The Carbon and Land Footprint of Certified Food Products

Valentin Bellassen, Marion Drut, Federico Antonioli, Ružica Brečić, Michele Donati, Hugo Ferrer-Pérez, Lisa Gauvrit, Viet Hoang, Kamilla Knutsen Steinnes, Apichaya Lilavanichakul, et al.

The carbon and land footprint of certified food products

Authors: Valentin Bellassen, Marion Drut, Federico Antonioli, Ružica Brečić, Abdoul Diallo, Michele Donati, Hugo Ferrer-López, Lisa Gauvrit, Viet Hoang, Kamilla Knutsen Steinnes, Apichaya Lilavanichakul, Edward Majewski, Agata Malak-Rawlikowska, Konstadinos Mattas, An Nguyen, Ioannis Papadopoulos, Jack Peerlings, Bojan Ristic, Marina Tomić Maksan, Áron Török (Corvinus University of Budapest, Hungary and ECO-SENSUS Research and Communication, Hungary), Gunnar Vittersø

Journal of Agricultural & Food Industrial Organization 2021

https://doi.org/10.1515/jafio-2019-0037

Abstract

The carbon and land footprint of 26 certified food products – geographical indications and organic products and their conventional references are assessed. This assessment goes beyond existing literature by 1) designing a calculation method fit for the comparison between certified food and conventional production, 2) using the same calculation method and parameters for 52 products – 26 Food Quality Schemes and their reference products – to allow for a meaningful comparison, 3) transparently documenting this calculation method and opening access to the detailed results and the underlying data, and 4) providing the first assessment of the carbon and land footprint of geographical indications. The method used is Life Cycle Assessment, largely relying on the *Cool Farm Tool* for the impact assessment. The most common indicator of climate impact, the carbon footprint expressed per ton of product, is not significantly different between certified foods and their reference products. The only exception to this pattern are vegetal organic products, whose carbon footprint is 16% lower. This is because the decrease in greenhouse gas emissions from the absence of mineral fertilizers is never fully offset by the associated lower yield. The climate impact of certified food per hectare is however 26% than their reference and their land footprint is logically 24% higher. Technical specifications directly or indirectly inducing a lower use of mineral fertilizers are a key driver of this pattern. So is yield, which depends both on *terroir* and farming practices. Overall, this assessment reinforces the quality policy of the European Union: promoting certified food is not inconsistent with mitigating climate change.

1. Introduction

The global food system – including land-use changes, production of inputs and post-farm emissions – is responsible for 22%-37% of global greenhouse gas emissions (Rogissart et al., 2019a). Even in industrialized countries like France, it is responsible for 24% of national emissions, excluding land-use changes. Such a large responsibility imposes the food system as a cornerstone of climate action. European and national climate strategies have acknowledged this necessity and aim at halving agricultural emissions by 2050 (European Commission, 2018; MTES, 2018).

Consumer choices are increasingly put forward as a promising way towards this goal (Hoolohan et al., 2013; Moran et al., 2018; Poore and Nemecek, 2018; Smith et al., 2019). However, without proper price signals reflecting the carbon footprint of products, information is necessary to drive these choices. Pilot initiatives for carbon footprint labelling on products have been launched in 13 countries, with limited demonstrated success so far (Rogissart et al., 2019b). An alternative to carbon footprint labelling is to provide information on the carbon footprint of food quality schemes (FQS) which consumers already recognize. This is one of the objectives of this article.

Two of the most recognized FQS are organic products and geographical indications. The organic label certifies environmentally friendly farming practices and in particular the absence of chemical inputs. Geographical indications (GIs) guarantee either the location of food processing (Protected Geographical Indication - PGI) or the location of both farmers and processors (Protected Designation of Origin - PDO). While these two labels are not well recognized *per se* by European consumers (Hartmann et al., 2019), the products they certify and protect such as Parmigiano Reggiano and Comté cheeses are very famous in their respective countries, if not internationally. Indeed, organic products and geographical indications already capture 4% and 5.7% respectively of total food retail sales in European countries where data is available (Chever et al., 2012; FiBL, 2017). This is why we estimate the carbon and land footprints of certified food products in this paper, differentiating three FQS: organic, PDO, and PGI.

The carbon footprint of organic products has already been investigated in many studies over the last 15 years, although even reviews and meta-analysis fail to come to a consensual conclusion. Mondelaers et al. (2009) finds that the carbon footprint of organic products is worse than conventional ones, whereas Meier et al. (2015), Tuomisto et al. (2012) and Clark and Tilman (2017) are inconclusive, arguing that the result may depend on product types. Most interestingly, Meier et al. (2015) concludes that it is not yet possible to draw a conclusive picture on the topic, because detailed calculation methods and parameters – often not fully transparent in published articles – likely overlook important differences between organic and conventional production.

To our knowledge, the carbon footprint of geographical indications has however never been investigated. This is a serious knowledge gap as these quality schemes capture a much higher market share than organic products in some food categories with a heavy carbon footprint. In France for example, GI cheeses capture 11.4% of the market (INAO, 2019) – which is close to the EU average of 10% (Chever et al., 2012) – while the market share of organic cheese is below 2% (Lambotte et al., 2020). Most studies on the carbon footprint of organic farming identify the trade-off between lower emissions per hectare and lower yield as key in determining whether organic products are more climate-friendly on a *per ton* basis. Another objective of this article is to assess to which extent this trade-off can be generalized to other FQSs such as geographical indications.

In this article, we estimate the carbon and land footprint of FQS, and test whether it is significantly different from similar conventional products. The sectoral specificities as well as differences between FQS are investigated. We also conduct a sensitivity analysis on our results. We thus go beyond existing literature by 1) following the recommendations of Meier et al. (2015) to design a calculation method fit for the comparison between organic and conventional production, 2) using the same calculation method and parameters for 52 products – 26 FQSs and their reference products, similar but non-certified – to allow for a meaningful comparison, 3) transparently documenting this calculation method and opening access to the detailed results and the underlying data, and 4) providing the first assessment of the carbon and land footprint of geographical indications.

2. Material and methods

2.1. Method

2.1.1. Following LCA principles

The carbon and land footprints are common indicators of the environmental impact of food products. The carbon footprint estimates the amount of greenhouse gas (GHG) emissions which are emitted for each ton of final product, expressed in $tCO_2e\ ton^{-1}$. Similarly, land footprint estimates the amount of land which is necessary to produce one ton of final product, expressed in $ha\ ton^{-1}$. For estimating the carbon and land footprints of food products, we follow the attributional life cycle assessment principles, as laid out in the ILCD handbook (JRC, 2010). The functional unit is defined as one ton of final product (eg. ripened cheese, fish sauce, …). Alternatively, we use one hectare of agricultural land as the functional unit, yielding a *per hectare carbon footprint* expressed in $tCO_2e\ ha^{-1}$. Although the ton of final product is the most common functional unit in the literature, the combined use of both functional units is recommended (Lambotte et al., 2021; van der Werf et al., 2009).

The system boundaries are cradle to processing plant: feed production, farming operations and processing activities are included. Emissions from transportation between different levels of the value chain up to the final retailer are estimated but excluded from most of the analysis. Given that the majority of transport-related emissions come from exported products, they are considered not to be an intrinsic characteristic of the products. They are therefore analysed in detail in a separate article dedicated to foodmiles and transport emissions (Drut et al., this issue). Finally, when a process jointly generates several products – eg. milk and meat from dairy cows – its impact is allocated in proportion to the economic value of the products.

2.1.2. Cool Farm Tool and *ad hoc* changes

Customized public calculation tool

The inventory analysis and in particular the data collection strategy is presented in section 2.2. For the impact assessment, we use a customized form of the Cool Farm Tool version 2.0 beta 3 (Hillier et al., 2011), as recommended by the review of Colomb et al. (2012) when the system boundaries extend beyond the farm. The Cool Farm Tool mostly follows the IPCC guidelines (IPCC, 2006) to estimate greenhouse gas emissions from the agricultural sector and implements them into an Excel file. Different spreadsheets are used for different emissions sources: crops, livestock, processing, transport, … They are ultimately aggregated into a synthesis spreadsheet and allocated between different co-products if relevant. The tool covers the most important emission sources from cradle to retail stores. In particular, emissions occurring during the production of farm inputs such as fertilizers are included, as well as processing and transport.

Here we slightly adapt the tool to better estimate the differences between certified and non-certified food products. Most importantly, data on both the dry matter intake and the animal productivity are collected which allows to account for differences on these aspects between FQSs and their reference products as recommended by Meier et al. (2015). In the case of dairy cows, feed digestibility, a highly uncertain parameter according to IPCC (2006), is therefore be adjusted so that these two values be consistent. Minor changes and corrections are also implemented (correction of the methane conversion factor for dairy cows, of the formula summing up emissions from manure management, setting the share of female offspring to one to restrict the system boundaries to the relevant product, allowing for the attribution of some “feed-emissions” to pasture, and allowing for up to 10 user-defined feed types).

Sensitivity analysis

The calculator used in this study – the Cool Farm Tool (Hillier et al., 2011) – makes two key assumptions that are worth exploring through a sensitivity analysis. Firstly, unlike the IPCC (IPCC, 2006) and most life cycle assessments but similarly to more recent works (Carlson et al., 2016), the calculator uses a non-linear relationship between N_2O emissions and fertilizer use derived from Bouwman et al (2002). As a result, even fields where no fertilizer is applied emit some N_2O and the marginal impact of one kilogram of nitrogen increases with the total amount applied. Applying both relationships to national or regional averages in nitrogen fertilization, Carlson et al. (2016) finds that the non-linear relationship decreases carbon footprint estimates by 30%.

Secondly, the emissions stemming from the application of organic fertilizers such as manure and compost are attributed to the crop they fertilize rather than to the production – generally livestock – which generated them. While this approach is the most frequently used in the life assessment literature and is retained by the IPCC for inventories (IPCC, 2006), it is questionable: manure is often waste produced in excess by livestock farms and breeders are usually happy to get rid of it for free. Therefore, attributing all its emissions to the production which generated them may be warranted, which would likely decrease the carbon footprint of organic products.

In the *Tier 1 N_2O* scenario, the non-linear relationship between nitrogen inputs and N_2O emissions used in the Cool Farm tool is replaced by an IPCC Tier 1[1] estimate. In *manure allocation* scenario, the traditional allocation of manure emissions is modified: direct and indirect N_2O emissions from field application of manure – usually allocated to crops growing in the field – are allocated to the animals generating the manure. The rationale is that in many cases, manure is overabundant and herders either give it away for free or even pay for its collection and disposal. In those cases where manure is closer to a waste from animal production than from an input to vegetal production, common LCA practice recommends that its impact be attributed to animal production (JRC, 2010). The methods to compute these alternative estimates of N_2O emissions from nitrogen inputs are detailed in annex 2.2.

2.2. Data collection strategy

2.2.1. Choice of products and their references

26 FQS products were selected for this study (Table 1). Choices aimed at a diversity of sectors – animal, vegetal and unfed seafood/fish – and FQS – organic, PDO, PGI – while taking into account country-specific constraints (some FQS simply do not exist in some countries for some sectors). Ultimately, the cases are evenly distributed across FQS, while regarding sectors, the unfed seafood/fish sector has much few cases (3) than the vegetal and animal sectors (Table 3).

[1] The IPCC distinguishes three types of methods – Tiers – for estimating greenhouse gas emissions. Tier 1 is the most generic type. Tier 2 often requires country-specific parameters and Tier 3 mostly consists in complex and customized models.

Table 1. Sample characteristics. Red, green and blue lines highlight the sector (animal, vegetal and seafood respectively). The indicated turnover is either at processing or farm level, whichever is higher. Arfini and Bellassen (2019) provides a detailed description of each value chain, its structure, its governance and its sustainability performance.

Product name	Country	Product description	Type of FQS	Processed?	Turnover of FQS (M€ yr-1)	Reference product	Turnover of reference product (M€ yr-1)
Dalmatian prosciutto	Croatia	Dry pork ham	PGI	Yes	4.20	Local non-PGI firm	7.45
PDO olive oil	Croatia	Olive oil	PDO	Yes	0.25	National average	53.33
Comte cheese	France	Hard pressed cooked cheese from cow milk	PDO	Yes	504.19	Similar uncertified cheese (Emmental) or national average (cow cheese)	1 203.10
Organic flour	France	Wheat flour	Organic	Yes	34.80	National average	5 180.00
Saint-Michel bay bouchot mussels	France	Mussels produced on "bouchots"	PDO	No	25.45	National average (TSG Bouchot mussels)	116.13
Organic rice	France	Rice	Organic	Yes	17.64	Non-organic rice (mostly PGI)	28.43
Organic pork	Germany	Raw meat	Organic	Yes	69.00	National average	11 571.53
Organic yoghurt	Germany	Organic yoghurt from cow milk	Organic	Yes	387.00	National average	8 995.00
Zagora apples	Greece	Apple	PDO	No	10.11	Kissavos apples (non-GI apples from another region)	1.40
Kastoria apples	Greece	Apple	PGI	No	7.50	Kissavos apples (non-GI apples from another region)	1.40
Gyulai sausage	Hungary	Sausage	PGI	Yes	55.00	Non-PGI Hungarian sausage	277.87
Kalocsai paprika powder	Hungary	Paprika powder	PDO	Yes	10.75	Imported Chinese pepper milled in Hungary	8.30
Parmigiano Reggiano cheese	Italy	Hard pressed cooked cheese from cow milk	PDO	Yes	1 009.94	Biraghi cheese (similar non-PDO cheese)	4 636.55
Organic tomatoes	Italy	Organic tomato	Organic	No	68.57	Conventional processed tomatoes in the same region (Emilia-Romagna)	595.35
Opperdoezer Ronde potatoes	Netherlands	Early potato	PDO	No	2.77	Regular potato in neighbouring IJsselmeerpolders region	491.54
Lofoten stockfish	Norway	Dried fish	PGI	No	71.24	Clipfish (cod)	409.79
Organic salmon	Norway	Salmon	Organic	Yes	144.71	Conventional salmon	6 613.31
Organic pasta	Poland	Pasta	Organic	Yes	0.52	Simulated conventional farms with sample characteristics	3.63
Kaszubska	Poland	Strawberry	PGI	No	0.64	National average	164.83
Sjenica cheese	Serbia	Sheep cheese	PGI	Yes	1.21	National average (cow cheese)	396.18
Organic raspberries	Serbia	Frozen rapsberries	Organic	Yes	4.37	National average	144.46
Sobrasada Porc Negre	Spain	Raw, cure saussage from pork meat	PGI	Yes	1.80	National average	10.92
Ternasco de Aragon	Spain	Unprocessed lamb meat	PGI	No	16.97	Non-PGI lamb in the same region (Aragon)	48.06
Thung Kula Rong-Hai Hom Mali rice	Thailand	Rice	PGI	No	300.74	Non certified rice from the same region (90% of GI rice is organic as well)	10 195.05
Doi Chaang coffee	Thailand	Coffee	PGI	Yes	756.00	Non-PGI coffee from the same province	20.00
Phu Quoc Fish	Vietnam	Fish sauce	PDO	Yes	3.43	Non-PDO fish sauce from same region	20.03
Buon Ma Thuot coffee	Vietnam	Coffee	PGI	Yes	89.58	Non-PGI coffee from Dak Lak province in Vietnam	732.49

In order to mitigate the influence of other possible drivers of carbon footprint than the participation to a FQS, such as country- or region-specific features, only the difference between a FQS and its reference product is analysed. This strategy is similar to the rationale of *controlled trials*. For this reason, we have paid close attention to select only products with an appropriate reference. For instance, for Sjenica cheese, a sheep-milk cheese from Serbia initially in the sample, the only possible reference within Serbia was a cow-milk cheese. Because the difference then lies more in the difference between sheep and cow systems than between FQS and non-FQS systems, this product is removed from the analysis. To improve efficiency and comparability, guidelines were provided for data collection (eg. relying to the extent possible on secondary data, interviewing key stakeholders in the value chain, …) and for the selection of the reference, non-FQS, product. Note that the reference product can be an actual non-certified product (eg. Kissavos apples) or the average conventional product in the same country.

With regards key input variables to the CoolFarmTool, FQS crops or fodder receive much less mineral nitrogen, much more organic nitrogen and their yield is 19% lower than reference crops or fodder (Table 2). In the animal sector, livestock density is also substantially lower in FQS farms. The descriptive statistics of the absolute value of these variables is provided in annex 3. The key data sources used for each product is provided in annex 4. Arfini and Bellassen (2019) provides a detailed description of each product, its value chain, its governance and its overall sustainability performance assessment.

Table 2. Descriptive statistics of the relative difference between FQS and their reference products

Variable	n	min	Q1	median	Q3	max
Mineral nitrogen	23	-100%	-100%	-49%	-3%	37%
Organic nitrogen	23	-84%	-6%	41%	675%	Inf%
Crop or fodder yield	23	-61%	-36%	-19%	0%	115%
Amount of final product per hectare	23	-87%	-49%	-20%	-6%	127%
Amount of raw product per ton of final product	26	-12%	0%	0%	22%	319%
Share of co-products in total value	26	-67%	0%	0%	0%	Inf%
Livestock density	8	-72%	-28%	-21%	-9%	0%
Renewal rate	8	-18%	-3%	-1%	0%	4%
Dry matter intake of breeding adults	8	-46%	-0%	0%	3%	38%
Milk production	3	-10%	-7%	-4%	-4%	-4%
Lifetime of fattening adults	6	-25%	-15%	0%	18%	180%
Dry matter intake of fattening adults	6	-3%	0%	4%	23%	74%

2.2.2. Data collection

In order to be able to collect data on the 52 products within a reasonable amount of time, the variables requested the Cool Farm Tool were divided in two categories, based on the literature (eg. Röös et al., 2014; Weber and Matthews, 2008):

- ✓ Key variables, which are the focus of the data collection effort;
- ✓ And secondary variables, which are collected only if readily available from existing datasets. Otherwise, default values from the literature are used.

Key variables are the variables which have been shown to be paramount drivers of carbon footprint. Even when land-use related emissions are excluded, 68% of food system emissions occur at farm level (Rogissart et al., 2019a). This is why most of these key variables are at this level: yield, fertilizer inputs, diesel use, herd size, … At processing level, only energy use is always considered a key variable. Additional key variables are added for products for which previous studies identified specific important emission sources (eg. waste water treatment for coffee processing, refrigeration gases for fish capture, …).

The life cycle inventory of animal products should in principle include data on herd sizes. However, while the method was being tested over three pilot cases, data on typical lifetimes for the different animal stages (eg. heifer, dairy cow, cull cow) proved to be more readily available and reliable than data is on the size of a farm herds (eg. when fattening and breeding does not occur in the same farm). However, both datasets are equivalent: from the lifetime data, a typical herd structure is reconstructed, starting with an arbitrary number (100) of reproductive females. In the dairy sector, emissions from males are neglected. On the one hand, overlooking reproductive males slightly underestimates farm emissions. On the other hand, leaving out male offspring overestimates the share of milk in farm emissions as part of the emissions of pregnant animals could arguably be allocated to the gestation of male calves. However, both sources of emissions are small and likely more or less offset one another.

All mineral nitrogen inputs are assumed to be ammonitrate and emissions from the production of other chemical inputs are neglected. Emissions from the energy required to spreading of all inputs, as well as other field practices such as irrigation or tillage, are accounted through the quantities of energy requirements from all energy types (gas, diesel, electricity, …). Land-use related emissions are not estimated because consensual and simple methods are not yet available.

The detailed list of variables, separated between key and secondary for each food sub-sector as well as all the spreadsheets including the raw data, their source, and the resulting estimated carbon footprints can be downloaded at https://www2.dijon.inra.fr/cesaer/informations/food-sustainability-indicators/.

2.2.3. Quality control procedure

Finally, for each product, a thorough quality check procedure was implemented to limit the risk of misreporting data. The three key aspects of this procedure were 1) to record all data, their date and source in a shared spreadsheet, 2) to separate the person who collected data from the person who estimated the carbon footprint, and 3) to come up with a written and consensual interpretation of the results between these people (annex 1). More details on the data collection procedure are provided in Bellassen et al. (in press).

2.3. Statistics

Non-parametric tests relying on rank are used: the Wilcoxon signed-rank test to test whether the median is different from zero and the Kruskal-Wallis test to test whether different groups belong to the same populations. Indeed, given the small sample size – 26 at most – these tests are much less sensitive to outliers than classical parametric tests. They also don't rely on a normality assumption which is difficult to ascertain in small samples.

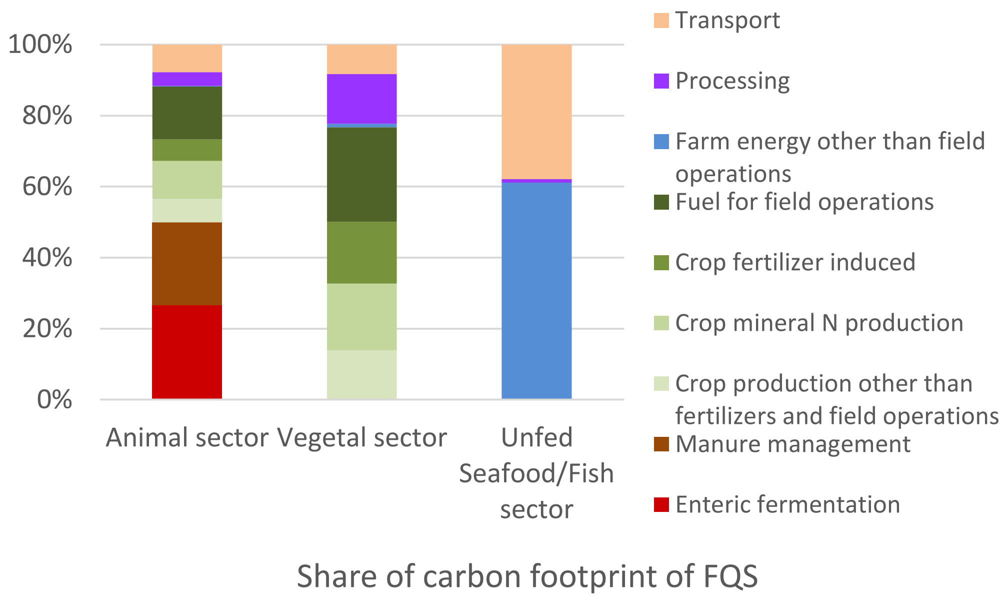
3. Results

3.1. Most emissions occur before the farm gate

For animal products, 92% of the carbon footprint is emitted before the animal or its products leave the farm (Figure 1). This figure drops to 84% for vegetal products and to 61% for unfed seafood and fish. This dominance of farm processes and fertilizer production is consistent with the literature (Rogissart et al., 2019a; Röös et al., 2014; Weber and Matthews, 2008). In the vegetal sector, the production and use of fertilizers are responsible for around 37% of emissions from crop production and fuel use for field operations are responsible for 27%. The rest comes from crop residues, background emissions and, in the case of flooded rice, anaerobic methanogenesis. In the animal sector, enteric fermentation and manure management are responsible for half of the emissions, while fertilizers and fuel for crop operations emit 16% and 15% respectively.

Because transport represents only a small fraction – except for the 3 unfed seafood/fish cases – of the carbon footprint for most FQS products and because the system boundaries on which it is has been assessed is not strictly identical across cases, it is not considered in the rest of the analysis. The carbon and land footprint of each FQS and its reference product, broken down into the same categories of emission sources, is available in annex 1.

Figure 1. Average composition of the carbon footprint of FQS products (per ton of finished product)

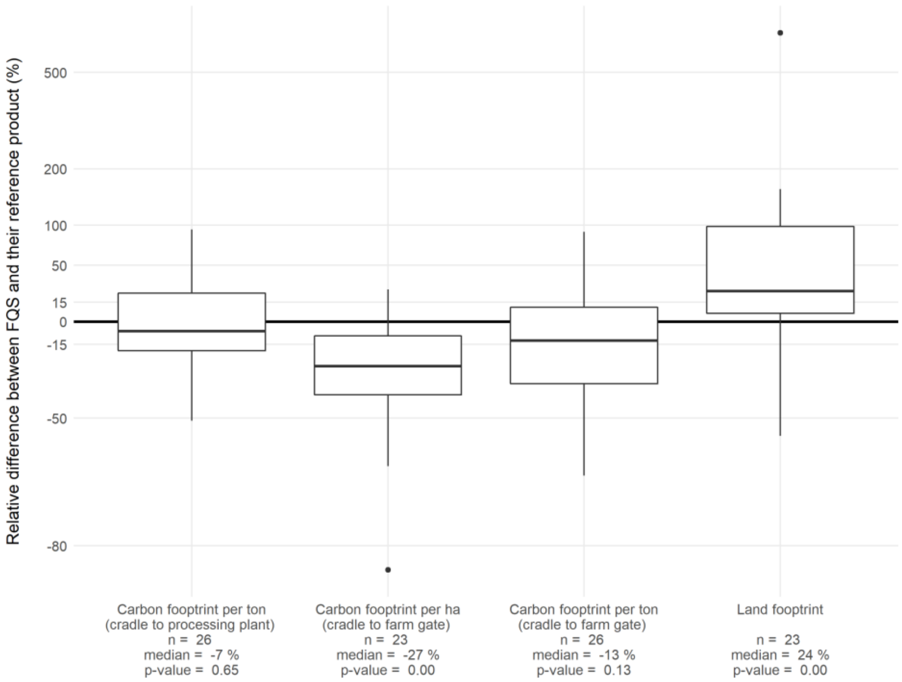


3.2. Carbon and land footprint of food quality scheme

When the most common definition of carbon footprint is used – that is using one ton of final product as the functional unit, the median difference between FQSs and their reference products is not significantly different from zero (Figure 2). The comparison is however more favourable to FQSs when it is performed at the level of original products (eg. milk for cheese, wheat for flour, …): more than two third of FQSs are not substantially worse than their reference at farm level. The climate impact of FQSs becomes clearly lower than their reference when the comparison is made on an area basis: the median difference is then significantly lower. The land footprint of FQS is clearly higher than their reference: the median difference is 24% higher, with three fourth of the FQSs having a substantially higher land footprint.

Figure 2. Carbon footprint and land footprint of Food quality schemes

The p-value indicates the probability that the median is different from zero (Wilcoxon signed-rank test). Boxes indicate the first and third quartiles with the median as a vertical bar within them. Whiskers indicate the largest values which is not further than 1.5 times the interquartile distance from the box. Points are outliers: the two points correspond to the Sobrasada Porc Negre case where FQS pigs live twice longer than their conventional counterparts.



3.3. Carbon and land footprint per subgroup

There is no clear-cut difference in carbon footprint per ton of final products between most of the different FQS and sectoral categories. The only exception are products which are at the same time vegetal and organic (Table 3). They have a significantly lower (-16%, p-value = 0.06) carbon footprint than their reference. When raw products are considered instead of final products, the median difference between FQS and their reference drops from -7% (p-value = 0.62) to -13% (p-value = 0.13), primarily due to FQS animal products whose difference in carbon footprint drops from +12% (p-value = 0.25) to no difference (0%, p-value = 0.65) (Table S 2).

On a per hectare basis however, the GHG emissions of FQS value chains are 26% smaller than their reference (p-value < 0.005, Figure 1), which is significantly different from zero. The difference between vegetal organic products and their reference is even starker (-54%). The per hectare carbon footprint of vegetal organic products is also significantly lower than the two other vegetal subgroups (Table S 1). The per hectare carbon footprint of the entire organic subgroup is also significantly lower than that of the PDO and PGI subgroups (-22% and -14% respectively).

Table 3. Difference in carbon footprint per ton of final product for different categories

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test.

Subgroup	Nb of cases	Median difference	Probability that the median is different from 0
All	26	-7%	0.62
Organic	8	-15%	0.11
PDO	8	-3%	0.84
PGI	10	10%	0.85
Animal	9	12%	0.25
Unfed Seafood/Fish	3	-6%	0.50
Vegetal	14	-15%	0.30
Animal_Organic	3	-13%	1.00
Animal_PDO	2	39%	1.00

Animal_PGI	4	21%	0.13
Unfed Seafood/Fish_PDO	2	-3%	1.00
Unfed Seafood/Fish_PGI	1	-48%	na
Vegetal_Organic	5	-16%	0.06
Vegetal_PDO	4	-10%	1.00
Vegetal_PGI	5	-14%	0.63

3.4. *Sensitivity analysis*

Our sensitivity analysis does not alter the key results in terms of carbon footprint difference between FQS and their reference products (annex 2). Some p-values are however altered: the median carbon footprint per ton of the entire organic category becomes significantly lower than its reference when a closed nitrogen cycle is ensured and emissions from organic fertilizers are attributed to animals (from -15% to -19%, p-value decreases from 0.11 to 0.05). To the contrary, the median carbon footprint per ton of the animal category becomes significantly higher than its reference (+12%, p-value decreases from 0.2 to 0.07) when the non-linear estimate of N_2O emissions from fertilizers is replaced by the IPCC tier 1 method.

Replacing the non-linear estimate of N_2O emissions from fertilizers by the IPCC tier 1 method also increases the absolute carbon footprint of products by an average 13% (Table 4). This increase is most pronounced for organic products, possibly because of their low yield. To the contrary, ensuring a closed nitrogen cycle and attributing emissions from organic fertilizers to animals does not substantially alter the absolute values of carbon footprint. The most impacted category is again organic products although this time the alternative method decreases its average footprint by -7% as the emissions from the large amounts of organic nitrogen imported from other farms are being attributed to exporting farms.

Table 4. Relative difference in carbon footprint (tCO2e ton-1) in the sensitivity analysis

	Tier1 N_2O		Manure allocation	
Subcategory	**FQS**	**Reference**	**FQS**	**Reference**
Animal sector	12%	4%	-2%	-2%
Vegetal sector	17%	16%	-4%	-1%
Unfed Seafood/Fish sector	0%	0%	0%	0%
Geographical indications	7%	8%	-1%	-2%
of which PDO	1%	4%	-1%	-1%
Organic	28%	14%	-7%	0%
All	13%	10%	-3%	-1%

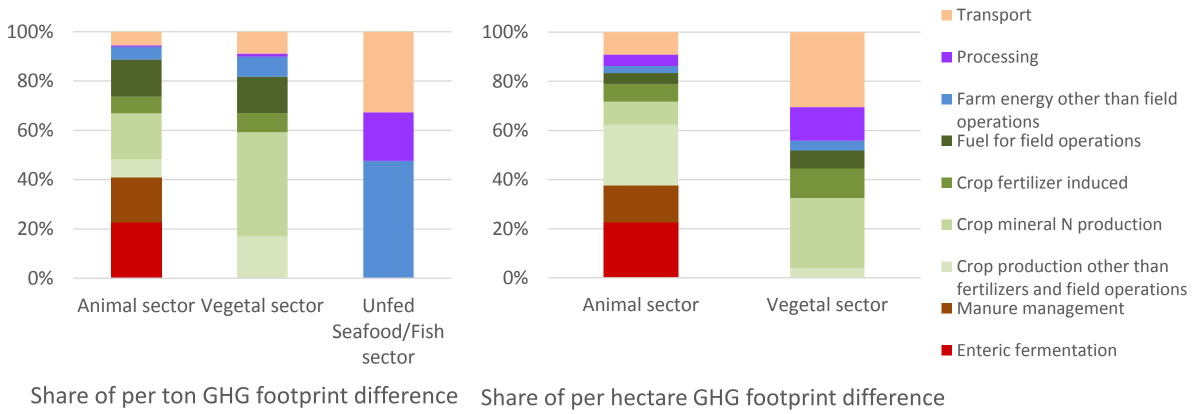
4. Discussion

4.1. *The role of technical specifications: fertilizer use, product concentration and animal efficiency*

In vegetal sectors, the bulk of the differences in carbon footprint is driven by fertilizer production and use (Figure 3). In 10 out of 23 relevant cases (unfed seafood is irrelevant), the technical specifications of FQS products play a direct role in driving fertilizer use down compared with conventional products: mineral fertilizers are forbidden in organic production and in PGI TKR Hom Mali rice, although they are partly substituted with organic ones, and limited in PDO Comté cheese. In an additional 5 cases, the specifications are indirectly driving down the use of mineral fertilizers, either through feed composition (e.g. ban on maize silage promoting alfalfa for PDO Parmigiano Reggiano and outdoor rearing for PGI Sobrasada Porc Negre), promotion of manure as a substitute (PGI Kaszucka strawberries and PDO Kalocsai paprika) or shorter growing season (PDO Opperdoezer potatoes). In two other cases, the lower and more efficient use of fertilizers does not directly stem from the technical specifications but is indirectly related to the FQS via the access to technical advice by cooperatives involved in the FQS (Kastoria and Zagora apples). Note that in the remaining 6 cases however, the FQS influences neither directly nor indirectly the use of fertilizers. That is, among others, the case of PGIs (5 out of 9 relevant PGIs neither directly nor indirectly impact fertilizer use). The only case where fertilizer use is higher than the reference is Doi Chaang coffee. Thus, although higher FQS prices are in theory an incentive for farmers to increase productivity at the intensive margin, we do not observe an overall increase in fertilizer use in FQS farms.

Figure 3. Average contribution of each emission source to the difference in carbon footprint

"Crop production other …" includes crop residues, background emissions and, in the case of flooded rice, anaerobic methanogenesis. The notion of "per hectare GHG footprint" is meaningless for unfed seafood and fish, hence their absence from the right hand side of the figure.



In animal cases, two other important drivers come to play. The first is simply product concentration: given that Parmigiano Reggiano cheese is drier than its reference cheese, twice as much milk is required to make one ton of Parmigiano Reggiano cheese than to make one ton of its reference. Accordingly, its carbon footprint is almost twice higher. Although three other products also require substantially more raw material per ton of final product than their reference – Dalmatian ham, Gyulai sausage and organic pasta – this is not a general trend. Several FQS products such as Croatian olive oil or Comté cheese even require less raw material than their reference thanks to a higher processing efficiency or a higher quality of the raw material.

The second pertains to how efficiently the animal herd transforms feed into food. The more ingested matter is required per unit of food, the more GHG are emitted from enteric fermentation, manure management, and, of course, feed production. On these aspects, FQS tend to perform worse than their reference although for a variety of reasons, often related to technical specifications. Sobrasada pigs for example live twice longer and exercise much more than their reference, thus "wasting" much more feed in maintenance and exercise. Similar although less pronounced differences drive a lower feed to food conversion efficiency in organic yoghurt, Comté cheese and organic pork. In the latter, the lower number of piglets per sow also increases the relative "deadweight" of sows on the carbon footprint of fattened pig meat.

Finally, the technical specifications of 5 out of the 26 products – including 4 vegetal products – either directly or indirectly promote manual harvest or processing. In two instances, solar drying is also required. These requirements lower energy use and therefore the carbon footprint, but the impact is small due to the small share of emissions directly related to energy use even when both farm and processing levels are included.

4.2. Yield and terroir

Another important factor driving the differences in per ton carbon footprint and in land footprint is yield. Many studies on organic products have already identified the trade-off between lower emissions per hectare and lower yield as the key driver of the difference in carbon footprint between organic and conventional products. This is particularly true when the consequences of lower yield on indirect land-use change are included in the assessment (Bellora and Bureau, 2016; Searchinger et al., 2018).

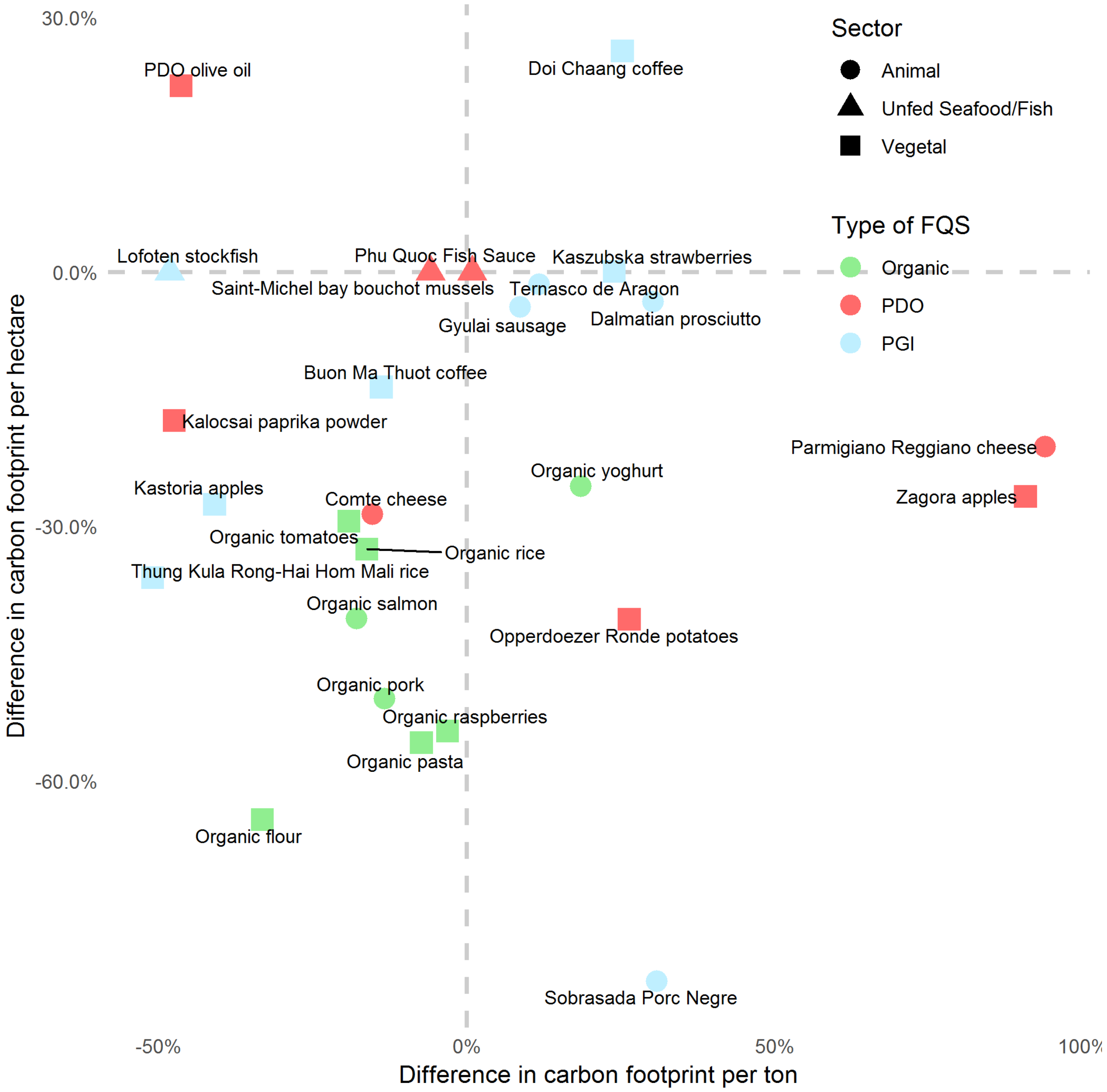
In our sample, we also find a 19% lower yield – median difference – for FQS crops and fodder. The difference is more pronounced for organic value chains (-33%) than for PDO (-18%) and PGI (0%). In terms of final product, the difference is even larger: -40% for organic, -34% for PDO and -12% for PGI. These yield differences are consistent with the results of existing meta-analysis for organic products (Ponisio et al., 2015; Seufert et al., 2012). This explains why, although only two FQS have a higher per hectare carbon footprint, the difference in per ton GHG emissions between FQS and their reference is more evenly distributed around zero (Figure 4).

The pedo-climatic conditions or *terroir* as they are referred to in the GI literature[2] (eg. Belmin et al., 2018) often drive this difference in yield, but it can go either way. In some cases such as Croatian olive oil, Kastoria apple or Kalocsai paprika powder, the pedo-climatic conditions allow for higher yield in the FQS. To the contrary, the pedo-climatic specificities of Zagora apple and Doi Chaang coffee constrain their yield.

Naturally, this *terroir* effect interacts with crop practices: irrigation and higher technicity certainly help Kalocsai paprika farmers in achieving higher yields while the shorter growing season mandated by the Opperdoezer potato technical specifications necessarily reduces crop yield.

[2] Rigorously speaking, *terroir* is a combination of pedo-climatic conditions and traditional know-how.

Figure 4. Differences in carbon footprint per hectare and per ton



4.3. *Unfed seafood and fish*

The unfed seafood and fish sector has a peculiar carbon footprint pattern because the two usually dominant emission sources – namely enteric fermentation and fertilizer use – do not occur in this sector. As a result, their carbon footprint – largely driven by diesel use for boat operation – is modest compared to other animal products. Differences in carbon footprint between FQS and their reference are negligible for mussels and fish sauce, but more substantial for stockfish. Most of the advantage of Lofoten stockfish pertains to lower fuel needs to capture the fish because the technical specifications request that fishermen fish "around Lofoten and Vesteralen". To a lesser extent, energy savings at processing level – such as sun drying and the absence of freezing for Lofoten stockfish – also contribute to improve the carbon footprint of Lofoten stockfish.

4.4. *The carbon and land footprint of organic products*

Our findings reinforce existing evidence that organic vegetal products have a lower carbon footprint than their conventional reference while animal organic products have a similar, if not higher, carbon footprint. Indeed, Meier et al. (2015) reports a negative median difference for both organic arable crops and organic fruits and vegetables (-7 % and -3% respectively) whereas it is positive for their animal products counterparts, with the exception of dairy products (- 1%). In Clark and Tilman (2017), the only category whose carbon footprint is significantly lower than conventional farming is *fruits*, whereas the only category whose carbon footprint is significantly higher than conventional farming is *dairy and eggs*. In Tuomisto et al. (2012), the only category whose carbon footprint is significantly lower than conventional farming is *other crops* while the only category whose carbon footprint is significantly higher than conventional farming is *pork*. The additional evidence we bring is stronger – all 5 vegetal organic products have a lower carbon footprint than their reference, possibly because we apply a consistent method across cases, therefore limiting the noise that heterogeneity of methods creates in existing meta-analysis.

Our results also confirm past evidence from these meta-analysis of a lower carbon footprint of organic products on a per hectare basis, as well as a higher land footprint. Although we focused on the indicator per ton as most of the literature, both product-based and area-based indicators may be relevant, depending on hypotheses on the elasticities of demand. Indeed, if demand is infinitely elastic or if there is no substitute for FQS products, consumers fully adjust to any change in the quantity produced and the product-based indicators are irrelevant. To the contrary, if demand is inelastic or if standard products are perfect substitutes for FQS products, a reduced production of FQS products is offset by an increase in production elsewhere, diminishing the relevance of area-based indicators. Several elements argue for imperfect substitutability and non-zero demand elasticity, justifying the relevance of the *per hectare* metric. First, FQS are substantially more expensive than conventional products so if they were perfect substitutes, FQS would disappear. Second, the diet of consumers eating a higher share of organic products has been shown to contain fewer animal-based products, incidentally leading to a lower carbon footprint of the diet as a whole (Baudry et al., 2019; Lacour et al., 2018).

4.5. Methodological issues & limits

4.5.1. System boundaries and unaccounted factors

While the system boundaries retained for this study – from cradle to processing plant gate – is already wider than many existing studies, we have enough data to expand it to transport-related emissions in several cases. However, the extent to which this information is policy-relevant is debatable: consumers broadly know where the product comes from – and hence how far it travelled before reaching the retail outlet – and they also broadly know how much they emitted to reach the retail outlet (consumer transport weighs around one third of transport-related emissions from the food system (Barbier et al., 2019)). And in any case, transportation only represents 10-15% of food-related emissions (Figure 1, (Barbier et al., 2019)). This is why this article focuses on emissions without transport, while another contribution is dedicated to foodmiles and transport-related emissions (Drut et al., this issue).

Biomass and soil carbon changes have also not been considered. These changes are indeed negligible when land use and management is kept constant over long time periods (IPCC, 2006; Pellerin et al., 2019). This is not true however in the first decades following change. Thus, organic farming has been shown to increase soil carbon stocks by an average 0.07-0.27 tC ha^{-1} $year^{-1}$, although this is likely an indirect consequence of higher manure inputs and crop rotations than a direct effect of the technical specifications (Gattinger et al., 2012). In addition, conversions from conventional systems to FQS would, in many cases, involve sowing grasslands over cropland which would increase soil carbon stocks (Lambotte et al., 2021). To the contrary however, such conversions would often result in decreasing yields, which in turn are predicted to have a negative effect on biomass and soil carbon stocks through indirect land-use changes (Bellora and Bureau, 2016; Searchinger et al., 2018). Therefore, there is no obvious prediction as to how including biomass and soil carbon changes would impact our results.

Although the carbon footprint estimates of both FQS and their reference products can be substantially changed by our sensitivity analysis on the method (up to an average +13% for FQS products for the *Tier1* scenario), these alterations largely cancel out and the differences in carbon footprint between FQS and their reference product are robust. In all three scenarios, the carbon footprint per ton of FQS is not significantly different from their reference products, except for vegetal organic products. Other second order factors such as crop residue management (except for rice), type of mineral fertilizer (eg. ammonitrate vs urea) or juvenile death rate have been overlooked in this analysis and could be included in future refinements. However, these second order factors are not expected to change the absolute values by more than a few percentage points and therefore, similarly to our sensitivity analysis, we can expect that their accounting would not alter our results.

4.5.2. Improving the statistical surveys of the agri-food sector

While the data collection procedure followed common guidelines and includes thorough quality checks, data sources and sample sizes were allowed to vary from one case to the other in order to fit with the national circumstances. Some cases – e.g. organic pork, organic yoghurt, Comté cheese, … – were able to rely on secondary data bolstered by a large sample size while other cases had to collect primary data on a small – usually five to ten – sample of farms (e.g. Kastoria and Zagora apple, organic raspberries, …). This heterogeneity clearly generates some noise. The most straightforward way to remove this pitfall would be a systematic identification of firms involved in other FQS than organic within EU-wide statistical surveys such as the Farm Accountancy Data Network and the Statistical Business Survey. Furthermore, these surveys would also need to be slightly modified in order to cover the key drivers of carbon footprint: as demonstrated by the FP7 FLINT project (Vrolijk et al., 2016), deriving environmental indicators from the current FADN is not straightforward.

Similarly, common generic guidelines were followed by the different people responsible for data collection to select the reference product, but again, this does not mean that reference products meet the exact same criteria across case studies (Table S 12). For example, while the reference product was preferably produced in the same administrative region as the FQS, it sometimes proved to be impossible to find a substantial production of the non-certified product within the region. One remedy could be to explore the sensitivity of results to a systematic use of national averages derived from large databases such as FADN, AROPAj and Mueller et al. (2012). Indeed, this would provide a more homogeneous reference but often at the expense of regional matching and updated data.

5. Conclusion

The most common indicator of climate impact, the carbon footprint expressed per ton of product, is not significantly different between FQS and their reference products. The only exception to this pattern are vegetal organic products, whose carbon footprint is 16% lower. This is because the GHG gains from the absence of mineral fertilizers is never fully offset by the associated lower yield. Although there were weak signals consistent with this finding in the literature, the consistent method deployed in this study over a large sample of products substantially strengthens the evidence. In organic value chains, the yield versus emissions trade-off tilts differently for vegetal products – lower carbon footprint despite lower yield – than for animal products – equal or higher carbon footprint despite lower per hectare emissions. In addition, the climate impact of FQS per hectare is however lower than their reference and their land footprint is logically higher.

Technical specifications directly or indirectly inducing a lower use of mineral fertilizers are a key driver of this pattern. So is yield, which depends both on *terroir* and farming practices. One can think that the much higher heterogeneity of requirements in geographical indications, compared with organic farming, explains why the median difference in carbon footprint is lower for the latter.

Overall, this assessment reinforces the quality policy of the European Union: despite generally lower yields, the carbon footprint per ton of FQS is similar to their reference products and the carbon footprint per hectare is lower. Therefore, although mitigating climate change is not an objective of FQS, we demonstrate that promoting FQS is at least not inconsistent with climate change mitigation – when climate impact is assessed on a *per ton* basis – or even synergetic – when climate impact is assessed on a *per hectare* basis. Our results also support the recent initiative from the French FQS agency (INAO) to promote the inclusion of environmental considerations in technical specifications (INAO, 2016): we show that specifications directly or indirectly limiting the use of mineral fertilizers are an important driver of the difference in carbon footprint. The best example of this driver is vegetal organic products for which the ban on mineral fertilizers results in a lower carbon footprint, despite the associated lower yield.

The main limit of this assessment is the small sample size. Two avenues can be explored to overcome this limit. First, given that the calculator we used is publicly available and that all raw data and results can be downloaded, one can hope that new FQS products can be assessed with the same method by other researchers or practitioners, increasing sample size. Second, as the FADN is scheduled to include physical variables in 2020, it may be sufficient to conduct the assessment at farm level which would open the access to a very large sample. This would however require that the involvement of a farm in a FQS is informed in the FADN, which for the moment is only the case for organic farming.

Acknowledgements

This project has received funding from the European Union’s Horizon 2020 research and innovation programme under grant agreement No 678024. The authors would like to thank all the people and institutions who collected or provided raw data for this publication. Unfortunately, they are too numerous to be exhaustively listed here but they will hopefully recognize themselves through the list of data sources and the associated book. Statistical analysis were conducted using the R software.

References

ADEME, 2017. Agribalyse v1.3 [WWW Document]. URL http://www.ademe.fr/expertises/produire-autrement/production-agricole/passer-a-laction/dossier/levaluation-environnementale-agriculture/loutil-agribalyser

Agreste, 2011. Enquête Pratiques culturales 2011. Ministère de l’agriculture, de l’agro-alimentaire et de la forêt, Paris, France.

Agreste, 2010. Pratiques culturales 2006 (No. 8), Agreste Les Dossiers. Ministère de l’agriculture, de l’agro-alimentaire et de la forêt, Paris, France.

Almeida, J., Achten, W.M.J., Verbist, B., Heuts, R.F., Schrevens, E., Muys, B., 2014. Carbon and Water Footprints and Energy Use of Greenhouse Tomato Production in Northern Italy: Carbon and Water Footprints, and Energy Use of Tomato Production. Journal of Industrial Ecology 18, 898–908. https://doi.org/10.1111/jiec.12169

ARAL, 2017. Costo razioni bovine [WWW Document]. URL http://www.aral.lom.it/web/Pagine/leggi_area.asp?ART_ID=2029&MEC_ID=143&MEC_IDFiglie=347 (accessed 11.4.17).

Arfini, F., Bellassen, V. (Eds.), 2019. Sustainability of European Food Quality Schemes: Multi-Performance, Structure, and Governance of PDO, PGI, and Organic Agri-Food Systems. Springer International Publishing, Cham. https://doi.org/10.1007/978-3-030-27508-2

Ari Tchougoune, J.-C., M.;. Mouret, 2018. Pour une gestion économe de la fertilisation phosphatée des rizières. Cardère Editeur Educagri, Avignon FRA, Dijon FRA, pp. 177–184.

Aubin, J., Fontaine, C., Callier, M., Roque d’orbcastel, E., 2018. Blue mussel (Mytilus edulis) bouchot culture in Mont-St Michel Bay: potential mitigation effects on climate change and eutrophication. The International Journal of Life Cycle Assessment 23, 1030–1041. https://doi.org/10.1007/s11367-017-1403-y

Barbier, C., Couturier, C., Pourouchottamin, P., Cayla, J.-M., Silvestre, M., Pharabod, I., 2019. L’empreinte énergétique et carbone de l’alimentation en France, Club Ingénierie Prospective Energie et Environnement. IDDRI, Paris, France.

Barbier, J.M., 2018. Améliorer l’efficience de la fertilisation azotée des rizières. Cardère Editeur Educagri, Avignon FRA, Dijon FRA, pp. 161–175.

Basset-Mens, C., van der Werf, H.M.G., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. Agriculture, Ecosystems & Environment 105, 127–144. https://doi.org/10.1016/j.agee.2004.05.007

Batalla, I., Knudsen, M.T., Mogensen, L., Hierro, Ó. del, Pinto, M., Hermansen, J.E., 2015. Carbon footprint of milk from sheep farming systems in Northern Spain including soil carbon sequestration in grasslands. Journal of Cleaner Production 104, 121–129. https://doi.org/10.1016/j.jclepro.2015.05.043

Baudry, J., Pointereau, P., Seconda, L., Vidal, R., Taupier-Letage, B., Langevin, B., Allès, B., Galan, P., Hercberg, S., Amiot, M.-J., Boizot-Szantai, C., Hamza, O., Cravedi, J.-P., Debrauwer, L., Soler, L.-G., Lairon, D., Kesse-Guyot, E., 2019. Improvement of diet sustainability with increased level of organic food in the diet: findings from the BioNutriNet cohort. The American Journal of Clinical Nutrition 109, 1173–1188. https://doi.org/10.1093/ajcn/nqy361

Bellora, C., Bureau, C., 2016. How green is organic ? Indirect effects of making EU agriculture greener (Presented at the 19th Annual Conference on Global Economic Analysis, Washington DC, USA). Global Trade Analysis Project (GTAP), Department of Agricultural Economics, Purdue University, West Lafayette, IN.

Belmin, R., Casabianca, F., Meynard, J.-M., 2018. Contribution of transition theory to the study of geographical indications. Environmental Innovation and Societal Transitions 27, 32–47. https://doi.org/10.1016/j.eist.2017.10.002

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Modeling global annual N2O and NO emissions from fertilized fields: N2O and NO emissions from fertilizers. Global Biogeochemical Cycles 16, 28-1-28–9. https://doi.org/10.1029/2001GB001812

CA Occitanie, CER Occitanie, 2016. Agriculture bio - Edition 2016, Les dossiers d'Agri'scopie. Chambres d'agricultures Occitanie et CER Occitanie.

CA Rhône-Alpes, 2012. Agriculture biologiques - Fiches technico-économiques - Blé tendre. Chambres d'agricultures Rhône-Alpes.

Carlson, K.M., Gerber, J.S., Mueller, N.D., Herrero, M., MacDonald, G.K., Brauman, K.A., Havlik, P., O'Connell, C.S., Johnson, J.A., Saatchi, S., West, P.C., 2016. Greenhouse gas emissions intensity of global croplands. Nature Climate Change 7, 63–68. https://doi.org/10.1038/nclimate3158

Chever, T., Renault, C., Renault, S., Romieu, V., 2012. Value of production of agricultural products and foodstuffs, wines, aromatised wines and spirits protected by a geographical indication (GI). European Commission & AND International, Brussels, Belgium.

CITEPA, 2013. Rapport national d'inventaire pour la France au titre de la convention cadre des Nations Unies sur les changements climatiques et du protocole de Kyoto. CITEPA, Paris, France.

Clark, M., Tilman, D., 2017. Comparative analysis of environmental impacts of agricultural production systems, agricultural input efficiency, and food choice. Environ. Res. Lett. 12, 064016. https://doi.org/10.1088/1748-9326/aa6cd5

Clune, S., Crossin, E., Verghese, K., 2017. Systematic review of greenhouse gas emissions for different fresh food categories. Journal of Cleaner Production 140, 766–783. https://doi.org/10.1016/j.jclepro.2016.04.082

Colomb, V., Bernoux, M., Bockel, L., Chotte, J.L., Martin, S., Martin-Phipps, C., Mousset, J., Tinlot, M., Touchemoulin, O., 2012. Review of GHG calculators in agriculture and forestry. ADEME, IRD, and FAO.

Delmotte, S., 2011. Participatory assessment of scenarios: perspectives for agricultural systems in Camargue, South of France. (Theses). Montpellier SupAgro.

Espinoza-Orias, N., Stichnothe, H., Azapagic, A., 2011. The carbon footprint of bread. Int J Life Cycle Assess 16, 351–365. https://doi.org/10.1007/s11367-011-0271-0

European Commission, 2018. In-depth analysis in support of the Commission communication COM(2018) 773 - A Clean Planet for all - A European long-term strategic vision for a prosperous, modern, competitive and climate neutral economy. European Commission, Brussels, Belgium.

FiBL, 2017. FiBL Statistics [WWW Document]. URL https://statistics.fibl.org/ (accessed 5.24.19).

Fritsche, U.R., Eberle, U., 2009. Greenhouse-Gas Emissions from the Production and Processing of Food, Working paper. Oeko Institut e.V., Darmstadt, Germany.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mäder, P., Stolze, M., Smith, P., Scialabba, N.E.-H., Niggli, U., 2012. Enhanced top soil carbon stocks under organic farming. PNAS 109, 18226–18231. https://doi.org/10.1073/pnas.1209429109

Giovannucci, D., Lewin, B., Swinkels, R., Varangis, P., 2004. Socialist Republic of Vietnam Coffee Sector Report. SSRN Electronic Journal. https://doi.org/10.2139/ssrn.996116

Hartmann, M., Yeh, C.H., Amilien, V., Csillag, P., Filipovic, J., Giraud, G., Kuč, V., Menozzi, D., Poschwatta, O., Roos, G., Saïdi, M., Tocco, B., Veneziani, M., Vreden, T., 2019. Report on quantitative research findings on European consumers' perception and valuation of EU food quality schemes as well as their confidence in such measures.

Hillier, J., Walter, C., Malin, D., Garcia-Suarez, T., Mila-i-Canals, L., Smith, P., 2011. A farm-focused calculator for emissions from crop and livestock production. Environmental Modelling & Software 26, 1070–1078. https://doi.org/10.1016/j.envsoft.2011.03.014

Hokazono, S., Hayashi, K., 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. Journal of Cleaner Production 28, 101–112. https://doi.org/10.1016/j.jclepro.2011.12.005

Hoolohan, C., Berners-Lee, M., McKinstry-West, J., Hewitt, C.N., 2013. Mitigating the greenhouse gas emissions embodied in food through realistic consumer choices. Energy Policy 63, 1065–1074. https://doi.org/10.1016/j.enpol.2013.09.046

Hou, Y., Bai, Z., Lesschen, J.P., Staritsky, I.G., Sikirica, N., Ma, L., Velthof, G.L., Oenema, O., 2016. Feed use and nitrogen excretion of livestock in EU-27. Agriculture, Ecosystems & Environment 218, 232–244. https://doi.org/10.1016/j.agee.2015.11.025

IDELE, 2012. Alimentation des bovins : rations moyennes et autonomie alimentaire. Institut de l'élevage.
IDELE Franche-Comté, 2016. Les cas types laitiers - chiffres clés. Institut de l'élevage de Franche-Comté.
INAO, 2019. Les produits sous signe d'identification de la qualité et de l'origine - Chiffres clés 2017. INAO, Paris, France.
INAO, 2016. L'agro-écologie s'installe dans les SIQO. La lettre de l'INAO 1.
IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Chapter 4: Agriculture, Forestry and Other Land Uses (AFOLU). IGES, Hayama, Japan.
Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Revisiting the Life Cycle Assessment of mussels from a sectorial perspective. Journal of Cleaner Production 18, 101–111. https://doi.org/10.1016/j.jclepro.2009.10.009
Italian FADN, 2014. Italian Farm Accountancy Data Network.
Jayet, P.-A., 2017. AROPAj model [WWW Document]. URL https://www6.inra.fr/basc/Recherche/Modeles/AROPAJ
JRC, 2010. International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. European Commission - Joint Research Centre - Institute for Environment and Sustainability, Luxembourg. Publications Office of the European Union.
Juin, H., 2015. Les pertes alimentaires dans la filière Céréales. Innovations Agronomiques 48, 76–96.
Killian, B., Rivera, L., Soto, M., Navichoc, D., 2013. Carbon Footprint across the Coffee Supply Chain: The Case of Costa Rican Coffee. Journal of Agricultural Science and Technology B 3 151–170.
Knudsen, M.T., Yu-Hui, Q., Yan, L., Halberg, N., 2010. Environmental assessment of organic soybean (Glycine max.) imported from China to Denmark: a case study. Journal of Cleaner Production 18, 1431–1439. https://doi.org/10.1016/j.jclepro.2010.05.022
Kool, A., Blonk, H., Ponsioen, T., Sukkel, W., Vermeer, H.M., Vries, J.W. de, Hoste, R., 2009. Carbon footprints of conventional and organic pork. Assessments of typical production systems in the Netherlands, Denmark, England and Germany. Blonk Milieu Advies [etc.], Gouda [etc.].
Lacour, C., Seconda, L., Allès, B., Hercberg, S., Langevin, B., Pointereau, P., Lairon, D., Baudry, J., Kesse-Guyot, E., 2018. Environmental Impacts of Plant-Based Diets: How Does Organic Food Consumption Contribute to Environmental Sustainability? Front. Nutr. 5. https://doi.org/10.3389/fnut.2018.00008
Lambotte, M., De Cara, S., Bellassen, V., 2020. Once a quality-food consumer, always a quality-food consumer? Consumption patterns of organic, label rouge, and geographical indications in French scanner data. Rev Agric Food Environ Stud. https://doi.org/10.1007/s41130-020-00121-z
Lambotte, M., De Cara, S., Brocas, C., Bellassen, V., 2021. Carbon footprint and economic performance of dairy farms: The case of protected designation of origin farms in France. Agricultural Systems 186, 102979. https://doi.org/10.1016/j.agsy.2020.102979
Leip, A., Weiss, F., Wassenaar, T., Perez, I., Fellmann, T., Loudjani, P., Tubiello, F., Grandgirard, D., Monni, S., Biala, K., 2010. Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS). European Commission, Joint Research Centre.
Lesschen, J.P., van den Berg, M., Westhoek, H.J., Witzke, H.P., Oenema, O., 2011. Greenhouse gas emission profiles of European livestock sectors. Animal Feed Science and Technology 166–167, 16–28. https://doi.org/10.1016/j.anifeedsci.2011.04.058
Lindenthal, T., Markut, T., Hörtenhuber, S., Rudolph, G., Hanz, K., 2010. Klimabilanz biologischer und konventioneller Lebensmittel im Vergleich. Ökologie und Landbau 153, 51–54.
Maina, J., Mutwiwa, U., Gareth, K., Githiru, M., 2016. Evaluation of Greenhouse Gas Emissions along the Small-Holder Coffee Supply Chain in Kenya 2, 111–120.
Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., Stolze, M., 2015. Environmental impacts of organic and conventional agricultural products – Are the differences captured by life cycle assessment? Journal of Environmental Management 149, 193–208. https://doi.org/10.1016/j.jenvman.2014.10.006
Meisterling, K., Samaras, C., Schweizer, V., 2009. Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. Journal of Cleaner Production 17, 222–230. https://doi.org/10.1016/j.jclepro.2008.04.009
Mondelaers, K., Aertsens, J., Van Huylenbroeck, G., 2009. A meta-analysis of the differences in environmental impacts between organic and conventional farming. British Food Journal 111, 1098–1119. https://doi.org/10.1108/00070700910992925
Monfreda, C., Ramankutty, N., Foley, J.A., 2008. Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000: global crop areas and yield in 2000. Global Biogeochemical Cycles 22, n/a-n/a. https://doi.org/10.1029/2007GB002947
Monier, C.-T., C.;. Mouret, J.C.;. Vadon, A.;. Soulard, 2018. La paille de riz en Camargue : du sous-produit au coproduit. Cardère Editeur Educagri, Avignon FRA, Dijon FRA, pp. 273–292.
Moran, D., Wood, R., Hertwich, E., Mattson, K., Rodriguez, J.F.D., Schanes, K., Barrett, J., 2018. Quantifying the potential for consumer-oriented policy to reduce European and foreign carbon emissions. Climate Policy 1–11. https://doi.org/10.1080/14693062.2018.1551186
MTES, 2018. Projet de Stratégie Nationale Bas-Carbone. Ministère de la transition écologique et solidaire, Paris, France.
Mueller, N.D., Gerber, J.S., Johnston, M., Ray, D.K., Ramankutty, N., Foley, J.A., 2012. Closing yield gaps through nutrient and water management. Nature 490, 254–257. https://doi.org/10.1038/nature11420

Noponen, M.R.A., Edwards-Jones, G., Haggar, J.P., Soto, G., Attarzadeh, N., Healey, J.R., 2012. Greenhouse gas emissions in coffee grown with differing input levels under conventional and organic management. Agriculture, Ecosystems & Environment 151, 6–15. https://doi.org/10.1016/j.agee.2012.01.019

Odegard, I., Bijleveld, M., Naber, N., 2015. Food Commodity Footprints. Global GHG footprints and water scarcity footprints in agriculture. CE Delft, Delft, The Netherlands.

Opio, C., Gerber, P., Mottet, A., Falcucci, A., Tempio, G., MacLeod, M., Vellinga, T., Henderson, B., Steinfeld, H., 2013. Greenhouse gas emissions from ruminant supply chains – A global life cycle assessment. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Passion Céréales, 2017. Des chiffres et des céréales.

Pellerin, S., Bamière, L., Launay, C., Martin, R., Schiavo, M., Angers, D., Augusto, L., Balesdent, J., BASILE-DOELSCH, I., Bellassen, V., Cardinael, R., Cécillon, L., Ceschia, E., Chenu, C., Constantin, J., Darroussin, J., Delacote, P., Delame, N., Gastal, F., Gilbert, D., Graux, A.-I., Guenet, B., Houot, S., Klumpp, K., LETORT, E., Litrico, I., Martin, M., Menasseri-Aubry, S., Meziere, D., Morvan, T., Mosnier, C., ROGER-ESTRADE, J., Saint-André, L., Sierra, J., Therond, O., Viaud, V., Grateau, R., Le Perchec, S., Savini, I., Rechauchère, O., 2019. Stocker du carbone dans les sols français, quel potentiel au regard de l'objectif 4 pour 1000 et à quel coût ? (Contract). Agence de l'Environnement et de la Maîtrise de l'Energie.

Pelletier, N., Tyedmers, P., 2007. Feeding farmed salmon: Is organic better? Aquaculture 272, 399–416. https://doi.org/10.1016/j.aquaculture.2007.06.024

Ponisio, L.C., M'Gonigle, L.K., Mace, K.C., Palomino, J., de Valpine, P., Kremen, C., 2015. Diversification practices reduce organic to conventional yield gap. Proc. R. Soc. B 282, 20141396. https://doi.org/10.1098/rspb.2014.1396

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science 360, 987–992. https://doi.org/10.1126/science.aaq0216

RD&T, 2012. Carbon emissions from chilled and frozen cold chains for a typical UK Sunday roast chicken meal.

RIAS Inc., 2016. Comparing the Environmental Footprint of B.C.'s Farm-Raised Salmon to OtherFood Protein Sources. BC Salmon farmers association.

Ribaudo, F., 2011. Ribaudo F. (2011), Prontuario dell'Agricoltura Italiana (Handbook of Italian Agriculture), Hoepli. ed. Milano, Italy.

Rinaldi, S., Barbanera, M., Lascaro, E., 2014. Assessment of carbon footprint and energy performance of the extra virgin olive oil chain in Umbria, Italy. Science of The Total Environment 482–483, 71–79. https://doi.org/10.1016/j.scitotenv.2014.02.104

Ripoll-Bosch, R., de Boer, I.J.M., Bernués, A., Vellinga, T., 2011. Greenhouse gas emissions throughout the life ycle of Spanish lamb-meat: A comparison of three production systems. Options Méditerranéennes. Séries A. Mediterranean Seminars 125–130.

Rodríguez, A.B., Bodas, R., Fernández, B., López-Campos, O., Mantecón, A.R., Giráldez, F.J., 2007. Feed intake and performance of growing lambs raised on concentrate-based diets under cafeteria feeding systems. animal 1, 459. https://doi.org/10.1017/S1751731107683803

Rogissart, L., Foucherot, C., Bellassen, V., 2019a. Estimating greenhouse gas emissions from food consumption: methods and results. I4CE, Paris, France.

Rogissart, L., Foucherot, C., Bellassen, V., 2019b. Food policies and climate: a literature review. I4CE, Paris, France.

Röös, E., Karlsson, H., 2013. Effect of eating seasonal on the carbon footprint of Swedish vegetable consumption. Journal of Cleaner Production 59, 63–72. https://doi.org/10.1016/j.jclepro.2013.06.035

Röös, E., Sundberg, C., Hansson, P.-A., 2014. Carbon Footprint of Food Products, in: Muthu, S.S. (Ed.), Assessment of Carbon Footprint in Different Industrial Sectors. Singapore.

Röös, E., Sundberg, C., Hansson, P.-A., 2011. Uncertainties in the carbon footprint of refined wheat products: a case study on Swedish pasta. The International Journal of Life Cycle Assessment 16, 338–350. https://doi.org/10.1007/s11367-011-0270-1

Ruini, L., Ferrari, E., Meriggi, P., Marino, M., Sessa, F., 2013. Increasing the Sustainability of Pasta Production through a Life Cycle Assessment Approach, in: Prabhu, V., Taisch, M., Kiritsis, D. (Eds.), Advances in Production Management Systems. Sustainable Production and Service Supply Chains. Springer Berlin Heidelberg, Berlin, Heidelberg, pp. 383–392. https://doi.org/10.1007/978-3-642-41263-9_48

SARF, 2012. Carbon Footprint Of Scottish Suspended Mussels And Intertidal Oysters (No. SARF078). Scottish Aquaculture Research Forum.

Searchinger, T.D., Wirsenius, S., Beringer, T., Dumas, P., 2018. Assessing the efficiency of changes in land use for mitigating climate change. Nature 564, 249. https://doi.org/10.1038/s41586-018-0757-z

Seufert, V., Ramankutty, N., Foley, J.A., 2012. Comparing the yields of organic and conventional agriculture. Nature 485, 229–232. https://doi.org/10.1038/nature11069

Smith, W.K., Nelson, E., Johnson, J.A., Polasky, S., Milder, J.C., Gerber, J.S., West, P.C., Siebert, S., Brauman, K.A., Carlson, K.M., Arbuthnot, M., Rozza, J.P., Pennington, D.N., 2019. Voluntary sustainability standards could significantly reduce detrimental impacts of global agriculture. PNAS 201707812. https://doi.org/10.1073/pnas.1707812116

SORS, 2015. Agricultural database, Statistical Office of Serbia.

STUARD, 2017. Experimental farm network [WWW Document]. URL www.stuard.it (accessed 10.1.18).

Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Does organic farming reduce environmental impacts? – A meta-analysis of European research. Journal of Environmental Management 112, 309–320. https://doi.org/10.1016/j.jenvman.2012.08.018

Vagnoni, E., Franca, A., Breedveld, L., Porqueddu, C., Ferrara, R., Duce, P., 2015. Environmental performances of Sardinian dairy sheep production systems at different input levels. Science of The Total Environment 502, 354–361. https://doi.org/10.1016/j.scitotenv.2014.09.020

van der Werf, H.M.G., Kanyarushoki, C., Corson, M.S., 2009. An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. Journal of Environmental Management 90, 3643–3652. https://doi.org/10.1016/j.jenvman.2009.07.003

Venkat, K., 2012. Comparison of Twelve Organic and Conventional Farming Systems: A Life Cycle Greenhouse Gas Emissions Perspective. Journal of Sustainable Agriculture 36, 620–649. https://doi.org/10.1080/10440046.2012.672378

Vrolijk, H., Poppe, K., Keszthelyi, S., 2016. Collecting sustainability data in different organisational settings of the European Farm Accountancy Data Network. Studies in Agricultural Economics 118, 138–144. https://doi.org/10.7896/j.1626

Wang, X., Zou, C., Zhang, Y., Shi, X., Liu, J., Fan, S., Liu, Y., Du, Y., Zhao, Q., Tan, Y., Wu, C., Chen, X., 2018. Environmental impacts of pepper (Capsicum annuum L) production affected by nutrient management: A case study in southwest China. Journal of Cleaner Production 171, 934–943. https://doi.org/10.1016/j.jclepro.2017.09.258

Warner, D.J., Davies, M., Hipps, N., Osborne, N., Tzilivakis, J., Lewis, K.A., 2010. Greenhouse gas emissions and energy use in UK-grown short-day strawberry (Fragaria xananassa Duch) crops. The Journal of Agricultural Science 148, 667–681. https://doi.org/10.1017/S0021859610000493

Weber, C.L., Matthews, H.S., 2008. Food-Miles and the Relative Climate Impacts of Food Choices in the United States. Environmental Science & Technology 42, 3508–3513. https://doi.org/10.1021/es702969f

West, P.C., Gerber, J.S., Engstrom, P.M., Mueller, N.D., Brauman, K.A., Carlson, K.M., Cassidy, E.S., Johnston, M., MacDonald, G.K., Ray, D.K., Siebert, S., 2014. Leverage points for improving global food security and the environment. Science 345, 325–328. https://doi.org/10.1126/science.1246067

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities (No. Defra Research Project IS0205). Cranfield University and Defra.

Winther, U., Ziegler, F., Skontorp Hognes, E., Emanuelsson, A., Sund, V., Ellingsen, H., 2009. Carbon footprint and energy use of Norwegian seafood products (No. SFH80 A096068). SINTEF.

World Bank, 2017. World Bank VITS, average of import data over 2013-2016.

Ytrestøyl, T., Aas, T.S., Åsgård, T., 2015. Utilisation of feed resources in production of Atlantic salmon (Salmo salar) in Norway. Aquaculture 448, 365–374. https://doi.org/10.1016/j.aquaculture.2015.06.023

Marion Drut, Federico Antonioli, Michael Boehm, Ružica Brečić, Liesbeth Dries, Hugo Ferrer-López, Lisa Gauvrit, Viet Hoang, Kamilla Knutsen Steinnes, Apichaya Lilavanichakul, Edward Majewski, Orachos Napasintuwong, An Nguyen, Kallirroi Nikolaou, Ioannis Papadopoulos, Bojan Ristic, Burkhard Schaer, Torvald Tangeland, Marina Tomić Maksan, Péter Csillag, Áron Török, Efthimia Tsakiridou, Mario Veneziani, Gunnar Vittersø, Valentin Bellassen (this issue). Foodmiles: the logistics of food chains applied to Food Quality Schemes. Journal of Agriculture and Food Industrial Organization.

Bellassen, V., Arfini, F., Antonioli, F., Bodini, A., Boehm, M., Brečić, R., Chiussi, S., Csillag, P., Donati, M., Dries, L., Drut, M., Duboys de Labarre, M., Ferrer, H., Filipović, J., Gauvrit, L., Gil, J. M., Gorton, M., Hoàng, V., Hilal, M., Knutsen Steinnes, K., Lilavanichakul, A., Malak-Rawlikowska, A., Majewski, E., Monier-Dilhan, S., Muller, P., Napasintuwong, O., Nikolaou, K., Nguyen, M., Nguyễn Quỳnh, A., Papadopoulos, I., Peerlings, J., Török, A., Poméon, T., Ristic, B., Schaer, B., Stojanovic, Z., Tocco, B., Tomic Maksan, M., Veneziani, M., Vitterso, G. (in press). Sustainability performance of certified and non-certified food. Research Data Journal for the Humanities and Social Sciences.

Destatis, 2017. „Land- und Forstwirtschaft, Fischerei -Betriebe mit ökologischem Landbau“, Agrarstrukturerhebung, Statistisches Bundesamt, Fachserie 3, Reihe 2.2.1; 28.6.2017 page 43

Gorn A. (2017). AMI Markt Bilanz Öko-Landbau 2017 (eBook)

Mesić et al. (2014). Studija Utjecaj poljoprivrede na onečišćenje površinskih i podzemnih voda u Republici Hrvatskoj. Sveučilište u Zagrebu Agronomski fakultet

Grubić, G (2012), Ishrana ovaca, Poljoprivredni fakultet, Beograd. http://www.agrif.bg.ac.rs/files/profiles/31/1073/Ishrana%20ovaca.pdf

Poljosfera (2017). http://www.poljosfera.rs/agrosfera/agro-teme/stocarstvo/znacaj-dubrenja-livada-i-pasnjaka/

Statistical Office of the Republic of Serbia (SORS), 2015. www.stat.gov.rs

Jaume, J., Joy, S., & González, J. (2012). Presentation’s effect of granulated or wet barley during the finishing phase on the productive yield of Majorcan Black Pig. Options Méditerranéennes. Séries A. Mediterranean Seminars, 101, 341–343. Retrieved from http://om.ciheam.org/om/pdf/a101/00006704.pdf

Toshiyuki Nagumo, Shintaro Tajima, Sanae Chikushi & Ayako Yamashita (2013) Phosphorus Balance and Soil Phosphorus Status in Paddy Rice Fields with Various Fertilizer Practices, Plant Production Science, 16:1, 69-76

Srisompun, O. et al. (2017). Impacts of post harvest management and storage of rice farmers in Northeast Thailand

NDF (2018). Norwegian Directorate of Fisheries. https://www.fiskeridir.no/Yrkesfiske/Statistikk-yrkesfiske/Omregningsfaktor

Cargill Aqua Nutrition (2017). Sustainability Report. Retrieved from: https://www.cargill.com/doc/1432118057937/aquaculture-sustainability-report-2017.pdf

Oraqua (2013). Assessment of organic aquaculture for further development of European regulatory framework. Deliverable 7.2. Progress report and financial report for 1st period. Retrieved from: https://www.oraqua.eu/content/download/110517/1547852/file/D7.2+Progress+report+first+period.pdf?version=1

FAO (2018). Cultured Aquatic Species Information Programme. Salmo salar. Text by Jones, M. In: FAO Fisheries and Aquaculture Department [online]. Rome. Updated 1 January 2004. [Cited 19 September 2018]. Retrieved from: http://www.fao.org/fishery/culturedspecies/Salmo_salar/en

KTBL, 2017: „Ökologischer Landbau – Daten für die Betriebsplanung im ökologischen Landbau“, in „Datensammlung Kuratorium für Technik und Bauwesen in der Landwirtschaft (KTBL)“, Darmstadt, 2nd edition 2017

LfL, 2017: „Agrarmärkte 2017“, Bayrische Landesanstalt für Landwirtschaft (LfL), Munich; December 2017

BÖLW, 2016: „Zahlen, Daten, Fakten zur Bio-Branche 2016”, Bund für Ökologische Lebensmittelwirtschaft (BÖLW), Berlin, 2016

Thünen, 2017: "OekoBuchführung, table 131- Betriebe des ökologischen Landbaus nach Betriebsformen im Vergleich zu konventionell wirtschaftenden Betrieben", Excel-file with results from Testbetriebsnetz , Thünen/BMEL; downloaded on 26.4.2018: https://www.bmel-statistik.de/landwirtschaft/testbetriebsnetz/testbetriebsnetz-landwirtschaft-buchfuehrungsergebnisse/

Hülsbergen K-J, Rahmann G (eds.) (2013): „Klimawirkungen und Nachhaltigkeit ökologischer und konventioneller Betriebssysteme – Untersuchungen in einem Netzwerk von Pilotbetrieben“; Thünen Rep 8 (Johann Heinrich von Thünen-Institut), 412p, Braunschweig

Warnecke, Schulz, Paulsen et al., 2014 (Original file in: Hülsbergen K-J, Rahmann G (eds.) (2013) Klimawirkungen und Nachhaltigkeit ökologischer und konventioneller Betriebssysteme – Untersuchungen in einem Netzwerk von Pilotbetrieben. Braunschweig: Johann Heinrich von Thünen-Institut, 412p, Thünen Rep 8

Annex 1. Product by product footprints and their interpretation

Annex 1.1. Carbon and land footprint of each FQS product and its reference

See carbon_and_land_footprint_of_FQS.xlsx.

Annex 1.2. Product by product interpretation of the carbon footprint

This annex provides a case by case interpretation of the carbon footprint of each FQS product and its difference with its reference product. As part of the quality procedure, this interpretation was agreed upon between the person in charge of data collection for the product and the person in charge of the estimation of the carbon footprint. More details on each product can be found in Arfini and Bellassen (2019).

Product	Country	Interpretation
Dalmatian prosciutto	Croatia	The carbon footprint ($tCO_2e\ t^{-1}$) of PGI ham is 7% higher than its reference, despite a slightly lower footprint of the fresh meat used for PGI ham. This is largely due to the technical specifications which require a more intense drying for the PGI. Therefore, an accounting unit like $tCO_2e\ kcal^{-1}$ may yield results similar to those of fresh meat. The lower footprint of fresh meat is mostly due to manure management: Hungarian pig farms – from which most of the PGI fresh meat come from – use more solid manure systems than their Croatian counterparts. These small differences are to be taken cautiously due to a larger use of default values in PGI estimates than in the reference. Our estimates for fresh meat – 2.04 and 2.24 tCO_2e t of liveweight^{-1} for PGI and reference respectively – are at the lower end of the literature which ranges from 2.1 to 11.9 tCO_2e ton^{-1} pork meat (Clune et al., 2017; Meier et al., 2015).
PDO olive	Croatia	The carbon footprint (kg $CO_2e\ t^{-1}$) of PDO olive oil is 45% lower than its reference.

oil		The PDO has a much lower carbon footprint than the reference, mostly thanks to the higher olive yield and lesser use of energy for soil and plant preparation for production. The order of magnitude is comparable[3] to the 3.52 kg CO_2e t^{-1} reported by Rinaldi et al. (2014) for the cultivation stage in Italy. The overall footprint Croatian olive oil is much lower though, due to the absence of freezing in the Croatian process.
Comte cheese	France	The carbon footprint of Comté is 15% lower than its reference, mostly due to a higher processing efficiency (10 liters per kg of Comté instead of 12 per kg of Emmental). Indeed, at farm level, the carbon footprint of milk is almost the same (1.131 and 1.126 tCO_2e t of $milk^{-1}$ respectively): while the higher share of pasture saves some emissions from fertilizer and machinery, these savings are offset by a 4% lower milk productivity of cows and by a higher share of rapeseed in the ration. The carbon footprints at farm level are within the 0.52-2 tCO_2e t of $milk^{-1}$ literature range (Meier et al., 2015).
Organic flour	France	The carbon footprint (without transport) of organic bread is 34% lower than the reference (162 vs 246 $kgCO_2$e ton of $bread^{-1}$). The difference in per hectare emissions is even higher, mainly due to the absence of mineral fertilizers, but the much higher yield of conventional wheat (4 vs 7.6 tons ha^{-1}) partly offsets this benefit. This is consistent with Meisterling et al. (2009) which also finds a better carbon footprint for organic flour. Note that the carbon footprint we find for conventional bread is almost equal to the value reported by Meisterling et al. (2009) and slightly lower than Espinoza-Orias et al. (2011).
Saint-Michel bay bouchot mussels	France	The carbon footprint indicator calculated takes into account the production stage and is basically based on the energy consumption at farm level in the FQS and the reference. There is practically no difference between the PDO and the reference (184 and 195 kgCO2e ton-1 of fresh mussel respectively), which is not surprising as the energy inputs are similar and because nothing in the technical specifications seems likely to have an impact on the carbon footprint. One could have expected higher fuel use in the PDO due to the higher use of amphibious boats in the Mont Saint Michel bay (large foreshore, long distances to the bouchots), but it does not materializes in the accounts of mussels farms. The results are towards the lower end of the literature: SARF (2012) reports 252 kgCO2e/ton of fresh suspended mussels and Winther et al (2009) reports 165 kgCO2e/ton of fresh mussels (shell included). This makes sense as we do not account for the CFP of materials (ropes, etc.) and because the French electricity mix is much less carbon intensive than average. Aubin et al. (2018) reports 9,5 kgCO2e/ton of fresh mussels when including C sequestration not only in shell but also in wooden bouchots. The high values from Irribaren (2010) were disregarded because the extremely high energy consumption involved is deemed unrealistic.
Organic rice	France	The carbon footprint of organic rice is 14% lower than its reference (0.99 and 1.16 tCO_2e/ton of processed rice respectively). The bulk of the difference is explained by the lower use of fertilizer in organic rice, and in particular the absence of mineral fertilizers banned in the technical specifications. Both products are in the lower part of the literature range – 0.66 to 5.69 tCO_2e/ton (Clune et al., 2017; Odegard et al., 2015) – which is explained by flooding which is only intermittent Camargue and by the refined estimate of N_2O emissions we use, which accounts for crop type and is therefore much lower than most existing LCAs. Hokazano et al. (2012) find a 33% higher carbon footprint for organic rice in Japan, explained by much higher methane emissions from the flooding techniques which weighs more on the lower yield organic rice.
Organic pork	Germany	The carbon footprint (excluding transport) of organic pork is 8% higher than its conventional reference (3.7 vs 4 tCO_2e ton^{-1} pork meat). These values are in the

[3] Assuming a density of 0.92 kg/L.

		lower range of the literature which ranges from 2.1 to 11.9 $tCO_2e\ ton^{-1}$ pork meat (Clune et al., 2017; Meier et al., 2015). The small net difference between organic and conventional pork results from two balancing differences. On the one hand, the carbon footprint of organic feed is twice lower per ton of dry matter, thanks for the absence of mineral fertilizer and the use of waste fishmeal. On the other hand, total intake is 40% higher, and emissions from enteric fermentation and manure management are also substantially higher, because fattening pigs live longer and are more active, and because of the lower pigs/sows ratio. A similar tradeoff is also reported by Kool et al. (2009) and by Basset-Mens and van der Werf (2005): both studies report a lower carbon footprint per ton of feed for the organic chain although the carbon footprint of feed as a category is almost the same between organic and conventional as organic pigs require more feed per ton of final product. The difference in performance between organic and conventional is within the literature range of -11% to 73% (Kool et al., 2009; Meier et al., 2015). It is lower than the 35% found by Kool et al. (2009) for Germany, despite many similarities in input data for three main reasons: 31% of the diet of organic pigs comes from straw and fishmeal which are assumed to be waste and have no carbon footprint. If instead we assume that fishmeal is fished for the sole purpose of feeding pigs, then the carbon footprint of organic pork becomes 20% higher than its reference. The second reason is that Kool et al. (2009) uses the IPCC Tier 1 approach to estimate N_2O emissions from fertilizer use which results on average in 30% higher estimates (Carlson et al., 2016). Finally, Kool et al. (2009) uses lower pigs/sows ratio of 6.6 (organic) to 7.3 (conventional) which increases the weight of sows emissions per ton of meat and consequently increases the feed/meat ratio.
Organic yoghurt	Germany	The carbon footprint of organic yoghurt is 18% higher than its conventional reference, with 1.48 and 1.25 tCO2e ton-1 yoghurt respectively. Most of this difference is explained by the 18% lower feed-to-milk conversion efficiency of organic cows. The second most important contributing factor is the higher allocation to meat in the conventional system: if 100% of the footprint is allocated to yoghurt, the organic system is only 13% higher. This mostly stems from the price premium of organic milk being much higher than the price premium of organic cull cows. The carbon footprint of feed is slightly lower for organic feed, despite the higher amount required per ton of yoghurt. But the carbon footprint of feed is only 7-9% of the total carbon footprint of yoghurt and it is offset by a higher share of more digestible grains in the diet of conventional cows. The carbon footprints at farm level are within the 0.52-2 tCO2e t of milk-1 literature range (Meier et al., 2015). Lindenthal et al. (2010) find however a lower carbon footprint for organic yoghurt in Austria. Their results are largely driven by the accounting of land-use related emissions and sequestration: 400 kgCO2/ha sequestration for organic feed (excluding grassland) while conventional feed fields emit an average 202 kgCO2/ha.
Zagora apple	Greece	The carbon footprints (excluding transport) of the Zagora apple and its reference, 326 and 177 $kgCO_2e\ ton^{-1}$ respectively, are within the literature range of 70-890 $kgCO_2e\ ton^{-1}$ (ADEME, 2017; Clune et al., 2017). The key driver of the 84% higher footprint of the PDO is its 61% lower yield. The lower yield is mainly attributable to less intensive practices imposed by the technical specifications: absence of mechanization for harvest, use of a refined fertilization strategy based on measured leaf nitrogen content, etc. In terms of fuel use, the absence of mechanization for harvest is offset by the higher fuel requirements of long range hoses used instead of tractors for fertilizer and pesticide spraying.

Kastoria apple	Greece	The carbon footprints (excluding transport) of the Kastoria apple and its reference, 100 and 177 $kgCO_2e\ ton^{-1}$ respectively, are within the literature range of 70-890 $kgCO_2e\ ton^{-1}$ (ADEME, 2017; Clune et al., 2017). Two main factors explain the 44% lower footprint of the PGI: lower use of fertilizers and higher yield. The higher yield is mainly attributable to better pedo-climatic conditions but the lower use of fertilizers is more related to the PGI: while the technical specifications do not mention fertilizers, FQS farmers all use a refined fertilization strategy based on measured leaf nitrogen content. This strategy has been so widely adopted because it is paid for by the local cooperative as part of the quality management of the PGI product.
Gyulai sausage	Hungary	The carbon footprint ($tCO_2e\ t^{-1}$) of PGI sausage is 11% higher than its reference, despite a similar footprint of the fresh meat used for PGI ham. This is largely due to the technical specifications which require a more intense drying for the PGI. Therefore, an accounting unit like $tCO_2e\ kcal^{-1}$ may yield results similar to those of fresh meat. Our estimate for fresh meat[4] – 2.7 tCO_2e t of fresh $meat^{-1}$ for both PGI and reference – is at the lower end of the literature which ranges from 2 to 11.9 $tCO_2e\ ton^{-1}$ pork meat (Clune et al., 2017; Lesschen et al., 2011; Meier et al., 2015).
Kalocsai paprika powder	Hungary	The carbon footprint of the raw PDO pepper and its reference – 94 and 223 $kgCO_2e\ ton^{-1}$ respectively – are comparable, although somewhat lower than the only literature reference of 368 $kgCO_2e\ ton^{-1}$ (Wang et al., 2018). The 43% difference – 1 and 1.7 $tCO_2e\ ton^{-1}$ respectively – found for the paprika itself (excluding transport) is explained by two main drivers: a twice larger use of mineral fertilizers in China – where the reference pepper is assumed to be produced – than in Hungary, and a twice higher yield in Hungary. Fuel use for cropping, one hundred times more important in Hungary, does not offset the first two drivers of carbon footprint.
Parmigiano Reggiano cheese	Italy	The carbon footprint of Parmigiano is 79% higher than its reference, mostly due to its higher density (16.7 liters of milk per kg of Parmigiano instead of 7.7 liters per kg of Biraghi cheese). To the contrary, at farm level, the carbon footprint of milk is 18% lower for Parmigiano (1.6 and 1.95 tCO_2e t of $milk^{-1}$ respectively). The two main drivers of this difference are the longer lifetime of Parmigiano cows, which lessens the “carbon deadweight” of unproductive heifers and cull cows, and the diet composition. Parmigiano cows eat substantially more alfalfa and mowed grass which are less fertilized and require less fuel for field operations than silage maize. Parmigiano breeders also obtain slightly higher yields for some crops such as alfalfa. The difference in diet composition is largely due to the technical specifications which limit many components (maize, soy, cereals, …) but not alfalfa and grass. The carbon footprints at farm level are within the 0.52-2 tCO_2e t of $milk^{-1}$ literature range (Meier et al., 2015).
Organic tomato from Emilia Romagna	Italy	The carbon footprints of fresh organic tomatoes and their reference, 18 and 34 $kgCO_2e\ ton^{-1}$ respectively, are lower than the literature range of 150-6,000 $kgCO_2e\ ton^{-1}$ (Clune et al., 2017). This large literature range is focused tomatoes grown in heated greenhouses where most of the carbon footprint comes from greenhouse construction and heating (Almeida et al., 2014; Röös and Karlsson, 2013). Open field Italian tomatoes are thus logically below the range. The bulk of the 48% difference between organic and conventional tomatoes is explained by the absence of synthetic nitrogen fertilizers for organic tomatoes. This gain is only marginally offset by the 13% lower yield of organic tomatoes. The integration of processing diminishes the difference, the carbon footprint of processed organic tomatoes being 18% lower than their reference, with 147 and 180 $kgCO_2e\ ton^{-1}$ respectively.
Opperdoezer Ronde potato	Netherlands	The carbon footprint of the PDO is 35% higher than the reference – 70 and 52 $kgCO_2e\ ton^{-1}$ respectively. Indeed, the higher yield of the reference more than compensates for its higher use of mineral fertilizers. The lower yield of the PDO largely stems from the technical specifications: as an “early potato”, the

[4] 1.56 $tCO_2e\ t^{-1}$ liveweight, 0.57 conversion factor from liveweight to fresh meat from organic pork case.

		Opperdoezer has a shorter growth period than common consumption potatoes. The lower fertilizer use is likely an indirect consequence of this shorter growth period: the Opperdoezer would not have time to profit from higher amounts of fertilizers. For the same reason, the diesel use per hectare for cultivation and the electricity use for storage are also lower. However, both are on the lower end of the literature which ranges from 80-360 kgCO_2e ton^{-1} (Clune et al., 2017; Meier et al., 2015). Indeed, potato cooling which usually weights around 50% of the energy demand is 100 times less carbon intensive in the Netherlands than in the UK (Hillier et al., 2011).
Lofoten stockfish	Norway	The carbon footprint of the PGI is 48% lower than its reference – 0.68 and 1.31 tCO_2e ton edible (rehydrated) fish^{-1} respectively. Indeed, PGI fishermen use 33% less fuel to capture the fish because the technical specifications request that they fish "around Lofoten and Vesteralen". Moreover, thanks to the lower fishing distance, they do not refrigerate the fish, neither on board nor when landed, whereas for the reference product, half the fish is cooled on boats. This results in additional 0.31 tCO_2e ton edible (rehydrated) fish^{-1} from the production of refrigerant liquid for the reference. Sun drying of the PGI does not improve substantially its carbon footprint as drying is only a minor component of the footprint and because the Norwegian electricity mix is dominated by hydro power. Both values are close to the carbon footprint obtained by Winther et al. (2009) for Norwegian clipfish (2.06 tCO_2e ton edible (rehydrated) fish^{-1} without transport but with all fish refrigerated).
Organic salmon	Norway	Similarly to previous studies (Pelletier and Tyedmers, 2007; Winther et al., 2009), feed production concentrates the lion's share of farmed salmon's carbon footprint. The carbon footprint (excluding transport) of organic salmon is 14% smaller than its conventional reference, with 0.89 vs 1.03 tCO_2e ton gutted fish^{-1}. This is driven by the absence of mineral nitrogen fertilizers for feed production (12-57% lower footprint of organic feed), although the lower feed yields and, more importantly, the higher use of fishmeal largely offset this benefit. These results are at the lower end of the 1.5 – 6.6 tCO_2e ton live fish^{-1} range in the literature (RIAS Inc., 2016), due to the use of Bouwman's equation for the estimation of N_2O emissions (Carlson et al., 2016) instead of the more simple IPCC Tier 1 method. These results rely heavily on the assumption that fishmeal is composed of fish captured for the sole purpose of feeding salmons, rather than composed of trimmings from fish processing. In the latter case, the carbon footprint of organic salmon would be half that of its reference, although both footprints would be much lower than the current estimates.
Organic pasta	Poland	Excluding transport, the carbon footprint of organic pasta is 7% lower than its reference (0.80 and 0.87 tCO2e ton-1 of pasta respectively). Most of this difference is driven by the absence of mineral fertilizers and pesticides in the cultivation of organic wheat. However, the lower yield of organic wheat partly offset these benefits. Processing represents 47% of the emissions of organic pasta. The use of energy per ton of output is higher in the case of organic pasta, due to a smaller scale of production and the use of traditional technologies. However the reference pasta generates higher emissions because of the greater share of electricity in the total energy input, which is coal-based in Poland. Both products are within the range found in the literature regarding value of carbon footprint in pasta production: 0.9 (Fritsche and Eberle, 2009), 1.3 (Ruini et al., 2013) or 0.5 (Röös et al., 2011) tCO2e ton-1 of pasta. The farm-level footprint is similar to (Röös et al., 2011) and at the lower end of the range from (Ruini et al., 2013), which can be explained by the relatively low amount of mineral fertilizer use.
Kaszubska strawberries	Poland	The carbon footprint of Kaszubska strawberries is 14% higher than the reference (122 vs 107 tCO_2e ton of strawberry^{-1}). The difference in per hectare emissions is in favour of the FQS, mainly due to the lower amount of fuel use for crop operations, but the higher yield of reference strawberries (8.9 vs 11 tons ha^{-1}) offsets this benefit. The lower amount of fuel use can be explained by the higher use of manpower for field operations (eg. manual planting, less mechanical weeding, manual harvest, …) and by improved logistics thanks to the fields being close to the

		farms. Our estimates are at the lower end of the 0.1-1.2 tCO_2e ton^{-1} range reported by Warner et al. (2010). Indeed, Warner et al. (2010) finds that pesticides, plastic use for greenhouses and bags and peat use substantially weight on the carbon footprint of UK strawberries whereas they are neglected in our estimate: no fumigation is necessary so pesticide use is much lower, and peat, greenhouses and crop bags are not used.
Sjenica cheese	Serbia	The carbon footprint of the sheep cheese FQS 85% higher than the reference cow cheese (21.7 vs 11.7 tCO_2e ton^{-1} of cheese). The reference value is within the range of existing studies, both for cow milk and cow cheese. So is the FQS: with 5.3 tCO_2e ton^{-1} of milk whereas existing studies range from 2-5 tCO_2e ton^{-1} (Batalla et al., 2015; Leip et al., 2010; Opio et al., 2013; Vagnoni et al., 2015). The large difference comes from the higher efficiency of cow herds in transforming fodder into milk: while the carbon footprint of each ton of fodder is similar between FQS and reference, ewes need three times more fodder to produce the same amount of milk than cows. While the diet of FQS ewes contains a higher share of grass and a lower share of maize than the reference, the associated carbon benefits are offset by the yield of the dominant forage in both diets – grass – which is twice higher for the reference without much more fertilizer use. This is due to the plateau land of the Sjenica region which is much less productive than the national average reference, as well as the combination of alfalfa with grass in the reference.
Organic raspberries	Serbia	The carbon footprint of organic raspberries is 5% lower than the reference (316 vs 333 $kgCO_2e$ ton of $raspberry^{-1}$). The difference in per hectare emissions is much higher, mainly due to the absence of mineral fertilizers, but the much higher yield of conventional raspberries (2.7 vs 5.7 tons ha^{-1}) largely offsets this benefit. Relatively large processing emissions due to freezing, which are the same for organic and conventional products, also reduce the advantage of organic raspberries in relative terms. The comparison with the literature is challenging as the carbon footprint of raspberries has never been investigated to our knowledge. Our estimates are within the 0.2-0.8 tCO_2e ton^{-1} literature range for red fruits. They are consistent with Venkat (2012) which finds a 31% and 13% lower carbon footprint for organic strawberries and blueberries respectively.
Sobrasada of Mallorca	Spain	The carbon footprint (excluding transport) of PDO sausage is 44% higher than its reference. This is mostly due to the characteristics of the Porc negre breed whose sows lay less than half the number of piglets that reference sows do and whose fattening pigs live around three times longer than reference pigs before being slaughtered. Despite the lower carbon intensity of one ton of fodder in the PDO, PDO pigs end up needing around three times as much of it per ton of sausage. Similarly, as pigs spend most of their time outside, the manure management system is emits less per ton of manure in the PDO, but longer lifetime and larger intake generate much more manure per ton of sausage in the PDO. Our estimate for fresh meat[5] – 4.4 and 3.1 tCO_2e t of fresh $meat^{-1}$ for the PDO and the reference respectively – is within the literature range of 2 to 11.9 tCO_2e ton^{-1} pork meat (Clune et al., 2017; Lesschen et al., 2011; Meier et al., 2015).
Ternasco de Aragon	Spain	The carbon footprint of Ternasco lamb is 59.3 tCO_2e ton^{-1} of meat, that is 12% higher than its non-PGI reference from the same region. The difference in carbon footprint is mostly due to the lower weight at slaughter of reference lambs in order to meet the technical specifications. Because lambs eat much less and live much shorter than ewes, the carbon footprint of system is dominated by the "deadweight" of juvenile and reproductive ewes. As a result, a 12.5% lower amount of meat produced per ewe FQS directly translates into a higher carbon footprint per ton of meat. Both values are within the 38.9-56.7 tCO_2e ton^{-1} of meat range reported by Ripoll-Bosh et al. (2011) for Spanish lamb.
Thung Kula Rong-Hai	Thailand	The carbon footprint (excluding transport) of GI rice is 51% lower than its reference (180 and 366 $kgCO_2e$/ton of processed rice respectively, excluding

[5] 1.87/0.56. 0.56 is the conversion factor from liveweight to fresh meat ratio in organic pork case.

(TKR) Hom Mali rice		transport-related emissions). The bulk of the difference is explained by the absence of mineral fertilization in GI rice. This is related to the fact that most GI producers also have an organic or "good agricultural practices" certifications. The higher yield obtained by GI producers – possibly due to better soils and higher farming skill – reinforces the benefit from the absence of mineral fertilization. Both products are much lower than the literature range – 0.66 to 5.69 tCO_2e/ton (Clune et al., 2017; Odegard et al., 2015) because both are rainfed whereas the literature focuses on the more common flooded rice which generates substantial methane emissions.
Doi Chaang coffee	Thailand	The carbon footprint (excluding transport) of the PGI coffee is 26% higher than its reference (7.6 vs 6.1 tCO_2e ton^{-1} of ground coffee). The bulk of this difference is due to higher yields for the reference coffee, although the higher use of fertilizers for the PGI coffee also plays a role. Because of lower yields and higher fertilizer use, these values are at the higher end of the literature range (perimeter restricted to the farming and processing stages) despite the efficient aerobic wastewater treatment: 7-8 tCO_2e ton^{-1} of coffee parchment[6] in Kenya where yields are almost twice higher (Maina et al., 2016), 1.68 tCO_2e ton^{-1} of green coffee in Costa Rica (Killian et al., 2013) where yields may reach 9 ton of coffee cherries per hectare (Noponen et al., 2012).
Phu Quoc Fish Sauce	Vietnam	The carbon footprint (excluding transport) is largely driven by fuel use in fishing boats. It is 2-4 times lower than Norwegian captured fish (Winther et al., 2009), but Norwegian fishermen may go further away to catch their fish. The carbon footprint of the PDO is 1% higher, because the lower amount of fuel used to catch fish, related to the restricted and nearby fishing area, is offset by the lower processing efficiency: only 0.26 ton of sauce and some co-products per ton of raw anchovy versus 1.09 for the reference product.
Buon Ma Thuot coffee	Vietnam	The carbon footprint of the PGI coffee is 15% lower than its reference (2.2 vs 2.7 tCO_2e ton^{-1} of ground coffee). Most of the difference comes from the lower use of mineral fertilizers in the PGI which is largely due to farmers belonging to GI-associated cooperatives. These cooperative provide advice on optimizing fertilization and substituting mineral fertilizers with organic ones. This effect is reinforced by lower electricity use to roast the coffee in the PGI, explained by the larger and more modern facilities than the reference. Both values are comparable to the 2.43 tCO_2e ton^{-1} of packaged roasted coffee[7] reported by Killian et al. (2013), using the same 0.75 kg roasted coffee per kg green coffee ratio as in Buon Ma Thuot coffee.

Annex 2. Subgroup statistics and sensitivity analysis

Annex 2.1. Subgroup statistics

Table S 1. Subgroup statistics for the per hectare carbon footprint

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-26%	0.00	
Organic	8	-46%	0.01	PDO, PGI

[6] 0.2 t of coffee parchment per t of coffee cherry.

[7] 1.82 tCO2e / t of packaged roasted coffee (packaging = 0.13), final_prod_ratio of approximately 0.75 for BMT coffee from green to roasted coffee beans.

PDO	8	-19%	0.14	Organic
PGI	10	-4%	0.08	Organic
Animal	9	-25%	0.00	None
Vegetal	14	-28%	0.01	None
Animal_Organic	3	-41%	0.25	None
Animal_PDO	2	-25%	0.50	Vegetal_Organic
Animal_PGI	4	-4%	0.13	None
Vegetal_Organic	5	-54%	0.06	Animal_PDO, Vegetal_PDO, Vegetal_PGI
Vegetal_PDO	4	-22%	0.38	Vegetal_Organic
Vegetal_PGI	5	-14%	0.44	Vegetal_Organic

Table S 2. Subgroup statistics for the per ton carbon footprint of raw products

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-13%	0.13	
Organic	8	-17%	0.08	None
PDO	8	-8%	0.55	None
PGI	10	4%	0.77	None
Animal	9	0%	0.65	Unfed Seafood/Fish
Unfed Seafood/Fish	3	-32%	0.25	Animal
Vegetal	14	-19%	0.22	None
Animal_Organic	3	-17%	1.00	None
Animal_PDO	2	-5%	1.00	None
Animal_PGI	4	11%	0.25	Unfed Seafood/Fish_PDO, Vegetal_Organic
Unfed Seafood/Fish_PDO	2	-31%	0.50	Animal_PGI
Unfed Seafood/Fish_PGI	1	-32%	na	None
Vegetal_Organic	5	-22%	0.06	Animal_PGI
Vegetal_PDO	4	-15%	1.00	None
Vegetal_PGI	5	-15%	0.63	None

Annex 2.2. Sensitivity analysis

2.2.1. Tier1 N_2O

Detailed method

In the *Tier 1 N_2O* scenario, the non-linear relationship between nitrogen inputs and N_2O emissions used in the Cool Farm tool is replaced by an IPCC Tier 1[8] estimate using Equation 1 from IPCC (2006).

Equation 1.

$$N2O = [F_{SN} \times (EF_1 \times (1 + Frac_{GASF}) + EF_5 \times Frac_{LEACH}) + F_{ON} \times (EF_1 \times (1 + Frac_{GASM}) + EF_5 \times Frac_{LEACH}) + F_{PRP} \times (EF_{PRP} + EF_1 \times Frac_{GASM} + EF_5 \times Frac_{LEACH}) + F_{CR} \times (EF_1 + EF_5 \times Frac_{LEACH})] \times \frac{44}{28}$$

Equation 2.

$$F_{CR} = (Y \times s + i) \times Frac_{renew} \times \left((1 - Frac_{remove}) \times N_{AG} + R_{BG-BIO} \times N_{BG}\right)$$

Where N_2O is the amount of N_2O (t N_2O yr-1) emitted, F_{SN} and F_{ON} are the amount of synthetic and organic nitrogen inputs (t N yr-1), F_{PRP} is the nitrogen input from urine and dung deposited on pasture (t N yr-1), $Frac_{GASF}$ and $Frac_{GASM}$ are the fraction of volatilised nitrogen for synthetic and organic inputs respectively, $Frac_{LEACH}$ is the fraction of nitrogen inputs leached as nitrate, F_{CR} is the amount of nitrogen in crop residues (t N yr-1), EF_1, EF_5 and EF_{PRP} are emission factors given in Table S 3, Y is crop yield (kgDM ha-1), s and i are slope and intercept parameters given in table 11.2 in IPCC (2006), $Frac_{renew}$ is the frequency of re-seeding, $Frac_{remove}$ is the fraction of above-ground residues removed from the field, N_{AG} is the nitrogen content of above-ground crop residues (kg N kgDM-1), R_{BG-BIO} is the ratio of below-ground residues over above-ground biomass and N_{BG} is the nitrogen content of below-ground crop residues (kg N kgDM-1).

Table S 3. Parameters for *Tier1 N_2O* sensitivity analysis

Parameter	Description	Value	Unit
EF_1	Emission factor for nitrogen inputs to soil	0.003 for flooded rice 0.01 otherwise (IPCC, 2006)	kg N_2O-N (kg N)$^{-1}$
EF_{PRP}	Emission factor for urine and dung deposited on pasture	0.02 for cattle, poultry and pigs 0.01 for sheep and others (IPCC, 2006)	kg N_2O-N (kg N)$^{-1}$
EF_5	Emission factor for leached nitrogen	0.0075 (IPCC, 2006)	kg N_2O-N (kg N)$^{-1}$
$Frac_{GASF}$	fraction of volatilised nitrogen for synthetic inputs	0.1 (IPCC, 2006)	No unit
$Frac_{GASM}$	fraction of volatilised nitrogen for organic inputs	0.2 (IPCC, 2006)	No unit
$Frac_{LEACH}$	fraction of nitrogen inputs leached as	0.3 (IPCC, 2006)	No unit

[8] The IPCC distinguishes three types of methods – Tiers – for estimating greenhouse gas emissions. Tier 1 is the most generic type. Tier 2 often requires country-specific parameters and Tier 3 mostly consists in complex and customized models.

	nitrate		
$Frac_{renew}$	frequency of re-seeding	1 for annual crops 0.25 for alfalfa and pasture	No unit
$Frac_{remove}$	fraction of above-ground residues removed from the field	0.82 (Agreste, 2010)	No unit
N_{AG}	nitrogen content of above-ground crop residues	crop-specific value from table 11.2 in IPCC (2006)	kg N kgDM-1
R_{BG-BIO}	ratio of below-ground residues over above-ground biomass	crop-specific value from table 11.2 in IPCC (2006)	No unit
N_{BG}	nitrogen content of below-ground crop residues	crop-specific value from table 11.2 in IPCC (2006)	kg N kgDM-1

Results

Table S 4. Subgroup statistics for the per ton carbon footprint of processed products (Tier1 N_2O)

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-3%	0.96	None
Organic	8	-12%	0.95	None
PDO	8	-3%	1.00	None
PGI	10	11%	1.00	None
Animal	9	12%	0.07	Unfed Seafood/Fish
Unfed Seafood/Fish	3	-6%	0.50	Animal
Vegetal	14	-12%	0.39	None
Animal_Organic	3	12%	0.75	None
Animal_PDO	2	40%	1.00	None
Animal_PGI	4	21%	0.13	Unfed Seafood/Fish_PDO, Vegetal_Organic
Unfed Seafood/Fish_PDO	2	-3%	1.00	Animal_PGI
Unfed Seafood/Fish_PGI	1	-48%	1.00	None
Vegetal_Organic	5	-13%	0.63	Animal_PGI
Vegetal_PDO	4	-8%	0.63	None
Vegetal_PGI	5	-11%	0.63	None

Table S 5. Subgroup statistics for the per hectare carbon footprint (Tier1 N_2O)

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups

All	26	-21%	0.00	
Organic	8	-31%	0.01	PDO, PGI
PDO	8	-20%	0.06	Organic
PGI	10	-9%	0.06	Organic
Animal	9	-21%	0.00	None
Vegetal	14	-31%	0.00	None
Animal_Organic	3	-22%	0.25	None
Animal_PDO	2	-23%	0.50	None
Animal_PGI	4	-10%	0.13	Vegetal_Organic
Vegetal_Organic	5	-31%	0.06	Animal_PGI
Vegetal_PDO	4	-29%	0.25	None
Vegetal_PGI	5	-11%	0.31	None

Table S 6. Subgroup statistics for the per ton carbon footprint of raw products (Tier1 N_2O)

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-9%	0.35	
Organic	8	-16%	0.74	None
PDO	8	-9%	0.31	None
PGI	10	5%	0.85	None
Animal	9	9%	0.36	Unfed Seafood/Fish
Unfed Seafood/Fish	3	-32%	0.25	Animal
Vegetal	14	-16%	0.39	None
Animal_Organic	3	9%	0.75	None
Animal_PDO	2	-3%	1.00	None
Animal_PGI	4	11%	0.25	Unfed Seafood/Fish_PDO
Unfed Seafood/Fish_PDO	2	-31%	0.50	Animal_PGI
Unfed Seafood/Fish_PGI	1	-32%	na	None
Vegetal_Organic	5	-18%	0.63	None
Vegetal_PDO	4	-14%	0.63	None
Vegetal_PGI	5	-12%	0.63	None

2.2.2. Manure allocation to animals

Detailed method

For vegetal products, the spreading of organic fertilizers is therefore assumed to result in no emissions. For animal products, the amounts of organic fertilizers spread in fields is adjusted to fit the estimated amount of nitrogen excreted by the animals and not volatilized or leached during manure management operations (Equation 3 from IPCC (2006)).

Equation 3.

$$N_{spreading} = N_{intake} \times (1 - N_{retention}) \times (1 - Frac_{LossMS})$$

$$N_{spreading} = DMI \times \frac{CP}{6.25} \times (1 - N_{retention}) \times (1 - Frac_{LossMS})$$

where $N_{spreading}$ is the amount of nitrogen available to be spread in kgN animal^{-1} yr^{-1}, N_{intake} is the amount of nitrogen ingested in kgN animal^{-1} yr^{-1}, $N_{retention}$ is the amount of nitrogen retained by animals in kgN animal^{-1} yr^{-1}, $Frac_{LossMS}$ is the fraction of nitrogen lost (volatilized or leached) in manure management systems, DMI is the dry matter intake in kg animal^{-1} yr^{-1}, and CP is the crude protein fraction.

Table S 7. Parameters for *manure allocation* sensitivity analysis

Parameter	Description	Value	Unit
$N_{retention}$	Amount of nitrogen retained by animals	0.2 for dairy cattle 0.07 for other cattle 0.3 for pigs and poultry 0.1 otherwise (IPCC, 2006)	kgN animal^{-1} yr^{-1}
$Frac_{LossMS}$	Fraction of nitrogen lost (volatilized or leached) in manure management systems	0 when grazing 0.5 for non-grazing pigs 0.4 for non-grazing other animals (IPCC, 2006)	No unit
DMI	Dry matter intake	Raw data	kg animal^{-1} yr^{-1}
CP	crude protein fraction	0.16 (Hou et al., 2016)	No unit

For instance in the Comté cheese supply chain, the average amount of manure to be spread has been estimated at 72 kgN ha^{-1} (Lambotte et al., submitted), while our estimate using these IPCC default values is very close at 78 kgN ha^{-1}. However, the IPCC default values may overestimate nitrogen losses in management systems – and therefore underestimate manure to be spread – in animal products where grazing is less important: for example, the average nitrogen loss in French manure management systems (excluding grazing) is 33% (CITEPA, 2013), which is lower than the 40-50% default values use here (Table S 7).

Results

Table S 8. Subgroup statistics for the per ton carbon footprint of processed products (manure allocation)

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-11%	0.58	None
Organic	8	-19%	0.05	None
PDO	8	-3%	0.84	None

PGI	10	10%	1.00	None
Animal	9	12%	0.30	Vegetal
Unfed Seafood/Fish	3	-6%	0.50	None
Vegetal	14	-20%	0.24	Animal
Animal_Organic	3	-18%	0.75	None
Animal_PDO	2	41%	1.00	None
Animal_PGI	4	19%	0.13	Unfed Seafood/Fish_PDO, Vegetal_Organic
Unfed Seafood/Fish_PDO	2	-3%	1.00	Animal_PGI, Vegetal_Organic
Unfed Seafood/Fish_PGI	1	-48%	na	None
Vegetal_Organic	5	-24%	0.06	Animal_PGI, Unfed Seafood/Fish_PDO
Vegetal_PDO	4	-9%	1.00	None
Vegetal_PGI	5	-14%	0.63	None

Table S 9. Subgroup statistics for the per hectare carbon footprint (manure allocation)

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-25%	0.00	
Organic	8	-42%	0.01	PDO, PGI
PDO	8	-16%	0.14	Organic
PGI	10	-9%	0.08	Organic
Animal	9	-24%	0.00	None
Vegetal	14	-30%	0.01	None
Animal_Organic	3	-42%	0.25	None
Animal_PDO	2	-23%	0.50	Vegetal_Organic
Animal_PGI	4	-12%	0.13	Vegetal_Organic
Vegetal_Organic	5	-56%	0.06	Animal_PDO, Animal_PGI, Vegetal_PDO, Vegetal_PGI
Vegetal_PDO	4	-21%	0.38	Vegetal_Organic
Vegetal_PGI	5	-14%	0.44	Vegetal_Organic

Table S 10. Subgroup statistics for the per ton carbon footprint of raw products (manure allocation)

The *Pr_median_not_zero* column indicates the p-value of the two-sided Wilcoxon signed-rank test. The *Different_from_subgroups* column lists the subgroups of the same type from which the subgroup is significantly different (p-value of the paired Kruskal-Wallis test lower than 0.1).

Subgroup	Nb_of_cases	Median_difference	Pr_median_not_0	Different_from_subgroups
All	26	-16%	0.09	
Organic	8	-23%	0.04	None
PDO	8	-7%	0.55	None
PGI	10	2%	0.92	None
Animal	9	1%	0.82	Unfed Seafood/Fish
Unfed Seafood/Fish	3	-32%	0.25	Animal

Vegetal	14	-32%	0.15	None
Animal_Organic	3	-19%	0.75	None
Animal_PDO	2	-4%	1.00	Vegetal_Organic
Animal_PGI	4	10%	0.25	Unfed Seafood/Fish_PDO, Vegetal_Organic
Unfed Seafood/Fish_PDO	2	-31%	0.50	Animal_PGI
Unfed Seafood/Fish_PGI	1	-32%	na	None
Vegetal_Organic	5	-44%	0.06	Animal_PDO, Animal_PGI
Vegetal_PDO	4	-15%	1.00	None
Vegetal_PGI	5	-16%	0.63	None

Annex 3. Descriptive statistics (absolute values)

Table S 11. Descriptive statistics of the products (pooling FQS and their reference products)

Variable	n	mean	median	std. dev.	min	max
Mineral nitrogen (kgN ha-1)	46	88.81	60.44	87.28	0.00	314.44
Organic nitrogen (kgN ha-1)	46	36.91	22.17	53.74	0.00	228.63
Crop or fodder yield (t ha-1)	46	9.40	4.06	16.32	1.35	79.00
Amount of final product per hectare (t ha-1)	46	5.26	1.73	10.75	0.04	53.50
Amount of raw product per ton of final product (t t-1)	52	10.26	1.71	19.47	0.88	90.54
Share of co-products in total value	52	0.07	0.02	0.12	0.00	0.56
Livestock density (LU ha-1)	16	0.98	0.97	0.44	0.39	2.18
Renewal rate (% yr-1)	16	0.17	0.19	0.03	0.10	0.20
Dry matter intake of breeding adults (tDM head-1 yr-1)	16	5.34	2.48	5.95	0.80	19.40
Milk production (kg day-1)	6	6.52	6.44	0.32	6.12	7.03
Lifetime of fattening adults (yr)	12	0.53	0.38	0.30	0.21	1.04
Dry matter intake of fattening adults (tDM head-1 yr-1)	12	0.617	0.786	0.432	0.005	1.200

Annex 4. Data sources and reference products

Table S 12. Data sources and reference product

Case studied	Country	Reference product	Most important data sources
Dalmatian ham	Croatia	Local non-PGI firm	FQS: FADN, Jayet (2017), Mueller et al. (2012) Reference: Interviews, Mueller et al. (2012)

PDO olive oil	Croatia	National average	FQS: interviews Reference: Mesic et al. (2014)
Comte cheese	France	National average (cow cheese)	FQS: IDELE France-Comté (2016), Agreste (2011), ADEME (2017), interviews Reference : IDELE (2012), Agreste (2011), ADEME (2017)
Organic flour	France	National average	CA Rhône-Alpes (2012), CA Occitanie et CER Occitanie (2016), Agreste (2011), Passion Céréales (2017), Juin (2015), Espinoza-Orias et al. (2011), interviews
Saint-Michel bay bouchot mussels	France	National average (TSG Bouchot mussels)	Interviews and accountancy data from farms (averaged over 2011-2014) and one processor (2017)
Camargue rice	France	Non-organic rice (mostly PGI)	Delmotte (2011), Ari Tchougoune (2018), Barbier (2018), Monier (2018), interviews
Organic pork	Germany	National average	Kool et al. (2009), Gorn (2017), Destatis (2017), Ecoinvent, Knudsen et al. (2010), interviews
Organic yoghurt	Germany	National average	Kool et al. (2009), Knudsen et al. (2010), KTBL (2017), BOLW (2016), Thünen (2017), Hülsbergen & Rahmann (eds.) (2013), Warnecke et al. (2014), interviews
Zagora apple	Greece	Kissavos apples (non-GI apples from another region)	Interviews and accountancy data from farms and cooperatives
Kastoria apple	Greece	Kissavos apples (non-GI apples from another region)	Interviews and accountancy data from farms and cooperatives
Gyulai sausage	Hungary	Non-PGI Hungarian sausage	FADN, Jayet (2017), Mueller et al. (2012), World Bank (2017), Kool et al. (2009), interviews
Kalocsai paprika powder	Hungary	Imported Chinese pepper milled in Hungary	FQS: interviews Reference: Wang et al. (2018)
Parmigiano Reggiano cheese	Italy	Biraghi cheese (similar non-PDO cheese)	FQS and reference: Italian FADN (2014), Ribaudo (2011), ARAL (2017).
Organic tomato from Emilia Romagna	Italy	Conventional processed tomatoes in the same region (Emilia-Romagna)	FQS and reference: STUARD (2017), interviews, accountancy data from Consorzio Casalasco del Pomodoro
Opperdoezer Ronde potato	Netherlands	Regular potato in neighbouring IJsselmeerpolders region	FQS: Interviews, MsC thesis Ref: KWIN-AGV 2015 report
Lofoten stockfish	Norway	Clipfish (cod)	NDF (2018), Winther et al. (2009), interviews
Organic farmed salmon	Norway	Conventional salmon	Cargill Aqua Nutrition (2017), Ytrestoyl et al. (2015), Williams et al. (2006), Knudsen et al. (2010), Oraqua (2013), NDF (2018), Winther et al. (2009), FAO (2018), interviews
Organic pasta	Poland	Simulated conventional farms	Interviews and accountancy data from farms and processing plants

		with sample characteristics	
Kaszubska strawberries	Poland	National average	FQS: survey of 14 farmers and interviews of producers association REF: national statistics and interviews
Sjenica cheese	Serbia	National average (cow cheese)	FQS: Grubić (2012), Serbian FADN, Poljosfera (2017), agroklub, SORS (2015), interviews Ref: Interviews, Serbian FADN, West et al. (2014), Lesschen et al. (2011)
Organic raspberries	Serbia	National average	SORS (2015), RD&T (2012), Serbian FADN (2015)
Sobrasada of Mallorca	Spain	National average	Interviews and accountancy data, Jaume (2017), Monfreda et al. (2008), Mueller et al. (2012), West et al. (2014) , Lesschen et al. (2011), Jayet (2017), FADN
Ternasco de Aragon	Spain	Non-PGI lamb in the same region (Aragon)	Rodriguez et al. (2007), Monfreda et al. (2008), Mueller et al. (2012), West et al. (2014), Lesschen et al. (2011), interviews, Opio et al. (2013)
Thung Kula Rong-Hai (TKR) Hom Mali rice	Thailand	Non certified rice from the same region (90% of GI rice is organic as well)	Interviews of farmers, millers and other stakeholders, Toshiyuki et al. (2013), Srisompun et al. (2017)
Doi Chaang coffee	Thailand	Non-PGI coffee from the same province	Interviews of farmers, millers and other stakeholders, Giovanucci et al. (2004)
Phu Quoc Fish Sauce	Vietnam	Non-PDO fish sauce from same region	Interviews and firm accountancy data
Buon Ma Thuot coffee	Vietnam	Non-PGI coffee from Dak Lak province in Vietnam	Interviews, Giovanucci et al. (2004)