Trade-offs between higher productivity and lower environmental impacts for biodiversity-friendly and conventional cattle-oriented systems

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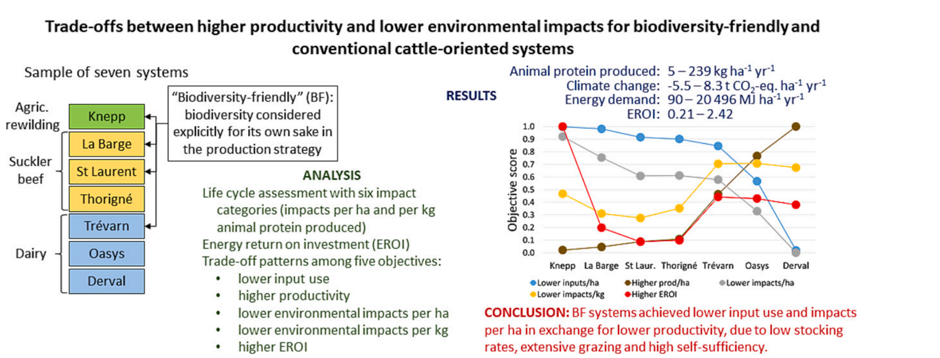
Aymeric Mondière [a], Michael S. Corson [a,*], Julie Auberger [a], Daphné Durant [b], Sylvain Foray [c], Jean-Francois Glinec [d], Penny Green [e], Sandra Novak [f], Frédéric Signoret [g], Hayo M.G. van der Werf [a]

[a] INRAE, Institut Agro Rennes-Angers, SAS, 35000 Rennes, France
[b] INRAE, DSLP, 17450 Saint Laurent de la Prée, France
[c] INNOVAL, 35538 Noyal sur Vilaine, France
[d] Trévarn farm, 29800 Saint Urbain, France
[e] Knepp Castle Estate, Horsham RH13 8LJ, United Kingdom
[f] INRAE, Experimental unit of Lusignan, 86600 Lusignan, France
[g] La Barge farm, 85690 Notre Dame des Monts, France

HIGHLIGHTS

- Biodiversity loss is a key environmental issue of livestock production.
- Livestock systems with a wide range of biodiversity friendliness were assessed.
- Productivity and environmental impacts per ha of land were lower for biodiversity-friendly systems than for other systems.
- Land abandonment and new compensation payments may provide opportunities for biodiversity-friendly livestock systems.
- Agricultural rewilding's potential forms and value for farmers remain to be explored further.

GRAPHICAL ABSTRACT



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ABSTRACT

CONTEXT: Biodiversity loss caused by livestock production is a key environmental issue. Biodiversity-friendly livestock systems aim to favour biodiversity mainly for its own sake. Environmental impacts of biodiversity-friendly livestock systems have not been studied, nor have trade-offs between higher productivity and lower environmental impacts of these systems.

OBJECTIVE: This study aimed to (i) assess the productivity and environmental impacts of a sample of cattle-oriented production systems representing a wide range of external inputs, productivity and consideration of biodiversity; (ii) identify trade-offs among the objectives of lower input use, higher productivity, lower environmental impacts and higher energy return on investment (EROI); (iii) relate the systems' production strategies to the trade-offs identified and (iv) identify perspectives for biodiversity-friendly livestock systems.

* Corresponding author.
E-mail address: michael.corson@inrae.fr (M.S. Corson).

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METHODS: This study assessed the productivity and environmental impacts of a sample of seven cattle-oriented production systems: an agricultural rewilding system (biodiversity-friendly, in England), three suckler beef systems (two of them considered biodiversity-friendly, in France) and three dairy systems (one biodiversity-friendly and one conventional, in France). Life cycle assessment (LCA) was applied to assess six environmental impacts (i.e. terrestrial acidification, freshwater eutrophication, marine eutrophication, land occupation, and in particular, climate change and energy demand) and was combined with calculation of EROI.
RESULTS AND CONCLUSIONS: Per hectare of land occupied to produce livestock and their feed, production of human-edible animal protein and environmental impacts of these systems increased as energy demand (i.e. input use) increased. Per hectare, human-edible animal protein production, energy demand and climate change impact (considering carbon dynamics) ranged from 5 kg, 90 MJ and −5.5 t CO_2-eq., respectively, for the agricultural rewilding system to 239 kg, 20,496 MJ and 8.3 t CO_2-eq., respectively, for the conventional French dairy farm. Patterns of trade-offs varied among the production strategies of the farms. The four biodiversity-friendly farms in the sample had low productivity but also low environmental impacts per ha, especially for climate change and energy demand, due to being extensive systems with relatively high self-sufficiency. Biodiversity-friendly farms also had higher EROI than other farms that produced the same products. Using LCA to assess biodiversity-friendly systems, in particular agricultural rewilding, challenges its ability to consider natural baseline emissions and non-provisioning ecosystem services.
SIGNIFICANCE: These results emphasise the need to consider the multiple functions of these systems and their overall environmental performances and sustainability. Current economic and social trends may provide opportunities for biodiversity-friendly livestock systems as a possible future for livestock systems.

1. Introduction

Biodiversity loss, one of the main environmental issues worldwide, is occurring more rapidly than ever before (Díaz et al., 2019). As agriculture occupies 40% of the world's ice-free land, the biodiversity status of agricultural land is important (Foley et al., 2011). Overall, 77% of agricultural land is used to feed livestock (Ritchie and Roser, 2019), and livestock production is the main cause of global land-use change, including deforestation, which is a major cause of biodiversity loss (Alexander et al., 2015). Livestock production also contributes to other impacts that pose threats to biodiversity, such as climate change (e.g. from enteric methane emissions) (Reid, 2006), acidification (e.g. from manure ammonia emissions) (Azevedo et al., 2013), eutrophication (e.g. from nitrate leaching) (Hautier et al., 2009) and energy consumption (e.g. from production of farm inputs), the last of which drives the previous impacts (Spangenberg, 2007).

Low-input livestock systems can help maintain biodiversity in agricultural landscapes (Kleijn et al., 2009), in particular by replacing external inputs with ecological processes (Therond et al., 2017). They can be considered "nature-based" or "agroecological" systems, which aim to maintain a certain level of biodiversity useful for their production (Dumont et al., 2013; Sabatier et al., 2015), often by maintaining a low stocking rate (Michalk et al., 2019) and a high percentage of permanent grassland (Duru et al., 2019), a habitat with a high potential to host biodiversity (Bengtsson et al., 2019). Among these systems, those based on grassland are key for maintaining biodiversity (Rodriguez-Ortega et al., 2014), while other systems favour ecological processes on the farm through crop-livestock interactions (Martin et al., 2016; Ryschawy et al., 2012).

Other types of livestock systems, called "biodiversity-friendly" in this study, aim to favour biodiversity, both above-ground and below-ground, for its own sake as well as its functional value for production (Mondière et al., 2022). Certain farmers aim first to conserve and restore natural ecosystems (e.g. flood meadows, create ponds) and secondly to maintain an economically viable production (Mondière et al., 2022; Paysans de Nature, 2021). Others consider livestock species as ecosystem engineers in rewilding projects (Gordon et al., 2021), with the goal to rebuild a natural ecosystem by restoring natural processes and reducing human management (Carver et al., 2021). These approaches that combine restoration of ecological processes with some degree of livestock production have been called "domesticated rewilding" (Thomas, 2021) or "agricultural rewilding" (Corson et al., 2022), an emerging form of land use conceptually positioned between agroecology and rewilding.

The biodiversity potential of farms can be assessed directly using field surveys of plant and/or animal species richness and diversity. In a companion study (Mondière et al., 2023a), we assessed avian biodiversity of a sample of seven cattle-oriented systems (four of them biodiversity-friendly) in France and England, and estimated their degree of "ecological integrity". The biodiversity-friendly farms tended to have more bird species than the other farms (Mondière et al., 2023a), but their relative environmental impacts were unknown, like those of biodiversity-friendly livestock systems in general. Likewise, trade-offs between the objectives of higher livestock production and lower environmental impacts of these systems had not been assessed. Life cycle assessment (LCA) is a methodological framework often used to estimate environmental impacts of livestock systems (McClelland et al., 2018), including those of agroecological systems based on crops/grassland and livestock (Casasús et al., 2012; Du et al., 2022; Lemaire et al., 2014). To this end, the present study focused on the productivity and environmental impacts (estimated using LCA) of the same sample of farms.

The objectives of this study were to (i) assess the productivity and environmental impacts of this sample of cattle-oriented production systems representing a wide range of external inputs, productivity (i.e. production of human-edible animal protein per ha) and consideration of biodiversity; (ii) identify patterns of trade-offs among the objectives of lower input use, higher productivity, lower environmental impacts and higher energy return on investment (EROI), with a focus on climate change and energy demand; (iii) relate the systems' production strategies to the trade-offs identified and (iv) identify perspectives for biodiversity-friendly livestock systems, in particular for their productivity and environmental impacts.

2. Methods

2.1. Farm sample and data collection

To establish the farm sample, we sought farms whose farmers were motivated to reflect on relations between their production strategy and its environmental impacts, and who could provide the large amount of data necessary for LCA. We included only farms with dairy or suckler beef cattle (the most common ruminant systems in our region) and intended to select no more than three of each, given the time and effort needed to perform a full LCA of each one. We first enlisted experimental farms known to us (operated by INRAE or the Chamber of Agriculture of the Pays de la Loire region), and their farmers helped identify other potential farms (commercial or experimental) to include. We aimed to select farms that would represent their type of system well (e.g. agricultural rewilding, dairy or suckler beef system). Ultimately, the farm sample contained seven experimental or commercial farms with a wide range of (i) use of external inputs and (ii) strategies for considering environmental issues (Table 1) in oceanic climate regions of France and England (Fig. 1): one agricultural rewilding system (Knepp (K)), three suckler beef systems (La Barge (B), Saint Laurent de la Prée (S) and Thorigné d'Anjou (Th)) and three dairy systems (Trévarn (Tr), Oasys (O) and Derval (D)). Because the concept of agricultural rewilding is relatively new, only one such farm (K) was included in the sample. Although most of its livestock were longhorn cattle (3-year mean of ca. 65), K also contained traditional breeds of pigs (ca. 14) and ponies (ca. 11, unharvested), as well as fallow deer (ca. 245) and red deer (ca. 31) (Table S1), that grazed and browsed freely throughout the year.

Four farms – K, B, S and Tr – considered biodiversity (primarily above-ground species of wild animals, plants and insects associated with agricultural land uses) explicitly in their production strategies; consequently, we considered these farms as "biodiversity-friendly". This classification was supported by results of point counts of bird species performed on each farm, which identified 30–64 species among the farms, of which K, B, S and Tr had 49, 54, 64 and 33, respectively (Mondière et al., 2023a) (Table 1).

For the experimental farms (S, Th, O and D), annual data (i.e. precise values for technical data, mean values for production data) were collected for a three-year period – 2018-2020 – except for D (2016–2018), which installed an anaerobic digester in 2018 that required adapting its production strategy from 2018 to 2020 to the new equipment. For the commercial farms (K, B and Tr), annual data (i.e. estimated mean values for technical data, mean values for production data) for 2018–2020 were collected by interviewing the farmer. The data described all farm inputs, livestock, land use, operations, machines, buildings, resource use and production (Table S2).

2.2. Production strategies

Productivity and production strategies varied greatly among the systems, even within each type of system (suckler beef or dairy) (Table 2). The farms lay on a gradient from extremely extensive (< 0.18 livestock units (LU) ha^{-1}) to more intensive (ca. 1.30 LU ha^{-1}), but none

Fig. 1. Locations of the seven farms in the sample.

Table 1
Description of the seven farms in the sample (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval). AR: agricultural rewilding.

Type	Farm	Product(s)	Overall production strategy	Livestock and feeding	Soil and climate context	Biodiversity considered?	Bird species[a]
AR	K	Organic meat	Restore ecological processes while producing a small amount of meat (Corson et al., 2022)	Traditional breeds of cattle, pigs and ponies, as well as fallow and red deer, graze and browse 450 ha of fenced grass-, shrub- and woodland with no other feed	Heavy clay soil, wet winter and dry summer	The main objective	49
Suckler beef	B	Organic meat	Conserve natural environments while producing meat in an economically viable way (Paysans de Nature, 2021)	Maraîchine cows fed mainly from permanent grassland (grazed and cut) in a marshland context	Hydromorphic soil, wet winter and dry summer	In the overall strategy	54
	S	Organic meat and crops	Obtain data on agroecological practices in a marshland context (Roche et al., 2022)	Maraîchine cows fed mainly from permanent grassland and protein crops produced on the farm in a marshland context	Hydromorphic soil, wet winter and dry summer	In the overall strategy	64
	Th	Organic meat	Optimise use of grassland resources in the feeding strategy (Farm-XP, 2021)	Limousine cows fed from multispecies temporary grassland and protein crops produced on the farm	Shallow sandy-loam soil, alternating wet and dry periods, dry summer	Not specifically	39
Dairy	Tr	Organic milk and meat	Follow agroecological practices based on grassland management (Glinec, 2019)	Crossbred cows (mainly Prim'Holstein, Jersey and Montbeliarde) fed mainly from permanent grassland (grazed and cut)	Schist fine-silt soil, high annual rainfall and sometimes dry summer	In the overall strategy	33
	O	Milk and meat	Adapt milk production to climate change (Novak et al., 2020)	Crossbred cows (Prim'Holstein, Jersey and Scandinavian Red) fed mainly from a variety of forage crops and grassland produced on the farm, with a low percentage of off-farm concentrate feed	Dry clay-loam soil, dry summer	Not specifically	30
	D	Milk and meat	Obtain data on milking, use of new technologies and energy transition (Farm-XP, 2022)	Prim'Holstein cows fed from forage produced on the farm (temporary grassland and maize silage) and off-farm concentrate feed	Loam soil, wet winter and dry summer	Not specifically	29

[a] Number of bird species observed during point counts (Mondière et al., 2023a).

Table 2
Three-year mean production and livestock-feeding management data of the seven farms in the sample (2018–2020, except for Derval (2016–2018)) (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval). Production data are expressed per ha of the total area used for the animal production system (off- and on-farm). AR: agricultural rewilding.

Type of system			AR	Suckler beef			Dairy		
Name			K	B	S	Th	Tr	O	D
Biodiversity-friendliness[a]			++	+	+	–	+	–	–
Milk produced (t FPCM[b] ha^{-1})			–	–	–	–	3.10	5.12	6.71
Animal live weight produced (kg ha^{-1})			54.2	114.2	218.2	272.5	97.3	163.1	197.3
Human-edible animal protein[c] produced (kg ha^{-1})			5.0	11.0	21.0	26.0	110.6	183.1	238.6
Mean herd composition (number)	Cows		24.1	50.0	48.0	70.0	70.0	69.0	86.2
	Heifers		40.8	60.0	42.0	71.5	33.0	49.5	56.6
	Other species[d]		300.9	–	–	–	–	–	–
	Sold	Cull cows	9.3	8.0	16.0	26.0	11.0	18.7	27.0
		Heifers ≥ 2 yr old	14.7	10.0	8.3	1.0	–	3.0	4.3
		Heifers < 2 yr old	–	–	3.0	–	–	4.0	–
		Steers	–	–	–	17.3[e]	–	–	–
		Grazing calves	–	25.0	20.0	14.0	–	–	–
		Milk-fed calves	–	–	15.3	5.7	15.0	–	–
		15-day-old calves	–	–	–	–	44.0	32.0	41.3
		Other species[d]	101.0	–	–	–	–	–	–
Annual feed ration for productive cows (kg dry matter day^{-1})	Annual crop silage[f]		–	–	–	–	–	2.9	10.9
	Grass silage		–	–	–	1.3	–	0.1	3.0
	Wrapped bales		–	–	1.7	–	6.8	4.7	–
	Hay		–	4.4	6.2	4.5	0.9	1.1	–
	Grass grazed		–	9.7	8.0	6.3	7.4	6.0	2.6
	Cereals		–	–	–	–	–	–	1.1
	Concentrate feed made on-farm		–	–	0.6	0.1	–	–	–
	Concentrate feed made off-farm		–	–	–	–	–	0.85	2.7
Grazing management	% of year outside		100	75	59	67	67	50	41
	Stocking rate (LU ha^{-1})[g]		0.18	0.53	0.63	0.89	0.79	1.11	1.30
Type of manure stored[h]			None	S	CS	S	S, SL	CS, SL	S, SL
Total area used for animal production (ha)	Crops/temporary grassland on-farm		–	8.0	18.0	118.0	12.0	85.0	90.4
	Permanent grassland on-farm		–	154.0	105.0	29.0	91.0	2.0	1.8
	Off-farm		–	–	–	–	0.8	5.2	21.5
	Total		450.0	162.0	123.0	147.0	103.8	92.2	113.7

[a] Biodiversity-friendliness: ++ = favouring biodiversity is the priority; + = farming practices are adapted to favour biodiversity; – = no specific efforts to favour biodiversity.
[b] FPCM: fat- and protein-corrected milk.
[c] Based on the protein content of FPCM (33 g l^{-1}) and human-edible meat (including edible offal and co-products used in the human food sector) (depending on the type and age of animals (Laisse et al., 2018); see Table S4 for details).
[d] Pigs, ponies (not sold), fallow deer and red deer; see Table S1 for details.
[e] Steers fattened with concentrate feed produced on-farm.
[f] Maize, sorghum and maslin in Oasys; maize in Derval.
[g] Livestock units (LU) per ha of on-farm area used to feed the livestock.
[h] S: solid, CS: composted solid, SL: slurry.

of them were very intensive, unlike other systems in this region of western Europe (Foray et al., 2022; Salou et al., 2017) (Table 2). According to the French system, 1 LU is equivalent to a 600 kg dairy cow that produces 3000 kg of milk and consumes 4500 kg of forage dry matter per year (Benoit and Veysset, 2021). See Table S3 for LU equivalents of the species and categories of animals present on the farms.

The agricultural rewilding system (K), based on a very-low-input production strategy, produced only animals (54.2 kg live weight ha^{-1}) on a large area (450 ha) at a very low stocking rate (0.18 LU ha^{-1}) (Table 2). Suckler beef systems (B, S and Th) produced cattle at higher but moderate productivities (from 114.2 (B) to 272.5 (Th) kg live weight ha^{-1}) and a range of stocking rates (from 0.53 (B) to 0.89 (Th) LU ha^{-1}). Their cattle grazed from spring to mid-autumn, with a variable percentage of permanent grassland (from 20% (Th) to 95% (B)), which resulted in a variety of self-sufficient feeding strategies (from a grass-based feed ration with only hay and grazed grass (B) to a mixed feed ration with several forms of grass and on-farm concentrate feed (S and Th)). The four biodiversity-friendly systems (K, B, S and Tr) were less productive than the other systems.

Production strategies also differed in the kinds of animals sold in addition to cull cows, mainly unfattened grazing calves for B and S, and unfattened grazing calves and fattened steers for Th. Dairy systems (Tr, O and D) produced milk and cattle with a wide range of productivity (from 3.10 (Tr) to 6.71 (D) t FPCM ha^{-1} and from 97.3 (Tr) to 197.3 (D) kg live weight ha^{-1}), grazing durations (from 41% (D) to 67% (Tr) of the year) and stocking rates (from 0.79 (Tr) to 1.30 (D) LU ha^{-1}). Their differences in crop rotations illustrated differences in their feeding strategies (i.e. based only on grass, mainly grazed permanent grassland (Tr); based on grass and different types of silage (O and D), with use of more off-farm concentrate feed in D). Production of human-edible animal protein ranged from 5 (K) to 239 (D) kg ha^{-1}.

2.3. Life cycle assessment

2.3.1. System boundaries

To smooth out annual production fluctuations, LCA was performed for each of the three calendar years of data collected, and the results were then averaged to yield a temporal boundary of one "average" year. The conceptual boundary lay from the cradle (i.e. resource extraction) to the farm gate (i.e. emissions and products that left the farm). Because the LCA considered only animal products, cash and forage crops grown on the farms but not used to feed the animals were excluded; thus, the LCA focused exclusively on the livestock system (Fig. 2).

2.3.2. Functional units

We defined four functional units (FUs), which are measures that reflect functions of the systems studied and are used to standardise the

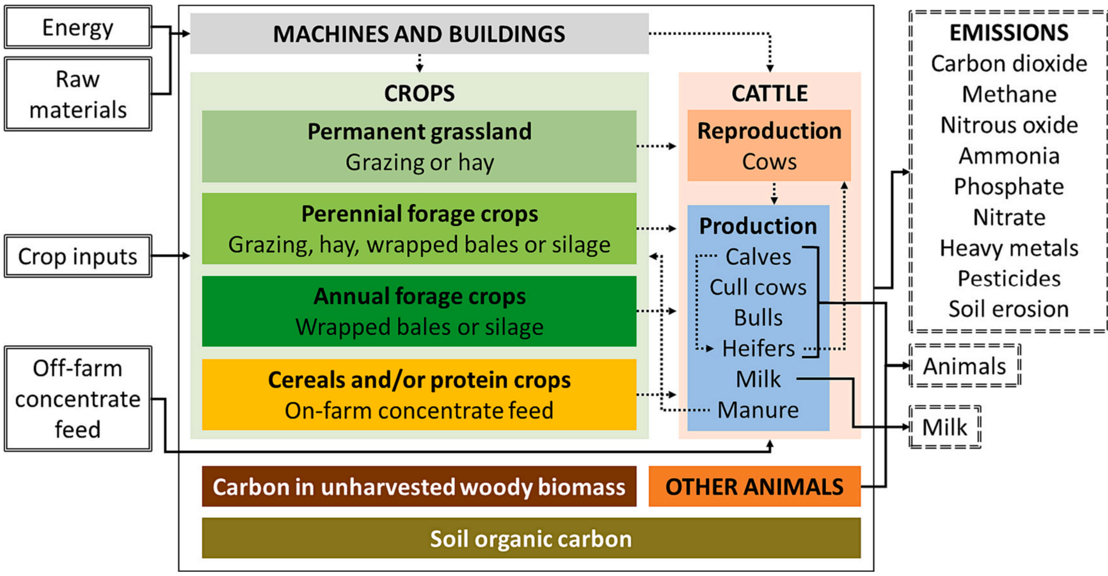

Fig. 2. Inputs, internal components and outputs of the livestock systems studied. Only the agricultural rewilding system contained unharvested woody biomass and other animals. Solid arrows are flows to or from the system, while dotted arrows are internal flows.

values of estimated impacts in order to compare systems to each other on an equivalent basis (Jolliet et al., 2015). Three FUs considered the systems' functions of producing milk, livestock and protein. For milk, the FU was 1 kg of fat- and protein-corrected milk (FPCM, kg), which standardised milk production (M, litres) by its fat content (FC, % by mass) and protein content (PC, % by mass):

$$FPCM = M \times (0.1226 \times FC + 0.0776 \times PC + 0.2534) \times 1.03 \quad (1)$$

For livestock, the FU was 1 kg of live weight. To combine milk and livestock production into a common FU, 1 kg of protein was used, because protein production is the main function of the dairy sector (Kyttä et al., 2021), and protein is the main nutrient in animal products (Gheewala et al., 2020). To convert FPCM to human-edible protein, the protein content of FPCM (33 g kg^{-1}) was used. To convert live weight to protein, live weight was converted to the mass of human-edible protein (including that in edible offal and co-products used in the human food sector) using coefficients of Laisse et al. (2018) (Table S4). The fourth FU considered the systems' function of managing land and was 1 ha of land occupied (on-farm and, for the production of inputs, off-farm) for one year (i.e. ha.year).

2.3.3. *Allocation among co-products*

If a system produces more than one product, several methods exist to estimate each co-product's impacts. One approach is to attribute the system's resource use and pollutant emissions to the co-products using allocation methods. For the agricultural rewilding and suckler beef systems, no allocation was necessary, as livestock were the only product. For dairy systems, using the biophysical allocation method of the AGRIBALYSE® method (Koch and Salou, 2020), impacts of productive dairy cows were allocated to milk and calves proportional to the energy required to produce these two products, while those of renewal heifers were allocated to cull cows.

2.3.4. *Life cycle inventories*

2.3.4.1. MEANS InOut software. Life cycle inventories (LCIs), which list and quantify the flows of resources used and pollutant substances emitted by the system (Jolliet et al., 2015), were modelled using the web-based software *MEANS InOut* v3.5, which facilitates and streamlines generation of LCIs of components of farming systems (Auberger et al., 2018). For all systems except K, LCIs of each field and each cow category (e.g. heifers, calves, dairy cows, fattening cows) were created. Direct emissions (e.g. carbon dioxide, methane, nitrous oxide, ammonia, phosphate, nitrate, heavy metals, pesticides, eroded soil) from and resource use (e.g. diesel, electricity, natural gas, water) for the production of crops (i.e. permanent grassland, annual and perennial forage crops, cereals and protein crops) and cattle were estimated using emissions factors and models in AGRIBALYSE® v3.0 models (Koch and Salou, 2020), except for nitrate leaching from grassland, soil organic carbon (SOC) dynamics and carbon (C) storage in unmanaged woody biomass. Background processes (e.g. for production of inputs) from ecoinvent v3.5 (Moreno-Ruiz et al., 2018) and AGRIBALYSE® v3.0 (Koch and Salou, 2020) databases were used.

Due to the unique management strategies and animal products of K, its LCIs were built differently. First, emissions factors of European extensive livestock systems (IPCC, 2019b) for each animal species (i.e. cattle, pigs, ponies and deer) and category (for cattle) were used to estimate direct emissions and manure emissions, without detailing on-farm feed production (only grazing and browsing) or animal feed using *MEANS InOut* (Table S1). Emissions factors varied as a function of live weight. An LCI of K's area was built using *MEANS InOut* to consider direct phosphate emissions and the few indirect emissions.

2.3.4.2. Nitrate leaching from grassland. Nitrate leaching from grazed permanent and temporary grassland (kg nitrogen (N) ha^{-1} yr^{-1}) was estimated as a function of the stocking rate (LU.days ha^{-1} yr^{-1}) according to Vertès et al. (2007):

$$\text{Nitrate} - \text{N leached} = 8.77 \times e^{0.003 \times [stocking\ rate]} \quad (2)$$

It was calculated for each grazed field when data were available and for the mean grazed area of the system when not. Because all farmers in the sample applied <50 kg N ha^{-1} yr^{-1} of fertiliser (organic or inorganic) to each of their grasslands, and did so only during periods with low leaching risk (spring or late summer, when rainfall is low and grassland N requirements are high), nitrate leaching due to fertiliser applications was assumed to be close to zero and thus negligible, based on expert opinion.

2.3.4.3. Dynamics of soil organic carbon and carbon storage in unmanaged woody biomass. SOC dynamics can influence net greenhouse gas (GHG) emissions of agricultural systems but are rarely considered in LCA studies (Knudsen et al., 2019). Similarly, while biogenic C stored in agricultural products is usually ignored because most is re-emitted within one year, the biogenic C stored in biomass that is not harvested can be considered to be stored for the long term (McKechnie et al., 2011). Because many factors influence SOC dynamics, we used three methods to consider (or not) the variability and uncertainty in their estimates:

Method 1: No consideration of SOC dynamics

Method 2: Rates of SOC sequestration or loss estimated according to French "4 per 1000" data (Pellerin et al., 2019) (Table S5):
- +259 kg C ha^{-1} yr^{-1} for arable crop rotations that include temporary grassland
- −91 kg C ha^{-1} yr^{-1} for arable crop rotations that do not include temporary grassland
- +189 and + 396 kg C ha^{-1} yr^{-1} for permanent grassland that is managed intensively or extensively, respectively

Method 3: For arable land, SOC sequestration or loss rates estimated as in method 2. For permanent grassland and vegetation naturally established on arable land, however, SOC sequestration rates estimated as a function of the age of land use (Pellerin et al., 2019) and an equation of Poeplau et al. (2011): +1042, +784, +424 and 0 kg C ha^{-1} yr^{-1} for age classes 5–25, 26–50, 51–120 and > 120 years, respectively (Table S6).

In K, because the animals graze permanent grassland but also browse shrubland and woodland, the entire area was included in the system boundaries. Since the beginning of K's rewilding scheme in 2001, shrubland and woodland have become established on 99.9 and 35.2 ha, respectively, of former arable land and permanent grassland (Czura, 2021). Because this woody biomass was not harvested, the C it stored was included in calculations. For the pre-existing woodland (49.7 ha) and the newly-established woodland (35.2 ha), IPCC (2019a) data were used to estimate the annual rate of C storage.

First, total annual biomass growth was estimated from annual above-ground biomass growth and the ratio of below- to above-ground biomass in a natural broadleaf forest in a temperate oceanic European climate. Then, the C fraction in this biomass (assuming that below-ground biomass had the same C fraction as above-ground biomass) was used to estimate annual C storage in all woodland biomass (Table S7). For the three types of shrublands that became naturally established, Czura (2021) estimated the storage of above-ground C at K using allometric equations (one per type) that had been developed in previous studies and LIDAR data from K. The same ratio of below- to above-ground biomass was then used to estimate annual C storage in all shrubland biomass (Table S7).

2.3.5. Life cycle impact assessment

Life cycle impact assessment, which classifies resource use and pollutant emissions into multiple impact categories to estimate potential impacts (Jolliet et al., 2015), was then performed. All calculations were performed using SimaPro v8.0 (PRé Sustainability, Amersfoort, Netherlands), which was also used to connect each system's LCIs of crop and animal production. To remain consistent with the main impacts of livestock systems usually assessed (McClelland et al., 2018), the present study focused on climate change, terrestrial acidification, freshwater eutrophication, marine eutrophication (these four from ReCiPe 2016 v1.06 (Huijbregts et al., 2016)), cumulative energy demand (Frischknecht et al., 2015) and land competition (CML non-baseline v3.05 (Van Oers, 2016)) (Table S8). The biogenic C in products and the energy content in biomass used in the production system (e.g. crop inputs, crop or grassland production) were excluded.

2.3.6. Contribution analysis

Contribution analysis was performed to assess the contribution of the system's processes to each impact (Table S9), especially that of C dynamics to the climate change impact and elements of production strategies (e.g. off-farm concentrate feed) to all impacts. For each impact, only processes that contributed >0.1% were included in a process group; all others were grouped in "other". In a context of energy scarcity, interest in the amount of energy consumed by agricultural production is increasing (Benoit and Mottet, 2023). Consequently, the energy consumed to produce crops used to feed cattle was aggregated not into each crop (i.e. four "forage" and one "on-farm concentrate feed" process groups) (as done in most studies), but in "direct energy consumption", in order to aggregate all energy consumed by the system into a single process group.

2.4. Energy return on investment

To analyse trade-offs between higher productivity and lower energy consumption, we calculated the EROI as the ratio of energy that left the system to energy that entered the system (Cleveland et al., 1984). The energy that left the system was calculated from the mass and energy content of the human-edible meat and milk sold (Laisse et al., 2018), while the energy that entered the system was assumed to equal the system's cumulative energy demand.

2.5. Identification of trade-offs

We used the quantitative results to identify patterns of trade-offs among five objectives, one based only on inputs (i.e. lower input use), three based only on outputs (i.e. higher productivity and lower environmental impacts per ha and kg) and one based on both (i.e. higher EROI). In this context, we defined a trade-off as having a characteristic that lies closer to one objective (e.g. production of human-edible protein per ha that is higher) in exchange for a characteristic that lies further from another objective (e.g. environmental impacts per ha that are higher).

To this end, we considered six indicators of input use per ha (i.e. electricity, diesel, N and P in inorganic fertilisers, herbicides, and off-farm concentrate feed), human-edible protein produced per ha, five impact categories per ha (considering climate change estimated using method 3 and excluding land competition per ha, which is theoretically the same for all systems), all six impact categories per kg of protein (again considering climate change estimated using method 3) and EROI. To avoid negative values for climate change, we added the absolute value of the climate change impact for K (per ha and kg) to those of all farms, thus setting the climate change impact of K to zero. We then normalised the indicators for each objective (from 0 to 1) relative to the largest value of each indicator in the sample, using "1 – <normalised value>" for the indicators of the three objectives to decrease (i.e. inputs per ha and impacts per ha and kg). We then calculated the mean of the indicators of these three objectives, transforming them into an "objective score", and used the normalised values of human-edible protein produced per ha and EROI as the other two objective scores. Values closer to 1 indicated that the indicator or objective score lay closer to the objective relative to the other farms in the sample for the three years studied. Finally, we grouped systems with similar patterns of trade-offs and related the systems' production strategies to each pattern, based on the assumption that farmers could choose practices that moved them toward each objective independently.

3. Results

3.1. Main inputs used and pollutants emitted

Differences in production strategies among systems caused large differences in the main inputs used and pollutants emitted (Table S10). K used only 1.2 l of diesel ha^{-1}, and its pollutant emissions per ha were much lower than those of the other systems. As all suckler beef systems were self-sufficient in feed (i.e. 100% of the area used was on-farm) and fertiliser, their only inputs were electricity and diesel, with much higher consumption in Th and S than in B. The degree of self-sufficiency varied among dairy systems. Tr used only 19 kg of off-farm concentrate feed ha^{-1}, whereas O and D used 233 and 1218 kg ha^{-1}, respectively. Tr was self-sufficient in fertilisers, whereas O used 9.7 kg N ha^{-1} and 1.3 kg P ha^{-1}, and D used 8.6 kg N ha^{-1} and 22.0 kg P ha^{-1}. O and D also applied herbicide active ingredients. The percentage of on-farm area in the total area used was 99%, 94% and 81% for Tr, O and D, respectively. Most dairy systems consumed more energy than suckler beef systems did, with diesel consumption of 38.9 (Tr) to 99.7 (D) l ha^{-1} and electricity consumption of 204.5 (Tr) to 396.5 (D) kWh ha^{-1}. Pollutant emissions per ha were lowest for K, intermediate for the suckler beef systems and highest for the dairy systems. Among suckler beef systems, emissions were lowest for B. Among dairy systems, emissions were highest for D, followed by O and Tr (except for those of N_2O, for which O emitted more than D). Biodiversity-friendly systems, except sometimes S, had lower input use and pollutant emissions than the other systems.

3.2. Environmental impacts

Climate change impacts per ha excluding C dynamics followed the same trend as GHG emissions (Tables 3 and S10). They were lowest for K, lower for B than for S and Th among suckler beef systems, and lower for Tr than for O and D among dairy systems. The impact of D was 15.9 times as high as that of K. Impacts per kg of protein were highest for suckler beef systems, intermediate for K and lowest for dairy systems (Table 3). Consideration of C dynamics decreased the climate change impact of each system and somewhat modified the ranking of the systems by impact magnitude, for both FUs (Table 3). The method used to estimate SOC dynamics influenced estimated impacts (Fig. 3). Compared to method 2 (based on Pellerin et al., 2019), method 3 (based on Poeplau et al., 2011 and Pellerin et al. (2019)) estimated less SOC sequestration for systems B and S, similar SOC sequestration for systems O and D, and more SOC sequestration for systems K (including C storage in biomass), Th and Tr. For K, considering C dynamics resulted in negative impacts (Table 3), unlike for the other systems.

Terrestrial acidification impacts per ha followed the same trend as ammonia emissions, the main pollutant contributing to this impact. K had the lowest impact, followed by B, Tr, Th, S, O and D. The impact per ha of D was 8.7 times as high as that of K (Table 3). The impact per kg of protein was highest for K, intermediate for suckler beef systems and lowest for dairy systems. Among suckler beef systems, B and Th had similar impacts, while that of S was highest. Among dairy systems, O had lower impact than D. The impact per kg of protein of K was 9.1 times as high as that of O.

Freshwater eutrophication impacts per ha followed the same trend as phosphate emissions, the main pollutant contributing to this impact, with a large difference between the lowest and highest impacts (167.6 times as high for D as for K) (Table 3). As for terrestrial acidification, impacts per kg of protein were similar within each type of system (6.25–7.03 and 1.54–2.17 g P-eq. for suckler beef and dairy systems, respectively) and lowest for K (0.62 g P-eq.). Consequently, the difference between the lowest and highest impacts was large (11.4 times as high for B as for K).

Marine eutrophication impacts per ha followed the same trend as nitrate emissions, the main pollutant contributing to this impact, with a smaller difference between the lowest and highest impacts (3.7 times as high for D as for K) than those for the other impact categories (Table 3). Impacts per kg of protein were similar for dairy systems (48.0–49.3 g N-eq.), but more variable for suckler beef systems (181.7–360.0 g N-eq.) and highest for K (636.1 g N-eq.), which had an impact 13.3 times as high as that of Tr.

Energy demand per ha followed the trend of the inputs consumed, with higher demand of the systems that were not self-sufficient in feed or fertiliser (O and D) and a large difference between the lowest and highest impacts (227.5 times as high for D as for K) (Table 3). Energy demand per kg of protein was variable for suckler beef systems (78.2–175.5 MJ), less variable for dairy systems (74.2–85.9 MJ) and lowest for K (17.9 MJ).

Because land competition is expressed in m^2year, occupation of 1 ha

Table 3
Environmental impacts per functional unit (FU) (1 ha or 1 kg human-edible animal protein produced) of the seven systems (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval). Impacts per ha are based on the total area used for animal production (off- and on-farm). AR: agricultural rewilding, MC: Multiplier coefficient, equal to the highest impact divided by the lowest impact (excluding negative impacts for climate change).

Impact category		FU	AR	Suckler beef			Dairy			MC
			K	B	S	Th	Tr	O	D	
Biodiversity-friendliness[a]			++	+	+	–	+	–	–	
Climate change (kg CO_2-eq.)	Excluding carbon (C) dynamics	*ha*	562	2553	4420	4670	4488	7469	8949	15.9
		kg protein	111.6	231.7	210.4	179.7	40.6	40.8	37.5	6.2
	C dynamics estimated using Pellerin et al. (2019)	*ha*	−3928	1128	3044	3623	3724	6410	8319	7.4
		kg protein	−780.9	102.4	144.9	139.4	33.7	35.9	34.9	4.3
	C dynamics estimated using Poeplau et al. (2011) and Pellerin et al. (2019)	*ha*	−5466	1800	4213	3157	2328	6598	8342	4.6
		kg protein	−1086.6	163.3	200.6	121.5	21.1	36.0	35.0	9.5
Terrestrial acidification (kg SO_2-eq.)		*ha*	7.99	12.12	28.11	27.73	25.18	33.08	69.58	8.7
		kg protein	1.59	1.10	1.34	1.07	0.23	0.18	0.29	9.1
Freshwater eutrophication (g P-eq.)		*ha*	3.10	77.42	131.39	165.88	170.80	290.70	518.66	167.6
		kg protein	0.62	7.03	6.25	6.38	1.54	1.59	2.17	11.4
Marine eutrophication (g N-eq.)		*ha*	3200	3967	4963	4722	5305	9084	11,845	3.7
		kg protein	636.1	360.0	236.3	181.7	48.0	49.6	49.7	13.3
Energy demand (MJ)		*ha*	90	862	3679	4030	8203	13,902	20,496	227.5
		kg protein	17.9	78.2	175.1	155.1	74.2	75.9	85.9	9.8
Land competition (m^2year)		*ha*	10,000	10,053	10,130	10,137	9883	10,246	10,102	1.0
		kg protein	1987.9	912.4	482.2	390.0	89.4	56.0	42.3	46.9

[a] Biodiversity-friendliness: ++ = favouring biodiversity is the priority; + = farming practices are adapted to favour biodiversity; – = no specific efforts to favour biodiversity.

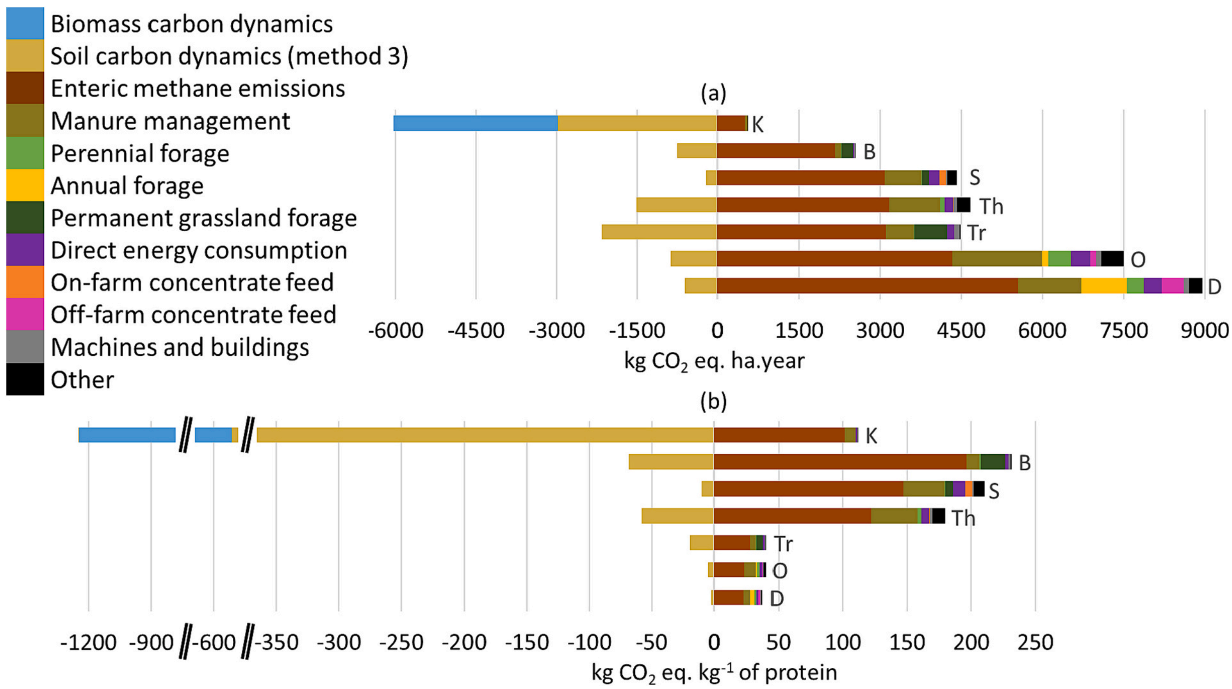

Fig. 3. Contribution analysis for the climate change impact of the seven systems (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval) per (a) ha and (b) kg of human-edible animal protein. To increase readability of the graphs, the contributions of soil carbon dynamics shown (always negative) are only those calculated according to method 3 (Poeplau et al. (2011) and Pellerin et al. (2019)). See Table S11 for detailed results of methods 2 (Pellerin et al. (2019)) and 3.

of the system for one year should equal 10,000 m^2yr. Here, this impact varied slightly ($\leq$ 2.5%) from this theoretical value due to using estimated data for feed rations and crop yields. Per kg of protein, land competition was inversely correlated with the productivity of the systems, with large differences between the lowest and highest impacts (46.9 times as high for K as for D) (Table 3).

3.3. Focus on climate change and energy demand

3.3.1. Climate change

According to both methods for estimating SOC dynamics, all systems sequestered C, which represented from 4% of the overall climate change impact for O and D, for which C sequestration in the soil of crop rotations with temporary grassland was moderate, to 90% for K (due to C sequestration in the soil of 350.5 ha of former arable land and 184.8 ha of woody biomass) (Fig. 3, Table S11). For all systems, C sequestration depended on the assessment method and the percentage and age of permanent grassland. It corresponded to nearly 50% of the impact of GHG emissions for Tr (according to method 3) (Fig. 3) and B (according to method 2) (Table S11).

For all systems, enteric methane contributed most to the climate change impact – from 56% (O) to 90% (K) – due to the relative contributions of the other process groups (no other large contributions for K due to its production strategy) (Fig. 3). The next largest contributor was manure management, which was related to the quantity and type of manure, its storage and its excretion in grassland and animal housing. It contributed from 8% (K, with year-round grazing) to 22% (O, composted manure and slurry used as fertiliser, and the influence of feeding a high percentage of legume forage) (Fig. 3). The relative contribution of the other process groups depended on the feeding strategy (e.g. forages, concentrate feed, buildings, direct energy consumption by machines used for crop and grassland management), with total contribution ranging from 1% (K) to 22% (D). Feeding strategies differed among systems, being based mainly on permanent grassland in K, B and Tr; on permanent grassland, on-farm concentrate feed and perennial forage in S and Th (the former with more permanent grassland and on-farm concentrate feed); and on perennial and annual forage and off-farm concentrate feed in O and D (the latter with more annual forage and off-farm concentrate feed) (Fig. 3). Climate change impact per ha increased as protein production increased, and at a higher rate for K and the suckler beef systems (B, S and Th) than for the dairy systems (Tr, O and D) (Fig. 4).

3.3.2. Energy demand

Direct energy consumption, mainly of diesel and electricity, contributed most (from 49% (D) to 70% (O)) to energy demand for all systems (Fig. 5). The other contributors were related to feeding strategies, including off-farm concentrate feed (0.5% for O to 5.0% for D) and the energy required to produce inputs for annual forage (31% for D, 4% for O, and 1% for S), perennial forage (4.5% for D and 8.5% for O, 3.7% for Tr and 5.7% for S, 2.6% for Th,) and permanent grassland forage (25.2% for Tr, mainly due transport (20 km one way), liming and plastic bale wrap) (Fig. 5). Finally, the energy required to construct and maintain machines and buildings contributed from 7% for D to 40% for K (i.e. the all-terrain vehicle used for herd management). Its relative contribution was related directly to the amount of crop and/or herd management.

Per kg of protein, there was no clear relation between energy demand and protein production, with the energy demand of D (the most productive system) 4.8 times as high as that of K (the least productive system) and the energy demand of B similar to that of Tr (Fig. 5), which was 10 times as productive as B (Table 2). In contrast, per ha, energy demand and protein production were strongly correlated, with energy demand strongly driving production, which varied greatly in the sample (Fig. 6). According to the high coefficient of determination (0.97) of the linear regression calculated between energy demand and protein production in the sample, producing each additional kg of protein required ca. 81 MJ of energy (Fig. 6).

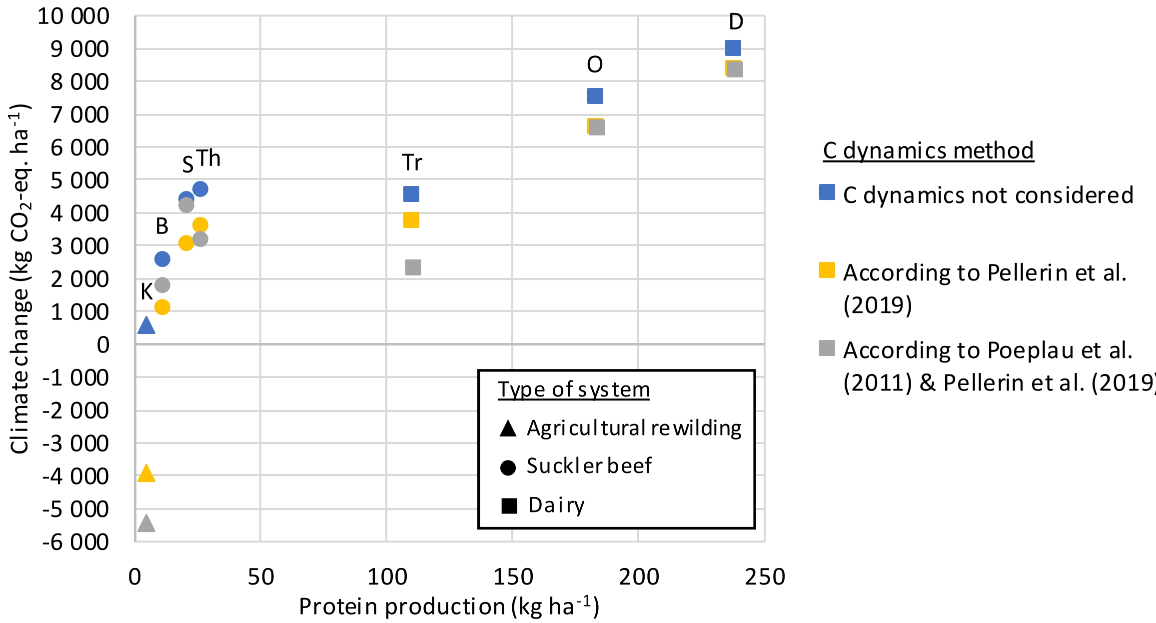

Fig. 4. Climate change impact per ha as a function of human-edible animal protein production per ha of the seven systems (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval). Colours of symbols identify the three methods used to consider (or not) carbon (C) dynamics in the soil and unharvested woody biomass.

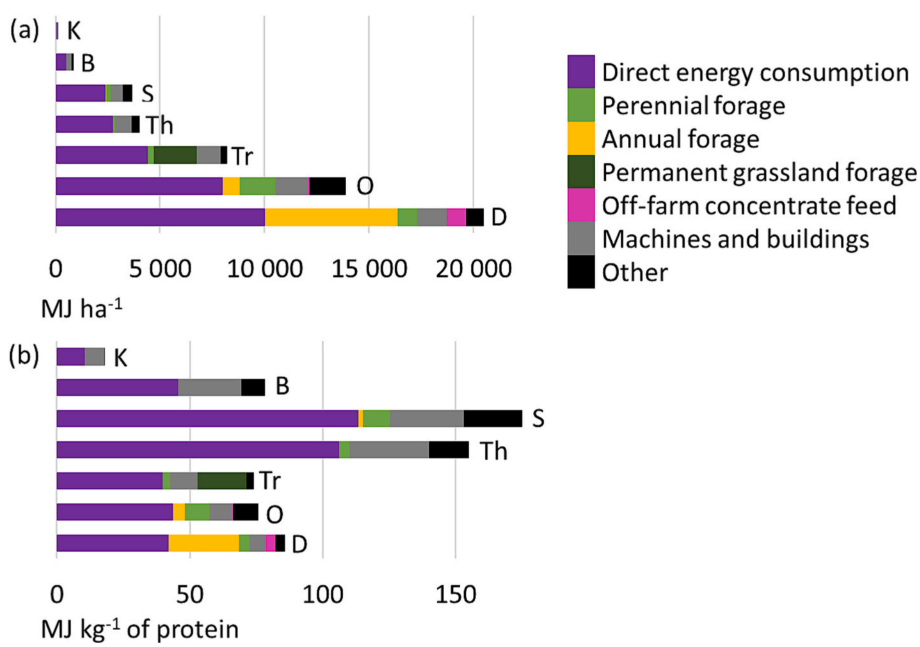

Fig. 5. Contribution analysis of energy demand of the seven systems (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval) per (a) ha and (b) kg of human-edible animal protein. See Table S11 for details.

3.4. Energy return on investment

The agricultural rewilding system (K) had the highest EROI (2.42), followed by the dairy systems, for which EROI was highest for Tr (1.07), followed by O (1.04) and D (0.92), the highest-input dairy system (Fig. 7, Table S12). Among the suckler beef systems, EROI was highest for B (0.48), the lowest-input suckler beef system, followed by Th (0.24) and S (0.21) (Fig. 7, Table S12).

3.5. Trade-offs

We identified four patterns of trade-offs in these systems considering the five objectives (i.e. lower input use, higher productivity, lower environmental impacts per ha and kg of human-edible protein and higher EROI) (Table S13), and related each pattern to one or more production strategies (Fig. 8):

1. *Trade-off pattern that prioritised productivity*: higher productivity and mostly lower impacts per kg in exchange for higher input use and impacts per ha. This trade-off was found in the intermediate- and high-input dairy systems (O and D), whose production strategies were based on feed rations with temporary grassland and annual forage, a high stocking rate and fertiliser and feed inputs. These strategies were similar to those of average specialised lowland dairy

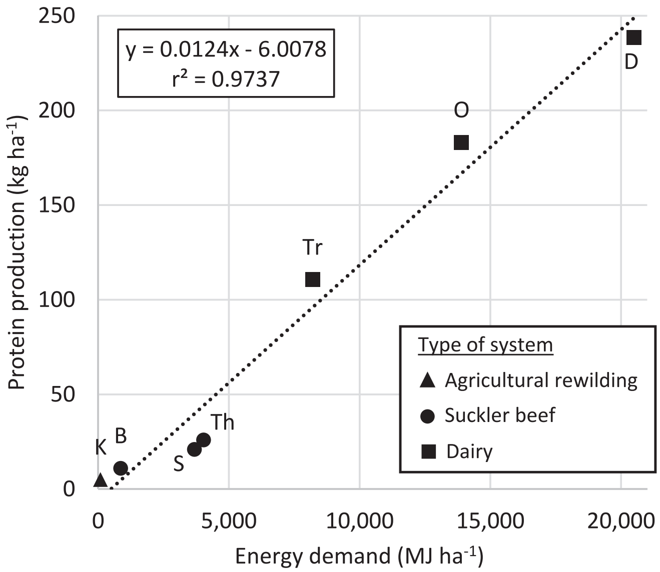

Fig. 6. Production of human-edible animal protein per ha as a function of energy demand per ha of the seven systems (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval).

systems in France (IDELE, 2020), and environmental impacts of similar strategies have been assessed in many studies (Jan et al., 2019; Salou et al., 2017; Thomassen et al., 2008).

2. *Trade-off pattern that favoured low-input land management*: lower input use and impacts per ha in exchange for lower productivity and EROI. This trade-off was found in two suckler beef systems (S and Th), whose production strategies were based mainly on feed rations with grassland and mixed protein-cereal crops to supplement the feed ration in a self-sufficient way at a moderate stocking rate. These strategies resulted in higher self-sufficiency but lower productivity than those of average specialised suckler beef systems in France (IDELE, 2020), and environmental impacts of similar strategies have been assessed in several studies (Bragaglio et al., 2020; Casey and Holden, 2006).
3. *Trade-off pattern that favoured low-impact land management:* lower input use and much lower impacts per ha and higher EROI than other systems of the same type in exchange for lower productivity. This trade-off was identified in the lowest-input and biodiversity-friendly suckler beef and dairy systems (B and Tr, respectively), which were particular due to (i) basing their feed ration only on grassland, (ii) their lower productivity, due to a stocking rate lower than those of other systems of their type (i.e. suckler beef (Bragaglio et al., 2020) or dairy (Salou et al., 2017)) and (iii) for Tr, the stronger influence of soil C dynamics on its climate change impact (O'Brien et al., 2014).
4. *Trade-off pattern that prioritised low-impact land management:* much lower input use, impacts per ha and impacts per kg of protein for climate change (negative impact when considering C dynamics) and energy demand, and much higher EROI in exchange for much lower productivity. This trade-off was identified in the agricultural rewilding system (K), whose strategy of prioritising biodiversity was based on minimising human management over a large area, which minimised energy consumption. It also had production based only on grazing and browsing, a very low stocking rate and a strong influence of C dynamics (in soil and biomass) on its climate change impact. Its production was the lowest of all systems, and this trade-off pattern has not yet been analysed in the literature because the interest in agricultural rewilding is recent (Corson et al., 2022).

4. Discussion

4.1. Interpretation of the main results

The large differences in stocking rate (0.18–1.30 LU ha^{-1}), energy demand (90–20,500 MJ ha^{-1}) and feeding strategy (from 100% grazing and browsing to a mixture of grazing, maize silage and concentrate feed)

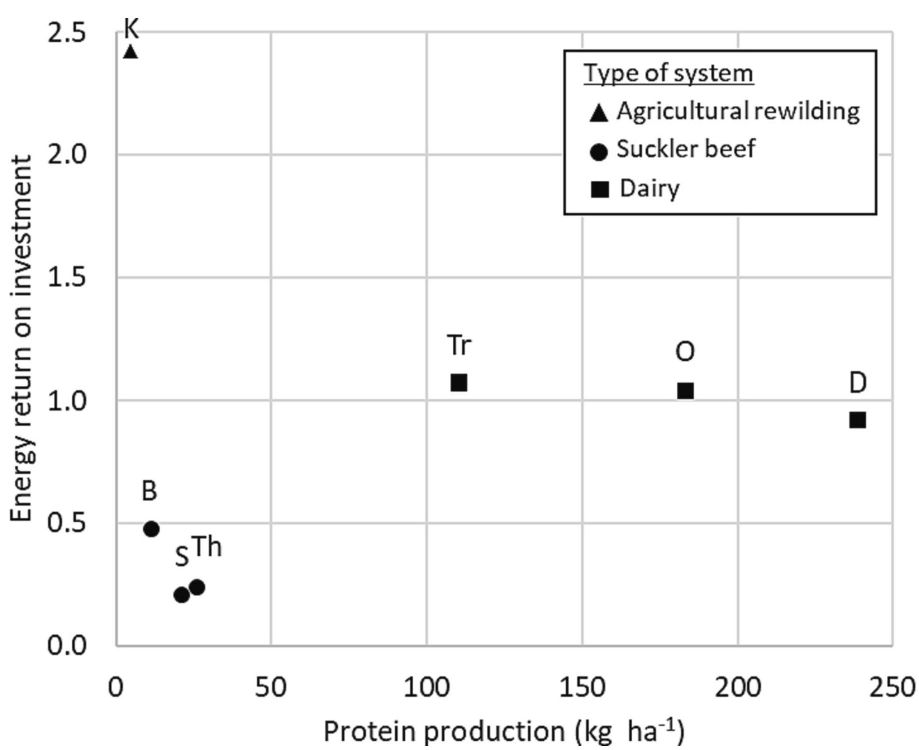

Fig. 7. Energy return on investment (i.e. energy in human-edible meat and milk divided by energy demand) as a function of human-edible animal protein production per ha of the seven systems (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval).

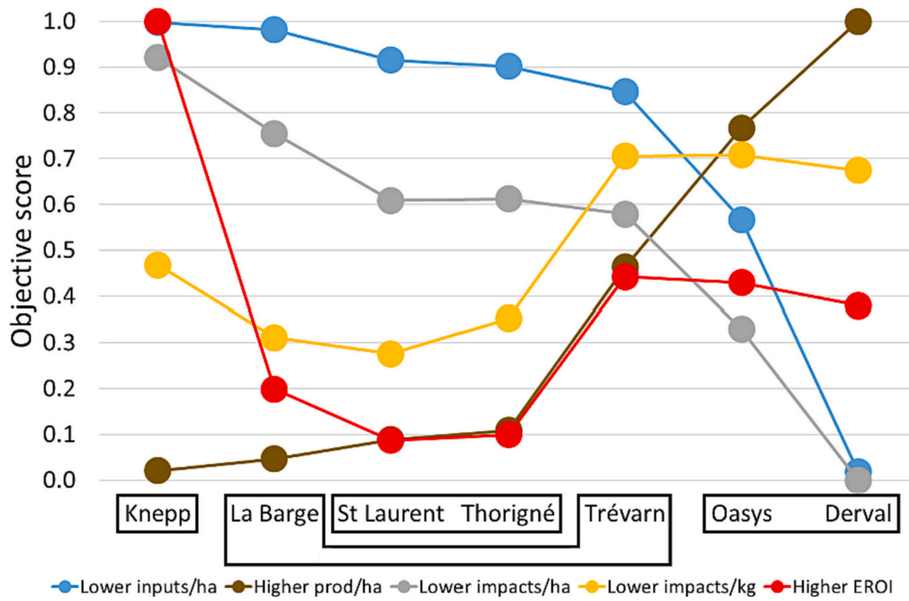

Fig. 8. Estimated "objective scores" of the seven systems in the sample. Values closer to 1.0 indicate that the objective score lay closer to the objective relative to the other farms in the sample for the three years studied. Boxes enclose systems that had similar patterns of trade-offs (either absolute or, for La Barge and Trévarn, relative to those of others of the same type (suckler beef or dairy)).

explain the large differences in productivity among systems, such as a 48-fold range of human-edible animal protein production per ha (5–239 kg ha^{-1}) (i.e. land-use efficiency), not only between suckler beef and dairy systems, but also within each of these types of system.

Impacts per ha varied greatly among the systems, especially energy demand, freshwater eutrophication and climate change, the last of which depended on the method used to estimate SOC dynamics. Impacts per kg of protein varied less but still greatly, and most so for land competition, due to differences in productivity. Compared to dairy systems, suckler beef systems tended to have lower impacts per ha, but higher impacts per kg of protein, which confirms literature results about impacts of producing protein in different animal products (Nijdam et al., 2012). Agricultural rewilding was unique, with extremely low impacts per ha and the lowest impact per kg of protein for climate change, freshwater eutrophication and energy demand. When C dynamics were considered, its climate change impact value was negative. The biodiversity-friendly systems, except sometimes S, had lower impacts per ha than the other systems did.

As in many studies, the small sample size and focus on a specific geographic area limits how broadly the results can be generalised to similar systems elsewhere. A different sample of farms would doubtless have yielded somewhat different estimates of productivity and impacts, but as each farm was chosen carefully to represent its type(s), we think that the same potential trade-offs would have been identified. In addition, this study focused exclusively on productivity and environmental impacts; consequently, more comprehensive assessment of biodiversity-friendly systems would require that future studies consider social and/or economic impacts.

4.2. *Relations between strategies and climate change and energy demand impacts*

For the climate change impact, contribution analysis showed the importance of the method used to estimate SOC dynamics. C sequestration in the soil decreased considerably the impact of the systems with a high percentage of permanent grassland (B, S, Th and Tr) and, along with C storage in biomass, decreased greatly that of the agricultural rewilding system (K). Differences in impact estimated by the two methods reflected differences in the percentage and age of permanent grassland among the systems. Research on C sequestration in agricultural systems continues, and as no consensus has been reached on how to predict it most accurately, the predictions of the methods that we used reflect this uncertainty by providing a range of potential C sequestration. Contribution analysis also confirmed the importance of enteric methane emissions and manure management, regardless of the type of system and its productivity (Gill et al., 2010). The other contributors reflected the technical characteristics and production strategies of the systems, which differed in their feed rations and the area and inputs used for the rations, highlighting that biodiversity-friendly systems used fewer inputs but more area than other systems of the same type. The climate change impact as a function of protein production differed between suckler beef and dairy systems. The systems that had the highest inputs in the sample (O and D) corresponded to average or even low-input dairy systems at the national scale in France (IDELE, 2020). The relations between productivity and climate change impact generally confirmed those described in the literature for suckler beef and dairy systems (Herron et al., 2021; Jan et al., 2019) and quantified them for very-low- to low-input systems (represented by three biodiversity-friendly systems in the sample (K, B and Tr)), which remain largely undocumented.

For energy demand, contribution analysis showed the importance of direct energy consumption regardless of the system, whereas in the literature, the main contributors to energy demand were related to feed (produced on- or off-farm) (Berton et al., 2016; Carvalho et al., 2022; Guerci et al., 2013; Salou et al., 2017; Zanni et al., 2022). Feed likely contributed less in our study due to (i) higher self-sufficiency of the systems studied, even for conventional system D, than the systems analysed in the literature and (ii) the fact that we assigned the energy consumed to produce crops used to feed cattle to direct energy consumption rather than to these crops. The latter factor, a methodological choice, seemed justified in order to compare the total amount of energy consumed directly on each farm by not dividing it among five groups of cattle feed.

4.3. *Energy return on investment*

Despite having extremely low productivity, K had the highest EROI because it used nearly no inputs. In comparison, dairy systems always had higher (and less variable) EROI than suckler beef systems because producing milk greatly increases the amount of human-edible protein produced from cattle-oriented systems. EROI has rarely been used to assess livestock systems, but it serves as a useful indicator of the energy efficiency of producing agricultural products. For comparison, we calculated the EROI of the seven composite dairy systems in France assessed by Salou et al. (2017): highland, organic, grass-based, intensive grass-based, maize-based, intensive maize-based and very intensive maize-based. The highland system represented dairy production in mountainous regions (from the AGRIBALYSE® database), while the other systems were developed from data on 69 lowland dairy systems throughout the country using factorial analysis and hierarchical clustering. Their EROI ranged from 0.51 to 0.80 and were lower than those of the dairy systems in the sample (0.92–1.06) (Table S12), due mainly to having an energy demand (14319–36,771 MJ ha^{-1}) higher than those of all systems in the sample except D (20 496 MJ ha^{-1}), in part due to lower self-sufficiency (e.g. their percentage of on-farm area in the total area ranged from 67 to 97%). Among systems in the sample of the same type, biodiversity-friendly systems had the highest EROI among suckler beef systems (i.e. EROI of B was double that of Th) and among dairy systems (i.e. EROI of Tr was 16% higher than that of D).

4.4. Trade-off patterns

LCA highlighted the importance of considering two functions of farming systems – agricultural production and land management (Salou et al., 2017) – and that trade-offs among lower input use, higher productivity, lower environmental impacts and higher EROI depended on the function assessed. The first two trade-offs identified have already been identified by studies that assessed effects of intensification on environmental impacts of cattle production (Bava et al., 2014; Bragaglio et al., 2018; Salou et al., 2017). The last two trade-offs concerned three of the sample's four biodiversity-friendly systems, which had much lower productivity but higher EROI than other systems in the sample (K) or than those of the same type (B and Tr). For some of these systems, the energy in their animal products even exceeded their energy demand (EROI >1), which could be an argument for accepting the trade-off of lower productivity. These latter patterns of trade-offs thus contrast with those of the current dominant livestock systems, which could shift the debate from relations between productivity and impacts (Dumont et al., 2019; Herrero et al., 2009) to relations between energy efficiency and land-use efficiency.

4.5. Relations between environmental impacts and biodiversity

Unlike the companion study (Mondière et al., 2023a), the present study assessed biodiversity only indirectly, by estimating potential environmental impacts on it. The impacts per ha of all of the farms in the sample were lower than those of intensive dairy and suckler beef systems in the same region, which reflected their lower degree of intensification, which tends to correlate with higher biodiversity (Emmerson et al., 2016). All biodiversity-friendly farms in the sample (as well as Th) were organic; thus, the lack of applied synthetic agrochemicals (especially pesticides) may have benefitted local biodiversity, although these benefits may be smaller than those provided by a mosaic of natural habitats at the landscape level and sowing a variety of crops in smaller fields (Tscharntke et al., 2021).

In the companion study (Mondière et al., 2023a), the four biodiversity-friendly farms tended to host more bird species (mean of 50.0) than the three other farms (mean of 32.7). Another companion study (Mondière et al., 2023b) assessed biodiversity on two of the biodiversity-friendly farms (S and Tr) somewhat more directly than the present study did, by estimating provision of five regulation and maintenance ecosystem services: global climate regulation, maintenance of chemical condition of freshwaters, mass stabilisation/control of erosion rates, pollination/seed dispersal and pest/disease control. In the study, the Ecological Focus Area calculator (Tzilivakis et al., 2015) was adapted to include permanent grassland and used to assess the influence of the land use and management of each field of the farm, as well as semi-natural areas, on these ecosystem services. Good impact scores for most fields, especially for pollination/seed dispersal and pest/disease control, indicated a good ability to host biodiversity, given biodiversity's strong influence on ecosystem services (Balvanera et al., 2006).

4.6. Perspectives

4.6.1. Implications for LCA methodology

Late Pleistocene and pre-industrial ruminant densities have been estimated at ca. 10 t live weight ha^{-1} (Manzano et al., 2023), which corresponds to the livestock density at K (ca. 10.8 t ha^{-1}). In the LCA framework, animals that rely on natural ecosystems are part of the ecosphere, and thus their emissions are excluded when assessing impacts of products derived from them, such as their meat (Fiala et al., 2020). As the agricultural rewilding system of K appears to have similar functioning and, presumably, pollutant emissions as natural ecosystems, its emissions associated with on-farm biophysical processes (e.g. CH_4, NO_3) could be considered part of the ecosphere, with only the emissions associated with human activities (e.g. herd management, fencing) being part of the technosphere. Pardo et al. (2023) recently applied a similar approach in an LCA study of transhumant sheep. Implementing this approach would equate products of K to wild game and thus decrease their impacts to near zero, which raises the question of how to consider natural baseline emissions in LCA studies.

In recent years, several LCA studies have allocated impacts of suckler beef and dairy cattle systems to non-provisioning ecosystem services that they provide (e.g. Bragaglio et al. (2020); von Greyerz et al. (2023)). In these studies, economic allocation was used based on the value of animal products produced and, as a proxy of the economic value of other ecosystem services, that of compensation payments. In these studies, up to 48% of climate change impacts of suckler beef farms was allocated to non-provisioning ecosystem services. Given the low productivity of K, whose main objective was to favour biodiversity, applying this approach to it might result in allocating an even higher percentage of impacts to these ecosystem services. This raises the question, which has been raised for extensive agricultural systems in previous studies (Salou et al., 2017; van der Werf et al., 2020), of whether it makes sense to express impacts of such systems only per kg of product.

4.6.2. The future for biodiversity-friendly livestock systems

The current agricultural context could provide opportunities for biodiversity-friendly livestock systems. First, consumption of animal products must be decreased in developed countries to achieve sustainable and healthy food systems (Willett et al., 2019), and many studies (Henchion et al., 2021; Schiavo et al., 2021) predict or recommend such a decrease. Secondly, large areas of agricultural land are abandoned in Europe (van der Zanden et al., 2017). The higher EROI for a given type of system and self-sufficiency in feed and fertilisers of biodiversity-friendly livestock systems makes them resilient to rising energy costs (Benoit and Mottet, 2023) and other disruptive events (Dardonville et al., 2022). The European Union's (EU) Common Agricultural Policy and Green Deal could also help these systems decrease their environmental impacts and increase their provision of ecosystem services (Gargano et al., 2021). However, given the relatively low productivity of biodiversity-friendly livestock systems, their development needs to be discussed in light of food security at national and global levels. For example, a recent study (Schiavo et al., 2021) estimated that if the EU were to transition to a largely agroecological system of agricultural production (e.g. phasing out soya bean imports and application of pesticides and inorganic fertilisers; expanding semi-natural habitats; extensifying livestock production) by 2050, it could meet the nutrient and calorie requirements of the EU population and even become a net exporter of calories, but only if the population's mean diet decreased its content of animal protein and calories.

5. Conclusion

The seven livestock-production systems assessed had a wide range of input use, productivity, environmental impacts (per ha and kg) and EROI. The four biodiversity-friendly livestock systems generally used fewer inputs, which resulted in lower impacts per ha and higher EROI, especially compared to systems of the same type. Productivity and environmental impacts per ha were lowest for biodiversity-friendly systems. They were particularly low for the agricultural rewilding system, whose main objective was to favour biodiversity and, to the best of our knowledge, had not been documented for this type of system. The results emphasised the need to analyse biodiversity-friendly systems and the complex relations between their functions more deeply to consider their overall environmental performances and sustainability. Current trends, such as the need to reduce consumption of animal products, land abandonment and new sources of income from compensation payments of the EU Green Deal may provide opportunities for biodiversity-friendly livestock systems.

CRediT authorship contribution statement

Aymeric Mondière: Methodology, Formal analysis, Writing – original draft. **Michael S. Corson:** Methodology, Formal analysis, Writing – review & editing. **Julie Auberger:** Software, Writing – review & editing. **Daphné Durant:** Data curation, Writing – review & editing. **Sylvain Foray:** Data curation. **Jean-Francois Glinec:** Data curation. **Penny Green:** Data curation. **Sandra Novak:** Data curation, Writing – review & editing. **Frédéric Signoret:** Data curation. **Hayo M.G. van der Werf:** Methodology, Formal analysis, Writing – review & editing.

Declaration of Competing Interest

The authors declare no competing interests.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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