Estimating Biodiversity Impact from Agricultural Statistics: an application to Food Quality Schemes in France

Sarah HUET, Clélia SIRAMI, Abdoul DIALLO, Julie REGOLO, Ainhoa IHASUSTA, Ludovic ARNAUD and Valentin BELLASSEN

Abstract

Although the agri-food system contributes to the five main drivers of biodiversity loss, impact assessment of food products remains limited either to *in situ* measurements that do not allow for generalization, or to systematic models that are not validated by *in situ* data. Here we describe the BVIAS (Biodiversity Value Increment from Agricultural Statistics) model, which allows estimating the biodiversity impact of all major food products based on accountancy data and public statistics. BVIAS is calibrated based on the most relevant large-scale studies and meta-analysis. It is then used to find out whether major Food Quality Schemes (FQSs) - Organic farming, Label Rouge (LR) and Geographical Indications (GIs) - have a lower impact on biodiversity than their conventional counterparts. Consistently with *in situ* data, we find that organic farms, as well as those producing Comté (Protected Designation of Origin), have less biodiversity impact on a per hectare basis. This local benefit is however more than offset by lower yields, resulting in no difference or a higher impact per ton. Taking into account the main drivers of biodiversity losses related to agriculture, relying on quantitative data for a large sample of farms and calibrating our model based on relevant large-scale studies and meta-analysis, we therefore propose here an objective, robust and operational method to estimate the impact of food products on biodiversity for use in environmental labeling schemes or other purposes.

1. Introduction

Biodiversity erosion is one of the most important environmental crisis with climate change. While the impacts of current climate change trajectories are estimated at several tens of percentage points of GDP (Rose et al., 2022), pollinator loss alone amounts to 1-2% of GDP, and about €4 billion for a country like Germany (Lippert et al., 2021). IPBES identifies five drivers of biodiversity loss (IPBES, 2019): land-use change (*30*%), direct exploitation (*23*%), climate change (*14*%), pollution (*14*%) and invasive species (*11*%). Agriculture heavily contributes to three of these five drivers (land-use change, climate change and pollution), and is, with direct exploitation, the main economic sector responsible for the global erosion of biodiversity (Maxwell et al., 2016; Tilman et al., 2017).

Environmental labeling of agricultural products is one of the policies that aims to reduce the impact of agriculture on biodiversity. Although consumer information appears to trigger only small short-term changes in food choices (De Marchi et al., 2023; Dubois et al., 2021), environmental labeling opens the way to several indirect long-term effects. It encourages producers to change their practices, and processors to change product formulations to improve their ratings, and can be used as a support for other policies (e.g., minimum rating requirement for public order, tax based on rating, etc.).

At the European level, the Commission adopted a proposal for a Directive in March 2023 requiring all companies wishing to claim a better environmental performance to use the life cycle analysis framework, and for example the EU Product and Organization Environmental Footprint (PEF and OEF) method, to objectify the claim. France has taken a further step with the 2021 Climate and Resilience Act ("LOI n° 2021-1104 du 22 août 2021 portant lutte contre le dérèglement climatique et renforcement de la résilience face à ses effets (1)," 2021), which plans to make environmental labeling mandatory on all food products after an experiment scheduled to last for 5 years. The proposal of Ecoscore by ADEME, which aimed to inspire the future government tool, was criticized by several stakeholders for being limited to a life cycle analysis of products ("Améliorer l'évaluation environnementale des viandes," 2020). Their main argument was that life cycle analysis fails to account for the impact of food products on biodiversity. The Scientific Council for Experimentation highlighted that life cycle analysis does address three of the five main drivers of biodiversity loss, but recognized the value of complementing life cycle analysis on specific points (Soler et al., 2021). In view of this controversy, it is urgent to propose an objective, robust and operational method for calculating the impact of food products on biodiversity. This is the aim of this paper.

We believe that environmental labeling should be based on a biodiversity indicator that meets five key requirements (Table 1):

- *Rely on an explicit and operational definition of biodiversity*. The term biodiversity is very polysemous. For this reason, it is important to specify an operational definition adapted to the context (Santana, 2014). This definition must also keep an intuitive link with the main issues related to biodiversity erosion such as loss of species or ecosystems (and especially those that provide important services such as pollinators).

- *Address the main drivers of biodiversity loss related to agriculture*, namely land-use change, climate change and pollution.

- *Rely on data on biodiversity or practices that are measured and representative at the plot level* to estimate the impact of actual rather than potential practices on biodiversity.

- *Allow for the evaluation of any food product.* The environmental labeling aims to be mandatory and the model must therefore be applicable to any product based on currently available data, differentiating both product types (e.g., lentils versus chicken) and production modes of the same product (e.g., conventional versus organic wheat).

- *Rely on a validation of the estimated impact based on in situ biodiversity measurements.* There are always two ways to assess an impact: *in situ* measurement and modeling. In the second case, an essential criterion of robustness is the validation of the model, at least on predicted variables for which *in situ* measurements of biodiversity are available (e.g., biodiversity per unit area).

Three main types of methods for assessing the impact of food products on biodiversity can be distinguished: *in situ* observations of biodiversity by species counting in ecosystems, modelling the impact of agricultural practices on biodiversity, and modelling the impact of FQS specifications on biodiversity (Table 1). The adequacy of these three methods with the five key requirements of environmental labeling is summarized in the following three paragraphs and in Table 1.

In situ observations clearly meet the data and validation criteria. However, they are available on a limited number of sites, making it difficult to assess all food products. The most comprehensive meta-analyses in terms of taxa studied allow an average effect to be estimated by type of product and by production modes. Tuck et al. (2014) studied organic and conventional fields for five crop types, and estimated that the specific diversity is on average 30% higher in organic fields. Species number is a relatively explicit definition of biodiversity and has an intuitive link with the main issues related to biodiversity erosion as long as one remains within the same ecosystem. This link is however less than true when comparing different components of ecosystems because we are then comparing communities where each species does not support the same share of ecosystem resilience and function (Santana, 2014; Sarkar, 2002). Finally, *in situ* observations have the major disadvantage of not taking into account yield, and therefore the amount of land used, which is by far the first driver of biodiversity loss according to the IPBES

(2019). Indeed, biodiversity is often expressed per unit area in these studies, and therefore attributes the same impact to two production modes for the same products (e.g., wheat), even if one occupies twice as much land as the other. This paradox is difficult to manage in environmental labeling because two tons of wheat have the same impact as one ton of wheat, provided the amount of inputs per hectare is the same.

Modelling based on agricultural practices data is the method used in life cycle analyses. Using such model, Crenna et al. (2019) showed that the impact on biodiversity in the EU food system is mainly caused by animal products (70-75% of the total impact), and more specifically by pork (19-23%) and beef (21-25%). Read et al. (2022) showed that the risk of species extinction caused by American food consumption could be reduced by 30% by adopting the EAT-Lancet flexitarian diet (Willett et al., 2019), and up to 45% by reducing waste. However, models used in these studies (Chaudhary et al., 2015; Curran et al., 2014; De Baan et al., 2013) poorly take into account differences in practices for the same product, using only three levels of «intensity», without taking into account agroecological infrastructures, plot size, etc. This results in very low sensitivity to differences in practices per unit area (Wermeille et al., 2023). To remedy this, Lindner et al. (2019) proposed a model that takes into account 14 agricultural practices in addition to the type of ecosystem (grassland vs arable land vs forest). Using a simplified version adapted to variables available in the Agribalyse life cycle inventory database, Lindner et al. (2022) concluded that organic farming reduces the biodiversity impact of the kilogram of wheat (-*33*%) or the liter of milk (-*27*%), but increases the impact of the kilogram of chicken (+*33*%). In all cases, these model predictions are not calibrated based on *in situ* observations of biodiversity.

Modelling from specifications, so far implemented only in the grey literature, offers an assessment of the potential impact of restrictions placed in the specifications of FQSs. Alliot et al. (2021) concluded that the specifications associated with organic farming (AB, Demeter, and Nature et Progrès FQSs) strongly limit damages to biodiversity (score between 3/5 and 4/5), that the Protected Designation of Origin (PDO) Comté and the Bleu Blanc Cœur FQS limit them moderately (2/5) and that other FQSs (e.g., HVE, Zero Pesticides, Label Rouge) do not limit them. This approach has the advantage of being applicable to any FQS, but ignores the practices actually implemented. For example, the Comté PDO's specifications that limit mineral fertilization to 50 kg N ha-1 do not provide information on the quantities actually used, possible effects on greater crop diversity or on the ratio between organic and mineral nitrogen. Moreover, this approach suffers from the same limitation as *in situ* observations on the non-consideration of land consumption and the same limitation as other models on the absence of calibration.

Table 1: ***Environmental Labeling Requirements and Existing Methods.*** *The color code gives the authors' judgment on the degree of adequacy of each type of method for each criterion (requirement fulfilled, partially fulfilled or not fulfilled), based on the example cited at the top of the table.*

	In situ observations (e.g., Tuck *et al.*, 2014)	Modeling from specifications (e.g., Alliot *et al.*, 2021)	Modeling from agricultural practices data (e.g., Lindner and Koch, 2022)	BVIAS (this study)
Explicit and operational definition of biodiversity	Species diversity, but dominated by studies on small soil organisms.	Implicit definition	Degradation of natural state (or risk of extinction of species for other studies)	Calibrated on meta-analysis or large-scale studies based on species diversity of multiple taxa related to ecosystem services (e.g., pollinators) or ecosystem functioning (e.g., vertebrates).
Addressing land use	Consideration of landscape effects. No consideration of yields	Consideration of landscape effects. No consideration of yields	No consideration of landscape effects. Consideration of yields	Consideration of landscape effects and yields
Addressing climate change	No	No	Not in Lindner et al. (2022), but considered by other models	No
Addressing pollution	Consideration of all pollutants, with effect size	Consideration of the main pollutants, no effect size	Consideration of the main pollutants, no effect size	Consideration of the main pollutants, with effect size
Based on measured data, biodiversity or practices at the farm level	Yes (biodiversity), but lack of representativeness (small sample size, only some productions)	No (specification at FQS level)	Yes (practices), but lack of representativeness (life cycle inventories)	Yes (practices), measured on all farms or a representative sample (FADN, AC, LPIS)
Allow a default estimate on any food product	No (too few measurements to differentiate products)	Differentiates many FQSs, but not products	Differentiates main products, and organic FQS	Differentiates main products, and main FQSs
Calibration based on biodiversity measurements	Yes (intrinsically)	No	No	Yes (large-scale studies and meta-analysis)

Finally, for the three families of methods, appraising the representativeness of their results for a given product type or FQS is currently challenging. *in situ* measurements, as well as life cycle inventories, are generally carried out on a limited number of farms without any explicit indication of their representativeness. This limit may become even more critical as the pressure of environmental labeling may lead some stakeholders to provide life cycle inventories with flattering examples for their sector. Modeling from specifications is inherently representative of a FQS but only for mandatory or prohibited practices and not for recommendations and unmentioned practices.

The increasing availability of data collected for public statistics and agricultural policies allows comparing data on practices for a high number of farms, including farms under FQSs. It also allows using propensity score matching for comparing farms with similar structural characteristics, thereby getting closer to the causal impact of FQSs. Some studies have already linked data from the Agricultural Accountancy Data Network (FADN) and the National Institute of Origin and Quality (INAO) to assess the economic effectiveness of certified farms. Jeanneaux et al. (2018) showed that some FQSs do not generate better profitability, because the price premium is fully compensated by a lower technical efficiency (e.g., Label Rouge or organic broilers). Sengel et al. (2022) estimated that dairy farms under GI generate 30% higher income per unit of work than conventional farms. This difference, mainly due to Franche-Comté GIs, rises to 40% after restricting the comparison to farms with comparable structure and location, following a «matching» procedure.

Our study proposes an objective, robust and operational method to calculate the impact of food products on biodiversity. This method is applied to the main French agricultural productions, distinguishing the main traceable FQSs (organic farming, Label Rouge and GIs), in the context of environmental labeling. Based on the BVI model (Lindner et al., 2019), our method overcomes three main limitations raised when considering environmental labeling (Lindner et al., 2022). By relying on agricultural statistics (e.g., FADN, Agricultural Census), we access large sample sizes and operating characteristics that make it possible to objectify the choice of the counterfactuals to FQS farms, following a "matching" procedure. We account for landscape effects on biodiversity by mobilizing the French Land Parcel Information System (LPIS) and semi-natural elements database (BD Haies). Finally, by calibrating predicted differences in impacts based on the orders of magnitude established from *in situ* measurements of biodiversity in the literature, we improve the empirical relevance of estimated impacts.

2. Materials & Methods

2.1 The BVI model

The BVIAS model builds on the BVI model as originally selected by ADEME for environmental labeling (Lindner et al., 2022). This model considers biodiversity in the plot and its adjacent semi-natural elements (SNE), and takes into account land use (permanent grassland versus arable land versus natural ecosystem), landscape effects and intensity of farming practices. BVIAS accounts for eight practices with a major impact on biodiversity within each land use category: application of nitrogen fertilizers (quantity and quality), hedge density, use of plant protection agents, duration of ground cover, tillage, mean field size and crop diversity (the last four only apply to arable land).

The key concept which is retained from the original BVI model is that the optimal biodiversity - normalized to a value of 1 - is obtained in a natural ecosystem, and that other land-uses - here arable land and permanent grassland - are associated with a specific range of biodiversity values based on the literature (Gallego-Zamorano et al., 2022). The actual position of a given plot within that range depends on the intensity of the height practices.

To aggregate the impact of farming practices into a synthetic biodiversity value, the BVIAS model first considers their normalized intensity values in the interval [0.1], the minimum intensity corresponding to 0 and the maximum corresponding to 1. To exclude possible outliers, the *95*percentile of observed intensity is set as a threshold above which values are normalized to 1. The contribution to the biodiversity value of each practice is then calculated by applying a response function specific to each parameter Eq. 1.

$$BVC_{i,l,v} = \gamma_{l,v} + \epsilon_{l,v} \cdot \exp\left(- \frac{\left| \left(x_{i,l,v}^{\delta_{l,v}} - \beta_{l,v} \right)^{\alpha_{l,v}} \right|}{2\sigma_{l,v}^{\alpha_{l,v}}} \right) (1)$$

where BVC is the biodiversity value contribution (without dimension, 1 = maximum contribution of practice, 0 = minimum contribution) of a practice v for each observation i in a land use type l, x is the normalized practice intensity, and $[\alpha, \beta, \delta, \epsilon, \gamma, \sigma)$ are the contribution function constants.

Whereas the original BVI model uses a simple average of BVCs to calculate the land use specific biodiversity value (BV_{LU}), BVIAS relies on a weighted average of BVCs (Eq. 2, where z is the practice weight), based on their relative importance as a driver of biodiversity loss (see Section 2.8).

$$BV_{LU} = \sum z_{l,v} \cdot BVC_{i,l,v} \quad (2)$$

As mentioned above, BV_{LU} is then projected into the range of possible biodiversity values for this particular land use type (i.e., arable or grassland), in order to obtain a standardized biodiversity value (BV_{loc}; Eq. 3). Unlike Lindner et al. (2019)'s seminal method, we do not correct for possibly «more valuable biomes » (e.g., tropical moist climate > temperate moist climate). Finally, the impact on biodiversity value (BVI) of one hectare is defined as $BVI_{ha} = 1 - BV_{loc}$.

$$BV_{loc,i,l} = LU_{min,l} + BV_{LU,i,l} \times (LU_{max,l} - LU_{min,l}) \quad (3)$$

Contrarily to the original BVI model, we calibrated all contribution function constants, aggregation weights and land use biodiversity value ranges based on literature values (see Section 2.8).

2.2 Allocation of biodiversity impact to farm products

To assign the previously calculated impact per hectare to different farm products, we follow the common life cycle analysis (LCA) recommendations of restricting the scope when possible and allocating where not (*International Reference Life Cycle Data System (ILCD) Handbook general guide for life cycle assessment*, 2010).

For crops, we use the crop practices survey (Ministère De L'Agriculture (SSP), 2019) to «separate systems» to the extent possible, i.e., allocating farm-level inputs to each specific crop within a given farm (see Section 2.4). The impact on biodiversity of one kilogram of crop is then simply calculated as: $BVI_t = BVI_{ha} / Y$ where Y is the yield of the crop in ton/ha.

For animal products, we use scope restriction to assess the impact of different production workshops (e.g., crops, cattle and pigs), but use an economic allocation method to assess the impact of co-products from the same workshop (e.g., milk and cull cow meat from dairy cattle). The impact on biodiversity of one kilogram of animal product is therefore calculated in two steps. Firstly, we estimate the workshop impact by summing up the impact of livestock feed consumed by the workshop: the so-called pseudo-farm encompasses both the farm areas used to produce feed and the areas outside the farm necessary for the production of purchased feed. Secondly, the workshop impact is economically allocated to the different co-products (e.g., milk and meat), based on the sale value. This economic allocation of impact – i.e.,in proportion to the turnover generated by each product – is both the most common choice in LCAs and the best approximation of the share of responsibility in generating the impact (Koch and Salou, 2020).

2.3 Data used

2.3.1 Source databases

Reporting on specific impacts of FQSs on biodiversity requires first to identify farms with certified products and to estimate farming practices applied in these farms.

To identify farms with certified products, we use the Quality and Origin Information Signs (SIQO; i.e., FQSs) database from the National Institute of Designations of Origin (INAO) linked to the Agricultural Census (AC).

The estimation of farming practices requires data on the area of different crop production, the quantity and type of inputs used, the average number of different livestock categories, the amount and type of feed, and the quantity and value of production. We use the French Farm Accountancy Data Network (FADN), which collects these data annually on a representative sample of more than 7,500 large and medium-sized farms throughout metropolitan France, to estimate farming practices applied on each farm (Fig. 1).This large sample size allows for statistically robust comparisons between productions with and without FQS. In some cases, we use other data sources (e.g., Chenu and Butault (2015) for share of tillage in off-road diesel use) to predict farming practice intensity (e.g., L of diesel used for tillage) based on the raw FADN data (e.g., L of consumed diesel).

We use the French Land Parcel Information System (LPIS; IGN (2020)) and the Hedges layer of the BD TOPO® database (IGN and ASP, 2020) to calculate mean field size, hedge density (i.e., hedge linear meter per hectare) and crop diversity (Shannon Index) for each farm. We also use raster data from Sentinel-2 (European Space Agency, 2022) to estimate ground cover (i.e., number of days per year with vegetation, see Section 2.5 and Section A.1.1.4).

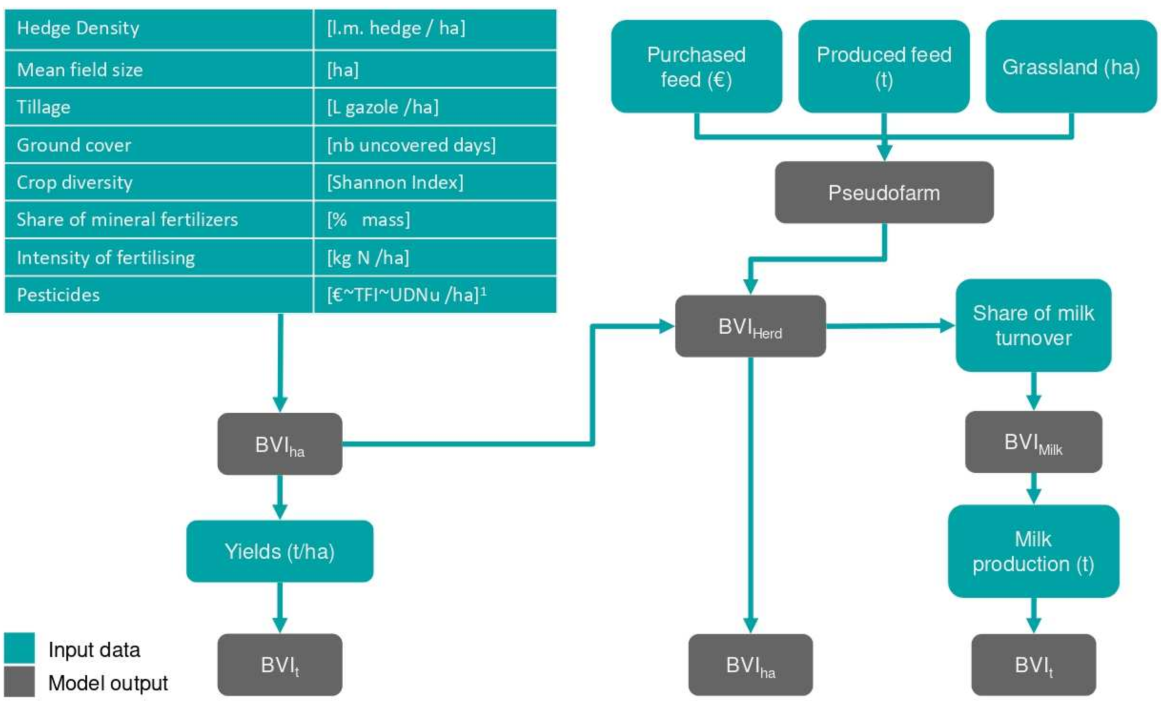

*Figure 1: **Simplified diagram of agricultural practice modeling** from FADN data and their integration with the BVIAS model.* ❑[1] $€ \sim TFI \sim UDNu$*: estimated average toxicity of one euro of pesticide based on Treatment Frequency Index (TFI) and Unit Dose Number (UDNu) among the ten most sold pesticides.*

2.3.2 Matching AC, SIQO and FADN databases

The matching of AC and FQS (SIQO from INAO) data was realized prior to this study by using the SIRET number (French System for Identification of Establishments) of agricultural holdings (Corre et al., 2023). This step allowed us to identify farms with production involved in FQSs. We then matched the 2020 FADN and AC data by comparing the SIRET, Pacage (identifier linked to the Common Agricultural Policy) and SIREN (French System for Identification of Business Register) numbers of agricultural holdings. The matched dataset thus includes 7,292 farms, located throughout the metropolitan territory, representing 99% of farms registered in the 2020 FADN. A description of the matched dataset is provided in Table 2.

Table 2: ***Descriptive statistics*** *(average) per technical-economic orientations (OTEX) and FQSs.*

Technical Orientation	Variable	Conventionnel	Organic Farming	Other FQSs
Field crops	Farm number	729	23	112
	Crop area (ha)	131.437	117.464	159.778
	Permanent grassland area (ha)	13.995	9.857	22.269
	Dairy herd size (number of cow)	8.63	-	32.872
	Wheat yield (kg/ha)	6,795.883	3,626.158	6,295.49
	Milk yield (L/cow)	4,587.618	-	6,005.217
Dairy cattle	Farm number	464	70	305
	Crop area (ha)	72.049	58.518	57.533
	Permanent grassland area (ha)	46.078	55.914	66.901
	Dairy herd size (number of cow)	75.1	64.337	72.357
	Wheat yield (kg/ha)	6,625.307	3,748.108	6,128.158
	Milk yield (L/cow)	7,047.4	4,812.512	6,731.377
Mixed cattle	Farm number	130	7	84
	Crop area (ha)	74.541	50.357	71.584
	Permanent grassland area (ha)	87.322	109.111	93.837
	Dairy herd size (number of cow)	57.275	53.79	64.391
	Wheat yield (kg/ha)	6,499.929	2,188.915	5,931.669
	Milk yield (L/cow)	6,428.421	4,464.584	6,409.946
Mixed crop and/or mixed livestock	Farm number	481	34	237
	Crop area (ha)	101.036	65.099	104.98
	Permanent grassland area (ha)	40.314	40.588	47.897
	Dairy herd size (number of cow)	64.512	42.75	62.607
	Wheat yield (kg/ha)	6,984.972	2,928.303	6,062.121
	Milk yield (L/cow)	7,613.625	3,524.284	7,111.986

2.3.3 Matching the FADN-AC-FQS database with the LPIS

To estimate landscape related variables for each farm, we first intersected our matched FADN-AC-FQS dataset with the 2020 French Land Parcel Information System (LPIS) (IGN, 2020), then add the hedge layer of the BD Topo®(IGN and ASP, 2020) and Sentinel-2 data (European Space Agency, 2022).

2.4 Estimation of farming practices in crops

The BVIAS model considers eight input variables, including the intensity of four within-field practices and four landscape variables (see Section 2.5): tillage, total nitrogen fertilization (mineral and organic), share of mineral nitrogen fertilization, use of pesticides, hedge density, mean field size, ground cover and crop diversity. The distribution of each variable is reported in Table A3, Table A4 and Fig. A1.

2.4.1 Tillage

Tillage intensity (expressed in L of diesel used for tillage/ha) is estimated from the farm's off-road diesel consumption, which is allocated to the different crops according to their area. Then, we subtract the average amount of off-road diesel per hectare used by farms which practice direct seeding, i.e., cultivate crops without tillage (Chenu and Butault, 2015). The implicit assumption is that the average off-road diesel use in these farms broadly corresponds to the amount of diesel used for other interventions than tillage. Note that tillage generally represents *43*% of off-road diesel consumption due to mechanization in cereal fields (Chenu and Butault, 2015), so even the total consumption is already highly correlated with tillage intensity.

2.4.2 Nitrogen fertilization

The intensity of nitrogen fertilization is estimated for both mineral and organic fertilizers. For mineral fertilizers, the amount of mineral nitrogen brought to the farm, directly reported in the FADN, is allocated to the different crops using national mineral nitrogen input averages as a distribution key (Ministère De L'Agriculture (SSP), 2019) (see "PKGC_N_ferti" sheet of Appendix C).

For organic fertilizers, as the FADN variable is not well-informed, we estimate these inputs in two different ways, depending on whether or not the farm has livestock.

For farms without livestock, a standard value, equal to the national average of organic nitrogen input per crop for farms that do not produce organic manure (Ministère De L'Agriculture (SSP), 2019) (see "PKGC_N_ferti" sheet of Appendix C)), is added to the different crops, distinguishing between organic and conventional farms.

For farms with livestock, we consider that the total amount of nitrogen excreted by the farm herd is spread on-farm. The amount of nitrogen excreted by livestock is calculated in accordance with IPCC recommendations (IPCC, 2019, 2006) and then allocated among the different crops, using the national averages of organic nitrogen input per crop of farms producing their own manure as a distribution key (Ministère De L'Agriculture (SSP), 2019) (see "PKGC_N_ferti_org" sheet in Appendix C). The calculation of nitrogen excretion by animals is based on the livestock population reported in FADN and the estimation of the feed quantity and nutritional quality of the ration provided to these animals (see Section 2.7.1).

If the resulting amount is smaller than both the national average of organic nitrogen inputs and the lower bound of the 95% confidence interval of the average of organic nitrogen inputs of farms producing their own organic manure (Ministère De L'Agriculture (SSP), 2019), we assign the lowest of these two values instead (implicitly concluding that despite having a few animals, these farms import manure).

At the opposite, if the resulting amount is higher than both the national average of organic nitrogen inputs and the higher bound of the *95*% confidence interval of the average of organic nitrogen inputs of farms producing their own organic manure (Ministère De L'Agriculture (SSP), 2019), we assign the highest of these two values instead (implicitly concluding that these farms have too many animals per hectare and therefore export manure).

In addition, the proportion of mineral nitrogen on total nitrogen input to crops is used as a fully-fledged parameter of the model (Lindner et al., 2019).

2.4.3 Application of pesticides

The intensity of the application of pesticides is estimated using the purchased value (€) in pesticides recorded in the FADN. This value is allocated to the different crops using the national average Treatment Frequency Index (TFI) as a distribution key (Ministère De L'Agriculture (SSP), 2019) (see "IFT_ref" sheet of Appendix C). We accounted for the lower toxicity of products used in organic farming by correcting the value in pesticides of organic farms by the ratio between the average toxicity of products used in organic farming and the average toxicity of products used in conventional agriculture.

To assess the average toxicities of products used either in organic or conventional farming, we estimate the toxicity by dose of four among the top ten best-selling products in each of these production modes (France and Ventes de produits phytopharmaceutiques par les Distributeurs agréés, 2020). We select these four products for the availability of data to determine their toxicity, by multiplying their "freshwater ecotoxicity" characterization factor (Andreasi Bassi et al., 2023) by their unit dose ("Arrêté du 27 avril 2017 définissant la méthodologie de calcul et la valeur des doses unités de référence des substances actives phytopharmaceutiques," 2017). In line with the ADEME proposal for environmental labeling ("Ingrédients agricoles - inventaires mobilisés (impacts ACV) ," 2024), the characterization factor is doubled for organic molecules compared to inorganic molecules (e.g., copper). Then, we calculate the average toxicity of products used in organic or conventional agriculture as the weighted average of the toxicity of the four products by the corresponding number of unit doses (NODU) used in France in 2020 (see "A.5.1_substance_top10" sheet of Appendix C). As a result, we estimate that each euro spent on plant protection in organic farms is 3.82 times less toxic than a plant protection euro in conventional farms.

2.5 Estimation of landscape variables

We estimate four landscape variables from the pairing between the French Land Parcel Information System (IGN, 2020) (LPIS), the Hedge layer of the BD Topo® (IGN and ASP, 2020), The Sentinel-2 raster data (European Space Agency, 2022) and our FADN-AC-FQS matched database: - Hedge density is the ratio of the sum in linear meters of hedge to the utilized agricultural area (UAA) of the holding (see Section A.1.1.1). - Mean field size is the ratio of the UAA to the number of plots, calculated based on the French Land Parcel Information System (LPIS; IGN (2020); see Section A.1.1.2). - Crop diversity is the Shannon diversity index of arable land use types of each farm based on the FADN data (see Section A.1.1.3). - Ground cover is the average number of days where soils is not covered by vegetation based on the Sentinel-2 (S2) raster data (European Space Agency, 2022) (see Section A.1.1.4)

2.6 Estimation of farming practices in grasslands

Permanent grasslands are not ploughed, do not receive pesticides, and are not associated with any crop diversity. The effect of mean field size on biodiversity is also considered negligible in grasslands. Permanent grasslands also have a year-round ground cover. These five variables are therefore weighted to zero for the calculation of BV_{LU} in grasslands. Note that temporary grasslands are considered as crops are they are often part of crop rotation.

2.7 Estimation of herding practices from accounting data

Estimating the quantity and nutritional quality of livestock feed is required to estimate the amount of nitrogen excreted by livestock according to Tier 2 IPCC recommendations (IPCC, 2019, 2006). This estimate also allows for the quantification of crop practices associated with animal feed, either on-farm (in the case of on-farm produced and consumed feed) or off-farm (i.e., purchased feed). We thus reconstitute a «pseudo-farm» gathering both on- and off-farm area cultivated to feed the herd.

2.7.1 Quantity and nutritional quality of livestock feed

The quantity of feed purchased is estimated by dividing the value of the FADN purchased concentrated feedstuffs and coarse fodder variables by the 2020 Farm Inputs Purchase Price Index (IPAMPA) for animal feed ("Indice annuel des prix d'achat des moyens de production agricole (IPAMPA) - Aliments des animaux Insee," 2020). This total kg feed per farm is then allocated to different feed types (e.g., soft wheat grains, soybean meal and corn silage) using the volumes consumed nationally as a distribution key (Sailley et al., 2021) (see "TT_feed_purchased" sheet of Appendix C). The amount of on-farm crops ingested by livestock is estimated by subtracting the quantity of crops sold to the quantity of crop produced, which are variables reported in the FADN. The amount of grass produced on the farm's grassland is estimated from national average grassland yields (Agreste, 2020) (see "yield_SAA_Agreste_2020" sheet in Appendix C). The sum of the on- and off-farm feed quantities gives the total amount of feed consumed on the farm. This total feed quantity is then distributed among the different animals using the average French ration in tons of dry matter per animal and per year as a distribution key (Jayet et al., 2023; see "AROPAJ_France" sheet in Appendix C). The proportion of raw protein provided by the feed is estimated for each feed type from their composition and nutritional values in dry matter (Tran, 2002) (see "feed_table_all_as_DM" sheet of Appendix C).

2.7.2 Crop practices for animal feed production

The crop practices applied to on-farm feed areas are derived directly from available data for the farm (see Section 2.4 and Section 2.5). With the exception of soybean, the crop practices applied to the areas cultivated for the production of purchased feed are approximated to the average practices in France calculated per crop in our study, distinguishing organic and conventional agriculture. The share of purchased feed allocated to soybean meal (see Section 2.7.1) is assumed to be imported from Brazil (Overmars et al., 2015). The BVC of Brazilian soybeans is taken from Lindner et al. (2022) and scaled to the values estimated in our study by correcting for the BVI ratio of conventional French wheat calculated in our study on that estimated by Lindner et al. (2022) for the four practices estimated in Lindner et al. (2022) (See "soybean" sheet in Appendix C). For the landscape variables, that are not present in Lindner et al. (2022), we approximate the Brazilian soybean production to the third most intensive quartile of the French wheat production practices.

2.8 Model calibration and validation according to literature

For each land use type (arable and grassland), three sets of parameters must be calibrated: the range of possible BV_{loc} values per land-use, the parameters of functions used to calculate BVCs in Eq. 1, and the weight of each practice when aggregating BVCs into BV_{loc}. Because we define biodiversity as ecosystem resilience and well function, we select large-scale studies or meta-analysis which can constrain these sets of parameters based on taxa that correspond well to this definition, namely large umbrella species (vertebrates), aquatic species of neighboring streams which concentrate landscape effects, pollinators and the wild plants that support them, etc. Symmetrically, we exclude studies focused on soil microorganisms whose abundance and diversity are primarily enhanced by manure, despite the detrimental impact it can have on other species at landscape level. The range of possible BV_{loc} are derived on the wild plants abundance and diversity reported by Gallego-Zamorano et al. (2022) (Table A1; see Section A.1.2). The two other sets of parameters are calibrated to minimize the average distance between our results and literature values on plants, birds, bees, butterflies, hoverflies, carabids, spiders, bats, bush-crickets, other arthropods and microbes (Beketov et al., 2013; Sánchez-Bayo and

Wyckhuys, 2019; Sirami et al., 2019; Tuck et al., 2014; Vallé et al., 2023) (Table A2; see Section A.1.2). This calibration step uses the optim function from the R base package (Team, 2023).

2.9 Comparison of practices and their impact on biodiversity between certified and conventional products

2.9.1 Choice of FQSs

The choice of FQSs studied was made based on a representativeness criterion: all FQSs with at least 30 farms in the FADN are retained. FQSs with geographical overlap are grouped according to the most frequent designation, after verification that the «farm» sections of the technical specifications are similar. For example, all cheeses with a protected designation of origin (PDO) produced in Franche-Comté are grouped under the name “Comté & Morbier” (PDO Comté, PDO Morbier, PDO Mont d’Or or Vacherin du Haut-Doubs), those with a PDO or Protected Geographical Indication (PGI) produced in Savoie are grouped under the name “Savoie cheeses” (PGI Emmental de Savoie, PGI Raclette de Savoie, PGI Tomme de Savoie, PDO Reblochon or Reblochon de Savoie), those produced in Auvergne under the name “Bleu d’Auvergne & Cantal” (PDO Bleu d’Auvergne, PDO Cantal or Fourme de Cantal).

2.9.2 Comparison of averages between certified and conventional production

The impact of FQS products on biodiversity (BVI_t and BVI_{ha}) is compared to the impact of their conventional counterparts in two complementary manners. To compare multiple conventional and FQS averages, we perform a Tukey HSD using the agriculturae package under R (version 1.3-7;Mendiburu and Yaseen (2020)). However, in order to compare the impact of FQSs caeteris paribus, closer to a causal effect of converting a farm to a given FQSs, we need to control for structural characteristics that are unlikely to be affected by a conversion to or from an FQS. Following Lambotte et al. (2023), we estimate a propensity score based on farm and herd size, technical-economic orientation, whether it is located in a mountain area or not («Mountain Area» category of the natural handicap compensation from the Common Agricultural Policy), farmer age and educational level (see Appendix B). We then apply a simple matching procedure, select for each certified farm the three conventional counterfactuals that are located in the same NUTS2 region and have the closest propensity scores in the same NUTS 2 region. A paired t-test is then applied to compare the average of differences between each FQS farm and its counterfactuals, adjusted for multiple testing by a Bonferroni correction. The same procedure is repeated for agricultural practices in order to interpret the BVI results.

3. Results

3.1 A representative sample of French certified farms

In 2020, *136,777* farms of the French AC’s population declared an FQS (*33*% of the *416,478* recorded farms). Non-organic FQSs are over-represented in our final database, with *40*% of the sample whereas they only represent *26*% of AC farms. Note that *42*% of them are only registered in FQS for wines and other spirits. Excluding the wine and other spirits sectors, our database still records *1,693* farms offering *287* different non-organic certified products, summing up *23*% of the FADN sample. On the other hand, representation of organic farms in our database is close to that of the AC, with *9*% of organic farms in the AC’s population and *11*% our sample. Ultimately, a BVI score is estimated for the productions of *5,511* farms, *7*% of which are organic and *27*% of which are registered in another FQS (excluding wine and other spirits). The *66*% of farms which are not registered as FQS producers are hereafter called «conventional farms».

3.2 Fertilization, pesticides and hedges are the most impacting farming practices

The calibration procedure (see Section 2.8) notably reduces the mean distance between predicted impacts and average *in situ* measurements from the literature, from 1.431×10^{-4} to 3.509×10^{-5} (Table A5). Thus optimized, the model orders the relative effect size of cropland practices - quantified as the change in BVI resulting from a change from the 95th to the 5th intensity percentile of each practice (Table 3) - as follows: fertilization quantity and pesticides dominate, with an effect size around *0.1*, followed by hedges (*-0.04*) and crop diversity (*-0.03*). The impact of other practices is very limited, below *0.005*. On the other hand, both the total amount of fertilization and the share of mineral fertilizers have similar effect size as grassland practices (*0.25* and *0.21*, respectively; Table 3).

This ranking is partly driven by the calibrated weights of each practice (Table A6), which end up being very different from the equal weighting of the original BVI model (Lindner et al., 2019). Similarly, once optimized, some contribution functions are substantially different from the original ones (Fig. A2). For instance, twice as many hedges are needed to reach the maximum BVC while six times less fertilization degrades the BVC down to zero in arable land. On the opposite, the same amount of fertilization degrades the BVC in grassland less faster than in arable land while the slightest share of mineral fertilization drops grassland BVC down to zero[1].

Table 3: ***Effect size of each variable in each land use type****, before and after optimization, calculated as:*
$BVI_{ha}\left(\text{Variable } 95^{th} \text{percentile, other variable medians}\right) - BVI_{ha}\left(\text{Variable } 5^{th} \text{percentile, other va}\right.$

Land use type	Variable	Description	Effect size before optimization	Effect size after optimization	Change in effect size (after / before)
Arable	A.2.1	Hedge Density (linear m / ha)	-0.028	-0.039	1.4
	A.2.2	Mean Field Size (ha)	0.004	0.005	1.371
	A.3.1	Tillage (L diesel / ha)	0.011	0.005	0.436
	A.3.2	Soil Cover (Number of uncovered day)	0.003	1.959 × 10^-8	6.396 × 10^-6
	A.3.3	Crop Diversity (Shannon Index)	-0.032	-0.029	0.923
	A.4.3	Share of mineral fertilization (%)	0.011	0.005	0.436
	A.4.5	Nitrogen fertilization (kg N / ha)	0.079	0.097	1.225
	A.5.1	Pesticides (€~TFI~UDNu / ha)	0.073	0.096	1.327

[1] Note that a grassland with the same BV_{LU} (weighted mean of BVCs) as a cropland has a higher biodiversity value (BV_{loc}) due to higher range of grasslands.

Grassland	A.4.3	Share of mineral fertilization (%)	0.119	0.212	1.778
	A.4.5	Nitrogen fertilization (kg N / ha)	0.137	0.246	1.803

3.3 Food quality schemes: lower impact per hectare but higher impact per unit of product

For a given area of production (e.g., one hectare), organic farming significantly mitigates the biodiversity impact of conventional soft wheat fields to *0.62*, down from *0.70* on average (i.e., before propensity score matching; p-value < *0.0001*, Table A7, Fig. 2). The same applies to milk production: while the hectare mobilized for conventional milk production has an impact of *0.62*, its organic counterpart has a significantly lower impact of *0.50* (p-value < *0.0001*). For milk production, only geographical indications (GIs) of mountain areas (Comté & Morbier PDOs, Bleu d'Auvergne & Cantal PDOs, and Savoie cheese GIs) have a significantly lower average impact per hectare than conventional. This is particularly the case for Franche-Comté PDOs, whose impact per hectare is *11*% lower than the average French conventional milk production (p-value < *0.0001*). The other FQSs (e.g., Beurre de Charentes – Poitou PDO, soft wheat Label Rouge) have an impact per hectare similar to the conventional average. After propensity score matching however, the difference is almost halved for Comté & Morbier farms which are no longer significantly different from their conventional counterparts (*7*%, p-value = *0.19*). Conversely, the difference between organic and conventional farms, for both wheat and milk, is barely affected by the matching procedure.

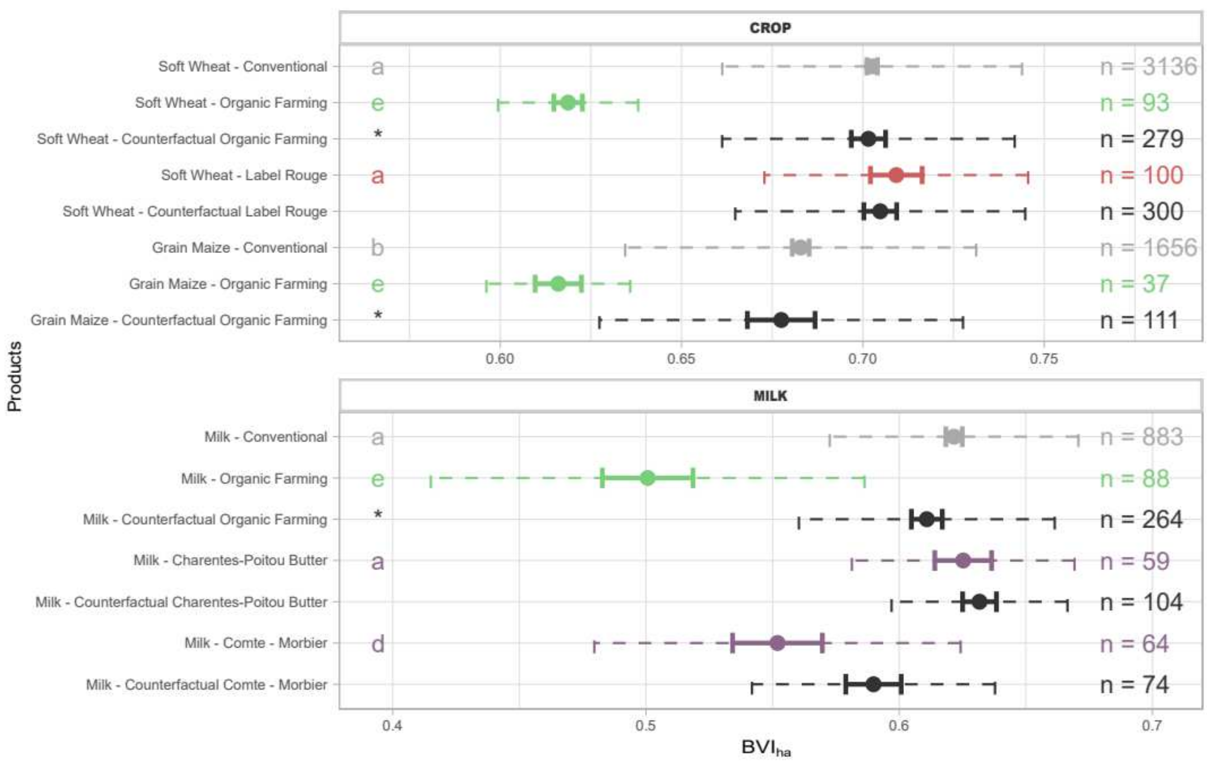

Figure 2: ***Biodiversity impact per unit of area (hectare) for some FQSs and conventional productions.*** *The mean (point), its 95% confidence interval (solid bar) and the standard deviation (dotted bar) are presented here as well as sample sizes (n). Statistical groups of averages before matching are indicated with letters (Tukey HSD p-value ≤ 0.05) and significant post-matching differences with a * (paired t-test, Bonferroni adjusted p-value ≤ 0.05). The full results for all FQSs and products studied are presented in Table 11 and Table 12.*

Table 4: ***Comparison of average scores for common wheat and milk*** *calculated either from Agribalyse ($BVI_{loc,ha}$ from Lindner et al. (2022)) or FADN (BVI_{ha}, BVI_t, BVI_{kcal} and $BVI_{kg\,protein}$ from this study). Number of kcal and kg of protein per product are retrieved from Agence nationale de sécurité sanitaire (2020) (Table A10). * Initial value for 1 kg of Comté, corrected for the ratio of 10 liters of milk required to produce 1 kg of Comté (Husson et al., 2019)*

Product	FQS	$BVI_{loc,ha}$	BVI_{ha}	$BVI_{t\ of\ wheat}$	$BVI_{t\ of\ milk}$	BVI_{kcal}	$BVI_{kg\ of\ protein}$
Soft Wheat	Conventional	0.28	0.70	0.13		3.8×10^{-8}	1.4×10^{-3}
	Organic	0.08-0.11	0.62	0.24		7.0×10^{-8}	2.5×10^{-3}
Milk	Conventional	0.22	0.62		0.24	4.2×10^{-7}	7.3×10^{-3}
	Organic		0.50		0.32	5.7×10^{-7}	9.7×10^{-3}
	Comté PDO	0.1*	0.55		0.30	5.3×10^{-7}	9.1×10^{-3}
	Charentes - Poitou Butter PDO		0.63		0.24	4.2×10^{-7}	7.3×10^{-3}

Some certified products however have a significantly higher impact than conventional ones when considering their impact per unit of product. Indeed, the impact of one ton of organic soft wheat, organic milk and Comté & Morbier PDO is *85*% (p-value = *0.0016*), *33*% (p-value < *0.0001*) and *25*% (p-value = *0.0002*) higher than the average ton of conventional product, respectively (Fig. 3, Table 4, Table A7 and Table A8). Results are not affected by the matching procedure, except for Comté & Morbier PDO milk, whose relative difference of *18*% with its conventional counterpart is no longer significant (p-value = *0.93*). Note that some differences with counterfactuals become non-significant (e.g., for milk products) because the sample size of conventional farms is drastically reduced by the matching procedure, but their average BVI_t remains very close to the value calculated using the whole sample average (Fig. 3, Table A7 and Table A8). FQSs whose BVI_{ha} is similar to conventional farms also generate products with a similar BVI_t (e.g., Label Rouge soft wheat and Charentes - Poitou Butter PDO)

Finally, comparing plant - here wheat - and animal - here milk - products, one notes that plant products have a lower impact per comparable unit of product (Table 4). Although the average hectare of land used for crop production has a slightly higher impact (*11*% between conventional wheat and milk production; Table 4), the impact of animal products is 5 to 11 times higher than plant products when other functional units such as kg protein or kcal are considered, respectively (Table 4). This is also the case for soft wheat and milk organic production (Table 4).

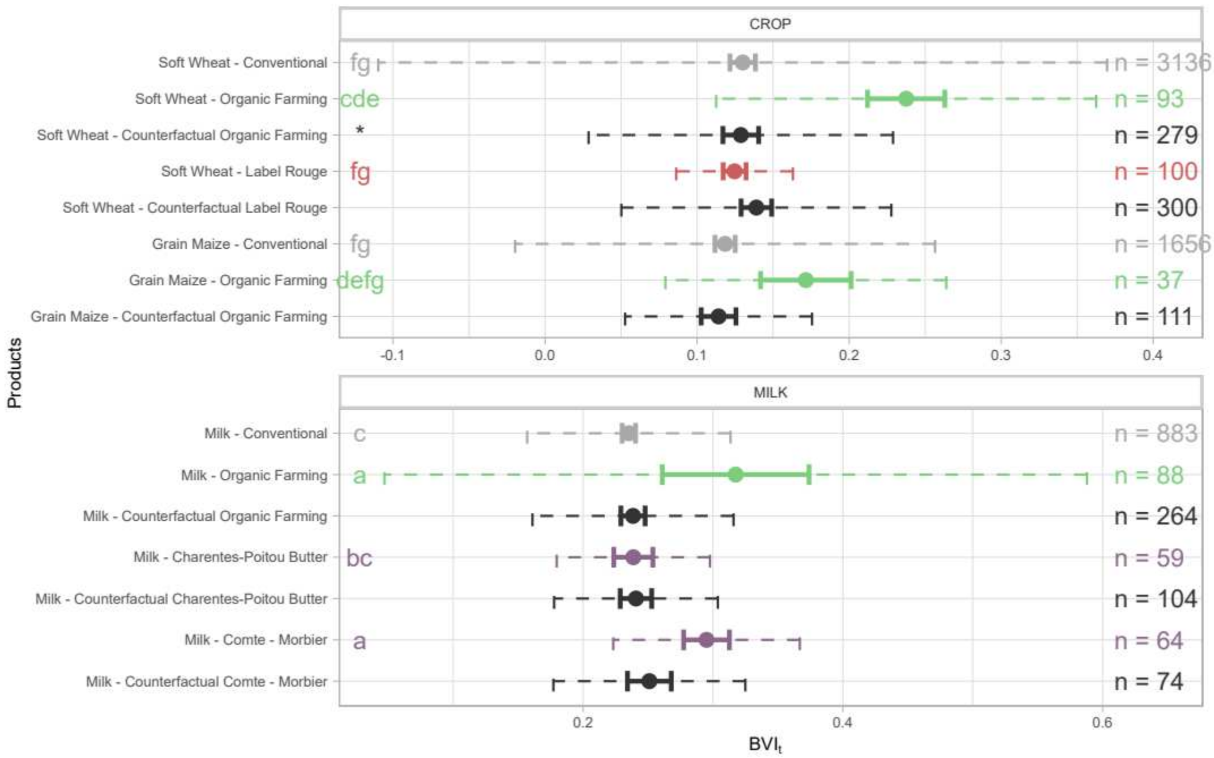

Figure 3: ***Biodiversity impact per unit of product (ton) for some FQSs and conventional products.*** *The mean (point), its 95% confidence interval (solid bar) and the standard deviation (dotted bar) are presented here as well as sample sizes (n). Statistical groups of averages before matching are indicated with letters (Tukey HSD p-value ≤ 0.05) and significant post-matching differences with a * (paired t-test, Bonferroni adjusted p-value ≤ 0.05). The full results for all studied products are presented in Table A7 and Table A8.*

3.4 Comparison of farming practices

For cereal production, only practices from organic farms differ significantly from conventional ones (Fig. 4 and Table A7). First, organic farms do not use nitrogen mineral fertilizers but only organic fertilizers and apply in fine almost one third less total nitrogen fertilizers than conventional farms (-*32*% and -*38*% for wheat and grain maize, respectively). Second, organic cereal production consumes *96*% less pesticides than conventional ones. This reduction in inputs (both fertilizers and pesticides) is concomitant with a much lower yield of organic cereals (-*49*% and -*37*% for wheat and grain maize, respectively). Conversely, no difference in input use or yield is recorded for soft wheat production under Label Rouge. For tillage or landscape variables, almost no difference is noted between conventional and FQSs productions, with the exception of *20*% larger mean field size for Label Rouge soft wheat (Fig. 4 and Table A7).

For milk production, feed content clearly differs between certified and conventional farms (Fig. 5 and Table A8). On average, conventional farms rely on *22*% of permanent grassland and *13*% of temporary grassland to feed their livestock and complement with *43*% of concentrates (Fig. 5 and Table A8). Meanwhile, the share of permanent grassland rises up to *48*% in Comté & Morbier, the share of temporary grassland up to *29*% in organic farming and the share of concentrates drops down to *25*% in organic farming (Fig. 5 and Table A8). In addition, farms under organic farming, Comté & Morbier, Bleu d'Auvergne & Cantal and Savoie cheeses FQSs have a lower livestock density than conventional farms (from *0.7* to *0.9* livestock unit of dairy cow / ha of main forage area on average for these FQSs against *1.4* for conventional farms; Fig. 5 and Table A8). Among FQSs, organic farming also stands out in terms of feed autonomy, with *73*% of feed produced on-farm compared to *57*% for conventional farms. This greater feed autonomy comes from a higher share of temporary grassland and legumes area, and a lower use of purchased soybean and overall concentrates (Fig. 5 and Table A8). However, some FQSs have lower yields, with lower milk productivity per hectare of pseudofarm. These lower yields have two causes. For organic farms, the main reason is the lower productivity of cows which may be caused by the lower share of concentrates in their feed while the livestock density per ha of pseudo-farm is almost equal to the conventional average (Fig. 5 and Table A8). For Comté & Morbier, Bleu d'Auvergne & Cantal and Savoie cheeses however, cows are almost as productive as in conventional farms, but livestock density (i.e., the number of cows per hectare of pseudofarm) is substantially lower (Table A8). Unlike all other FQSs, Charentes-Poitou Butter PDO farms do not differ in any significant way from conventional farms, except for a *20*% higher share of forage maize (Fig. 5 and Table A8).

Besides, farming practices applied for the production of dairy cow feed do not differ between FQS and conventional ones, except for organic farming and Comté & Morbier FQSs (Table A9, Fig. A3, Fig. A4, Fig. A5 and Fig. A6). First, organic and Comté & Morbier farms use fewer mineral fertilizers, leading to an overall nitrogen fertilization decrease of *24*% and *9*% compared to their counterfactuals, respectively (Table A9). Secondly, as for cereal production, organic feed is produced with fewer pesticides and higher ground cover (Table A9). Surprisingly, organic farms purchase feed from plots surrounded by *33*% more hedges, reflecting the higher share of rough feed (such as hay) purchased instead of concentrates.

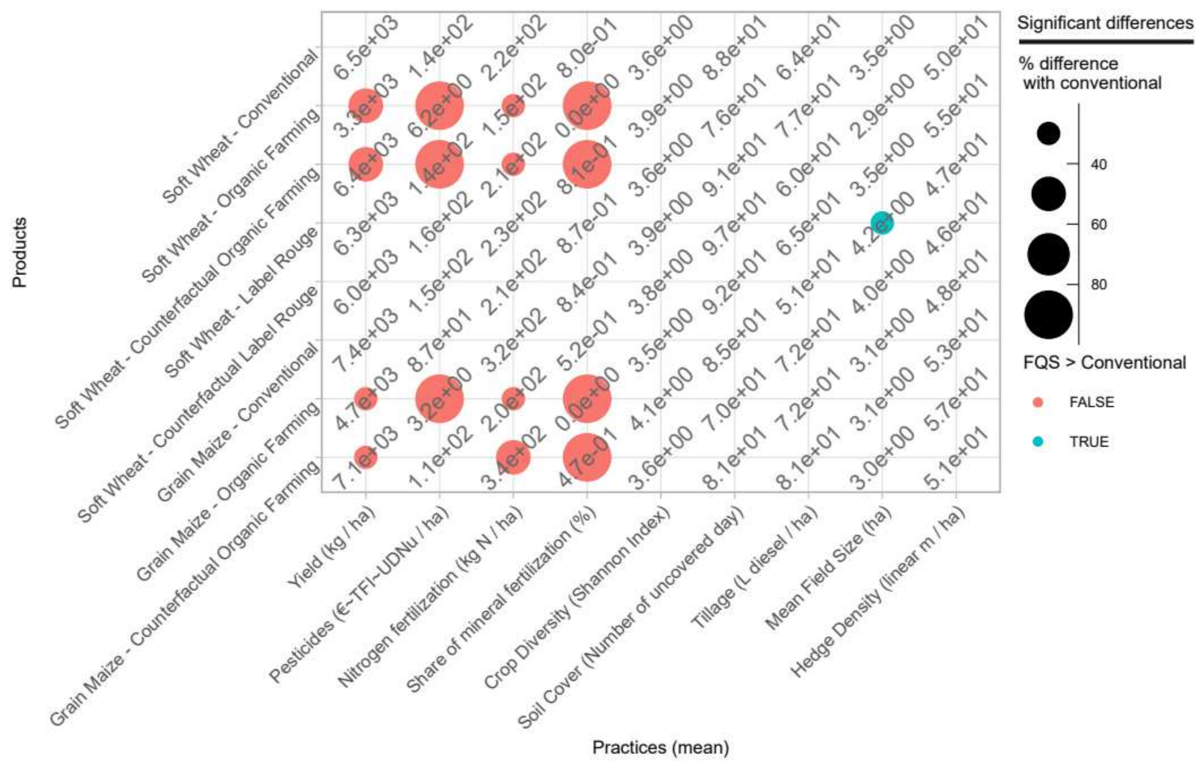

Figure 4: ***Crop practices for some FQSs and conventional products.*** *The mean is displayed with a dot for significant differences between FQS and either conventional (displayed on FQS row; Tukey HSD p-*

value ≤ 0.05) or counterfactual (displayed on counterfactual row; paired t-test, Bonferroni adjusted p-value ≤ 0.05). The dot color depends on whether the comparison between the FQS and either conventional or counterfactual is positive (blue dot) or negative (red dot). The dot size depends on the proportion of the difference between the FQS and either conventional or counterfactual. The full results for all FQSs and products studied are presented in Table A7.

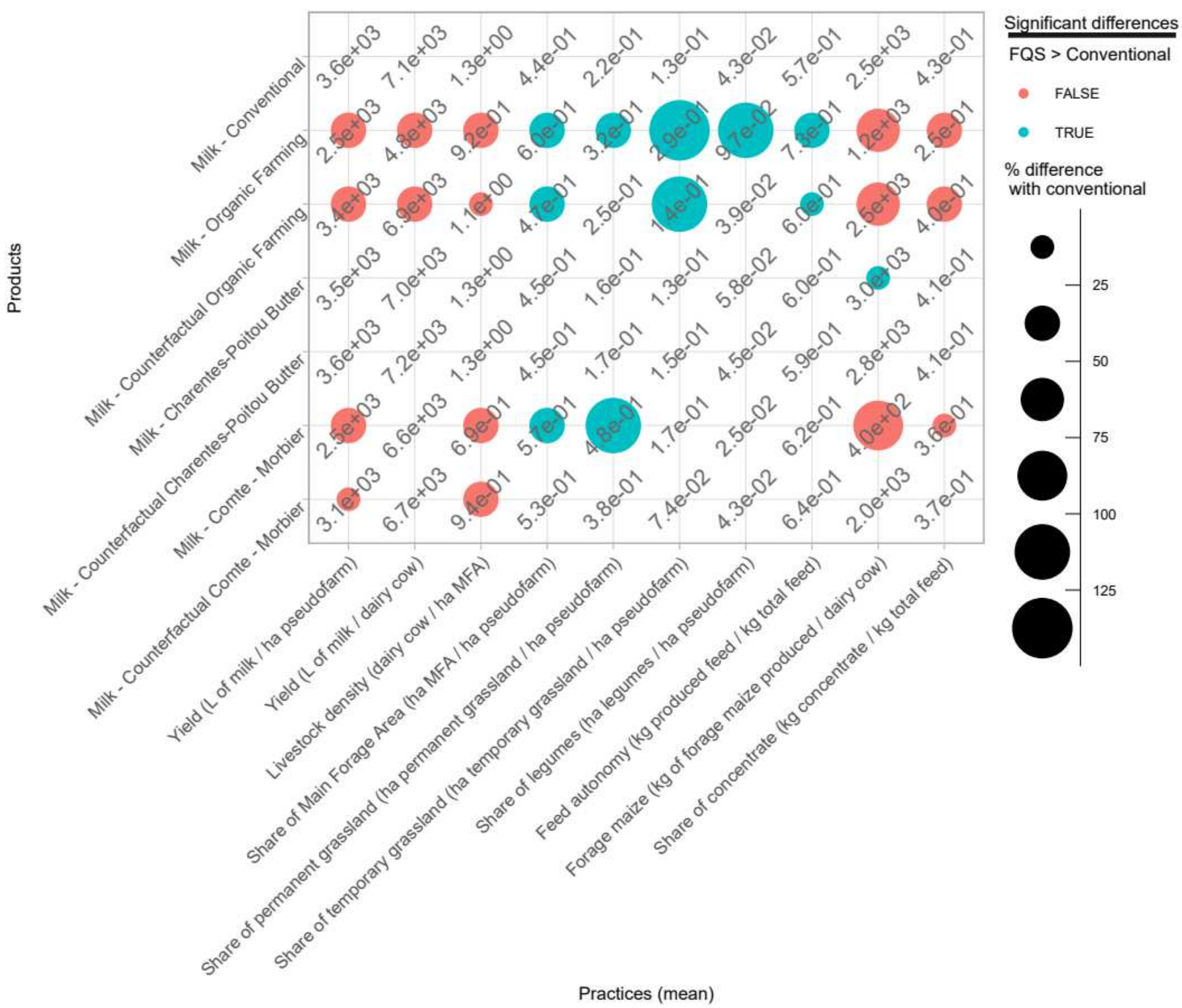

*Figure 5: **Husbandry practices for some FQSs and conventional milk productions.** The mean is displayed with a dot for significant differences between FQS and either conventional (dot displayed on FQS row; Tukey HSD p-value ≤ 0.05) or counterfactual (dot displayed on counterfactual row; paired t-test, Bonferroni adjusted p-value ≤ 0.05). The dot color represents whether the comparison between the FQS and either conventional or counterfactual is positive (blue dot) or negative (red dot). The dot size depends on the proportion of the difference between the FQS and either conventional or counterfactual. The full results for all practices, FQSs and products studied are presented in Table A8. The feed production practices are presented in Table A9, Fig. A3, Fig. A4, Fig. A5 and Fig. A6.*

4. Discussion

4.1 Differences in biodiversity impact: comparison with the literature

4.1.1 Food quality scheme vs conventional fields and products

We estimate that, for the ten studied products, organic farming has a lower impact per hectare on biodiversity than conventional farming, ranging from a *3*% to *20*% difference. This is logically consistent with the literature based on *in situ* measures used for calibration (Tuck et al., 2014). Indeed, the estimated biodiversity value ($BV_{loc}=1-BVI_{ha}$) of organic wheat fields is *42*% higher than the BV_{loc} of conventional fields (Table A5), close to the meta-average of *34*% reported by Tuck et al. (2014). It is however substantially lower that estimated by Lindner et al. (2022) based on the Agribalyse life cycle impact (LCI) database: while Lindner et al. (2022) found that impact on biodiversity is on average *50*% lower in organic farming than in conventional farming (Table 4). This difference comes from two main improvements in our approach. Firstly, we included landscape variables in our model, and they have similar values in organic and conventional farms (Table A7). For hedge density in particular, which is the most impacting landscape variable (Table 3), we found no significant difference between organic and conventional farms. This result is consistent with OFB's (French Office for Biodiversity) findings based on all LPIS plots (Transition écologique et al., 2023). However, contrary to OFB's study, we do not find a substantial difference in crop diversity between organic and conventional productions. This difference can most likely be explained by the difference in study scale: OFB aggregates all organic or conventional land at the small agricultural region level whereas we compute farm-level diversity indicators. Indeed, when we compute the Shannon index of crop diversity at larger scales, such as counties and regions, it becomes higher for organic than conventional productions (e.g., cereals), and comparable to OFB's results. This can most likely be interpreted as showing that inter-farms diversity is higher in the organic sector, but not intra-farm diversity. Secondly, we calibrate the BVIAS model based on *in situ* measurements, resulting in substantial changes in parameters compared with Lindner et al. (2022), in particular for hedge density, mean field size, fertilization and pesticides which effect size increased by *33*% on average (Fig. A2 and Table 3). For other FQSs, the only point of comparison is the semi-quantitative study by Alliot et al. (2021), whose ranking is similar to ours: organic and Comté PDO obtain higher biodiversity scores ("good", 4/5) than Cantal PDO ("weak", 1/5).

On the opposite, we estimate that organic farming has a similar or higher impact per ton of product than conventional farming. Studies on impacts per unit of product (e.g., cereal or milk) are scarce and poorly consistent with meta-analyses comparing organic and conventional farming. Indeed, we found only two studies comparing the impact per unit of product between organic and conventional farming (Lindner et al., 2022; Tuomisto et al., 2012)). Tuomisto et al. (2012) reported an impact *26*% lower for organic crops. However, they used input data that do not correspond to actual farms but "typical averages" and, more importantly, they consider that biodiversity is three-fold higher in organic fields, based on De Schryver et al. (2010), a study limited to plants in England, which is at odds with the *30-40*% range found in recent meta-analysis relying on many more taxa and geographical areas (Tuck et al., 2014). Lindner et al. (2022) reported a *17*%, *25*% and *31*% lower biodiversity impact for organic maize, wheat and sunflower, respectively. However, they show differences in impact per kg that would imply *1.5-2* times less impact on biodiversity per hectare, i.e., an average *71*%-*167*% higher biodiversity value per hectare, based on an average biodiversity value of *0.32* for arable land, which is again at odds with recent meta-analysis.

We found four studies estimating differences in biodiversity impact per kg of milk between organic and conventional farms. All report a lower impact of organic farming per kg of milk: *-76*% in Guerci et al. (2013), *-60*% in Mueller et al. (2014), *-77*% in Knudsen et al. (2019), and *-42*% in Lindner et al. (2022). The first two derive their characterization factors from De Schryver et al. (2010). For the arable crops, they therefore suffer from a lack of consistency with recent meta-analysis on biodiversity levels in conventional and organic arable crops (see previous paragraph). Some of the difference is driven by a larger share of permanent grassland in organic farms (e.g., *23*% vs *13*% in Mueller et al. (2014), *63*% vs *23*% in Guerci et al. (2013)), whereas in our – much larger – sample, the average share of permanent grassland is the same for organic and conventional farms. The bulk of the difference however comes from a much lower impact of organic grassland on biodiversity (positive impact, characterization factor = -*0.02*) compared with conventional grassland (characterization factor = *0.28*). This implies a *42*% higher biodiversity value of organic grassland, which is at odds with the *17*% difference reported by Tuck et al. (2014) for grassland. The finding that an organic meadow with a stocking rate of *0.9* livestock units per hectare has a higher biodiversity value than natural habitat (De Schryver et al., 2010) is also somewhat counter-intuitive. In Knudsen et al. (2019), the substantially positive characterization factors of most grasslands are even more disturbing: although livestock densities are not shown, they are likely close to or higher than *1* based on the reported milk production per hectare. Note that one paper compares the biodiversity impact of Italian PDO milk to Italian conventional milk, finding a *45-200*% higher impact of PDO milk (Battini et al., 2016).

In a nutshell, while our findings are – by construction – consistent with the literature regarding the biodiversity impact of FQS land (BVI_{ha}), we diverge from the very few studies which ventured to estimate the biodiversity impact of FQS products (BVI_t). Differences between our results and previous studies are probably largely driven by the calibration of our model using large-scale studies and meta-analysis. Moreover, our reliance on numerous real farms instead of a few, sometimes simulated, ones probably also weighed on the results. The lack of consideration by previous studies of landscape variables – which tend to be similar between organic and conventional farms – may also contribute to their overestimation of the mitigation of biodiversity impacts by organic farms.

4.1.2 Plant vs animal products

We found that animal -here milk- products have a higher impact per ton than cereals (e.g., *0.24* versus *0.13* for conventional milk and wheat, respectively). This difference would increase three-fold if considering the amount of calories in each product (e.g., 4.2×10^{-7} versus 3.8×10^{-8} for conventional milk and wheat, respectively). This result is consistent with the literature. For instance, Lindner et al. (2022) also found a twice as high impact for milk product than cereal-based one. Similarly Read et al. (2022) found that beef and pork meat alone explained *44*% of the impact on biodiversity of food consumption in Europe. Besides, Read et al. (2022) show that adopting a diet reducing the consumption of animal products, such as the EAT Lancet Planetary Health diet or the recommended vegetarian diet of the US Department of Agriculture (USDA), nationwide would almost halve the biodiversity footprint of food consumption.

4.2 Direct and indirect role of FQSs specifications

The differences we observed in terms of biodiversity impact per hectare between FQSs and conventional agriculture can be explained by differences in estimated practices, which are consistent with the technical specifications of FQSs. For crop production, the use of pesticides as well as the share of mineral fertilization are the main factors differentiating organic from conventional farming in our study. Furthermore, as organic farms do not use mineral nitrogen fertilizers and do not replace all these inputs by organic manure, their nitrogen balance per hectare is lower than those of conventional productions. These results are consistent with specifications of organic farming, which prohibits both the use of synthetic pesticides and of mineral nitrogen fertilizer but allows the use of organic nitrogen. However, while organic specifications recommend preserving semi-natural elements and cultivating a high variety of crops ("Production biologique et étiquetage des produits biologiques," 2018), we do not find any differences in hedge density, crop diversity, nor field size between organic and conventional farms. Conversely, the production of soft wheat in Label Rouge does not use different practices than conventional one. This is consistent with this FQS specifications, which focus mainly on post-harvest processes, product quality and organoleptic properties, without heavy constraint on wheat production (“Arrêté du 10 octobre 2022 portant homologation du cahier des charges du label rouge n° LA 09/05 « Farine de blé »,” 2022).

For milk production, the main practices that differ between certified and conventional husbandries relate to feed content. Organic farms and those certified in Comté & Morbier PDO have a share of temporary and permanent grassland twice as large as conventional farms. Moreover, organic farms rely less on concentrates and purchased feed, increasing their autonomy. Again, it is reasonable to assume that the specifications explain some of these differences. For instance, while the *60*% threshold of self-sufficiency is a requirement for organic husbandry practices, specifications do not set any requirements on the share of permanent grassland (“Production biologique et étiquetage des produits biologiques,” 2018). The higher presence of temporary grassland may be an indirect consequence of the absence of synthetic pesticides and mineral fertilizers. Indeed, the insertion of temporary grassland in rotations is an agroecological strategy recognized to offset the use of synthetic inputs (Franzluebbers and Gastal, 2019). Besides, specifications for the Comté PDO require that most feed comes from the geographical area, that each cow have at least one hectare of grassland and that temporary grasslands represents a maximum of 15% of the farm forage area (“Cahier des charges de l’appellation d’origine « Comté»,” 2015).

4.3 Delicate balance between pollution and land consumption

Differences between impacts per hectare and impacts per ton illustrate the importance of the balance between pollution and land consumption when quantifying environmental impacts in a way that is compatible with the environmental labeling of food products. This delicate balance is already well documented in the literature, both in conceptual debate on land sharing vs sparing (Erm et al., 2023; Gong et al., 2022; Ricciardi et al., 2021) or prospective scenarios on food system sustainability and nutrition security (Mouel et al., 2018) and in empirical applications such as the climate impact (Bellassen et al., 2021) or water impact (Bodini et al., 2021) of food products.

In our sample, the yield difference between organic and conventional cereals (e.g., -*48*% for soft wheat, -*37*% for grain maize) is substantially larger than the values of -*20*% (Ponisio et al., 2015) and -*25*% (Seufert et al., 2012; Smith et al., 2019) reported in global meta-analysis. However, these values correspond to averages across crop types, but when considering only wheat, they estimate a difference of -*35*% to -*40*% (Seufert et al., 2012; Smith et al., 2019). Moreover, meta-analyses are mainly based on experimental plots where organic yields are probably overestimated compared to real farms where the substitution of inputs by human labor is more difficult. Our results are very similar to other studies on cereals based on French in-farm surveys (Coinon, 2022; Dubosc et al., 2016).

The balance between pollution and land consumption is particularly exacerbated for soft wheat while slightly less critical for milk and other crop products: a ton of organic milk and grain maize are *33*% and *50*% more impactful, respectively, while a ton of organic wheat is *84*% more impactful than their conventional counterfactuals. This is partly due to the lower yield gap between organic and conventional milk production: -*30*% in our present study (when considering the whole pseudo-farm), which is similar to Meier et al. (2015) but lower than the -*52*% reported by Lambotte et al. (2023). For other dairy GIs, we find that yield per hectare of pseudo-farm is *20*% lower for Comté & Morbier and there was no significant difference for other GIs, close to the median difference of -*4*% reported by Bellassen et al. (2021) for 3 GI animal products. At the opposite, we found a yield gap of -*48*% for organic soft wheat compared to the conventional counterfactuals, which is between the -*35*% reported by Walder et al. (2023) and the -*54*% reported by Gabriel et al. (2013). These differences between the two studies could be explained by the location of their sampling sites: while Walder et al. (2023) sampled fields in Switzerland, Gabriel et al. (2013) studied landscapes in England.

The greater or lesser compensation between pollution and land consumption is highly sensitive to the difference in biodiversity levels between natural land and cropland (Gong et al., 2022). In empirical papers selected to calibrate the BVIAS model, the three-fold difference between the average BV_{loc} of cropland and "natural" land (Gallego-Zamorano et al., 2022) largely trumps the *34*% difference between conventional and organic cropland (Tuck et al., 2014) and the *35*% difference between fields with low versus high pesticide use (Beketov et al., 2013). This three-fold difference is also reported in formally elicited expert judgments on pollinator abundance (Alejandre et al., 2023). More generally, this three-fold difference is consistent with the strong emphasis put by the IPBES on land consumption/protected areas compared to pollution and its mitigation in terrestrial ecosystems (IPBES, 2019) and also fuels the "land sparing vs sharing" debate.

4.4 Yields and rebound effects

Beyond our modeling framework, changes in yield may have consequences on biodiversity due to rebound effects. In a general equilibrium model or consequential LCA, lower yields could be less damaging to biodiversity than in our model, for example if lower yields result in higher prices and demand is elastic (hence decreases) and supply increases at the intensive margin. Reversely, lower yields could be more damaging to biodiversity, for example if demand is inelastic and agriculture encroaches on the margins of natural ecosystems with high diversity (e.g., tropical forest). To our knowledge, studies on this subject are rare for climate impact (Bellora and Bureau, 2016; Searchinger et al., 2018) and even rarer for biodiversity (Desquilbet et al., 2017). Nonetheless, Desquilbet et al. (2017) estimated that even with the slightest price elasticity of demand for agricultural goods and lower productions cost for conventional farming, which is the case for most food products, as well as with a relationship between biodiversity and yield that is not highly convex, there is a rebound effect of intensive agriculture, with an increase in the size of the market and therefore more negative impacts on biodiversity than extensive agriculture.

Along the lines of Gong et al. (2022), if the proportion of land "freed" by the deconversion of one hectare of organic wheat which is affected to additional food production instead of conservation is higher than *83*% - our estimated difference in BVI_t for wheat, the "consequential" impact of deconversion on biodiversity becomes negative. These considerations do not matter much in the context of environmental labeling because whatever happens to the land that would not be used for a given product thanks to higher yield would be attributed to the products of this land, be they food, fiber or nothing in the case of protected areas (in which case the impact is nil). They may however matter for other policies such as subsidies for organic farming.

4.5 Implications for environmental labeling and other policies

Our study goes beyond existing studies based on *in situ* measurement or life cycle assessments, which often cover a small number of farms. Indeed, we estimate practices and biodiversity impact for dozens of productions throughout France and compare them between certified and conventional farms with similar geographical and technical-economic characteristics. This large-scale study is made possible by the use of FADN data, which provides data on a uniquely large sample of productions and farms.

Our study provides an objective, robust and operational method to calculate the impact of food products on biodiversity from agricultural accounting data. We apply this method to the comparison between certified and conventional products, as well as between different products (e.g., milk vs wheat). However, this method could also be applied to other types of differences (e.g., between regions, between farm sizes, etc.), provided that accounting data is available for a sufficient number of farms in each of the groups to be compared. It is also compatible with the environmental labeling and the LCA framework since impacts on biodiversity accumulate with quantities consumed. In the context of environmental labeling, and if considering the impact per kg, our study suggests that FQSs farms do not deserve a better grade than conventional farms. They may not deserve a worst grade neither, since the impact difference between FQS and conventional products is smaller than the between product differences (i.e., milk versus crops).

Despite the trade-offs between biodiversity impact and yields, we think organic farming should still be supported for several key reasons. Although organic agriculture has, generally but not always, a higher impact on biodiversity per kilogram of product due to lower yields, its consistent lower per-hectare impact makes it more beneficial for local biodiversity conservation. In such context, the impact per unit of land is crucial for maintaining local ecosystems. Moreover, at the consumer level, the disadvantage of organic products could be mitigated by the diets of their consumer. Organic consumers typically have a lower overall environmental impact, as they tend to consume more plant-based foods and fewer animal products, reducing their overall ecological footprint (Baudry et al., 2019). This is consistent with supply and demand modeling showing that the higher price resulting from the lower profitability of extensive farming, such as organic one, reduces the demand for animal products, which has higher price elasticity than plant-based products (Desquilbet et al., 2017).

There are therefore many reasons and contexts for policies supporting organic farming and organic product consumption. For instance, the French government passed a law in 2009 to promote organic farming on drinking water catchment basins, since this production method effectively prevents water pollution ("LOI n° 2009-967 du 3 août 2009 de programmation relative à la mise en œuvre du Grenelle de l'environnement (1)," 2009). This is of peculiar importance as organic farming practices align with the precautionary principle in public health (Kim et al., 2017; Mesnage and Antoniou, 2018), help mitigate risks from poorly studied pesticide additive and synergistic effects (Le Magueresse-Battistoni et al., 2018; Relyea, 2009) and provide organic food which consumption is beneficial for health by reducing pesticide exposure (Jiang et al., 2024; Velimirov et al., 2010).

4.6 Main limits

4.6.1 Climate change and water use

In our study, two of the five main threats to biodiversity listed by the IPBES are taken into account: land use and pollution. Of the remaining three, only two are substantially influenced by agriculture: climate change and direct resource exploitation (more specifically freshwater use for agriculture). Climate change is the third most important threat to biodiversity and is expected to become the main threat by the end of the century (IPBES, 2019). The food sector is responsible for *26*% of global greenhouse gas emissions, and these emissions are mostly farm-based (Poore and Nemecek, 2018). Similarly, agriculture is the sector with the largest consumption of freshwater, with, for instance, *62*% of the water consumed in France in 2020 (Arambourou et al., 2024).

The estimation of greenhouse gas emissions and water consumption from different agricultural products and its inclusion in a biodiversity impact model such as BVIAS seems therefore essential in the medium term. Note however that these additions would likely not change drastically the difference between organic and conventional products as their carbon and water footprint tend to be similar (Bellassen et al., 2021; Bodini et al., 2021). Including these drivers of biodiversity loss would also contravene the principle of separation of impacts in LCAs. There is however a necessary trade-off between this principle and the display of transparent results on environmental impacts which are recognized and matter for society and policymakers.

4.6.2 Off-field effects and non-linear impacts of pollutants

One intrinsic limitation of BVIAS, environmental labeling or any generic method to capture the environmental impacts of food production is the difficulty to account for non-linear impact of pollutants at landscape level. This is obvious for nitrate pollution and the use of pesticides, which have both biodiversity and health impacts. Nitrate pollution becomes exponentially problematic with quantities of excess nitrogen (Peyraud et al., 2012). For this reason, two neighboring livestock farms have more impact than the same farms with more distance from one another. This could in turn generate counter-intuitive impacts for geographical indications. Comté PDO for example was found here to have a lower BVI_{ha} (Table A8) and a lower overall amount of nitrogen fertilization (Table A9), but its higher profitability drives a concentration of milk farms in its denominated area, which in turn could be locally detrimental to health and the environment. Regional factors could be a solution to factor this in but would themselves raise their lot of technical and ethical questions.

Another particularly acute limit comes from the fact that the literature used for calibration only partially takes into account the impacts of practices outside the field, in particular the impact of pesticides and nitrogen in space and time. The volatilization rate of pesticides can reach 90% of the doses applied for some molecules (Calvet, 2005) and residues of about *80*% of applied pesticides can be detected (with half of these found as transformation products) with a persistence of more than a decade (Chiaia-Hernandez et al., 2017). Moreover, the negative impact of pesticides on biodiversity outside crop fields is far from zero. For example, the number of plant species per *4*m\2 plot decreases from more than *30* species in a permanent grassland adjacent to an organic crop field to *20* species in a permanent grassland adjacent to a conventional crop field (Schöpke et al., 2023). This negative effect of conventional agriculture on the flora persists at least up to *50* meters inside the permanent grassland. Other studies have shown the negative effect of conventional farming and/or pesticides on biodiversity in hedges, grassy strips or even forests adjacent to crops (Batáry et al., 2010; Caprio et al., 2015; Holzschuh et al., 2008; Rundlöf et al., 2008).

However, the number of taxonomic groups and EBVs (Essential Biodiversity Values), the distances from the plot, and the pedoclimatic contexts covered by these studies remain too limited to include this effect in our model. Nonetheless, a back-of-the-envelope calculation based on them shows that the difference of biodiversity impact between organic and conventional wheat could be reduced by *30*% in a typical landscape with *10*% of semi-natural habitat and even be totally cancelled in a landscape with *40*% of semi-natural habitat (see “off_field_impact” sheet in Appendix C), supporting once again the promotion of organic farming in natural reserves.

Similarly, no interactions are assumed between pollutants. For example, BVIAS does not condition the positive impact of hedges on biodiversity to pesticides not exceeding a given threshold, or vice-versa. This limit is not intrinsic however: if such interactions were properly documented by *in situ* ecological studies, including them in the model would be rather straightforward.

4.6.3 Scarcity of consolidated evidence and practice proxies

Finally, a major limit of our study comes from the small set of large-scale studies and meta-analysis available to calibrate the model. We found only five studies focusing on biodiversity variables and taxa that are representative of social concerns regarding biodiversity. Indeed, the majority of *in situ* studies of biodiversity impact focus on soil organisms, and in particular soil micro-organisms (Babin et al., 2023), which are not reliable proxies of important services such as pollination or concerns such as species extinction. More importantly, the services provided by soil organisms are largely internal in economics terms (i.e.,they directly benefit farmers and land managers).

Nonetheless, we show here that the use of accounting data to inform environmental labeling is reachable, robust and promising. However, accounting data is often only a proxy for the intensity of a practice we are trying to characterize. For example, for tillage, although the estimated diesel consumption for ploughing is consistent with national averages (Chenu and Butault, 2015), we were unable to estimate whether it was deep or shallow. Similarly, for organic fertilizer inputs from farms without livestock, as the FADN variable for this input is very poorly informed, we had no choice but to apply a standard value equal to the national average (Ministère De L'Agriculture (SSP), 2019). Recognizing the potential of these accounting data for assessing the environmental impact of agriculture, members of the European Commission and the European Parliament reached an agreement to amend the FADN regulation ("Conversion of the Farm Accountancy Data Network into a Farm Sustainability Data Network," 2023). This new text aims to transform FADN into a Farm Sustainability Data Network (FSDN) to better reflect the objectives of the "From Farm to Fork" strategy. This amendment extends the collection of data, previously limited to microeconomics and accounting data, to include environmental data that allows for the estimation of greenhouse gas emissions and carbon storage, soil health, water use and the adoption of agro-ecological practices. Such additional inputs would allow for a more precise estimation of the within-field agricultural practices, as well as other variables impacting biodiversity such as ground cover with intercropping and landscape variables.

Similarly, but of much lower importance, the FADN is not built to be representative of specific schemes, such as organic farming or FQS and only sample farms above *25,000*€ of standard production. Thus, FQS with few or very small farms might not be represented in the FADN. Overall, however, the French FADN, which covers more than *90*% of the national standard production and more than *90*% of the agricultural area, is representative of the bulk of agricultural products.

5. Conclusion

We show here that organic agriculture and some other FQS such as Comté & Morbier PDO have a lower impact per hectare on biodiversity. However, their lower yield more than offset such advantage when looking at the per ton impact, resulting in a higher impact per ton for organic wheat and milk. We also show that only specifications lead to effective differences in practices. For instance, organic productions do use less pesticides and mineral fertilizers, as mandated by specifications. On the other hand, organic productions do not have more hedges, smaller fields or a higher crop diversity, as these practices are only mentioned in specifications and not mandated. Besides, our method substantially improves existing literature thanks to the unprecedentedly large sample at the farm scale we used, allowing for the consideration of land use and yields, major pollution sources with effect size and landscape effects. Furthermore, we calibrated our model on meta-analysis or large-scale studies based on species diversity of multiple taxa related to ecosystem services (e.g., pollinators) or ecosystem functioning (e.g., vertebrates). Finally, we show that the difference in impact between animal products, such as milk, and plant products, such as wheat or grain maize, is much higher than the difference between the FQS and conventional versions of the same product. This demonstrates that to limit our impact on biodiversity, reducing our consumption of animal products is a much higher priority than choosing a different version of what we already eat.

For these reasons, we think that our results are robust enough to either support the biodiversity component of an environmental labeling score, or to recommend the absence of a "biodiversity" bonus for French FQSs in such a scheme. As all scientific results, this conclusion could be reversed in the future, in particular after enlarging the geographical scope – from France to the EU for example – or after considering a wider range of products, and in particular meat and eggs. New biodiversity measurements could also lead to changes in the calibration of the BVIAS model, and therefore to changes in its conclusions.

More importantly, this conclusion pertaining to the biodiversity component of environmental labeling should not be recklessly extended to other policies. For example, our results justify support to organic farming and a few other FQSs when the per hectare impact is more pertinent, such as in natural reserves. If the objective is to maximize biodiversity within the perimeter of the reserve, and the choice is only between FQS and conventional productions, yield becomes irrelevant and organic farming or Comté PDO are to be favored. More generally, the costs and benefits of FQSs are not limited to biodiversity.

6. Funding

This project benefited from funding by ADEME, project "Impact des modes de production des produits alimentaires sous label sur la biodiversité" (BiodivLabel, grant agreement No. 2203D0043), and by the European Commission, EJP Soil projects Road4Scheme and SERENA (grant agreement No. 862695).

7. Acknowledgements

We would like to warmly thank Christian Bockstaller for his advice, help and proofreading. We would also like to thank Fabrice Vinatier for his valuable feedback on the impact of practices on biodiversity. More generally, we would like to thank all the members of the BiodivLabel collective expertise who contributed by their proofreading or discussions to enrich this work, especially Cecile Bessou, Emmanuelle Porcher, Anne Farugia, Catherine Donnars and Guy Richard.

8. References

Agence nationale de sécurité sanitaire, de l'environnement. et du travail (Anses)., de l'alimentation, 2020. Table de composition nutritionnelle des aliments Ciqual.

Agreste, 2020. Statistique agricole annuelle (SAA) - Cultures développées (hors fourrage, prairies, fruits, fleurs et vigne) - 2020 - Départements.

Alejandre, E.M., Scherer, L., Guinée, J.B., Aizen, M.A., Albrecht, M., Balzan, M.V., Bartomeus, I., Bevk, D., Burkle, L.A., Clough, Y., Cole, L.J., Delphia, C.M., Dicks, L.V., Garratt, M.P.D., Kleijn, D., Kovács-Hostyánszki, A., Mandelik, Y., Paxton, R.J., Petanidou, T., Potts, S., Sárospataki, M., Schulp, C.J.E., Stavrinides, M., Stein, K., Stout, J.C., Szentgyörgyi, H., Varnava, A.I., Woodcock, B.A., Bodegom, P.M. van, 2023. Characterization Factors to Assess Land Use Impacts on Pollinator Abundance in Life Cycle Assessment. Environmental Science & Technology 57, 3445–3454. https://doi.org/10.1021/acs.est.2c05311

Alliot, C., Feige-Muller, M., Mc Adams-Marin, D., Gissinger, A., Benoit, G., Segré, H., Ly, S., Duval, L., Laroche, K., Dalle, S., Ducos, L., D'halluin, J., Uthayakumar, T., Legeay, M., 2021. Etude de démarches de durabilité dans le domaine alimentaire - Rapport d'analyse transverse. BASIC, Greenpeace, WWF.

Améliorer l'évaluation environnementale des viandes, 2020. Interbev.

Andreasi Bassi, S., Biganzoli, F., Ferrara, N., Amadei, A., Valente, A., Sala, S., Ardente, F., 2023. Updated characterisation and normalisation factors for the environmental footprint 3.1 method. Publications Office of the European Union, Luxembourg.

Arambourou, H., Ferrière, S., Oliu-Barton, M., 2024. Prélèvements et consommations d'eau : Quels enjeux et usages ? (No. 136). France Stratégie.

Araya, S., Ostendorf, B., Lyle, G., Lewis, M., 2018. CropPhenology: An R package for extracting crop phenology from time series remotely sensed vegetation index imagery. Ecological Informatics 46, 45–56. https://doi.org/10.1016/j.ecoinf.2018.05.006

Arrêté du 10 octobre 2022 portant homologation du cahier des charges du label rouge n° LA 09/05 « Farine de blé », 2022.

Arrêté du 27 avril 2017 définissant la méthodologie de calcul et la valeur des doses unités de référence des substances actives phytopharmaceutiques, 2017.

Babin, C., Espagnol, S., Aubin, J., 2023. Effects of agricultural practices on biodiversity. A review.

Bamière, L., Bellassen, V., Angers, D., Cardinael, R., Ceschia, E., Chenu, C., Constantin, J., Delame, N., Diallo, A., Graux, A.-I., Houot, S., Klumpp, K., Launay, C., Letort, E., Martin, R., Mézière, D., Mosnier, C., Réchauchère, O., Schiavo, M., Thérond, O., Pellerin, S., 2023. A marginal abatement cost curve for climate change mitigation by additional carbon storage in French agricultural land. Journal of Cleaner Production 383, 135423. https://doi.org/10.1016/j.jclepro.2022.135423

Batáry, P., Matthiesen, T., Tscharntke, T., 2010. Landscape-moderated importance of hedges in conserving farmland bird diversity of organic vs. Conventional croplands and grasslands. Biological Conservation 143, 2020–2027. https://doi.org/10.1016/j.biocon.2010.05.005

Battini, F., Agostini, A., Tabaglio, V., Amaducci, S., 2016. Environmental impacts of different dairy farming systems in the Po Valley. Journal of Cleaner Production 112, 91–102. https://doi.org/10.1016/j.jclepro.2015.09.062

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

Beketov, M.A., Kefford, B.J., Schäfer, R.B., Liess, M., 2013. Pesticides reduce regional biodiversity of stream invertebrates. Proceedings of the National Academy of Sciences 110, 11039–11043. https://doi.org/10.1073/pnas.1305618110

Bellassen, V., Drut, M., Antonioli, F., Brečić, R., Donati, M., Ferrer-Pérez, H., Gauvrit, L., Hoang, V., Knutsen Steinnes, K., Lilavanichakul, A., Majewski, E., Malak-Rawlikowska, A., Mattas, K., Nguyen, A., Papadopoulos, I., Peerlings, J., Ristic, B., Tomić Maksan, M., Török, Á., Vittersø, G., Diallo, A., 2021. The Carbon and Land Footprint of Certified Food Products. Journal of Agricultural & Food Industrial Organization 19, 113–126. https://doi.org/10.1515/jafio-2019-0037

Bellora, C., Bureau, C., 2016. How green is organic ? Indirect effects of making EU agriculture greener, in: 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.

Bockstaller, C., Sirami, C., Sheeren, D., Keichinger, O., Arnaud, L., Favreau, A., Angevin, F., Laurent, D., Marchand, G., Delaroche, E., Ceschia, E., 2021. Apports de la télédétection au calcul d'indicateurs agri-environnementaux au service de la PAC, des agriculteurs et porteurs d'enjeu. Innovations Agronomiques 83, 43. https://doi.org/10.15454/ns4w-a314

Bodini, A., Chiussi, S., Donati, M., Bellassen, V., Török, Á., Dries, L., Ćorić, D.S., Gauvrit, L., Tsakiridou, E., Majewski, E., Ristic, B., Stojanovic, Z., Roig, J.M.G., Lilavanichakul, A., An, N.Q., Arfini, F., 2021. Water Footprint of Food Quality Schemes. Journal of Agricultural & Food Industrial Organization 19, 145–160. https://doi.org/10.1515/jafio-2019-0045

Cahier des charges de l'appellation d'origine « Comté», 2015.

Calvet, R., 2005. Les pesticides dans le sol: Conséquences agronomiques et environnementales. France Agricole Editions.

Caprio, E., Nervo, B., Isaia, M., Allegro, G., Rolando, A., 2015. Organic versus conventional systems in viticulture: Comparative effects on spiders and carabids in vineyards and adjacent forests. Agricultural Systems 136, 61–69. https://doi.org/10.1016/j.agsy.2015.02.009

Chaudhary, A., Verones, F., De Baan, L., Hellweg, S., 2015. Quantifying Land Use Impacts on Biodiversity: Combining Species–Area Models and Vulnerability Indicators. Environmental Science & Technology 49, 9987–9995. https://doi.org/10.1021/acs.est.5b02507

Chenu, C., Butault, J.-P., 2015. Action 3 : Développer les techniques culturales sans labour pour stocker du carbone dans les sols, in: Agriculture Et Gaz à Effet de Serre: Dix Actions Pour Réduire Les Émissions, Matière à Débattre & Décider. Éditions Quae, Versailles.

Chiaia-Hernandez, A.C., Keller, A., Wächter, D., Steinlin, C., Camenzuli, L., Hollender, J., Krauss, M., 2017. Long-Term Persistence of Pesticides and TPs in Archived Agricultural Soil Samples and Comparison with Pesticide Application. Environmental Science & Technology 51, 10642–10651. https://doi.org/10.1021/acs.est.7b02529

Coinon, M., 2022. Essais sur les externalités spatiales associées à l'agriculture biologique (These de doctorat). Bourgogne Franche-Comté.

Corre, T., Pomeon, T., Regolo, J., 2023. Méthode de sirétisation ODR.

Crenna, E., Sinkko, T., Sala, S., 2019. Biodiversity impacts due to food consumption in Europe. Journal of Cleaner Production 227, 378–391. https://doi.org/10.1016/j.jclepro.2019.04.054

Curran, M., Hellweg, S., Beck, J., 2014. Is there any empirical support for biodiversity offset policy? Ecological Applications 24, 617–632. https://doi.org/10.1890/13-0243.1

De Baan, L., Mutel, C.L., Curran, M., Hellweg, S., Koellner, T., 2013. Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction. Environmental Science & Technology 47, 9281–9290. https://doi.org/10.1021/es400592q

De Marchi, E., Cavaliere, A., Pucillo, F., Banterle, A., Nayga, R.M., 2023. Dynamics of demand-side and supply-side responses to front-of-pack nutrition labels: A narrative review. European Review of Agricultural Economics 50, 201–231. https://doi.org/10.1093/erae/jbac031

De Schryver, A.M., Goedkoop, M.J., Leuven, R.S.E.W., Huijbregts, M.A.J., 2010. Uncertainties in the application of the species area relationship for characterisation factors of land occupation in life cycle assessment. The International Journal of Life Cycle Assessment 15, 682–691. https://doi.org/10.1007/s11367-010-0205-2

Desquilbet, M., Dorin, B., Couvet, D., 2017. Land Sharing vs Land Sparing to Conserve Biodiversity: How Agricultural Markets Make the Difference. Environmental Modeling & Assessment 22, 185–200. https://doi.org/10.1007/s10666-016-9531-5

Dubois, P., Albuquerque, P., Allais, O., Bonnet, C., Bertail, P., Combris, P., Lahlou, S., Rigal, N., Ruffieux, B., Chandon, P., 2021. Effects of front-of-pack labels on the nutritional quality of supermarket food purchases: Evidence from a large-scale randomized controlled trial. Journal of the Academy of Marketing Science 49, 119–138. https://doi.org/10.1007/s11747-020-00723-5

Dubosc, N., Glandières, A., Roubière, M., 2016. Agriculture bio. CA Occitanie; CER Occitanie.

Erm, P., Balmford, A., Holden, M.H., 2023. The biodiversity benefits of marine protected areas in well-regulated fisheries. Biological Conservation 284, 110049. https://doi.org/10.1016/j.biocon.2023.110049

European Space Agency, 2022. Sentinel-2 MSI Level-2A BOA Reflectance. https://doi.org/10.5270/S2_-znk9xsj

France, E., Ventes de produits phytopharmaceutiques par les Distributeurs agréés, B.N. des, 2020. BNV-D Traçabilité - Ventes 2020 par substance, France entière.

Franzluebbers, A.J., Gastal, F., 2019. Building Agricultural Resilience With Conservation Pasture-Crop Rotations, in: Agroecosystem Diversity. Elsevier, pp. 109–121. https://doi.org/10.1016/B978-0-12-811050-8.00007-8

Gabriel, D., Sait, S.M., Kunin, W.E., Benton, T.G., 2013. Food production vs. Biodiversity: Comparing organic and conventional agriculture. Journal of Applied Ecology 50, 355–364. https://doi.org/10.1111/1365-2664.12035

Gallego-Zamorano, J., Huijbregts, M.A.J., Schipper, A.M., 2022. Changes in plant species richness due to land use and nitrogen deposition across the globe. Diversity and Distributions 28, 745–755. https://doi.org/10.1111/ddi.13476

Gong, S., Hodgson, J.A., Tscharntke, T., Liu, Y., Werf, W. van der, Batáry, P., Knops, J.M.H., Zou, Y., 2022. Biodiversity and yield trade-offs for organic farming. Ecology Letters 25, 1699–1710. https://doi.org/10.1111/ele.14017

Guerci, M., Knudsen, M.T., Bava, L., Zucali, M., Schönbach, P., Kristensen, T., 2013. Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy. Journal of Cleaner Production 54, 133–141. https://doi.org/10.1016/j.jclepro.2013.04.035

Holzschuh, A., Steffan-Dewenter, I., Tscharntke, T., 2008. Agricultural landscapes with organic crops support higher pollinator diversity. Oikos 117, 354–361. https://doi.org/10.1111/j.2007.0030-1299.16303.x

Husson, E., Delesse, L., Paget, A., Courbou, R., Bellassen, V., Drut, M., 2019. PDO Comté Cheese in France, in: Arfini, F., Bellassen, V. (Eds.), Sustainability of European Food Quality Schemes: Multi-Performance, Structure, and Governance of PDO, PGI, and Organic Agri-Food Systems. Springer International Publishing, Cham, pp. 405–426. https://doi.org/10.1007/978-3-030-27508-2_21

IGN, 2020. Registre Parcellaire Graphique (RPG).

IGN, ASP, 2020. BD TOPO®.

Indice annuel des prix d'achat des moyens de production agricole (IPAMPA) - Aliments des animaux Insee, 2020.

Ingrédients agricoles - inventaires mobilisés (impacts ACV) : Ecobalyse, 2024.

International Reference Life Cycle Data System (ILCD) Handbook general guide for life cycle assessment: Detailed guidance, 2010. Publications Office, Luxembourg, First edition.

IPBES, 2019. Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Zenodo. https://doi.org/10.5281/ZENODO.3831673

IPCC, 2019. EMISSIONS FROM LIVESTOCK AND MANURE MANAGEMENT, in: 2019 Refinements of 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

IPCC, 2006. EMISSIONS FROM LIVESTOCK AND MANURE MANAGEMENT, in: IPCC Guidelines for National Greenhouse Gas Inventories.

Jayet, P.-A., Petsakos, A., Chakir, R., Lungarska, A., Cara, S.D., Petel, E., Humblot, P., Godard, C., Leclère, D., Cantelaube, P., Bourgeois, C., Clodic, M., Bamière, L., Fradj, N.B., Aghajanzadeh-Darzi, P., Dumollard, G., Isbasoiu, A., Adrian, J., Pilchak, G., Bounaffaa, M., Barberis, D., Assaiante, C., Ollier, M., Henry, L., Florio, A., Chiadmi, I., Gossiaux, E., Ramirez, E., Gérard, M., Reineix, J., Zuravel, O., Baldi, L., Weng, M., 2023. The European agro-economic model AROPAj (report). INRAE-PSAE. https://doi.org/10.17180/nxw3-3537

Jeanneaux, P., Blasquiet-Revol, H., Gillot, M., 2018. La compétitivité hors coût des exploitations agricoles françaises :une analyse des effets des signes de qualité et d'origine. UMR Territoires (AgroParisTech, INRAE, Université Clermont-Auvergne, VetagroSup).

Jiang, B., Pang, J., Li, J., Mi, L., Ru, D., Feng, J., Li, X., Zhao, A., Cai, L., 2024. The effects of organic food on human health: A systematic review and meta-analysis of population-based studies. Nutrition Reviews 82, 1151–1175. https://doi.org/10.1093/nutrit/nuad124

Kim, K.-H., Kabir, E., Jahan, S.A., 2017. Exposure to pesticides and the associated human health effects. Science of The Total Environment 575, 525–535. https://doi.org/10.1016/j.scitotenv.2016.09.009

Knudsen, M.T., Dorca-Preda, T., Djomo, S.N., Peña, N., Padel, S., Smith, L.G., Zollitsch, W., Hörtenhuber, S., Hermansen, J.E., 2019. The importance of including soil carbon changes, ecotoxicity and biodiversity impacts in environmental life cycle assessments of organic and conventional milk in Western Europe. Journal of Cleaner Production 215, 433–443. https://doi.org/10.1016/j.jclepro.2018.12.273

Koch, P., Salou, T., 2020. AGRIBALYSE®: Methodologie, Volet Agriculture – Version 3.0. ADEME, Angers.

Lambotte, M., De Cara, S., Brocas, C., Bellassen, V., 2023. Organic farming offers promising mitigation potential in dairy systems without compromising economic performances. Journal of Environmental Management 334, 117405. https://doi.org/10.1016/j.jenvman.2023.117405

Le Magueresse-Battistoni, B., Vidal, H., Naville, D., 2018. Environmental Pollutants and Metabolic Disorders: The Multi-Exposure Scenario of Life. Frontiers in Endocrinology 9. https://doi.org/10.3389/fendo.2018.00582

Lindner, J.P., Fehrenbach, H., Winter, L., Bloemer, J., Knuepffer, E., 2019. Valuing Biodiversity in Life Cycle Impact Assessment. Sustainability 11, 5628. https://doi.org/10.3390/su11205628

Lindner, J.P., Koch, P., Fehrenbach, H., Buerck, S., 2022. Bringing the Biodiversity Value Increment method to Agribalyse. ADEME.

Lippert, C., Feuerbacher, A., Narjes, M., 2021. Revisiting the economic valuation of agricultural losses due to large-scale changes in pollinator populations. Ecological Economics 180, 106860. https://doi.org/10.1016/j.ecolecon.2020.106860

LOI n° 2009-967 du 3 août 2009 de programmation relative à la mise en œuvre du Grenelle de l'environnement (1), 2009.

LOI n° 2021-1104 du 22 août 2021 portant lutte contre le dérèglement climatique et renforcement de la résilience face à ses effets (1), 2021.

Maxwell, S.L., Fuller, R.A., Brooks, T.M., Watson, J.E.M., 2016. Biodiversity: The ravages of guns, nets and bulldozers. Nature 536, 143–145. https://doi.org/10.1038/536143a

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

Mendiburu, F. de, Yaseen, M., 2020. Agricolae: Statistical Procedures for Agricultural Research.

Mesnage, R., Antoniou, M.N., 2018. Ignoring Adjuvant Toxicity Falsifies the Safety Profile of Commercial Pesticides. Frontiers in Public Health 5. https://doi.org/10.3389/fpubh.2017.00361

Ministère De L'Agriculture (SSP), 2019. Pratiques culturales sur les grandes cultures - 2017. https://doi.org/10.34724/CASD.56.3033.V1

Mouel, C.L., Lattre-Gasquet, M.D., Mora, O., 2018. Land Use and Food Security in 2050: A Narrow Road.

Mueller, C., De Baan, L., Koellner, T., 2014. Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study. The International Journal of Life Cycle Assessment 19, 52–68. https://doi.org/10.1007/s11367-013-0638-5

Overmars, K., Edwards, R., Padella, M., Gerdien Prins, A., Marelli, L., 2015. Estimates of indirect land use change from biofuels based on historical data. European Commission. Joint Research Centre. Institute for Energy; Transport, LU.

Peyraud, J.-L., Cellier, P., Donnars, C., 2012. Les flux d'azote liés aux élevages : Réduire les pertes, rétablir les équilibres.

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. Proceedings of the Royal Society B: Biological Sciences 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

Read, Q.D., Hondula, K.L., Muth, M.K., 2022. Biodiversity effects of food system sustainability actions from farm to fork. Proceedings of the National Academy of Sciences 119, e2113884119. https://doi.org/10.1073/pnas.2113884119

RÈGLEMENT (UE) 2018/848 DU PARLEMENT EUROPÉEN ET DU CONSEIL du 30 mai 2018 relatif à la production biologique et à l'étiquetage des produits biologiques, et abrogeant le règlement (CE) no 834/2007 du Conseil, 2018.

REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL amending Council Regulation (EC) No 1217/2009 as regards conversion of the Farm Accountancy Data Network into a Farm Sustainability Data Network, 2023.

Relyea, R.A., 2009. A cocktail of contaminants: How mixtures of pesticides at low concentrations affect aquatic communities. Oecologia 159, 363–376. https://doi.org/10.1007/s00442-008-1213-9

Ricciardi, V., Mehrabi, Z., Wittman, H., James, D., Ramankutty, N., 2021. Higher yields and more biodiversity on smaller farms. Nature Sustainability 4, 651–657. https://doi.org/10.1038/s41893-021-00699-2

Rose, S., Diaz, D., Carleton, T., Drouet, L., Guivarch, C., Méjean, A., Piontek, F., 2022. Cross-Working Group Box ECONOMIC: Estimating Global Economic Impacts from Climate Change, in: Climate Change 2022 – Impacts, Adaptation and Vulnerability: Working Group II Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change.

Rundlöf, M., Nilsson, H., Smith, H.G., 2008. Interacting effects of farming practice and landscape context on bumble bees. Biological Conservation 141, 417–426. https://doi.org/10.1016/j.biocon.2007.10.011

Sailley, M., Cordier, C., Courtonne, J.-Y., Duflot, B., Cadudal, F., Perrot, C., Brion, A., Baumont, R., 2021. Quantifier et segmenter les flux de matières premières utilisées en France par l'alimentation animale. INRAE Productions Animales 34, 273–292. https://doi.org/10.20870/productions-animales.2021.34.4.5396

Sánchez-Bayo, F., Wyckhuys, K.A.G., 2019. Worldwide decline of the entomofauna: A review of its drivers. Biological Conservation 232, 8–27. https://doi.org/10.1016/j.biocon.2019.01.020

Santana, C., 2014. Save the planet: Eliminate biodiversity. Biology & Philosophy 29, 761–780. https://doi.org/10.1007/s10539-014-9426-2

Sarkar, S., 2002. Defining "Biodiversity"; Assessing Biodiversity. The Monist 85, 131–155. https://doi.org/10.5840/monist20028515

Schöpke, B., Wesche, K., Wulf, M., 2023. Dry grasslands adjacent to organic fields have higher plant diversity − Even far into their interior. Agriculture, Ecosystems & Environment 357, 108672. https://doi.org/10.1016/j.agee.2023.108672

Searchinger, T.D., Wirsenius, S., Beringer, T., Dumas, P., 2018. Assessing the efficiency of changes in land use for mitigating climate change. Nature 564, 249–253. https://doi.org/10.1038/s41586-018-0757-z

Sengel, S., Midler, E., Depeyrot, J.-N., 2022. Performances des exploitations laitières bovines sous indications géographiquesAgreste, la statistique agricole (No. 173). Centre d'études et de prospective.

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

Sirami, C., Gross, N., Baillod, A.B., Bertrand, C., Carrié, R., Hass, A., Henckel, L., Miguet, P., Vuillot, C., Alignier, A., Girard, J., Batáry, P., Clough, Y., Violle, C., Giralt, D., Bota, G., Badenhausser, I., Lefebvre, G., Gauffre, B., Vialatte, A., Calatayud, F., Gil-Tena, A., Tischendorf, L., Mitchell, S., Lindsay, K., Georges, R., Hilaire, S., Recasens, J., Solé-Senan, X.O., Robleño, I., Bosch, J., Barrientos, J.A., Ricarte, A., Marcos-Garcia, M.Á., Miñano, J., Mathevet, R., Gibon, A., Baudry, J., Balent, G., Poulin, B., Burel, F., Tscharntke, T., Bretagnolle, V., Siriwardena, G., Ouin, A., Brotons, L., Martin, J.-L., Fahrig, L., 2019. Increasing crop heterogeneity enhances multitrophic diversity across agricultural regions. Proceedings of the National Academy of Sciences 116, 16442–16447. https://doi.org/10.1073/pnas.1906419116

Smith, O.M., Cohen, A.L., Rieser, C.J., Davis, A.G., Taylor, J.M., Adesanya, A.W., Jones, M.S., Meier, A.R., Reganold, J.P., Orpet, R.J., Northfield, T.D., Crowder, D.W., 2019. Organic Farming Provides Reliable Environmental Benefits but Increases Variability in Crop Yields: A Global Meta-Analysis. Frontiers in Sustainable Food Systems 3. https://doi.org/10.3389/fsufs.2019.00082

Soler, L.-G., Aggeri, F., Dourmad, J.-Y., Hélias, A., Julia, C., Nabec, L., Pellerin, S., Ruffieux, B., Trystram, G., Van Der Werf, H.M.G., 2021. L'Affichage Environnemental des Produits Alimentaires. ADEME.

Team, Q.D., 2022. QGIS Geographic Information System.

Team, R.C., 2023. R: A Language and Environment for Statistical Computing.

Tilman, D., Clark, M., Williams, D.R., Kimmel, K., Polasky, S., Packer, C., 2017. Future threats to biodiversity and pathways to their prevention. Nature 546, 73–81. https://doi.org/10.1038/nature22900

Tran, G., 2002. Tables of composition and nutritional values of feed materials INRA CIRAD AFZ.

Transition écologique, M. de la, ADEME, OFB, 2023. GT Ecologue affichage environnemental n°3.

Tuck, S.L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L.A., Bengtsson, J., 2014. Land-use intensity and the effects of organic farming on biodiversity: A hierarchical meta-analysis. Journal of Applied Ecology 51, 746–755. https://doi.org/10.1111/1365-2664.12219

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

Vallé, C., Le Viol, I., Kerbiriou, C., Bas, Y., Jiguet, F., Princé, K., 2023. Farmland biodiversity benefits from small woody features. Biological Conservation 286, 110262. https://doi.org/10.1016/j.biocon.2023.110262

Velimirov, A., Huber, M., Lauridsen, C., Rembiałkowska, E., Seidel, K., Bügel, S., 2010. Feeding trials in organic food quality and health research. Journal of the Science of Food and Agriculture 90, 175–182. https://doi.org/10.1002/jsfa.3805

Walder, F., Büchi, L., Wagg, C., Colombi, T., Banerjee, S., Hirte, J., Mayer, J., Six, J., Keller, T., Charles, R., Heijden, M.G.A. van der, 2023. Synergism between production and soil health through crop diversification, organic amendments and crop protection in wheat-based systems. Journal of Applied Ecology 60, 2091–2104. https://doi.org/10.1111/1365-2664.14484

Wermeille, A., Gaillet, G., Asselin, A.-C., 2023. Améliorer la prise en compte de la biodiversité à "la parcelle" en analyse de cycle de vie pour Agribalyse. Sayari.

Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L.J., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J.A., Vries, W.D., Sibanda, L.M., Afshin, A., Chaudhary, A., Herrero, M., Agustina, R., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S.E., Reddy, K.S., Narain, S., Nishtar, S., Murray, C.J.L., 2019. Food in the Anthropocene: The EAT–Lancet Commission on healthy diets from sustainable food systems. The Lancet 393, 447–492. https://doi.org/10.1016/S0140-6736(18)31788-4