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ABSTRACT

Life cycle assessment (**LCA**) and ecosystem services assessment (**ESA**) are often used for environmental assessment. LCA has been increasingly used over the past two decades to assess agri-food systems and has established that ruminant products have higher impacts per kg of protein than products from monogastric species. Conversely, ESA is used less but is likely to rank ruminant systems higher than monogastric systems, as the former often include grasslands that can provide high levels of regulating ecosystem services (**ESs**). Here, we applied both methods to a selection of contrasting meat-oriented animal-production systems that included either ruminants or monogastrics (6 of each). We considered 16 environmental impact categories in the LCA and two functional units: 1 kg of human-edible protein (**HEP**) and 1 m^2yr of land occupied. We used the life-cycle inventory step of LCA to characterise the land occupation of the systems, i.e. the land cover types used, such as croplands and grasslands. Based on these land covers and quantification of the ES they provide, we performed ESA. We estimated that ruminant systems had higher environmental impacts than monogastric systems per kg of HEP for all 16 LCA impact categories studied. For example, for ruminants and monogastrics, mean greenhouse gas (**GHG**) emissions were 280 vs 32 kg CO_2-eq., respectively (P = 0.002), and mean fossil energy use was 351 vs 189 MJ, respectively (P = 0.009). The trend was the opposite for impacts per m^2yr, with mean GHG emissions of 0.50 vs 0.57 kg CO_2-eq. (P = 0.485) and mean fossil energy use of 0.71 vs 3.63 MJ (P = 0.002) for ruminants and monogastrics, respectively. We also estimated that ruminant systems had a higher capacity to supply regulating ES than monogastric systems did, with mean scores of 2.4 and 1.2, respectively (P = 0.002), due to multiple types of grasslands in ruminant systems. Applying both LCA and ESA to a range of contrasting animal-production systems was a novelty of this study, and ESA indicated that ruminant systems have higher positive environmental contributions than monogastric systems. The study also found that LCA and ESA frameworks can agree or disagree on the assessments of animal-production systems depending on functional unit used (i.e. agreement per unit of land occupied but disagreement per unit of HEP).

Implications

Environmental assessment is crucial for investigating the sustainability of animal production. Life cycle assessment, which focuses on negative environmental impacts, is commonly used to assess animal-production systems. Ecosystem services assessment is more recent and focuses on positive contributions (e.g. carbon sequestration). Here, we applied both life cycle assessment and ecosystem services assessment to a range of contrasting animal-production systems that included either ruminant or monogastric animals. Ruminant systems had better performances in the ecosystem services assessment, whereas monogastric systems had lower life cycle assessment impacts per kg of protein. Animal-production systems can thus differ in their positive contributions to the environment and negative environmental impacts.

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Introduction

Life cycle assessment (**LCA**) and ecosystem services assessment (**ESA**) are two frameworks commonly used for environmental assessment. LCA is a standardised method based on inventorying material and energy flows involved in all stages of the life cycle of a given product (e.g. manufacturing, use) to estimate environmental impacts (**EIs**) of a wide range of products in multiple economic sectors (e.g. services, industry, agriculture). The method is historically product-oriented and originated in industry in the 1960 s (Huppes and Curran, 2012). It uses well-established software, such as SimaPro and OpenLCA, and inventory databases, such as ecoinvent (Frischknecht et al., 2007) and Agribalyse (Koch and Salou, 2020). It is increasingly used to assess EIs of agri-food systems (Merida et al., 2022; van der Werf et al., 2020), and the number of peer-reviewed studies published in English on these systems increased from 1 to 1 040 per year, from 1992 to 2018 (van der Werf et al., 2020). LCA estimates EIs such as the greenhouse gases (**GHGs**) emitted or energy used to produce, for example, 1 kg of wheat or beef (Poore and Nemecek, 2018). According to LCA's product-oriented approach, agricultural systems that perform better are usually high-input industrial systems in which the use of inputs (e.g. feed, fertilisers) is optimised to produce as much output as possible (van der Werf et al., 2020).

Ecosystem services assessment is a more recent framework that became popular in the 2000 s and originated in ecology and economics (Costanza et al., 1997, 2017; Millennium Ecosystem Assessment, 2005). It assesses ecosystem services (**ESs**) that represent the "ecological characteristics, functions, or processes that directly or indirectly contribute to wellbeing" (Costanza et al., 2017). ESA distinguishes several types of ES, including provisioning ES (**PES**), regulating & maintenance ES (**RES**) and cultural ES (Haines-Young and Potschin, 2018). PES represents the production of physical commodities (e.g. grain, wood, meat), and ESA usually assesses production per ha per year. RES represents ESs that help stabilise biophysical processes (e.g. erosion prevention, climate regulation), and ESA usually uses proxies to quantify these processes per ha per year as well (e.g. t of carbon sequestered per ha and year). Cultural ESs represent non-material ESs that contribute to mental or cognitive well-being (e.g. providing a pleasant hiking experience or aesthetic scenery), and these ESs are the most difficult to quantify, but they can be assessed based on visitation rates of sites per year, for example. ESA can be performed using software and databases that characterise and quantify ES such as INVEST and the ES Valuation Database, respectively, but to our knowledge, these tools are much less widespread than LCA tools and used mainly by researchers. According to ESA, agricultural systems that perform better are based on semi-natural resources and processes, such as organic croplands (Boeraeve et al., 2020; Sandhu et al., 2010) or extensive grassland-based sheep- or cattle-grazing systems (Leroy et al., 2024; Dumont et al., 2019).

It is well established that LCA EIs of animal-production systems calculated per kg of product depend on the animal species assessed. Meat from monogastric species (e.g. chickens, pigs) has much lower LCA EIs than that from ruminant species (e.g. sheep, cattle) (Poore and Nemecek, 2018; Flachowsky et al., 2017; de Vries and de Boer, 2010). For example, producing 1 kg of meat from chicken, pigs and cattle emits 3.7–6.9, 3.9–10.0 and 14.0–32.0 CO_2-eq., respectively (de Vries and de Boer, 2010). The pattern is similar for energy use (de Vries and de Boer, 2010) and land use (Poore and Nemecek, 2018; Flachowsky et al., 2017; de Vries and de Boer, 2010). Ruminants' higher LCA EIs are due mainly to the methane that they emit during rumination, which contributes greatly to GHG emissions (de Vries and de Boer, 2010), and to their lower feed-use efficiency and fecundity, which means that they require more resources over their lifetimes and reproductive cycles (de Vries and de Boer, 2010). Conversely, as mentioned, the permanent grasslands in the most extensive sheep and cattle farms can provide a relatively high level of RES, whereas the crops that provide the ingredients for monogastric feed provide lower levels of RES (Schils et al., 2022; Burkhard et al., 2012). Grassland-based extensive ruminant systems are therefore likely to have higher scores for RES than monogastric systems do.

Life cycle assessment and ecosystem services assessment can thus provide opposite assessments of animal-production systems, which makes integrating them an important research issue (Taelman et al., 2024; Bergez et al., 2022; De Luca Peña et al., 2022; Alejandre et al., 2019; Boone et al., 2019; Brandão and i Canals, 2013; Koellner et al., 2013). Doing so is especially challenging because the LCA and ESA communities have few connections (VanderWilde and Newell 2021). To integrate these frameworks, Rugani et al. (2019) developed a cause-effect chain model that represents the effects of production systems on ES through a "cascade" approach, integrating processes that impact ecosystems. This approach uses land-cover (**LC**) types to characterise effects on ecosystems (e.g. cropland, grasslands, forest), as the capacity to supply ES can be quantified as a function of LC (Stoll et al., 2015; Burkhard et al., 2012, 2014). For example, forest provides more RES than grasslands, but grasslands provide more RES than croplands (Burkhard et al., 2012, 2014).

Here, we applied this cascade approach to compare the potentially opposite assessments of LCA and ESA results, with intensive monogastric systems having higher LCA EIs than extensive ruminant systems but providing lower levels of RES. The objectives of this study, based on a sample of contrasting meat-production systems, were to (i) apply LCA and ESA simultaneously to animal-production systems using the cascade approach, (ii) assess the strengths and limits of this approach and (iii) examine the implications of combining LCA and ESA for environmental assessment of animal-production systems.

Material and methods

The production systems selected

Due to the differences in LCA EIs as a function of animal species and management intensity, and those in ESA due to the use of grasslands, we based this study on a range of contrasting animal-production systems distributed along two axes (Fig. 1): (1) species (i.e. two ruminants – sheep and cattle – and two monogastrics – chickens and pigs) and (2) management intensity and the relative area of grasslands in feeding systems. The production systems and their related data were selected from Agribalyse 3.01, a life cycle inventory database of agri-food products in France (Koch and Salou, 2020). The systems described in the database include all processes involved in animal production from the cradle (i.e. resource extraction) to the farm gate (i.e. emissions and products that leave the farm) (Koch and Salou, 2020), such as production of feed (including crop production, land use and processing) and its transport to the farm. They also include all GHG emissions, including direct emissions from animals (e.g. enteric methane) and indirect emissions from manure management (e.g. N_2O). Agribalyse uses the functional unit (FU) of 1 kg of liveweight at the farm gate.

For each of the four species, we selected three systems with different management intensities, based on input levels (conventional or organic) and/or the livestock stocking rate (Donald et al., 2001). We included systems with permanent grasslands for grazing or outdoor runs, the latter of which are present in some organic or quality-oriented monogastric systems (i.e. "Label

Management and grassland axis

Animal species axis

Indoor lamb - conventional	Lowland lamb - organic	Pastoral lamb - organic
Young bull (stocking rate > 1.2 LU/ha) - conventional	Young bull (stocking rate < 1.2 LU/ha) - conventional	Dairy cull cow - organic
Pig - conventional	"Label Rouge" pig - conventional	Pig - organic
Chicken - conventional	"Label Rouge" chicken - conventional	Chicken - organic

Fig. 1. Animal-production systems studied from the Agribalyse 3.01 life cycle inventory database (LU: livestock unit).

Rouge", based on specific guidelines validated by a French national institute). To ease understanding, we simplified the names of the Agribalyse systems (see Supplementary Material S1 for the original names and details about the systems).

Life cycle assessment and ecosystem services assessment of the selected systems

We used life cycle inventory data from Agribalyse 3.01 to estimate the LCA EIs of the selected systems, using OpenLCA 1.11 software and the Environmental Footprint 3.0 method, which estimates EIs for 16 EI categories (described in the Results). The latter also provides sub-categories for some of them, such as climate change (i.e. GHG emissions), which is divided into "biogenic" (e.g. from methane), "fossil" and "land use and land-use change". We kept the distinctions among these three sources of GHGs due to the large contribution of livestock farming to GHG emissions and the GHGs' differing lifetimes and dynamics in the atmosphere (Lynch et al., 2020).

We also used these inventory data to estimate levels of ES based on the LCs involved in animal production, in accordance with Rugani et al. (2019). These LCs are available from the input inventories of Agribalyse and depend on the feed required by the animals in the systems. They are expressed per unit area*time of land occupation (m^2yr).

As monogastrics consume mainly grain from annual crops, their main associated LC is arable land and as ruminants consume mainly forage, either as grazed grass or hay, their main associated LC is temporary or permanent grassland. Some ruminants also consume maize silage or grain-based concentrates and are thus associated with a mixture of grassland and arable land. Each system is therefore associated with a specific profile of land occupation (**LO**), characterised by proportional areas of multiple LCs. We thus assessed LO profiles using Agribalyse data and removed LCs that covered less than 1% of LO, as most were non-agricultural lands irrelevant to the study (e.g. roads, mines, construction sites, landfills).

Ecosystem service scoring

We assessed six ESs – Air and climate regulation, Erosion control, Pollination, Nursery (e.g. habitat provision), Maintenance of soil quality and Regulation of water quality – based on the availability of quantitative data for the LCs (section 2.3.2). As recommended by Rugani et al. (2019), we quantified these ES from LC using "capacity matrices" (Burkhard et al., 2012), a well-established tool (Campagne et al., 2020; Jacobs et al., 2015; Stoll et al., 2015; Burkhard et al., 2014) with LCs in rows and ES in columns. Each cell in a matrix contains a score from 0 (no capacity) to 5 (very high capacity) that represents the capacity of a given LC to supply a given ES (Table 1). We focused on PES and RES in this study due to the difficulty in quantifying CES for LCs. CESs are usually studied at the landscape scale (Plieninger et al., 2015; Rodríguez-Ortega et al., 2014) and depend on the mosaic of LCs (Chai-allah et al., 2023; Oteros-Rozas et al., 2018).

Calculating the score of provisioning ecosystem services

To estimate the PES score of production system p (PES_p^{score}), we assessed each system's capacity to supply meat per unit area*time in accordance with ESA. We considered that PES_p^{score} was inversely proportional to the total area*time required to produce meat; thus, it illustrated land-use efficiency. We calculated PES_p^{score} based on a reference system that represented the most land-use-efficient way to produce human-edible protein (**HEP**). We defined this reference system by examining intensive egg- and milk-production systems in Agribalyse 3.01, which are the most land-use-efficient sources of animal protein in the global agri-food system (Poore and Nemecek, 2018; Flachowsky et al., 2017; de Vries and de Boer, 2010).

We based calculations on HEP instead of live weight to use a relevant metric for comparison across animal species. This unit is commonly used in agri-food studies (Laisse et al., 2019; Poore and Nemecek, 2018; Flachowsky et al., 2017; de Vries and de Boer, 2010) because not all animal parts are human-edible (e.g. skin, hooves, bones, egg shells), and among the parts that are, the proportion of protein varies (Laisse et al., 2019; Flachowsky et al., 2017; de Vries and de Boer, 2010).

We calculated the area*time required to produce 1 kg of HEP in production system p ($AProt_p$) as follows (Eq. (1):

$$AProt_p = A_p/(Prop_p^{edible} \times Prop_p^{protein}) \quad (1)$$

where A_p is the total area*time in m^2yr used by system p to produce 1 kg of liveweight, estimated by Eq. (2), and $Prop_p^{edible}$ and $Prop_p^{protein}$ are the human-edible proportion of liveweight and protein proportion of this human-edible fraction, respectively.

We obtained values for the last two variables from Laisse et al. (2019), who provided a range of human-edible proportions from "meat only" to "total consumable". As the latter included co-products used in agro-industries such as gelatine, we selected an intermediate proportion that represented the "meat and offal" fraction, which we considered to be the main targets of animal production for human consumption.

Table 1
Scores of regulating & maintenance ecosystem services by land cover on a scale from 0 (no capacity to supply the ecosystem service) to 5 (very high capacity) in the context of evaluating animal production systems.

Land cover	Erosion control 2.2.1.1	Pollination 2.2.2.1	Nursery (habitat provision) 2.2.2.3	Soil quality 2.2.4.1 2.2.4.2	Water quality 2.2.5.1 2.2.5.2	Carbon stock 2.2.6.1	Mean RES score
Construction	0.0	0.0	0.0	0.0	0.0	0.0	0.00
Urban discontinuous	0.3	0.8	1.4	0.1	0.1	0.1	0.47
Annual crops legume conv. BR	0.0	0.0	1.5	0.5	0.0	2.0	0.67
Annual crops conv. FR	0.5	0.5	1.0	0.0	1.0	2.0	0.83
Annual crops legume conv. FR	1.0	0.5	1.5	1.0	0.0	2.0	1.00
Annual crops org. FR	0.5	1.5	2.5	1.0	1.5	2.5	1.58
Annual crops legume. org. GLO	1.0	2.0	3.5	2.0	0.5	2.5	1.92
Annual crops legume. org. FR	1.0	2.0	3.5	2.0	0.5	2.5	1.92
Temporary grassland without clover conv. FR	2.0	0.5	2.0	2.5	2.0	1.5	1.75
Temporary grassland + clover conv. FR	2.0	1.5	2.5	2.5	1.0	1.5	1.83
Temporary grassland without clover org. FR	2.0	1.5	3.0	3.0	3.5	1.5	2.42
Temporary grassland + clover org. FR	2.0	1.5	3.0	3.0	1.5	2.0	2.17
Permanent grassland without clover conv. FR	2.0	2.0	2.5	3.0	3.0	2.5	2.50
Permanent grassland + clover conv. FR	2.0	2.0	3.0	3.5	2.0	2.5	2.50
Permanent grassland without clover org. FR	2.0	2.5	4.0	3.5	3.0	2.5	2.92
Permanent grassland + clover org. FR	2.0	2.5	4.0	3.5	1.0	3.0	2.67
Mountain meadows – FR	2.5	3.5	4.5	4.0	4.0	1.5	3.33
Forest – GLO	4.75	2.5	4.25	4.5	5.0	4.0	4.17

Abbreviations: RES = regulating & maintenance ecosystem services; conv. = conventional; org. = organic; BR = Brazil; FR = France; GLO = global.
See Supplementary Material S2 for the biophysical proxies used to derive the RES scores.
The code(s) below each RES refer to the CICES classification (Haines-Young and Potschin, 2018).

Variable A_p was calculated as follows (Eq. (2)):

$$A_p = \sum_{n=1}^{n=N_p} LC_{n,p} \tag{2}$$

where $LC_{n,p}$ is the area*time of LC n (m^2yr) in the N_p LCs of the LO profile of production system p, obtained from Agribalyse (n differs by p).

Finally, the PES_p^{score} was calculated as follows (Eq. (3):

$$PES_p^{score} = 5 \times A_{ref}/AProt_p \tag{3}$$

where A_{ref} is the area*time used to produce 1 kg of HEP from the reference system, identified as eggs from caged hens, which was lower than that of milk (26.5 and 32.5 m^2yr, respectively). The reference system thus had the highest possible score of 5, in accordance with the ES scoring range described (Burkhard et al., 2012, 2014). For our systems, for example, the PES score of 0.16 for the lowland organic lamb system was calculated as 5 × 26.5 / 815.7, with 815.7 being the area*time (m^2yr) that the system required to produce 1 kg of HEP.

Calculating the score of regulating and maintenance ecosystem services

We calculated the RES score of production system p (i.e. RES_p^{score}) as the mean RES score of the LCs in the LO profile weighted by the area*time of the LCs, as follows (Eq. (4):

$$RES_p^{score} = \sum_{n=1}^{n=N_p} (LC_{n,p} \times RES_{LC_{n,p}}^{score}) / \sum_{n=1}^{n=N_p} LC_{n,p} \tag{4}$$

where $RES_{LC_{n,p}}^{score}$ is the mean RES score of the LC n in the LO profile of production system p (mean score of the six ES studied).

For example, the RES score of 2.44 for the lowland organic lamb system was calculated as (15 × 1.58 + 330 × 2.17 + 470 × 2.67) / (15 + 330 + 470), where 15, 330 and 470 are the area*time (m^2yr) of annual crops, temporary grasslands and permanent grasslands, respectively, and 1.58, 2.17 and 2.67 are their RES scores (unitless), respectively. The RESs were selected based on a comprehensive literature search (Supplementary Material S2), which identified available biophysical proxies for RES that had been quantified for the LCs of interest in the study (e.g. arable crops, grasslands). The values of the ES proxies defined were then standardised to the 0–5 scale (Table 1).

Life cycle assessment of the systems

To perform LCA of the systems, we used two FUs – kg of HEP and m^2yr of LO – because intensive systems tend to have lower LCA EIs per kg of HEP than extensive systems, as mentioned, whereas the large areas occupied by low-input and extensive systems may decrease the LCA EIs per m^2yr of LO of these systems.

The LCA EIs per kg of HEP of system p ($EI\_HEP_{p,i}$) were calculated as follows (Eq. (5):

$$EI\_HEP_{p,i} = EI_{p,i}/(Prop_p^{edible} \times Prop_p^{protein}) \tag{5}$$

where $EI_{p,i}$ is the EI per kg of liveweight of category i among the 16 EI categories of the Environmental Footprint 3.0 method for system p, with the unit depending on the EI.

The LCA EIs per m^2yr of LO of system p ($EI\_LO_{p,i}$) were calculated as follows (Eq. (6):

$$EI\_LO_{p,i} = EI\_HEP_{p,i}/AProt_p \tag{6}$$

Statistical analysis of data

To assess potential differences in LCA and ESA results between ruminant and monogastric systems, we tested the differences in LCA EIs (per unit of HEP or LO), ES scores and the total area*time required to produce 1 kg of HEP using the Mann-Whitney U test. We also assessed the agreement or disagreement between LCA and ESA using Spearman correlation tests between LCA EIs (per unit of HEP and LO), PES and RES for the 12 systems. We used these non-parametric tests to avoid problems with potentially non-normal distributions in the animal-production systems selected.

Results

Land occupation profiles and ecosystem services

The ruminant and monogastric systems had contrasting LO profiles (Fig. 2, Table 2). The ruminants occupied significantly more land (P = 0.002) than monogastrics did to produce 1 kg of HEP (mean of 856 and 62 m^2yr). The systems sometimes differed greatly; for example, the pastoral organic lamb system used nearly 100 times as much land as the conventional chicken system to produce 1 kg of HEP. The LO profiles also differed in their cropland/-grassland composition, reflecting differences in the diets of ruminants and monogastrics (Table 2). Monogastric LO profiles consisted mainly of cropland, reflecting the importance of grain in their diets, even though grasslands were marginally present in certain monogastric systems in the form of outdoor runs. Conversely, ruminant LO profiles were dominated by grasslands of different types (i.e. temporary and permanent) or mountain meadows, reflecting the importance of forage in their diets.

These LO profiles were reflected in their RES scores, which were higher for ruminant systems than for monogastric systems (mean of 2.4 and 1.2, respectively; P = 0.002). This was due to the higher RES scores of semi-natural habitats in ruminant LO profiles, such as permanent grasslands (Table 1). Conversely, the PES scores of monogastric systems were higher than those of monogastric systems (mean of 2.6 and 0.2, respectively; P = 0.005) because of the smaller amount of land required to feed these animals (Table 2). These differences in PES and RES scores illustrate clear trade-offs between the two types of animal-production systems (Fig. 3).

Life cycle assessments by functional unit

As expected, ruminant systems had higher mean LCA EIs per kg of HEP than monogastric systems did for all 16 LCA EIs studied (Table 3). For example, producing 1 kg of HEP emitted 21–53 and 218–363 kg of CO_2-eq. for monogastrics and ruminants, respectively (detailed results in Supplementary Material S3). This climate change EI was lowest for conventional chicken and highest for organic pastoral lamb. Similarly, producing 1 kg of HEP used 145–253 and 241–439 MJ of fossil resources (i.e. energy) for monogastrics and ruminants, respectively (Supplementary Material S3). The only difference from this ranking was for the land use and land-use change sub-category of climate change, with monogastrics emitting more GHGs from this source due to consuming Brazilian soya beans, some of which are assumed to be associated with relatively recent deforestation.

The LCA EIs per unit of LO had patterns opposite to those per kg of HEP: monogastric systems had higher EIs than ruminant systems did for all 16 EIs studied, most of them significantly so (Table 3), due to the larger amount of land used by ruminant systems. The only difference from this ranking was for the biogenic sub-category of climate change, with ruminants emitting more GHGs from this source (specifically, enteric methane) than monogastrics.

These differences between the two animal types according to the two FUs were reflected in the correlations. Overall, 15 of the 19 LCA EIs (including the three climate change sub-categories) had negative correlations with themselves when expressed according to the other FU (8 of them significant), indicating that they had opposite patterns (Table 4). Among the LCA EIs without negative correlations were the biogenic and land use and land-use change

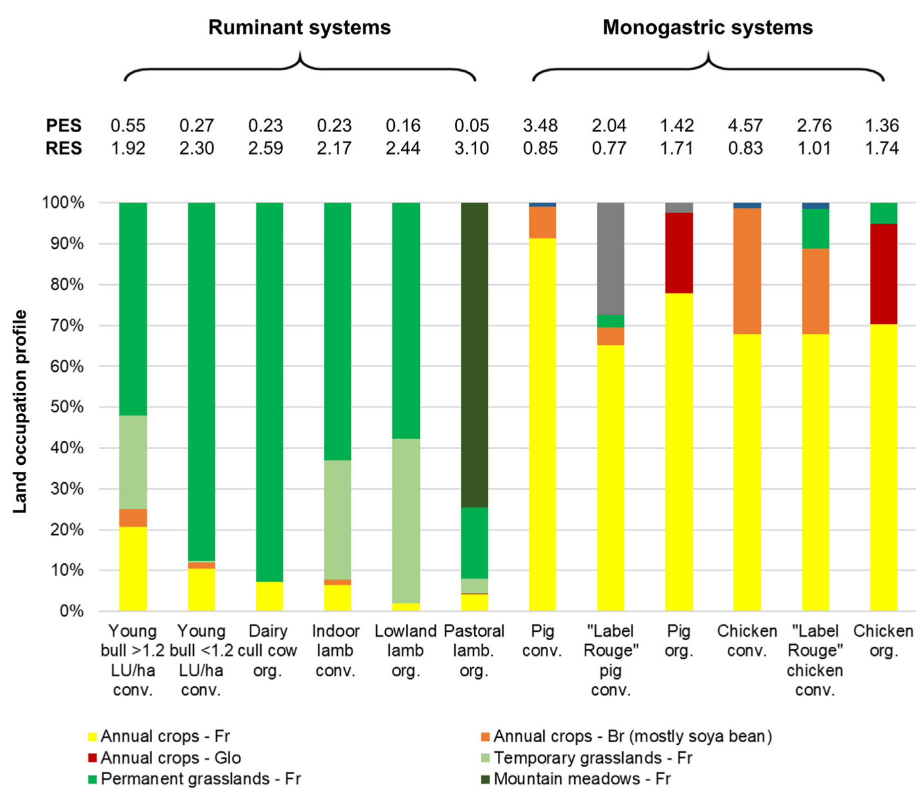

Fig. 2. Simplified land occupation (LO) profiles of the animal-production systems studied (m^2yr required to produce 1 kg of human-edible animal protein). PES: score of provisioning ecosystem service; RES: score of regulating & maintenance ecosystem service. Maximum score = 5. LU: livestock unit; Fr: France; Br: Brazil; Glo: global; conv.: conventional; org.: organic.

Table 2
Land occupation profiles of conventional and organic animal-production systems, land required to produce 1 kg of liveweight and 1 kg of human-edible animal protein, and ecosystem services scores.

Animal type	**Ruminant**						**Monogastric**					
Production system	**Young bull > 1.2 LU/ha conv.**	**Young bull < 1.2 LU/ha conv.**	**Dairy cull cow org.**	**Indoor lamb conv.**	**Lowland lamb org.**	**Pastoral lamb. org.**	**Pig conv.**	**“Label Rouge” pig conv.**	**Pig org.**	**Chicken conv.**	**“Label Rouge” chicken conv.**	**Chicken org.**
Provisioning ES score (–)	0.55	0.27	0.23	0.23	0.16	0.05	3.48	2.04	1.42	4.57	2.76	1.36
Regulating & maintenance ES score (–)	1.92	2.30	2.59	2.17	2.44	3.10	0.85	0.77	1.71	0.83	1.01	1.74
Grasslands (temporary and permanent) and meadows (%)	75%	88%	93%	92%	98%	96%	0%	3%	0%	0%	10%	5%
Human-edible proportion of animal live weight (–)	0.46	0.46	0.40	0.38	0.38	0.38	0.53	0.53	0.53	0.50	0.50	0.50
Protein proportion of the human-edible proportion (–)	0.16	0.16	0.16	0.18	0.18	0.18	0.16	0.16	0.16	0.18	0.18	0.18
Total land required per kg live weight (m^2yr)	17.4	35.5	36.5	39.3	55.8	166.9	3.2	5.4	7.8	2.6	4.3	8.8
Total land required per kg of human-edible protein (m^2yr)	240.0	488.1	577.6	575.2	815.7	2 439.7	38.0	64.7	93.2	28.9	48.0	97.2
Construction	–	–	–	–	–	–	–	1.3	–	–	–	–
Urban discontinuous	–	–	–	–	–	–	–	16.4	2.3	–	–	–
Annual crops (mostly soya bean) – conv. – Br	10.3	7.2	–	8.3	–	–	3.0	2.8	–	8.9	10.1	–
Annual crops – conv. – without legumes – Fr	49.8	51.1	–	36.7	–	–	33.9	40.5	–	19.4	32.5	–
Annual crops – conv. – with legumes – Fr	–	–	–	–	–	–	0.8	1.7	–	0.2	–	–
Annual crops – org. – with legumes (soya bean) – Glo	–	–	–	–	–	4.3	–	–	18.2	–	–	23.9
Annual crops – org. – without legumes – Fr	–	–	40.2	–	14.9	100.7	–	–	47.3	–	–	63.9
Annual crops – org. – with legume – Fr	–	–	1.2	–	–	–	–	–	25.3	–	–	4.4
Temporary grassland – conv. –without clover – Fr	–	–	–	–	–	–	–	–	–	–	–	–
Temporary grassland – conv. with clover – Fr	54.8	2.1	–	168.1	–	–	–	–	–	–	–	–
Temporary grassland – org. –without clover – Fr	–	–	–	–	–	–	–	–	–	–	–	–
Temporary grassland – org. with clover – Fr	–	–	–	–	330.4	90.2	–	–	–	–	–	–
Permanent grassland – conv. –without clover – Fr	125.0	427.9	–	362.1	–	–	–	–	–	–	–	–
Permanent grassland – conv. with clover – Fr	–	–	–	–	–	–	–	1.9	–	–	4.7	–
Permanent grassland – org. –without clover – Fr	–	–	–	–	–	–	–	–	–	–	–	–
Permanent grassland – org. with clover – Fr	–	–	536.2	–	470.4	426.8	–	–	–	–	–	5.0
Mountain meadow – org. – Fr.	–	–	–	–	–	1 817.7	–	–	–	–	–	–
Forest – Intensive (softwood or hardwood) – conv. – Glo	–	–	–	–	–	–	0.3	–	–	0.4	0.7	–

Abbreviations: LU = livestock unit; conv. = conventional; org. = organic; ES = ecosystem service; Br = Brazil; Fr = France; Glo = global.

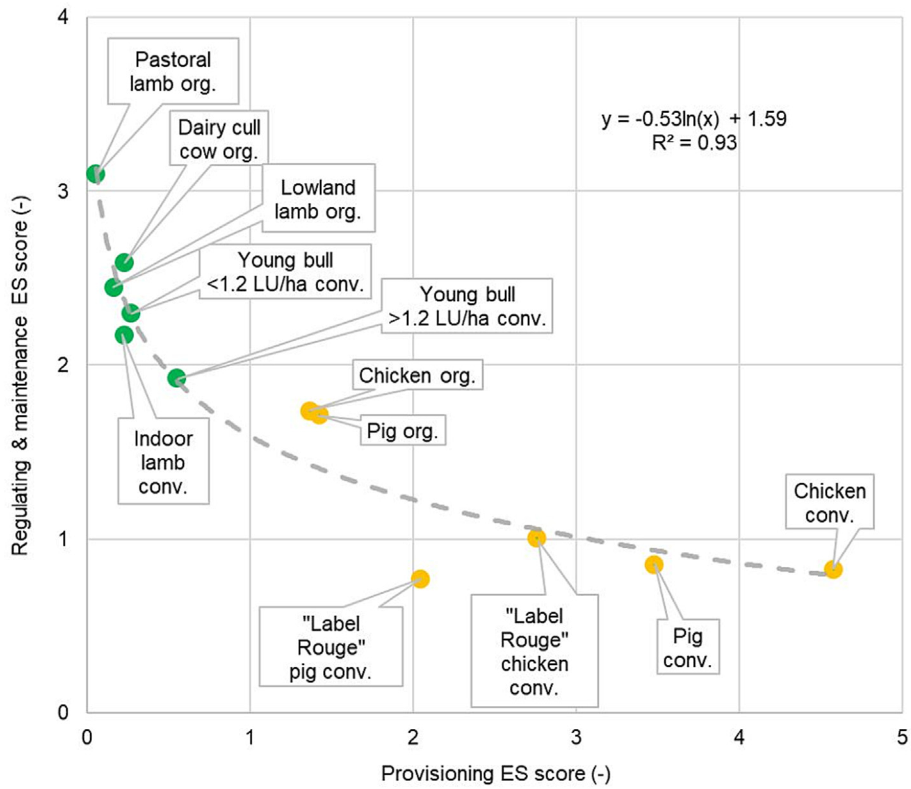

Fig. 3. Trade-off between provisioning and regulating & maintenance ecosystem services (ESs) scores of ruminant (green) and monogastric (yellow) production systems. LU: livestock unit; org.: organic; conv.: conventional.

Table 3
Mean life cycle assessment environmental impacts expressed per (A) kg of human-edible protein or (B) m^2yr of land occupation in the context of evaluating animal production systems.

Environmental impact category	(A) Functional unit: human-edible protein (1 kg)				(B) Functional unit: land occupation (1 m^2yr)			
	Ruminant	Monogastric	RMSE	P[1]	Ruminant	Monogastric	RMSE	P[1]
Acidification (mol H + eq.)	3.5	0.9	0.71	0.002	6.6E-03	1.6E-02	3.61E-03	0.009
Climate change (kg CO_2-eq.)	280	32	32.6	0.002	0.5	0.6	0.23	0.485
Climate change – Biogenic	213	6	25.4	0.002	3.8E-01	1.1E-01	1.68E-01	0.041
Climate change – Fossil	64	23	12.4	0.004	1.2E-01	4.0E-01	8.58E-02	0.002
Climate change – Land use and land-use change	2	3	2.1	0.699	6.5E-03	6.3E-02	4.14E-02	0.015
Ecotoxicity, freshwater (CTUe)	1.4E+03	7.0E+02	6.61E+02	0.589	3.2	14.8	6.57	0.026
Eutrophication, freshwater (kg P eq.)	1.5E-02	6.9E-03	4.51E-03	0.004	2.4E-05	1.3E-04	3.09E-05	0.002
Eutrophication, marine (kg N eq.)	1.1	0.3	0.60	0.002	1.6E-03	4.2E-03	5.52E-04	0.002
Eutrophication, terrestrial (mol N eq.)	15.7	4.0	3.20	0.002	2.9E-02	7.0E-02	1.58E-02	0.009
Human toxicity, cancer (CTUh)	6.06E-08	3.27E-08	5.28E-08	0.180	9.9E-11	5.4E-10	1.32E-10	0.002
Human toxicity, non-cancer (CTUh)	1.97E-06	1.67E-06	5.10E-06	0.937	2.1E-09	2.7E-08	1.50E-08	0.026
Ionising radiation (kBq U-235 eq.)	4.6	3.1	1.30	0.240	9.9E-03	6.0E-02	1.98E-02	0.002
Land use (Point)	3.1E+04	4.3E+03	2.18E+04	0.009	37.1	67.6	16.67	0.002
Ozone depletion (kg CFC11 eq.)	4.3E-06	1.8E-06	5.15E-07	0.002	8.6E-09	3.4E-08	1.09E-08	0.004
Particulate matter (disease incidence)	2.4E-05	6.1E-06	4.85E-06	0.002	4.4E-08	1.1E-07	2.46E-08	0.009
Photochemical ozone formation (kg NMVOC eq.)	3.4E-01	8.0E-02	3.55E-02	0.002	6.3E-04	1.5E-03	4.73E-04	0.015
Resource use, fossils (MJ)	351	189	58.7	0.009	0.7	3.6	1.15	0.002
Resource use, minerals and metals (kg Sb eq.)	8.5E-05	3.3E-05	1.15E-05	0.002	1.7E-07	6.3E-07	1.84E-07	0.002
Water use (m^3 deprivation)	45	33	16.7	0.394	0.1	0.6	0.25	0.002

Abbreviations: CTUe = comparative toxic unit for ecosystems; CTUh = comparative toxic unit for humans.
[1] Significant differences were assessed using the Mann-Whitney U test. Ruminant species: cattle and sheep; Monogastric species: pig and chicken.

sub-categories of climate change, which had specific patterns, as mentioned (i.e., enteric methane emissions of ruminants and consumption of soya beans by monogastrics, respectively).

Relationships between life cycle and ecosystem services assessment

The relationships between LCA EIs and ES scores depended on the FU of the LCA (per unit of HEP or LO) and the ES type (RES or PES). PES scores were usually negatively correlated with LCA EIs per kg of HEP, meaning that the systems that occupied the least land had the lowest LCA EIs per kg of HEP (Table 4). Conversely, RES scores were usually positively correlated with LCA EIs per kg of HEP, meaning that the systems with the highest RES capacity had the highest LCA EIs per kg of HEP (Table 4). As expected, the LCA EIs per unit of LO had the opposite trend: PES scores were positively correlated with all LCA EIs per unit of LO, indicating

Table 4
Spearman correlations (A) for each life cycle assessment environmental impact between its expressions according to two functional units (per kg of human-edible protein and per m^2yr of land occupation) and (B) between life cycle assessment environmental impacts and scores of each type of ecosystem service (provisioning and regulating & maintenance) in the context of evaluating animal-production systems.

Environmental impact category	(A) Correlation between the two FUs		(B) Correlation between LCA EIs and ES							
			FU: HEP (1 kg)				FU: LO (1 m^2yr)			
			PES		RES		PES		RES	
	Corr.	P	Corr.	P	Corr.	P	Corr.	P	Corr.	P
Acidification (mol H+ eq.)	−0.79	0.004	−0.87	0.000	0.83	0.001	0.95	<0.0001	−0.90	<0.0001
Climate change (kg CO_2-eq.)	−0.23	0.471	−0.81	0.002	0.84	0.001	0.65	0.026	−0.49	0.110
Climate change – Biogenic	0.64	0.028	−0.85	0.001	0.85	0.001	−0.31	0.319	0.37	0.237
Climate change – Fossil	−0.73	0.010	−0.84	0.001	0.83	0.001	0.96	<0.0001	−0.90	<0.0001
Climate change – Land use and land-use change	0.72	0.011	0.50	0.104	−0.41	0.184	0.92	<0.0001	−0.85	0.001
Ecotoxicity, freshwater (CTUe)	0.50	0.104	0.22	0.499	−0.17	0.604	0.93	<0.0001	−0.85	0.001
Eutrophication, freshwater (kg P eq.)	−0.73	0.010	−0.80	0.003	0.83	0.002	0.97	<0.0001	−0.87	0.000
Eutrophication, marine (kg N eq.)	−0.71	0.012	−0.94	<0.0001	0.86	0.001	0.85	0.001	−0.85	0.001
Eutrophication, terrestrial (mol N eq.)	−0.78	0.004	−0.87	0.000	0.83	0.001	0.94	<0.0001	−0.89	<0.0001
Human toxicity, cancer (CTUh)	−0.16	0.619	−0.50	0.099	0.57	0.059	0.88	0.000	−0.78	0.004
Human toxicity, non-cancer (CTUh)	0.64	0.030	0.00	1.000	0.12	0.716	0.71	0.013	−0.59	0.049
Ionising radiation (kBq U-235 eq.)	−0.06	0.869	−0.13	0.700	0.20	0.528	0.99	<0.0001	−0.93	<0.0001
Land use (Point)	−0.54	0.071	−0.81	0.002	0.75	0.007	0.86	0.001	−0.92	<0.0001
Ozone depletion (kg CFC11 eq.)	−0.58	0.052	−0.71	0.012	0.77	0.005	0.95	<0.0001	−0.89	<0.0001
Particulate matter (disease incidence)	−0.79	0.004	−0.87	0.000	0.83	0.001	0.95	<0.0001	−0.90	<0.0001
Photochemical ozone formation (kg NMVOC eq.)	−0.67	0.020	−0.82	0.002	0.85	0.001	0.94	<0.0001	−0.88	0.000
Resource use, fossils (MJ)	−0.48	0.121	−0.59	0.046	0.68	0.019	0.97	<0.0001	−0.90	<0.0001
Resource use, minerals and metals (kg Sb eq.)	−0.71	0.012	−0.76	0.007	0.80	0.003	0.99	<0.0001	−0.92	<0.0001
Water use (m^3 deprivation)	−0.06	0.852	−0.30	0.342	0.40	0.201	0.96	<0.0001	−0.89	<0.0001

Abbreviations: FU = functional unit; LCA = life cycle assessment; EI = environmental impact; ESs = ecosystem services; HEP = human-edible protein; LO = land occupation; CTUe = comparative toxic unit for ecosystems; CTUh = comparative toxic unit for humans; PES = provisioning ecosystem services; RES = regulating & maintenance ecosystem services.

that systems that produced more HEP per unit of LO, had higher EIs per unit of LO. In other words, systems with high PES scores, such as conventional monogastric systems that produce large amounts of HEP per unit of LO also have high LCA EIs per unit of LO, while systems with low PES scores, such as pastoral organic sheep, also have low LCA EIs per unit of LO. Conversely, RES scores were negatively correlated with all LCA EIs per unit of LO.

The correlations between LCA EIs per unit of LO and either PES or RES scores were particularly strong: 31 of the 38 (i.e. for 19 LCA EIs including the three climate change sub-categories) correlations exceeded |0.85|. We examined the inversion of the correlations between RES and three specific LCA EIs (i.e. climate change – fossil; resource use, minerals and metals; and resource use, fossils) according to the two FUs (Fig. 4), because they represent mechanisms that could explain the strength of the correlations, e.g. industrial impacts of animal-production systems on ecosystem ecological integrity (See Supplementary Material S4 for all LCA EIs according to the two FUs compared to RES). These three EIs per unit of LO had a particularly strong linear or non-linear relationship with the RES score (R^2 = 0.78, 0.81 and 0.85, respectively).

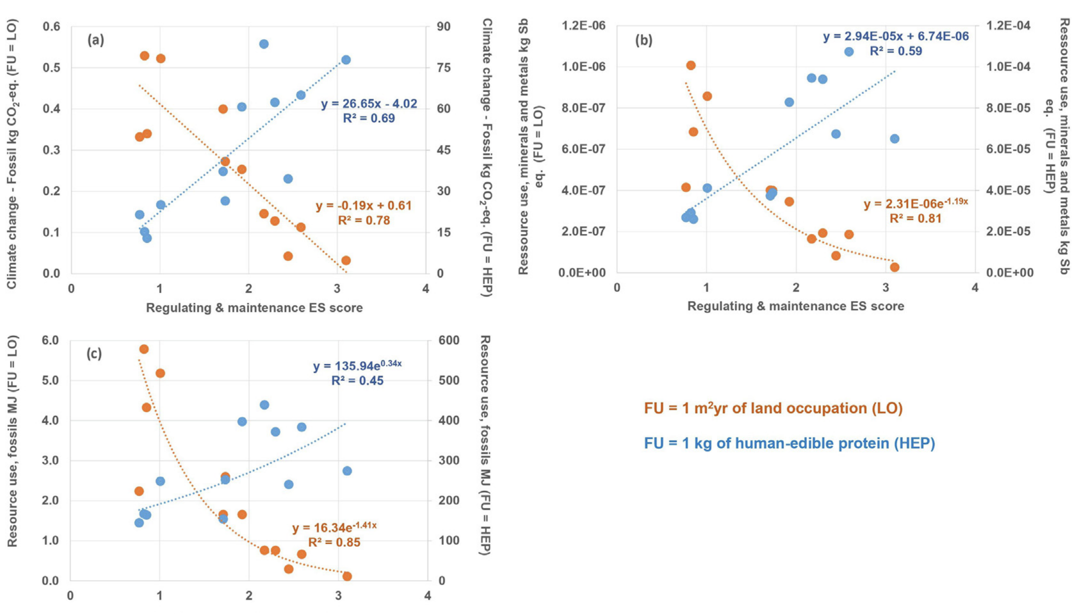

Fig. 4. Regressions between scores of regulating & maintenance ecosystem services (ESs) and environmental impacts in life cycle assessment impact categories of (a) climate change – fossil, (b) resource use, minerals and metals and (c) resource use, fossils, according to two functional units (FUs) in the context of evaluating animal-production systems.

Discussion

Combined assessment using ecosystem services and life cycle assessment

The first objective of this study was to assess a range of ruminant and monogastric systems using LCA and ESA simultaneously. We estimated that LCA EIs per kg of HEP were higher for ruminant systems than those of monogastric systems, which is consistent with previous studies (Poore and Nemecek, 2018; Flachowsky et al., 2017; de Vries and de Boer, 2010). Conversely, LCA EIs per unit of LO had the opposite trend, due to the larger extents of ruminant systems. Ruminant systems had in addition a higher capacity to supply RES, due to the amount of semi-natural LCs in their LO profiles, represented by permanent grasslands and mountain meadows. Overall, the capacity of animal-production systems to supply RES is positively or negatively correlated with LCA detrimental EIs per kg of HEP or per unit of LO, respectively. The agreement or disagreement between LCA and ESA thus depends on the FU used, which is this study's first contribution.

These results describe distinct patterns of positive and negative contributions of animal-production systems. Although LCA can include negative emissions (e.g. carbon sequestration) and ESA can include disservices (e.g. nuisance of mosquito bites), LCA focuses on pollutant emissions and resource depletion, considered negative, whereas ESA focuses on ecosystem services, considered positive. The distinct trends estimated using LCA and ESA thus show that ruminant and monogastric systems provide distinctly different negative and positive contributions. Quantifying this dual pattern is this study's second contribution.

Methodological strengths and limits

The second objective of this study was to assess strengths and limits of the cascade approach for combining LCA and ESA for environmental assessment of animal-production systems.

Strengths

One major strength is that the cascade approach inventoried the LCs used in animal-production systems, which provided a basis for quantifying both types of ES. This quantification allowed trade-offs between ES to be assessed, as illustrated by the clear opposition of PES and RES for the animal-production systems selected (Fig. 3). Another strength is that the approach used FUs as distinct as kg of product, commonly used in LCA, and ecosystem type, commonly used in ESA (van der Werf et al., 2020). To date, using multiple FUs has remained a methodological challenge, and several studies, including that of Rugani et al. (2019), recommended doing so through land use (Taelman et al., 2024; VanderWilde and Newell, 2021). We followed this recommendation by relating kg of HEP produced to the distribution of ecosystem types, described using LCs, in LO profiles. Reconciling these contrasting FUs through land use following the cascade approach is this study's third contribution.

Limits

Application of the cascade method in this study also identified three potential limits. The first was to weigh the mean RES score by LC areas, which conceals potential interactions between LCs when supplying ES. For example, planting grassland strips can significantly increase erosion prevention in annual crops (Schulte et al., 2017), which shows that the level of RES provided by a given LC can depend on other adjacent LCs. To address this limit, the distribution of LCs in the LO profile could be represented explicitly when calculating the weighted mean of RES scores, for example by adapting an index that is proportional to the number of LCs in the LO profile and represents the evenness of their distribution (e.g. Shannon index).

A second limit can come from the fact that LCs can have a spatial distribution of thousands of km (e.g. soya bean crops in Brazil and grasslands in France), with potentially different levels of ES demand. This did not limit this study, however, as we used scores of ES-supply capacity (following Rugani et al. (2019)), which represent the potential amount of ES that ecosystems could supply, not the demand for these ES (Burkhard et al., 2014). For example, an ecosystem with a high density of pollinating insects can have a high ES-supply capacity but zero ES demand if no plants around it require pollination. ES matrices could be used to improve this approach, as they can also represent ES demand (Burkhard et al., 2012, 2014).

A third limit was choosing the production of eggs from caged hens as the reference system. We selected this system to focus on animal production, but we could have selected a protein-crop system (e.g. pea, soya bean) instead. Protein-crop systems require much less land to produce HEP than animal systems (Poore and Nemecek, 2018), but selecting one would have changed the distribution of PES scores. However, differences in quality between plant and animal protein are currently debated, as animal protein is digested more easily and has a more favourable amino-acid composition (Beal et al., 2023; McAuliffe et al., 2023). This subject is the focus of recent "nutritional LCA" studies (McAuliffe et al., 2020, 2023; Sonesson et al., 2019), which may help select a reference protein-production system more objectively and may ultimately help refine how PES scores are calculated.

Inferring ecosystem services from life cycle assessment

The third objective of this study was to discuss the implications of combining LCA and ESA for environmental assessment of animal-production systems. We found strong correlations between ES and certain LCA EIs per unit of LO, which could be used to infer the capacity to supply RES from LCA. RES can be difficult to assess, as they require large amounts of field work and specialised skills to measure characteristics such as soil organic carbon, the ground covered by vegetation or nectar resources to assess carbon sequestration, erosion control or pollination, respectively (Richter et al., 2021). Conversely, PESs are easier to assess, as they are based on agricultural yields, which are routinely measured. We found strong correlations between RES scores and fossil GHG emissions, fossil energy use and use of minerals and metals per unit of LO, and these LCA EIs could be used to estimate levels of RES. These EIs are also meaningful because they presumably represent the amounts of industrial machinery and chemical inputs used to modify ecosystems to produce agricultural products, which in turn decreases the ecological integrity of these ecosystems and thus their capacity to provide RES (Burkhard et al., 2012). These LCA EIs per unit of LO, which are related to the use of fossil energy, minerals and metals, may thus be used to assess RES indirectly. LCA, with its longer history and more established software and databases, could therefore be used to infer ESA, which has fewer operational tools available.

Conclusion

By applying the cascade approach of Rugani et al. (2019), we used LCA inventory data to identify LO profiles of animal systems, which made it possible to perform ESA, which is a novelty of this study. We thus benefitted from the well-established LCA methods and tools to perform ESA. We estimated strong correlations between LCA EIs and ES, which indicated that monogastric systems had the lowest LCA EIs per kg of HEP but also the lowest capacity to supply RES. Ruminant systems had the opposite trend, with the highest LCA EIs per kg of HEP and the highest capacity to supply RES, due to their grasslands. This study thus highlighted trade-offs between positive and negative environmental contributions of animal-production systems.

We also found that LCA and ESA frameworks can agree or disagree on the assessments of animal-production systems depending on the functional unit used (i.e. agreement per unit of LO but disagreement per unit of HEP). Specifically, we estimated that high LCA EIs per unit of LO were negatively correlated with the capacity to supply RES, which could illustrate decreased ecological integrity. It would be interesting to study the strength of these correlations further so that LCA databases and tools could be used to provide operational proxies for assessing RES.

Supplementary material

Supplementary material to this article can be found online at https://doi.org/10.1016/j.animal.2024.101368.

Ethics approval

Not applicable.

Data and model availability statement

The inventory data used to perform the LCA are available in the Agribalyse database, publicly available for research, and the data related to RES scores are available in the tables of the article or in the Supplementary Material.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work the author(s) did not use any AI and AI-assisted technologies.

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Declaration of interest

This study was funded by Interbev, the French professional union of bovine and ovine meat and livestock. The authors declare that they have no known competing personal relationships that could have appeared to influence the work reported in this article.

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