# Food, climate and biodiversity: A trilemma of mineral nitrogen use in European agriculture

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**Abstract**

Mineral nitrogen (N) application in agriculture has significantly increased food production over the past century. However, the intensive use of N-fertilizers also impacts negatively the environment, notably through greenhouse gas emissions and biodiversity loss and remains a major challenge for policymakers. In this paper, we explore the effects of a public policy aiming at halving agricultural mineral nitrogen use across the European Union (EU). We investigate the impacts on food security, climate mitigation, and biodiversity conservation and we analyse the potential trade-offs and synergies between them. Despite the uncertainties associated with monetary valuation and the choice of modeling approach, our results show that climate-and-biodiversity-related benefits of halving N use in EU agriculture more than offset the decrease in agricultural benefits.

Keywords: agriculture, land use, mineral nitrogen pollution, climate, biodiversity.

JEL Classification: Q11, Q12, Q15, Q18, Q52, Q53, Q54

# Contents

# 1 Introduction

Mineral nitrogen use in agriculture is at the heart of the trilemma of simultaneously ensuring climate-change mitigation, biodiversity conservation, and food security (WBGU, 2020). Indeed, mineral fertilizer is essential for feeding around half of the world's population (Erisman et al., 2008), and will be fundamental to ensuring global food security for the remainder of the 21st century. However, nearly half of the nitrogen fertilizer used is not taken up by crops and is lost into the ecosystems (Billen et al., 2013). This makes agriculture the largest global source of nitrogen pollution, leading to negative effects on land, water, biodiversity, human health, and exacerbation of climate change impacts (Sutton et al., 2013, 2021).

Historically, the regulation of agricultural fertilizer-related pollution has been aimed at different types of pollutants and on different geographical scales (EU, national and regional). For example, in response to the increasing eutrophication of water bodies, the EU adopted the Nitrates Directive (Council of the European Communities, 1991). However, 30 years after its adoption, nutrient inputs to agricultural land are generally still excessive (Eurostat, 2018). The Common Agricultural Policy (CAP) could be instrumental in supporting the sustainable use of fertilizers in agriculture, enabling farmers to maintain their productivity while reducing the harmful effects of pollution. The latest CAP reform (to be implemented from 2023) requires that Member States contribute to the Green Deal via national agricultural policies. The Farm to Fork Strategy and Biodiversity Strategy for 2030 are at the heart of the European Green Deal aiming at making food systems fair, healthy, and environmentally-friendly. These strategies pursue a reduction in nutrient losses (nitrogen and phosphorus) by at least 50% by 2030 relative to 2020.

Several recent studies have assessed the economic consequences of the reduction of chemical inputs in agriculture, in particular nitrogen, as part of the European Green Deal (Beckman et al., 2020; Barreiro Hurle et al., 2021a; Bremmer et al., 2021; Lungarska et al., 2022). These studies agree that the reduction of chemical inputs will be detrimental to the European agricultural sector with a reduction in domestic production, a loss of trade competitiveness, and an increase in food prices for the consumer. As pointed out by Barreiro-Hurle et al., these conclusions result from standard economic reasoning and are therefore not particularly surprising. However, these studies focus only on the impacts on production and competitiveness and overlook the potential benefits on climate and biodiversity.

In this paper, we fill this gap by studying trade-offs and synergies between food security[1],

[1] According FAO (1996): "Food security exists when all people, at all times, have physical and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life". Because of lack of data and our modeling framework, we restrict in this paper food security to only one

climate-change mitigation[2] and biodiversity impacts of halving fertilizer use in EU agriculture. Indeed, fertilizer use in agriculture is at the intersection between these crucial issues and the literature dealing with each of these relationships is rich. First, the transformation of atmospheric nitrogen (N) into its reactive forms and its use as fertilizer (Hager, 2009) has been one of the 20th century's major technological advances, allowing agriculture to feed the growing global population (Chang et al., 2021). Second, nitrogen losses are one of the most influential global drivers of anthropogenic biodiversity loss together with habitat destruction and climate change (Dise et al., 2011). Nitrogen deposition can directly damage vegetation, eutrophicate ecosystems and alter nutrient ratios in soil. This has a profound impact on the functioning and species richness of ecosystems. Moreover, N-fertilizer use in agriculture is responsible for large emissions of nitrous oxide ($N_2O$), the third most important greenhouse gas (GHG) after carbon dioxyde ($CO_2$) and methane ($CH_4$) with a global warming potential 265 times greater than that of $CO_2$ over 100 years (IPCC, 2013). Approximately 70% of global anthropogenic $N_2O$ over the decade 2007-2016 are due to agriculture. Global $N_2O$ emissions have increased by 10% between 1980 and 2016, mainly due increased use of N-fertilizers in agriculture (Tian et al., 2020).

The trade-offs between food security, climate mitigation, and biodiversity have generated lively debates in both the scientific (Obersteiner et al., 2016; Prudhomme et al., 2020b) and recently in policy arenas on reducing the greening of the CAP due to the current food crisis. The interplay between food security, climate change, and biodiversity makes it difficult to simultaneously meet targets on these three dimensions as described by a recent joint IPCC/IPBES report (Pörtner et al., 2021). This calls for an integrated approach.

Such an integrated approach requires a common metric to make the impacts on food security, climate change, and biodiversity comparable. Beyond the methodological challenges it implies, this raises the question of the possibibility of substitutions between these three dimensions. This question is related to that of the very nature of sustainability and echoes the debate between of strong vs. weak sustainability (Neumayer, 2003; Dietz and Neumayer, 2007). Strong sustainability consists of considering each indicator in its own unit of measurement and the level of degradation of that indicator is estimated in relation to a reference level. An example of this approach is the environmental sustainability gap framework (Ekins et al., 2003). At a global scale, according to Steffen et al. (2015), two planetary boundaries (Rockström et al., 2009) have been crossed with a high risk of serious impacts: biodiversity loss, and nitrogen and phosphorus biogeochemical flows. Weak sustainability consists of valuing the indicators in monetary terms

dimension namely "agricultural production" in the EU. This is a restrictive vision of food security which assumes that access, stability and quality are not an issue in the EU, and that food loss and waste are limited.

[2]In this paper, "Climate change mitigation" takes into account both non-$CO_2$ greenhouse gas emission ($N_2O$ and $CH_4$) reductions and carbon sequestration.

so that they can be compared using a common unit that reflects the relative importance attached to each indicator (Gutés, 1996).

In this case, the studies seek to make cost-benefit analyses where the benefits of mineral nitrogen fertilization are compared in monetary terms to the damage it causes. On the one hand, Van Grinsven et al. (2013) estimated the social costs of N-related environmental impacts on atmospheric and water pollution affecting ecosystems and human health in the EU-27 in 2008 at 75-485 billion €[/yr]. A slightly narrower but similar range of estimates (70-320 billion €/yr) is reported by Sutton et al. (2013). This social cost is expected to more than double by 2050 relative to 2010 (Brink et al., 2011). On the other hand, N-related benefits, are estimated at 10-50 billion €/yr (Van Grinsven et al., 2013, , including agricultural, ecosystem and climate benefits) and 20-80 billion €/yr (Brink et al., 2011, , agricultural benefits only). These ranges of estimates, although uncertain, suggest that the environmental costs of N-fertilizers may well outweigh their benefits in terms of agricultural and food production.

The objective of this paper[3] is to analyse the impacts of a scenario involving halving mineral fertilizer use in Europe on food production and land use and on environmental indicators (GHG emissions, carbon sequestration and biodiversity). This paper synthesizes the results obtained in two previous studies (Lungarska et al., 2022 and Devaraju et al., 2020) and goes beyond them by identifying the trade-offs and synergies between food security, climate, and biodiversity indicators. Our assessment of the impacts of halving mineral nitrogen is conducted in three steps. First, we examine the results for each of the food/climate/biodiversity dimensions separately. Second, we analyse the potential trade-offs and synergies between these three dimensions. Finally, we carry out a cost-benefit analysis of these impacts by assigning a monetary value to each of these dimensions. Our results show that reducing by half the use of N-fertilizers in the EU would affect negatively EU agricultural productivity and production. However, it would be accompanied by an increase in the number of species and a decrease in net $CO_2$ emissions. The overall social impact of halving mineral nitrogen use is found to be positive, the reduction of environmental costs exceeding EU agricultural losses.

The paper is organized as follows. Section 2 provides a description of the economic and biophysical models (subsection 2.1), links between them (subsection 2.2) as well as how we simulate a 50% decrease in nitrogen use in each model (subsection 2.3) and finally comparisons of indicators (subsection 2.4). Section 3 presents and discusses the simulation results. Section 4 concludes the study.

[3]This paper presents the synthesis of results of the STIMUL (Scenarios Towards Integrating MUlti-scale Land-use tools) project. For more information see `https://www6.versailles-grignon.inrae.fr/economie_publique/Projets/STIMUL`

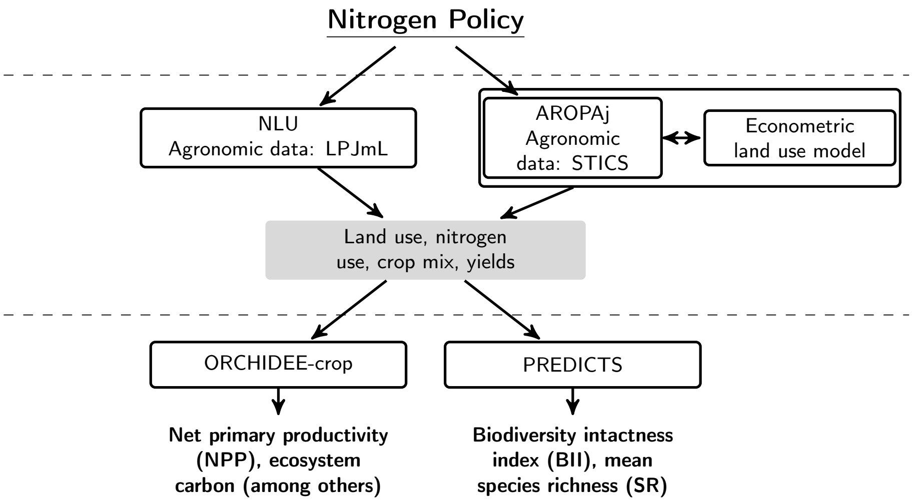


Figure 1: Models (in boxes), outputs (gray background when serving later as inputs, boldfaced otherwise) and data flows

# 2 Models, methods and scenarios

We analyse the impacts of halving mineral nitrogen use and the trade-offs between food production, biodiversity conservation, and carbon stocks by combining several existing models into a three-layer modeling architecture. First, we use two different economic models to simulate land use change under a 50%-reduction in N use in Europe (see Section 2.1.1). The resulting land-use scenarios are then used as inputs by (i) a global model of biodiversity (see Section 2.1.2) and (ii) a process-based land model that includes a representation of farmland (see Section 2.1.3) that simulates the change in carbon stocks in soil and vegetation. Each model operates at different temporal and spatial resolutions. For this reason, we use specific methodologies to connect each part of our modeling architecture (see Section 2.2).

## 2.1 Description of models

Our economic assessment is based on the results of two economic models–AROPAj and NLU–. These two models both include a rich technical content in terms of crop and livestock management, but are based on different modeling approaches (supply-side vs. partial equilibrium) and operate at different scale and resolution. In a previous study (Lungarska et al., 2022), both models were run to simulate the impacts of the same scenario of a 50%-reduction in mineral nitrogen use. The simulated impacts in terms of land use and land use change differ markedly from one model to the other. In this study, we therefore consider the results of both models to inform on the full range of possible outcomes.

The models used in our study and the interactions between them are summarized in Figure 1.

### 2.1.1 Economic models

AROPAj **(Agriculture, Recomposition de l'Offre et Politique Agricole)** (Jayet et al., 2018) is a microeconomic model of European agricultural supply that describes the behavior of individual farms representative of the diversity of farming contexts across the EU. As a supply-side model, farmers are assumed to be price takers and input and output prices are exogenous. The model does not represent the demand for agricultural products and the equilibrium. In this framework, reducing the nitrogen input per ha and consequently yields has a direct negative impact on revenues. In some cases, the costs associated with cultivation of land and crop protection (among others) can no longer be covered by the revenues associated with certain plots of agricultural land when the N reducing policy is implemented. As a result, such plots are set aside in order to limit economic losses.

On the other hand, NLU (Souty et al., 2012) is a partial equilibrium model at global scale in which supply needs to match demand. Here the profitability of farming activities is maintained through price increase and land/fertilizer substitution. Only a share of the production is displaced outside the EU because of lower competitiveness of European agriculture on international markets.

The two economic models used in this study have different focuses and therefore provide two different perspectives of the economic response to the N reduction policy. AROPAj focuses on Europe with a detailed description of local farm types while NLU entails a global perspective. Also, AROPAj has a finer representation of crop choices while NLU include finer details on the nitrogen balance. Lungarska et al. (2022) show that the two models yield two contrasted pathways regarding the reduction in N-fertilizer: a massive land abandonment with a large reduction in agricultural production (AROPAj) and an extensification of crop production with a smaller reduction in agricultural production (NLU). Thus, the use of two economic models allows us to evaluate these two pathways.

A short description of the economic models is provided in Table 1. See Jayet et al. (2018) and Souty et al. (2012) for a more exhaustive description of these models.

**AROPAj model** is built on data from the Farm Accountancy Data Network (FADN) and models agricultural supply by (group of) farmers, representative at the FADN regional level (similar to the EU NUTS2 level). Each agent in the model is maximizing its gross margin, which is the difference between production income and variable costs (such as nitrogen input costs). The mathematical programming structure of the model aims to solve this maximization problem while respecting a number of constraints associated with physical processes and the EU

Table 1: Description of economic models

| Models | Econometric models | AROPAj | NLU |
|---|---|---|---|
| **Scale** | France/EU | EU | World |
| **Resolution** | grids/NUTS3 | EU 27 and >1800 agents; 15 crop and pasture classes | 12 regions of the world. 60 land classes per region |
| **Tools** | Discrete choice models, land use share models, spatial econometric models | Mixed Linear Programming | partial equilibrium model of land-use. |
| **Input data** | Corine Land Cover, agricultural rents (€/ha), forest rents (€/ha), population density (hab/ha), land quality (text, WHC,...) , elevation, slope, climate variables | FADN, IPCC (GHG), European Soils Database | biomass demand for food and bioenergy products<br>population<br>forest area<br>price of chemical inputs<br>carbon price |
| **Output data** | Prediction of land use (agri, forest, pasture, urban) under different scenarios | crop production and acreage,<br>GHG, livestock,<br>mineral and organic fertilizers,<br>water demand<br>land shadow price | food price<br>land rent<br>crop yields<br>cropland area<br>pasture area<br>international trade in agricultural products<br>$CO_2$ and non-$CO_2$ emissions |
| **Simulated scenarios** | Climate scenarios, policy scenarios | CAP, climate scenarios,<br>tax schemes (N pollutants, GHG),<br>incentives (specific / energy crops) | Diet scenario<br>Climate scenarios (impact or mitigation)<br>Bioenergy scenario |

Common Agricultural Policy (CAP).

One of the strengths of a supply-side model like AROPAj is the fine description of production processes which is calibrated at the representative farm level. By using nitrogen-water dose-response functions derived from the crop model STICS (Simulateur mulTidisciplinaire pour les Cultures Standard, Brisson et al., 2009, 2003; Humblot et al., 2017) in AROPAj, the input level choice by farmers is endogenous under the assumption that farmers maximize their gross margin. This way, we can estimate the impact of reducing nitrogen input via an input taxation policy or simply by limiting the amount of nitrogen when defining the mathematical problem solved by AROPAj.

Another important aspect of AROPAj for this study is the dual (shadow) value of agricultural land. This value is associated with the total area constraint of the model: farmers allocate their land to different crops but they cannot exceed the total area at their disposal. The dual value associated with this constraint is a measure of the additional benefit to farmers if they could operate on one additional hectare of land. In microeconomic terms, this corresponds to the marginal profit of the land production factor. For this reason, land shadow price is important to us when modeling land use econometrically (Lungarska et al., 2022; Lungarska and Chakir, 2018).

It is worth noting that the model does not account for market feedback following the reduction in mineral nitrogen use and its repercussions in terms of yields and crop mix.

**Econometric land use model** Most econometric land use models are explanatory models, seeking to estimate statistically the drivers of land use allocation. These models target decision-support, for example by simulating the impact of public policies (pasture subsidy, deforestation tax) or climate change scenarios on land use allocation (Chakir and Le Gallo, 2021). The econometric land use model used in this paper was first developed at the French level by Chakir and Lungarska (2017) and then extended to the EU level in Lungarska et al. (2018). This model explains land use by land rents and pedo-climatic variables for the following classes: cropland, forest, pasture, and urban. Agricultural land rents are approximated by the land shadow price from AROPAj. This allows us to estimate land use allocation under different climate/policy/economic scenarios (Lungarska and Chakir, 2018; Bayramoglu et al., 2020).

**NLU model** In NLU, the agricultural sector is divided into 12 regions of the world, interconnected by international trade. NLU provides a simple representation of the main processes of agricultural intensification for crop and livestock production: the substitution between i) land

and fertilizer[4] for the crop sector and ii) grass, food crops, residues and fodder for the livestock sector. It does so by minimizing the total production cost using a supply-use equilibrium between food and bioenergy markets. A detailed description can be found in Souty et al. (2012) or in Brunelle et al. (2015).

Production intensification in the crop sector is modeled using a non-linear response of yield to fertilizer inputs. The asymptote of this function corresponds to the potential crop yield given by the vegetation model LPJmL (Bondeau et al., 2007). The yield-fertilizer relationship is calibrated on the N, P, K fertilizer consumption values calculated from FAOSTAT data. Nutrients are represented as complementary inputs without any possibility of substitution between them. Yield-fertilizer relationship parameters (minimum yield and slope at the origin) are calibrated so as to minimize the error between modeled and observed crop yields over the 1961-2006 period. NLU includes a nitrogen balance based on Zhang et al. (2015) which represents the different sources and outputs of nitrogen in the cropping system. Because NLU works in partial equilibrium, it does not take into account the feedback effects of a reduction in nitrogen fertilizer use on fertilizer markets.

Two categories of crops are distinguished in NLU: “dynamic” crops, corresponding to most annual crops (cereals, oilseeds, sugar beet and cassava), and “other” crops corresponding mostly to perennial crops (e.g. sugar cane, palm oil and some fodder crops). All categories of crops are aggregated based on their calorific values.

In both AROPAj and NLU models, animal production is taken into account as well as on-farm reused part of the cereal production, industrial feed and pastures. In addition nitrogen sourced in manure accounts in the crop-related nitrogen balance.

Because of the type of model used here, simulations presented here only inform about some of the potential consequences of an ambitious reduction in mineral N use in the absence of radical technological changes like agroecology or organic farming.

### 2.1.2 Biodiversity model

PREDICTS (Projecting Responses of Ecological Diversity In Changing Terrestrial Systems) is a modeling framework for simulating biodiversity impacts (Purvis et al., 2018) which considers land-use to be the main driver of biodiversity losses (Foley, 2005). The input land-use maps are converted into impacts on biodiversity through global estimation of two indicators of local

[4] For a given quantity of agricultural product, farmers have to combine land and inputs. The amounts of these factors of production are supposedly decided in a rational way where the objective is to reduce costs. If input prices increase (via a tax, for instance), farmers should cultivate a greater area of land in order to attain the same level of production. Thus the reduction in input (the intensive margin of agriculture) could lead to an increase in land cultivated (the extensive margin of agriculture).

biodiversity - the Biodiversity Intactness Index (BII) and the Species (plant, animal and fungi) Richness (SR) index - applying a mixed-effect modeling structure (Hill et al., 2018) to the PREDICTS database (Hudson et al., 2017).

The PREDICTS database provides a wide coverage of species (abundance and occurrence data for over 50,000 species) and in terms of geographical coverage (over 30,000 sites in nearly 100 countries). It also provides information not only on biodiversity pressures such as land-cover conversion (classified into eight categories in this version: primary vegetation, secondary vegetation, N-fixing crops, perennial crops, annual crops, pasture, rangeland and urban) or indicators of habitat fragmentation such as distance to road, but also on the intensity of the associated land use (the urban and cropland land cover classes are classified into 3 levels of intensity and the pasture class into 2 levels). This allows the impact of agricultural intensification on biodiversity to be taken into account. A more precise description of the PREDICTS's models (biodiversity models) used in this study is provided in Prudhomme et al. (2020b).

### 2.1.3 Carbon cycle model

The carbon cycle model used is the process-based land model ORCHIDEE-CROP that integrates crop-specific phenology based on STICS (Brisson et al., 2009, 2003). Carbon allocation is based on the plant-based hybrid model from the original ORCHIDEE allocation scheme (Friedlingstein et al., 1999) for natural plants while for crops it uses the crop specific formulation of STICS providing leaves, root, shoot biomass, grain formation and maturity, litter production. Litter and soil carbon decomposition are parameterized similarly on both natural crop land. The harvest date is calculated as being after grains have reached maturity (Wu et al., 2016) and harvest is exported away from the grid cell. The model has no explicit nitrogen cycle but accounts empirically for the effect of N fertilization by increasing the Rubisco- and light-limited leaf maximum photosynthetic rates as a function of the amount of N applied, according to a Michaelis-Menten function. Some agricultural practices like ploughing intensity are accounted indirectly through the turnover time of agricultural soil carbon, N fertilizer input per hectare is accounted for, and productive cultivars is modeled indirectly by lengthening the grain filling duration and by altering the harvest index. Irrigation is modelled to account whenever there is water demand for crops in the model. This version of the model currently uses three crop varieties: C3 winter, C3 summer and C4 summer. Pastures and other C3 and C4 herbaceous vegetation categories are modeled as C3 or C4 natural grass types. Forests are classified as broadleaf or needle leaf, deciduous or evergreen, temperate and boreal. Up to 11 non-cropland vegetation types can co-exist with crops on a grid point of the model. A gridded simulation of ORCHIDEE-CROP requires 30-minutes time step meteorological forcing (air temperature, specific humidity, incoming shortwave and longwave radiation, rainfall), which comes from gridded climate analysis data or atmospheric models.

## 2.2 Links between models

### 2.2.1 Link between AROPAj and the econometric land use model

As mentioned before, one of the outputs of AROPAj is the land shadow price. We use this as a proxy for agricultural land rent in the econometric land use model that allocates land between different uses on the basis of the relative revenues for each of them. Thus, the econometric land use model allows us to render the extent of agricultural land as being endogenous to the nitrogen policy scenario. The other AROPAj outputs are then extrapolated by calculating values per hectare (ha) (yield per ha, percentage of pasture land per ha, nitrogen use per ha, etc.) and then multiplying them by the new acreage resulting from the econometric land use model.

A more precise description of the link between the land-use models (NLU and AROPAJ) and the ORCHIDEE model is provided in Devaraju et al. (2020).

### 2.2.2 Links between economic models and the PREDICTS biodiversity model

The impact on biodiversity indicators of halving mineral N fertilizer application to croplands in Europe is estimated through the land-use and the land-use intensity changes computed by NLU and AROPAj. Land-use is here taken in the sense of land-cover and land-use intensity is represented by the crop yield in cropland and the stocking rate for pasture.

NLU and AROPAj provide both land use maps and yield maps which are then used by the biodiversity models. The integration is one way and a biodiversity or soil carbon change does not affect production.

The correspondence between the land uses of the economic models (NLU or AROPAj) and the land uses employed in the PREDICTS models are described in Table 2. In NLU, the “dynamic” and “other” crops Souty et al., 2012 are divided into PREDICTS crops categories (perennial, annual and nitrogen-fixing crops) based on their relative proportion in the crop mix in the reference year (Ramankutty et al., 2008). For AROPAj, the land categories are directly aggregated to PREDICTS categories through the econometric land use model.

The correspondence between the classes of cropland intensities in NLU and the PREDICTS intensities (minimum, light and intense) is achieved thanks to a generalized additive model (GAM, see Prudhomme et al. (2020b) for more details). The cropland intensity in NLU is specified by 60 land classes defined in the reference year according to their potential yield (Brunelle et al., 2018) as provided by the LPJmL model Bondeau et al. (2007). A land class includes all the land with similar potential yields. Pastures in the NLU mixed crop-livestock and pastoral production systems are aggregated into a single pasture category. In PREDICTS, pastures include rangeland, “light” and “intense” pastures. Among the NLU aggregated pasture categories, rangeland areas are defined on the basis of a reference rangeland map (Hurtt et al., 2011). For the remaining pastures, livestock density is defined on the basis of FAO livestock density maps (Robinson et al., 2014). In the reference year, a GAM is computed to match the relative proportion of “light” and “intense” pasture with livestock density maps.

A similar method is used for AROPAj, but the livestock density is directly provided by AROPAj. The AROPAj yield expressed in tons of fresh matter is converted into a yield in calories using the dry matter content and the energy content per ton of dry matter from FAOSTAT. We chose to classify fallow as an annual crop with low land-use intensity.

Table 2: Correspondence table between economic model (NLU and AROPAj) and biodiversity model (PREDICTS) land-uses

| | PREDICTS | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Cropland | | | Grassland | | Natural vegetation | | Urban |
| | Annual | Nfixing | Perenial | Pasture | Rangeland | Secondary | ¨Primary | Urban |
| NLU | cassava, maize, sugarbeet, sunflower millet, rapeseed,Vegetables, sunflower,rice,barley, rye,sorghum,sugarcane other cereals,other roots | fieldpea,groundnut soybean | Fruits, FruitsNuts Olives, oilpalm | pasture mixed land, pastoral pasture | pastoral pasture | Forest | Forest | Urban |
| AROPAj | Durum wheat, Tender wheat, Winter barley, Oats,Rice,Potato, Other cereals, Maize, fallow, Beetroot, rapeseed, | Soybean, Other legumes | Perennial | Pasture | Rangeland | Forest Other ecosystems | Forest Other ecosystems | Urban |

### 2.2.3 Links between economic models and the carbon cycle model

The impact on ecosystem carbon of halving mineral N fertilizer application is simulated by forcing ORCHIDEE-CROP with the land-use change computed by AROPAj and NLU. In order to link the land use maps from AROPAj and NLU models with the carbon cycle model ORCHIDEE-CROP, the first step is to match the land uses between the three models. ORCHIDEE-CROP is a global land surface model that aggregates diverse plants into specific plant types based on climatic conditions. They are named Plant functional types (PFTs), a system used by climatologists to classify plants according to their physical, phylogenetic and phenological characteristics as part of an overall effort to develop a vegetation model for use in land use studies and climate models. Plant functional types provide a finer level of modeling than biomes. Both AROPAj and NLU crops are grouped in ORCHIDEE-CROP as C3 winter crops, C3 summer crops and C4 summer crops. Pastures and rangeland are grouped into C3 and C4 natural grasslands depending on the climate classification. Finally, forest and other natural areas of NLU and AROPAj are classified as natural forest plant types (classified as broadleaf or needle leaf, deciduous or evergreen, temperate and boreal). Note that the fallow land described in AROPAj that is part of crops is classified in the plant type "grass" in ORCHIDEE. Urban land is included in the baresoil class type as this version of ORCHIDEE does not include urban areas and, with respect to carbon sequestration, baresoil is closer to urban behavior than any other ecosystem. Land use maps for the two scenarios (baseline and halving N) are used as inputs to the ORCHIDEE-CROP simulations from both AROPAj and NLU.

Table 3: Correspondence table between economic models (NLU and AROPAj) and carbon cycle model (ORCHIDEE-CROP) land-uses

| | ORCHIDEE-CROP | | | | | |
|---|---|---|---|---|---|---|
| | | Cropland | | | Natural vegetation | Other land |
| | C3 Winter crop | C3 Summer crop | C4 Summer crop | C3, C4 Natural grassland | Temperate and Boreal Needleleaf, Broadleaf, Evergreen and summergreen trees | Baresoil |
| NLU | rapeseed<br>wheat | millet, rice, soybean,<br>sugarbeet, sunflower | maize | pasture<br>Vegetables<br>Pasture mixed land<br>Pastoral pasture | Forest | Urban, other land |
| AROPAj | Durum wheat,<br>Tender wheat,<br>Winter barley,<br>rapeseed | rice<br>Spring barley<br>Soybean,<br>Other cereals<br>sunflower | maize | pasture<br>rangeland<br>fallow<br>Beetroot<br>Potato | Forest | Urban<br>Other ecosystems |

## 2.3 Simulated scenarios

### 2.3.1 The baseline scenario

The two economic models are based on different hypotheses but also on different data sources. The initial land use allocation is based on FADN and Corine Land Cover data (for AROPAj and econometric land use model respectively) and on FAO data (for NLU).

In the NLU baseline situation, cropland and pasture represent some 174 million ha in Europe. The amounts of mineral and organic N are 12.5 and 4.7 **Teragram nitrogen per year** (TgN/yr) respectively. The yield per ha is 2.56 tDM/ha/yr and the total crop production is 331 million tDM/yr.

For AROPAj and the econometric land use model, there are some 194 million ha of cropland and pastures in Europe. Nitrogen input is 18 TgN/yr for mineral N and 1.8 TgN/yr for animal manure. The yield per ha is on average 3.86 tDM/ha/yr and the total crop production is 475 million tDM/yr[5].

[5] The differences between the two models are exaggerated by the recalculation of land use acreage by the econometric land use model and the extrapolation of AROPAj values (see section 2.2.1). Without the econometric land use model, cropland and pastures represent 124 million ha, the N mineral use is practically the same as in NLU and the total crop production is 338 million tDM/yr.

### 2.3.2 Scenario of mineral nitrogen reduction in the EU

There are two ways to simulate the -50% reduction in N scenario in AROPAj. The first consists of running a benchmark simulation and then imposing a 50% reduction in N on all farms (functioning here as one and thus sharing the effort) as a technical limitation to their activity. The second is to increase the price of the N input to a level that leads to total N use in EU agriculture being halved. We conducted our current simulations using the first method. AROPAj economic agents have different options for adapting their activity to the new constraints. Since the choice of N input is endogenous thanks to the dose-response functions derived from STICS, farmers can simply reduce the level of input (adaptation on the intensive margin). They can also switch from one more N intensive crop to another that requires less input (adapting production on the extensive margin).

In NLU, the -50% of N is simulated by artificially increasing the fertilizer price. This orientates farmers' behavior towards land-intensive, rather than N intensive, practices. We use a stepwise procedure to select the fertilizer price level consistent with a -50% reduction in N. As there is no possibility of substitution between nutrients, the reduction in N implies an equivalent reduction in P and K. In both economic models the -50% reduction in N is assumed to take place in 2012.

The mineral nitrogen reduction scenario is denoted "HalfN" in the rest of the paper.

## 2.4 Comparison of indicators

### 2.4.1 Food, climate and biodiversity indicators

Two indicators are used to assess the impact of the scenarios on food: food production and total economic surplus. These indicators are outputs of the NLU and AROPAj models. Food production can be compared with domestic food demand to estimate food sovereignty indicators. It also allows food production to be compared with a purely biophysical approach without taking price effects into account. Production is not associated with any demand in AROPAj, while it is associated with a price-inelastic food demand in NLU. In NLU, however, European production may not match domestic demand if Europe imports its production to satisfy domestic demand or if it starts producing and exporting to satisfy demand from other countries. Since there is no consumer represented in AROPAj, the change in economic surplus corresponds simply to the change in farmers' profit (here the gross margin). In NLU, the change in economic surplus is the sum of the surplus change from the consumer side (estimated by linear approximation) and the producer side. Profit in AROPAj and NLU takes into account not only changes in production, but also changes in feed prices and costs.

Biodiversity is represented by two complementary indicators: species richness (SR) and the biodiversity intactness index (BII). The BII indicator is defined as the average similarity between an ecosystem and a primary natural ecosystem, weighted by the abundance of a taxonomically and ecologically broad set of species in an area, relative to their abundances in an intact reference ecosystem. The SR reports the number of species, relative to the number expected in a natural system.

Impacts of scenarios on climate are assessed through biomass production and soil carbon quantification. For biomass production we look in particular at the ecosystem Net Primary Productivity (NPP) indicator. NPP reflects the carbon assimilated by the biomass or vegetation through photosynthesis that is available for allocation after accounting for autotrophic respiration. An increase in NPP permits the allocation of carbon for new leaves, roots and stems of the plants that could lead to an increase in biomass and sequester carbon in soils. The other indicator is soil carbon: carbon sequestration in soils reflects the accumulation of carbon into the ecosystem and carbon loss in soils reflects the $CO_2$ emission from the ecosystem.

### 2.4.2 Conversion of indicators into monetary units

To compare the impact of halving mineral nitrogen fertilization in Europe on each dimension of the agriculture-climate-biodiversity trilemma, we express each impact of the HalfN scenario in monetary units within a cost-benefit analysis. This weak sustainability type analysis consists of monetising environmental externalities, such as environmental pollution, and adding them to agricultural production already monetized through the existence of markets. The assessment of environmental externalities as monetary values is challenging and requires an adaptation of our methodological framework: firstly, we rely solely on the outputs of the NLU model since it provides a more accurate description of N flows than AROPAj; secondly, the assessment of biodiversity in monetary values is based on Van Grinsven et al. (2013) rather than on the outputs of the PREDICTS model. The methodology proposed by Van Grinsven et al. is well established in the field and seems to us to be more robust than a coarse approximation based on PREDICTS' outputs in monetary units.

The benefits considered in this analysis correspond to the producer profits attributable to the use of nitrogen and the consumer surplus. Details of the calculus of profits and consumers surplus are presented in Appendix B. The costs are the social cost of nitrogen and the social cost of carbon. The social cost of nitrogen is calculated using the same method as Van Grinsven et al. (2013) to allow a comparison of this study's results with theirs. The social cost of nitrogen can be broken down into a human health component (corresponding to the nitrogen pollution of surface waters used as drinking waters, ozone depletion due to NOx and the impact of NH3 volatilization on health), ecosystem quality (eutrophication of aquatic and terrestrial environments by leached nitrogen, redeposition of volatile NH3) and climate (cooling effect of NOx and NH3 due to ozone depletion, warming effect of $N_2O$ as a powerful greenhouse gas). For this purpose, we calculate: (i) $N_2O$ emissions not only from crop fertilization (Tier I of the IPCC method), but also from manure management (Tier II of the IPCC) as described in Prudhomme et al. (2020a), (ii) NH3 emissions from nitrogen volatilization during crop fertilization (IPCC Tier I), and (iii) NOx emissions which represent 30% of NH3 emissions. Biodiversity is valued at the marginal cost of maintaining surface waters against eutrophication of 18/kgN lost (range 10-30€/kgN), plus the marginal cost of maintaining ecosystems against N deposition of 2€/kgN (range 2-10€/kgN) according to Van Grinsven et al. (2013) (See Table 4). Carbon is valued by taking the average value between the social cost of carbon with a 2% discount rate as advocated by Stern and Stiglitz (2021): 195\$/t$CO_2$ (234€/t$CO_2$ using the euro-dollar parity of 1.2 in 2012) and the social cost of carbon with a 10% discount rate as advocated by Nordhaus (2014): 31\$/t$CO_2$ (37€/t$CO_2$ using the euro-dollar parity of 1.2 in 2012).

Table 4: Cost of nitrogen pollution (NOx,NH3 and $N_2O$) expressed in € per kg **of reactive nitrogen (Nr)** emitted, used or produced based on Van Grinsven et al. (2013). Due to the high uncertainty associated with these costs, we provide here the lower and the upper bound of these costs.

| | Human health | Ecosystems | Climate |
|---|---|---|---|
| $N_r$ to water | 0 to 4 | 5 to 20 | - |
| $NH_3 - N$ to air | 2 to 20 | 2 to 10 | 3 to 0 |
| $NO_x - N$ to air | 10 to 30 | 2 to 10 | 9 to 2 |
| $N_2O - N$ to air | 1 to 3 | - | 4 to 17 |

# 3 Results

## 3.1 Simulation results for individual food/climate/biodiversity objectives

### 3.1.1 Food production, associated land use and economic surplus

Both economic models show that the HalfN scenario leads to a reduction in domestic agri-food production, with however substantial divergence in the magnitude of the production loss: -34% in AROPAj vs -12% in NLU compared to the no-policy scenario. This discrepancy stems from the two different modeling approaches.

In NLU, the change in relative prices between land and fertilizer incites the farmers to substitute between the two inputs. The reduction in fertilizer use leads to decrease in crop yields of -18% which in turn drives an expansion of cropland by 3% at the expense of pasture (-6%). This mechanism reflects an increased market share for low-input farming. There is no change in forest areas in NLU, as they are exogenously driven.

On the other hand, the supply-side nature of AROPAj results in large areas of agricultural land being left fallow, as in some cases, the costs associated with cultivation of land and crop protection (among others) can no longer be covered by the revenues associated with certain plots of agricultural land. The greatest impact in terms of production concern oilcrops and tubers (-41% and -35% respectively). The lowest impact is on leguminous crops (-13%) since their ability to fix nitrogen directly from the atmosphere limits the need for it to be supplied as an external input. These impacts are the repercussions of the reduction in crop yields (-12% overall) and the reduction in cropland of -4% benefiting pastures and forests. However, the major land use change, predicted by AROPAj and extrapolated by the econometric land use models, is an increase in fallow land in the EU, where some 26 million ha (or some 20% of the initial cropland area) are no longer put into production. Nevertheless, this land use is transitional. The abandoned land will either become forest or rangeland or be converted to less nitrogen-intensive agricultural practices such as organic farming. Since the latter use is still ill-represented in official statistics, AROPAj cannot yet account for this possible development.

Since there is no consumer represented in AROPAj, the change in economic surplus corresponds simply to the change in farmers' profit (here the gross margin). In AROPAj, the profit decreases by 37.8 Billions€(Bn€) in direct response to the decrease in production. In NLU, the change in economic surplus affects both consumers and producers. In total, NLU estimates a loss in surplus by -7.7Bn€ due to the N reduction policy. As the agri-food demand is mainly rigid in NLU (only the demand for feed is somewhat flexible), the loss in surplus is exclusively at the expense of the consumers.

**Effects on other chemical inputs** The use of pesticides is correlated to the use of mineral nitrogen in NLU. The dose response functions in NLU link use of all these chemical inputs with the yield. Thus, a change of mineral fertilizer use in NLU is associated with a change in pesticide consumption systematically by construction. The effect of a change in the consumption of other inputs such as pesticides on biodiversity is in this modelling framework mixed with the effect of the change in consumption of mineral fertilizer through the change in yield.

Crop protection expenses are also accounted for in the AROPAj model as quasi-fixed costs associated with a given crop. Given the constraint in mineral N use, farm types can either switch between crops (some being more or less vulnerable and thus more or less pesticides-intensive) or stop production on given areas (when the quasi-fixed costs cannot be covered by income). In this sense and since an important span of area is set aside, pesticide consumption is also lowered.

| | Geographical Scale | |
|---|---|---|
| Indicator | EU | Global |
| Food | | |
| AROPAj | | |
| Production | -34% | - |
| Change in prod. surplus | -37.8Bn€ | - |
| NLU | | |
| Production | -12% | 0.3% |
| Change in tot. surplus | -7.7Bn€ | |
| Biodiversity | | |
| based on AROPAj | | |
| SRI | 1.9% | |
| BII | 2% | |
| based on NLU | | |
| SRI | -0.4% | |
| BII | 1.1% | |
| Climate | | |
| based on AROPAj | | |
| NPP | 1.85% | - |
| Soil Carbon | 4.89% | - |
| Non-$CO_2$ emissions | -10.9% | - |
| based on NLU | | |
| NPP | -0.13% | - |
| Soil carbon | -0.49% | - |
| Non-$CO_2$ emissions | -35.9% | 1.4% |

Table 5: Summary of results for each individual objective in percent change from the no policy scenario or in billions of euros (€bn)

### 3.1.2 Biodiversity impacts

In the case of the AROPAj land-use scenario, the PREDICTS model simulates increase in both BII and SR (respectively 2.0 and 1.9% at the EU level) due to halving of N fertilizer (Table 5). In AROPAj, the increase in forest, pasture and other herbaceous vegetation areas at the expense of cropland leads to an increase in the number of species (increase in SR).

In the case of the NLU land-use scenario, PREDICTS simulates on average a small increase in BII and a decrease in the relative number of species (+1 and -0.4% respectively ) in the HalfN scenario compared to the baseline. The decrease in SR due to a decrease in cropland yield is here partially offset by the replacement of pasture (an ecosystem with high species richness) by cropland (an ecosystem with lower species richness). Here, the small increase in BII is due to the replacement of the pasture ecological communities (very different to the ones found in the primary natural ecosystem) by cropland ecological communities (more similar to ones found in the primary ecosystem).

### 3.1.3 Carbon impacts

Carbon assimilated by the biomass or vegetation through photosynthesis in the ORCHIDEE-CROP carbon cycle model is referred to as NPP as defined above.

Halving of N fertilizer over Europe contributes to a significant increase in total net primary production 38.45 million tons of C per year ($MtCyr^{-}1$) for the AROPAj scenario (1.85%, see Table 5) . This increase in NPP contributes to an increase in soil carbon sequestration of ~4.89%. A much higher increase is simulated if we look at France alone, i.e. ~2.53% increase in NPP and ~6% increase in soil carbon sequestration. This is the consequence of increased forest, pasture, grasslands and other herbaceous vegetation due to halving N fertilizer in AROPAj.

The NLU land-use change scenario simulates, on average, a contrasting response in total NPP (Table 5) when compared with the AROPAj land-use scenario. A reduction in N fertilizer causes a decrease in total primary production of up to -2.71 MtCyr-1 across Europe (-0.13%). A decrease in NPP contributes to a decrease in soil carbon sequestration (-0.49%). For France, the NPP decrease is -0.31% and the soil carbon sequestration decrease is -0.6%. The decrease in NPP and soil carbon for the NLU scenario is mainly due to the loss of herbaceous vegetation and pasture areas resulting form the increase in cropland areas and to the lower production on cropland due to the decreased fertilization.

## 3.2 Trade-offs and synergies between the three dimensions of the trilemma

The relationships between the food/climate/biodiversity objectives are analysed through 6 indicators (see Fig. 2): food production, economic surplus, non-$CO_2$ greenhouse gas emissions ($N_2O$ and $CH_4$), carbon sequestration in soils and vegetation, species richness (SRI) and intactness index (BII) (for details of the calculation of each indicator, see Section 2.4). The variation in each of these indicators is analysed with respect to desired objectives: increase in food production and economic surplus, decrease in non-$CO_2$ emissions, increase in carbon in soil and vegetation, increase in SRI and BII.

We distinguish three types of relations: (i) trade-off occurs when the two indicators vary in opposite directions with respect to the desired objective (e.g. food production decreases and non-$CO_2$ emissions decrease); (ii) positive synergy occurs when the indicators both show an improvement towards the desired objective (e.g. non-$CO_2$ emissions decrease and SRI increases) and (iii) negative synergy occurs when the indicators both show a deterioration with respect to the desired objective (e.g. food production and economic both declines).

The two modeling frameworks used in this paper - AROPAj/ORCHIDEE/PREDICTS and NLU/ORCHIDEE/PREDICTS - lead to sharply contrasting results regarding the relationships between indicators. When the AROPAj model is used, we obtain positive synergies between climate and biodiversity, trade-offs between climate/food and food/biodiversity and a negative synergy between the food indicators: food production and economic surplus. On the other hand, when the NLU model is used, we obtain mainly trade-offs and negative synergies, and only one positive synergy between BII and non-$CO_2$ emissions.

This difference between the models' results can be explained by two contrasting views of how the policy is implemented: with AROPAj, the use of mineral nitrogen is reduced mainly by reducing agricultural production (which drops by -37%). This leads to significant areas of agricultural land being abandoned. As a result, food indicators deteriorate sharply, while climate/biodiversity indicators improve substantially. In the case of the NLU model, the situation is more complex. Food production decreases, but to a lesser extent than in AROPAj. The decrease in nitrogen consumption leads to a decrease in yields, and consequently to an increase in cropland area (+3%). As a result, we have here ambiguous results on the climate and biodiversity indicators: the decrease in mineral nitrogen improves the non-$CO_2$ emissions indicator and the BII, while the changes in agricultural surface area degrade the carbon sequestration in soils and vegetation and the SRI.

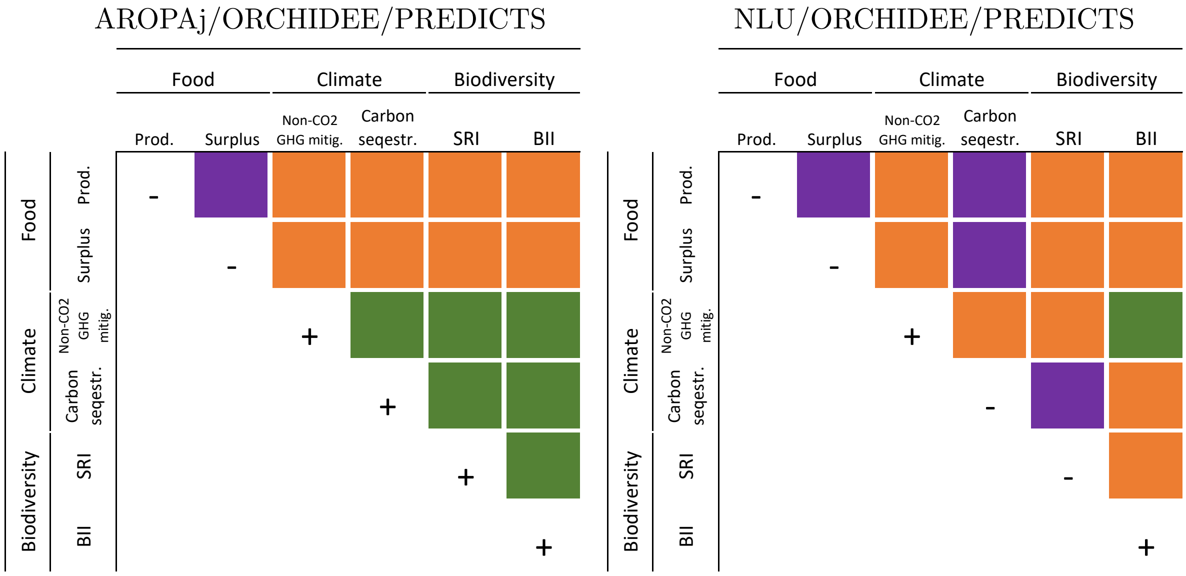


Figure 2: Trade-offs between food, climate and biodiversity. A green cell indicates that both variables improve in the HalfN compared to the baseline scenario (positive synergy), a purple cell indicates that both variable decreases in the HalfN compared to the baseline scenario (negative synergy) and an orange cell indicates that one variable decreases and one variable increases in the HalfN compared to the baseline scenario (trade-off). On the diagonal, a plus indicates that the variable is higher in the HalfN scenario than in the baseline and a minus indicates that the variable is lower in the HalfN scenario than in the baseline

### 3.3 An attempt at cost-benefit analysis of mineral nitrogen use

To compare the benefits and costs associated with a 50% reduction in the use of mineral nitrogen, we monetize the pollution associated with the use of synthetic nitrogen in agriculture and the benefits associated with the use of synthetic nitrogen. To calculate the social cost of nitrogen, we use Van Grinsven et al. (2013)'s method as it covers the different impacts of nitrogen pollution. The costs per unit of pollution (Table 4) are estimated from a method of monetary valuation of the externalities associated with the associated pollution. Other methods can be used leading to different costs. Due to the uncertainty of these costs, we use the minimum and the maximum cost per unit of pollution provided by Van Grinsven et al. (2013). The costs associated with ecosystem pollution, climate and health are thus represented with uncertainty bars (Fig.3). The cost ranges chosen, although very wide and not allowing for a precise evaluation, nevertheless allow us draw conclusions on the environmental benefits of halving nitrogen fertilization. Recent monetary costs associated with maintaining water quality through biodiversity are however within the range proposed in this study (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, IPBES., 2018).

Economic valuation of carbon sequestration was done using the social costs of carbon from Nordhaus (2014). We did not apply discount factors associated with taking into account 2050 emissions and 2020 emissions in the same indicator. Similarly, we have only applied a single carbon price and not a carbon price trend. Finally, we could have chosen to calculate a valuation of the carbon price using market prices such as those of the EU-ETS market. But these are well below the carbon price compatible with the Paris agreements.

Finally, the economic valuation of agricultural production was performed by taking into account agricultural profits enabled by the use of nitrogen and consumer surplus losses. We do not take into account the impact of agricultural production on the economic activity of other sectors such as the food industry or the input-producing industry. The partial equilibrium models used in these studies do not allow the economic activity of other sectors than the agricultural sector to be taken into account.

Our results show that social cost of nitrogen is much lower in the HalfN scenario (21-143 Bn€) than in the baseline (49-302 Bn€) (see Figure 3). This cost reduction is much bigger than the economic losses associated with the reduction of agricultural production (9 Bn€) in the HalfN scenario compared to the baseline (see Section A in SI for more details).

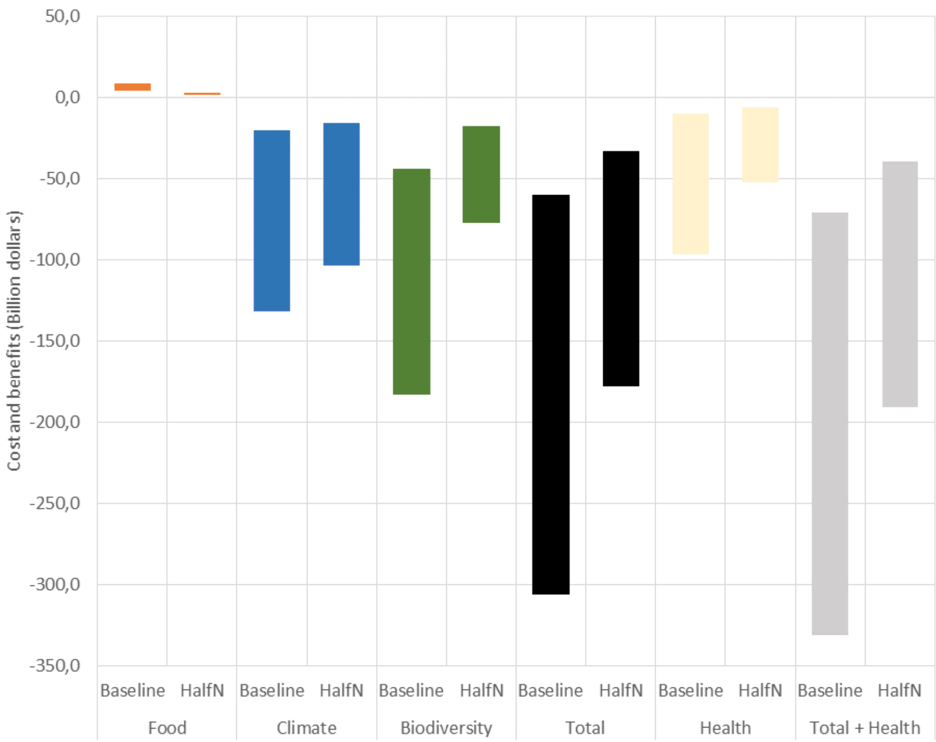


Figure 3: Food, climate, biodiversity and total impact of the baseline and the HalfN scenarios (in monetary terms Bn€) for the NLU results. A negative impact corresponds to a cost and a positive impact corresponds to a benefit. The color bars correspond to the range in costs evaluations. Paler yellow corresponds to health impacts.

**Comparison of the cost-benefit analysis to other studies** Despite the limitations of the economic evaluation mentioned above, we have improved several aspects of the estimation of costs and benefits. Concerning the estimation of the social cost of nitrogen, we find similar results to Sutton et al. (2011) and Van Grinsven et al. (2013) for the baseline (Table 6). The main contributions of our estimates lie in the estimation of agricultural benefits and carbon sequestration associated with land use change in the HalfN scenario compared to the baseline scenario. In addition to the agricultural benefits usually used in other studies (Table 6), we also estimated the losses to the consumer associated with the use of poorer quality land in the HalfN scenario compared to the baseline (- 12,5 Bn€). These results allow us to improve the social cost analysis of nitrogen use in agriculture (Kanter et al., 2021).

| | Baseline | Sutton et al. (2011) | Van Grinsven et al. (2013) | Halving N |
|---|---|---|---|---|
| **Agricultural benefits** | 31 | 20 to 80 | 20 to 80 | 40 |
| Cost for human health | 10 to 97 | 9 to 79 | 10 to 70 | 6 to 53 |
| Cost for ecosystems | 44 to 183 | 21 to 90 | 35 to 115 | 18 to 77 |
| Cost for climate | -6 to 23 | 2 to 5 | -10 to 5 | -4 to 14 |
| **Social cost of nitrogen** | 49 to 302 | 32 to 174 | 35 to 230 | 21 to 143 |

Table 6: Comparison of the cost-benefit analysis of the baseline scenario (in 2012) compared to other cost-benefit analyses (in 2008) for the agricultural sector in Bn€.

Although our assessment is surrounded by large uncertainties, the interest of our approach is to quantify fundamental trade-offs between food security, climate change, and biodiversity. These trade-offs are utterly important for the policy-making process (as exemplified by the current situation) and the use of a common metric–although imperfect–to resolve these trade-offs is also of great importance. Although the use of a single metric raises methodological and conceptual issues about how to think about nature. This is why we remained cautious in our phrasing and spoke of "an attempt to economic evaluation"; we think the uncertainty in the economic estimates that we show is in itself an interesting result. Finally, our assessment contributes to the set of assessments we identified in Table 6 and is useful in this for moving towards more reliable estimates.

# 4 Conclusion

The objective of this paper was to analyse the impacts of a public policy aimed at halving mineral nitrogen use in European agriculture. We investigated the impacts of such a policy from the economic and environmental perspectives. We consider a modelling framework based on a set of models: i) the European agricultural supply-side model AROPAj (combined with a spatial econometric land use model at the EU level), ii) the global partial equilibrium agricultural model NLU iii) ORCHIDEE-crop carbon cycle model; and iv) PREDICTS biodiversity simulation model. Our modelling framework allows us to examine the results for each of the trilemma food/climate/biodiversity impacts separately and to analyse the potential trade-offs and synergies between them. We then use these results to carry out a cost-benefit analysis by assigning a monetary value to each dimension of the trilemma.

The main contribution of our paper is to show that a mineral N reduction policy leads to a complex set of trade-offs and synergies between food/climate/biodiversity objectives, which depends fundamentally on the type of economic response of the agricultural sector to the mineral N reduction. In a nutshell, a significant decrease in agricultural production as simulated by the AROPAj model leads to better results in terms of climate and biodiversity and to positive synergies between these two dimensions. A more moderate reduction in agricultural production as simulated by the NLU model gives mixed results on the different economic and environmental indicators and a majority of trade-offs between food/climate/biodiversity objectives.

Based on the results of the NLU model, we performed a cost-benefit analysis to investigate the trade-offs between food/climate/biodiversity on a monetary basis. Our results show that halving nitrogen use in EU agriculture will decrease agricultural benefits, but it will also reduce costs on carbon and ecosystems with an overall positive impacts (including human health). The reduction of nitrogen fertilization in Europe leads to a considerable decrease in the costs to society of environmental damages related to their use in agriculture. But these environmental damages are not currently marketed, whereas the production losses associated with reduced nitrogen fertilisation are. Our results therefore suggest paying farmers for the environmental services they would render to society by reducing their mineral fertilisation by half. This payment would, according to this study, largely compensate the economic losses for farmers associated with lower agricultural production in Europe. This payment would be borne by tax-payers who will benefit from halving nitrogen fertilisation through improved freshwater and air quality. From a policy-making point of view, it seems essential to analyze the impacts of mineral nitrogen reduction at the finest possible geographical scale to take into account the large variability in the location of the costs and benefits of mineral nitrogen reduction. This will guarantee a proper valuation of the ecosystem services provided by farmers (Burkhard et al., 2018) and a good articulation with existing territorial policies (Lungarska and Jayet, 2018).

Our framework could be extended and several remaining questions could be considered in future work. First, a possible extension of our work is to consider a dynamic framework. Indeed, our static modelling framework prevents us from taking into account the evolution of agricultural policies such as the CAP and the effect of climate change on yield in the baseline. This means that we probably overestimate agricultural production compared to 2030 (Barreiro Hurle et al., 2021b). Second, we don't take into account in our framework the effect of indirect land use change (iLUC) on climate and biodiversity. Our results show an increase in the global agricultural production by +0.3% to overcome EU agricultural losses, which could have environmental impacts in terms in terms of biodiversity and carbon. Finally, our framework allow to analyse only impacts on agricultural sector, a general equilibrium model would be necessary to estimate the overall impacts on the other sectors (Bellora and Bureau, 2013). We hope that our analysis will pave the way for further research that overcomes these limitations and sheds new light on these important questions.

# 5 Declarations

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### Ethics

This article does not contain any studies involving human participants performed by any of the authors.

### Conflicts of interest/Competing interests

On behalf of all authors, the corresponding author states that there is no conflict of interest.

### Consent to participate

On behalf of all authors, the corresponding author states that all authors consent to participate.

### Consent for publication

On behalf of all authors, the corresponding author states that all authors consent to publication.

### Availability of data and material

Available on request.

### Code availability

Not applicable

### Authors' contributions

RP, RC, AL, TB and ND led the formal analysis and the writing. RP, RC, AL, TB, ND, NDN, PAJ, SDC, JCB participated to the design of the research.

### Disclaimer

The authors only are responsible for any omissions or deficiencies.

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# A Cost-Benefits Analysis

Table 7: Details of the cost-benefit analysis

| | | Minimum | Maximum |
|---|---|---|---|
| Food | Baseline | 4.2 | 4.2 |
| | HalfN | 1.4 | 1.4 |
| Climate | Baseline | -20.8 | -131,6 |
| | HalfN | -16.4 | -103.5 |
| Biodiversity | Baseline | -44,2 | -182,7 |
| | HalfN | -18,4 | -77,2 |
| Total | Baseline | -60.8 | -310.1 |
| | HalfN | -33.4 | -179.3 |
| Health | Baseline | -10,2 | -96,7 |
| | HalfN | -6,2 | -52,5 |
| Total + Health | Baseline | -71.1 | -331.5 |
| | HalfN | -39.6 | -190.8 |

# B Profits and consumer surplus calculus

In Section 3.3, we consider the profit attributable to the use of nitrogen, which is calculated based on the NLU's outputs as follows:

$\pi_N = [\frac{\rho_{min}}{\rho}\frac{N_{free}}{NPK_{free}} + \frac{\rho-\rho_{min}}{\rho}\frac{N_{synthetic}}{NPK_{synthetic}}] * \pi$

with $\pi_N$: profit attributable to nitrogen, $\rho_{min} : yieldgetwithoutcostlyinputs$,$\rho$: actual yield, $N_{free}$: free nitrogen (manure, deposition, biological fixation), $NPK_{free}$: freen NPK, $\pi$:profit, $N_{synthetic}$: synthetic nitrogen,$NPK_{synthetic}$: synthetic NPK. See Table 5 for numerical values from NLU.

The consumer surplus is calculated as the difference in food cost (price times consumption distinguished between vegetal and animal food) between the HalfN scenario and the baseline.

Table 8: Elements used to compute profit attributable to the use of nitrogen in the NLU model.

| | Baseline | Halving mineral N |
|---|---|---|
| Min yield (Mkcal/ha/yr) | 3.96 | 3.96 |
| Actual yield (Mkcal/ha/yr) | 10.4 | 8.6 |
| Crop profits (billion $/yr) | 41.6 | 46.2 |
| Free nitrogen rate (kgN/ha/yr) | 61.3 | 56.7 |
| Synthetic nitrogen rate (kgN/ha/yr) | 124.7 | 43.9 |
| Free PK rate (kgPK/ha/yr) | 1.2 | 3.3 |
| Synthetic PK rate (kgPK/ha/yr) | 50.9 | 13.2 |