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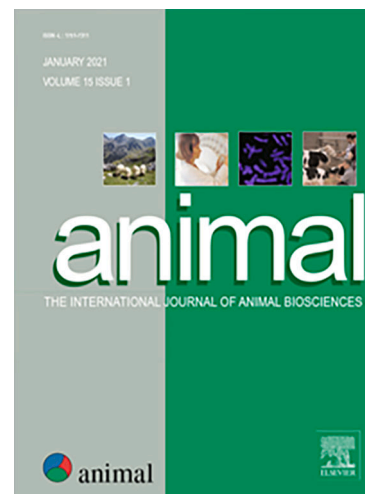
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Environmental trade-offs of meeting nutritional requirements with a lower share of animal protein for adult subpopulations

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Highlights:

- Life Cycle Assessment was applied to observed and optimized French population diet
- Diet with low animal protein share has mixed environmental impacts
- Climate change, acidification, land occupation and energy use decreased
- Water scarcity and land-use impacts on biodiversity increased
- Changing agricultural practices and dietary recommendations remains an objective

Abstract

Decreasing the share of protein contributed by animal-based foods is recommended to move towards more sustainable and healthier diets. This study aimed to assess potential environmental impacts of diets with a lower share of animal protein. The diets were modeled to include the minimum share of animal protein in total protein that met nutrient requirements and did not increase costs. The new diets also minimized the difference in the quantity of food from those of observed (**OBS**) diets. They were modeled for five adult subpopulations (defined by sex and age) using mathematical optimization. The model was created by combining the INCA2 database (to model OBS diets in the French population) and a database of 207 food items to adjust nutritional and price parameters. All modeled diets satisfied nutritional

and cost constraints. A low-animal-protein (**LAP**) diet was identified for each subpopulation by progressively decreasing the share of animal protein by steps of 5% until the recommended quantity of protein and/or consumption constraints were no longer satisfied. Potential environmental impacts of the LAP diets in eight impact categories were calculated using life cycle assessment (**LCA**) and life cycle inventories from Agribalyse® 3.0. A LAP diet for the entire population was calculated as a weighted mean of the subpopulations' LAP diets. The share of animal protein decreased from 70% in the OBS diet to 50% in the LAP diet. Compared to the OBS diet, the LAP diet decreased five environmental impacts: climate change (greenhouse gas emissions), acidification (emissions of acidifying compounds) and land occupation (all by more than 30%), cumulative energy demand (by 23%) and marine eutrophication (by 13%). Conversely, it increased three environmental impacts: freshwater eutrophication and water use (both by ca. 40%) and biodiversity damage potential (potential loss of species associated with land use) (by 66%). These results suggest that decreasing the share of animal protein to 50% is compatible with nutritional requirements, affordability and consumption constraints, but would have mixed effects on the environment.

Keywords: life cycle assessment, diet, biodiversity, water use, climate change

Implications

Decreasing the share of animal protein in the human diet is recommended for human and planetary health. But, does optimizing nutritional requirements by reducing the proportion of animal protein systematically improve the environmental balance of the human diet? Based on French consumption data, this study found that diets optimized to be healthy and affordable and have a lower share of animal protein decreased several life cycle assessment environmental impacts, such as climate change, but increased impacts on biodiversity and water use. Nutritional recommendations and agricultural practices must evolve to achieve the dual objective of human and planetary health.

Introduction

The current food system in high-income countries is a major threat to sustainable development, and dietary changes are needed to keep the Earth system within planetary boundaries (Aiking and de Boer, 2020; Campbell et al., 2017; Lucas et al., 2021; Sala et al., 2020; Willett et al., 2019). The environmental pressure of the global food system is driven mainly by the share of animal products (e.g., meat, fish, eggs, milk, dairy products) in the human diet (Xu et al., 2021). In particular, the livestock industry contributes greatly to greenhouse gas emissions, land competition, water use, eutrophication, acidification (Gerber et al., 2013; Poore and Nemecek, 2018) and impacts on biodiversity (Corson et al., 2022; Steinfeld et al., 2006). In this context, the projected increase in demand for animal protein over the coming decades due to population growth and rising living standards (FAO, 2017a) poses a major challenge for future sustainability. In Western countries, mean protein intake exceeds the minimum quantity recommended by nutritionists (Andreoli et al., 2021) and is provided mainly (55-73%) by animal products (Halkjær et al., 2009; Hoy et al., 2021). Decreasing the consumption of protein, as well as the share of protein from animal products, could therefore have a large impact by decreasing environmental

impacts (Springmann et al., 2018). Thus, shifting the shares of animal and plant products in favor of the latter is a key target of food policies to increase sustainability (Lonnie and Johnstone, 2020; Willett et al., 2019). Nevertheless, animal products provide important micronutrients such as vitamins B12 and D, riboflavin, calcium, iron and zinc (Fehér et al., 2020). In addition, drastically decreasing their intake may decrease diet affordability (Monsivais et al., 2015) and acceptability (Fehér et al., 2020).

Linear optimization of diets is a powerful method for identifying the dietary changes needed to achieve environmentally friendly diets without compromising other dimensions of diet sustainability, such as nutritional adequacy, cultural acceptability (i.e., avoiding large differences from observed diets) and economic affordability (Gazan et al., 2018b). From observed food-consumption data, diet modeling attempts to modify the quantity of food consumed to create a new diet that optimizes (minimizes or maximizes) an objective function while satisfying a set of nutritional and non-nutritional constraints. Previous modeling studies consistently predicted that transition towards a more sustainable diet would require adopting a diet with fewer animal products and more fruits and vegetables, pulses and cereals than the current diet (Gazan et al., 2018b; Wilson et al., 2019). However, the share and type of animal products in the optimized diets and their environmental impacts differed among studies due to differences among the optimization models and their input data (e.g., food consumption data or tables, nutritional recommendations, environmental impacts of foods, prices), due mainly to differences in model settings and data sources (Gazan et al., 2018b; Wilson et al., 2019). For example, in France, a 30% decrease in greenhouse gas emissions was found to be possible while meeting nutritional requirements and differing little from current diets (Perignon et al., 2016). However, another modeling study found that environmental impacts of diets could be decreased by more than 50% in more "disruptive" scenarios, which drastically decreased the share of animal protein (Seconda et al., 2021). Nevertheless, in these studies, decreasing the share of animal protein was a consequence of model parameterization, and none of them specifically examined the environmental impacts of forcing a decrease in the share of animal protein in a nutritionally adequate diet.

Life cycle assessment (**LCA**) has been used several times to assess human diets (Chai et al., 2019; Martin and Brandao, 2017; Munoz et al., 2010). However, doing so challenges LCA methodology, notably the use of relevant functional units (Sonesson et al., 2019) and generation of optimized diets for a large population (Larrea-Gallegos and Vazquez-Rowe, 2020), and the data that LCA requires (e.g., lack of representative databases of food impacts and population consumption habits). These issues were recently discussed to provide recommendations for developing nutritional LCA (McLaren et al., 2021). Which impact categories to use to assess environmental impacts of the food system remains an open question and is often limited to the carbon footprint (Kovacs et al., 2021). Consequently, LCA of agriculture does not consider all environmental impacts or ecosystem services associated with agriculture (van der Werf et al., 2020). This is especially true for impacts on biodiversity, since the loss of biodiversity is a major threat for the sustainability of food systems. Despite recent methodological developments (Crenna et al., 2019; Crenna et al., 2020), applications of nutritional LCA remain scarce, due mainly to the need for suitable life cycle inventories (**LCIs**) (Sanyé-Mengual et al., 2022), in particular differentiation of land-use classes (e.g., annual crops, permanent crops,

organic farms) (McLaren et al., 2021). Impacts on biodiversity must be considered at the diet level to assess the consequences of transitions in dietary habits.

Aware of the importance of protein in human nutrition and its potential environmental impacts via animal production, we performed an initial study using mathematical optimization of diets to explore the minimum share of animal protein that met all food nutrient recommendations (Vieux et al., 2022). We applied this study to the French population as an example of a European high-protein diet and because data on this topic were easily available. We used mathematical optimization to decrease the share of animal protein while meeting nutritional recommendations for the French population, satisfying consumption constraints and not increasing the cost (Vieux et al., 2022). Using this method, an initial study was performed to compare environmental impacts of an optimized low-animal-protein (LAP) diet to those of an observed (OBS) diet, as well as related diets for adult subpopulations in France (Tharrey et al., 2022). Eight LCA environmental impacts were estimated, including land-use impacts on biodiversity.

Since the study of Tharrey et al. (2022), we detected inconsistencies in the data used. We thus corrected and published a complete version of the databases and recalculated the impacts (Le Féon et al., 2022; Le Féon et al., 2023). Because the results changed, so did the conclusions. The present study updates the study of Tharrey et al. (2022) with more complete methods and discussion.

The objective of this study was to explore potential trade-offs between food practices considered environmentally friendly (i.e., decreasing the share of animal protein in the diet) and multiple environmental impacts for a plausible national-scale diet. This study is addressed in particular to scientists and decision-makers working for a more sustainable food supply.

Material and methods

The method consisted of three main steps: (1) defining the diets to assess as combinations of food ingredients based on adult subpopulations, (2) adapting Agribalyse[®] LCIs and extracting data per unit of product and (3) assessing eight environmental impacts associated with the selected diets. The method had several substeps (Fig. 1), described as follows.

Population and nutritional data

Data on dietary intakes used to design the optimized diets came from the second French cross-sectional Individual and National Study on Food Consumption (INCA2, 2006-2007) performed by the French Agency for Food, Environmental, and Occupational Health and Safety (AFSSA, 2009) (**OBS** diet). Only data collected from adults aged ≥ 18 years old ($n = 2624$) were used in the present study. Subpopulations of interest were defined by sex and age (< 65 years and ≥ 65 years) because of their different nutritional needs (ANSES, 2016). The subpopulation of women was subdivided as a function of iron requirements (ANSES, 2019), considering 50 years old as the age of menopause, which resulted in a total of five subpopulations: women < 50 years old, women 50-64 years old, women ≥ 65 years old, men < 65 years old and men ≥ 65 years. Like Vieux et al. (2022), we calculated

the nutritional composition of the foods consumed by participants in the INCA2 study (expressed per 100 g of edible portion) from the French reference database of the nutritional composition of foods commonly consumed in France (CIQUAL). The phytate content of foods came from the International Framework of Food Data Systems (INFOODS, 2015). For each subpopulation, the recommended total protein intake was estimated from the mean body weight of the French population. The mean prices of foods were calculated using data from the 2006 Kantar Worldpanel survey of food purchases in France (Gazan et al., 2018a; Kantar Worldpanel, 2015).

The database of food nutrition and cost was aggregated into a list of 212 frequently consumed food items built using a matching method for compiling food metrics (Gazan et al., 2018a). For each subpopulation, the mean daily food intake was estimated by aggregating the foods consumed into the list of 212 food items (Gazan et al., 2018a). Alcohol items were removed from the analysis, which yielded a final database of 207 food items. The food items were categorized into three levels: food groups ($n = 9$), food subgroups ($n = 26$) and food families ($n = 54$).

The minimum share of animal protein that met all food nutrient recommendations was estimated using mathematical optimization to decrease the share of animal protein while meeting nutritional recommendations, satisfying consumption constraints and not increasing the cost (**LAP** diet) (Vieux et al., 2022). See Vieux et al. (2022) for details of the modeling method and the parameters considered.

Life cycle assessment and environmental impact categories

Environmental impacts of foods were estimated using LCA, a multi-criteria and multi-stage environmental assessment framework defined by ISO standard 14044 (ISO, 2006). LCA quantifies all relevant emissions and resources consumed throughout the life cycle of a product or service (e.g., production, distribution, use and end-of-life phases) (European Commission et al., 2010).

Goal and scope

The objective of the study was to compare potential environmental impacts of the LAP diet to those of the OBS diet. The system boundaries were those of the food items in the Agribalyse database: the value chain from “cradle to plate”. The functional unit was “Provide a daily diet to consumers that meets recommended nutrient intake and remains affordable”.

Life cycle inventory

LCIs of input and output flows came from Agribalyse 3.0, the LCI database of food items produced and consumed in France (Koch and Salou, 2020; Asselin-Balençon et al., 2020). For 2500 food items from the CIQUAL database, Agribalyse provided “cradle-to-plate” LCIs that included agricultural production, processing, transport, packaging, distribution, retailing, consumer use, end of life and waste disposal (Asselin-Balençon et al., 2020). All transportation was included, except that from retail to consumer. All waste (ingredients and food) was included, except that at consumers’ places of consumption. The sources of the main ingredients were defined in detail by combining data from technical institutes and expert knowledge (Asselin-Balençon et al., 2020). For other ingredients, five-year mean (2009-2013)

data from FAOSTAT (2019) were used. For some ingredients (i.e., tomatoes, strawberries, chicken and beef), a distinction was made between direct consumption and processing/catering. For example, all chicken and beef intended for direct consumption came from France. The Agribalyse LCIs used proxies when the country of origin was unknown or the corresponding data were missing. Exports and stocks were excluded (Asselin-Balençon et al., 2020). The diets included 686 foods and were averaged to 212 food items using relations from the French Food Agency (Gazan et al., 2018a) to connect the food-composition and environmental-impact databases. See Le Féon et al. (2023) for details.

Life cycle impact assessment

Based on the Agribalyse LCIs, seven midpoint impact categories were estimated that covered the main potential environmental impacts of food production: climate change (**CC**), marine eutrophication (**ME**), freshwater eutrophication (**FE**), acidification (**AC**), water scarcity (**WS**), land occupation (**LO**) and cumulative energy demand (**CED**) (Table 1). To estimate impacts of food production on biodiversity, an eighth impact category – biodiversity damage potential (**BDP**) – was used to express the potential loss of species associated with land use (Table 1). Land-use impacts on biodiversity were assessed using the approach of Knudsen et al. (2017), who calculated BDP as the product of a characterization factor, area and duration of each land-use type:

$$\text{BDP} = A * t * \text{CF}$$

where A is the area of the land use type, t the time considered

and CF the characterisation factor related to the land use type

The characterization factor expresses the plant species richness associated with a land-use type for European agriculture compared to that of a reference situation (i.e., temperate-broadleaf and mixed-forest biome) and estimates the potential fraction of species lost. It usually ranges from 0 (no loss) to 1 (total loss) but can become negative when the land-use type potentially supports more plant species than the reference. The characterization factors of Knudsen et al. (2017) were supplemented with previously published characterization factors for tropical annual crops (Mueller et al., 2014) and vineyards and orchards (Koellner and Scholz, 2008). The characterization factors used are available in Supplementary Table S1. To apply BDP, some land occupation flows of Agribalyse LCIs needed to be modified, which also resulted in adapting the method used to estimate LO, as documented in a public dataset (<https://doi.org/10.57745/HZSRHZ>) associated with a data paper (Le Féon et al., 2023). Impacts at the food level were combined to obtain impacts at the diet level using attributional LCA, which was calculated using SimaPro 8.4 (PRé Sustainability, Amersfoort, the Netherlands).

Interpretation

We compared impacts of the two diets and contributions of the food categories in the diets to these impacts, and examined the key mechanisms that influenced differences between the diets.

We used mathematical optimization of diets (Gazan et al., 2018b) to model nutritionally adequate diets that satisfied consumer constraints (quantities and costs) and contained less protein than OBS diets. The OPTMODEL procedure of SAS/OR software (SAS institute, Cary, NC, USA) was used to run linear programming models. For each subpopulation, new diets were modeled by modifying the quantities of each of the 207 food items in the OBS diets. All optimized diets were constrained to have the same energy intake as OBS diets, not exceed the cost of OBS diets and meet a set of 31 nutritional recommendations (i.e., energy, carbohydrates, fiber, water, lipids, saturated and essential fatty acids, sugar, sodium, 11 vitamins and 8 minerals). Nutrient reference values (**NRVs**) were taken from the French nutritional recommendations available at the beginning of the study (ANSES, 2016, 2019). See Table S2 for all constraints and corresponding thresholds and Vieux et al. (2022) for details of the nutritional constraints. Because the NRVs for EPA+DHA (500 mg/d) and vitamin D (15 µg/d) could not be met in either the OBS or optimized diets, their minimum constraints were lowered to 250 mg/d (European Union reference intake) (European Food Safety Authority, 2017) and 5 µg/d (considered adequate by some national and international food and public health agencies (ANSES, 2021)), respectively, to allow the models to find a solution. For calcium, the NRV was weighted by the percentages of individuals older and younger than 25 years old in the subpopulation. Sodium and fortified products (i.e. supplemented with essential nutrients) could not exceed those in the OBS diets. In accordance with French NRVs, the recommended quantity of zinc was set as a function of the quantity of phytate (ANSES, 2016). Drinking water could not exceed its NRV in order to keep the optimized diets from including too much of it. In accordance with the French Agency for Food, Environmental and Occupational Health & Safety, fish intake could not exceed 200 g/week in order to decrease health risks due to contaminant exposure (ANSES, 2012).

To avoid too much difference from the OBS diet, two non-nutritional constraints were added: (1) the optimized quantity of each food item could not exceed the 95th percentile among consumers (to keep the maximum portion of a food item realistic) and (2) the quantity of each food category (i.e., group, subgroup and family) was constrained between the 5th and 95th percentiles of that observed in the entire population. To consider diet affordability (and indirectly, acceptability), the cost of the diet could not exceed that of the OBS diet of the corresponding subpopulation. The objective function was designed to minimize the difference from the OBS diet (D), which was calculated as the sum of the absolute differences between the quantity of each of the 207 food items in the OBS diets and the quantity of that food item in the optimized diet:

$$\text{Min } D = \sum_{i=1}^{207} |Q_i^{\text{opt}} - Q_i^{\text{obs}}|$$

Where Q_i^{opt} and Q_i^{obs} are the optimized and observed quantities of food item i , respectively.

For each subpopulation, the first optimized diet had no constraints on the share of animal protein. Then, the share of animal protein was progressively decreased by steps of 5% until no solution was found. The optimized diet with the lowest share of animal protein selected for a subpopulation – the LAP diet – was determined according to three criteria: (1) adequate total protein content (i.e., must not fall below the minimum quantity recommended), (2) little difference in food weight from that of the OBS diet (i.e. the objective function value (total deviation from the OBS diet) could not exceed the quantity of food consumed in the OBS diet (in kg)) and (3) little loss of dietary diversity in the optimized diet (i.e., the number of food items could not decrease by more than 15% for each 5% decrease in the share of animal protein). See Vieux et al. (2022) for details of the criteria used to select the LAP diet by subpopulation. The compositions (in g/d) of the OBS and LAP diets were compared by food groups, subgroups and families. The composition and impacts of a mean LAP diet of the entire population were then calculated by averaging data of the LAP diets of the subpopulations, weighted by the prevalence of the corresponding subpopulation in the French population: women < 50 years old (32.1%), women 50-64 years old (11.5%), women ≥ 65 years old (7.8%), men < 65 years old (40.4%) and men ≥ 65 years (8.2%).

Results

Protein content of the observed and low animal protein diets for the five subpopulations and the entire population

All LAP diets met the recommended total protein intake. Depending on the subpopulation, food weight differed from that of OBS diets by 1.12 kg (for men ≥ 65 years old) to 1.94 kg (for women < 65 years old), and the numbers of food items decreased by 30% (for men ≥ 65 years old) to 60% (for women ≥ 65 years old) (Table 2). The share of animal protein in the LAP diets varied among the subpopulations of men (from 45% (< 65 years old) to 60% (≥ 65 years old)) and women (from 50% (50-64 years old) to 55% (< 50 and ≥ 65 years old)) (Table 2). Total protein content (in g) was similar between the OBS and LAP diets for women < 50 and ≥ 65 years old, but decreased in LAP diets for the other subpopulations. For the entire population, the mean share of animal protein decreased from 70% in the OBS diet to 50% in the LAP diet.

Difference in nutrient and food-group contents between observed and low animal protein diets for the entire population

By design, the LAP diet of the entire population met all NRVs, but its OBS diet did not meet NRVs for fiber, ALA, EPA+DHA, pantothenic acid, folates, vitamins C and D, calcium, iron, iodine or magnesium (Table S3). Both diets cost 5.80 €/d. The food and energy contents of the OBS and LAP diets of the entire population varied among the food groups (Table 3). Compared to the OBS diet, the LAP diet contained more fruits and vegetables (+103%); pulses, potatoes and unrefined grain products (+142%); milk (+222%) and eggs (+96%) and less meat (-66%); cheese (-97%); mixed dishes and sandwiches (-47%); sweet products (-34%) and sugary drinks (-40%). Among food families, poultry, lean fish, nuts, table sugar and spices disappeared from the LAP diet, and substitutions were predicted within some subgroups (Table S4). For example, lean fish was replaced by fatty fish (23.0 g/d; +184%), nuts were replaced by dried fruits (4.1 g/d; +486%), and red meat (cattle,

sheep or goat meat) decreased to 8.7 g/d (-85%), while offal increased to 15.7 g/d (+481%).

Nutrient and food-group contents and costs of OBS and selected LAP diets varied among the subpopulations (Tables S3 and S4). The nutrients that increased the most in the LAP diets were ALA, DHA+EPA, fiber, folate, copper and vitamins A, B12 and D for all subpopulations, and iron for women only. For all subpopulations, the LAP diet contained more fruits and vegetables, dairy products and starchy foods (except the latter in men \geq 65 years old) and less of the other food groups.

Environmental impacts of observed and low animal protein diets of the entire population

The environmental impacts of individual food items are available in the dataset of Le Féon et al. (2022), described by Le Féon et al. (2023). The impacts of the OBS diet were 5.3 kg CO₂ eq/d for climate change (CC), 0.07 mol H⁺ eq/d for acidification (AC), 0.9 g P eq/d for freshwater eutrophication (FE), 20.6 g N eq/d for marine eutrophication (ME), 6.1 m³/d for water scarcity (WS), 5.9 m²/y/d for land occupation (LO), 116.5 MJ/d for cumulative energy demand (CED) and 0.7 PDF per m²/y/d for biodiversity damage potential (BDP) (Fig. 2). The LAP diet decreased five impacts (AC (-39.5%), LO (-34.9%), CC (-29.8%), CED (-23.5%) and ME (-12.8%)) and increased three impacts (BDP, WS and FE, by 66.1%, 41.2% and 36.1%, respectively). For all impact categories except BDP and AC, the food groups “Meat/Fish/Eggs” and “Fruits and vegetables” together contributed the most to impacts of the LAP diet (from 43.6% to CC to 74.9% to WS). The main contributing food groups were “Meat/Fish/Eggs”, “Water and drinks” and “Dairy products” for AC and “Fruits and vegetables” and “Grains and other starchy products” for BDP.

When examining the contribution of food families of the “Meat/Fish/Eggs” and “Fruits and vegetables” groups to impacts of the diets of the entire population, the LAP diet’s decrease in AC, LO, CC, CED and ME was driven mainly by the strong decrease in red meat, whose contribution decreased by nearly 66% (Fig. 3). The increased contribution of fresh fruits and fatty fish (exclusively salmon from aquaculture, data not shown) explained most of the LAP diet’s higher WS and FE, respectively. The large decrease in permanent grassland area due to consuming less red meat and increase in arable land needed to meet the higher demand for vegetables increased BDP greatly.

Differences between the OBS and LAP diets were relatively similar for each subpopulation. Men < 65 years old had the OBS diet with the highest impacts (except for BDP) and the largest decrease in impacts between the OBS and LAP diets due to the largest decrease in the share of animal protein (Figure S1). FE and WS of the LAP diet increased more for women than for men, while ME increased slightly for adults > 65 years old but decreased for the other subpopulations.

Discussion

Influence of the database used on general trends in environmental impacts

Based on mathematical optimization, we predicted that a strong decrease in the share of animal protein in the diet of the adult population in France from 70% to 50%,

while meeting nutritional requirements, satisfying consumption constraints and not increasing cost, had mixed effects on the environment. The optimized LAP diet decreased five environmental impacts: climate change (CC), acidification (AC) and land occupation (LO) by more than 30%, cumulative energy demand (CED) by 23% (by 6.5% in Tharrey et al. (2022)) and marine eutrophication (ME) by 12%. However, it increased freshwater eutrophication (FE), water scarcity (WS) and biodiversity damage potential (BDP) by 36%, 41% and 66% (by 71.3% in Tharrey et al. (2022)), respectively. Using Agribalyse 3.0, which is the largest LCI database of agri-food products consumed in France, allowed us to represent well a wide range of food products items in France. Agribalyse is based on average agricultural practices, however, which may have induced some limitations, since averaging practices does not reflect local production practices or contexts. Nevertheless, Agribalyse reflects specific characteristics of France (e.g., a large share of beef coming from dairy cattle). Agribalyse has inherent limitations due to methodological choices made during its development (e.g., system boundaries, data collected, assessment models), which influenced the estimated input and output flows in its LCIs. In particular, several flows were not considered (e.g., particulate emissions from on-farm activities, carbon sequestration in the soil without land-use change) (Koch and Salou, 2020), but they should be considered in future versions of Agribalyse.

The key influence of red meat consumption

We predicted that the LAP diet's decrease in red meat strongly decreased its impacts, which agrees with previous diet-optimization studies in which reducing red meat in the diet strongly decreased CC, CED and LO (Kesse-Guyot et al., 2021; Seconda et al., 2021), AC (Perignon et al., 2016) and excessive flows of nitrogen and phosphorus to soil and water that cause AC and eutrophication (Barré et al., 2018; Gephart et al., 2016). Indeed, beef cattle use more land and emit more greenhouse gases than other livestock, such as dairy cattle, poultry and pigs (Poore and Nemecek, 2018) (e.g., ca. 60% for red meat in the present study, data not shown). Consequently, these results also agree with studies that compared meat-based meals to vegetable-based meals (Takacs et al., 2022).

Water consumption: a point of concern

A review of diets in high-income countries found that studies that modeled these diets tended to agree that decreasing the share of animal-based foods and increasing the share of fruits and vegetables to follow national dietary recommendations would have both health and environmental benefits (Steenon and Buttriss, 2021). Nevertheless, by considering only a few environmental impacts, they did not provide an overview of impacts of diets on the environment or the risks of potential trade-offs between these impacts. In the present study, although several impacts of the LAP diet decreased, FE and WS increased by more than 30% due to higher contents of fatty fish and fruits, respectively. The literature on sustainable diets indicates that increasing consumption of certain vegetables, fruits or nuts in human diets often uses more water than the quantity of water saved by decreasing consumption of animal products (Springmann et al., 2018; Willett et al., 2019). Results of the present study are consistent with this increase in "blue" water use. Since agricultural irrigation uses 70% of the world's freshwater (FAO, 2017b), achieving a sustainable food system within planetary water boundaries requires

increasing water-use efficiency by decreasing food waste and improving agricultural water management (Jägermeyr et al., 2015; Willett et al., 2019). The increase in FE was due mainly to emitting more phosphorus to water from crop fertilization and direct emissions of aquaculture fish waste to water. In Agribalyse, the impacts of fertilizing crops with livestock manure are associated with the crops, not the livestock (Koch and Salou, 2020).

Negative impacts on biodiversity

An unexpected result of this study was that the LAP diet increased impacts of land use on biodiversity greatly (+66%) because it contained more vegetables and less red meat. Biodiversity is essential for human well-being and ecosystem functioning, and the expansion of agricultural land and intensive farming strongly influence it (Steinfeld et al., 2006; UNEP et al., 2015). However, considering biodiversity loss in LCA remains challenging due to the complexity of assessing biodiversity (Curran et al., 2011; Winter et al., 2017). Many studies have attempted to estimate impacts on biodiversity in LCA using indicators such as species richness, abundance or functional diversity (Curran et al., 2016). In the present study, we used the approach of Knudsen et al. (2017), which quantifies the potentially disappeared fraction of species of a given land use compared to a reference ('temperate broadleaf and mixed forest' for Europe). In particular, it differentiates the main crop and grassland categories. We supplemented their approach with characterization factors for imported crops, especially from tropical areas. As additional limitations, the literature discusses focusing on plant species alone and not considering advantages of mixed landscapes (Kok et al., 2020). This approach is based on averaging observed performances and, unlike other methods (Lindner et al., 2019), considers only land use, not direct relations between agricultural practices and biodiversity. The characterization factors of Knudsen et al. (2017) cannot represent variability within a land-use type, although the intensity of production seemed to influence those for , as stated in their article: "However, for conventional agriculture the CFs for 'Arable' and 'Pastures' seems to be influenced significantly by the intensity of the production. If such information is available, this could further be taken into account". Although it does not consider variability among systems, it has been considered sufficient for assessing land-use impacts of crop and livestock systems on biodiversity, such as when comparing organic and conventional agricultural practices (Nitschelm et al., 2021). The BDP includes only land-use impacts on biodiversity, but agriculture, especially livestock production, exerts other pressures on biodiversity worldwide, especially through climate change, pollution and deforestation (Steinfeld et al., 2006). Applying LCA to agriculture still has limitations, especially for the most extensive or organic practices (van der Werf et al., 2020). Since the present study focused on France, the decrease in red meat consumption decreased grassland more than it may have in other countries, which helped explain the predicted impacts on biodiversity. A decrease in red meat consumption, however, is not the general trend at the global scale. The decrease in meat consumption decreased LO greatly. However, since permanent grassland can host much more biodiversity than any other agricultural land use (Alkemade et al., 2009), the increase in LO translated into an equivalent increase in BDP. Although livestock production is a leading driver of the current biodiversity crisis (Steinfeld et al., 2006), livestock farming systems are complex and generate multiple trade-offs or synergies with other ecosystem services (FAO, 2020). In particular, extensive livestock grazing systems, which represent 34%

of livestock and 31% of the utilized agricultural area in Europe (Dumont et al., 2019), have high nature value and strongly influence biodiversity in agricultural environments through grassland management practices (e.g., grazing frequency, mowing dates, fertilization) (FAO, 2020; Gaujour et al., 2012; Sabatier et al., 2015).

Methodological choices in Life Cycle Assessment

We selected the most relevant impact categories for agri-food systems; thus, some indicators of the Environmental Footprint (EF) method recommended by the European Commission were excluded from this study (e.g., ionizing radiation), while indicators from other methods were added (i.e., BDP). The EF method can also be aggregated into a single score by weighting normalized values of the midpoint indicators. Due to uncertainty in EF indicators and their weights, we did not calculate a single score, but it could be included in future studies. Other emerging indicators and causal chains should also be considered (e.g., impacts of fisheries on aquatic biodiversity). Since this study used attributional LCA, no future changes in practices or proportion between production types (e.g. organic, conventional) were considered. Thus, when the quantity of a food item in the LAP diet changed, the LCA was based on average French markets, without considering that a change in demand could change agricultural practices (e.g., decreasing meat consumption would decrease grassland area). Two main advances are needed to develop studies in this direction: (1) representing more agricultural systems in databases and (2) developing consequential LCI to make prospective studies of these systems more robust. Characterization methods should also include more specific characterization factors (e.g., the BDP method differentiated organic vs. conventional fruits but not the wide range of existing systems). LCA results are partly driven by the functional unit(s) chosen. In this study, we used food items from Agribalyse, which uses 1 kg of product as the functional unit. This functional unit has limitations, since it tends, for example, to favor systems that produce high yields without considering potential benefits (e.g. ecosystem services) besides the main product. In the belief that agri-food systems need to be transformed even more, other functional units should be investigated (e.g., ha), to represent the multifunctionality of agri-food systems (Van der Werf et al., 2020). Finally, this study was based on food items from Agribalyse, whose LCIs technically cannot be disaggregated by life cycle stage. Thus, potential changes in the contribution of specific stages due to changes in diets could not be assessed. Doing so would be interesting in future studies, however, since some of the impact could be transferred from one stage to another.

Issues when modeling environmental impacts and nutrition together

Since we aimed to predict environmental impacts of diets modeled to contain the minimum share of animal protein that met all food nutrient recommendations without increasing the cost, we decided not to use environmental impact indicators as constraints in the model. Several modeling studies have tried to identify more sustainable diets by predicting environmental impacts of following existing national dietary guidelines for food categories, deriving theoretical diets that meet nutritional recommendations and satisfy environmental constraints, or comparing environmental and health benefits of specific plant-based diets (e.g. vegetarian, vegan) (Steenson and Buttriss, 2021). One study estimated that a diet composed of 12% animal protein, especially from milk, would use agricultural land in the Netherlands most

efficiently by optimizing the use of co-products from human food and forage from grasslands (Van Kernebeek et al., 2016), but it did not consider nutrition constraints. At the global scale, it has been suggested that a strong decrease, or even elimination, of meat in diets accompanied by an overall decrease in animal protein could sustain human health and protect the planet (Springmann et al., 2018; Willett et al., 2019). However, the optimized diets did not meet requirements for certain nutrients (i.e., calcium, riboflavin and vitamin B12) and did not consider animal-specific micronutrients (e.g., vitamin D, iodine) (Springmann et al., 2018). Furthermore, such a dietary shift is unlikely to be affordable for much of the world's low-income population (Hirvonen et al., 2020). Results of the present study suggest that such a large decrease in the consumption of animal-based food would lead to nutritional deficiencies, the need for a nutrient-supplementation policy and/or the need for extreme dietary changes.

Toward more complete consideration of subpopulations

The models used in this study were unable to decrease the share of animal protein below ca. 50% without compromising nutritional requirements or increasing cost (except for a share of 45% for men < 65 years old). These results agree with those of de Gavelle et al. (2017), who found 50% as the minimum share at which requirements for protein and indispensable amino acids of the French population could be met. A larger decrease in the share of animal protein from 60% down to 20% was identified in another French study that used diet optimization to model nutritionally adequate diets with lower environmental impacts and the same cost, based on data from the Nutrinet-Santé cohort (Seconda et al., 2021). In that study, however, animal-specific nutrients (e.g., vitamin D, long-chain omega-3 fatty acids) were not constrained, and the minimum protein recommendation was set at 0.66 g/kg body weight regardless of age category (Seconda et al., 2021), which may have given its model the ability to achieve such a low share. Modeling studies need to consider subpopulations to tailor healthier and more sustainable diets to these subpopulations' dietary requirements and habits (Steenon and Buttriss, 2021). One innovation of the present study was to estimate a weighted mean diet for the entire population that considered specific requirements of subpopulations, including women of reproductive age and the elderly. While men < 50 years old could decrease their intake of animal protein greatly, we predicted that older populations could do so only moderately due to their higher protein requirements (ANSES, 2019).

Study limitations

This study had some limitations; for example, the dietary data collected in 2006-2007 may no longer have represented current food consumption. However, two representative surveys of food consumption in France performed in 2006 and 2015 (ESEN, 2017) identified no major differences. Likewise, food prices and their structure have changed over the past 15 years, and not considering these changes may have influenced the results. However, when the cost constraint was removed, expensive foods (e.g., oysters, foie gras) were introduced into the optimized diets (data not shown), which would not have been acceptable in the real world, either for affordability or availability. Effects of nutrient bioavailability and relations among co-products (e.g., milk and beef, meat and offal) on dietary changes also need to be considered. However, using the same dietary data, a previous study also achieved a

50% share of animal protein when modeling sustainable diets (i.e. meeting nutrient requirements, satisfying consumption and cost constraints and decreasing greenhouse gas emissions) considering nutrient bioavailability and co-product relations (Barré et al., 2018). Despite consumption constraints in the present study, some food families (e.g., poultry, lean fish) were eliminated from the LAP diet, making it unlikely that the general population would follow it. Similarly, milk was increased more than threefold, while cheese was nearly eliminated (since it contains more salt and fatty acids and is more expensive), which does not follow current consumption of dairy products in France. Finally, due to the lack of sufficiently robust data, we were unable to quantify uncertainty in the LCA models.

Conclusions

While shifting toward a more plant-based diet is promoted, especially in Western countries, the optimal share of animal protein compatible with a sustainable diet has yet to be determined. The study found that a 50% share of animal protein compatible with nutritional requirements and affordability would have mixed effects on the environment compared to the observed French diet. The trade-offs among several environmental impacts highlight the importance of considering resource (especially water) and energy consumption, nutrient and pollutant emissions and their impacts on biodiversity when designing sustainable diets. Shifting the shares of plant and animal products in diets may require transforming agricultural practices and food systems greatly to address the major concerns of climate change, biodiversity preservation and water consumption. These objectives cannot be reached without considering the dietary habits of populations and the regions in which they live, including the availability of their resources. Assessment methods will need to change to better consider cause-effect relations between agricultural and food practices and their environmental, increased BDP greatly and economic impacts. In particular, impacts on biodiversity need to be studied in depth to be able to move from recommendations to practical solutions. Future modeling studies should add environmental constraints to nutritional constraints to estimate the multiple effects of diet changes on environmental impacts. Doing so would contribute knowledge needed to find trade-offs between a change in diets acceptable to the general population and environmental impacts of these diets, especially in the face of the challenges of climate change, biodiversity preservation and water scarcity, as well as to help maintain economic vitality in regions that differ greatly in agronomic potential.

Ethics approval

Not applicable

Data and model availability statement

Data are deposited in an official repository: Recherche.data.gouv.fr, named Datasets on the nutritional and environmental (including biodiversity) characteristics of food products consumed in France. The data are fully and freely available at this address: <https://doi.org/10.57745/HZSRHZ>

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Declaration of interest

The authors declare the following financial interests/personal relationships, which may be considered as potential competing interests: As a sponsor, INTERBEV was provided the first view of the study's results but was not involved in its analysis or conclusions.

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Table 1. Environmental impact categories and models used in Life Cycle Assessment applied in the study

Impact category	Impact assessment model	Unit	Description	Source
Climate change (CC)	Global Warming Potential over a 100-year horizon	kg CO ₂ eq	Indicator of potential global warming due to emissions of greenhouse gases to the air, due mainly to combustion of fossil fuels (e.g., coal, oil, natural gas).	EF 3.0 method (European Commission, 2019)
Acidification (AC)	Accumulated exceedance model	mol H ⁺ eq	Indicator of the potential acidification of soils and water due to the release of gases (e.g., nitrogen oxides, sulfur oxides), which contribute to a decline of coniferous forests and increase in fish mortality. Its main causes are combustion processes in electricity production, heating and transport.	EF 3.0 method (European Commission, 2019)
Freshwater eutrophication (FE)	EUTREND model	g P eq	Indicator of the excessive enrichment of freshwater ecosystems with nutrients due to the emission of nitrogen or phosphorus compounds. Eutrophication promotes the growth of green algae, which can leave water without enough oxygen for fish to survive. Its main causes are sewage treatment plants for urban and industrial waste and leaching from agriculture land.	EF 3.0 method (European Commission, 2019)
Marine eutrophication (ME)	EUTREND model	g N eq	Indicator of the enrichment of marine ecosystems with excessive nutrients due to the emission of nitrogen compounds. Its main causes are agricultural use of fertilizers and combustion processes.	EF 3.0 method (European Commission, 2019)

Water scarcity (WS)	AWARE	m ³ water use related to local WS	Indicator of the relative quantity of water used, based on regionalized water-scarcity factors.	Boulay et al. (2018)
Land occupation (LO)	CML-IA	m ² y	The area of land involved in production processes for a specific duration	Guinée et al. (2022)
Cumulative energy demand (CED)	CED	MJ	Indicator of all forms of energy consumed throughout the life cycle of a product or service, including non-renewable resources (e.g., fossil fuels, uranium) and renewable resources (e.g., wood, biomass from agriculture, wind and solar energy, geothermal energy, hydropower)	VDI-Richtlinien (1997)
Biodiversity damage potential (BDP)	BDP	PDF ¹ per m ² y	Indicator of the total loss of biodiversity in agricultural systems due to land use. It is estimated as the difference in plant species richness between a target land-use type and a reference biome (i.e., temperate broadleaf and mixed forest)	Knudsen et al. (2017); Koellner and Scholz (2008); Mueller et al. (2014)

¹ PDF: potentially disappeared fraction

Table 2. Protein content of the observed (OBS) and low-animal-protein (LAP) diets¹ for the five subpopulations and entire population of France

Journal Pre-proofs

Subpopulation	Recommended total protein intake (g/d) ²	Total protein (g)		Share of animal protein (%)		Food weight of the diet (kg/d)		Sum of absolute differences from the OBS diet ³	Number of food items in the diet		Difference from the OBS diet
		OBS	LAP	OBS	LAP	OBS	LAP		OBS	LAP	
Women < 50 years old	52.0	69.5	69.8	71.0	55.0	2.35	2.38	1.94	207	82	-125
Women 50-65 years old	56.2	71.5	66.4	71.6	50.0	2.54	2.37	1.91	207	82	-125
Women ≥ 65 years old	65.5	68.2	68.4	68.7	55.0	2.37	2.37	1.75	205	80	-125
Men < 65 years old	64.5	93.9	67.1	70.8	45.0	2.53	2.98	1.69	207	117	-90
Men ≥ 65 years old	78.8	88.0	79.3	67.8	60.0	2.44	2.96	1.12	207	146	-61
Entire population ⁴	60.8	81.0	69.0	70.5	50.8	-	-	-	-	-	-

¹ Satisfying all nutritional and non-nutritional constraints, followed by a 5% stepwise decrease in the share of animal protein until at least one of the three criteria used to select one optimized diet per subpopulation (i.e., total protein intake greater than that recommended, total difference from the OBS diet < total observed quantity of foods, and < 15% decrease in the number of modeled foods per 5% decrease) was not satisfied.

² Estimated from the mean BW of each subpopulation

³ Sum of the absolute differences between the quantity of each of the 207 food items in the OBS diet and the quantity of each in the optimized diet

⁴ Mean weighted by the percentage of the corresponding subpopulation in the entire population

Table 3. Food quantity and energy content in observed (OBS) and low-animal-protein (LAP) diets for the entire population of France

Food group	Quantity (g/d)		Energy content (kcal/d)	
	OBS	LAP	OBS	LAP
Fruits and vegetables	338.7	686.3	150.9	328.8
Fruits (without juice)	141.1	332.1	81.1	195.9
Nuts and seeds	2.3	4.1	10.1	10.9
Vegetables	195.3	350.1	59.7	122
Grains and other starchy products	231.0	300.9	478.2	604.7
Breakfast cereals	4.5	3.3	18.8	13.9
Pulses, potatoes and unrefined grain products	71.7	173.2	98.3	269.4
Refined grains products	154.7	124.5	361.1	321.4
Dairy products	192.4	348.2	195.8	155.9
Cheese	30.1	0.9	97.7	3.4
Milk	86.9	279.7	42	103.4
Yogurt	75.3	67.6	56.1	49.1
Meat/Fish/Eggs	156.5	94.4	304.1	208.8
Eggs	14.2	27.8	25.3	42.6

Fish	30.2	28.6	44.8	52.3
Meat	112.1	38.0	234	113.9
Meat and dairy imitates	3.7	3.6	2.4	2.3
Meat imitates	0.2	0.1	0.3	0.2
Dairy imitates	3.5	3.5	2.1	2.1
Mixed dishes and sandwiches	123.6	65.4	210.2	124.4
Animal-based dishes	94.6	58.7	176	117.3
Plant-based dishes	29.0	6.7	34.2	7.1
Sweet products	105.1	69.9	329.4	259.7
Biscuits, sugar	32.3	5.9	122.9	31.3
Desserts	15.5	8.8	21.6	11.7
Cakes, tarts	57.3	55.2	184.9	216.7
Added fats	41.4	32.6	225.7	225.8
Sauces and seasoning	7.9	3.5	6.4	3.5
Animal fats	12.4	4.6	74.1	27.4
Vegetable fats	21.1	24.5	145.2	194.9
Water and drinks	1263.0	1065.8	50.4	36.7
Sugary drinks	75.9	45.4	25	15.6
Water	774.7	617.5	0	0

Fruit juices	54.1	44.6	23.8	19.5
Tea, coffee	358.3	358.3	1.6	1.6

Figure captions

Figure 1. Method used to estimate environmental impacts of nutritionally adequate diets with a lower share of animal protein in the diet of the adult population of France (INCA2; AFSSA (2009))

Figure 2. Eight environmental impacts of observed (OBS) and optimized low-animal-protein (LAP) diets for the entire population of France.

Abbreviations: PDF = potentially disappeared fraction; m²y = square meter multiplied by year

Figure 3. Contribution of food families in the (A) “Meat/Fish/Eggs” and (B) “Fruits and vegetables” groups to the eight environmental impacts of the observed (OBS) and optimized low-animal-protein (LAP) diets for the entire population of France.

Abbreviations: PDF = potentially disappeared fraction; m²y = square meter multiplied by year

