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RESEARCH ARTICLE

# Defining cost-effective ways to improve ecosystem services provision in agroecosystems

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## Abstract

Mitigating climate change through the adoption of environmental-friendly agricultural practices also affects biodiversity and the provision of other non-marketed ecosystem services (hereafter, ES). In this paper, we investigate a method to identify cost-effective strategies to improve the provision of these ES. We model the link between agricultural practices and the provision of ES, to illustrate the general antagonism between agricultural production and the provision of non-marketed ES, as well as synergies among the latter. We run efficiency analyses on the simulated agroecological data to explore the interactions among ES and identify efficient bundles of ES. Improving the provision of non-marketed ES comes at a cost in terms of production. The bundle of ES provided by an alternative management option has an opportunity cost corresponding to the profit loss compared to the most profitable management option. We determine which strategy costs less to improve the provision of non-marketed ES: to adopt a given set of agroecological practices over the whole agricultural area, or to dedicate only a part of the landscape to the provision of the non-marketed ES. This result is helpful to determine if agroenvironmental policies should target large areas with uniform low requirements, or several smaller areas with higher environmental conditions. It can be used to determine cost-effective ways to mitigate climate change through agricultural practices reducing greenhouse gases emissions and increasing carbon storage in soil while maintaining other ES.

**Keywords** Agriculture · Ecosystem-services bundles · Production possibility frontiers · Efficiency analysis · Cost-effectiveness · Land-sparing/land-sharing

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## Introduction

Agricultural intensification has resulted in an increase of the quantity of food supplied by agroecosystems, along with a decline in biodiversity and the provision of many biodiversity-based ecosystem services (hereafter, ES). Pesticide inputs, reduced crop diversity, and the removal of many semi-natural areas (hedges, field margins, wetlands) have had dramatic effects on farmland biodiversity, such as insects and in particular pollinators (Deguines et al., 2014), farmland birds (Burel et al., 1998; Donald et al., 2001; Wretenberg et al., 2006), and some farmland-specialist mammals (de la Peña et al., 2003; Pocock & Jennings, 2008). Soil organic matter has declined, as well as soil biodiversity (Matson, 1997). Water bodies are contaminated by pesticides, nutrients causing eutrophication, and sediments. Agriculture is also responsible for an important share of the emissions of greenhouse gases, either through agricultural practices (fuel, fertilisers) or carbon release due to the degradation of soil organic matter. This trend is not sustainable, both for the ecosystems themselves, and also for food production: the decline in pollinator populations and soil fertility, as well as the human-induced climate change are a threat to maintaining yields in Europe (Deguines et al., 2014; Stoate et al., 2001; Tan et al., 2005).

The agriculture sector is challenged to contribute to the mitigation of climate change, both by reducing its greenhouse gases emissions and by storing carbon in agricultural soils. However, only agricultural production generates profit. A farmer considering only his private profit has interest to focus on agricultural commodities production and little incentive to provide more of the other ES. Due to the functioning of ecosystems, there is globally a trade-off between agricultural production and the other ES, so that intensive agricultural practices are associated with low provision of non-marketed ES, due to a degradation of the corresponding ecosystems and related biodiversity. Changes in agricultural practices are therefore required to balance commodity production, the mitigation of climate change, and the provision of biodiversity-based ES.

Agroenvironmental policies are gradually implemented to enhance the provision of non-marketed ES, for example via the AgroEnvironment Climate Schemes of the Common Agricultural Policy in the EU. Changes in agricultural practices impact several (often all) ES provided in agroecosystems: even when they only target one of them, the multiple and complex interactions among ES create side effects on the other ES. Lindenmayer et al. (2012) show that a policies aiming at increasing carbon storage to mitigate climate change may alter biodiversity if biodiversity is not taken into account in the policy design. On the contrary, building on the existing synergies among different policy objectives is a good way to design more effective policies (Zhang & Pagiola, 2011). In order to choose the management option with the greater climate benefits, its performance must be assessed by considering effects on all ES. The option maximising a given ES may achieve low levels of many others. However, if many assessments integrate more than two ES, only few studies aiming at defining optimal strategies to provide ES include more than two ES.

Besides, changing agricultural practices incurs a cost, either because of a lower yield or because of additional management costs. No matter if the cost is supported by the farmer or the taxpayer via agricultural subsidies, in the interest of the whole society, it should be minimised. As stated by Naidoo et al. (2006), integrating the cost in conservation planning is crucial to maximise biodiversity conservation benefits when the conservation budget is limited. This recommendation is followed by some recent publications studying the provision of single ES (Ruijs et al., 2013), but few frameworks account for the cost of providing bundles of ES in interactions (Ruijs et al., 2017).

Our paper proposes a way to explore and summarise the complex interactions (antagonisms and synergies) among ES in agroecosystems, and determine cost-effective ways to increase the provision of non-marketed ES. To illustrate this approach, we generate data from a simple agroecological model assessing the effect of various agricultural practices on the provision of five ES: agricultural production, climate regulation (green house gases emissions and carbon storage in agricultural soils), pollination potential, water quality, and soil fertility (through the evolution of soil organic matter). We focus on the provision of ES that do not show spatial interactions, i.e. in economic terms, with constant returns to scale. We consider different land uses (grassland versus cropland) and agricultural practices, including (reduced) tillage, fertilisation, pesticide use, biomass input, and non-crop habitats, and assess the effect of these agroecological practices on the level of ES provision. By combining all these practices, we obtain 122 management options (2 for grassland and 120 for cropland) and the corresponding data set of ES bundles. We describe the synergies and antagonisms among these ES with usual techniques (correlation coefficients, production possibility set and frontiers) and show their limitations. As ES are interdependent, it may not be possible to find a strategy maximising each of them separately, and a desired strategy should be chosen by considering the joint provision of all ES. To do so, we use efficiency analysis to summarise these complex interactions and determine which management options provide ES efficiently. Last, we introduce the opportunity cost of providing bundles of ES, and determine cost-effective strategies to provide ES. As ES are provided as bundles, the cost of their provision cannot be attributed to individual ES. While the opportunity cost of ES is a concept used in the literature, we argue that given the multiple and complex interactions among ES, the calculation of the opportunity cost of individual ES makes little sense. We introduce the opportunity cost of providing a bundle of ES as a new way of measuring of the cost of providing ES. Representing the landscape as a homogeneous piece of land on which different agricultural management options are adopted with variable shares,[1] it is possible to compare two types of strategies to increase the provision of non-marketed ES and determine which type is the most cost-effective: (i) strategies consisting in adopting a single management option conciliating the provision of all ES, including agricultural production, on the whole landscape (i.e. a land-sharing strategy), against (ii) strategies consisting in combining different management options on the landscape, dedicating a part of it to the provision of non-marketed ES only (i.e. a land-sparing strategy).

Our approach is interdisciplinary and combines strengths from agroecological modeling and efficiency analysis. The links between changes in agricultural practices

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[1] We consider the composition of the landscape, but not its structure (spatial arrangement).

and the joint provision of several ES are encompassed within our integrated agroecological model. We present how the data from the model's output can be explored thanks to efficiency analysis tools, such as data envelopment analysis. Even if we used modeling data for an illustrative purpose, the tools we discuss could be applied to field agroecological data (or to the data of more realistic models). The main contribution of the paper is to advocate for the use of efficiency analysis tools to

- study the interactions among several ES, i.e. to describe the synergies and antagonisms among ES in a multidimensional way,
- to determine efficient land management options, i.e. the combinations of agricultural practices that produce the highest levels of ES
- to provide a clear-cut condition to determine which strategy is cost-effective to increase the provision of non-marketed ES: *(i)* adopting a management option conciliating the provision of all ES on the whole landscape, or *(ii)* dedicate part of the landscape to the provision of non-marketed ES only. This last result can straightforwardly be used to determine the cost-effective ways to mitigate climate change in agricultural landscape while supporting other biodiversity-based ES.

## Modeling framework

We develop a simple agroecological model to simulate the effect of agricultural practices on the provision of multiple ES, and study the interactions among these ES. The model is built on representations of various ecological dynamics borrowed from the agroecological modeling literature. It aims at synthesising in a convenient way the complex linkages between agricultural practices and ES, without relying on a specific case study. Its structure is also designed to easily simulate farmer's profit alongside ES provided by different agricultural management options (a feature which will be used in the "Cost-effective way to provide a bundle of ecosystem services" section).

In this section, we first present the model ("Agroecological-economic model" section) and then present the usual ways to study the synergies and antagonisms in the provision of ES ("Studying the interactions among ecosystem services" section), emphasising their limits.

### Agroecological-economic model

The agroecological model simulates agroecological processes to associate levels of ES to different agricultural management options. Management options correspond to combinations of agricultural practices; they influence the provision of ES by the agroecosystem.

The model is run on an agricultural area of any size (for a field, a landscape, or a small agricultural region), for an annual time period. The area is characterised by a given potential yield and initial soil organic matter content, considered homogenous over space.[2] Each agricultural management option can be adopted on a share of the

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[2] The case of agricultural regions with heterogeneous soil quality will be considered in future research.

agricultural area, in the spirit of land-use share models (Lichtenberg, 1989; Feng & Babcock, 2010; Lankoski et al., 2010; Lankoski & Ollikainen, 2011). When a single management option is adopted over the whole area, one gets a “homogenous” landscape. When several options are implemented in various proportions over the area, one gets a “heterogeneous” landscapes. Besides, we consider no interactions among agricultural management options (e.g. spatial spillovers or externalities), so that the bundles of ES provided by heterogeneous landscapes are linear combinations of the bundles of ES corresponding to each of the management option in the landscape.[3]

The model is designed and parametrised so as to represent an agroecosystem with arable crops and pasture, corresponding to the Northern half of France, flat areas. Given the assumption that the characteristics of the area are homogenous over space, the area can represent up to Small Agricultural Region.[4]

Our purpose is not to be highly realistic and fit a particular case-study, but rather to derive some general results using an economic approach to analyse the trade-offs among ES and the cost of their provision. As such, the model does not aim at accounting for precise representation of agroecological processes, but is intended to summarise patterns established in the agroecological literature as a support to an economic analysis. Our analysis includes the most important ecological processes for agriculture (reviewed for example by the Millennium Ecosystem Assessment, Cassman et al. (2005)). To avoid an inaccurate representation of processes that are too complex for the model, we exclude ES depending on the relief and hydro-geological structure of the landscape (flood control, water quantity regulation), on precise crop species (for example, the provision of genetic biodiversity by agroecosystems, or pharmaceuticals), or which depend on the spatial configuration of the landscape (biological control).

This model is thus a mean to formalise the links between agricultural practices and the provision of ES and to generate data to be used to illustrate the challenges of studying the potential dilemmas among agricultural production and environmental issues, such as climate change mitigation and the provision of other biodiversity-based ES.

### The model simulates the provision of five ecosystem services

The following ES are studied as outputs of the agroecosystem:

- *agricultural production*, aggregating the different agricultural outputs (grain, crop residues, fodder) by means of prices;
- *climate regulation*, through the accounting of greenhouse gases emissions due to agricultural practices and to the contribution of the agroecosystem to greenhouse gas exchange with the atmosphere (net emissions of greenhouse gases or net storage of carbon in agricultural soil);

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[3] The case of spatial interactions will be considered in future research.

[4] Small Agricultural Regions (SAR) correspond to homogeneous agricultural areas in France. The whole metropolitan area is divided into 713 SAR.

- the evolution of soil organic matter, which provides a proxy for both *soil fertility*, reflecting the potential to provide nutrients for current and future plants, and to *carbon storage in agricultural soil*, which is another metric of interest for climate change mitigation.
- *pollination potential*, i.e. the capacity of the landscape to offer suitable habitat and foraging resources to pollinators, and more generally to host wild insects;
- *water quality*, measuring the amount of pollutants potentially leached to water bodies (mineral nitrogen, pesticide residues, and organic particles).

There is a scientific debate over whether some of the mentioned ES can be considered as such (e.g. pollination or soil fertility), since they are intermediate agroecological processes delivering an indirect benefit to humans via an increase in the provisioning service (Haines-Young & Potschin, 2013). These ES benefit the farmer through production, but they can also benefit the whole society through other channels. For example, an increase in soil organic matter benefits the farmer through an increase of soil fertility, but it also benefits the society as it corresponds to carbon storage in agricultural soils (climate change benefit), the increase in soil biodiversity, and the reduction of water pollution (which potentially benefits aquatic biodiversity). The pollination potential benefits the farmers whose crops depend on pollination, but it also benefits society through the associated biodiversity preservation. We include all these desirable agroecosystem processes as "agroecosystem outputs" and refer to them as "ecosystem services" in our analysis, to acknowledge the fact that they are both desirable and under threat, and that they could be favored by agroenvironmental policies.

### Ecosystem services provision depends on agricultural practices

In European intensive agricultural landscapes, available alternative management options to enhance the provision of ES generally rely on changes in agricultural practices rather than converting land to a natural reserve, rewilding or reforestation. Such practices have a great impact of the delivery of ES, but are often overlooked by studies based on land use Bennett et al. (2009).

In our model, the drivers of ES provision are agricultural practices: we consider 122 possible management options, which are combinations of the following agricultural practices:

- *Land-use* (cropland/grassland): Land can be allocated either to cropland or grassland. Cropland is dedicated primarily to agricultural crop production. It can be more or less intensive, depending on the management options described below. Grassland stands for a more extensive land-use, without input of synthetic fertilisers, pesticides nor tillage. We consider two grassland options, with or without livestock, to account for the impacts of livestock on nitrogen input and methane emissions. Grassland are management options per se and exclude any other choice listed below.
- *Pesticide intensity* (three levels, including zero pesticides): Pesticides are harmful pollinators and degrade water quality, but they are beneficial to agricultural production by controlling pest.

- *Fertiliser intensity* (five levels, including zero fertiliser): Fertilisers bring mineral nitrogen and increase crop production, but induce nitrate leaching into water bodies and greenhouse gases emissions.
- *Non-crop habitat* (yes/no): Farmers can take actions to support biodiversity and good water quality by creating non-crop habitats at the margins of their fields (flower strips, buffer strips, or hedges). They decrease the cultivated area. In our analysis, we consider NCH covering 5% of the cultivated area.
- *Crop residue restitution* (yes/no): In addition to the biomass of roots and lower parts of crops that remain in the field after harvest and contribute to form new soil organic matter, farmers can also decide to increase fresh biomass inputs by leaving crop residues.
- *Tillage regime* (conventional/reduced tillage): Reduced tillage (or conservation tillage) avoids digging deep into the soil and disturbing the soil ecosystem. It contributes to a slower degradation of soil organic matter, and has thus a negative short-term impact on nutrient delivery, but a positive long-term impact on soil fertility and soil carbon storage. It also reduces soil erosion and green house gases emissions through reduced fuel burning and carbon release from the soil.

### The model is run on different agronomic contexts

In our analysis, which simulates ES provision for 1 year, the agronomic context is assumed to be exogenous. This context is characterised by two variables having an impact on the provision of ES: soil quality and the stock of soil organic matter. Soil quality is an exogenous parameter $Q$ capturing all the characteristics that have an impact on the yield but cannot be modified by farmers' decisions (soil mineral composition and depth, slope, local climate and precipitations, etc.). It is defined in the model as potential yield, in $t\ ha^{-1}$, and calibrated to represent the observed range of potential yield for soft wheat in France. Soil organic matter is a state variable which is influenced by past agricultural practices, but considered as given ("initial stock") when the model runs for one period. In the following, we loosely use the term "agronomic context" to refer to these two variables.

To represent the fact that soil organic matter is inherited from past management decisions, we assume that the initial value of the stock of soil organic matter depends on soil quality. More precisely, the initial stock of soil organic matter is supposed to reflect the equilibrium stock in a situation where the prices and costs are stable over a long period, and farmers maximise the discounted value of intertemporal profits. Soil quality affects the yield and profit associated to agricultural practices and hence the agricultural practice chosen by the farmer, which plays a role in the evolution of the stock of soil organic matter and its long-term value. We solve this intertemporal maximisation problem using a Bellman algorithm for a range of soil quality, and use the corresponding soil organic matter equilibrium as a reference value for soil organic matter in the corresponding context.

To assess the sensitivity of our analysis to the context, we consider a range of agronomic conditions and run the simulations on 10 different agronomic contexts detailed in Appendix B. The predicted values for soil organic matter reproduce a range of observable situations in agricultural landscapes in Northern temperate regions, and are close to estimates of soil organic matter content mentioned in the gray literature (Conseil des Productions Végétales du Québec, 2000). For example, this representation predicts that soils with a good potential yield (higher than 5.5t/ha) have a relatively low percentage of soil organic matter (1.7%), because the most profitable option on good soils is intensive cropland, which leads to a low stock of soil organic matter. Conversely, on poor soils, grassland is the most profitable option, and thus the stock of soil organic matter is higher.

### Mathematical representation

The mathematical model representing the links among decision variables, exogenous parameters, and output indicators is based on equations borrowed from existing models in the agroecological literature. We detail the main traits of the model here. A complete description can be found in Appendix A.

The stock of soil organic matter evolves over time according to inflows (fresh biomass) and outflows (mineralisation, soil erosion). This representation is build upon the Hénin-Dupuis model (Hénin & Dupuis, 1945). Similar equations are used by Lifran et al. (2014) to study soil natural capital. The magnitude of both flows of soil organic matter depends on agricultural practices: crop residue restitution increases fresh biomass and reduces soil erosion; tillage increases mineralisation and soil erosion. The stock of organic matter is a proxy to carbon storage in agricultural soil, in line with the climate change mitigation objective.

Mineralisation of soil organic matter delivers carbon dioxide, which is one of the greenhouse gases we account for, and mineral nitrogen, which is then used by plants to grow. Available nitrogen from mineralisation can be completed by external inputs (synthetic fertilisers on cropland, manure on grassland with livestock). Crop yield is represented by a modified Mitscherlisch-Baule function with nitrogen as the only limiting input (if nitrogen is available in unlimited amount, the yield reaches its potential). Pest damage is a fixed proportion of potential yield, and can be reduced by pesticides, which show diminishing marginal efficiency.[5]

Plants take up only part of available nitrogen, which follows a linear function with plateau: above a certain threshold, no more nitrogen is taken up. Following the methodology proposed by IPCC (2006c), a fixed proportion of available nitrogen is emitted as nitrous oxide, another greenhouse gases we account for, and the remaining is assumed to be ultimately leached to water bodies.

Nitrous oxide is one of the greenhouse gases. Following the guidelines of IPCC for cropland and grasslands (IPCC 2006a, b), we count the emissions due to changes in soil carbon stock (proportional to soil organic matter), fossil fuel burned by farm machinery, and methane from livestock. Practices decreasing the stock of soil organic matter contribute to emit carbon dioxide. Conventional tillage contributes more to fossil fuel burning, and thus is assumed to emit more greenhouse gases.

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[5] Studying biological control is out of the scope of the present paper. It would require to include spatial spillovers and to consider the spatial structure of the landscape. Such an investigation is part of future research.

Water quality is an index which accounts for the presence of three agricultural pollutants: nitrogen, pesticides, and organic particles. The pollutants are expressed in terms of the worse situation and aggregated with a limiting factor approach, to avoid any substitution between pollutants and reflect the logic of water quality norms. This kind of aggregated index of water quality has been used by Canadian Council of Ministers of the Environment (2001). Non-crop habitats (NCH) limit the export of pollutants and facilitate their degradation, and thus increase water quality.

Pollination is modeled by an index which depends on agricultural practices (land-use, pesticides, non-crop habitat). We use the methodology proposed by Lonsdorf et al. (2009) and used later by Zulian et al. (2013), but without the spatial interactions over the landscape. The parameter values are taken from the appendix of Zulian et al. (2013): grasslands are the best-suited land-use; non-crop habitats are also beneficial. The impact of pesticides is added: we assume that they decrease pollinator potential by killing them and eliminating their forage.

In addition to the agroecological part of the model linking agricultural management to bundles of ES, the model also simulates the profit associated to each management option in each agronomic context (see "Cost-effective way to provide a bundle of ecosystem services" section).

Model calibration is detailed in Appendix A.3. Simulations are run with Matlab.

The agroecological model gives us the bundles of ES provided by each of the 122 management options, in different agronomic contexts. It is the (simulated) data set we use to study the interactions among ES and the influence of agricultural practices on them in the next section. We conduct the analysis on these simulated data, but the same could be done using data from another model or field data. No such comprehensive data set is available, however, and simulations allow us to explore all combinations of agricultural practices and their implications on the provision of multiple ES.

## Studying the interactions among ecosystem services

Ecosystems are defined through the interactions among living organisms (biodiversity) and with their environment, and thus ES exhibit multiple and complex links such as antagonisms and synergies (Bennett et al., 2009). The design of agro-environmental policies to mitigate climate change should take these interactions into account to target management options that contribute to the reduction of greenhouse gases emissions and carbon storage in agricultural soil while avoiding unwanted side effects on other ES. Our model aims at studying these interactions among ES.

Many assessments of multiple ES in agriculture areas have been carried out, either to compare the provision of ES by different areas and design conservation schemes, to assess to what extent targeting the provision of one ES also ensures high levels of other ES, or to assess the bundle of ES associated to different land-uses or land management options. Describing and representing the results of these assessments, i.e. multiple ES in different scenarios, locations, or management options, is a challenge and several methods are used in the literature. For example, visual methods can be used to represent different bundles of ES, such as flower diagrams (Raudsepp-Hearne et al., 2010) and maps (Nelson et al., 2009; Goldstein et al., 2012). They, however, do not make the comparison of different alternatives easy nor provide a clear view of the interactions among several ES.

A direct way to represent interactions among ES and to compare alternative bundles is to refer to production possibility sets, by considering a multidimensional space in which each dimension represents the level of provision of an ES. These sets can be represented graphically by plotting several bundles on a diagram which axes are levels of ES only for two (or three) ES, but conceptually production possibility sets can include as many dimensions as necessary. Numerical methods can then be used to analyse these sets.

In this section, we discuss some of these methods and use them to describe our simulated data. We then emphasise that it is difficult to go further than a descriptive analysis with these methods and advocate for the use of efficiency analysis tools for the identification of strategies maximising multiple ES.

### Correlation between ES

A first solution to study the joint provision of various ES is to assess *correlation coefficient* among pairs of ES, which informs on the general trend followed by one ES when another increases (see, e.g. Chan et al. 2006; Naidoo et al. 2008 who calculate correlation coefficients among ES using the level of ES in different locations of a landscape).

To gain a first insight into the interactions among ES in our model, we calculate the correlation coefficients among ES in our set of 122 bundles for one agronomic context (i.e. $Q = 9.9\ t\ ha^{-1}$, corresponding to context 8 described in Appendix B) are shown in Table 1. The model behaviour is similar in the other contexts. There is an antagonism (negative coefficient) between production and non-marketed ES (pollination, water quality, climate regulation, and soil fertility), and synergies (positive coefficient) among non-marketed ES. These features comply with general findings in the literature (Lee & Lautenbach, 2016), even if comparisons are limited by the fact that few quantitative multi-ES assessments involve the same ES and that different studies often do not compare the same alternatives. It emphasises that our modeling assumptions are somehow sensible.

However, correlation coefficients only show the general trend of the interactions among ES and rely on the assumption of a linear relation among the ES. Drawing directly production possibility sets and looking at their shape is sometimes more

**Table 1** Correlation coefficients among ES as simulated in the model

| ES | Pollination | Water quality | Climate regulation (GHG emissions) | Soil fertility (incl. carbon storage) |
|---|---|---|---|---|
| ag. production | −0.46 | −0.46 | −0.48 | −0.24 |
| Pollination | | 0.49 | 0.25 | 0.22 |
| Water quality | | | 0.39 | 0.31 |
| Climate regulation | | | | 0.86 |

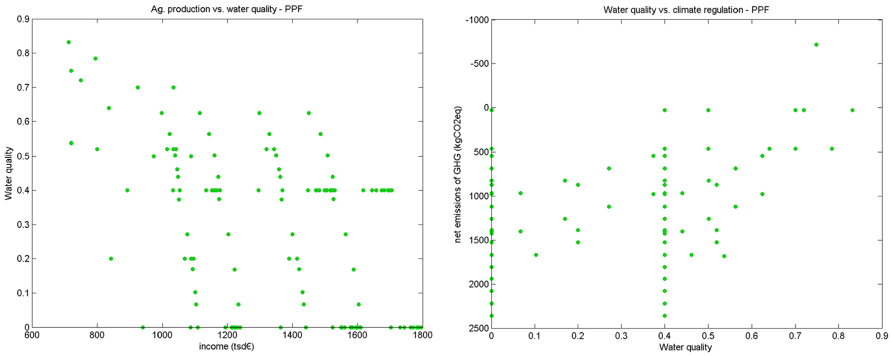


**Fig. 1** Illustration of interactions between ES with the shape of two production possibility sets resulting from the simulations: on the left, the antagonism between production and water quality and on the right the synergy between water quality and climate regulation

informative about the interaction among ES, especially when it is non linear (Bekele et al., 2013; Vallet et al., 2018). For example, the antagonism between production and water quality and the synergy between water quality and climate regulation in our data set are visible on Fig. 1, which moreover shows that the bundles cover a large range of ES provision levels.

These diagrams already show quite complex inter-dependencies which cannot just be summarised by determining synergies and antagonisms among pairs of ES. They show two dimensions, but hide the other dimensions; hence, it is impossible to know the other ES provided by a given bundle. As a consequence, the information drawn from these 2-dimensional cuts of the production possibility set or from correlation coefficients is not sufficient to determine which management options provide more of the ES.

## The production possibility frontier

Alternatively to the correlation coefficients or the shape of the production possibility set, interactions among ES can be assessed by the formal analysis of production possibility frontiers. This frontier connects the bundles of ES with a maximal provision of ES. It corresponds graphically to the envelope of the production possibility set. Bundles belonging to the frontier are efficient in a Pareto sense: there is no other bundle that achieves better on all ES simultaneously. Thus, among efficient bundles, increasing one ecosystem service requires to decrease at least another one. While correlation coefficients or the observation of the shape of the production possibility set consider all the bundles of ES, the production possibility frontier focuses on efficient bundles. Thus, the assessed interactions are not exactly the same in both cases. The observations on the frontier represent bundles that maximise ES, and thus its shape represents unavoidable trade-offs that cannot be solved by reducing inefficiency (Lester et al., 2013).

Three characteristics of the production possibility frontier are useful to study antagonisms and synergies among ES: its length, its slope, and curvature. They lead

**Fig. 2** PPF exhibiting a **concave** antagonism between ES, where intermediate management options dominate combinations of extreme options

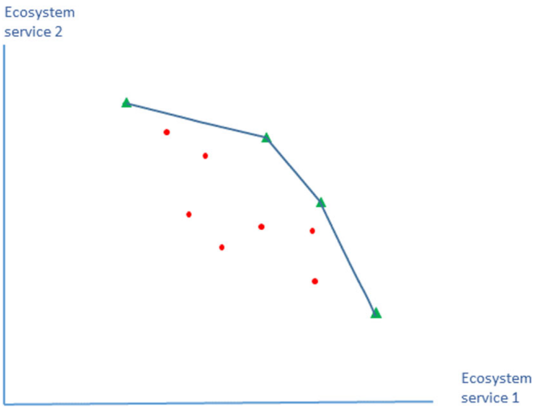


**Fig. 3** PPF exhibiting a **convex** antagonism between ES, implying combinations of extreme options dominate intermediate options

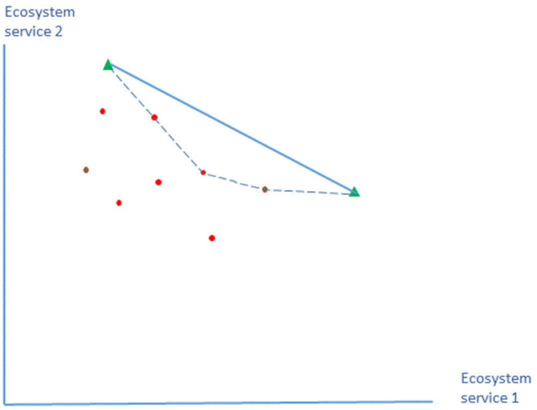
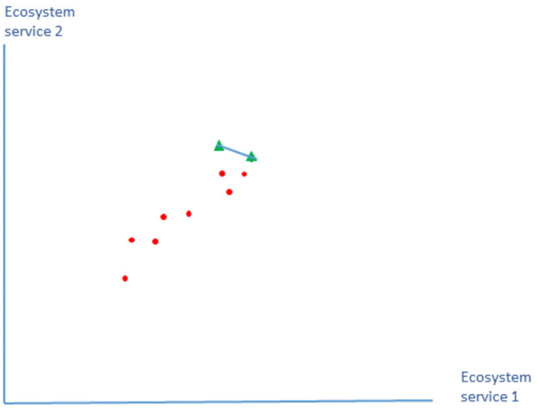


**Fig. 4** PPF exhibiting a synergy between ES. Illustrative examples of the different shapes of production possibility frontiers (PPF). In these figures, bundles of ES are represented with red dots and efficient bundles with green triangles. The PPF joins the efficient bundles



to distinguish three different shapes, corresponding to a concave antagonism, a convex antagonism, or a synergy. These three cases are illustrated schematically in Figs. 2, 3, and 4.

A synergy is a globally positive relationship among several ES, while an antagonism is a negative relationship. A synergy implies that maximising one ES ensures the maximisation of the other(s) and hence there is no major difficulty in defining which strategy maximises the provision of ES. It is characterised by a very short frontier composed of close bundles. On the contrary, an antagonism among ES (downward-sloping, long frontier) calls for a trade-off. In this case, the slope and curvature of the production possibility frontier inform respectively on the strength of the antagonism and the strategy to maximise the provision of ES.

Many authors use production possibility frontiers in the literature on the optimal provision of ES (see the discussion in Vallet et al., 2018). They generally estimate a functional form for the frontier and represent it explicitly, and then study its slope and curvature to assess interactions among ES. As synergies do not pose much problem in the maximisation of ES, the literature tends to focus on issues related to antagonist ES.

**Slope of the production possibility frontier.** This slope indicates how strong the antagonisms are. Wossink and Swinton (2007) conceptually relate the slope of the frontier to the strength of regulation needed to provide ES: the steeper the slope of the frontier between production and non-marketed ES, the larger the loss farmers bear to increase the latter, and thus the stronger the incentives need to be in order to convince them. For mild antagonisms or even synergies, information campaigns are enough, while strong antagonisms require compensation payments. Sauer and Wossink (2013) apply this framework to a case study by estimating production possibility frontiers in England over fields included or not in agroenvironmental schemes. They assess whether the commodity production and the provision of non-marketed ES stay in synergy or in an antagonism and what the marginal costs of providing non-marketed ES are (the monetary value of production lost when increasing non-marketed ES). In a rather similar approach, Ruijs et al. (2013) determine and estimate the production possibility frontier of agricultural production, carbon sequestration, cultural ES, and biodiversity for an area spanning several East-European countries. Their objective is to measure the foregone production associated to an increase of each non-marketed ES across the studied area, and to find the areas where the trade-off is the least severe, and where the provision of the ES should be enhanced. Ruijs et al. (2017) analyse the opportunity cost of ES provision using a semi-parametric technique to construct multi-dimensional frontiers using data provided by biophysical models.

**Curvature of the production possibility frontier** Since many authors find an antagonism between marketed agricultural output and non-marketed ES or biodiversity, the issue is to find the best compromise between contradictory objectives. One question is to assess whether both types of outputs should be provided together or separately, and relates to the curvature of the frontier. Whether the frontier is concave (outward-bending) or convex (inward-bending) informs on the optimal strategy, as described in the land-sparing/land-sharing literature (Green et al., 2005).

When the production possibility frontier is convex as in Fig. 3, linear combinations of extreme bundles dominate intermediate bundles, meaning that a linear combination of extreme strategies provides more ES than strategies conciliating the provision of several ES jointly. Such a linear combination is interpreted as the division of the landscape into different areas each devoted to a land use performing very well with respect to the provision of one ES. For example, Phalan et al. (2011) find a convex relationship between food production and biodiversity preservation, and suggest to produce food as intensively as possible on a small amount of land in order to spare as much land as possible for wild nature. This is the *land-sparing* strategy, which provides both more commodities and biodiversity than a biodiversity-friendly agricultural land use adopted on the whole landscape (the *land-sharing strategy*).

The opposite conclusion emerges when the production possibility frontier is concave (outward-bending) as in Fig. 2. Finding a concave frontier, Polasky et al. (2008) conclude that the trade-off between marketed and non-marketed outputs (e.g. biodiversity) is less severe for high levels of production. Increasing biodiversity protection implies less yield loss in productive agricultural areas than in less productive areas. This favors the *land-sharing* strategy: marketed production can be combined with the provision of non-marketed outputs on the same land, and extreme bundles of outputs should be avoided unless very unbalanced preferences exist (i.e. the desired bundle of ES is composed of much of one ES and very little of the others).

These results hold only when there are no interactions among neighbour land uses. In this case, mixed strategies at a landscape scale provide a linear combination of the associated bundles of ES in the proportions in which the land uses are implemented. This is not the case when there are spatial spillovers, neighbouring effects, or size effect etc. Indeed, in the case of spatial interactions among land uses, the land use in one area determines the ES in that area but also in neighbouring areas, and the bundle of ES resulting from a patchwork of land uses depends not only on the area covered by each land use, but also on their spatial arrangement. In this case, not all linear combinations of bundles of ES may exist (Brown et al., 2011) and, if they exist, it is not straightforward to determine which arrangement of land uses could provide them. Therefore, the use of the curvature of the frontier in the land-sharing/land-sparing debate should be restricted to spatial scales or ES for which no interactions among land uses occur (Kremen, 2015). This is the case in our model.

Altogether, the analyses mentioned above focus on the description of interactions among pairs of ES and derive implications for the maximisation of their provision. They mostly rely on the graphical representation of the production possibility frontier. They enable to draw interesting conclusions about strategies to maximise ES provision, but it becomes difficult to extend them to more than two or three ES. Approaches studying the provision of all ES jointly are needed. For this purpose, more general quantitative methods such as efficiency analysis can be used.

## Efficiency and cost-effectiveness analyses

In this section, we study the interactions among ES in our data set through efficiency analysis, and more specifically data envelopment analysis. Compared to other multicriteria decision tools, efficiency analysis has the advantage to rely on raw data without any need to aggregate ES or to simplify their interactions by considering them two-by-two. This is particularly interesting since any aggregation implies assumptions over which ES should be prioritised and influences the results in a partly arbitrary way. Besides, it is also possible to analyse all dimensions altogether, compared to correlations which are always pairwise. Efficiency analysis is also an interesting tool to select management options maximising the environmental outcomes (Ferraro, 2004). We first describe data envelopment analysis ("Data envelopment analysis" section). We then use it to study the shape of the production possibility frontier and interpret the results in terms of strategies to maximise the provision of ES ("Efficient bundles of ecosystem services" section). We last consider the cost of implementing alternative management options for farmers, and provide a rule to determine cost-effective strategies to provide a given bundle of non-marketed ES ("Cost-effective way to provide a bundle of ecosystem services" section).

## Data envelopment analysis

Efficiency analysis techniques such as data envelopment analysis rely on production theory in economics, which puts a theoretical framework on the transformation of inputs into outputs. Here, we interpret the provision of ES — including agricultural commodities — by agroecosystems as the process of producing ES ("outputs" in the production theory terminology). This production relies on land ("input" in the production theory terminology), which is allocated to different management options (different "production processes" in the production theory terminology). The production possibility set corresponds to the various bundles of ES which can be produced with a given amount of land, each bundle corresponding to a different management option.

As the rationale of our analysis is to assess which management option maximises the provision of ES on the available land, we shall first focus on the provision of ES by the landscape/agroecosystem. Land is considered as the only input at the landscape scale. In the "Efficient bundles of ecosystem services" section, labor, capital, pesticides, or fertilisers belong to the different technologies: we don't seek to minimise them per se, and their detrimental consequences are already embedded in the ES provided by each management option. For example, a management option characterised by heavy use of pesticides will correspond to a bundle of ES with a higher production but lower levels of water quality and pollination. In a second time ("Cost-effective way to provide a bundle of ecosystem services" section), we shall consider these other inputs through their influence on agricultural profit, within our cost-effectiveness analysis.

Data envelopment analysis is an appropriate tool to answer this question, as its principle is to find for each bundle of ES (called an *observation* in the data envelopment analysis framework) to what extent the outputs (the ES) could be increased by using the input (land) differently, while staying inside of the production possibility set[6] (Coelli et al., 2005). It is a non-parametric technique, and thus imposes no functional form on the data.

Among the possible specifications, we choose a directional data envelopment analysis whose direction is the evaluated observation. This means that we examine, for each management option $k$ and the associated bundle of ES, denoted by the vector $Y_k$, if a linear combination of other management options performs better in terms of ES provision in the sense that it makes it possible to increase the production of all the ES by the highest possible proportion or, equivalently, to produce the same bundle of ES with as less land as possible. The resulting *inefficiency score* $\beta_k$ is interpreted as a potential proportional increase in all ES.[7]

Formally, this is done through a linear optimisation problem under constraint on the production level. For each management option (observation) $k = 1, \ldots, N$, the optimisation problem reads

$$\begin{aligned} \max_{\mu_i} \quad & \beta_k \\ \text{s.t.} \quad & \sum_{i=1}^{N} \mu_i Y_i \geq (1 + \beta_k) Y_k \end{aligned} \tag{1}$$

where the vector $Y_i$ stands for the bundle of ES provided by the alternative options $i$. The right-hand side of the constraint has the proportional form $(1 + \beta_k)Y_k$ where $\beta_k$ represents the *inefficiency score* associated to observation $k$ and is expressed as a percentage by which all ES could be increased at the same time with respect to the observed vector $Y_k$. For the efficient observations, it equals 0.

Each observation $k$ is associated to an efficient benchmark, which is the linear combination of other management options $i = 1, \ldots, N$ producing the efficient bundle $(1 + \beta_k)Y_k$. The optimal share of each alternative option $i$ is given by the shadow-value $\mu_i$. All ES are jointly produced by the combined management options, and the resulting bundle is the weighted sum of the bundles $Y_i$.

Using this approach, **we perform two analyses** on the 122 simulated bundles of ES. First, we run the data envelopment analysis on the five ES (agricultural production and the four non-marketed ES) to find out which ones are efficient and describe the overall interactions among our set of ES, and in particular the necessary trade-offs between production and other services. Second, we run the data envelopment analysis on the four non-marketed ES only, excluding agricultural production. This allows us to examine the interactions among the non-marketed services and determine if they can be provided jointly or not. This also allows us to determine cost-effective ways to provide given bundle of non-marketed ES. Detailed outputs are presented in the Appendices D and E.

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[6] This set is understood here as the space delimited by the linear combinations of all bundles. This definition of the production possibility set is somewhat particular, but we explain it in the "Efficient bundles of ES and associated management options" section below.

[7] This specification is invariant to translations, which allows us to translate the values for climate regulation and soil fertility in order to get rid of negative values.

## Efficient bundles of ecosystem services

### Shape of the production possibility frontier and trade-offs among ES

Data envelopment analysis allows us to identify the efficient agricultural management options and associated bundles of ES among the 122 simulated bundles in our model. Efficient bundles are the ones that maximise the provision of ES on a given agricultural area, in the sense that no other (combination of) management option(s) produces more of all the services on the same area. Thus, they can be considered as the management options that make an efficient use of scarce land.

The number and relative position of efficient bundles of ES characterise the shape of the production possibility frontier. A large number of efficient bundles indicate a concave antagonism (see Fig. 2). A small number of efficient bundles indicate either a synergy if the efficient bundles are quite similar in terms of ES provision (Fig. 4), or a convex antagonism if the bundles are quite different in terms of provided ES (Fig. 3).

By doing the efficiency analysis on a subset of ES, it is possible to investigate further which ES are in synergy and in antagonism: if removing one ES from the analysis strongly reduces the number of efficient bundles, this ES was standing on a concave antagonism with the other(s).

As mentioned above, we run the analysis twice, first with all five ES (including production) and then only with the four non-marketed ES. From our results, we can state that

1. Only few bundles are efficient when considering the five ES, so that most of the management practices are not: at least 100 out of 122 bundles are inefficient. This means that some options provide a higher level of all services than most options, which indicates room for efficiency and the possibility to improve jointly the provision of all ES with respect to inefficient management options.
2. While the number of efficient bundles is rather restricted, they cover a large range of agricultural practices and ES levels, illustrating that many different compromises between agricultural production and other ES are possible. Among efficient bundles of ES, some show intermediate levels of all ES. This is an indication that the relationship between agricultural production and the other ES is rather concave, and that some management options conciliating production and non-marketed ES are efficient.
3. When considering only non-marketed services (i.e. excluding agricultural production from the analysis), there are only two efficient options with rather similar levels of ES. Non-marketed ES are maximised by grassland and by the least intensive cropland, with reduced tillage, low fertilisation and pesticide use, biomass input, and NCH. This illustrates the synergy among non-marketed ES, which are jointly produced by the same agricultural practices. It also illustrates the general antagonism between production and other ES. This confirms the interpretation of the correlation coefficients and the shape of the production possibility set in the "Correlation between ES" section.

These results apply for all the agronomic contexts we analysed (see Appendix D), although efficient bundles differ among contexts. More precisely, results differ between contexts with high and low potential yield. In contexts with low potential yield, grassland has a higher production than many cropland options while also providing much more non-marketed ES. Hence, grassland is efficient compared to many cropland options, including intensive ones, and far less options are efficient. However, all efficient options in low-yield contexts are also efficient in high-yield contexts.

### Efficient bundles of ES and associated management options

The analysis of the slope and curvature of production possibility frontier enables us to describe interactions among ES and to determine strategies to maximise their provision, as presented in the "The production possibility frontier" section. Many management options lead to inefficient bundles of ES, so that despite the general antagonism between production and non-marketed services, non-marketed services can be increased sometimes without yield loss, for example through agroecological practices.

In our model representing homogenous agricultural areas, any agricultural management option can be adopted on any proportion of the area: a landscape is modeled as a linear combination of management options. Each management option, if implemented on the whole area (thus creating a homogeneous landscape), provides one of the 122 simulated bundle of ES. In the case in which several management options are adopted over the landscape (heterogeneous landscape), given the absence of interactions among management options in our model, the bundle of ES provided is the linear combination of ES bundles provided by the management options composing the landscape. All possible linear combinations of management options and thus of ES bundles are allowed. Data envelopment analysis considers all these linear combinations of ES bundles as belonging to the production possibility set. Running data envelopment analysis enables to compare each management option to any linear combination of other options, and thus to consider all possible landscapes.[8] The efficient benchmark matching an inefficient option is an heterogeneous landscape providing more ES in the same proportion. The efficient bundles, as well as the efficient benchmark of any bundle, represent the efficient ways to conciliate agricultural production and the provision of non-marketed ES.

Each (combination of) management option(s) provides a bundle of ES, where the various ES stand in different proportions. Overall, a large range of ES proportions is covered by all our management options, up to extreme orientations with very low or very high agricultural production. Efficiency analysis isolates efficient bundles and (combinations of) management options, but makes no assumption over which of the efficient bundles or which proportions of ES are better. This depends entirely

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[8] Data envelopment analysis determines the efficiency scores only for homogeneous landscapes. Heterogeneous landscapes only serve for determining the efficiency of homogeneous landscapes

on social preferences, that is, the relative importance given to the different ES in social choice.

The previous efficiency analysis identified which options maximise the ES provided by a given amount of land. However, one criteria is not accounted for: the cost of providing non-marketed ES. Hence, efficient bundles of ES may not be cost-effective. This is what we examine in next subsection.

### Cost-effective way to provide a bundle of ecosystem services

Changing the bundle of ES has a cost due to yield loss or the extra cost of alternative agricultural practices. Even if land is scarce, this cost is likely to be more limiting than land, whether it is supported by the farmer or by public budget via subsidies. As a consequence, to focus on realistic strategies, the opportunity cost should be considered as a criterion to minimise along with the maximisation of the provision of non-marketed ES, as shown by Naidoo et al. (2006). We now explore cost-effective strategies to provide non-marketed ES, computing the opportunity cost of bundles of ES.

### The opportunity cost of bundles of ecosystem services

In economics, the opportunity cost is defined as the monetary loss incurred when giving up a profitable option. For farmers, changing agricultural practices is likely to cause a loss of profit, either because of additional costs (e.g. implementing a hedge) or because of a lower yield. More precisely, if we assume that farmers behave as rational economic agents and choose the most profitable management option,[9] the *statu quo* is the most profitable option, and any change in the bundle of ES induces a cost. This cost corresponds to the profit gap compared to the most profitable management option, and is supported either by the farmer or by the rest of the society when it is compensated by subsidies. The way of sharing this cost does not change the cost itself, so that from the society's point of view it is interesting to seek to minimise this cost.

The notion of opportunity cost has been used in the literature to measure the cost of providing non-marketed ES. For example, Ruijs et al. (2013) express the foregone production related to an increase in one ES in monetary units by means of the crop price. The same approach is followed by Bostian and Herlihy (2014) to value the trade-off between production and an index of wetland condition (see also Ruijs et al. (2017)). This way to define opportunity cost is, however, raising two issues.

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[9] This is of course an approximation, as farmers may consider other criteria than profit (working time, tediousness, environmental preferences, etc.) or behave sub-optimally. We, however, consider the maximisation of profit as a rather good approximation of farmer's behaviour for our research question, and it corresponds to the logic behind common agroenvironmental policies in the EU.

First, these authors only look at the foregone production, whereas the opportunity cost is defined as the profit loss and hence does not depend only on the revenue stemming from production but also on the management costs. In the end, the opportunity cost of a more productive option could be positive because of increased costs (e.g. fertiliser use). To overcome this limit, we consider the difference in profit.[10] Second, the several ES are provided as bundles by common agroecological processes, so that it is impossible to attribute the opportunity cost to the level of one ES in particular. The opportunity cost depends directly on the agricultural practices which provide a whole bundle of interdependent ES, not separated ES. This issue is well known in economics, in the case of joint production: Baumgärtner et al. (2001) state that "From the firm's point of view, the allocation of costs between joint products is essentially arbitrary", and it is also the case of ES provided by a landscape. As a consequence, we propose to consider the opportunity cost of **bundles** of ES.

**Definition [Opportunity cost of bundles of ES]** *We define the opportunity cost of each bundle of ES as the difference between the gross margin of the corresponding management option and that of the most profitable option* (statu quo).

By considering the associated change in the bundle of ES, we can conduct a cost-effectiveness analysis over the possible bundles of ES.

### Simulation of the opportunity cost

Our model allows us to compare the gross margin of all (combinations of) management options: it equals revenues from the sale of agricultural products (fodder, crop, crop residues) minus management costs which depend on the agricultural practices. Revenues equal production times an exogenous price, for each type of production. Each management option has a different agricultural production and different management costs, and thus a different gross margin. Prices and costs have been calibrated based on aggregate and farm-level data from the North of France. Appendix C shows the simulated profit of each option, in a context with good potential yield (i.e. $Q = 9.9\ t\ ha^{-1}$, corresponding to context 8 Appendix B).

For farmers maximising profit, we can define the *statu quo* as the most profitable management option, which depends on the agronomic context. Because of the management costs, production and profit are not perfectly correlated, and hence the most profitable management option is not necessarily efficient in terms of ES provision (including agricultural production).

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[10] This is also in line with the principles of agroenvironmental subsidies in the EU, which aim at compensating foregone profit, encompassing both reduced production and additional costs incurred by the agricultural practices.

### Most profitable management options:

- For agronomic contexts characterised by low potential yields (i.e. contexts 1 to 5 in Appendix B, corresponding to potential yields up to $Q = 6.6\ t\ ha^{-1}$ for our parameters values), the most profitable management option corresponds to Grassland without livestock (management option # 1 in Appendix C). This option is efficient in terms of ES provision.
- For agronomic contexts characterised by high potential yields (i.e. contexts 6 to 10 in Appendix B, corresponding to potential yields above $Q = 6.6\ t\ ha^{-1}$ for our parameters values), the most profitable management option corresponds to a quite intensive cropland, with tillage, no agroecological practice (biomass input, NCH), pesticide use, and more or less fertilisation depending on the potential yield (management option # 47 for contexts 6 to 8, with limited fertilisation, and management option # 71 for contexts 9 and 10, with higher fertilisation). These profit-maximising options are not efficient in terms of ES provision.

These results underline the fact that improving ES provision may not be possible in some contexts (e.g. in areas where grassland is the most profitable option), whereas improving ES in more productive contexts is feasible but induce a cost and requires agroenvironmental policies. The cost of providing additional ES with respect to the *statu quo* option, through a given (agroecological) management option, is then determined by the opportunity cost of this option.

In the following, we focus on agronomic contexts with rather high potential yields, where the most profitable option is not efficient, and explore what are cost-effective ways to provide more ES. Adopting an agroenvironmental perspective, we consider only options providing more non-marketed ES than the *statu quo* and exclude the few management options providing less non-marketed ES than the *statu quo*.

### Comparing the cost of different strategies to improve the provision of non-marketed ecosystem services

We aim at determining cost-effective ways to increase non-marketed ES provision with respect to the most profitable option. For each bundle of ES, we determine the cost-effective strategy to provide (at least) that level of non-marketed ES. This analysis is useful to identify strategies that maximise the provision of ES at the lowest possible cost.[11]

As the opportunity cost of a bundle is defined as a difference in terms of profit with respect to the *statu quo*, most profitable option, for consistency we shall now consider changes in the provision of ES relatively to the levels provided by the *statu*

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[11] The current European budget for agro-environmental policies is too small to cover all the land concerned by their implementation. Over the period 2007–2012, only 25% of the agricultural area was covered by agro-environmental schemes in the EU (Duval et al., 2016), although maximising the provision of ES probably means enrolling a greater area.

*quo* management option. Each management option thus corresponds to an alternative to the *statu quo* characterised by an opportunity cost and a variation of ES provision.

The efficiency analysis of the "Efficiency and cost-effectiveness analyses" section identified that only two management options maximise non-marketed ES provision: grassland and the least intensive cropland. Thus, all other options are inefficient to provide non-marketed ES, in the sense that the same bundle of ES could be provided by a combination of the two efficient options on a smaller area. To improve the non-marketed ES provision of a given area, there are thus two strategies:

- adopting an ES-improving management option on the whole area (akin to a land-sharing strategy, in which all the ES are provided jointly by a homogenous landscape), or
- dedicating part of the landscape to the provision of non-marketed ES, and leaving the rest under the statu quo (akin to a land-sparing strategy, in which parts of a heterogeneous landscape are specialised in the provision of non-marketed ES).

For each management option, which of these strategies induce the lowest opportunity cost for a given increase in non-marketed ES? Depending on the relative costs of the two strategies, it is possible that even if implemented on less land, the efficient alternative is more costly to provide the same amount of ES.

To answer this question, we run again a data envelopment analysis with the four non-marketed ES, but expressing the provision of ES as a difference to the *statu quo* levels.[12] The data envelopment analysis identifies the same bundles maximising non-marketed ES as previously (grassland and the least intensive cropland), but above all it defines an *efficient benchmark* for every other (inefficient) management option, as well as an efficiency score.[13] The *efficient benchmark* of a management option $k$ is the efficient combination of the two efficient options (grassland and extensive cropland) that increase the ES with respect to the *statu quo* in the same proportions than the inefficient management option $k$. The efficiency score $\beta_k$ is the proportion by which the increase in ES can be enhanced, or equivalently, $1/(1+\beta_k)$ is the area needed to achieve the same improvement with the efficient benchmark (the rest of the area remaining under the *statu quo* management option, incurring no opportunity cost and no change in the ES provision).

Adopting a given management option $k$ on one unit of land is associated to an opportunity cost $C_k$ and change in ES provision $\Delta\text{ES}_k$, while adopting the efficient alternative on 1 unit of land costs $C_e$ and provides $(1+\beta_k)\Delta\text{ES}_k$ (second and third columns of Table 2). To compare both solutions for a given increase in ES, we con-

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[12] We thus run a directional data envelopment analysis which direction is given by the variation of non-marketed ES of each option with respect to the *statu quo*.

[13] Efficient bundles are the same as in the previous data envelopment analysis run, only the scores and composition of efficient benchmarks change, because we now consider differences to the *statu quo* and not absolute levels of ES.

**Table 2** Opportunity cost and ecosystem services provided by an option and its efficient alternative

| | Homogeneous landscape option $k$ | Efficient benchmark | Heterogeneous landscape efficient alternative providing as much ES as option $k$ |
|---|---|---|---|
| Land use shares | $\mathrm{LU}_k$ on 1 unit of land | $LU_e = \sum \mu_k LU_k$ on 1 unit of land | $LU_e$ on $1/(1+\beta_k)$ unit of land (*statu quo* on the rest) |
| Cost | $C_k$ | $C_e = \sum \mu_i C_i$ | $C_e/(1+\beta_k)$ |
| Ecosystem services difference with the *status quo* | $\Delta\mathrm{ES}_k$ | $(1+\beta_k)\Delta\mathrm{ES}_k$ | $\Delta\mathrm{ES}_k$ |

sider the adoption of the efficient alternative on $1/(1+\beta_k)$ units of land (fourth column of Table 2),the rest of the land remaining in the *statu quo* (profit-maximising option) with no cost and no change in the ES provision.

To assess which solution is least costly to provide ES, we compare the cost of option $k$ (i.e. $C_k$) and the cost of its efficient alternative providing the same amount of ES (i.e. $C_e/(1+\beta_k)$). It leads to the following result, which offers a decision rule to the adoption of cost-effective strategies to enhance ES provision.

**Result [Cost-effective decision rule]** A management option $k$ improving ES provision with respect to the *statu quo* (i.e. the profit maximising option) should be implemented on a given area to produce jointly all the ES (land-sharing strategy) only if its opportunity cost $C_k$ satisfies $C_k \leq C_e/(1+\beta_k)$, where $C_e$ is the opportunity cost of its *land-efficient* benchmark and $\beta_k$ is its efficiency score. Otherwise, its is less costly to adopt its efficient benchmark on a share $1/(1+\beta_k)$ of the area, i.e. to devote this share of the area to the management options providing the most non-marketed ES, the share $\beta_k/(1+\beta_k)$ remaining under the *statu quo* management option (land-sparing strategy).

This is illustrated on Fig. 5, which represents the strategy of adopting any management option $k$ on the left, and its efficient benchmark on the right-hand side. The ES provided (measured as the difference compared to statu quo) are represented by the (lower) green area and the opportunity cost by the (upper) orange area. On the right-hand panel, the spare land is cultivated under the *statu quo* (white area), and given that both ES and opportunity cost are expressed as the difference with the *statu quo*, spare land provides no additional ES and incurs no additional cost. To equal the increase in ES provided by adopting option $k$ on the whole area, the land on which its efficient alternative has to be adopted is limited to a share $1/(1/\beta_k)$ of the area, as represented by the green arrow, so that both green areas are equal. Determining which solution (option $k$ or its efficient benchmark) is cheaper to provide the ES

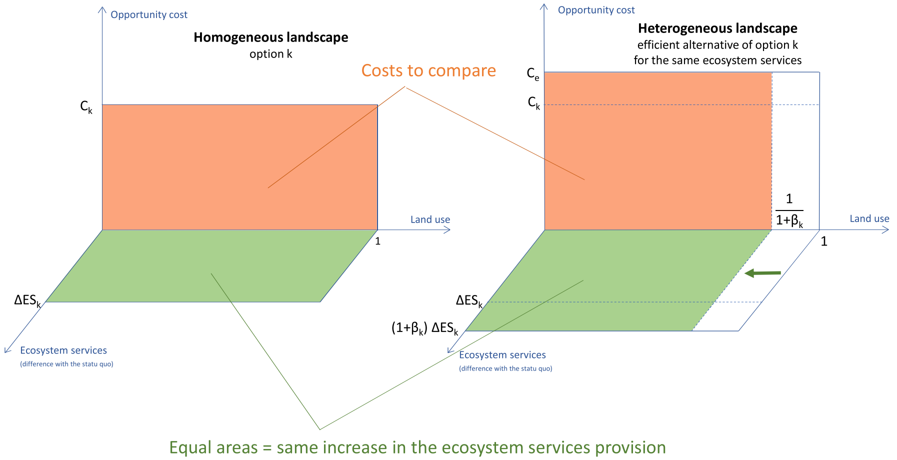

**Fig. 5** Illustration of the comparison of a given management option (on the left-hand side) and its “efficient alternative” (on the right-hand side). The level of $\Delta$ES (green areas in the lower part of the figure) is identical in the two options, by construction of the efficient alternative. The cost of implementation can differ, the option with the lowest cost (orange areas in the upper part of the figure) is cost-effective

bundle is equivalent to determining which orange area is the smallest. The solution is obviously to adopt the efficient benchmark when $C_e < C_k$ (not the case on the figure), but it depends on $\beta_k$ when $C_e > C_k$, as represented on Fig. 5.

This analysis has similarities with the land-sharing/land sparing debate: in the case the efficient alternative is chosen, the “spared” land is cultivated according to the *statu quo* (most profitable management option), and thus the provision of non-marketed ES on the one hand and the provision of agricultural commodities on the other hand are spatially separated. However, our methodology is a way to adapt the land-sparing/land-sharing debate to multiple ES, while emphasising the importance of the opportunity cost of their provision. This analysis thus emphasises that the opportunity cost of changing agricultural management is crucial in a world where the budget for the provision of non-marketed ES is scarce.

The output of this analysis is detailed in Appendix E for agronomic context 8 (i.e. $Q = 9.9\ t\ ha^{-1}$), i.e. a context of high soil quality. For each of the management options that provide more ES than the most profitable option (intensive cropland - management option # 47, in this agronomic context), we define an efficient benchmark, i.e. the combination of grassland (management option # 1) and low-input cropland (management option # 6) that provides the same improvement in terms of ES as the considered management option but on as less land as possible. Given the corresponding inefficiency score $\beta$, we know that the considered management option uses $(1 + \beta)$ more land than its efficient benchmark. As the efficient management options in terms of ES provision are also those with the highest opportunity costs, we need to compare this land ratio to the cost ratio $C_e/C_k$, which measure the extra cost of the efficient benchmark. Whenever $(1 + \beta) > C_e/C_k$, implementing the efficient alternative on a sub-part of the area (land-sparing strategy, with a sub-part of the landscape being used to improve the provision of ES efficiently) is more cost-effective than implementing the considered management option on the whole landscape. Otherwise, the considered management option should be applied on the whole area (land-sharing strategy).

This analysis shows that for about half of the 122 options, adopting the option on the whole area is less costly than adopting the efficient alternative on part of the land (green-colored lines in Appendix E). This is also true for options that were not identified as efficient in the first efficiency analysis with all 5 ES including agricultural production. There is no clear pattern that could explain in which case each strategy is less costly. This depends both on the option considered (and thus on the proportion in which the ES are increased), and on the agronomic context. This is also the case for the management options associated with the largest benefits on climate change mitigation, which combine reduced tillage and biomass input (crop residue restitution) for cropland. Depending on the other practices, which influence both the intensity of production and the provision other ES provision, the cost-effective strategy to mitigate climate change can either correspond to a land-sharing or land-sparing strategy.

In spite of these mild results, we notice two facts. First, in contexts with medium potential yield, the best strategy is often to adopt the efficient alternative (very extensive management on part of the land and *statu quo* on the rest), due to lower opportunity costs of efficient management options. Second, in contexts with high potential yield, both strategies can be interesting, depending on the targeted increase in non-marketed ES. A homogeneous landscape (land-sharing strategy) is more often the best strategy than in the contexts with medium potential yield. This result supports the idea that land-sharing might not be necessarily a bad strategy in intensive agricultural landscapes when one considers adoption costs, but there is no one-size-fits-all solution.

## Concluding remarks

This paper studies the interactions among five biodiversity-based ecosystem services (ES) provided by agroecosystems, by means of efficiency and cost-effectiveness analyses. We use data envelopment analysis, which offers a good way to consider several dimensions of multicriteria decision-making problems without aggregating them. The originality of this paper is also to consider agricultural practices as drivers of the provision of ecosystem services in agroecosystems, which is more realistic compared to many assessments that rely on land-use or land-cover data only.

We simulate the bundles of ES provided by different management options with an integrated agroecological model. The model aims at representing the antagonism between production and non-marketed ecosystem services and the synergy among non-marketed ecosystem services highlighted in the literature. Then, we run an efficiency analysis to identify efficient management options (i.e. combinations of agricultural land uses and practices that maximise ecosystem services provision) and study the interactions among the simulated ES. We find that although few bundles of ES are efficient, they cover a large range of options, from very productive ones to very extensive ones, including intermediate levels, which seems to indicate that providing commodities and non-marketed ecosystem services jointly (on the same land) could be an efficient management strategy.

We also define the opportunity cost of providing a particular bundle of ES rather than choosing the most profitable management option. We use this measure to do a cost-effectiveness analysis. We determine the less costly management strategy to achieve a given improvement of the provision of ES. In particular, for given targets in terms of increase of ES, we determine whether it is cost-effective to spare a part of the area to increase the provision of the ES only (land-sparing strategy), compared to implementing a management option conciliating the provision of ES and agricultural production on the whole area (land-sharing strategy).

Although our analysis does not deal with incentives, the results can be used to identify key recommendations in designing agro-environmental policies aiming at compensating farmers for changes in their agricultural practices.

Our analysis underlines the crucial role of the agronomic context in the determination of efficient management strategies: when the potential yield is low, farmers already tend to choose management options that provide many non-marketed ES (e.g. grassland or low-input cropland), so that agro-environmental policies would bring few additional gains in terms of ES in these contexts. This is one of the criticisms on the lack of efficiency of the agro-environmental schemes in the EU (Kleijn & Sutherland, 2003). Agro-environmental schemes encouraging the provision of non-marketed ES in areas with low potential yield may result in windfall effects.

In the perspective to regulate the provision of multiple ES, such as in our analysis of five ES, efficiency analysis shows that few management options deliver efficient bundles of ES. This underlines the need for the environmental goals of policies to be clearly identified, and the need to consolidate knowledge on the link between agricultural management and the ES provided, in order to identify the efficient management options. Given the strong and complex interactions among ES, we think identifying efficient bundles of ES is a way to design effective agro-environmental schemes that tackle all environmental issues in agriculture in a consistent way.

Our approach also makes clear that cost-effective management options are not always the same as the efficient ones, so that with a fixed budget for agro-environmental policies, the cost of changing agricultural management should also be a decision criterion.

Nevertheless, our approach relies on a simple model, and therefore suffers some limitations. First the costs considered in the analysis do not include transaction costs, costs linked to the transition from one management system to another (investments, education etc.), nor related non-monetary hurdles (risk aversion, role of habits etc.). Thus, the opportunity costs we consider do not equal the subsidies needed to make farmers change their practices, but only the social cost of promoting an increase in non-marketed ecosystem services.

Another limit of our work is that we consider homogenous agricultural regions. Future research will extend the analysis to the case of heterogeneous areas, with different yield potentials, in order to determine spatially explicit strategies to enhance ecosystem services provision in an efficient way.

## Appendix A. Agroecological model: mathematical details

### A.1 Notations

We denote a particular field/area by the index $x$ and a particular time by the index $t$.[14] Management options are denoted by $k$. They correspond to a combination of agricultural practices. At each time, each field/area has a management option denoted by $k(x, t)$.

Control variables correspond to agricultural practices. We consider the following practices:

- The land use $U = \{G; C\}$, corresponding to the choice between grassland or cropland
- pesticide intensity, with three levels: $FTI = \{0; 1; 2\}$ (no pesticides, medium, or high use)
- fertiliser intensity: $F = \{0; 1; 2; 3; 4\}$ (from no fertiliser to high input)
- presence of non-crop habitat such as grass or flower strips: $NCH = \{0; 1\}$ (no; yes)
- biomass input such as crop residues or cover crops: $BI = \{0; 1\}$ (no; yes)
- tillage regime: $T = \{0; 1\}$ (conventional tillage; conservation/reduced tillage)

A management option $k(x, t)$ is thus a combination within the set $U \times FTI \times F \times NCH \times BI \times T$. Grassland options always coincide with no pesticide use, no fertiliser input, no NCH, no biomass input and no tillage. Considering all possible combinations, we obtain 122 management options.

There is a single state variable in the model, the soil organic matter $SOM_x(t)$.

### A.2 Equations for ES assessment

**Soil organic matter and nitrogen** The evolution of soil organic matter $SOM_x(t)$ is given by

$$SOM_x(t+1) = SOM_x(t) - (m_{k(x,t)} + \lambda_{k(x,t)})SOM_x(t) + I_{k(x,t)} \tag{2}$$

The mineralisation rate $m_{k(x,t)}$ and the organic matter leaching rate $\lambda_{k(x,t)}$ depend on the management option $k$. So does the input of organic matter $I_{k(x,t)}$, which is the sum of crop residues and non-harvested part of the plants.

The total available (mineral) nitrogen is

$$N_{x,t} = \frac{c_3}{c_2} \cdot m_{k(x,t)}SOM_x(t) + f_{k(x,t)} + LN_{k(x,t)} \tag{3}$$

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[14] The model can run on multiple locations and time. We use it mostly considering a single piece of land at a given time in this paper, except for the sensitivity analysis of the agronomic context and for the dynamic computation of initial soil organic matter.

where $c_3$ and $c_2$ are conversion parameters to calculate the amount of nitrogen in soil organic matter, $f_{k(x,t)}$ is the mineral nitrogen from applied fertiliser, and $LN_{k(x,t)}$ is the mineral nitrogen stemming from livestock (if relevant).

Nitrogen emitted as nitrous oxide is proportional to total mineral nitrogen

$$N_{Ax,t} = \beta N_{x,t} \tag{4}$$

with $\beta$ is the rate of denitrification.

Crops take up part of the nitrogen available for plants, i.e. $N_{x,t} - N_{Ax,t}$. Up to a certain amount $N^*$, nitrogen uptake by crops is proportional to the nitrogen available. Above this threshold, the nitrogen uptake slows gradually down as nitrogen available for plants increases.

$$\begin{cases} N_{Px,t} = \gamma (N_{x,t} - N_{Ax,t}) & \text{for } N_{x,t} - N_{Ax,t} < N^* \\ N_{Px,t} = \gamma N^* + \frac{\gamma (N_{x,t} - N_{Ax,t} - N^*)}{1+\epsilon (N_{x,t} - N_{Ax,t} - N^*)} & \text{for } N_{x,t} - N_{Ax,t} \geq N^* \end{cases} \tag{5}$$

where $\gamma$ is the nutrient use efficiency, and $N^*$ and $\epsilon$ parameters determining the shape of this hyperbolic function.

Eventually, the remaining nitrogen $N_{Wx,t}$ is leached to water bodies:

$$N_{Wx,t} = N_{x,t} - N_{Ax,t} - N_{Px,t} \tag{6}$$

**Greenhouse gases** Greenhouse gases come from 4 sources: emission of nitrous oxide $N_{Ax,t}$, changes in soil organic carbon stock $\Delta SOC_{x,t}$, fossil fuel burning $FC_k$, and methane emitted by livestock $\text{methane}_{k(x,t)}$.

$$GHG_{x,t} = g_1 c_4 N_{Ax,t} + FC_{k(x,t)} + \Delta SOC_{x,t} + g_2.\text{methane}_{k(x,t)} \tag{7}$$

where $g_1$ is the global warming potential of nitrous oxide and $c_4$ the conversion parameter of nitrogen into nitrous oxide, and $g_2$ is the global warming potential of methane.

The change in soil organic carbon is proportional to the change in soil organic matter, with $c_3$ the carbon content of organic matter:

$$\Delta SOC_{x,t} = SOC(t+1) - SOC(t) = c_3(SOM_x(t+1) - SOM_x(t)) \tag{8}$$

It is an indicator of carbon storage.

**Plant growth - production** In cropland, potential yield $Y_{1x,t}$ depends nitrogen intake $N_{Px,t}$ and soil quality $Q_x$

$$Y_{1x,t} = Q_x(1 - \exp^{-n_2 N_{Px,t}}) \tag{9}$$

with $n_2$ the marginal effect of nitrogen on yield.

Non-crop habitats reduce cultivated area and thus the potential yield after accounting for the area really cultivated is

$$Y_{2x,t} = Y_{1x,t}(1 - e \cdot NCH_{k(x,t)}) \tag{10}$$

with $e$ the proportion of the field dedicated to non-crop habitat.

Eventually, damage due to pests reduces yield, the final yield equals.

$$Y_{3x,t} = Y_{2x,t}(1 - D_{k(x,t)}) \tag{11}$$

Pests and damage $D_{k(x,t)}$ are supposed to be proportional. Pests feed on crop, so that their carrying capacity depends on the yield, and thus damage is expressed as a fraction of yield. This fraction only depends on the intensity of pesticides.

Crop residues are proportional to crop yield.

$$Y_{Rx,t} = \rho Y_{3x,t}(1 - BI_{k(x,t)}) \tag{12}$$

where $BI$ is the binary associated to the crop residue restitution, which equals 1 if crop residues are left on the field.

**Water quality** Water quality over the landscape is given by

$$W_t = \min\{PL_t; NL_t; ML_t\}(1 - w\sum_x NCH_{k(x,t)}) \tag{13}$$

where $PL_t$, $NL_t$, and $ML_t$ are functions expressing pollutant loads in the landscape and $w$ the reduction of pollutants export due to semi-natural elements (in percentage).

Water quality score of the landscape for pesticides:

$$PL_t = \frac{\sum_x FTI_{k(x,t)} - \underline{PL}}{\overline{PL} - \underline{PL}} \tag{14}$$

with $\underline{PL}$ and $\overline{PL}$ minimum and maximum levels of pesticide load over the landscape. Here the minimal load is achieved when no farmer uses pesticides and maximal if every farmer uses pesticides.

Water quality score of the landscape for nutrients:

$$NL_t = \frac{\sum_x N_{Wx,t} - \underline{NL}}{\overline{NL} - \underline{NL}} \tag{15}$$

Again, $\underline{NL}$ and $\overline{NL}$ describe the minimal and maximal nutrient loads of the landscape. $\overline{NL}$ corresponds to a landscape with high levels of fertilisers, soil organic matter, and conventional tillage.

Water quality score of the landscape for organic matter:

$$ML_t = \frac{\sum_x ML_{x,t} - \underline{ML}}{\overline{ML} - \underline{ML}} \tag{16}$$

with $ML_{x,t} = \lambda_{k(x,t)} SOM_x(t)$ the amount of soil organic matter leached on field $x$. $\underline{ML}$ and $\overline{ML}$ describe the minimal and maximal organic matter loads of the landscape. $\overline{ML}$ corresponds to a landscape with high levels soil organic matter and soil loss.

**Pollination** Pollination source score

$$PS_{x,t} = HF_{k(x,t)} \cdot HN_{k(x,t)} \cdot PM_{k(x,t)} \tag{17}$$

with $HF_{k(x,t)}$ the suitability in terms of floral resources, depending on the management $k$ (land-use, pesticide intensity, non-crop habitat), $HN_{k(x,t)}$ the suitability in terms of nesting which depends only on management option $k$ (land-use, pesticide intensity, non-crop habitat), and $PM_{k(x,t)}$ a multiplier representing the decreased mortality of pollinators in fields with medium intensity or no pesticides.

### A.3 Parameter values

| Parameter | Meaning | Value |
|---|---|---|
| $FA_{k(x,t)}$ | Foraging resources for pollinators (index) | 0.5 in grasslands<br>0.23 in croplands with non-crop habitat<br>0.2 in cropland without NCH |
| $NS_{k(x,t)}$ | Habitat suitability for pollinators (index) | 0.4 in grasslands<br>0.23 in cropland with NCH<br>0.2 in cropland without NCH |
| $PM_{k(x,t)}$ | Pesticide impact factor on pollinators (index) | 1.4 for no pesticides<br>1.2 for medium pesticide intensity<br>1 for maximal pesticide intensity |
| $m_{k(x,t)}$ | Mineralisation rate of organic nitrogen (fraction) | 0.015 for reduced tillage and grassland<br>0.019 for conventional tillage |
| $\lambda_{k(x,t)}$ | Erosion rate (fraction) | 0.00006 for grassland<br>0.00007 for cropland, reduced tillage, and with crop residue restitution<br>0.00009 for cropland, reduced tillage, and without crop residue restitution<br>0.0002 for cropland, conventional tillage, and with crop residue restitution<br>0.00025 for cropland, conventional tillage, and without crop residue restitution |
| $I_{k(x,t)}$ | Fresh organic matter inputs (t/ha) | 1.684 for grassland<br>1.348 for cropland, reduced tillage, and with crop residue restitution<br>1.142 for cropland, reduced tillage, and without crop residue restitution<br>1.321 for cropland, conventional tillage, and with crop residue restitution<br>1.059 for cropland, conventional tillage, and without crop residue restitution |
| $f_{k(x,t)}$ | Mineral nitrogen input from synthetic fertilisers (kg/ha) | 0 for no fertilisers<br>110 for fertiliser intensity 1<br>140 for fertiliser intensity 2<br>170 for fertiliser intensity 3<br>200 for fertiliser intensity 4 |
| $LN_{k(x,t)}$ | Mineral nitrogen input from livestock on grassland (kg/ha) | 62 if livestock impacts are accounted for |
| $c_3$ | Nitrogen fraction in soil organic matter | 0.65 |
| $\beta$ | Denitrification rate (fraction) | 1% |

| | | |
|---|---|---|
| $\gamma$ | Nutrient uptake coefficient (fraction) | 0.87 |
| $\epsilon$ | Nitrogen uptake parameter | 0.0032 |
| $N^*$ | Nitrogen uptake parameter | 206 |
| $c_4$ | Conversion parameter Elemental nitrogen into nitrous oxide (factor) | 1.57 |
| $c_5$ | conversion parameter Elemental carbon into carbon dioxide | 3.66 |
| $g_1$ | Global warming potential of nitrous oxide | 298 |
| $g_2$ | Global warming potential of methane | 34 |
| $FC_{k(x,t)}$ | Carbon dioxide due to fuel burning kg $CO_2$eq | 8 for grassland<br>120 for reduced tillage<br>150 for conventional tillage |
| Pesticide doses | Pesticide doses (standard doses) | 3 for medium pesticide intensity<br>6 for maximal pesticide intensity |
| $w$ | Reduction in pollutants due to non-crop habitats | 40% |
| $n_2$ | Marginal effect of uptaken nitrogen on yield | 0.015 |
| $e$ | Area covered by non-crop habitat (if any) | 5% |
| $D_{k(x,t)}$ | Proportion of potential yield lost due to pest damage | 0.3 if no pesticide are used<br>0.12 for medium intensity of pesticides<br>0.03 for maximal intensity of pesticides |
| $\rho$ | Crop - residue ratio | 1 |
| | Crop price (euro/t) | 170 |
| | Revenues of grassland (euro/ha) | 720 |
| | Base management costs (euro/ha) | 197 for any management option |
| | Fertiliser costs (euro/kg N) | 1.15 |
| | Pesticide costs (euro/standard dose for 1 ha) | 33 |
| | Mechanisation costs (euro/ha) | 150 for grasslands<br>225 for reduced tillage<br>300 for conventional tillage |
| | Costs of implementing non-crop habitat (euro/ha) | 35 |

## Appendix B. Agronomic contexts

The following table summarises the characteristics of the 10 agronomic contexts that we considered, with the exogenous soil quality index $Q$ (representing the potential yield) and the corresponding soil organic matter (SOM), defined as the equilibrium value reached by the dynamic Eq. 2 under the more profitable management option for each agronomic context.

| Context | $Q$ (t/ha) | $SOM_0$ (t/ha) |
|---|---|---|
| 1 | 4 | 111.8 |
| 2 | 5 | 111.8 |
| 3 | 5.2 | 104.4 |
| 4 | 5.5 | 75.7 |
| 5 | 6.6 | 75.7 |
| 6 | 7.7 | 75.7 |
| 7 | 8.8 | 75.7 |
| 8 | 9.9 | 75.7 |
| 9 | 11 | 75.7 |
| 10 | 12 | 75.7 |

Note that, although the decreasing stock of soil organic matter has an opposite effect on yield than the increasing soil quality, the effect of soil quality dominates the impact of soil organic matter, so that the yield increases in a monotonous way across agronomic contexts.

# Appendix C. Output of the simulations*

Option 2 is grassland with the impacts of livestock accounted for
Land use: G means grassland, C means cropland.
Tillage regime: R means reduced tillage, C means conventional tillage
Non-crop habitat and crop residue restitution: 0 means no, 1 means yes

| Option | Agricultural production (euro) | Poll (index) | Water quality (index) | GHG (t $CO_2$eq) | Evolution of soil fertility (ton) | Profit (euro) | Land use | Fertiliser intensity | Pesticide intensity | Tillage regime | Non-crop habitat | Crop residue restitution |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **1** | 720 | 0,200 | 0,75 | -714 | 0,536 | 373 | G | - | - | - | - | - |
| **2** | 720 | 0,200 | 0,54 | 1684 | 0,536 | 373 | G | - | - | - | - | - |
| **3** | 837 | 0,040 | 0,64 | 464 | 0,000 | 390 | C | 0 | 0 | R | 0 | 0 |
| **4** | 750 | 0,040 | 0,72 | 30 | 0,204 | 303 | C | 0 | 0 | R | 0 | 1 |
| **5** | 795 | 0,053 | 0,78 | 464 | 0,000 | 313 | C | 0 | 0 | R | 1 | 0 |
| **6** | 712 | 0,053 | 0,83 | 30 | 0,204 | 230 | C | 0 | 0 | R | 1 | 1 |
| **7** | 941 | 0,040 | 0,00 | 1423 | -0,391 | 474 | C | 0 | 0 | C | 0 | 0 |
| **8** | 843 | 0,040 | 0,20 | 870 | -0,130 | 376 | C | 0 | 0 | C | 0 | 1 |
| **9** | 893 | 0,053 | 0,40 | 1423 | -0,391 | 391 | C | 0 | 0 | C | 1 | 0 |
| **10** | 801 | 0,053 | 0,52 | 870 | -0,130 | 299 | C | 0 | 0 | C | 1 | 1 |
| **11** | 1088 | 0,033 | 0,50 | 464 | 0,000 | 542 | C | 0 | 1 | R | 0 | 0 |
| **12** | 975 | 0,033 | 0,50 | 30 | 0,204 | 429 | C | 0 | 1 | R | 0 | 1 |
| **13** | 1033 | 0,044 | 0,70 | 464 | 0,000 | 452 | C | 0 | 1 | R | 1 | 0 |
| **14** | 926 | 0,044 | 0,70 | 30 | 0,204 | 345 | C | 0 | 1 | R | 1 | 1 |
| **15** | 1223 | 0,033 | 0,00 | 1423 | -0,391 | 657 | C | 0 | 1 | C | 0 | 0 |
| **16** | 1096 | 0,033 | 0,20 | 870 | -0,130 | 530 | C | 0 | 1 | C | 0 | 1 |
| **17** | 1162 | 0,044 | 0,40 | 1423 | -0,391 | 561 | C | 0 | 1 | C | 1 | 0 |
| **18** | 1041 | 0,044 | 0,52 | 870 | -0,130 | 440 | C | 0 | 1 | C | 1 | 1 |
| **19** | 1213 | 0,029 | 0,00 | 464 | 0,000 | 568 | C | 0 | 2 | R | 0 | 0 |
| **20** | 1087 | 0,029 | 0,00 | 30 | 0,204 | 442 | C | 0 | 2 | R | 0 | 1 |
| **21** | 1153 | 0,038 | 0,40 | 464 | 0,000 | 473 | C | 0 | 2 | R | 1 | 0 |
| **22** | 1033 | 0,038 | 0,40 | 30 | 0,204 | 353 | C | 0 | 2 | R | 1 | 1 |
| **23** | 1364 | 0,029 | 0,00 | 1423 | -0,391 | 699 | C | 0 | 2 | C | 0 | 0 |
| **24** | 1222 | 0,029 | 0,00 | 870 | -0,130 | 557 | C | 0 | 2 | C | 0 | 1 |
| **25** | 1296 | 0,038 | 0,40 | 1423 | -0,391 | 596 | C | 0 | 2 | C | 1 | 0 |
| **26** | 1161 | 0,038 | 0,40 | 870 | -0,130 | 461 | C | 0 | 2 | C | 1 | 1 |
| **27** | 1174 | 0,040 | 0,37 | 978 | 0,000 | 600 | C | 1 | 0 | R | 0 | 0 |
| **28** | 1051 | 0,040 | 0,37 | 545 | 0,204 | 477 | C | 1 | 0 | R | 0 | 1 |
| **29** | 1115 | 0,053 | 0,62 | 978 | 0,000 | 507 | C | 1 | 0 | R | 1 | 0 |
| **30** | 998 | 0,053 | 0,62 | 545 | 0,204 | 389 | C | 1 | 0 | R | 1 | 1 |
| **31** | 1194 | 0,040 | 0,00 | 1938 | -0,391 | 600 | C | 1 | 0 | C | 0 | 0 |
| **32** | 1068 | 0,040 | 0,20 | 1385 | -0,130 | 475 | C | 1 | 0 | C | 0 | 1 |
| **33** | 1134 | 0,053 | 0,40 | 1938 | -0,391 | 506 | C | 1 | 0 | C | 1 | 0 |
| **34** | 1015 | 0,053 | 0,52 | 1385 | -0,130 | 386 | C | 1 | 0 | C | 1 | 1 |
| **35** | 1526 | 0,033 | 0,37 | 978 | 0,000 | 854 | C | 1 | 1 | R | 0 | 0 |
| **36** | 1366 | 0,033 | 0,37 | 545 | 0,204 | 693 | C | 1 | 1 | R | 0 | 1 |
| **37** | 1450 | 0,044 | 0,62 | 978 | 0,000 | 742 | C | 1 | 1 | R | 1 | 0 |

| Option | Agricultural production (euro) | Poll (index) | Water quality (index) | GHG (t $CO_2$eq) | Evolution of soil fertility (ton) | Profit (euro) | Land use | Fertiliser intensity | Pesticide intensity | Tillage regime | Non-crop habitat | Crop residue restitution |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **38** | 1297 | 0,044 | 0,62 | 545 | 0,204 | 590 | C | 1 | 1 | R | 1 | 1 |
| **39** | 1552 | 0,033 | 0,00 | 1938 | -0,391 | 859 | C | 1 | 1 | C | 0 | 0 |
| **40** | 1389 | 0,033 | 0,20 | 1385 | -0,130 | 696 | C | 1 | 1 | C | 0 | 1 |
| **41** | 1474 | 0,044 | 0,40 | 1938 | -0,391 | 747 | C | 1 | 1 | C | 1 | 0 |
| **42** | 1319 | 0,044 | 0,52 | 1385 | -0,130 | 592 | C | 1 | 1 | C | 1 | 1 |
| **43** | 1702 | 0,029 | 0,00 | 978 | 0,000 | 931 | C | 1 | 2 | R | 0 | 0 |
| **44** | 1523 | 0,029 | 0,00 | 545 | 0,204 | 752 | C | 1 | 2 | R | 0 | 1 |
| **45** | 1617 | 0,038 | 0,40 | 978 | 0,000 | 810 | C | 1 | 2 | R | 1 | 0 |
| **46** | 1447 | 0,038 | 0,40 | 545 | 0,204 | 641 | C | 1 | 2 | R | 1 | 1 |
| **47** | 1731 | 0,029 | 0,00 | 1938 | -0,391 | 940 | C | 1 | 2 | C | 0 | 0 |
| **48** | 1549 | 0,029 | 0,00 | 1385 | -0,130 | 758 | C | 1 | 2 | C | 0 | 1 |
| **49** | 1644 | 0,038 | 0,40 | 1938 | -0,391 | 818 | C | 1 | 2 | C | 1 | 0 |
| **50** | 1472 | 0,038 | 0,40 | 1385 | -0,130 | 645 | C | 1 | 2 | C | 1 | 1 |
| **51** | 1203 | 0,040 | 0,27 | 1119 | 0,000 | 595 | C | 2 | 0 | R | 0 | 0 |
| **52** | 1076 | 0,040 | 0,27 | 685 | 0,204 | 468 | C | 2 | 0 | R | 0 | 1 |
| **53** | 1143 | 0,053 | 0,56 | 1119 | 0,000 | 500 | C | 2 | 0 | R | 1 | 0 |
| **54** | 1023 | 0,053 | 0,56 | 685 | 0,204 | 380 | C | 2 | 0 | R | 1 | 1 |
| **55** | 1216 | 0,040 | 0,00 | 2078 | -0,391 | 588 | C | 2 | 0 | C | 0 | 0 |
| **56** | 1088 | 0,040 | 0,20 | 1525 | -0,130 | 460 | C | 2 | 0 | C | 0 | 1 |
| **57** | 1155 | 0,053 | 0,40 | 2078 | -0,391 | 492 | C | 2 | 0 | C | 1 | 0 |
| **58** | 1033 | 0,053 | 0,52 | 1525 | -0,130 | 370 | C | 2 | 0 | C | 1 | 1 |
| **59** | 1564 | 0,033 | 0,27 | 1119 | 0,000 | 857 | C | 2 | 1 | R | 0 | 0 |
| **60** | 1399 | 0,033 | 0,27 | 685 | 0,204 | 692 | C | 2 | 1 | R | 0 | 1 |
| **61** | 1486 | 0,044 | 0,56 | 1119 | 0,000 | 744 | C | 2 | 1 | R | 1 | 0 |
| **62** | 1329 | 0,044 | 0,56 | 685 | 0,204 | 587 | C | 2 | 1 | R | 1 | 1 |
| **63** | 1580 | 0,033 | 0,00 | 2078 | -0,391 | 853 | C | 2 | 1 | C | 0 | 0 |
| **64** | 1414 | 0,033 | 0,20 | 1525 | -0,130 | 687 | C | 2 | 1 | C | 0 | 1 |
| **65** | 1501 | 0,044 | 0,40 | 2078 | -0,391 | 739 | C | 2 | 1 | C | 1 | 0 |
| **66** | 1343 | 0,044 | 0,52 | 1525 | -0,130 | 581 | C | 2 | 1 | C | 1 | 1 |
| **67** | 1744 | 0,029 | 0,00 | 1119 | 0,000 | 938 | C | 2 | 2 | R | 0 | 0 |
| **68** | 1561 | 0,029 | 0,00 | 685 | 0,204 | 755 | C | 2 | 2 | R | 0 | 1 |
| **69** | 1657 | 0,038 | 0,40 | 1119 | 0,000 | 816 | C | 2 | 2 | R | 1 | 0 |
| **70** | 1483 | 0,038 | 0,40 | 685 | 0,204 | 642 | C | 2 | 2 | R | 1 | 1 |
| **71** | 1763 | 0,029 | 0,00 | 2078 | -0,391 | 937 | C | 2 | 2 | C | 0 | 0 |
| **72** | 1577 | 0,029 | 0,00 | 1525 | -0,130 | 751 | C | 2 | 2 | C | 0 | 1 |
| **73** | 1675 | 0,038 | 0,40 | 2078 | -0,391 | 814 | C | 2 | 2 | C | 1 | 0 |
| **74** | 1498 | 0,038 | 0,40 | 1525 | -0,130 | 637 | C | 2 | 2 | C | 1 | 1 |
| **75** | 1221 | 0,040 | 0,17 | 1259 | 0,000 | 579 | C | 3 | 0 | R | 0 | 0 |
| **76** | 1093 | 0,040 | 0,17 | 825 | 0,204 | 450 | C | 3 | 0 | R | 0 | 1 |
| **77** | 1160 | 0,053 | 0,50 | 1259 | 0,000 | 483 | C | 3 | 0 | R | 1 | 0 |
| **78** | 1038 | 0,053 | 0,50 | 825 | 0,204 | 361 | C | 3 | 0 | R | 1 | 1 |
| **79** | 1230 | 0,040 | 0,00 | 2219 | -0,391 | 567 | C | 3 | 0 | C | 0 | 0 |
| **80** | 1100 | 0,040 | 0,10 | 1666 | -0,130 | 438 | C | 3 | 0 | C | 0 | 1 |

| Option | Agricultural production (euro) | Poll (index) | Water quality (index) | GHG (t $CO_2$eq) | Evolution of soil fertility (ton) | Profit (euro) | Land use | Fertiliser intensity | Pesticide intensity | Tillage regime | Non-crop habitat | Crop residue restitution |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **81** | 1168 | 0,053 | 0,40 | 2219 | -0,391 | 471 | C | 3 | 0 | C | 1 | 0 |
| **82** | 1045 | 0,053 | 0,46 | 1666 | -0,130 | 348 | C | 3 | 0 | C | 1 | 1 |
| **83** | 1588 | 0,033 | 0,17 | 1259 | 0,000 | 846 | C | 3 | 1 | R | 0 | 0 |
| **84** | 1421 | 0,033 | 0,17 | 825 | 0,204 | 679 | C | 3 | 1 | R | 0 | 1 |
| **85** | 1508 | 0,044 | 0,50 | 1259 | 0,000 | 732 | C | 3 | 1 | R | 1 | 0 |
| **86** | 1350 | 0,044 | 0,50 | 825 | 0,204 | 573 | C | 3 | 1 | R | 1 | 1 |
| **87** | 1598 | 0,033 | 0,00 | 2219 | -0,391 | 837 | C | 3 | 1 | C | 0 | 0 |
| **88** | 1430 | 0,033 | 0,10 | 1666 | -0,130 | 669 | C | 3 | 1 | C | 0 | 1 |
| **89** | 1518 | 0,044 | 0,40 | 2219 | -0,391 | 722 | C | 3 | 1 | C | 1 | 0 |
| **90** | 1359 | 0,044 | 0,46 | 1666 | -0,130 | 562 | C | 3 | 1 | C | 1 | 1 |
| **91** | 1771 | 0,029 | 0,00 | 1259 | 0,000 | 931 | C | 3 | 2 | R | 0 | 0 |
| **92** | 1585 | 0,029 | 0,00 | 825 | 0,204 | 744 | C | 3 | 2 | R | 0 | 1 |
| **93** | 1683 | 0,038 | 0,40 | 1259 | 0,000 | 807 | C | 3 | 2 | R | 1 | 0 |
| **94** | 1506 | 0,038 | 0,40 | 825 | 0,204 | 630 | C | 3 | 2 | R | 1 | 1 |
| **95** | 1783 | 0,029 | 0,00 | 2219 | -0,391 | 922 | C | 3 | 2 | C | 0 | 0 |
| **96** | 1595 | 0,029 | 0,00 | 1666 | -0,130 | 735 | C | 3 | 2 | C | 0 | 1 |
| **97** | 1694 | 0,038 | 0,40 | 2219 | -0,391 | 798 | C | 3 | 2 | C | 1 | 0 |
| **98** | 1516 | 0,038 | 0,40 | 1666 | -0,130 | 620 | C | 3 | 2 | C | 1 | 1 |
| **99** | 1233 | 0,040 | 0,07 | 1400 | 0,000 | 556 | C | 4 | 0 | R | 0 | 0 |
| **100** | 1103 | 0,040 | 0,07 | 966 | 0,204 | 426 | C | 4 | 0 | R | 0 | 1 |
| **101** | 1172 | 0,053 | 0,44 | 1400 | 0,000 | 460 | C | 4 | 0 | R | 1 | 0 |
| **102** | 1048 | 0,053 | 0,44 | 966 | 0,204 | 336 | C | 4 | 0 | R | 1 | 1 |
| **103** | 1238 | 0,040 | 0,00 | 2359 | -0,391 | 541 | C | 4 | 0 | C | 0 | 0 |
| **104** | 1108 | 0,040 | 0,00 | 1806 | -0,130 | 411 | C | 4 | 0 | C | 0 | 1 |
| **105** | 1176 | 0,053 | 0,40 | 2359 | -0,391 | 444 | C | 4 | 0 | C | 1 | 0 |
| **106** | 1053 | 0,053 | 0,40 | 1806 | -0,130 | 321 | C | 4 | 0 | C | 1 | 1 |
| **107** | 1603 | 0,033 | 0,07 | 1400 | 0,000 | 827 | C | 4 | 1 | R | 0 | 0 |
| **108** | 1435 | 0,033 | 0,07 | 966 | 0,204 | 659 | C | 4 | 1 | R | 0 | 1 |
| **109** | 1523 | 0,044 | 0,44 | 1400 | 0,000 | 712 | C | 4 | 1 | R | 1 | 0 |
| **110** | 1363 | 0,044 | 0,44 | 966 | 0,204 | 552 | C | 4 | 1 | R | 1 | 1 |
| **111** | 1610 | 0,033 | 0,00 | 2359 | -0,391 | 814 | C | 4 | 1 | C | 0 | 0 |
| **112** | 1441 | 0,033 | 0,00 | 1806 | -0,130 | 645 | C | 4 | 1 | C | 0 | 1 |
| **113** | 1529 | 0,044 | 0,40 | 2359 | -0,391 | 698 | C | 4 | 1 | C | 1 | 0 |
| **114** | 1369 | 0,044 | 0,40 | 1806 | -0,130 | 538 | C | 4 | 1 | C | 1 | 1 |
| **115** | 1788 | 0,029 | 0,00 | 1400 | 0,000 | 913 | C | 4 | 2 | R | 0 | 0 |
| **116** | 1600 | 0,029 | 0,00 | 966 | 0,204 | 725 | C | 4 | 2 | R | 0 | 1 |
| **117** | 1699 | 0,038 | 0,40 | 1400 | 0,000 | 789 | C | 4 | 2 | R | 1 | 0 |
| **118** | 1520 | 0,038 | 0,40 | 966 | 0,204 | 610 | C | 4 | 2 | R | 1 | 1 |
| **119** | 1796 | 0,029 | 0,00 | 2359 | -0,391 | 901 | C | 4 | 2 | C | 0 | 0 |
| **120** | 1607 | 0,029 | 0,00 | 1806 | -0,130 | 712 | C | 4 | 2 | C | 0 | 1 |
| **121** | 1706 | 0,038 | 0,40 | 2359 | -0,391 | 776 | C | 4 | 2 | C | 1 | 0 |
| **122** | 1526 | 0,038 | 0,40 | 1806 | -0,130 | 596 | C | 4 | 2 | C | 1 | 1 |

# Appendix D. Efficient bundles of ecosystem services in all agronomic contexts

Yelllow-colored cells indicate which options are efficient in which agronomic context

| | Contexts | | | | | | | | | |
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# Appendix E. Output of the cost-effectiveness analysis

Bright green options (1 and 6) maximise non-marketed ecosystem services
Orange option (47) is the most profitable one, the opportunity cost and all ecosystem services are expressed as a difference with its levels. Grey options provide less ecosystem services than the most profitable option, and are excluded from the analysis.
Light green lines are the options for which the **efficient alternative** is less costly to provide non-marketed ecosystem services.
Light blue lines are the options for which the **original option** is less costly to provide non-marketed ecosystem services.

| **Option** | **Composition of the efficient alternative** | | | | **Opportunity cost (euro)** | **Opportunity cost of the efficient alternative** | **1+β** | **Cost ratio** | **Land use** | **Fertiliser intensity** | **Pesticide intensity** | **Tillage regime** | **Non-crop habitat** | **Crop residue restitution** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | First option | Proportion of first option | Second option | Proportion of second option | | | | | | | | | | |
| **1** | 1 | 1 | 0 | 0 | 566,51 | 566,51 | **1** | **1** | G | - | - | - | - | - |
| **3** | 1 | 0,01 | 6 | 0,- | 549,81 | 707,97 | **1,3** | **1,29** | C | 0 | 0 | R | 0 | 0 |
| **4** | 1 | 0,31 | 6 | 0,69 | 636,76 | 665,38 | **1,12** | **1,04** | C | 0 | 0 | R | 0 | 1 |
| **5** | 1 | 0,01 | 6 | 0,- | 626,64 | 707,82 | **1,06** | **1,13** | C | 0 | 0 | R | 1 | 0 |
| **6** | 1 | 0 | 6 | 1 | 709,24 | 709,24 | **1** | **1** | C | 0 | 0 | R | 1 | 1 |
| **7** | 1 | 1 | 0 | 0 | 465,98 | 566,51 | **5,15** | **1,22** | C | 0 | 0 | C | 0 | 0 |
| **8** | 1 | 1 | 0 | 0 | 563,62 | 566,51 | **2,48** | **1,01** | C | 0 | 0 | C | 0 | 1 |
| **9** | 1 | 0,17 | 6 | 0,83 | 548,01 | 684,61 | **2,04** | **1,25** | C | 0 | 0 | C | 1 | 0 |
| **10** | 1 | 0,1 | 6 | 0,9 | 640,76 | 695,44 | **1,58** | **1,09** | C | 0 | 0 | C | 1 | 1 |
| **11** | 1 | 0,55 | 6 | 0,45 | 397,8 | 630,75 | **1,57** | **1,59** | C | 0 | 1 | R | 0 | 0 |
| **12** | 1 | 1 | 0 | 0 | 510,83 | 566,51 | **1,39** | **1,11** | C | 0 | 1 | R | 0 | 1 |
| **13** | 6 | 1 | 0 | 0 | 487,18 | 709,24 | **1,19** | **1,46** | C | 0 | 1 | R | 1 | 0 |
| **14** | 1 | 0,37 | 6 | 0,63 | 594,56 | 656,42 | **1,14** | **1,1** | C | 0 | 1 | R | 1 | 1 |
| **15** | 1 | 1 | 0 | 0 | 282,82 | 566,51 | **5,15** | **2** | C | 0 | 1 | C | 0 | 0 |
| **16** | 1 | 1 | 0 | 0 | 409,75 | 566,51 | **2,48** | **1,38** | C | 0 | 1 | C | 0 | 1 |
| **17** | 1 | 0,05 | 6 | 0,95 | 378,96 | 701,71 | **2,07** | **1,85** | C | 0 | 1 | C | 1 | 0 |
| **18** | 1 | 0 | 6 | 1 | 4-,54 | 708,77 | **1,6** | **1,42** | C | 0 | 1 | C | 1 | 1 |
| **19** | 1 | 1 | 0 | 0 | 371,29 | 566,51 | **1,8** | **1,53** | C | 0 | 2 | R | 0 | 0 |
| **20** | 1 | 1 | 0 | 0 | 497,37 | 566,51 | **1,39** | **1,14** | C | 0 | 2 | R | 0 | 1 |
| **21** | 1 | 1 | 0 | 0 | 466,95 | 566,51 | **1,8** | **1,21** | C | 0 | 2 | R | 1 | 0 |
| **22** | 1 | 1 | 0 | 0 | 586,72 | 566,51 | **1,39** | **0,97** | C | 0 | 2 | R | 1 | 1 |
| **23** | 1 | 1 | 0 | 0 | 240,75 | 566,51 | **5,15** | **2,35** | C | 0 | 2 | C | 0 | 0 |
| **24** | 1 | 1 | 0 | 0 | 382,31 | 566,51 | **2,48** | **1,48** | C | 0 | 2 | C | 0 | 1 |
| **25** | 6 | 1 | 0 | 0 | 343,93 | 709,24 | **2,08** | **2,06** | C | 0 | 2 | C | 1 | 0 |
| **26** | 1 | 0,32 | 6 | 0,68 | 478,42 | 663,11 | **2,01** | **1,39** | C | 0 | 2 | C | 1 | 1 |
| **27** | 1 | 0,65 | 6 | 0,35 | 339,15 | 616,09 | **2,08** | **1,82** | C | 1 | 0 | R | 0 | 0 |
| **28** | 1 | 1 | 0 | 0 | 462,5 | 566,51 | **1,56** | **1,22** | C | 1 | 0 | R | 0 | 1 |
| **29** | 1 | 0,05 | 6 | 0,95 | 432,84 | 701,56 | **1,33** | **1,62** | C | 1 | 0 | R | 1 | 0 |
| **30** | 1 | 0,48 | 6 | 0,52 | 550,02 | 640,61 | **1,27** | **1,16** | C | 1 | 0 | R | 1 | 1 |
| **31** | 1 | 1 | 0 | 0 | 339,21 | 566,51 | **15** | **1,67** | C | 1 | 0 | C | 0 | 0 |
| **32** | 1 | 1 | 0 | 0 | 464,61 | 566,51 | **3,55** | **1,22** | C | 1 | 0 | C | 0 | 1 |
| **33** | 1 | 0,17 | 6 | 0,83 | 433,9 | 684,61 | **2,04** | **1,58** | C | 1 | 0 | C | 1 | 0 |
| **34** | 1 | 0,1 | 6 | 0,9 | 553,03 | 695,44 | **1,58** | **1,26** | C | 1 | 0 | C | 1 | 1 |
| **35** | 1 | 0,65 | 6 | 0,35 | 85,- | 616,09 | **2,08** | **7,16** | C | 1 | 1 | R | 0 | 0 |
| **36** | 1 | 1 | 0 | 0 | 246,34 | 566,51 | **1,56** | **2,3** | C | 1 | 1 | R | 0 | 1 |
| **37** | 6 | 1 | 0 | 0 | 197,29 | 709,24 | **1,33** | **3,59** | C | 1 | 1 | R | 1 | 0 |
| **38** | 1 | 0,48 | 6 | 0,52 | 349,63 | 640,61 | **1,27** | **1,83** | C | 1 | 1 | R | 1 | 1 |
| **39** | 1 | 1 | 0 | 0 | 80,07 | 566,51 | **36** | **7,08** | C | 1 | 1 | C | 0 | 0 |

| **Option** | **Composition of the efficient alternative** | | | | **Opportunity cost (euro)** | **Opportunity cost of the efficient alternative** | **1+β** | **Cost ratio** | **Land use** | **Fertiliser intensity** | **Pesticide intensity** | **Tillage regime** | **Non-crop habitat** | **Crop residue restitution** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | First option | Proportion of first option | Second option | Proportion of second option | | | | | | | | | | |
| **40** | 1 | 1 | 0 | 0 | 243,09 | 566,51 | **3,55** | **2,33** | C | 1 | 1 | C | 0 | 1 |
| **41** | 1 | 0,05 | 6 | 0,95 | 192,67 | 701,71 | **2,07** | **3,64** | C | 1 | 1 | C | 1 | 0 |
| **42** | 1 | 0 | 6 | 1 | 347,54 | 708,77 | **1,6** | **2,04** | C | 1 | 1 | C | 1 | 1 |
| **43** | 1 | 1 | 0 | 0 | 8,91 | 566,51 | **2,37** | **63,56** | C | 1 | 2 | R | 0 | 0 |
| **44** | 1 | 1 | 0 | 0 | 187,77 | 566,51 | **1,56** | **3,02** | C | 1 | 2 | R | 0 | 1 |
| **45** | 1 | 0,53 | 6 | 0,47 | 129,02 | 634,28 | **1,97** | **4,92** | C | 1 | 2 | R | 1 | 0 |
| **46** | 1 | 1 | 0 | 0 | 298,93 | 566,51 | **1,56** | **1,9** | C | 1 | 2 | R | 1 | 1 |
| **47** | 1 | 1 | 0 | 0 | 0 | 566,51 | **-** | **-** | C | 1 | 2 | C | 0 | 0 |
| **48** | 1 | 1 | 0 | 0 | 181,83 | 566,51 | **3,55** | **3,12** | C | 1 | 2 | C | 0 | 1 |
| **49** | 6 | 1 | 0 | 0 | 121,55 | 709,24 | **2,08** | **5,83** | C | 1 | 2 | C | 1 | 0 |
| **50** | 6 | 1 | 0 | 0 | 294,29 | 709,24 | **2,08** | **2,41** | C | 1 | 2 | C | 1 | 1 |
| **51** | 1 | 1 | 0 | 0 | 344,61 | 566,51 | **2,37** | **1,64** | C | 2 | 0 | R | 0 | 0 |
| **52** | 1 | 1 | 0 | 0 | 471,09 | 566,51 | **1,56** | **1,2** | C | 2 | 0 | R | 0 | 1 |
| **53** | 1 | 0,08 | 6 | 0,92 | 439,75 | 698,24 | **1,47** | **1,59** | C | 2 | 0 | R | 1 | 0 |
| **54** | 1 | 0,68 | 6 | 0,32 | 559,91 | 612,67 | **1,38** | **1,09** | C | 2 | 0 | R | 1 | 1 |
| **55** | - | - | - | - | 351,89 | - | **-** | **-** | C | 2 | 0 | C | 0 | 0 |
| **56** | 1 | 1 | 0 | 0 | 479,68 | 566,51 | **3,55** | **1,61** | C | 2 | 0 | C | 0 | 1 |
| **57** | - | - | - | - | 447,67 | - | **-** | **-** | C | 2 | 0 | C | 1 | 0 |
| **58** | 1 | 0,1 | 6 | 0,9 | 569,08 | 695,44 | **1,58** | **1,45** | C | 2 | 0 | C | 1 | 1 |
| **59** | 1 | 1 | 0 | 0 | 82,74 | 566,51 | **2,37** | **1,27** | C | 2 | 1 | R | 0 | 0 |
| **60** | 1 | 1 | 0 | 0 | 247,17 | 566,51 | **1,56** | **1** | C | 2 | 1 | R | 0 | 1 |
| **61** | 6 | 1 | 0 | 0 | 195,93 | 709,24 | **1,48** | **8,57** | C | 2 | 1 | R | 1 | 0 |
| **62** | 1 | 0,68 | 6 | 0,32 | 352,13 | 612,67 | **1,38** | **2,48** | C | 2 | 1 | R | 1 | 1 |
| **63** | - | - | - | - | 86,21 | - | **-** | **-** | C | 2 | 1 | C | 0 | 0 |
| **64** | 1 | 1 | 0 | 0 | 252,34 | 566,51 | **3,55** | **2,89** | C | 2 | 1 | C | 0 | 1 |
| **65** | - | - | - | - | 200,22 | - | **-** | **-** | C | 2 | 1 | C | 1 | 0 |
| **66** | 1 | 0 | 6 | 1 | 358,05 | 708,77 | **1,6** | **2,01** | C | 2 | 1 | C | 1 | 1 |
| **67** | 1 | 1 | 0 | 0 | 1,3 | 566,51 | **2,37** | **6,57** | C | 2 | 2 | R | 0 | 0 |
| **68** | 1 | 1 | 0 | 0 | 184,7 | 566,51 | **1,56** | **2,25** | C | 2 | 2 | R | 0 | 1 |
| **69** | 1 | 0,53 | 6 | 0,47 | 123,51 | 634,28 | **1,97** | **3,17** | C | 2 | 2 | R | 1 | 0 |
| **70** | 1 | 1 | 0 | 0 | 297,74 | 566,51 | **1,56** | **1,58** | C | 2 | 2 | R | 1 | 1 |
| **71** | - | - | - | - | 2,87 | - | **-** | **-** | C | 2 | 2 | C | 0 | 0 |
| **72** | 1 | 1 | 0 | 0 | 188,16 | 566,51 | **3,55** | **435,03** | C | 2 | 2 | C | 0 | 1 |
| **73** | - | - | - | - | 126 | - | **-** | **-** | C | 2 | 2 | C | 1 | 0 |
| **74** | 6 | 1 | 0 | 0 | 302,03 | 709,24 | **2,08** | **3,84** | C | 2 | 2 | C | 1 | 1 |
| **75** | 1 | 1 | 0 | 0 | 360,59 | 566,51 | **2,37** | **4,59** | C | 3 | 0 | R | 0 | 0 |
| **76** | 1 | 1 | 0 | 0 | 489,07 | 566,51 | **1,56** | **1,9** | C | 3 | 0 | R | 0 | 1 |
| **77** | 1 | 0,13 | 6 | 0,87 | 456,66 | 690,26 | **1,64** | **240,77** | C | 3 | 0 | R | 1 | 0 |
| **78** | 1 | 0,91 | 6 | 0,09 | 578,72 | 579,45 | **1,51** | **3,08** | C | 3 | 0 | R | 1 | 1 |
| **79** | - | - | - | - | 372,48 | - | **-** | **-** | C | 3 | 0 | C | 0 | 0 |
| **80** | 1 | 1 | 0 | 0 | 501,8 | 566,51 | **3,55** | **4,5** | C | 3 | 0 | C | 0 | 1 |
| **81** | - | - | - | - | 468,96 | - | **-** | **-** | C | 3 | 0 | C | 1 | 0 |

| Option | Composition of the efficient alternative | | | | Opportunity cost (euro) | Opportunity cost of the efficient alternative | 1+β | Cost ratio | Land use | Fertiliser intensity | Pesticide intensity | Tillage regime | Non-crop habitat | Crop residue restitution |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | First option | Proportion of first option | Second option | Proportion of second option | | | | | | | | | | |
| **82** | 1 | 0,13 | 6 | 0,87 | 591,81 | 563,99 | **1,78** | **1,87** | C | 3 | 0 | C | 1 | 1 |
| **83** | 1 | 1 | 0 | 0 | 93,16 | 566,51 | **2,37** | **1,57** | C | 3 | 1 | R | 0 | 0 |
| **84** | 1 | 1 | 0 | 0 | 260,19 | 566,51 | **1,56** | **1,16** | C | 3 | 1 | R | 0 | 1 |
| **85** | 1 | 0,13 | 6 | 0,87 | 207,56 | 564 | **1,64** | **1,24** | C | 3 | 1 | R | 1 | 0 |
| **86** | 1 | 0,91 | 6 | 0,09 | 366,23 | 566,24 | **1,51** | **0,98** | C | 3 | 1 | R | 1 | 1 |
| **87** | - | - | - | - | 102,63 | - | **-** | **-** | C | 3 | 1 | C | 0 | 0 |
| **88** | 1 | 1 | 0 | 0 | 270,74 | 566,51 | **3,55** | **1,52** | C | 3 | 1 | C | 0 | 1 |
| **89** | - | - | - | - | 217,54 | - | **-** | **-** | C | 3 | 1 | C | 1 | 0 |
| **90** | 1 | 0,02 | 6 | 0,98 | 377,25 | 563,69 | **1,8** | **1,12** | C | 3 | 1 | C | 1 | 1 |
| **91** | 1 | 1 | 0 | 0 | 8,95 | 566,51 | **2,37** | **1,21** | C | 3 | 2 | R | 0 | 0 |
| **92** | 1 | 1 | 0 | 0 | 195,25 | 566,51 | **1,56** | **0,96** | C | 3 | 2 | R | 0 | 1 |
| **93** | 1 | 0,53 | 6 | 0,47 | 132,5 | 565,13 | **1,97** | **6,07** | C | 3 | 2 | R | 1 | 0 |
| **94** | 1 | 1 | 0 | 0 | 309,49 | 566,51 | **1,56** | **2,18** | C | 3 | 2 | R | 1 | 1 |
| **95** | - | - | - | - | 17,2 | - | **-** | **-** | C | 3 | 2 | C | 0 | 0 |
| **96** | 1 | 1 | 0 | 0 | 204,71 | 566,51 | **3,55** | **2,73** | C | 3 | 2 | C | 0 | 1 |
| **97** | - | - | - | - | 141,34 | - | **-** | **-** | C | 3 | 2 | C | 1 | 0 |
| **98** | 6 | 1 | 0 | 0 | 319,47 | 709,24 | **2,08** | **1,94** | C | 3 | 2 | C | 1 | 1 |
| **99** | 1 | 1 | 0 | 0 | 383,28 | 566,51 | **2,37** | **5,52** | C | 4 | 0 | R | 0 | 0 |
| **100** | 1 | 1 | 0 | 0 | 513,04 | 566,51 | **1,56** | **2,09** | C | 4 | 0 | R | 0 | 1 |
| **101** | 1 | 0,35 | 6 | 0,65 | 479,94 | 564,63 | **1,82** | **2,6** | C | 4 | 0 | R | 1 | 0 |
| **102** | 1 | 1 | 0 | 0 | 603,21 | 566,51 | **1,56** | **1,5** | C | 4 | 0 | R | 1 | 1 |
| **103** | - | - | - | - | 398,11 | - | **-** | **-** | C | 4 | 0 | C | 0 | 0 |
| **104** | 1 | 1 | 0 | 0 | 528,4 | 566,51 | **3,55** | **63,29** | C | 4 | 0 | C | 0 | 1 |
| **105** | - | - | - | - | 495,03 | - | **-** | **-** | C | 4 | 0 | C | 1 | 0 |
| **106** | 1 | 0,17 | 6 | 0,83 | 618,81 | 564,11 | **2,04** | **2,89** | C | 4 | 0 | C | 1 | 1 |
| **107** | 1 | 1 | 0 | 0 | 112,31 | 566,51 | **2,37** | **4,28** | C | 4 | 1 | R | 0 | 0 |
| **108** | 1 | 1 | 0 | 0 | 281 | 566,51 | **1,56** | **1,83** | C | 4 | 1 | R | 0 | 1 |
| **109** | 1 | 0,35 | 6 | 0,65 | 227,47 | 564,63 | **1,82** | **32,83** | C | 4 | 1 | R | 1 | 0 |
| **110** | 1 | 1 | 0 | 0 | 387,72 | 566,51 | **1,56** | **2,77** | C | 4 | 1 | R | 1 | 1 |
| **111** | - | - | - | - | 125,59 | - | **-** | **-** | C | 4 | 1 | C | 0 | 0 |
| **112** | 1 | 1 | 0 | 0 | 294,97 | 566,51 | **3,55** | **4,01** | C | 4 | 1 | C | 0 | 1 |
| **113** | - | - | - | - | 241,09 | - | **-** | **-** | C | 4 | 1 | C | 1 | 0 |
| **114** | 1 | 0,05 | 6 | 0,95 | 402 | 563,77 | **2,07** | **1,76** | C | 4 | 1 | C | 1 | 1 |
| **115** | 1 | 1 | 0 | 0 | 26,33 | 566,51 | **2,37** | **1,48** | C | 4 | 2 | R | 0 | 0 |
| **116** | 1 | 1 | 0 | 0 | 214,48 | 566,51 | **1,56** | **1,1** | C | 4 | 2 | R | 0 | 1 |
| **117** | 1 | 0,53 | 6 | 0,47 | 150,74 | 565,13 | **1,97** | **1,18** | C | 4 | 2 | R | 1 | 0 |
| **118** | 1 | 1 | 0 | 0 | 329,48 | 566,51 | **1,56** | **0,94** | C | 4 | 2 | R | 1 | 1 |
| **119** | - | - | - | - | 38,84 | - | **-** | **-** | C | 4 | 2 | C | 0 | 0 |
| **120** | 1 | 1 | 0 | 0 | 227,75 | 566,51 | **3,55** | **1,42** | C | 4 | 2 | C | 0 | 1 |
| **121** | - | - | - | - | 163,62 | - | **-** | **-** | C | 4 | 2 | C | 1 | 0 |
| **122** | 6 | 1 | 0 | 0 | 343,09 | 709,24 | **2,08** | **1,34** | C | 4 | 2 | C | 1 | 1 |

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**Data availability** All data is included in the Appendix.

**Code availability** Available on request.

## Declarations

**Ethics declarations** Not applicable.

**Consent to participate** Not applicable.

**Consent to publication** Not applicable.

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