of integration are explained. Based on the understanding on how each study integrated the methods, a classification scheme was proposed to characterize the different integration approaches (section 5). A final section explains the way forward within SUMES for the integration of ESA and LCA results.

3. Methods to identify the socio-economic and environmental effects of offshore human activities

As discussed in D4.1, offshore wind energy produced at an offshore windfarm (OWF) has been identified as the first case study for SUMES. The production of offshore wind electricity induces both local effects, i.e. environmental (and socio-economic) impacts to local ecosystem services (e.g. fish provisioning, waste remediation), and global effects, i.e. environmental impacts to the global environment (e.g. climate change) (**Figure 1**). This section focusses mainly on two methods that are used in SUMES, Life Cycle Assessment (LCA) and Ecosystem Services Assessment (ESA), to assess complementary aspects towards a comprehensive environmental sustainability assessement of marine activities. While LCA tackles global environmental impacts associated to the entire life cycle of the OWF, ESA focuses on the impacts of the activity on local ecosystem services through change in ecosystem strucutre (**Figure 1**).

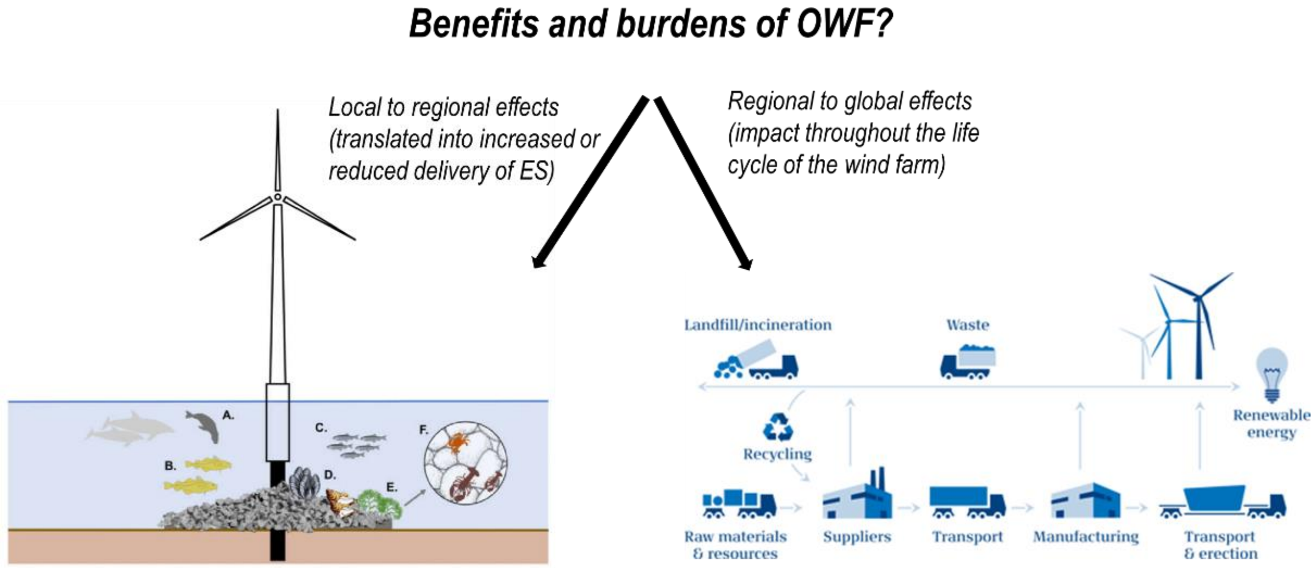

Figure 1 Scope of the environmental sustainability assessment performed in the offshore windfarm (OWF) case study.

3.1. Short introduction to LCA and its limitations

3.1.1. Principles of Life Cycle Assessment

Life Cycle Assessment (LCA) is an internationally standardized methodology to assess quantitatively the environmental performance of a product or a service (Guinée, 2002). The methodology is framed by the International Organization for Standardization (ISO), under the 14000 series (ISO, 2006). According to the ISO standards, LCA is organized in four steps (as presented in **Figure 2**): (1) the definition of the goal and scope of the study which includes the definition of the functional unit (i.e. the quantity of the product or service under study) and the system boundaries; (2) the development of a Life Cycle Inventory (LCI) consisting of an exhaustive dataset of elementary flows (*i.e.* materials and energy inputs and outputs) associated within the study's boundaries; (3) the impact assessment step where elementary flows from the LCI are multiplied with specific impact factors (i.e. characterization factors, CF) to assess their contribution to defined impact categories based on elementary flow-specific cause-effect chains (i.e. life cycle impact assessment, LCIA) and, (4) the interpretation of the three previous stages with global conclusions and recommendations as main outcomes (ISO, 2006). The stages are interdependent (i.e. double sided arrows in **Figure 2**): the completion of one stage informs on how to complete other stages.

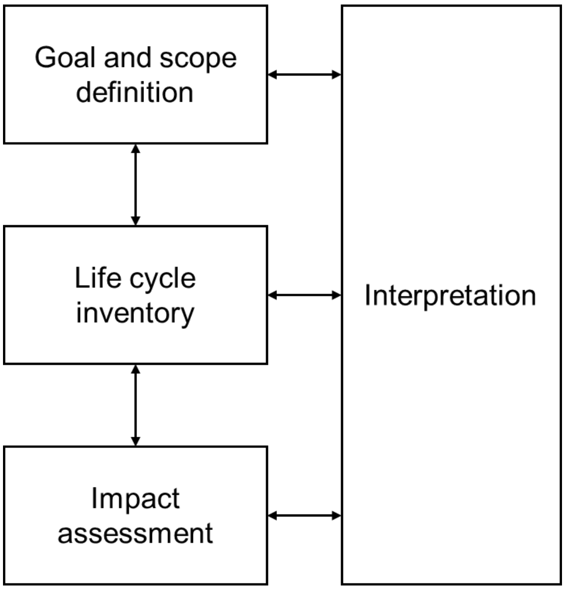

Figure 2 Stages of Life Cycle Assessment (LCA) according to ISO 14040.

The modelling of the environmental impacts in LCA relies on impact pathways that link specific environmental stressors caused by human activities (e.g. pollutants emissions, natural surface occupation) with one or multiple potential effects on the environment. These effects are classified into impact categories defined for two levels of aggregation: the midpoint level (i.e. aggregating directly the contribution of the LCI flows to a first set of impact categories) and the endpoint level (i.e. aggregating midpoint impacts into a reduced number of broader impact categories, called Areas of Protection – AoP) (**Figure 3**). Several quantitative life cycle impact assessment methods (LCIA) methods exist to quantify various impact categories (De Luca et al., 2022). For example, the European Commission developed its own LCA method, the Product Environmental Footprint (PEF or generally the EF framework), relying on multiple LCIA models that are recommended by the scientific community to assess impacts at midpoint level (Fazio et al., 2018). Another example, the ReCiPe 2016 method (Huijbregts et al., 2017), has 17 midpoint categories in total (e.g. global warming, land use, freshwater ecotoxicity). The most used AoPs in LCA are human health, ecosystem health and natural resources. However, other AoPs have been proposed 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, some methods aggregate the impacts into a single score to evaluate the overall environmental costs and benefits (Manfredi et al., 2012). Different normalisation and weighting methods have therefore been developed, of which the most often used ones are internal and external normalisation, the distance to target method, the panel preferences, the monetization and the evidence approaches (Sala et al., 2018). The further the aggregation is carried out, the more decision making is facilitated, however, it brings along increased uncertainty and reduced transparency (Préat, 2021).

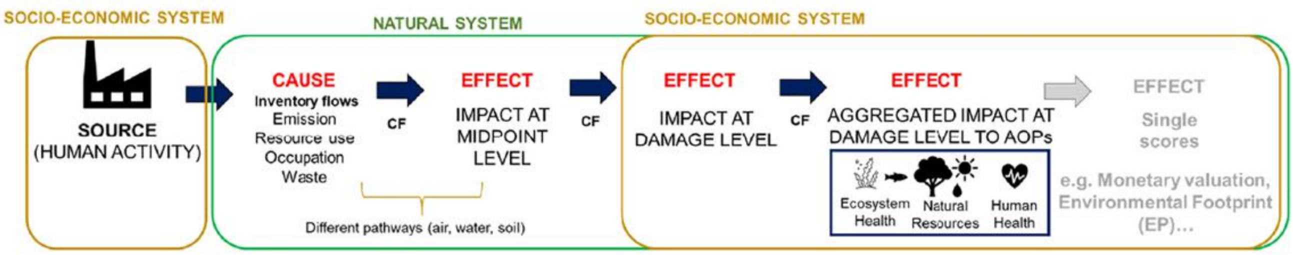

Figure 3 Representation of the cause-effect chain modelling in LCA. The yellow, green and blue boxes encompass the components of a socio-economic system, the components of a natural system and the entities that need to be protected respectively. The grey shadings at the end of the cause-effect chains represent the elements that are not always assessed in LCA (from De Luca et al., 2022).

3.1.2. Major limitations of Life Cycle Assessment

Four main limitations of LCA are discussed: (1) the implementation of spatially differentiated impact pathways, (2) the lack of coverage of LCA for marine impact categories, (3) the causal linearity of modelling the impact pathways and (4) the level of aggregation.

(1) Historically, LCA has been developed for the evaluation of global environmental impacts (e.g. global warming), using generic impact pathways (i.e. LCIAs). Regional impacts (e.g. freshwater consumption impacts), that differ according to the geographical location of the inventory flows, have been included in a second stage and rely on the development of regional LCIAs in LCA (Bulle et al., 2019; Patouillard et al., 2016). The regionalization of the cause effect chains is a main challenge in LCA to improve the spatial representativity of the results and many regional aspects are still overlooked such as impacts on regional biodiversity (Curran et al., 2016) or impact on local ecosystem services (Rugani et al., 2019). Overall, the difficulty to address regional parameters in LCA arises from the complexity of modelling site-specific ecological processes / interactions under specific disturbance (type, intensity) (De Baan et al., 2013; Chaplin-Kramer et al., 2017).

(2) Most of the impact categories included in LCA methods focuses on the terrestrial environment. Hence, marine impact categories are poorly covered in LCA results except for marine eutrophication and marine acidification which are included in some LCA methods such as ReCiPe and PEF (EC, 2017; Huijbregts et al., 2017).

(3) The LCIA follows a linear approach by multiplying the LCI with specific CFs to assess a marginal increase in an impact category. However, CFs are derived from non-linear models (e.g. CFs for biodiversity impacts rely on species-area models) but are used in a linear way in LCA (Heijungs, 2020). Those represent a marginal increase in their target impact category. This limits LCA to global impact catogories for which the contributions of the LCI flows are not affecting the capacity of the system under study to absorb specific impact and hence, the impact function (i.e. LCIA).

(4) Although aggregation is useful in LCAs to help simplify the communication of results, to identify trade-offs between different alternatives and to inform decision-makers, it is considered a controversial step in both methodologies (Buckwell et al., 2018; Kalbar et al., 2017; Prado et al., 2020). Aggregation in LCA has been criticized because of its subjectivity in the normalization and/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 are controversial and heavily criticized for ethical reasons (Sullivan, 2009; Sullivan and Hannis, 2017).

3.2. Short introduction to ESA and its limitations

3.2.1. Principles of Ecosystem Services Analysis

Ecosystem services (ES) are defined as the benefits people obtain from ecosystem functioning (Millenium Assessment, 2005). This concept extends the ecological dimension of ecosystems to a socio-economic perspective. Three main frameworks for classifying ecosystem services have been proposed and are commonly accepted (Millenium Assessment, 2005; Sukhdev and Kumar 2008; Willot et al., 2019). The terminology of ES varies accordingly but the main categories of ES are somehow similar amongst the three frameworks. The categories consist of provisioning services (e.g. fish production), cultural services (e.g. area for recreation), regulating services (e.g. water purification) and supporting / maintenance services. Depending of the framework considered, supporting / maintenance survices are integrated within the regulating services category.

Ecosystem Services Analysis (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 (De Luca et al., 2022). As explained in D1.1 and D1.2, the Ecosystem Services Cascade model of Haines-Young & Potschin (2010) has been widely adopted and describes that ES are supplied through ecosystem functioning, consisting to biological processes occuring within specific physicochemical conditions. The cause-effect pathway modelling in ESA is heterogeneous in the sense that 1) the analysis starts from changes (δ) on a biophysical structure without considering the cause (e.g. human activity) and 2) the level of aggregation of the results differs substantially (results per ES, per category of ES, monetized and through a single score, etc.).

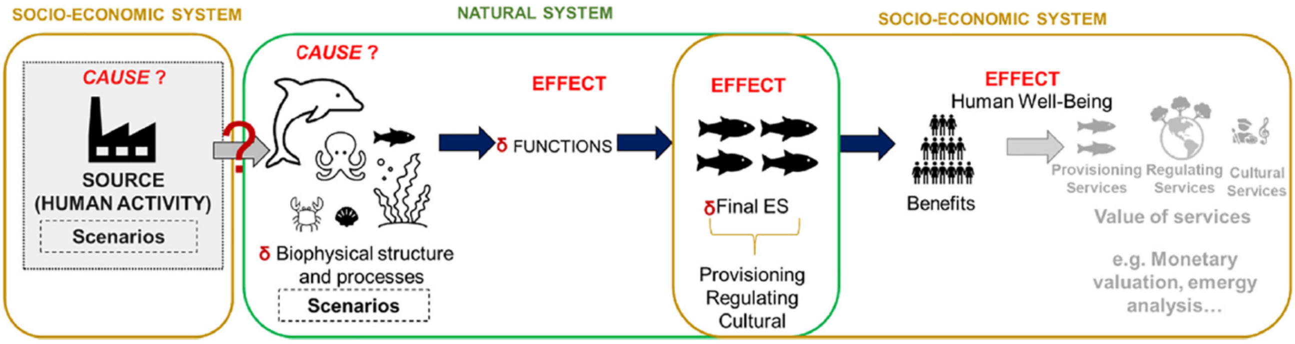

Figure 4 *Representation of the cause-effect chain modelling ESA. The yellow and green boxes encompass the components of a socio-economic system and the components of a natural system, respectively. The grey shadings at the end of the cause-effect chains represent the elements that are not always assessed in 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 be based on a change within a biophysical structure and process that is not related to human interventions (from De Luca et al., 2022).*

3.2.2. Major limitations of Ecosystem services analysis

The ESA also includes some limitations depending of the study, these concern (1) the use of qualitative methods, (2) the scope definition that does not necessarily includes human activities and, (3) the final aggregration of the results through monetization.

(1) Apart from quantitative, also qualitative methods are used in ESA to capture the changes within a local ecosystem and consequently changes to the supply of ES (Grêt-Regamey et al., 2017; Harrison et al., 2018; Hein et al., 2006). Qualitative methods lead to subjectivity in the interpretation of the results, which can be inaccurate. Also, most qualitative methods take a lot of time and do not offer any statistical representation.

(2) In addition, conducting an ESA does not necessarily mean taking human activities as such into account as a starting point of the cascade modelling, and therefore ESA would not be able to quantify the impact of human activities on the local environment. Rather, ESA starts from changes in ecological parameters and their effects on ES supply. One way to overcome this limitation is to make use of additional methods, that are able to quantify the impacts of human activities on ecological parameters. One possible method is Ecological Risk Assessment (ERA), which assesses the probability of adverse effects on organisms exposed to a toxic substance, to a biological stress or to a physical change (EPA, 2016a,b). Similar to LCA, ERA is an iterative process with distinct stages such as goal & scope definition, evaluation of exposure pathways and intensity (duration, frequency, magnitude) and identifying the reponse of the receptor (effect) depending of its sensitivity, a risk characterization stage of which the results are confronted to specific protection goals (e.g. biodiversity, ecosystem services) in accordance to policy targets (EFSA, 2016; De Luca et al., 2022). Depending in the ES under study, also other methods can be used; GIS layer overlaps, food web modelling, proxy indicators to address cultural ES changes due to OWFs such as willingness-to-pay to retain a natural seascape or bird spotting indicators linked to recreation.

(3) Monetization is often used to report changes in ecosystem services from a socio-economic perspective and also, to aggregate changes in different ES in one single value. However, this may raise some issues: except for provisioning most of ES do not present any market value and, the value of ES depends of the incomes of the country (Costanza et al., 1998; Daily, 1997). Different monetary valuation methods are used depending of the service, from direct market value for provisioning services to willingness to pay, hedonic pricing or contingent valuation for non-marketable services. The results are adapted according to the

purchasing power of the country (De Groot, Wilson, and Boumans, 2002). Furthermore, ESA dealing with future impacts on ES make use of discounting methods to assess the present value of future costs on ES, raising debate about intergenerational equity in function of the discounting factor selected (e.g. declining versus constant, high value versus low value) (Padilla, 2002; Rabl, 1996; Weitzman, 1994).

4. Integration of LCA and ESA in the state-of-the art

A review study was carried out by De Luca et al., (2022) in which the various integration approaches used to combine LCA and ESA were investigated. Mainly, the Web of Science was used as search engine for selecting most of the the literature and Google Scholar 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, 42 relevant scientific papers covering LCA-ES were selected to analyse their main characteristics and content. From these papers, a second selection was made, whereby only 25 papers with a case study were retained, leaving out the conceptual papers, as we only wanted to mention integration approaches that have been proven to be applicable. 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 of De Luca et al., (2022).

4.1. Integration methods used

Most of the papers found during the review study focused on incorporating ES as an additional impact pathway to traditional pathways in LCA (such as ozone depletion, particulate matter, climate change) or to change the modelling of traditional impact pathways (e.g. land use), and develop spatially-differentiated midpoint and/or endpoint CFs. Most of the studies do not report the impacts on ES in monetary terms and rather express them in physical units, at the structure/process level. A few papers are following the UN Environment Programme and Society of Environmental Toxicology and Chemistry (UNEP-SETAC) guidelines;

- Müller-Wenk and Brandão (2010) focused on the climatic impact of land use as determined by the CO_2 transfers between vegetation/soil and the atmosphere in the course of terrestrial release and re-storage of carbon. Compared with the potential natural vegetation as a baseline, land occupation and transformation may lead to a reduced storage of carbon in soil and vegetation, whereby the mobilized carbon is essentially transferred to the atmosphere, contributing to global warming. The size of this climatic impact is determined by the amount of carbon transferred per hectare, as well as by the duration of the carbon's stay in air.
- Brandão and Milà i Canals (2013) studied the impacts of land transformation and occupation on biotic production potential (BBP), and endpoint for the AoP natural resources, and used soil organic matter (SOM) as a proxy indicator for BBP. Characterisation factors for eight land use types at the climate region level were developed. Land-use-specific and biogeographically differentiated data on SOC is needed to make the BPP impact assessment operational.
- Saad et al. (2013) assessed potential land use impacts on ecosystem services (erosion regulation potential, freshwater regulation potential and water purification potential). Spatially differentiated CFs were calculated for each biogeographic region (Holdridge life regions, Holdridge life zones, and terrestrial biomes) along with a nonspatial world average level. In addition, seven land use types were assessed considering both land occupation and land transformation interventions.
- Muñoz et al. (2014) assessed the environmental impacts caused by bio-based ethanol production. Apart from midpoint impact categories of the ReCiPe method (Goedkoop et al., 2009), also novel impact categories on biodiversity and ecosystem services were addressed; biodiversity damage potential, climate regulation potential, biotic production potential, freshwater regulation potential, erosion regulation potential, water purification potential through physicochemical filtration and water purification potential through mechanical filtration.

- Cao et al. (2015) proposed a new LCIA method to estimate the decrease in value of the ecosystem services provided to society due to land use. Six midpoint land use indicators are proposed by the LULCIA project (BBP, ground water recharge, erosion regulation, mechanical and physicochemical water purification and climate regulation potential) and developed a new AoP named resources and ecosystem services. Indicators expressed in biophysical units are converted into monetary units based on the economic valuation of the reduction of a given ecosystem service. Impact scores are not only influenced by the biophysical specificity of the studied systems (e.g. crop yield affecting the inventory flow, type of biome affecting impact characterisation) but also by the local socioeconomic capacity to compensate for an ecosystem service loss and available compensation measures and technologies.

However, there were other papers who did not follow these UNEP-SETAC guidelines, nevertheless, they had similar ES-LCA integration goals;

- Zhang et al. (2010b) presented an Ecologically Based LCA (Eco-LCA) which includes a large number of provisioning, regulating, and supporting ecosystem services as inputs to a life cycle model. LCA methodology is combined with other methods such as network algebra, emergy, exergy, monetary valuation, HANPP, ecological footprint, etc. in order to account for ecosystem services. The resources are represented in diverse physical units and may be compared via their mass, fuel value, industrial/ecological cumulative exergy consumption or by normalization with total consumption of each resource or their availability. Such results at a fine scale provide insight about relative resource use and the risk and vulnerability to the loss of specific resources.
- Núñez et al. (2013) developed a globally applicable and spatially resolved method for assessing land use impacts on the erosion regulation ecosystem service. The regionalized assessment takes into account the differences in soil erosion-related environmental impacts caused by the great variability of soils. Life cycle inventory data of topsoil and topsoil organic carbon (SOC) losses were interpreted at the endpoint level in terms of the ultimate damage to soil resources and ecosystem quality.
- Arbault et al. (2014) demonstrated the feasibility of using an integrated earth system dynamic modeling perspective to retrieve time- and scenario-dependent CFs that consider the complex interlinkages between natural processes delivering ES. The Global Unified Metamodel of the Biosphere (GUMBO) model is used to quantify changes in ES production in physical terms – leading to midpoint CFs – and changes in human welfare indicators, which are considered here as endpoint CFs.
- Blanco et al. (2018) incorporated four ES (food production, carbon sequestration, tourism and recreation, flood protection) as a midpoint impact category in LCA; Physical models are first used to determine how physical units of ecosystems are transformed by industrial processes, while ES models are then used to determine the losses or gains of ES per ecosystem unit, and economic valuation is used to normalize and weigh the total ES losses/gains. Both spatial and temporal variability of ES production is taken into account, as well as socioeconomic aspects of ES use.
- Jeswani et al. (2018) assessed the life cycle impacts of land use on biodiversity and ecosystem services associated with the production of breakfast cereals. For biodiversity, the impacts on five taxonomic groups have been assessed: mammals, birds, vascular plants, amphibians and reptiles. For ecosystem services, the potential loss in the following services has been considered: biotic production, erosion resistance, groundwater regeneration, infiltration and physicochemical filtration. The LCIA methods for biodiversity and ecosystem services require identification and quantification of land occupation and land use change (LUC) in a spatially differentiated format.
- Liu et al. (2018) focussed on land use change impacts through the ecosystem service of carbon sequestration. The Century model was used to simulate the carbon dynamics for seven land use change scenarios under two climate change scenarios. The impact on carbon sequestration was calculated based on the difference of carbon sequestration between the land use change scenario and the corresponding baseline and the decay of CO2 in the atmosphere.

- van Zelm et al. (2018) developed CFs to be used in LCIA to determine the impact of crop cultivation on soil erosion. They derived spatially explicit erosion rates (kg of soil lost per kg of crop) as a function of crop choice and management practice on a global scale.
- Othoniel et al. (2019) proposed a new methodology to calculate midpoint and endpoint characterization factors for land use impacts on ecosystem services in LCA. A cause-effect chain of ES was established in line with traditional LCA to describe the impacts of land use and related land cover changes. CFs that are regionalized at two scales, for Luxembourg as a whole and for each of its 116 municipalities, were calculated. The calculated CFs enable the impact assessment of six land cover types on six ecosystem functions and two final ecosystem services (pollination and food production). Monetary valuation of ES changes is used.

Some papers focused on modifying the life cycle inventory step by using allocation as a way to integrate ES into LCA. Boone et al. (2019) proposed an allocation procedure based on the capacity of agricultural systems to deliver ES to divide the environmental impact over all agricultural outputs (i.e. provisioning and other ES). Allocation factors are developed for conventional and organic arable farming systems. Bragaglio et al. (2020) aimed to compare four beef-production systems in terms of global warming potential, acidification potential, eutrophication potential, and land occupation, before and after the economic allocation of the ecosystem service, such as preservation of biodiversity, conservation of landscapes, contribution to the socio-economic viability of many rural areas, and co-products (e.g., milk production and transformation) provided by each system.

Chaplin-Kramer et al. (2017) used the ES method to alter (replace or add new data to) the LCI of the traditional LCA impact pathways. The study presents advances for LCA that integrate spatially explicit modelling of land change and ecosystem services in a Land-Use Change Improved (LUCI)-LCA. Key elements of life cycle inventory in the agricultural stage of an attributional LCA were substituted with outputs from predictive land-change modelling (LCM) and spatially explicit ecosystem services modelling using the InVEST software.

On the other hand, Bruel et al. (2016) used the outputs of the LCI phase directly in the bio economic models of ES to evaluate changes in the supply of ES. Then, to obtain monetary indicators of ES, they evaluate the loss of benefits through monetization techniques.

Liu et al. (2020) applied the methodology of Rugani et al. (2019) abbreviated as ES-LCIA, that integrates the ES cascade framework within the LCIA cause-effect chain to assess the impacts on ES provisioning. The method was demonstrated using a case study of rice farming in the United States, China, and India. Four ES are considered, namely carbon sequestration, water provisioning, air quality regulation, and water quality regulation.

Some papers expanded the system boundaries of LCA to account for the supply and demand of ES. Liu et al. (2019) and Liu and Bakshi (2019) extend the framework of conventional process LCA to assess and include ecosystem models through techno-ecological synergies in life cycle assessment (TES-LCA). It accounts for local and absolute environmental sustainability by comparing the demand and supply of ES at multiple spatial scales. Schaubroeck et al. (2013)., expanded the boundaries of a product's life cycle which were first limited to the human/industrial system, to also the natural environment through a framework called Techno-Ecological Systems (TES). Other approaches proposed (1) using the LCI as an input for a bioeconomic model of ES Bruel et al. (2016). Furthermore, there are papers that build upon the results of LCA to obtain a value of an ES. Morales et al. (2020) carries out an LCA (water and energy depletion and climate change) on combined heat and power plant systems and calculates through the GHG emissions the amount of hectares needed for carbon storage (ES carbon sequestration). Xue et al. (2014) calculated the effect on the ES carbon sequestration in different tillage systems on the basis of an LCA study providing GHG emissions (carbon footprint). A carbon sustainability index was calculated as a final aggregated result. While most papers proposed or attempted to integrate parts of both LCA and ESA methodologies, two papers conducted LCA and ES assessments independently. Briones-Hidrovo et al. (2020) performed a quantitative analysis of the net environmental performance of hydropower, by integrating LCA and ES at the end of the cause-effect chain. The ReCiPe LCIA method is used, and 17 ES are tackled. A first screening of all impact categories and methods to avoid double counting is recommended. The net environmental performance is calculated based on the potential benefits (results of ESA) minus the potential burdens (results of LCA). Monetarisation is proposed to all categories of ESA and LCA. Viglia et al. (2013), on the other hand, integrated different environmental accounting methods with life cycle and ecosystem services assessment through multicriteria assessement to investigate the interplay of human activities and nature conservation in a natural national park in Poland.

4.2. Challenges for integrating ESA and LCA

Multiple aspects must be considered when integrating ES and LCA, those mostly arise from the different scopes of the two methods.

- Spatial and temporal scales used in ESA and LCA. The two methods (ESA and LCA) have been traditionally developed for different spatial and temporal scales. While LCA has been conceived to address cause-effect chains at global scale, ESA deals with varying spatial scales, depending of the case study. Furthermore, the temporal consideration in ESA is also case-study specific while LCA relies on standardized time horizons (Maia de Souza et al., 2018; Othoniel et al., 2016; Pavan and Ometto, 2018; Rugani et al., 2019).
- Final beneficiary of ES and scope of LCA. Depending of the ES under study, the socio-economic system benefiting from the ES might be limited to a narrowed number of final beneficiaries (i.e. local community). On the other hand, LCA presents a more global perspective with some midpoint (e.g. climate change) or endpoint (e.g. human health) impact categories (affecting society as a whole (Liu et al., 2020; Othoniel et al., 2016)).
- Metric for aggregating ES. The aggregation of multiple ES requiring making use of a single metric amongst ES, some ES can be ignored when it is not possible to express them into a common metric (Zhang et al., 2010a; Bakshi and Small, 2011). The commonly used aggregation metric for ES is their value expressed in monetary terms (see 3.2.2), yet several papers did not monetize ES due to concerns about under- or over-estimations of value (Cao et al., 2015). This limitation also holds for the aggregation of LCA impact cetegories.
- Compatibility with of ESA methods in LCA softwares. Some of the methods combining ESA and LCA 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). This is especially the case for ESA relying on dynamic models (e.g. Arbault et al., 2014) to include interconnections amongst ES in their evaluation (e.g. change or consumption of one ES affecting another ES).
- Overlap amongst ES and LCA impact categories. The ESA results and LCA results might overlap depending of the selected ES and impact categories. This leads to double-counting when combining and interpreting the results (Blanco et al., 2018; Briones-Hidrovo et al., 2020; Callesen, 2016; Rugani et al., 2019).
- Data limitations. Some papers encountered problems with the reliability of ES indicators compared to LCA 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).

5. Classifying integration approaches

Three main categories of ES-LCA integration approaches are identified according to De Luca et al. (2022) and considering further suggestion from Moraga et al. (in preparation) (**Figure 5**). In type A, "post-analysis", LCA and ESA are performed independently in parallel without any integration of the methods or aggregation of the results. In this case, there is no actual integration between the different phases of the methodologies nor a quantitative integration of both results. In type B, "aggregation", LCA and ESA are performed independently in parallel without any integration but the results are aggregated by introducing an additional quantitative or qualitative step (e.g. multicriteria decision aid analysis). The integration always occurs at the endpoint of the cause-effect chains. In type C, "driving method", ESA is integrated in LCA (i.e. including ES aspects in the goal and scope definition, LCI, LCIA and/or results interpretation) with LCA (i.e. the driving method) providing the main structure of the analysis.

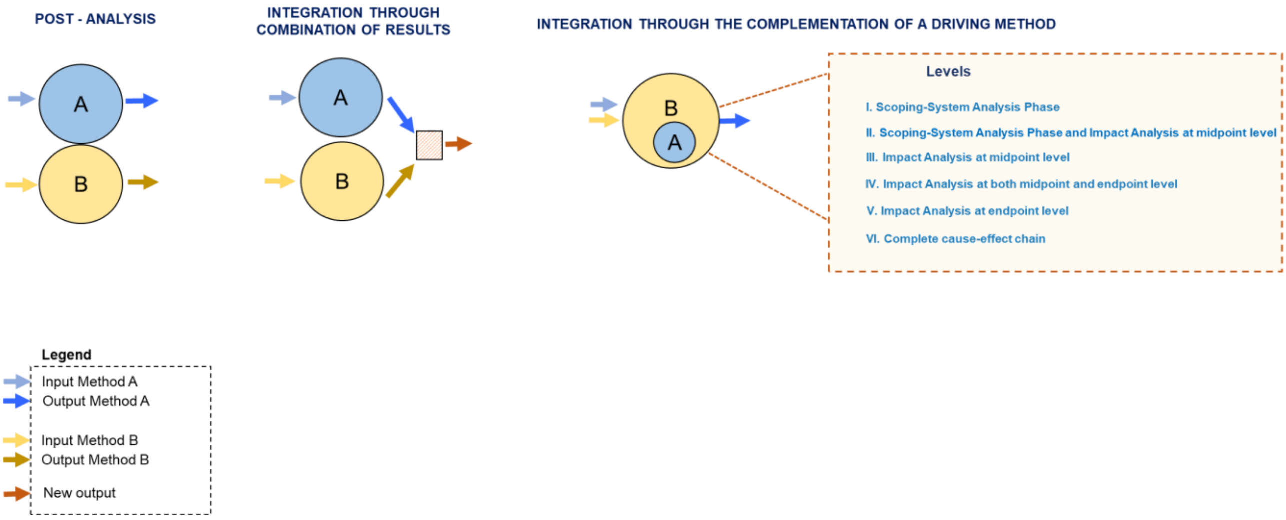

Figure 5 Integration approaches for Ecosystem Services Assessment and Life Cycle Assessment identified from the literature review.

Table 1 (adopted from the SI of De Luca et al. (2022) shows for each of the 25 studies to which integration category they belong. As a result of the literature review, it appears that the driving method is by far the most frequently used method for integrating ESA and LCA (**Table 1**). Most studies use LCA as the driving method to integrate ESA while only two studies follow ESA as driving method. On the other hand, ESA is integrated in LCA through different ways depending of the study: some studies consider ES as an additional midpoint impact category of LCA (e.g. Blanco et al., 2018; Cao et al., 2015; Arbault et al., 2014), other studies modify the LCI using allocation methods to integrate ES into LCA (e.g. Boone et al., 2019; Bragalio et al., 2020) and a few studies adapt the LCI with outputs from ES quantification tools (e.g., Chaplin-Kramer et al., 2017). In the integration category 'driving method', applied to 22 individual papers, it is also notable that LCA allows the connection of a human activity with ES mainly in the context of land occupation/transformation as a stressor. Furthermore, Table 1 shows that economic allocation or monetisation are strategies that are often used to perform the aggregation, mainly at the endpoint level.

As mentioned in De Luca et al. (2022), each of the integration approaches can have potential advantages and limitations. The *post-analysis* category is a simple integration technique as there is no quantification needed and avoids any potential conflicts regrding matching of methods having a different structure and scope, however, it can be debated if it can be considered as an actual integration of two or more methods and there is a risk of double counting (Muazu et al., 2021). Furthermore, it might lack the ability to give clear guidance to decision-makers and it can be time-consuming as two independent assessments need to be carried out. The integration through the *combination of results* has similar benefits (it can combine methods with different structures and scopes) and burdens (time-consuming and risk of double-counting) as post-analysis, however proves to be more informative to decision makes as it is able to aggregate results of separately performed studies through the use of indicators or other metrics, although the choice of the latter is not straightforward

(Linkov et al., 2017, Bakshi and Small, 2011, Zhang et al., 2010a). 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. Although the application of the integrated methods seems to be less time-consuming than when the methods are applied in parallel, its development is more complex (e.g. developing new cause-and-effect pathways and characterisation factors). If the integration method is not well developed, there is a risk of double counting (Muazu et al., 2021).

The selection of a particular integration category (Figure 5) seems to be depending on the final aim and scope of the study and has its particular consequences in the collection of data, impact assessment calculations, number of assumptions, system boundary and functional unit definition, and/or visualisation and interpretation of results of a comprehensive study.

Table 1 Overview of the different studies applying each of the three main methods to integrate ESA and LCA.

Method	Studies	Explaination of the integration
Post-analysis	Viglia et al. (2013)	Independent assessments. Joint interpretation of the results without a quantitative or qualitative integration.
Aggregation of the results	Briones-Hidrovo et al. (2020)	Independent assessments. Aggregated results from LCA and ESA are combined to calculate the net ecosystem value, expressed in monetary units.
	Xue et al. (2014)	Independent assessments. GHG emissions obtained from LCI inventory and used to calculate the carbon footprint. Valuation of carbon sequestration. Calculation of carbon sustainability index.
Complementation of a driving method	Arbault et al. (2014)	Driving method: LCA. Calculation of CFs at midpoint (physical units) and endpoint level (monetary units) of LCA. The CFs are calculated using ES models.
	Blanco et al. (2018)	Driving method: LCA. CFs at the midpoint level and monetary valuation method is used to normalize and weight the total ES losses or gains.
	Boone et al. (2019)	Driving method: LCA. ES supply scores are used to calculate allocation factors for LCA.
	Bragaglio et al. (2020)	Driving method: LCA. ES value is used for economic allocation of products (different monetary valuation techniques used).
	Brandão and Milà i Canals (2013)	Driving method: LCA. Calculation of CFs at the midpoint level.
	Bruel et al. (2016)	Driving method: ESA. Data from the LCI is used to complement the inputs for a bio-economic model.
	Cao et al. (2015)	Driving method: LCA. Impacts on ES at midpoint (physical units) and endpoint (monetary units) level in LCA.
	Chaplin-Kramer et al. (2017)	Driving method: LCA. Outputs from InVEST models are used to modify or add new data of the LCI.
	Jeswani et al. (2018)	Driving method: LCA. Impacts on ES at midpoint in LCA. Calculation of land occupation in LCI based on national data.
	Liu & Bakshi (2019)	Driving method: LCA. Modification of LCI and calculation of a sustainability metric based on the demand ans supply of ES.
	Liu et al. (2018)	Driving method: LCA. Calculation of CFs based on carbon dynamics model. No clear about modification on the inventory.
	Liu et al. (2019)	Driving method: LCA. Modification of LCI and calculation of sustainability metric based on the demand ans supply of ES. No ES values. Demand and supply at different scales. Allocation.
	Liu et al. (2020)	Driving method: ESA. Data from the LCI is used to complement the inputs for an agro-environmental model. Monetary valuation at the endpoint.
	Morales et al. (2020)	Driving method: Ecological funds. Results from LCA are being used to make an analysis of ecological fund flows.
	Müller-Wenk & Brandão (2010)	Driving method: LCA. CFs at midpoint level.
	Muñoz et al. (2014)	Driving method: LCA. Impacts on ES at midpoint level in LCA. Modification LCI inventory.
	Núñez et al. (2013)	Driving method: LCA. Impacts on ES ar midpoint level in LCA. Modification LCI inventory.
	Othoniel et al. (2019)	Drlving method: LCA. Calculation of CFs at midpoint (biophysical) and endpoint (monetary) level of LCA. The CFs are calculated using ES models.
	Saad et al. (2013)	Driving method: LCA. CFs at midpoint level

Schaubroeck et al. (2013)	Driving method: LCA. Modification of the LCI inventory
van Zelm et al. (2018)	Driving method: LCA. CFs at midpoint level.
Zhang et al. (2010a)	Driving method: LCA. Impacts at midpoint and endpoint level. Modification of the inventory.

6. Conclusion: identifying suitable integration techniques in the context of SUMES

This deliverable provides an overview of the state of integration of two specific methods (life cycle assessment and ecosystem services assessment) as both the local and global effects of human activities at sea (in this case offshore wind energy, cfr D4.1) need to be quantified and aggregated to facilitate understanding of sustainability results and better support decision-making related to the development of a sustainable blue economy.

By conducting a literature review, we identified different integration approaches, which also served as a basis to propose a classification of integration approaches. 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 the most recurrent type of integration, as it is found to be most 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 doublecounting impacts, therefore it is crucial to avoid any overlap when selecting relevant parameters and indicators (De Luca et al., 2022). The driving method is in most cases the LCA method, which is altered or complemented with part of the ESA method, either at the scoping phase, the life cycle inventory, midpoint and/or endpoint impact assessment or through the entire cause-and-effect chain. However, a full integration, in which all the components of one methodology are integrated with all the components from the other methodology, has not been performed so far (probably because of different complex stuctures of the methods). It can be questioned as well if this is needed, because a partial integration may already serve the goal to comprehensively asses environmental and socio-economic costs and benefits of human activities on the ecosystems, as intended in SUMES. Also notable is that most of the LCA-ES papers assessed mainly impacts on terrestrial ecosystems and not so much aquatic ecosystems, mainly because the development of impact pathways and CFs for sustainability assessment in the marine context needs further research (Woods et al., 2016). Therefore, as we deal with aquatic case studies in SUMES, it is predicted that we will also need to calculate additional CFs, others than those already available in classicial LCIA methods such as IPCC 2013 GWP and ReCiPe 2016.

Aggregation of indicators towards a single score is what many papers aimed for (e.g. Liu et al. (2020) and Blanco et al. (2018) for ESA; Roesch et al. (2020) for the current status of the normalization and weighting in LCA), i.e. indicator results are combined because they are presented in compatible units, where normalisation and/or weighting preceded (Gan et al., 2017). Providing a single score can be helpful in interpreting the sustainability results, certainly when compared to alternatives (e.g. offshore wind energy versus fossil-based energy). While there is no consensus on the selection of a specific method to aggregate LCA impact categories, the European Commission recommends a set of weighting factors to aggregate the environmental impact categories from its LCA method, the Product Environmental Footprint (PEF) (Sala et al., 2018). However, major drawbacks are the increased uncertainty and transparency of the single score and the fact that strong sustainability (no compensation allowed across different impacts) cannot be reached (Kalbar et al., 2017). While weighting factors for aggregating ES results with LCA impact categories have not been discussed yet, the monetary valuation of ES and LCA impact categories is promising to obtain aggregated monetary scores, combining results of both ESA and LCA. Different sets of monetary conversion factors for LCA midpoint categories have been proposed (e.g. Environmental Prices Handbook, the Handbook on the External Cost of Transport, Ecovalue, Ecotax or the Monetization of the MMG method[1]). Each monetary conversion factor relies on a specific approach[2] monetizing an LCA impact category from one of the following perspective: the instrumental value of ecosystem for humans or, the intrinsic value of ecosystems

[1] Method developed by OVAM for the monetization of indicators complians with CEN and CEN+ guidelines (Wille et al., 2017).

[2] e.g. abatement costs approach for the impact category global warming potential; future costs for extraction of raw materials approach for the impact category abiotic depletion of non-fossil resources.

(Rea and Munns, 2017). So far, there is no common agreement on the selection of a specific monetary conversion coefficient for each LCA impact category and hence, it is important to provide transparency on the coefficients and their underlying monetization approaches. While the final results are varying depending on the approach selected to derive the monetary conversion factors, it is also important to consider the country perspective (purchasing power effect) and the reference year in which monetary units are expressed (effect of inflation) (cfr. Arendt et al., 2020 for a comprehensive review of the monetization methods). If performed in a transparent way, monetizing ESA and LCA results has a strong potential to combine comprehensively the outputs of these two methods. In the upcoming D4.3, the sustainability assessment framework will be applied to the showcase (offshore wind energy from the farm) and the aggregation technique will be thoroughly explained and elaborated in this deliverable.

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