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Environmental sustainability of conventional and organic farming: Accounting for ecosystem services in life cycle assessment

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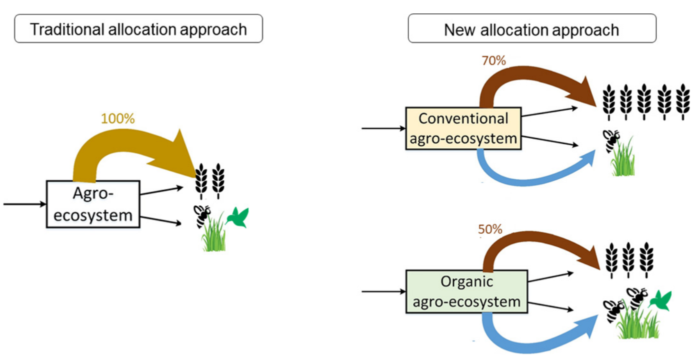
HIGHLIGHTS

- LCA does not account for all ecosystem services (ES) supplied by agroecosystems.
- This leads to varying LCA results when comparing organic and conventional food.
- The environmental impact should be allocated among all ES supplied by agroecosystems.
- An allocation approach based on the capacity to deliver ES is proposed.
- The approach allows to compare the impact of conventional and organic food systems.

GRAPHICAL ABSTRACT



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ABSTRACT

Today, there is an ongoing debate about the environmental sustainability of the products of organic farming. To compare the performance of conventional and organic farming systems regarding environmental impact and productivity, the comprehensive environmental assessment tool ‘life cycle assessment’ can be used. The lower crop yields attained by organic systems compared to conventional farming systems might, however, outweigh the benefits of the use of more environmental-friendly practices when evaluating the environmental impact per product unit. Although these practices are beneficial for the environment, which is reflected in the delivery of a range of ecosystem services (ES), the focus is traditionally put only on the (harvested) product. Because the agricultural product involves actually a bundle of ES, the impact should be allocated among the whole output

Abbreviations: CICES, Common International Classification of Ecosystem Services; ES, ecosystem services; ES_{cul}, cultural ecosystem service; ES_{prov}, provisioning ecosystem service; ES_{reg}, regulating and maintenance ecosystem service; EU, European Union; $f_{prov,con}$, allocation factor of provisioning ecosystem services for a conventional agro-ecosystem; $f_{prov,org}$, allocation factor of provisioning ecosystem services for an organic agro-ecosystem; $f_{reg,con}$, allocation factor of regulating and maintenance ecosystem services for a conventional agro-ecosystem; $f_{reg,org}$, allocation factor of regulating and maintenance ecosystem services for an organic agro-ecosystem services for an organic system; J_{ex}, Joules of exergy; LCA, life cycle assessment; $n_{prov,con}$, selected number of provisioning ecosystem services for a conventional agricultural system; $n_{prov,con}$, selected number of provisioning ecosystem services for an organic agricultural system; $n_{reg,con}$, selected number of regulating and maintenance ecosystem services for a conventional agricultural system; $n_{reg,con}$, selected number of regulating and maintenance ecosystem services for an organic agricultural system; RF, resource footprint; RF_a, allocated resource footprint; $RF_{a,con}$, allocated resource footprint related to a conventional agricultural system; $RF_{a,org}$, allocated resource footprint related to an organic agricultural system; RF_{con}, resource footprint related to a conventional agricultural system; RF_{org}, resource footprint related to an organic agricultural system; Y_{con}, yield under conventional farming practices; Y_{org}, yield under organic farming practices; *(capacity to supply* $ES_{prov})_{av,con}$, average capacity to deliver provisioning ecosystem services for a conventional agricultural system; *(capacity to supply* $ES_{prov})_{av,org}$, average capacity to deliver provisioning ecosystem services for an organic agricultural system; *(capacity to supply* $ES_{reg})_{av,con}$, average capacity to deliver regulating and maintenance ecosystem services for a conventional agricultural system; *(capacity to supply* $ES_{reg})_{av,org}$, average capacity to deliver regulating and maintenance ecosystem services for an organic agricultural system.

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of an agricultural system. In this study, we propose 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. Applying these allocation factors, we demonstrate that for about half of the studied food products (including maize, potato), organic farming has clear environmental benefits in terms of resource consumption in comparison to conventional cultivation methods. This allocation approach allows a more complete comparison of the environmental sustainability of organically and conventionally produced food.

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1. Introduction

For several decades, agricultural intensification was the answer to meet the growing demands for food, feed and fuel (Foley et al., 2011). Because intensification is characterized by land management aiming a maximization of productivity, often through the use of agrochemicals (e.g., fertilizers, pesticides), irrigation and mechanization, it also contributed to increased resource consumption (e.g., water, energy, minerals), water and soil degradation, and widespread pollution (Foley et al., 2011; National Research Council, 2010). Furthermore, intensive agricultural management is one of the drivers for, among others, biodiversity loss, erosion, changes to nitrogen and carbon cycles, and regime shifts in hydrological cycles, and in turn, for degradation of several ecosystem services (ES) (Lorenz and Lal, 2016; Meier et al., 2015; Sandhu et al., 2010a).

Organic agriculture is often put forward as a solution to reduce the negative impact of agriculture on the environment (Sandhu et al., 2010a; Seufert et al., 2012). It refers to systems targeting food production with minimal adverse impacts on ecosystems, animals and humans (National Research Council, 2010; Seufert et al., 2012). Instead of using synthetically produced inputs (fertilizers, pesticides), organic farm management practices rely on and benefit from biological cycles by e.g., appropriate selection of crop rotations and cover crops (soil fertility), well-considered choices regarding the timing of sowing and mechanical cultivation (weed control), and making use of biological control and natural pesticides (pest control) (Gomiero et al., 2011; Meier et al., 2015). The impact of the adapted management is often reflected in a reduction of greenhouse gas emissions and a better performance in terms of (but not limited to) biodiversity, water use efficiency, soil, water and air quality, and a variety of ES (Gomiero et al., 2011; Hole et al., 2005; Kremen and Miles, 2012; Lorenz and Lal, 2016). However, on average, a yield reduction of 20–40% is reported for arable crops in organic systems compared to conventional systems; with differences that strongly depend on site and system characteristics (Fedele et al., 2014; Gomiero et al., 2011; Kremen and Miles, 2012; Meier et al., 2015; Seufert et al., 2012; Winqvist et al., 2011). Thus, organic farming systems usually require more land to produce the same amount of output (e.g., food) and therefore, per product unit, their better environmental results might be cancelled out (Lorenz and Lal, 2016; Meier et al., 2015).

A comparison of the performance of conventional and organic farming systems regarding environmental impact and productivity, is ideally based on comprehensive environmental assessment tools, such as the widely applied technique life cycle assessment (LCA) (Fedele et al., 2014; Meier et al., 2015). LCA is a standardized tool to estimate the environmental burden of products (i.e. goods and services) from a lifecycle perspective (ISO, 2006). In agricultural context, often a cradle-to-farmgate analysis is conducted, then, a mass-based functional unit is regularly considered as most appropriate (Caffrey and Veal, 2013; Fedele et al., 2014). By doing so, the yield seems to be a crucial factor and might even be decisive (Boone et al., 2016; Noya et al., 2015). In order to conduct an LCA in a comparative context, goal and scope need to be clearly defined, i.e. a strict description of the system boundaries and the functional unit are then required. Taking this into account, Meier et al. (2015) made a review of LCA studies in which the environmental impact of products produced by conventional versus organic agriculture is compared. From this review, it can be deduced that for most LCA studies, lower yields attained in organic farming result in higher environmental impacts when evaluating per product unit.

However, by only focusing on the delivered (i.e. harvested) product units, the multifunctional role of agriculture is neglected: agriculture does not only provide food, feed or fuel (commodities), but also delivers numerous ES to society (non-commodities) in addition to the provisioning services (Meier et al., 2015; Power, 2010; Schader et al., 2012). To deal with multifunctionality in LCA (i.e. next to the function of food production, other functions are provided as well), allocation procedures can be applied. Allocation defines the share of the total environmental burden for each function that the production process fulfills (ISO, 2006). Though, in current agricultural LCAs, the impacts are usually expressed per unit of food/feed/fuel product without allocation between commodities and non-commodities (Meier et al., 2015). Therefore, the impact assigned to the product is overestimated. Because the term ‘agricultural product’ goes beyond the harvested product, and involves also a bundle of other ES delivered by agriculture, the impact should be allocated among the whole output of an agricultural system.

Several allocation procedures exist to partition the inputs and outputs among the various co-products. For instance, allocation can be based on physical relationships (e.g., mass, energy) or using other relationships (e.g., economic value of products) between the delivered products and functions (ISO, 2006). Since the output of an agricultural farming system entails a range of ES, it goes beyond the supply of products or energy and thus the first option cannot be applied. Next, defining the monetary value of specific ES might be challenging, because not all ES are yet represented quantitatively. While representing provisioning services in monetary values is the easiest, quantifying ES that are not available on the market have received less attention. Incomplete knowledge have resulted in often only qualitative representations. Attempts are made to define monetary values by valuation methods such as willingness-to-pay. However, this entails some problems and might lead to conflicting results. Economic valuation reflects human preferences, and as the general public opinion is incomplete, this will affect the estimated monetary values (Zhang et al., 2010). Furthermore, monetary values might fluctuate over time and region, which should be taken into account (Cao et al., 2015). Therefore, another allocation procedure is proposed. The capacity of an ecosystem to deliver a certain number of ES, can be estimated and evaluated. This provision of ES will vary among agro-ecosystems as this is strongly linked to natural conditions (e.g., land cover, hydrology, soil conditions, fauna, climate, …) and human impact (e.g., land use, pollution) (Burkhard et al., 2012). While some ecosystems will mainly focus on a good supply of provisioning ES (e.g., conventional agro-ecosystems), others aim to offer a more broad range of ES (e.g., organic agro-ecosystem) (Sandhu et al., 2010a). Therefore, we propose to rely on ES assessment, which is used to address the extent to which provisioning and other ES are supplied by an agro-ecosystem, as a basis to allocate the environmental impact among the food/feed/fuel product and non-commodities.

In this study, we want to stress that an agriculture product does not only refer to the harvested product but actually entails a range of ES. In particular, we rely on the concept of ES to strike a balance between the productivity and other non-commodities delivered by the agro-ecosystem in order to be able to compare the environmental sustainability of agricultural products produced in conventional and organic farming systems. Therefore, we propose an allocation procedure based on ES to divide the environmental impact over the whole set of agricultural outputs (i.e. provisioning and other ES) delivered by agriculture. As first step, the capacity of the terrestrial agro-ecosystems to supply a particular ES is evaluated and scored according to the approach presented by Burkhard et al. (2012). Next, the allocation factors for a conventional and organic farming system are developed, which are then applied on arable farming systems to compare the resource footprint of conventional and organic agricultural products (food, feed) retrieved from lifecycle databases.

2. Setting the scene

2.1. Organic farming in Europe: the context

Policy support and the growing interest of consumers for organic products, have resulted in a rapid expansion of the total area dedicated to organic agriculture during the past years (European Commission, 2018; European Commission, 2016; Sahm et al., 2013). In the European Union (EU-28), an increase from 5.0 million ha in 2002 to 11.9 million ha in 2016 was recorded, corresponding to 6.7% of the total utilized agricultural area in the EU, and this area is still expected to grow in the coming years (European Commission, 2016; Eurostat, 2018). Regarding 2016, the organically utilized agricultural area consisted of arable land (45%), permanent grassland (44%), and permanent crops such as vineyards and fruit trees (11%) (Eurostat, 2018).

2.2. Towards sustainable farming practices

Policy stimulates agriculture towards more sustainable farming practices. Environmentally sustainable agriculture is, however, a broad concept which can be referred to as "environmentally friendly methods of farming that allow the efficient production of crops or livestock while safeguarding the natural environment, i.e. without damage to the farm as an ecosystem, including effects on soil, water supplies, biodiversity, or other surrounding natural resources" (SAI Platform, 2018; United Nations, 2006). In this sense, organic farming is one of the options available to move in the direction of more sustainable farming systems, as, according to the definition of the United Nations, organic agriculture is a "holistic production management whose primary goal is to optimize the health and productivity of interdependent communities of soil life, plants, animals and people" (United Nations, 2006). Obviously, also conventional farming can implement environmentally friendly management practices (e.g., reduced or no tillage), contributing to an overall increase of the sustainability of the agricultural sector (Gomiero et al., 2011).

In conventional farming, the number of environmentally friendly management practices and the intensity of their application can vary extremely, while in the organic sector, strict regulation and certification mechanisms are in place leaving less choice to the farmer. It is therefore important to define more sharply the context of the work presented in this study. In what follows, we refer to 'conventional farming' as farming that aims for a maximum productivity while meeting the requirements of legislation (e.g., the Common Agricultural Policy) regarding environmental aspects, and refer to 'organic farming' as farming that is strictly regulated and needs to fulfill a list of specified requirements in order to get certified organic agricultural products (IFOAM, 2018). The main differences between organic and conventional arable farming as considered in this study are summarized in Table 1.

3. Material and methods

3.1. Ecosystem service scoring

3.1.1. Selection of ecosystem services related to terrestrial agro-ecosystems

Ecosystem services, which indicate the benefits that humans can obtain directly or indirectly from ecosystems (Costanza et al., 1997), can be classified according to several classification systems. In this study, we rely on the recently updated Common International Classification of Ecosystem Services (CICES) compiled by the European Environment Agency (Haines-Young and Potschin, 2018). They classify ES into 'provisioning', 'regulating and maintenance' and 'cultural' ES, referred to as ES_{prov}, ES_{reg}, and ES_{cul}, respectively. The most recent version is v.5.1, which encompasses both biotic and abiotic ES. CICES consists out of 90 class types referring to the detailed benefits humans can obtain from ecosystems.

In the context of terrestrial agro-ecosystems, relevant ES are pollination, pest, weed and disease control, nitrogen fixation, prevention of soil erosion, etc., which all provide critical inputs to agriculture to ensure the production of other ES and goods (e.g., food) (Gomiero et al., 2011; Kremen and Miles, 2012). The capacity of agro-ecosystems to provide ES is strongly influenced by farming system practices such as tillage, fertilization, crop rotation, etc. (Bai et al., 2018; Kremen and Miles, 2012). Consequently, the supply of ES is different for organic and for conventional systems. It is well established that the delivery of environmental benefits is higher for organic than conventional agriculture, as schematically shown in Fig. 1 (Sandhu et al., 2010a).

In this study, we only focus on biotic ES as the biotic provisioning role might be considered the main function of agricultural systems. Out of the list of CICES, a number of ES_{prov} and ES_{reg} relevant and appropriate to conventional and organic arable crop systems are selected (Table 2). The ES_{prov} refer to the supply of cultivated terrestrial plants and the contribution to genetic diversity by agro-ecosystems. For the selection of ES_{reg}, two main criteria are used. First, we included those ES that have an effect on critical aspects to ensure the provisioning of agricultural food and feed. Second, as emphasized by policy (e.g., the Common Agricultural Policy), agriculture can play an important role in society with respect to the climate. Therefore, those ES belonging to the group 'atmospheric composition and conditions' are also selected. The selection of relevant ES has been verified by a team of experts (ILVO, personal communication; JRC, personal communication).

We have only included ES_{prov} and ES_{reg}, because ES_{cul} are strongly related to human values and behavior, and patterns of e.g., economic organization. Therefore, perceptions of ES_{cul} to be (and the extent to be) of benefit to humans differ more among people or communities than it is the case for ES_{prov} and ES_{reg} (MEA, 2005). Assessment of ES_{cul} is thus rather subjective (Burkhard et al., 2012), and therefore not included in this study. For both conventional and organic agro-ecosystem, the same ES are selected, to cover the largest group of ES relevant for both. The extent to which the ES are supplied will however be different for the two types of agro-ecosystems (Fig. 1). Table 2 presents an overview of the selected ES according to the CICES classification.

Table 1
Main differences between organic and conventional arable farming practices, based on IFOAM (2018), National Research Council (2010), Soil Association (2018), and Viaene et al. (2016).

Organic	Conventional
Fertilization	
Mainly organic manure	Organic manure + high consumption of mineral fertilizers
Maintaining organic matter stock is very important	Organic matter stock is important but not the main priority
Compost is often applied	Limited use of compost
Limited use of natural and non-chemically treated mineral fertilizers	Intensively use of mineral fertilizers (mainly natural products chemically treated)
Crop protection	
Naturally derived plant protection products	Synthetically produced plant protection products
Mainly mechanically	Mainly chemically

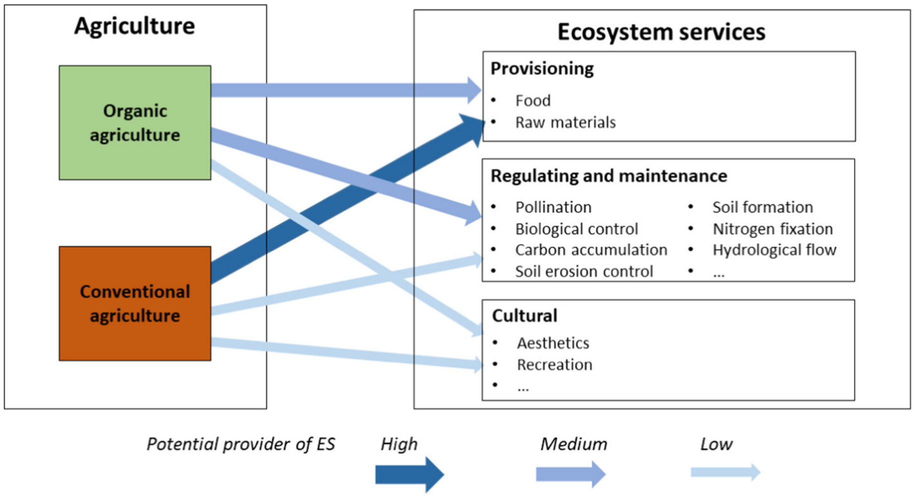

Fig. 1. Supply of ecosystem services by agriculture. Ecosystem services are classified according to CICES. Modified after Sadhu et al. (2010a).

3.1.2. Assigning scores to ecosystem services

The number of ES and the extent to which a particular area can supply ES are influenced by human interventions (e.g., human induced land cover, land use practices, impact of human activities on climate change). In this study, we use the approach presented by Burkhard et al. (2012). They evaluate several land cover classes according to their capacity to deliver a specific bundle of ES within a given time period. Therefore, Burkhard et al. (2012) propose a scale ranging from 0 to 5. Going in increasing order, the numbers correspond respectively to “no relevant capacity to supply the selected ES”, “low relevant capacity”, “relevant capacity”, “medium relevant capacity”, “high relevant capacity”, and “very high relevant capacity”. The distinguished land cover classes correspond to those suggested by the EU program CORINE in which 44 land cover classes are grouped into five main categories: artificial areas,

Table 2
Selected biotic ecosystem services (ES) related to arable conventional and organic agro-ecosystems. Next, also the capacity of the agro-ecosystem to supply provisioning (ES_{prov}) and regulating and maintenance (ES_{reg}) ES is indicated by values ranging from 0 to 5, going from no relevant capacity to very high relevant capacity, respectively. Scoring is based on expert knowledge and literature review (Appendix A).

Section	Division	Group	Class	Code	Example	Conventional	Organic
Provisioning	Biomass	Cultivated terrestrial plants for nutrition, materials or energy	Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes	1.1.1.1	Cereals	5	3
			Fibers and other materials from cultivated plants, fungi, … for direct use or processing	1.1.1.2	Flax	5	3
			Cultivated plants (including fungi, algae) grown as a source of energy	1.1.1.3	Miscanthus	2	1
	Genetic material from all biota	Genetic material from plants, algae or fungi	Seeds, spores and other plant materials collected for maintaining or establishing a population	1.2.1.1	Seed collection	2	4
Regulating and maintenance	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Control of erosion rates	2.2.1.1	The capacity of vegetation to prevent or reduce the incidence of soil erosion	0	1
			Hydrological cycle and water flow regulation	2.2.1.3	The capacity of vegetation to retain water and release it slowly	2	2
		Lifecycle maintenance, habitat and gene pool protection	Pollination	2.2.2.1	Providing a habitat for native pollinators	2	3
			Maintaining nursery populations and habitats (Including gene pool protection)	2.2.2.3	Providing nursery habitats	2	4
		Pest and disease control	Pest control (including invasive species)	2.2.3.1	Providing a habitat for native pest control agents	2	4
			Disease control	2.2.3.2	Presence of native disease control agents	2	4
		Regulation of soil quality	Weathering processes and their effect on soil quality	2.2.4.1	Inorganic nutrient release in cultivated fields	2	4
			Decomposition and fixing processes and their effect on soil quality	2.2.4.2	Decomposition of plant residue	2	4
		Atmospheric composition and conditions	Regulation of chemical composition of atmosphere and oceans	2.2.6.1	Sequestration of carbon in biomass and soil	1	2
			Regulation of temperature and humidity, including ventilation and transpiration	2.2.6.2	Evaporative cooling	1	1

agricultural areas, forests and semi-natural areas, wetlands, and water bodies (EEA, 1994). The values assigned to each combination of land cover class and ES are mainly derived by the authors as hypotheses of the potential capacity to deliver ES, based on experience from several case studies conducted in Europe and next verified by experts (Appendix A, Table A1) (Burkhard et al., 2012).

The approach used by Burkhard et al. (2012) is widely used for ES assessment. Therefore, in this study, an approach analogous to that of Burkhard et al. (2012) is used to evaluate the conventional and organic arable agro-ecosystems regarding their capacity to deliver ES. When possible, the scores of Burkhard et al. (2012) are adopted. However, Burkhard et al. (2012) rely on the Millenium Ecosystem Assessment classification for ES (while here, we use the CICES classification), and, the CORINE classification does not make a distinction between organic and conventional farming, so more values were required. Supplemental scoring is based on elaborated case studies related to the delivery of ES by conventional and organic farming systems (Kremen and Miles, 2012; Lorenz and Lal, 2016; National Research Council, 2010; Reganold and Wachter, 2016; Sandhu et al., 2007, 2010a, 2010b, 2015; Schader et al., 2012), after which the scores are verified by ES experts from national and international institutes (ILVO, personal communication; JRC, personal communication). The final scores are presented in Table 2 and in Appendix A, Table A2 (with indication of references).

These scores refer to regular conventional and organic agro-ecosystems. However, a high degree of variability exists within conventional or organic systems. For instance when aiming towards more sustainable farm practices, decisions regarding implementation of semi-natural elements such as vegetated field margins, hedgerows, etc., can be taken. Next, farmers need to make a range of choices regarding fertilization, crop rotation, etc. This will all influence the capacity to supply ES, and, consequently, the scores in Table 2. Therefore, per case study, the values are critically examined.

3.2. Case studies

3.2.1. Selection of products

We compare the environmental impact of products of which production data is available for both, conventional and organic farming. Agricultural food products produced by both conventional and organic systems, are e.g., available in the life cycle inventory databases Ecoinvent (Swiss Centre for Life Cycle Inventories, 2015) and Agribalyse (INRA, 2018; Koch and Salou, 2013). A selection of products is made (Table 3). In Ecoinvent, Swiss or global yield averages are available; we used the Swiss products as reference. Agribalyse is a French life cycle inventory database for which average yield data over several regions in France are used. The motivation for the selection as well as background information related to data collection are presented in Appendix B.

The yield data in Table 3 are generic and represent an average over several years (Appendix B), but they can differ in specific case studies. For instance, regarding the meta-analysis conducted by Seufert et al. (2012), the average yield ratio of organic to conventional is 75%. However, they emphasize that results are highly contextual and are strongly dependent on the management. In this study, we have retrieved data from life cycle inventory databases as the purpose of this study is to present a conceptual framework. The selection of ES as well as the scores assigned to them, are carried out from the viewpoint to represent regular conventional and organic agro-ecosystems in general. Furthermore, the values adopted from Burkhard et al. (2012) are based on Western-Europe case studies and do not represent any specific situation. Also the production processes in the life cycle inventory databases are meant to represent general farms. Therefore both the data inventory and scores of ES are characterized by the same level of detail. It is therefore important to keep in mind that if selecting and scoring of ES is performed for a specific case study, also the data inventory (including yield) need to be changed accordingly in order to calculate the environmental impact.

3.2.2. Life cycle impact assessment method

To account for the resource footprint of the production of food products, the Cumulative Exergy Extraction from Natural Environment (CEENE (2013)) method is used. This method quantifies the overall resource consumption by accounting for all exergy extracted from nature contained in the natural resources used throughout the supply chain (Alvarenga et al., 2013; Dewulf et al., 2007). The exergy of a resource is the maximum amount of useful work that can be obtained from this system or resource when it is brought to equilibrium with the surroundings through reversible processes in which the system is allowed to interact only with the environment. So exergy takes into account both the quality and the quantity of resources and is expressed in one common unit, i.e. joules of exergy (J_{ex}) (Dewulf et al., 2007). CEENE covers the following groups of natural resources: fossil fuels, nuclear resources, abiotic renewable resources (wind, geothermal and hydropower), metal ores (e.g. aluminum in bauxite), minerals, water resources, and land and biotic resources and atmospheric resources (Alvarenga et al., 2013; Dewulf et al., 2007).

4. Results and discussion

4.1. Methodological development to calculate an ecosystem services allocated resource footprint

The inputs related to farm management practices do not only result in the provisioning of the harvested product, but also contribute to the delivery of a range of other ES. Consequently, the environmental impact should not be fully allocated to the harvested product, which is, however, mostly done in agricultural LCAs when expressing the impact per product unit. In contrast, the environmental impact must be allocated among all outputs delivered by the agro-ecosystem.

In this study, we distinguish two groups over which the impact should be allocated: on the one hand, the supplied ES_{prov}, and on the other hand, the ES_{reg}. Therefore, we need to compute allocation factors

Table 3
Selected arable products of the databases Ecoinvent and Agribalyse with corresponding yield under conventional and organic farming practices, Y_{con} and Y_{org}, respectively.

Product	Yield under conventional farming (Y_{con}) (kg ha^{-1})	Yield under organic farming (Y_{org}) (kg ha^{-1})	Ratio Y_{org}/Y_{con} (%)	Database
Barley grain	6,828	4,153	61	Ecoinvent
Carrot	64,600	42,500	66	Agribalyse
Faba bean	5,070	2,500	49	Agribalyse
Maize grain	9,315	7,777	83	Ecoinvent
Maize silage	61,457	49,166	80	Ecoinvent
Potato	41,001	22,908	56	Ecoinvent
Protein pea	3,840	3,044	79	Ecoinvent
Rape seed	3,113	2,023	65	Ecoinvent
Rye grain	7,540	4,172	55	Ecoinvent
Triticale grain	5,200	3,000	58	Agribalyse
Wheat grain	6,425	4,069	63	Ecoinvent

that indicate the fraction of the impact that is assigned to ES_{prov} ($f_{prov,con}$ and $f_{prov,org}$ for a conventional and organic system, respectively). Consequently, the rest of the environmental impact is allocated to ES_{reg}, indicated by the allocation factors $f_{reg,con}$ (conventional system) and $f_{reg,org}$ (organic system). This relationship is indicated by Eqs. (1) and (2). The allocation approach is schematically represented in Fig. 2.

$$f_{prov,con} + f_{reg,con} = 1 \quad (1)$$

$$f_{prov,org} + f_{reg,org} = 1 \quad (2)$$

4.1.1. Calculation of allocation factors

In this study, we propose to allocate the environmental impact according to the capacity of an agro-ecosystem to deliver ES_{prov} and ES_{reg}. Because the capacity to supply ES_{prov} and ES_{reg} is different for a conventional and organic system (Fig. 1), the allocation factors will also be dissimilar, although the same procedure to compute the allocation factors is applied. First, relying on the scores assigned to the selected ES_{prov} and ES_{reg} (Table 2), we compute the average capacity to deliver ES_{prov}, called *(capacity to supply* $ES_{prov})_{av}$ for a conventional and organic system. The calculation is presented for a conventional system (Eq. 3). In a similar way, it can be calculated for an organic system. n_{prov} refers to the number of ES_{prov} selected in this study, corresponding to 4 for both the conventional and organic system.

$$(\text{capacity to supply } ES_{prov})_{av,con} = \frac{(\sum \text{capacity to supply } ES_{prov})_{con}}{n_{prov,con}} \quad (3)$$

Analogous to Eq. (3), the average capacity to supply ES_{reg} (*(capacity to supply* $ES_{reg})_{av}$) is calculated relying on the total capacity to supply ES_{reg} and n_{reg} (i.e. the number of ES_{reg} included in the assessment).

Then, the allocation factor indicating the fraction of the environmental burden assigned to the ES_{prov} for a conventional systems ($f_{prov,con}$) is calculated by Eq. (4).

$$f_{prov,con} = \frac{(\text{capacity to supply } ES_{prov})_{av,con}}{(\text{capacity to supply } ES_{prov})_{av,con} + (\text{capacity to supply } ES_{reg})_{av,con}} \quad (4)$$

Analogous, the allocation factor $f_{prov,org}$ for the ES_{prov} regarding an organic system can be calculated.

Based on Eqs. (1) and (2), $f_{reg,con}$ and $f_{reg,org}$, respectively, are then calculated.

For a conventional system, $f_{prov,con}$ and $f_{reg,con}$ equal 0.69 and 0.31, respectively. For the organic system; the allocation factors amount 0.49 and 0.51 for $f_{prov,org}$ and $f_{reg,org}$, respectively. So for a conventional system, two third of the input should be allocated to the ES_{prov}, being of main importance for a conventional farming system. In contrast, for an organic system, more than half of the inputs should be assigned to ES_{reg}, reflecting that the focus of organic farming is to deliver a range of ES_{reg} as well. The values are summarized in Table 4.

4.1.2. Calculation of the resource footprint

In this study, we calculate the resource footprint (RF) to assess the resource consumption related to the production of agricultural products. Relying on the allocation approach according to the ES theory, the new (allocated) RF (RF_a) of one agricultural product unit corresponds to the environmental impact assigned to ES_{prov} and is calculated by Eq. (5) regarding a conventional system and Eq. (6) for an organic system. RF_{con} and RF_{org} refer to the RF of a product without allocation between ES_{prov} and ES_{reg}.

$$RF_{a,con} = RF_{con} \cdot f_{prov,con} \quad (5)$$

$$RF_{a,org} = RF_{org} \cdot f_{prov,org} \quad (6)$$

4.2. Allocated resource footprint

The RF is determined by the life cycle impact assessment methodology CEENE. For almost all crops discussed in this study, the standard RF (i.e. 100% allocation to the product) is higher for one kg of product produced by organic farming practices compared to production by conventional practices (Table 5). For instance, one kg barley entails a resource consumption of 35.3 MJ_{ex} under conventional farming, compared to 54.3 MJ_{ex} under organic farming (Table 5), due to the lower Y_{org} than Y_{con} (Table 3). When applying Eq. (5) to obtain the allocated RF, RF_{con} is multiplied with 0.69 ($f_{prov,con}$); the RF is thus reduced by 31% and

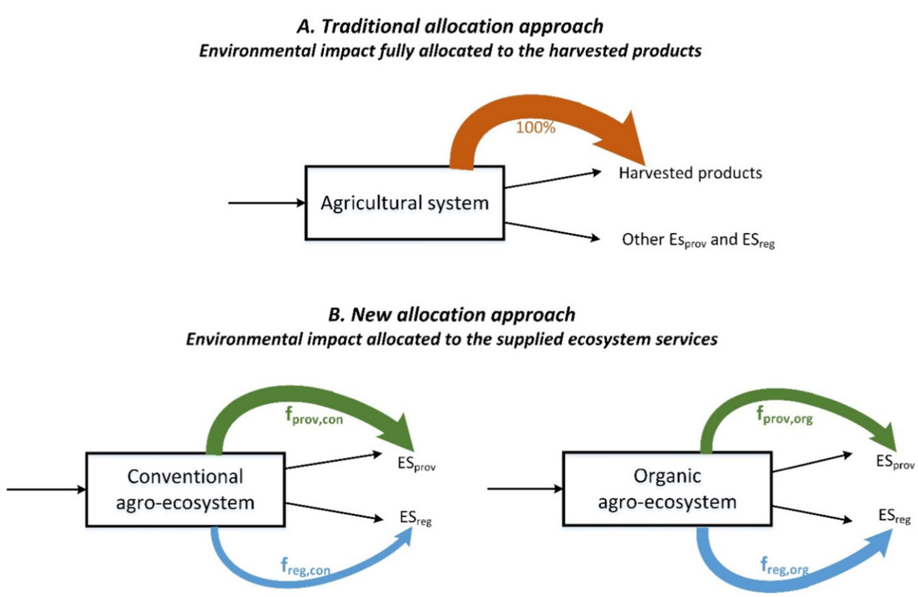

Fig. 2. Visualization of the allocation of the environmental impact to the output of an agro-ecosystem. The input includes provisioning (ES_{prov}) and regulating and maintenance (ES_{reg}) ecosystem services. A: allocation as traditionally applied in agricultural LCAs. B: new allocation approach for a conventional and an organic agro-ecosystem. The thickness of the arrows represents the share of the environmental burdens allocated to the products. $f_{prov,con}$ and $f_{reg,con}$ are the allocation factors indicating the fraction of the environmental burden allocated to ES_{prov} and ES_{reg}, respectively, for a conventional system. Analogous, $f_{prov,org}$ and $f_{reg,org}$ are the allocation factors for the organic system.

Table 4
Allocation factors f_{prov} and f_{reg} for conventional and organic farming systems and intermediate results. f_{prov} and f_{reg} indicate the share of the environmental impact that needs to be assigned to provisioning (ES_{prov}) and regulating and maintenance (ES_{reg}) ecosystem services, respectively.

		Conventional	Organic
ES_{prov}	n_{prov}	4	4
	$\sum$(capacity to supply ES_{prov})	14	11
	(capacity to supply $ES_{prov})_{av}$	3.50	2.75
ES_{reg}	n_{reg}	10	10
	$\sum$(capacity to supply ES_{reg})	16	29
	(capacity to supply $ES_{reg})_{av}$	1.60	2.90
Allocation factors	$\boldsymbol{f_{prov}}$	**0.69**	**0.49**
	$\boldsymbol{f_{reg}}$	**0.31**	**0.51**

only amounts 24.2 MJ_{ex} kg^{-1}. In the case of organic farming, the $RF_{a,org}$ equals 26.4 MJ_{ex} kg^{-1}. The allocation approach according to the ES concept contributes to a more balanced way to compare the RF related to conventional and organic barley production, as we now only account for the environmental impact that is related to the provisioning function of the agro-ecosystem. Even when allocation is performed, a higher RF is recorded for the organic product, but the difference in environmental impact between organic and conventional becomes smaller. The ratio of the RF of organic to conventional farming decreased from 154 to 109% (Table 5).

Because $f_{prov,org}$ is lower than $f_{prov,con}$, a higher reduction of the RF of organic products can be noticed. Consequently, the ratio of the $RF_{a,org}$ over $RF_{a,con}$, which gives an indication of the difference between RF of conventional and organic products, is smaller than the ratio of RF_{org} over RF_{con}. Only for two crops, the difference in RF between organic and conventional cultivation is less than 20%, namely for carrot (RF_{org} 19% lower than RF_{con}) and potato (RF_{org} 6% higher than RF_{con}). In contrast, the difference between $RF_{a,org}$ and $RF_{a,con}$ is smaller than 20% for 7 out of 11 products (Table 5). Through ES based allocation, we can deduce that the difference in environmental impact of conventional and organic products is actually smaller than generally accepted. However, the allocation procedure does not result in the conclusion that organic farming is always favored with respect to environmental sustainability. The standard (unallocated) RF is lower for almost all crops cultivated under conventional farming than when organically produced. However, the RF_a is for almost half of the crops lower when produced by organic instead of conventional practices (Table 5). Thus although for many crops less inputs of agro-chemicals and fuel are associated with organic farming practices, the RF_a is not for all cases lower for organic practices, which emphasizes the important effect of the yield on the impact results and the importance of efficient use of land resource when assessing the environmental sustainability.

Despite the fact that the allocation factors weigh the RF in terms of ES and will reduce the RF_{org} more than RF_{con}, cereals under conventional agriculture are preferable to organic cereals (except maize) regarding RF_a (Table 5). Whereas the resource consumption per area is quite similar for those cereals (Appendix C, Table C1), the lower yields obtained under organic practices (up to 45% lower) strongly affect the impact results (Table 3). In other words: the additional ES_{reg} provided by organically growing cereals do not outbalance the lower yield. An increase of Y_{org} or a reduction of RF_{org} for these cereals would be required to balance the impact results. For maize and peas, the differences in RF_a for conventional and organic production are small. While, analogous to the case of the cereals, the resource consumption per area is quite similar for maize and peas produced under conventional and organic agriculture, the differences between Y_{org} and Y_{con} are only about 20%, the minimum observed difference of the considered crops (Table 3). Because the yield is only slightly higher for conventional farming, it does not cancel out the additional ES_{reg} provided by the organic system, and thus the lowest RF_a are reported for organically produced maize and peas. More pronounced are the results for carrots and potato. Organically grown carrots have a lower RF than conventionally produced carrots, thanks to the greatly reduced resource consumption associated with organic cultivation (Table C1), which diminishes the effect of a lower Y_{org} compared to Y_{con} on the RF. Obviously, applying allocation results in a more pronounced environmental performance in favor of organic carrots. Also for potato a large difference between Y_{org} and Y_{con} can be reported, i.e. Y_{org} is 44% lower than Y_{con}, but (similar to carrots) this has been outweighed by the large difference in resource consumption per cultivated area (Table C1), which contributes to having the lowest RF_a under organic farming.

In Table 5, the allocation of the environmental impact to the ES_{prov} is given, in order to be able to compare the unallocated RF (all impact assigned to the harvested products) with the allocated RF. It clearly indicates that without allocation, an overestimation of the RF occurs. Next to that, also the resource consumption allocated to the delivered ES_{reg} can be calculated. In traditional LCA, this is not taken into account. These results are discussed in Appendix D.

4.3. A closer look at the allocation approach

4.3.1. A range of choices affecting the allocation factors

The allocation factors are a first attempt to make a more complete comparison of the environmental sustainability of organic and

Table 5
Resource footprint of conventional (RF_{con}) and organic systems (RF_{org}) and allocated RF when applying allocation based on ecosystem service scoring ($RF_{a,con}$ and $RF_{a,org}$ for conventional and organic system, respectively). The green colored numbers correspond to the lowest resource footprint (conventionally or organically produced product).

	Environmental impact fully allocated to the harvested products			Environmental impact allocated to the supplied provisioning ecosystem services		
Product	RF_{con} ($MJ_{ex}kg^{-1}$)	RF_{org} ($MJ_{ex}kg^{-1}$)	Ratio RF_{org}/RF_{con} (%)	$RF_{a,con}$ ($MJ_{ex}kg^{-1}$)	$RF_{a,org}$ ($MJ_{ex}kg^{-1}$)	Ratio $RF_{a,org}/RF_{a,con}$ (%)
Barley grain	35.3	54.3	*154*	24.2	26.4	*109*
Carrot	6.2	5.0	*81*	4.2	2.4	*57*
Faba bean	47.7	102.9	*216*	32.7	50.1	*153*
Maize grain	35.8	44.5	*124*	24.6	21.7	*88*
Maize silage	5.0	6.1	*121*	3.5	3.0	*86*
Potato	14.4	15.3	*106*	9.9	7.4	*75*
Protein pea	70.7	89.9	*127*	48.5	43.8	*90*
Rape seed	81.5	116.0	*142*	55.9	56.5	*101*
Rye grain	28.8	48.7	*169*	19.8	23.7	*120*
Triticale grain	45.9	85.1	*185*	31.5	41.4	*131*
Wheat grain	37.9	55.8	*147*	26.0	27.2	*104*

conventional production systems. The developed procedure, however, required some choices and assumptions.

First, we have made a general selection of ES related to agro-ecosystems. However, the number of selected ES (n_{prov} and n_{reg}) might be different when focusing on a particular case study about which more specific information regarding farm practices is available. Certain practices or decisions (e.g., greening measures) will vary (the focus of) the range of ES supplied by the farm system.

Per case study, the number of selected ES might vary, however, it is important to include the same ES (and thus the same number of ES) for organic and conventional farming in the comparison. Current agricultural LCAs generally evaluate the impact of the harvested product. In this research, however, we compare the environmental sustainability of 1 kg product, including the harvested product as well as other relevant ES_{prov}. To ensure that we make an unbiased comparison of the environmental sustainability of the product produced under conventional and organic agriculture, the same basket of products must thus be considered.

As discussed, the product in the case of allocation refers to all ES_{prov}. Most of the ES_{prov} selected in this study belong to the CICES division 'biomass', which gets a higher score for the conventional farming system, but one ES assigned to the division 'genetic material from all biota' is also included (Table 2). Because this ES gets a higher score for the organic than for the conventional system, the sum of the scores given to ES_{prov} becomes more balanced for both systems. Including both CICES divisions in our assessment is supported by the LCA viewpoint. Typically, LCA distinguishes three areas of protection, i.e. human health (e.g., life expectancy of humans), natural resources (e.g., resource availability e.g. biomass or water) and ecosystem quality (e.g., biodiversity) (Sonderegger et al., 2017). The area of protection natural resources includes biotic and abiotic resources. According to Taelman et al. (2016), LCA methods evaluating the impact of land use on biotic resources can be classified into methods accounting for a change in (1) biomass or (2) genetic resources, thus corresponding with the two divisions distinguished by CICES regarding the biotic ES_{prov}.

Finally, also the relative 0–5 scoring procedure retrieved from Burkhard et al. (2012) can have a large impact on the value of the allocation factors. If possible, we have adopted the scores from that study (Table A2). As the scores related to the organic system were missing, we needed to add them. So the scoring used in this work can be debated as it depends on the information considered by Burkhard et al. (2012) to define the scores for the conventional system, and on the experience and knowledge of the experts consulted to assign scores to the organic system. Therefore, it needs corroboration with e.g., an international expert panel before it may be considered in e.g., policy making. The consultation of a broad expert panel would allow to make a second table collecting the variability on the scores. This would make it possible to estimate the uncertainty of the results.

Today, the approach used by Burkhard et al. (2012) and applied in this study, is considered as the most commonly used ES assessment method. It is a quick assessment method that is easy to understand and communicate, and can highlight main issues. By making use of this evaluation method, it is possible to make a comprehensive assessment. In contrast, for some ES, quantitative data are still lacking. The method is flexible enough to integrate and rely on all kinds of data (models or measurements). However, there are some criticisms (Campagne et al., 2017). To overcome the main shortcoming in further research, further extension of experts to be involved can be proposed, e.g. through establishing an international panel. This would enable to make a second table collecting the variability on the scores and to estimate uncertainties.

It should be kept in mind that the main goal of this study is to offer and test a methodology to account for ES in LCA in order to comprehensively compare the sustainability of crops produced by conventional or organic farming. The applied approach including a thorough literature review and expert judgement, seems to be adequate to define the scores needed in this research. For any particular case study, even when the same ES are selected, these values should always be checked critically and, if needed, adapted. Indeed, some measures or choices of farmers might change the capacity of the ecosystem to supply ES. Interesting would be to compare the results with other sustainability assessment tools such as those obtained by the SMART (Sustainability Monitoring and Assessment Routine) farm tool, which enables the assessment of the sustainability performance of farms across different regions (Schader et al., 2016). However, the challenge before being able to perform comparisons is the compilation of an extensive data inventory required to calculate the resource footprint for both the conventionally and organically produced food product.

4.3.2. Broaden the applicability

In this study, we put the focus on arable crops. However, the concept could also be applied to grassland and permanent crops. This might involve a different number of ES, and a new scoring of the ES would be required.

Furthermore, in this study, only ES_{prov} and ES_{reg} are addressed, assuming ES_{cul} being of minor importance and more challenging to score. But for specific agro-ecosystems, also ES_{cul} could be of high relevance for humans. The environmental burden should then be allocated to ES_{prov}, ES_{reg} and ES_{cul}. This would imply a recalculation of the factors f_{prov} and f_{reg}, and f_{cul} (i.e. allocation factor indicating the share of environmental impact that need to be assigned to the ES_{cul}) should then be calculated. Consequently, the share of ES_{prov} would even further decrease. The approach in this study (excluding ES_{cul}) can thus be considered as representing the conservative approach. Scoring of ES_{cul} is however more difficult as it is often subjective and less scientific research is available compromising a good choice of score values.

In this study, both the scores and the resource footprint are defined for general cases of conventional and organic farming. However, each farmer has to make a range of decisions which might affect the supply of ES and the resource footprint. In further research, it would be interesting to offer a list of ES scores dependent on the applied farm practices. To do so, a range of scores should be evaluated by a broad expert panel.

4.3.3. Exploring other options to calculate the allocation factors

In this study, we rely on the average capacity to deliver ES_{prov} and ES_{reg} in the allocation approach. In this sense, an equal weight is attached to ES_{prov} and ES_{reg} (both can get a maximum score of 5). Another option could be to use the ratio of the total capacity to supply ES_{prov} and the total capacity of ES delivered in order to compute f_{prov}. Clearly, n_{reg} is higher than n_{prov}. Consequently, more weight is attached to the ES_{reg}. This would be reflected in the results: $f_{prov,con}$ would equal 0.47 and $f_{prov,org}$ only 0.28 (Appendix E, Table E1). Consequently, for almost all crops, the lowest RF_a is then reported for organically produced crops (Table E2). On the one hand, one could argue that the priorities are then clearly reflected in the allocation factors, but, on the other hand, the availability of ES_{reg} in CICES which can be associated with plant production systems, is higher than the number of biotic ES_{prov} relevant in the agricultural context. In addition, the number and the selection of ES depend on the choices made by the LCA practitioner.

A second option could be to give a weight to the bundle of ES_{prov} and ES_{reg}, instead of using the averages. For instance, the share of ES_{prov} to the total bundle of ES could be used as weighting factor. Or, this weighting factor could also be defined arbitrary, e.g., a weight of 0.6 could be assigned to ES_{prov} and thus 0.4 for ES_{reg}, estimating ES_{prov} of higher importance for agricultural production systems. Further research is needed to investigate the reliability of these options.

Another option for allocation could be to rely on the monetary value of ES instead of using the ES assessment approach of Burkhard et al. (2012). Cao et al. (2015) use monetary values to evaluate land use impacts on ES. The economic value corresponds to the economic costs to society to compensate for the loss in the ES provided. These authors rely on six existing biophysical impact indicators from soil ecological functions (e.g., biotic production potential (Brandão and Milà I Canals, 2013) and fresh water regulation potential (Saad et al., 2013)), which are then converted into economic valuation of ES loss. The advantage is that all ES are expressed in one single unit (Cao et al., 2015), as in the approach of Burkhard et al. (2012). The main difficulty is that no indicators are yet available for many ES. Thus, applying this methodology for the broad range of ES related to agro-ecosystems, as investigated in this study, is not yet possible.

4.4. The allocation approach in the life cycle assessment context

According to the ISO standards, system expansion is preferred above allocation (ISO, 2006). Other researchers debate this and state that the choice is being dependent on the situation (Zamagni et al., 2008). In the context of this study, it might not be easy to apply system expansion. One could suggest to add a certain area that can deliver the missing ES or yield. But the additional area to provide e.g. the missing yield will immediately deliver other ES as well. The supply of these ES will probably not exactly match the expectations (what and how much). So it will be difficult to obtain a comparable basket of products. Another option for system expansion could be to focus on the supply of ES. For instance, to improve the supply of the ES regarding soil quality, extra compost could be added. However, it might be complicated to define for each ES a particular method to improve the supply of ES. And again, this might influence other ES as well, influencing the basket of products.

ES assessment and LCA are clearly linked. While LCA will focus on the damage to the ecosystems by changing the supply of ES due to human interventions (negative effect), ES assessment presents the positive service delivered by ecosystems (positive impact). For several impact categories and ES, the same aspect is discussed, e.g. erosion, carbon sequestration. In the last decade, the importance to integrate ES in LCA has been acknowledged and is also emphasized by the UNEP-SETAC Life Cycle Initiative. In this context, guidelines to account for the impact of land use on biodiversity and ecosystem services are presented by Köllner et al. (2013). They propose a cause-effect chain in which biodiversity damage potential and ecosystem services damage potential are distinguished as main impacts (endpoint). The latter is based on the structure of the Millenium Ecosystem Assessment classification and is linked to the impacts of land use to the following ES and corresponding impact categories: potential to produce biomass (biotic production potential), the impact on climate by influencing the carbon sequestration (Climate Regulation Potential); the impacts on water quantity and quality (freshwater regulation and water purification potential), and the impacts on soil quantity and quality (erosion regulation potential). The ecosystem service damage potential depends then on the difference in quality between the system under study and a reference. CFs have been elaborated in Saad et al. (2013). They are clearly aiming to calculate the impact of land use on the potential to supply an ES. Köllner et al. (2013) mention that the list of impact categories considered can be extended, because up to now, only a limited number of impact categories are considered. A clear advantage of this approach is the fact that the impact on the included ES is studied in detail and is quantitatively determined. This method is undoubtedly different from the method presented in this study. Here, the focus is on the potential of ecosystems to supply ES. Therefore, ES are considered as (co-)products. This method does not allow to quantitatively define the impact of land use on the ES. Instead, scores are assigned to the ES based on expert opinions. One advantage of the method presented here is that it allows to include a broad range of ES, which is not yet possible for the method of Köllner et al. (2013). In addition, not only the impact of land use but also of other farm management decisions are considered in this assessment. So, here the focus is to address the multifunctional role of agriculture by a broad overview and qualitative estimation, while the approach suggested by Köllner et al. (2013) is more focussed on a quantitative estimation for specific ES.

5. Conclusions

In this study, we address the shortcoming that the multifunctional role of agricultural systems is often not integrated in LCA. This results in contra-intuitive results when comparing the environmental sustainability of organic and conventional agro-ecosystems. Often, a lower environmental burden is found for organic products when LCA results are considered per area and per year basis. However, lower yields are generally reported for organic farming systems. Hence, when evaluating the impact per product unit (e.g. per kg product), the highest impacts are then assigned to organically produced products. The product provided by an agricultural system is, however, more than the harvested product, and includes actually a bundle of ES. In this study, we propose allocation factors based on the capacity to supply ES to assign the environmental impact to, on the one hand, the produced ES_{prov}, and, on the other hand, the ES_{reg}. By doing so, we stress the multifunctional role of agriculture and acknowledge the efforts made by farmers that not only aim to increase the productivity but also environmental sustainability (e.g., practices to maintain a good soil quality). Therefore, the environmental impact should not only be allocated to the harvested product but to all ES supplied.

In this study, allocation factors are developed for arable land crops but they can easily be determined for permanent grassland and permanent crops. Ideally, guidance on how the scores would change when implementing nature-oriented measures or applying environmentally sound practices, should be developed, as this will affect the supply of ES. Further research is also needed about how ES can be weighted in relation to each other. This research is a good basis to further integrate the multifunctional role of agriculture in environmental sustainability assessments, and to demonstrate the value of LCAs to highlight efforts towards sustainable agriculture.

Declaration of Competing Interest

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

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Appendix A. Supplementary data

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