Economic incentives for the regulation of the provision of bundles of ecosystem services in agroecosystems

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1. General introduction

Agriculture provides directly or indirectly many goods and services to humans. Some of them are marketed and have a monetary value (crops, straw, fodder, livestock etc.). Many don't have any marketed value, although they contribute to human well-being: the potential to sequester greenhouse gases and mitigate climate change, the pleasure to observe a beautiful landscape etc. These goods and services are widely apprehended through the ecosystem services' framework (MEA). This framework considers all the services provided by ecosystem to humans, no matter whether they are marketed or not. It considers ecosystems as contributing to the human well-being.

In the twentieth century in Europe, agricultural intensification resulted in an increase of the marketed output supplied by agroecosystems, along with a degradation of the environment and a decline in all non-marketed goods and services. Pesticide inputs, reduced crop diversity and landscape uniformisation, and the removal of many semi-natural areas (hedges, field margins, wetlands...) had dramatic effects on farmland biodiversity: insects and in particular pollinators (Deguines et al., 2014), farmland birds (Burel et al., 1998; Donald et al., 2001; Wretenberg et al., 2006), and even some farmland-specialist mammals (de la Peña et al., 2003; Pocock and 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. 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). Recent assessments confirm the general decline in regulating ecosystem services, which are non-marketed (MEA; EFESE).

A crucial point in understanding and solving the decline in ES are the **interactions among ecosystem services**. Common agroecological processes determine the provision of all ecosystem services in agroecosystems, and create multiple and complex interactions among them. Hence, it is impossible to disentangle the provision of one ecosystem service from the provision of others, and to make one vary without making the others vary too (Bryan, 2013). In this thesis, we refer to **bundles of ecosystem services** to capture the fact that their provision is interdependant.

From an economic point of view, the strong interactions among ecosystem services and the decline in regulating ecosystem services are related. First, in the absence of agri-environmental policies, only agricultural production generates profit and there is no financial incentives (at least in the short term) to provide more of the regulating ecosystem services. A farmer considering only his private profit thus has interest to focus on agricultural commodities production. The regulating ecosystem services are public goods in economic terms: their provision benefits to more agents than those who bear the costs. Second, even if interactions among ES are complex, agricultural production globally stands in a trade-off with the other ecosystem services, so that the agricultural intensification has caused a decline in non-marketed ecosystem services as a side-effect. Changes in agricultural management are therefore required to find a balance between commodity production and maintaining other ecosystem services at a sustainable level. In particular, in the case the European intensive agricultural landscapes, this means changes in agricultural practices (i.e. implementing non-crop habitat, decreasing the use of fertilizers and pesticides) rather than changes in land-use (via conservation areas). Many changes in agricultural practices are possible that are associated to various changes in ecosystem services: reducing the use of fertilisers and pesticides have different implications than the implementation of hedges, even if they both increase non-marketed ecosystem services.

To push farmers to change their agricultural practices, agri-environmental policies are gradually implemented, for example via the Agri-Environmental Schemes of the Common Agricultural Policy in the EU. These policies aim at increasing the provision of ecosystem services, by compensating the costs of their provision.

These policies face several challenges.

First, the objective in terms of ES is multidimensional: they must fight the decline in many ES. Because of complex interactions among ES, it is not straightforward to identify which options provide the most ES and should be targeted by agri-environmental policies. Targeting an increase in each ES separately may be problematic, as the solutions to increase the provision of the different ES may not be coherent. Policies solving the provision of ES separately are likely to be incoherent. For example, an incentive to increase one ES may have negative effects on other ES (Lindenmayer et al., 2012). This is for example true for the theoretical solution stemming from economic literature on public goods. The pigovian solution of pricing public goods at their willingnes to pay by society may not achieve the desired solution if the targeted bundle of ES doesn't exist. The type of interactions among ES determines the strategies to provide ES. In particular in the case of trade-offs, it determines whether two ecosystem services should be provided together by a common option reaching a compromise, or whether they should be provided through different options, for example by dedicating part of the area to the provision of one ES and the other part to the provision of other(s). Depending on the interactions among ES, the best strategy can be a combination of several management options. The land-sparing/land-sharing debate is one example of this question: should biodiversity be provided together with agricultural production, by adopting less intensive agricultural practices over the whole landscape, or should the compromise rely on a spatial segregation with more intensive agricultural practices on one side and the spared land preserved only for biodiversity on the other side.

Second, increasing the provision of regulating ES comes at a cost. This cost comprises the loss in marketed output (e.g. due to a reduction in fertiliser use) and additionnal costs due to agricultural management (implementing a hedge etc.). This is the opportunity cost, the cost of changing agricultural practices to increase the provision of ES. From a welfare perspective, the total cost of ES provision should be minimised, whether agri-environmental policies make the farmer (via taxes) or the society (via subsidies to farmers) support it. In reality, agri-environmental policies are generally implemented through subsidies, and the public budgets are limited. In such a case, the cost must be considered in order to achieve the maximal provision of ES (Naidoo et al., 2006). Forgeting the cost automatically selects solutions that achieve high levels of ecosystem services for a given area, but which can be very costly. Considering the cost enables to consider solutions that provide less ecosystem services per area, but at a lower cost. The latter solutions could be implemented over a larger area for a given budget, and in the end achieve higher total provision of ES. Hence, policies must trade off the provision of multiple ES against its cost. Given the interactions among ecosystem services, it is impossible to calculate the cost of providing ecosystem services individually, measuring the cost can only be done for a bundle of ES. A given change in agricultural management impacts all ecosystem services. All dimensions must be considered together when designing agri-environmental policies and the solutions they should target.

The third issue is that policies are generally implemented by means of economic incentives, which raises several problems. Beyond law and norms, policy makers cannot dictate agricultural management to farmers, they can only design incentives so that farmers choose the targeted option. This introduces participation constraints into the design of policies: in order to implement an option, incentives must make this option the most attractive for farmers. While agri-environmental subsidies aim at compensating the opportunity cost, the budget needed to respect the participation constraint (policy budget) may be greater than the opportunity cost. If the policy maker could dictate what to do to farmers, they could give them exactly the opportunity cost, the policy budget would be exactly equal to the opportunity cost. The policy budget measures the real amount of money spent for the agri-environmental policy. It can be higher than the opportunity cost. Given that the total budget is limited, this reduces the total provision of ecosystem services. The policy budget should be the criteria considered in the design of policies. Accounting for farmers' decision-making and the way incentives work is therefore crucial in the design of agri-environmental policies.
Another problem related to the use of incentives is to choose on which element incentives should be based. Typically, policy makers can choose between action-based incentives or result-based ones. Action-based incentives compensate farmers for income loss on the basis of actions they take, i.e. agricultural practices. Result-based incentives pay farmers on the basis of results achieved, i.e. levels of ecosystem services they provide. Most current agri-environmental policies in the EU rely on action-based incentives, for example Agri-Environmental Measures of the Common Agricultural Policy. Result-based incentives correspond to the theoretical solution to solve the underprovision of public goods in an efficient way (Mas-Colell et al., 1995), and have gained some appeal in the last years, in particular concerning biodiversity (Musters et al., 2001; Schwarz et al., 2008). The choice of incentives may have consequences on the policy budget, especially if incentives are somehow dependent on each other. Result-based incentives may interact with each other (Bryan and Crossman, 2013; Huber et al., 2017) and create problems for their calibration. Few answers exist on this issue: theoretical economic studies don't consider complex interactions among multiple ecosystem services, while studies accounting for these interactions don't conclude on the consequences in terms of policy budget. The interactions among ecosystem services could undermine the theoretical efficiency of

result-based incentives compared to action-based incentives.

The fourth issue is the presence of heterogeneity. ES provided and the associated cost and policy budget vary spatially, or even accross time or farmers. This heterogeneity makes the determination of cost-efficient or budget-efficient management options more complicated. Incentives can not be calibrated for each particular case, a certain degree of uniformity is unavoidable, whether it is because of asymetry of information or not. Two solutions exist: reduce the variability or adopt incentives that better adapt to this heterogeneity. Reducing the variability is possible through the definition of incentives at a smaller spatial scale, and calls for participative design or mechanisms that make farmers reveal their costs and ecosystem services provided. On the other hand, designing policies to cope with heterogeneity can be done through self-screening contracts or complex incentives. In particular, result-based incentives are often cited as being able to select cost-efficient options in heterogeneous areas (Gibbons et al., 2011). However, this argument relies on analyses which don't encompass multiple ecosystem services and their interactions, which could change this conclusion.

There remains a knowledge gap concerning the consequences of interactions among ecosystem services on the regulation of ecosystem services' provision in agriculture.

Adressing these challenges requires interdisciplinary research, in particular by interlinking appropriate concepts and tools of economics and agroecology. Current economic approaches on the provision of ES rely often on simplistic assumptions about the functionning of agroecosystems (Derissen and Quaas, 2013; Hasund, 2013; White and Hanley, 2016). On the other side, studies focussing on the agroecological side often overlook basic economic concepts such as the existence and accurate definition of costs, or the importance of the policy instruments used to implement the agri-environmental policies. Moreover, when both detailed agroecological processes and important economic lessons are integrated, the existence of many ES and their complex interactions is not always accounted for in interdisciplinary studies.

This thesis deals with the design of economic incentives aimed at increasing the provision of non-marketed ES, and focuses on the existence of multiple ES and their interactions. It is declined in different research questions echoing the four challenges:

1. in presence of multiple and interacting ES, how to determine which options maximise the ES with a limited budget

2. how to implement these options with economic incentives, given the interactions among ES

3. what does heterogeneity change to the conclusions of previous questions

This thesis is mainly anchored in economics, but is also interdisciplinary. The research questions at the core of this thesis are therefore treated with different perspectives. First, from the perspective of microeconomic theory, the questions raise issues related to joint production and the regulation of public goods. We use conceptual tools of microeconomic theory to explore them theoretically. Second, we rely on an applied modelling approach to simulate the ES provided by different management options and the associated cost, and the economic incentives needed to implement these management options.

The contributions of this thesis are therefore at the crossing of agroecology and economics. The first contribution is in theoretical economics. We study how joint products complexify the regulation of public goods. Joint production refers to the interactions among outputs, here we apply it to ecosystem services. Many theoretical analyses exist about the underprovision of public goods, and the way to solve it with economic incentives, i.e. achieve economic optimum. Some analyses deal with the consequence of joint production on the economic optimum. The rare analyses that combine both issues consider specific types of joint production, which don't reflect the variety of types of joint production among ecosystem services. We explore the consequences of types of joint production such as synergies or non-convex relationships on the regulation of public goods.

Second, we base our analysis of cost-efficient strategies to provide ES on an explicit representation of interactions among ES and of the cost of ES provision. This represents a contribution in terms of applied economics. Existing literature often tends to include interaction among ES in a very simple way (Derissen and Quaas, 2013; Hasund, 2013; White and Hanley, 2016).

Third, this thesis brings sound economic elements to the agroecology literature exploring maximisation of ecosystem services. We use tools of efficiency analysis as a way to identify options maximising ecosystem services. Above all, we refine current analyses of agri-environmental policies by integrating useful economic concepts like opportunity cost and considering explicitely participation constraints.

While it is anchored in economics, the analysis presented in this thesis explicitely considers agroecological constraints as a basis for economic analysis. In this sense, it is apparented to ecological economics: ecological processes are regarded as constraints determining the economic processes. Our agroecological modelling framework has several original features. It is based on the simulation of agroecological processes, and considers agricultural practices (rather than land use) as key drivers for the provision of ecosystem services. Interactions among ecosystem services are not specified a priori, they emerge from the simulated functioning of the agroecosystem. Similarly, the relation between the provision of ecosystem services and the cost is not specified with a function. We also don't seek to estimate these relations, which keeps their non-smooth characteristics and is probably more realistic. The economic modelling part of the applied analysis is simple, but highlights important aspects of economic modelling often overlooked. In particular, we underline the importance of determining the statu quo (i.e. the agricultural management in absence of the implementation of policies) in order to determine the cost of providing ecosystem services, and the importance of participation constraints. Another originality of this thesis is to articulate several levels of analysis. The theoretical analysis doesn't enable to conclude, so that we use more applied modelling to deepen the analysis, and last relate our results with recommandations stemming from real policy analysis.

This thesis is organised as follows: Chapter 2 reviews agroecological literature dealing with the quantification and representation of multiple ES; Chapter 3 presents what theoretical economic concepts and literature say about the regulation of joint public goods; Chapters 4 and 5 present respectively the model and simulated data used in the applied analysis. Chapter 6 uses simulated data to explore how to define management options maximising ES for a limited budget, while Chapter 7 asks the means of implementing a management option with economic incentives. Chapter 8 introduces heterogeneity and its effects on the definition and implementation of management options maximising ES under a budget constraint. Last, Chapter 9 discusses the results in light of the recommandations over current agri-environmental policies.

Part I.

State of the art

2. Literature review: assessments and representations of multiple ES

The concept of ecosystem service has been introduced as way to acknowledge that ecosystems provide benefits to humans. It has been used in environmental economics to account explicitely for the contribution of ecosystems in human well-being. Ecosystem services are often defined as "the benefits people obtain from ecosystems" (Millenium Ecosystem Assessment, 2005a).

Agroecosystems provide many ecosystem services. Research on the provision of ecosystem services needs to assess the consequences of alternative management options or scenarios in terms of ecosystem services. Researchers first need to quantify multiple ecosystem services associated to the management options considered. To do so, they rely on different methods, often using simulation models. Then, they need to represent these multiple ES accross the alternative management options or scenarios. For this second step, they use different types of representations. In particular, they use so-called production possibility frontiers which are also a mean to represent interactions among ecosystem services.

This chapter offers in Section 2.1 an overview of different ways to gather or generate quantitative data about multiple ES, and in Section 2.2 the solutions used to represent interactions among ES. In Section 2.3, we focus on the representation of interactions among ES by means of the production possibility frontier. Last, in Section 2.4, we review some shared conclusions from these studies.

2.1. Assessing and modelling the provision of multiple ecosystem services

A wide range of methods have been used to quantify the provision of ecosystem services in agriculture. In particular, most rely on models, which differ by their degree of complexity in terms of processes represented and by the drivers they consider. Few assessments rely only on indicators measured in field studies like Raudsepp-Hearne et al. (2010), first because of the lack of appropriate data, and second because models enable sometimes to better approximate ES than available field data: the percentage of forested land is a rather poor proxy of the recreational ES, while models may better approximate it. Modelling is also used to explore scenarios and alternative management options, and to integrate the future changes in drivers (among them climate change, as in Kirchner et al. (2015)). The drivers of ES provision are more or less detailed. Many authors rely on land use and land cover, while other consider more detailed drivers such as agricultural practices, climate scenarios or agricultural public policy scenarios.

2.1.1. Assessments based on land use and land cover

Land cover refers to the physical land type such as forest or open water whereas land use refers to the way people are using the land (cultivated, urban...). Both types of data are easily assessed via remote-sensing and satellite data, and included in GIS, so that they are handy to access and handle. Many assessments have therefore used such data to quantify the provision of ecosystem services. In this kind of assessments, land use is split into classes and the provision of ecosystem services is assessed on this basis. Diverse classifications exist and categories can be more or less precise: Nelson et al. (2009) distinguishes 9 broad land use and land cover classes, while Goldstein et al. (2012) uses 31 classes adapted to the ecoregion considered. Naidoo et al. (2008) use the GLC2000 classification developed by the JRC to assess carbon storage, which comprises 43 land use classes (from water to many detailed vegetation cover types).

A few studies then directly measure or gather data linking one type of land use to the provision of multiple ecosystem services (Raudsepp-Hearne et al., 2010; Fontana et al., 2013). But most authors then engage in modelling to infere ecosystem service levels from the land use and land cover, using various models. Chan et al. (2006) use very simple "benefit functions", combining land use data with other simple statistics. Many other assessments couple land use data with more or less complex simulation models. Bateman et al. (2013) and the UK National Ecosystem Assessment use a large dataset to estimate econometric relationships between ecosystem services and land use, which are not based on the underlying biophysical processes. Most other cases involve models which try to mimic biophysical processes. For example, the modelling suite InVEST gathers a whole collection of separate modelling units of ecological processes based on land cover maps, thus offering estimates of many ES (see box below). Nelson et al. (2009) and Goldstein et al. (2012) rely on it to assess alternative development scenarios of particular two regions. Similarly, Ruijs et al. (2013) simulate ES from land use data with several models: the integrated IMAGE model (MNP, 2006) and the GLOBIO model for biodiversity (Alkemade et al., 2009), among others. Modelling enables to simulate ecosystem service levels, but the issue in these models based on land use/land cover is that they fail to account for other drivers than land use, and they weren't built to consider dynamics, spatial interactions or feedbacks between ecosystem services (Seppelt et al., 2011). In this sense, Eigenbrod et al. (2010) show that using proxies based on land use doesn't provide a sound basis for assessment of ES.

Existing integrated models of ecosystem services: InVEST, TIM

Developed by the natural capital project, **InVEST** (Integrated Valuation of Environmental Services and Trade-offs) is a suite of models used to map and value ecosystem services. It comprises 18 distinct models of ecosystem services provision across terrestrial, marine and freshwater ecosystems (including for example carbon sequestration, timber harvest, crop pollination, marine water quality, nutrient and sediment retention etc.). The drivers considered in the models are mainly land uses and spatially explicit variables associated to the geography (climate, precipitations, soil characteristics, slope etc.), so that the models can be used together with GIS data, and can easily provide maps of ES. The aim is to provide decision makers with clues about the trade-offs and synergies among ecosystem services (both in biophysical and monetary terms), and to predict the multi-ES impacts of land use scenarios. The 18 models can be run according to several levels of precision. They are also distinct, making it easy for users to use the ones they are interested in, but this setting neglects any direct interaction among the provision of several ecosystem services. Possibilities of linking several modules exist however for marine and freshwater modules. For details, see Sharp et al. (2014).

The Integrated Model (TIM), developped in the UK NEA, follows the same logic, but gives even more emphasis to the economic valuation process. It includes the main ecosytem services of terrestrial ecosystems (agricultural and timber production, water quality, GHG emissions, recreation), and even biodiversity. The methodology differs much from biophysical models such as InVEST. Instead of modelling the biophysical processes which deliver ecosystem services, the models are econometric relations between a service and its drivers, estimated with available data. A detailed description is provided in Bateman et al. (2014).

2.1.2. Assessments based on more detailed drivers

Other assessments try to take into account more detailed drivers, and in particular agricultural practices. They are associated with integrated or coupled models, and generally more complex. They better represent agroecological processes and have the advantage to capture feedbacks between ES, as well as dynamic and spatial interactions.

For example, Bekele et al. (2013) rely on SWAT, an agricultural simulation model to assess several ES and include a mix of crop types and agricultural practices as drivers. Following the same strategy, Kragt and Robertson (2014) assess the impacts of two agricultural practices with a rather simple model coupling. Going fruther into complexity, Balbi et al. (2015) integrate several separate models into a unique coding system, and account for drivers such as agricultural practices (irrigation, tillage, and application of both organic and mineral fertilizers) and various environmental conditions. Groot et al. (2012) built their own integrated model to assess ES, which accounts for many variations in farming practices (different crop rotations and crop areas, amounts of green manure and feed crops, number of animals...).

Other authors rely on complex model coupling to assess multiple ES, taking into account very detailed drivers (Schönhart et al., 2011; Kirchner et al., 2015). In both cases, the authors are able to simulate the impact of many management variables, policy scenarios, and external drivers and derive spatially explicit ES indicators for their case studies in Austria. However, their models rely on very extensive calibration and require very large databases to do so. What is more, the separate models have been designed separately for another goal and may be already complex, and coupling them increases this complexity. In this sense, Carpenter et al. (2009) criticise the use of model coupling because it doesn't fit the scope and content of conceptual frameworks as the MEA. They plead for integrated models designed especially for the research question, that account for non-linear and abrupt changes and quantify trade-offs between ES, such as the one used by Groot et al. (2012).

2.2. Quantifying and representing the interactions among multiple ES

Ecosystems are characterized by close interactions and multiple feedbacks between organisms and their environment, as a result, the multiple ES present in agricultural ecosystems are interacting with each other. These interactions may be due to a common driver influencing the provision of several ES, or to a direct influence of one ES on the other (Bennett et al., 2009).

Most authors of multi-ES assessments study the links among the diverse ES and try to represent them together, using different techniques. They distinguish between two types of relations among ES: **synergies** (positive relations among ES) and **trade-offs** (negative relations). The exact definitions and the representations of these interactions differ according to the type of available data and the approach. For example a synergy can correspond to a spatial congruence among several ES (several ES are provided on the same places), or to the co-evolution of two ecosystem services (when one ES increases, the other also increases). And the same holds for a trade-off.

2.2.1. Flower maps

The most basic representation of multiple ES accross multiple scenarios or land uses is to use "spider webs" or "flower" diagramms: levels of the multiple ES are represented as bars or points on multiple axes starting from the center of the flower or the spider web. One diagramm is drawn for each land use or scenario, which are then compared. Without further analyses, interactions among ES are difficult to assess. An example of such diagramms, taken from Raudsepp-Hearne et al. (2010) is shown on Figure 2.1.

2.2.2. Spatial correlations and comparison of maps

Assessments aiming at comparing ES accross several development scenarios or different conservation schemes rely on spatially explicit data of ES provision over a landscape. Even if they could compare ES levels accross different management scenarios, they often study the spatial congruence of ES. Such approaches generally refer to synergies and trade-offs among ES in the meaning of (spatial) correlations or congruence.

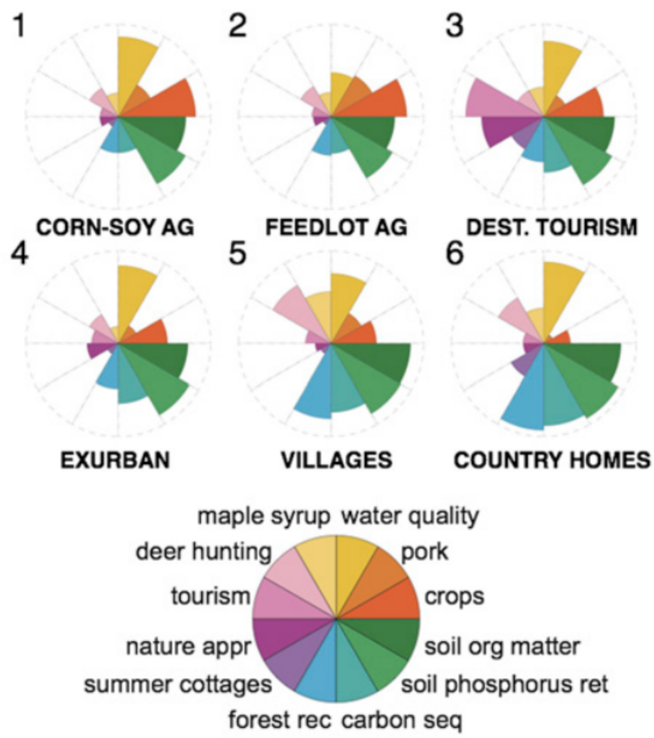

Figure 2.1.: *Flower diagramms in Raudsepp-Hearne et al. (2010)*

Chan et al. (2006); Naidoo et al. (2008); Ruijs et al. (2013) focus on finding the best conservation networks, and proceed in a rather similar way as above, simulating the outcomes in terms of ES levels in each spatial unit for different conservation scenarios. They get results in the form of maps of ES, and by overlapping these maps, they determine "hotspots" where many ES are provided simultaneously. These assessments also calculate correlations between the levels of ES provided accross all spatial units and scenarios. They are thus able to identify some trade-offs and synergies between ES, which are indeed spatial congruences and spatial mismatches between ES. As the correlations don't control for the specificity of each spatial unit and what drives it to provide the given ES levels, no conclusion can be made on what would happen to other ES if one ES were to be increased. Indeed, two ES like pollination and water quality may be spatially correlated (i.e. provided at the same place), but increasing pollination by sowing more pollinator-friendly flowers may not automatically result in increasing the water quality, because they depend on drivers that are not controlled for in the calculation of the correlation: all spatial units are compared with each other, no matter how heterogeneous they are.

Nelson et al. (2009) and Goldstein et al. (2012) study the ES provided by real landscapes under different management and development scenarios. They use InVEST model to simulate the ecosystem services provided in different scenarios of land use. To represent the results, they draw maps representing the change for each ES in each scenario and draw them side by side to be compared. These maps offer an easy representation of all consequences of every scenario over the whole landscape. In these two papers, the results are also analysed in terms of joint evolution of the different ES in each scenario, drawing some insights about trade-offs and synergies among ES. These trade-offs and

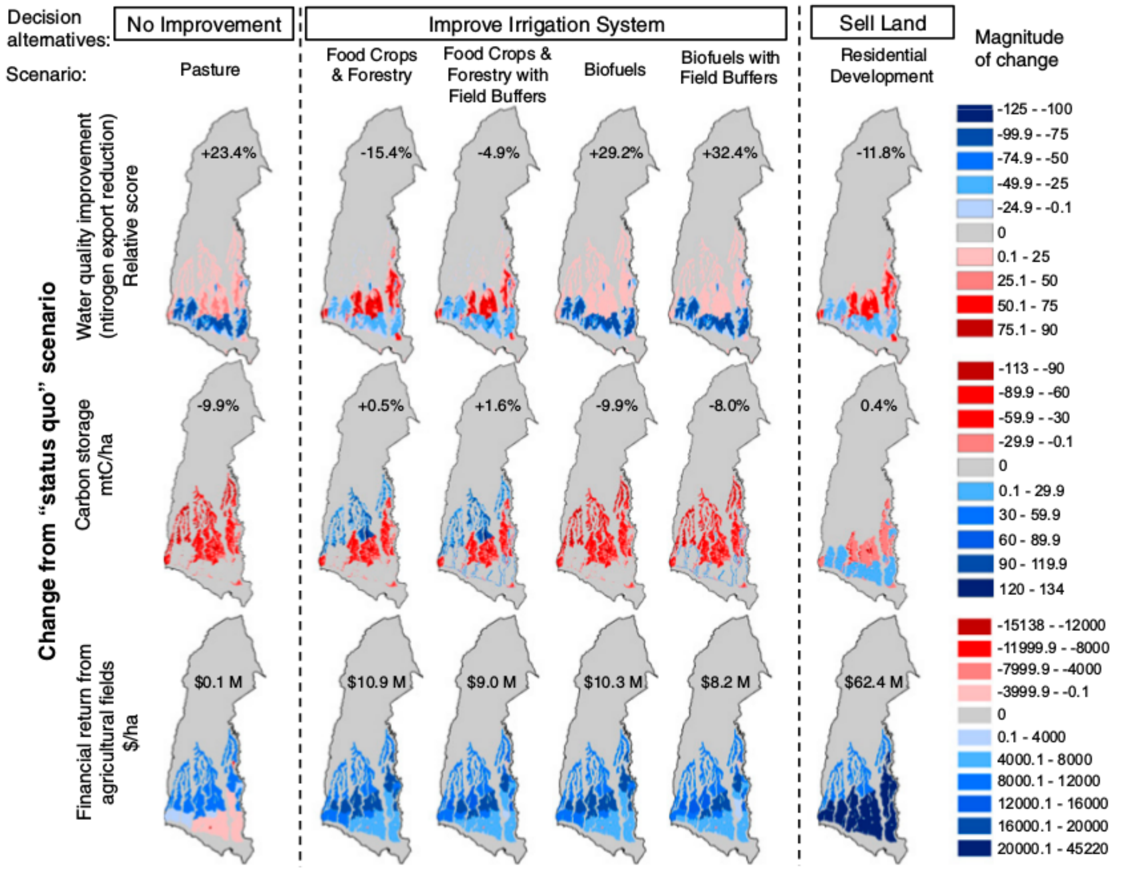

Figure 2.2.: *Maps of multiple ES in different land use scenarios in Goldstein et al. (2012)*

synergies identified depend of course on the landscape and the alternative scenarios, but contrary to the spatial congruences and mismatches, they inform rather well on the existing trade-offs and synergies incurred when choosing one or the other management scenario, because the alternatives compared are based on the same landscape. To illustrate, Figure 2.2 shows an example from Nelson et al. (2009).

However, both these approaches use simulation models that don't account for direct feedbacks between ES, and thus are not suited to analyse precisely the relations (synergies and trade-offs between ES).

2.2.3. Representations of the coevolution and production possibility sets

Production possibility sets are another way to represent the outcomes of alternative management scenario on multiple ES, which enables to better look at interactions among ES, and assess synergies and trade-offs more in the meaning of a co-evolution (the trend followed by one ES in reaction to an increase in another one). Production possibility sets are scatter plots with perpendicular axes representing the level of ES, and the outcome of each scenario plotted as a point in this space. The shape of the scatter plot reflects the trade-offs and synergies among ES, in the sense of coevolution: the trend of one ES when another varies. In particular, the non-dominated points (outer points) draw

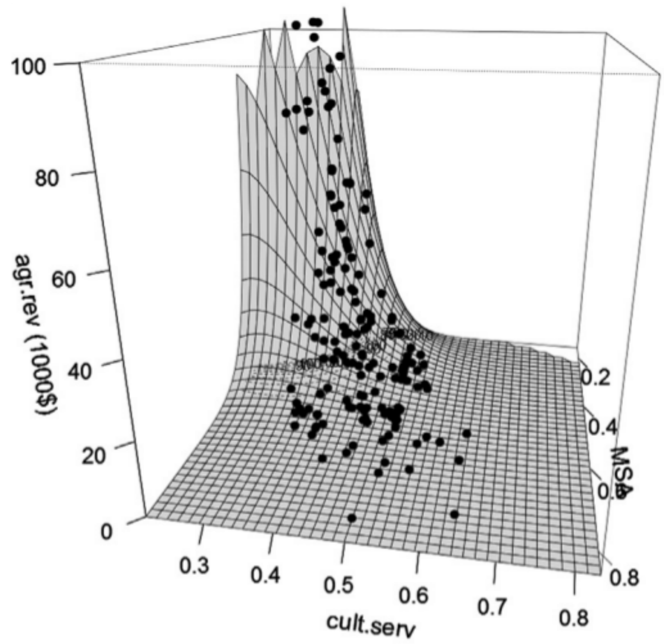

Figure 2.3.: *Production possibility set and estimated production possibility frontier in Ruijs et al. (2013)*

the so-called production possibility frontier, and represent the most interesting options, as they maximise all ecosystem services at the same time. The greater the number of possible alternative scenarios or management options included, the more complete the production possibility sets are, and the more representative the trade-offs and synergies are. By varying land use (Ruijs et al., 2013), agricultural practices (Bekele et al., 2013), or agricultural policy pathways (Groot et al., 2012; Schönhart et al., 2011; Kirchner et al., 2015), production possibility sets and frontiers give to see all possible outcomes and especially the efficient ones, and let trade-offs and synergies among ES appear. Figure 2.3 shows an example of a PPF found in (Ruijs et al., 2013).

2.3. Production possibility sets and frontiers applied to agriculture and ecosystem services

2.3.1. Use of PPF in the agricultural sustainability literature

PPF is a handy tool to represent trade-offs and synergies between ecosystem services, and possible outcomes of alternative management options. Using production possibility sets and frontiers meets several objectives.

First, the methodology is used to assess how management options score on several objectives: Kragt and Robertson (2014) consider only few management options to compare, but they are interested in their impact on several ecosystem services. By considering only few possible management options, their approach doesn't assess fully the interrelations among ecosystem services, but they show that some agricultural practices (as crop residue restitution) is beneficial for many ES. In particular, they are beneficial for both production and regulating ES, which globally stand in a trade-off[1].

Second, another objective of drawing a production possibilty set and identifying the frontier is to get an insight of its shape and of the underlying trade-offs and synergies. Using models simulating the outcomes resulting from spatially explicit combinations of land use, Polasky et al. (2008); Bekele et al. (2013); Teillard et al. (2016) assess the interrelations respectively between agricultural production and biodiversity, and among many ecosystem services [2]. The mere shape of the PPF serves then as a basis to analyse the interactions among ES. Teillard et al. (2016) use the shape of frontier to show that the strength of the identified trade-off among biodiversity and production varies according to the position in the production set, and to the set of management options considered (agricultural intensification, extensification, reallocation of land uses). Polasky et al. (2008) conclude that PPF between agricultural production and biodiversity is probably concave, such that a significant increase in biodiversity conservation can be achieved with a very limited loss of agricultural production. The same conclusion is reached by Bekele et al. (2013) for an index aggregating several ES. The same type of argument used in the land-sparing/land-sharing debate (see section 2.3.2). The shape of the PPF is also related to the economic theory, and some authors explore this link. Using theoretical biological functional forms, Brown et al. (2011) show that the production possibility sets can differ from the standard assumptions made in the microeconomic theory, and thus that the prescribed solutions can be ineffective or create contradictory effects. Ruijs et al. (2013) confirm this intuition. The authors estimate a production possibility frontier with 4 ecosystem services from simulated data, and identify that its shape is non-concave. As demonstrated by Brown et al. (2011), if the frontier is not concave, the determination of bundles that maximise ES cannot be made as easily. Groot et al. (2012) explore combinations of many management variables and simulate desirable characteristics of the management system (labor balance, agricultural output, environmental variables etc.) They use production possibility frontiers as a multi-objective optimization tool, even if dimensions considered are not all outputs in the strict sense.

Third, the interest of drawing the production possibility frontier is to identify the efficient bundles of ecosystem services, and the underlying configurations that lead to these bundles. Besides, if the current situation is comprised among the options represented, it is possible to compare it to the production possibility frontier. Since the frontier represents the efficient situations, it is a way to assess the efficiency of the reference situation. Groot et al. (2012) thus show that substantial improvements are possible simultaneously on every objective (profit, labor balance, nitrogen losses mitigation, organic matter balance), and identify which management options seem to be efficient. Similarly, Bostian and Herlihy (2014) explore in which extent wetland condition or agricultural production could be increased, either separately or together.

[1]Within a trade-off between two ecosystem services, inefficiency can make it possible to increase both ecosystem services simultaneously

[2]These authors explore and optimise over all possible landscapes to find the ones that provide the highest levels of production and environmental good. By doing so, they don't consider economic decision making, and can consider irrealistic landscapes (e.g. grassland or conservation land use on the most profitable lands), that cannot be achieved with economic incentives.

Fourth, some authors use the production possibility frontier to derive opportunity costs of providing non-marketed ecosystem services. This requires to approximate the frontier by specifying a functional form and estimating parameters, in order to get an differentiable equation of the frontier. Sauer and Wossink (2013) estimate a production possibility frontier with multiple inputs and multiple outputs, and look at the sign of partial derivatives to know if the agricultural output's value and the ecosystem service value (the payment for ES) show a trade-pff or a synergy. Bostian and Herlihy (2014) estimate the production possibility frontier of agricultural production and wetland condition and apply the duality theory to value improvements of wetland condition from the market value of agricultural production. Similarly, Ruijs et al. (2013) estimate the cost of providing non-marketed ES from the monetary value of lost production calculated with the slope of the estimated PPF. They use spatially explicit data, which enables them to go further and determine where the provision of ES is least costly in terms of agricultural production. All these estimated opportunity costs rely on the economic theory, which assumes that firms maximise their profit and usually run at full efficiency.

A fifth objective, related to the estimation of opportunity costs, is to use the slope of the frontier to assess which types of regulation instruments are needed to encourage the provision of non-marketed ES. Whether with theoretical developments (Smith et al., 2012) or with estimated frontier (Sauer and Wossink, 2013), the idea is rather simple: where increasing non-marketed ecosystem services translates into no loss, or a rather small loss of profit, information campaigns or technical advice may be enough to achieve an increase in ecosystem services. However, if the increase in ecosystem services comes with a sharp decrease in marketed commodities or a significant increase in costs, financial incentives are needed, which compensate for the (opportunity) cost of providing non-marketed ES.

Finally, PPF are a useful framework to create transdisciplinary discussions among ecologists and economists (Smith et al., 2012; Cavender-Bares et al., 2015). They are also able to gather representation of biophysical processes, as well as the representation of social preferences.

2.3.2. Links with the land sharing/land sparing debate

Articles arguing over land-sharing/land-sparing debate (Green et al., 2005; Phalan et al., 2011) use the shape of production possibility frontiers to derive land use strategies: a concave PPF implies land-sharing strategies while a convex one would stand for land-sparing ones.

This debate hides the sensitivity of such PPF to the hypotheses, the indicators of ES used, and to the spatial and temporal scale considered in the analysis. Moreover, the shapes of PPF considered are stylised, and in reality relations among ES can follow more complex shapes.

Kremen (2015) summarises the critisicms towards the land-sparing/land-sharing debate, and particularly the conclusions in favor of land sparing strategies. She underlines that the difference between the two strategies is often a matter of spatial and temporal scales: for example, according to the spatial resolution, forest patches in an agricultural landscape can be considered either as part of a land-sparing or land-sharing landscape. More interesting, papers finding superiority of land-sparing don't consider any feedbacks or interactions between the two types of land use (agricultural land use and conservation), whereas many different reasons for feedbacks exist: either via direct interactions (e.g. pesticides affecting neighbouring land), via land use change, displacement effects and so-called leakage, via global markets and price effects. It is also not sure that the land "spared" is actually used for biodiversity conservation, and indeed few examples really show this happening. To ensure that land-sparing strategies really translate into biodiversity conservation, studies need to go over a longer time period, because the response of species dynamics to land management is characterized by long time lags.

2.3.3. Links with eco-efficiency literature

Production possibility curves and frontiers are related to the (eco-)efficiency literature which stem from production theory in economics. The efficiency literature aims at measuring the efficiency of so-called decision-making units (for example, firms). Efficiency is measured relative to efficiency frontiers which are similar to production possibility frontiers. The efficiency frontier is composed of all efficient decision-marking units, defined as those for which no other unit generates as much output using less input, or generates more output with the same input. Efficiency is often measured through an output/input ratio, or with a score reflecting how much output can be increased (or input decreased) to achieve an efficient situation. The eco-efficiency literature derives from the efficiency literature, and integrates the use of natural resources or environmental variables as inputs or outputs in the efficiency measurement. The aim is to take into account the use of natural resources and the damaging environmental impacts in the production process, to assess if natural resources are used efficiently, or if the environmental impacts are reduced to the minimal possible level. The eco-efficiency is for example often measured by the ratio between output and environmental impacts, or the ratio between outputs and the natural resources used. The eco-efficiency score, as for the efficiency score, measures the proportional decrease in the use of natural resources or environmental impacts compared to eco-efficient production processes. The efficiency score is a measure relative to the efficiency frontier, and thus measuring efficiency scores requires to first estimate the efficiency frontier. An example of eco-efficiency analysis in agriculture is given by the paper of Beltrán Esteve et al. (2015). The authors assess the eco-efficiency of conventional and organic citrus farming, in terms of 6 different environmental impacts. Their results show for example that organic citrus cultivation is eco-efficient compared to conventional cultivation, but that within each technology (organic and conventional), the level of efficiency of the different farms are equivalent.

2.3.4. Limits to the use of PPF: irreversibilities

Simple representations of trade-offs (like spatial congruence and "flowers") are purely static and don't inform on the reversibility of coevolution of different ES. Neither do production possibility frontiers. As long as these tools are used to compare outcomes of future management scenarios departing from current situation, this is no problem. However, attention should be drawn on using them to compare future situations with each other or to predict outcomes of management options from another initial situation, i.e. moving along the PPF or going from one ES "flower" to another. Indeed, given the complex interactions, all outcomes may not be achievable from any initial conditions Brown et al. (2011); Smith et al. (2012). Irreversibilities may occur, e.g. once biodiversity has reached a low threshold, it may be difficult to enhance it again, even at the costs of trade-offs with agricultural production.

2.4. Stylised facts identified in the literature

The first general lesson from the analysis of trade-offs and synergies among ES is summarised by Carpenter et al. (2009): although some literature focus on win-win situations, trade-offs are the rule. More precisely, a general trade-off is identified between provisionning services and regulating and supporting services (Kirchner et al., 2015; Jiang et al., 2013; Raudsepp-Hearne et al., 2010; Goldstein et al., 2012; Nelson et al., 2009). This result is confirmed by the extensive literature review of Lee and Lautenbach (2016) and the meta-analysis of Howe et al. (2014). Many authors underline moreover that a substantial gain of regulating ES may be achieved for a small loss of provisioning ES (Polasky et al., 2008; Bekele et al., 2013; Balbi et al., 2015). Groot et al. (2012) even conclude that such a gain in regulating ES can be achieved with only minor changes in agricultural practices. This comes in line with some case studies showing that conservation agriculture can achieve provisionning services comparable to intensive agriculture (Badgley et al., 2007), even if this may require some entry costs to replenish the stock of SOM, semi-natural habitats. The relations among regulating or supporting services are quite context-dependant, conclusions are diverse. Nevertheless, assessments drawing PPF conclude to a positive relation between supporting ES (Ruijs et al., 2013). Wratten et al. (2012) also find a synergy between pollination potential, other ES like biological control, erosion control and aesthetic ES, and biodiversity. This is due to the fact that the main practices that favour pollination also benefit to other ES. More generally, the literature review of Lee and Lautenbach (2016) confirms that synergies seem to occur frequently among regulating services, and between regulating and cultural services. More precisely, the most robust synergies identified in the literature are the ones between habitat-related ES and most regulating ES, in particular soil formation.

However, reviewing the results in the literature, Bennett et al. (2009) states that most ES "were not good surrogates for one another" and authors relying on spatial congruence seem to find weak or no synergies among supporting ES (Chan et al., 2006; Naidoo et al., 2008; Ruijs et al., 2013), but conclude that locally, high spatial congruence and win-win possibilities exist. These results indicate that complex patterns may lie under the synergies, which are poorly identified by analyses relying on simple correlations. Indeed, as shown by Lee and Lautenbach (2016), the probability of identifying "no-effect" relationships among ES is increased by the use of correlation coefficients. Last, Bekele et al. (2013) show that non-linearities and non monotonous relationships arise very frequently with complex ecological interactions, while correlations implicitely assume a linear relation.

These conclusions are quite consensual in the agroecological literature, but contradictory results exist, that lead to identify some issues in building PPF. First, the range of options considered while building the PPF matters: Kragt and Robertson (2014) assess the impact of only two alternative agricultural practices (increasing the share of pasture in crop rotation and crop residue restitution rate) on production, carbon sequestration, N-mineralisation and groundcover. They find a potential synergy between production and regulating ES (carbon sequestration and N-mineralisation) when considering increases in the residue retention rate, but they find a trade-off when considering an increase of pasture phases. The wider the range of options, the more representative the relations identified. Second, the accurate definition of the spatial and temporal scale of trade-off assessment is important: in the long term, restoring the stock of soil organic matter is beneficial for agricultural production, but in the short term, it requires to increase input and decrease outputs of fresh organic matter with management options that can be in competition with production (reduced tillage, crop residue restitution...). Similar examples exist for the sensitivity to the spatial scale. Third, the relationships identified are of course sensitive to the measurement. For example, when agricultural production is measured by the agricultural income, the variability of the market prices of diverse crops can modify the measured agricultural production without any change in the biophysical variables.

We identified two interesting conclusion with this literature review. First, the use of models is helpful to assess the consequences of alternative management options and scenarios. More specifically, integrated models based on precise drivers such as agricultural practices are best suited to account for complex feedbacks among ES and assess interactions among multiple ecosystem services. Second, production possibility frontiers help represent these interactions and identify options maximising ES. They are also relate to economic analyses and current debates about the strategies to maximise ES.

3. The economics of regulating the provision of ecosystem services

This chapter presents how the regulation of ecosystem services in agriculture raises issues studied in economic theory.

A given agricultural landscape can provide many different bundles of ecosystem services, not only biomass for food, energy or other uses, but also regulating and cultural ecosystem services. Different management options provide different amounts of ES.

We describe the provision of ecosystem services as a production process, with one input being land and outputs being the ecosystem services. Management options (combinations of agricultural practices) represent the different production processes available, which each provide a different bundle of ecosystem services.

We detail in Section 3.1 which issues ecosystem services in agriculture encounter and in Section 3.2 the consequences for their regulation. We list some clues for their regulation in Section 3.3 and finally detail the knowldege gap and the contributions of this thesis in Section 3.4

3.1. Ecosystem services as joint public goods

Two issues make ecosystem services provided by agricultural landscapes interesting from an economic point of view (Wossink and Swinton, 2007). First, many ecosystem services are **public goods**, i.e. they benefit to more people than those who bear the costs of providing them (think of climate regulation). A wide stream of economic literature over public goods exist, and can be related to our analysis. The second reason is that the different ecosystem services are interdependant of each other, they are **joint products**. Increasing or decreasing one ecosystem service is likely to impact the other ones. That's the reason why we use the term **bundle of ecosystem services**.

The two issues have different implications. Public goods characteristics requires the regulation of their provision, and joint production adds constraints on it.

3.1.1. Optimal allocation

These economic issues are defined against an ideally operating economy. In theory, the optimal output bundle makes demand and supply functions coincide such that the marginal rate of substitution equals the marginal rate of transformation and the relative price of outputs. This optimal bundle maximises the welfare. In an example with two goods, this can written as follows:

$$MRS_{ji} = MRT_{ji} = p \tag{3.1}$$

The marginal rate of substitution MRS_{ji} represents the preferences of economic agents towards the two goods i and j, and more precisely how much good j they are ready to trade off against a marginal increase of good i. It equals to the ratio of marginal utilities, and thus represents the preferences of the agent, the demand side. The marginal rate of transformation MRT_{ji} represents the supply side and the production technology, i.e. how much of output j must be given up against a marginal increase the quantity of output i. p is the relative price, i.e. the value of output j relative to output i on the market.

The hypothesis of a perfect market and thus of unique price for all economic agents ensures the equality of MRS_{ji} and MRT_{ji} among all economic agents - at different levels of outputs if their utility and production functions differ. This hypothesis consists in three elements: (i) markets must be complete, with perfect information and no transaction costs; (ii) agents are price-taking, they cannot manipulate the price (no monopoly, no barriers to enter the market); and (iii) that local preferences are non-statiated locally, i.e. that it is always possible to find a bundle of outputs that agents prefer to the current one. On the supply side, this theoretical result also relies on another hypothesis, namely that the production function is quasi-concave, and that output are freely disposable: with a given production process, it is possible to produce less outputs than the maximal levels.

Compared to this ideal economy, the first issue encountered by ES in agriculture is that they are public goods, and thus the price p doesn't equal the marginal rate of substitution MRS_{ji} on the demand side. This is a violation of the hypothesis of completion of markets. The second issue is that joint production, which concerns the supply side, restricts the values taken by the marginal rate of transformation MRT_{ji}, so that the equality with MRS_{ji} transfers this constraint on the demand side. This is mainly related to the violation of the quasi-concavity and free disposability[3].

3.1.2. Public goods

Many ES are public goods: they benefit to all or at least many individuals, their use by an individual doesn't preclude the use by another (non-rivalry), and it is impossible or difficult to exclude anyone from their benefits (non-exludability). Because of the non-rivalry and non-excludability, their private provision constitutes a positive externality, which is not embedded in the decision of producers, and as a result, they are generally provided in smaller quantities than the optimum (Mas-Colell et al., 1995). As a consequence, their provision must be regulated.

For example, in the case of agroecosystems, carbon sequestration is a public good, as the mitigation of climate change benefits to everyone. Other outputs of agroecosystems are "impure" public goods, they have public goods characteristics, but are not totally non-rival or non-excludable. Pollination is partially non-rival and non-excludable: pollinators hosted in one field benefit to many neighbouring fields simultaneously, but the positive externality is limited by the flying distance of pollinators and their activity is limited by the size of their population. All ES except provisionning ones have public goods characteristics, even if few of them are "pure" public goods, i.e. are totally non-rival

[3] In environmental issues, it is often the case that a given production process can not produce less outputs than the maximal level, think for example of pollution

and non-excludable. Their private provision is a positive externality.

Economic theory states that public goods are underprovided in absence of regulation. In a framework with one public and one private good, each agent determines the allocation of the consumption and the production between the private and the public good by equating his/her marginal rate of substitution with the marginal rate of transformation. In the case of private goods, the market price reflects exactly the demand, and the sum of individuals' production or consumption is socially optimal.

Since the provision of public goods benefits to everyone, Samuelson (1954) shows that the socially optimal provision of public goods is determined by the equality between the *sum* of the marginal rates of substitution of all individuals and the marginal rate of transformation (Bowen-Lindahl-Samuelson condition). However, the real provision of public good is determined by agents considering only their own marginal rate of substitution and equating this MRS with the MRT. Hypotheses assume that the MRT is increasing in the output quantity while the MRS is decreasing, and the equality between individual MRS and MRT will be realised for a lower quantity of public goods than the social optimum (see figure 3.1).

In agriculture, the underprovision of public goods is well illustrated by the current unstainable level or decline in many regulating ecosystem services: pollination (Deguines et al., 2014), soil fertility (Stoate et al., 2001; Tan et al., 2005), water quality and climate regulation (Foley, 2005).

Because most ES have public goods characteristics, in the absence of any regulation, the bundle of ecosystem services provided by an agroecosystem doesn't correspond to the social preferences, and the regulator aims at correcting that with economic instruments (see section 3.2).

3.1.3. Jointness between ES

Another economic issue raised by the provision of ecosystem services is the jointness of outputs in the production process. Jointness in outputs means that the levels of the different outputs are not independent. These constraints on the supply side translate into constraints on the demand side. According Shumway et al. (1984), jointness occurs if the supply of one output depends on the price of another output. This definition is very broad, and applies to most production processes with several outputs. However, different types of joint production exist, which correspond to different situations and have different implications on the economic optimum.

An example of joint production of agriculture is the decline in many regulating ES caused by the increased provision of biomass due to agricultural intensification. Another example is the increased carbon sequestration due to reduced tillage, which is related to a better erosion control and an increase in other soil-related ecosystem services (Palm et al., 2014).

Jointness may come from two different causes (Havlík et al., 2005) : a fixed allocable factor or a non-allocable factor. In the case of a fixed allocable factor, the several outputs are produced with a common input available in limited quantity, which is allocated between the production of the different outputs. Producing more of one output requires to dedicate more input for its production, and thus less to the production of other outputs.

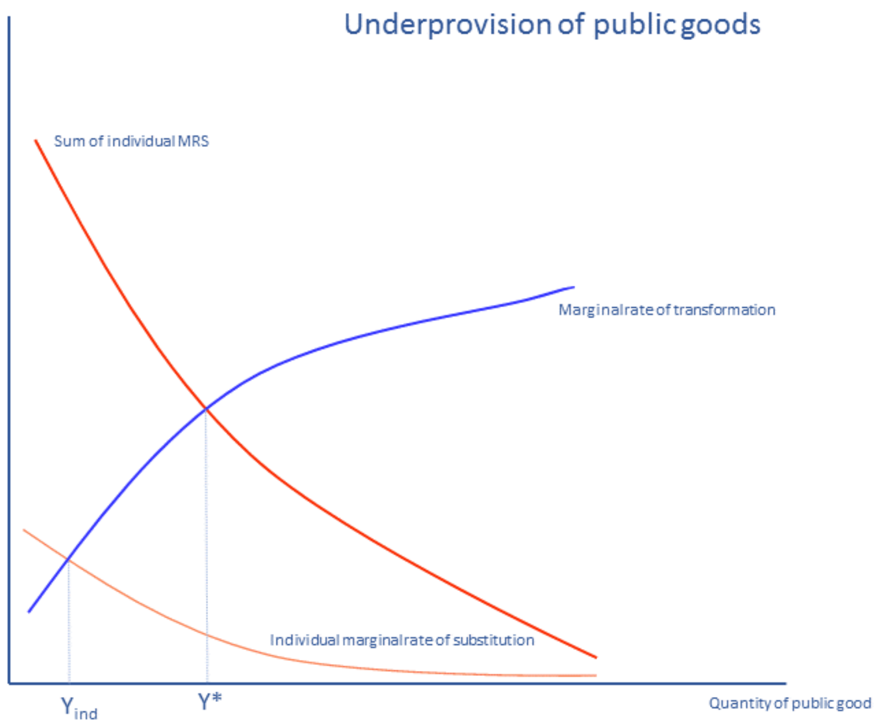

Figure 3.1.: *Underprovision of public goods*
Y^* denotes the socially optimal level of ecosystem service provided, while Y_{ind} stands for the level provided with all individuals optimising their private utility The marginal rate of substitution of each individual (light orange line) represents the amount of public good which has to be given up to keep utility constant while increasing the quantity of private good. The marginal rate of transformation (blue line) represents the amount of public good which has to be given up to keep input use constant while increasing the quantity of private good. As the public good becomes more abundant, the marginal rate of substitution (i.e. its marginal contribution to utility) decreases, while its marginal requirements in terms of input use inceases, driving the marginal rate of transformation upwards. The sum of the individual MRS (dark orange line) gives the amount of public good society as a whole is willing to trade off against an increase in private good, and its intersection with the marginal transformation rate determines the socially optimal level of public good Y^*. However, agents determine their individually optimal level of public good provision and thus choose level Y_{ind}, which results in a lower public good provision than the socially optimal level.

For example, in agroecosystems, ecosystem services are produced using land as common input[4], which is available in fixed quantity (Abler, 2004), so that increasing the provision of one ecosystem services by devoting more land to it (e.g. carbon sequestration) reduces the land available for the provision of other ecosystem services (e.g. agricultural production). This type of jointness results in trade-offs among ecosystem services. The second case, a non-allocable factor, arises when several outputs are produced with the same input, and it is impossible to disentangle the input's contribution to the production of each output. For example, a given field provides many ecosystem services simultaneously, it is impossible to determine which part of the field produces which ecosystem service, because it is impossible to produce one ecosystem service separately from the others. By varying agricultural management, it is possible to vary the levels of the different ecosystem services, but it is impossible to produce an ES without producing the others. This second case of joint production creates synergies among ecosystem services (e.g. synergy between pollination and erosion control). Boisvert (2001) distinguishes a third case, with similar implications, where one output contributes to the production of another, with the example of pollination and pollination-dependent crop production. In (agro)ecosystems, due to ecological interrelations, and the use of resources for competitive objectives, complex trade-offs and synergies arise among different ecosystem services, and providing more of one ES impacts the other ES (Bennett et al., 2009).

Jointness has an impact on output supply and input demand, and on the determination of economic optimum. In welfare economics, which deal with the determination of economic optimum and associated output levels, prices and input allocation, production functions are assumed to respect certain conditions. These assumptions make concave trade-offs the standard case of interrelations between outputs and are: i) fixed allocable input is the only form of joint production considered and ii) the higher the production of one output, the lower the additional output produced by an increase in input (diminishing marginal returns). The second condition means that for a given level of input, the higher the level of an output, the more severe is the trade-off with other outputs. These assumptions have the advantage to guarantee the existence and unicity of the economic optimum (the set of inputs, outputs and prices that maximises welfare). Other forms of joint production, like synergies among outputs, represent therefore a special case and have been studied as such by theoretical economists[5] Different authors show that synergies translate into additionnal constraints on the supply side and therefore must be reflected into constraints on the demand-side at the economic optimum. For example, Cornes and Sandler (1984) show that joint production (synergy) implies a relation between the marginal rates of substitution of the different outputs at the optimum: for each agent, the weighted sum of the MRS for each of the jointly produced goods should equal the MRT. It then translates into the optimal prices. To decentralise the optimum (i.e. implement the optimum by means of economic incentives), prices should then account for the relations of joint supply between outputs. In the case of the regulation of joints public goods, this implies that incentives are dependant from each other. Samuelson (1969) derives in the same way a geometrical relation between the demand curves in

[4]Alternatively, labour (working time) is also a common limited input

[5]as stated by Baumgärtner et al. (2001): "From then on, joint products were viewed from the analytical point of view – irrespective of their empirical relevance – as a 'peculiar case'".

the case of a perfect synergy. However, this work only considers the case of joint supply where two outputs stand in a perfect synergy, with fixed proportions, while in reality the relative proportions of outputs are more flexible.

The analysis of Shumway et al. (1984) allows for flexible forms and different causes of jointness and refers explicitely to the agricultural sector. It underlines that jointess due to fixed allocable input or interdependant production process implies a supplementary difficulty in determining and modelling the optimal allocation of inputs. Namely, it states that as soon as some joint production exists in the form of non-allocable input, it is impossible to specify separate production, profit and cost functions for the different outputs, and thus each input demand function and each output supply function is a function of all input and output prices, as well as input quantities. This conclusion also implies interrelations among incentives in the regulation of joint public goods. However, the authors underline the implications for the econometric estimation of production functions, and don't deal with the regulation of joint (public) goods.

Some authors compare theoretical implications of joint supply and public goods (e.g.Samuelson (1969)). However, they don't mix the two issues. Moreover, they only consider perfect synergies when dealing with joint supply, and other rather irrealistic assumptions about preferences of consumers. For example, to compare both settings, Ellickson (1978) assume that in the case of joint products, consumers only want to consume one of the joint products. Therefore, these theoretical works don't inform much on the combined stakes of joint production and public goods. The rare studies on the regulation of joint products rely on theoretical forms of production functions, which drive the results they obtain (see for example Cornes and Sandler (1984)), or conclude that the regulation of joint public bads requires unrealistic conditions (Baumgärtner et al., 2001).

However, it is interesting to note the conclusions of Holmstrom (1999): if two outputs are joint products, and one of the outputs is not perfectly observable (the measure can be manipulated, or doesn't capture the output well), it can be more effective to regulate the other output to achieve the desired bundle of outputs. Peterson et al. (2002) studies the multifunctionnality of agriculture through joint production and deals with regulation, and also focuses more on the implication of the fact that some outputs (environmental outputs for example) are non-observable. He proposes a different solution, which is to base the regulation on allocable inputs, regulating them separately according to the output they are dedicated.

Lessons from the literature on joint production Even when mixing the two issues, these last papers rely on restricting hypotheses about the production or transformation function linking outputs together, which drive their conclusions. We believe more realistic functional forms are needed to study ecosystem services in agriculture, but still keep in mind two important conclusions emerging from this literature

1. joint production, and in particular the shape of the transformation function, imposes new constraints on the demand side in order to achieve an optimal production of outputs

2. the supply function of a joint output depends on the prices of all other joint outputs

The first point is especially important since the shape of the transformation function driving the economic optimum is determined by agroecological processes and has thus no reason to comply with theoretical economic assumptions. The implication is that it is pointless to make the relative price coincide with the marginal rate of substitution in order to achieve the economic optimum if the corresponding marginal rate of transformation doesn't exist. In the following chapters, we adopt thus a two-step approach, where we take agroecological constraints has a starting point for the economic analysis, unlike many theoretical works in environmental economics.

The second point has an important implication for the instruments to regulate joint outputs, which we develop in the next subsections.

3.1.4. Jointness and the link with production possibility sets and frontiers

In the following paragraphs, we leave the demand side (preferences) aside, and thus move from "optimal bundles of outputs" (maximising outputs jointly and utility) to "efficient bundles" of outputs (only maximising outputs jointly). We assume implicitely that any efficient bundle of outputs corresponds to some preferences (maximises utility for these preferences).

More recently, a less theoretical literature has emerged which considers the problem of joint production of ecosystem services in agriculture via production possibility sets and production possibility frontiers (Smith et al., 2012; Bekele et al., 2013; Ruijs et al., 2013; Sauer and Wossink, 2013; Bostian and Herlihy, 2014). Production possibility sets and frontiers are a way to represent all the possible bundles of outputs together, without necessarily specifying a functional form for the joint supply of outputs. Their analysis also focuses on the supply side, and doesn't require the specification of a demand or utility function.

The production possibility set represents on a unique diagramm all combinations of outputs that can be produced with a given amount of input(s), with axes being the outputs. Each bundle represents a possible production process, and the whole set represents the available technology. The points lying the most towards the top-right corner, which form the boundary of the production set, belong to the production possibility frontier. These bundles are **efficient**: for each of them, there is no other bundle achieving better on all outputs. On the contrary, all other bundles of ecosystem services are inefficient: there exist for each of them a bundle providing a higher level of at least one ecosystem service without decreasing the other(s)[6].

The shape of the production possibility frontier reflects the interactions between the outputs, i.e. the constraints dictated by agroecological processes. Its slope corresponds theoretically to the marginal rate of transformation (MRT). For example, a downward-sloping curve indicates a trade-off (producing more of one output means less of the other), while a synergy or complementarity corresponds to an upward-sloping set and a

[6]We consider only good outputs and no bad ones, but this doesn't change the conclusions: bad outputs such as pollution can be considered as good ones such as environmental quality with a conceptual effort. Minimising pollution is equivalent to maximising environmental quality, the bundles maximising outputs are desirable and efficient bundles are the ones lying the most towards the upper-right corner.

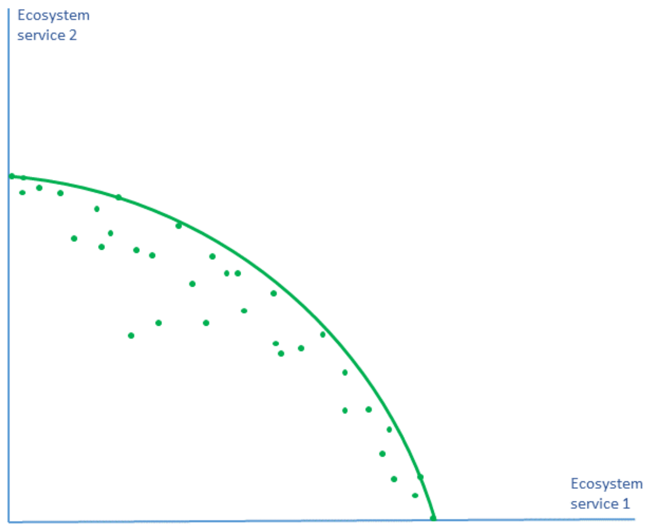

Figure 3.2.: *Example of a production possibility set and frontier of two outputs standing in a trade-off*

very narrow frontier.

For example, figure 3.2 represents a production possibility set and frontier with two outputs standing in a trade-off.

By definition, the production possibility frontier cannot be increasing: on an increasing PPF, one point necessarily would have higher levels of both outputs, which is in contradiction with the definition of efficient bundles belonging to the PPF[7]. In the standard framework of the microeconomic theory (Mas-Colell et al., 1995), three assumptions are generally made that allow for the standard shape of PPF represented on figure 3.2. First, the frontier is assumed to be concave (the set is assumed to be convex), in agreement with the assumption of diminishing marginal returns and the idea that "unbalanced" bundles are more costly to produce than "balanced" ones. Second, the assumption of free disposability ensures that interior, non efficient bundles of outputs exist. A third property of the production set is often assumed, additivity, which implies that the linear combination of any two bundles belonging to the production set belong also to the production set. All these assumptions also ensure that the production possibility set and frontier are complete, ranging from one axis to the other. The frontier on figure 3.2 complies with these assumptions. Haight (2007) derives theoretical ecological-economic production possibility frontiers which also comply to these assumptions. With simulated agroecological data, Bekele et al. (2013) estimate such forms of PPF between agricultural production and water quality.

However, the production possibility set is determined by the "technology", i.e. the biophysical processes in the case of ecosystem services, which lead to other forms of interactions than the classic trade-off described above. Many ES have common biophysical drivers and are related through ecological processes, and this creates complex trade-offs

[7]However, some authors estimating frontiers use a looser definition and consider also upward-sloping or backward bending frontiers (Havlík et al., 2005; Smith et al., 2012; Sauer and Wossink, 2013)

and synergies among them. For example, agricultural practices such as planting hedges or implementing flower strips are beneficial both for erosion control and pollination, and as a consequence these ES are likely to show a synergy (Wratten et al., 2012). Moreover, in the case of a trade-off, nothing in the biophysical processes guarantees that the resulting production possibility frontier complies with the hypotheses above, and is thus concave and complete. Threshold effects may also happen in ecological functions, as shown by Brown et al. (2011), creating abrupt changes and non-continuities in the shape of the production possibility set.

3.2. The regulation of joint public goods

Public goods are underprovided because their private provision is a positive externality, and the economic theory proposes several regulation instruments to correct the externality and achieve a higher provision of public goods. These regulation instruments can take various forms: economic incentives, norms, instruments relying on information or education etc. Here we focus on economic incentives, and consider two different types. First, the regulator can set output-based incentives (subsidies or taxes), which "internalise" the externality into producers' payoff, and thus integrate it in their decisions. Second, especially in the case of agriculture, the regulator often relies on incentives based on the agricultural practices, or input-based incentives.

However, joint production adds constraints on the supply of outputs, and thus interfere with the regulation of public goods. In this section, we review the two types of incentives and the issues caused by joint supply.

3.2.1. Solving the underprovision of public goods

Result-based or output-based incentives The theoretical litterature on public goods identifies result-based (or output-based) incentives as the solution to their underprovision. These incentives work by subsidising the provision of the public good (the output, or result of farmers's management) itself, i.e. giving it a price. They change the relative price of outputs, and thus the bundle of outputs which maximise the producer's profit. Ideally, to achieve the economic optimum, they must be calibrated according to the preferences (demand-side), so that the public good is provided at the exact level where the relative price equals both the marginal rate of transformation and the marginal rate of substitution. For example, to give farmers incentives to provide habitat for pollinators, the regulator can set up a subsidy which is proportional to the level of pollination potential provided by the farmer. Providing pollination service becomes more profitable, and the farmer will have an incentive to chose to increase the level of pollination potential of the output bundle. In theory, the rate of the subsidy should equal the positive externality of providing the public good. Subsidising the provision of public goods exactly mirrors the pigouvian tax in the case of a negative externality (Pigou, 1932).

By rewarding producers for the provision of an output, the regulator changes the relative price of outputs, and pushes the producer to provide more of the subsidized one. All bundles of the production possibility set are obtained for the same level of input and thus have the same production cost. Assuming that prices are fixed, maximising the profit is equivalent to maximise the outputs. Hence, a profit-maximising producer will choose an efficient bundle of outputs, i.e. that jointly maximise outputs. Graphically, all bundles of outputs associated to the same revenue (and thus profit) can be represented as a decreasing line (isoprofit line, in red on the figure 3.3), which slope is the relative price of outputs (the ratio of output prices). Maximising the profit given the achievable bundles of outputs means finding the bundle of the production possibility set which is on the highest isoprofit curve. By changing the slope of the isoprofit curve and the relative price of outputs, economic incentives change the most profitable output bundle, and then the quantities of the outputs provided (orange arrows on the axes).

This illustrates the second conclusion identified in the literature on joint production: relations among outputs impact the regulation of public goods by means of result-based incentives. In presence of joint products, the supply of one output depends on the price of all others. Result-based incentives work by changing the price of outputs, and therefore each result-based incentive impacts all joint outputs. Interactions among outputs (stemming from joint production) translate into interactions among result-based incentives. The several types of joint production won't have the same impact on the regulation of public goods by means of result-based incentives.

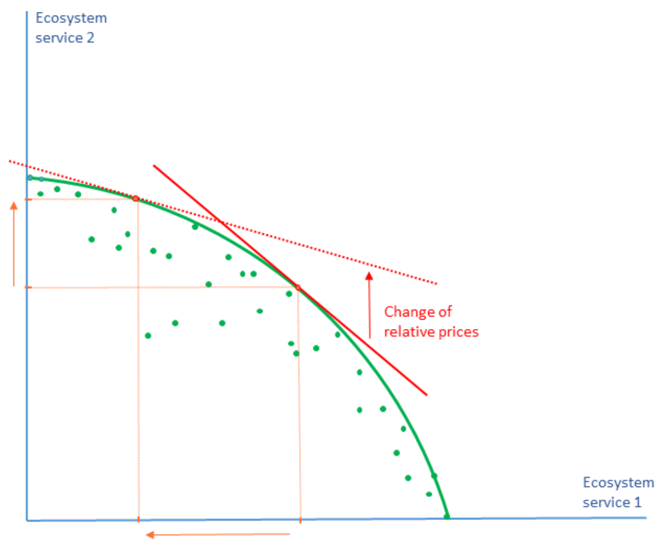

Figure 3.3.: *Example of a production possibility frontier corresponding to a standard case (concave trade-off)*

As illustrated on figure 3.3, if the production set is convex (and the frontier is concave), every efficient bundle of outputs is profit-maximising for some positive price (Mas-Colell et al., 1995). In other words, in the case of a concave production possibility frontier, producing more of one output (e.g. the public good) means producing less of the other(s), and by varying the relative price of the outputs, all bundles of the frontier can be achieved. Each relative price corresponds to one unique bundle on the frontier. As a consequence, in this standard case, result-based incentives work well.

However, we saw that this shape of PPF is restricting, as it relies on assumptions of convexity and completeness that may not hold in practice for agricultural outputs and

ES. Among others, it excludes the existence of synergies among outputs, which lead to non-complete production frontiers, and of non-convexities in the production set.

Action-based or input-based incentives Contrary to the logic of output-based (or result-based incentives), an alternative is to encourage production processes that provide much public goods. Thus, it means to favor directly some bundles of outputs with higher levels of public goods by subsidising the process itself. The term "action-based" incentive refers to the fact that these incentives target directly the production process, the actions taken by the producer, i.e. the agricultural practices. In pratice, this corresponds to the perspective taken by the Agri-Environmental Schemes in the Common Agricultural Policy in the UE, or to taxes on pesticides and fertilisers: the incentive targets the agricultural practices, and is in fact independant of the (environmental) output. Thus, to provide efficient bundles of output with action-based incentives, the regulator must know which actions provide efficient bundles of outputs, which is not the case with result-based incentives.

Action-based incentives are not studied much in the economic theory, but they are sometimes included in comparison with output-oriented incentives, either in the context of asymetries of information (Bontems and Bourgeon, 2000), or in the regulation of non-point source pollution (Shortle et al., 1998).

3.2.2. Non-standard production possibility frontiers and result-based incentives

There are many clues that agroecological processes don't correspond to a concave and comprehensive PPF, covering bundles with a large diversity of proportions of outputs. In particular, some authors have estimated PPF from ecosystem services assessments, and found ocurrences of synergies among ES and non-convexities in the production set (Ruijs et al., 2013). This echoes the first conclusion identified from the literature on joint production in subsection 3.1.3: joint production imposes new constraints on the supply side, which are passed on the demand side. This affects the functionning of result-based incentives.

Two types of issues can arise that make the frontier different from the well-behaved theoretical case, and interfere with the way result-based incentives regulate the provision of public goods.

First, if the outputs are characterized by a synergy, the production set will align along an upward-sloping curve, and the production possibility frontier will be very narrow. In this case, changing the relative price will not have many effects, as few bundles are efficient, and thus few bundles can be achieved by changing the relative price of outputs. Not every price corresponds to a marginal rate of transformation. The more complementary the outputs are, the less changing prices can change the respective level of outputs produced. Geometrically, it would correspond to a frontier limited to a vary narrow bow (see figure 3.4). In this case, standard price incentives change the bundle of outputs, but they must be calibrated with care to avoid ineffective changes in relative price. The issue with result-based incentives in the case of a synergy lies in the fact that the proportions of outputs cannot vary much (the marginal rate of transformation is very constrained) rather than in the inefficiency of the incentives itself.

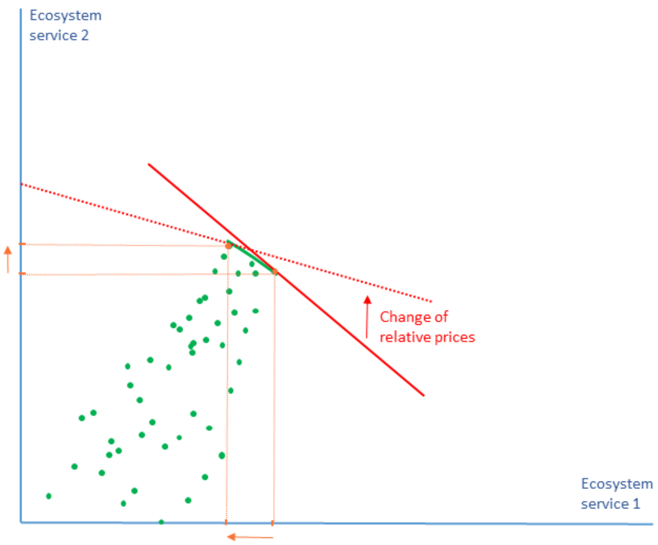

Figure 3.4.: *Example of a synergy between outputs and the limited effectivity of result-based incentives*

Second, ecological dynamics may lead to a non-concave frontier (non-convex production set). For example, Brown et al. (2011) shows that non-convexities in the ecological transformation function arise easily in ecological models, and that it could hinder the effectiveness of incentives based on outputs, or even lead to the worse solutions being chosen. Figure 3.5 provides an example where changing the relative price can lead to several solutions, with one of these solutions being inferior to the reference situation. In this case, the hypothesis of additivity is violated: not every linear combination of bundles belonging to the production possibility set belongs also to it.

The issue of non-convexity has long been identified as an issue in the efficient allocation by competitive markets. Chavas and Briec (2012) cite Guesnerie (1975) and states that "non-convexity is known to have adverse effects on the ability of competitive markets and decentralized decisions to support an efficient allocation."

Action-based incentives are not much studied in the theoretical literature over the regulation of public goods, neither the consequences of joint production. However, as action-based directly target production processes, it is likely that the relations among outputs caused by joint production don't affect their functionning.

3.2.3. Participation constraint, policy budget and second-best policies

Another aspect is left aside by theoretical analyses on the regulation of public goods, and could raise other problems for the regulation of joint public goods. It concerns both result-based and action-based incentives.

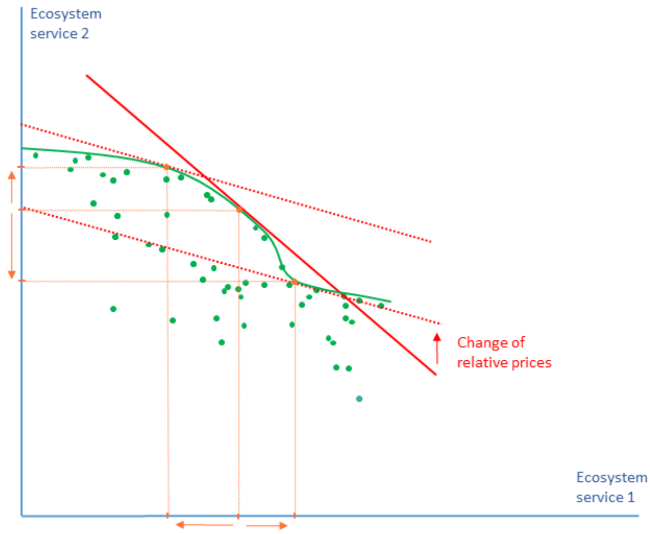

Figure 3.5.: *Example of a non-convex production set and the potential conterproductive effects of result-based incentives*

The theoretical framework presented above assumes that every cost stems from an input, and that all comparable production processes represented on the production possibility set use the same level(s) of input(s). Thus, it is assumed that all bundles of outputs bear the same production cost. In reality, changing the bundles of outputs without changing any variable associated to a cost is very restrictive. To study the regulation of the provision of public goods in agriculture, the framework must be more flexible, and allow the comparison of different management options that bear different costs, because of different levels of input-like variables (such as fertiliser intensity). In most studies estimating production possibility frontiers in agriculture, the only fixed input is land, and thus the production possibility set compares all options achieved with the same amount of land but various land uses or agricultural practices (Ruijs et al., 2013; Kragt and Robertson, 2014). Implicitely, these studies allow for various management costs (different land uses or agricultural practices bear different costs).

Allowing costs to vary among bundles of outputs makes the regulation of public goods more complex with result-based incentives. It also complexifies the regulation with action-based incentives if costs depend on other variables than those on which these incentives are based. To explain it, we need to introduce some new definitions. The regulator cannot dictate the bundle of outputs the producer must choose, he can only give incentives that make the producer choose this bundle. This is the **participation constraint**. We assume that the producer is profit-maximising, so that the participation constraint implies that the bundle chosen by the producer is the most profitable one. To illustrate this point, think that the set of incentives defined applies to every possible bundle of outputs and modifies its profit. The participation constraint is based on the ranking of bundles in terms of profit. Let us define the **opportunity cost** as the cost for the producer to change the bundle of outputs and provide public goods, and the **policy budget** the sum of incentives needed to make a producer choose a given bundle. The participation constraint implies that the sum of incentives is at least equal to the opportunity cost, but it can be greater.

When the production of any bundle is equally costly, maximising the profit is equal to maximising the revenues from outputs, and result-based incentives make profit-maximising farmers choose efficient bundles of outputs. In other words, the opportunity cost is simply equal to the difference in revenues from the outputs. To respect the participation constraint, the incentives must only compensate this difference related to the levels of outputs. There exist a set of result-based incentives that makes every efficient bundle of outputs the most profitable, so that respecting the participation constraint only requires a policy budget equal to the opportunity cost.

Differentiated production costs imply that farmers choose the bundle of outputs which not only maximises the revenues, but also minimises the production costs. It doesn't change the property that result-based incentives make farmers choose bundles which maximise ecosystem services, and result-based incentives now target cost-efficient bundles of outputs. However, when the cost of providing bundles of outputs varies, the opportunity cost also comprises the difference in production costs, and incentives must also compensate for this difference to respect the participation constraint. Result-based incentives change the profit of a bundle according to their output levels, but the production costs don't necessary vary according to the outputs. Hence, it may be that the set of incentives needed to implement one bundle leads to a policy budget higher than the opportunity cost, in order to respect the participation constraint (i.e. to make this bundle the most profitable). This is especially likely if the production cost of this bundle is high. Action-based incentives are also subject to the same effect: not all costs are related to agricultural practices, so that it is not straightforward to compensate production cost with action-based incentives.

The distinction between opportunity cost and policy budget leads to distinguish between **first-best** solutions and **second-best** ones. First-best solutions refer to solution that don't account for the participation constraint (i.e. cost-efficient bundles), while second-best solutions explicitely include the participation constraint (i.e. solutions which maximise ecosystem services while minimising the policy budget).

Accounting for the participation constraint via the policy budget is important, since it is possible that options that are cost-efficient are indeed very costly to implement with incentives. In particular, interactions among ecosystem services translate into interactions among result-based incentives, and could impact the policy budget.

3.3. Some elements from connex literature

Few theoretical economic studies mix the regulation public goods and joint production, and they don't deal with interactions among incentives and participation constraints. In this section we review some literature dealing with one or the other aspect of the regulation of ecosystem services in agriculture.

3.3.1. Comparison of both types of incentives

Some studies compare result-based and action-based incentives to regulate the provision of ecosystem services in agriculture and account for participation constraints. The first ones focus on the consequences of asymetry of information. The second group studies the case of the trade-off between agricultural production and biodiversity. We review them, and show that they don't account for every case of joint production.

Focus on asymetry of information The first group looks at asymetry of information and uncertainty (Shortle et al., 1998; Bontems and Bourgeon, 2000; Derissen and Quaas, 2013; White and Hanley, 2016). In economics, asymetry of information refers to the fact that the agent (here the farmer) has more information than the regulator. The asymetry of information in this case concerns the costs of providing the ecosystem services for the farmers, and the real actions taken by them. This information is either unknown from the regulator's point of view, or too costly to observe or measure. Asymetry of information can take two forms: adverse selection (due to asymetry of information, the regulator cannot select the cost-efficient solutions), and moral hazard (the regulator cannot be sure that agents do what they declare to do). For example, water pollution by agricultural inputs is a non-point source pollution and it is very difficult and costly to measure the contribution of each farmer to the pollution. This asymetry of information creates adverse selection because it is difficult to target areas or farmers where action is cost-efficient. If the asymetry of information concerns actions taken by farmers (it is difficult or costly to know which actions are taken), it creates moral hazard and the regulator is likely to subsidise farmers who don't take the actions they are supposed to.

Theoretical studies on this subject make use of principal-agent models, where the regulator is the principal and lacks information on the heterogeneous costs incurred to farmers for providing ES or the actions they really take. Most studies find that the two types of incentives have contradictory characteristics. Input-based regulation give incentives to farmers to reveal their true effort, and output-based regulation pushes them to reveal their true costs. Thus, White and Hanley (2016) find that the informational rent due to adverse selection is reduced in more cost-effective way by input-based incentives: when the regulator doesn't know if providing ES is costly for the farmer (in terms of land or effort productivity), regulation based on inputs is better adapted because inputs (such as pesticides or fertilisers) are easier to control. Moreover, in the presence of uncertainty on the outcome of actions taken by farmers, input-based also differ from output-based schemes in that they better reduce uncertainty for farmers, and thus risk-averse farmers prefer input-based incentives (Derissen and Quaas, 2013).

Similarly, Bontems and Bourgeon (2000) conclude that "input-based and output-based contracts may produce opposite incentives that the principal can exploit", and indeed all authors find that mixed strategies are more cost-effective.

However, these studies are based on very simplistic equations concerning the provision of ES. Derissen and Quaas (2013) for example assumes a linear relationship between a variable called effort and the results in terms of ES, and assumes that the farmer only provides one non-marketed ES. Because complex interactions among outputs impact the regulation with result-based incentives, these assumptions are likely to biais their results and overestimate the advantage of result-based incentives.

Contrary to this literature, we aim at considering multiple ES and their complex interactions. The asymetry of information between the farmer and the regulator is beyond the scope of this thesis.

Focus on the trade-off between agricultural production and biodiversity Other analyses are based on more precise representations of agroecological processes. For example, Sabatier et al. (2012) use an ecological-economic model to address the comparison of result-based and action-based constraints. They consider production and bird survival, which stand in a complex trade-off. Their results show that result-based constraints not only enable better economic performance, but also a better ecological performance. The key in this double benefit is the flexibility that result-based incentives allow for farmers, to better cope with fluctuations in the biophysical processes, due for example to precipitations and other external factors. Here, the relationship between production and biodiversity emerges from a quite realistic model, which allows for non-linear interactions. However, they consider only two objectives standing in a trade-off, agricultural output and biodiversity. They also don't consider participation constraints, since their approach is not based on incentives.

Gibbons et al. (2011) use a rather detailed model of costs related to the provision of biodiversity, and compare action-based and result-based incentives in heterogeneous landscapes. They discuss the advantages of the two types of incentives according to the type of heterogeneity, the sensitivity of biodiversity to conservation action, and the monitoring cost (the cost incurred by controlling that farmers comply with the conditions). The analysis includes a detailed representation of costs and participation constraint, but the only objective is biodiversity and the relation between profit and biodiversity is very smooth.

3.3.2. Interactions among result-based incentives

Last, some authors studied possible interactions among result-based incentives, and try to conclude on the consequences of these interactions. They account for participiption consraints by modelling decision-making as profit maximisation. They show that result-based incentives may interact with each other, and that ignoring it may lead to calibrate incentives so that the policy budget is higher.

Bryan and Crossman (2013) find that result-based incentives create interactions among ecosystem services: an incentive targeting one ecosystem service also impacts other ecosystem services. They conclude that these interactions could raise problems when calibrating incentives: in the case of a synergy among ecosystem services, neglecting the interaction means missing an opportunity to reduce the policy budget; while in the case of a trade-off, the interaction may blurr the policy budget needed, and inflate the different incentives.

Similarly, Huber et al. (2017) also find many interaction effects among incentives targeting different issues in agriculture, and conclude that such an approach based on the Tinbergen principle (one policy instrument for each issue) may not be the best solution if interactions are ignored.

Both analyses are interesting, but they don't go far enough in assessing the consequences of the interactions among incentives. In particular, they don't estimate policy budgets.

3.4. Contributions of this thesis

Existing literature doesn't deal with all aspects of the regulation of ecosystem services in agriculture. First, whether very theoretical or more applied analyses, most studies rely on unrealistic modelling of agroecological processes and interactions among ecosystem services. This underestimates the complexity of interactions among ecosystem services. Second, many studies forget to consider participation constraints, which may underestimate the policy budget and the consequences of interactions among ecosystem services on their regulation. Third, no study accounting for interactions among ecosystem services and participation constraints really analyse how both aspects play on the policy budget, or includes action-based incentives.

This thesis addresses these three gaps. It relies on realistic agroecological modelling allowing complex interactions among multiple ecosystem services. While the economic modelling of decision making is fairly simple, costs are specified in a realistic way, and it does include participation constraint in an explicit way. Last, we compare result-based and action-based incentives and the consequences of both interactions among ecosystem services and participation constraints on them.

3.4.1. Accounting for complex interactions among ecosystem services

The existing literature doesn't provide many answers about the regulation of joint public goods or relies on simplistic assumptions, and we believe that a more realistic modelling approach could explore these questions and provide new insights in this question. For example, we aim at extending approaches like Gibbons et al. (2011) with more ES, and more complex interactions.

Given complex interactions among ecosystem services, one issue is to define which options maximise the provision of ecosystem services while minimising its cost. We rely on realistic agroecological modelling to simulate the constraints imposed by agroecological processes and the resulting interactions among ecosystem services. We use efficiency analysis to identify which options maximise the provision of multiple and interacting ecosystem services. Our approach also simulates the cost associated to the bundles of ecosystem services, and we study how to define the cost of providing ecosystem services given the complex interactions among them.

These aspects are developed in Chapter 6.

3.4.2. Modelling incentives and participation constraint explicitely

The participation constraint is not included in analyses on the provision of ecosystem services in agriculture. When comparing different action-based and result-based incentives, no studies includes more than two ecosystem services with complex interactions.

Besides accounting for the cost of providing ecosystem services, we model decision-making and with it the implementation of management options by means of action-based and result-based incentives. We assess the importance of the participation constraint and the magnitude of the policy budget to compare action-based and result-based incentives.

In particular, we explore whether interactions among ecosystem services translate into interactions among result-based incentives and increase the policy budget.

We study this in Chapter 7.

3.4.3. Accounting for heterogeneity

Last, heterogeneity in costs and environemental benefits is likely to modify the definition of options maximising ES and their implementation.

We introduce heterogeneity and wonder in particular how result-based incentives cope with it.

This is developed in Chapter 8.

Part II.

Methods

4. Agroecological and economic model

This chapter details the agroecological and economic model to explore the theoretical economic questions mentionned in Chapter 3. The model simulates the levels of ecosystem services and profit associated to a range of agricultural practices. This model relies on current literature in agroecology, hence this chapter also reviews the state of the art in agroecological modelling: for each agroecological process included in our model, we review existing approaches. We first explain the general principles that led us to make use of a model, and later review state of the existing approaches and detail equations of our model. The calibration is detailed in Appendix 3.

4.1. Presentation of the model

The aim of this model is to simulate the impacts of several agricultural practices on a set of ecosystem services and on the farmer's profit, in a stylized way, on a imaginary agricultural area of arable crops and grasslands. The area can be of any size. It is assumed to be homogeneous in its characteristics, so that it is more adapted to represent a field or a small agricultural area. In the description of the model, we call it a field.

We follow an interdisciplinary approach: the provision of ES in agriculture is our case study, the research questions, concepts and analytical tools stem from economics. This model is an attempt to better account for the specificity of agroecological processes in economics. We depart from current approaches in economics by building our analysis on the constraints represented by agroecological processes. Hence, we distinguish two levels: the agroecological processes are determinant, and the economic analysis can only adapt to this reality.

Our model may not be realistic enough for agroecologists, but it represents a progress towards better accounting for the complexity of ecosystem services in economic research.

This model simulates levels of ES provided and the associated profit on one agricultural area, over a yearly time period.

In order to be generic, this model includes only two exogenous drivers. Varying them enable to represent various agronomic conditions.

4.1.1. Validity domain

A model is a simplified and approximate representation of the reality. We need to find a balance between a very accurate but complex model and a simpler and less accurate one (Groot et al., 2012). In accordance with the second point, we also need to define its validity domain as it can't be simple, apply to a very broad context and accurate. We are interested in representing the impacts of agricultural practices on ecosystem services in a stylized way, to disentangle the complex linkages. As the model serves more as a tool to get insights into the complex agroecological processes, and its purpose is to be the base of economic analysis, we want a fairly simple model, which behaviour we can easily understand. We also don't want exogenous variables to interfere too much, to be able to interpret the results of the simulations. We therefore favour simplicity and genericity over accuracy, even if simple doesn't mean generic (Evans et al., 2013). We also restrict the agroecosystem to a system of arable crops and pastures like the ones in Northern half of France, which are the representative land use of European temperate agriculture. To calibrate the model, we rely on winter wheat. This type of agroecosystem poses major threats to the environment, therefore, we build and calibrate the model on this type of agroecosystem, to assess the impacts of modifying current intensive agricultural management and explore the economic impact of some agroecological practices.

The model is deterministic, it entails no stochastic processes. Uncertainty is thus difficult to assess with it. However, by varying the exogenous driver(s), it can represent a certain form of variability. The model has been calibrated to include or not the impacts of livestock in terms of nitrogen (methane emissions, and fertilisation).

4.1.2. Special features of the model

This model has several original features. First, it considers agricultural practices as drivers. Compared to models based on land use, this enables a more precise assessment of alternative management options and associated ecosystem services. For example, it can account for the effect of pesticides on biodiversity or pollinators. The advantage is also to consider multiple ways to influence the provision of ecosystem services. Compared to models considering only a gradient of effort or binary decisions such as conservation versus developement, it features many possible management options with various effects on the different ecosystem services. Second, it is based on the simulation of agroecological processes, and lets interactions among ES emerge from the simulated agroecological processes through the combined effect of agricultural practices. It simulates the profit in the same way, without a pre-defined profit function. This feature enables to capture complex interactions among ecosystem services and complex link between the provision of ecosystem services and the profit, and it will drive the economic analysis. The third important feature is that the model is coherent in terms of nitrogen and carbon cycles.

4.1.3. Ecosystem services and agroecological processes studied

We study an agroecosystem with arable crops and pasture. In our analysis, we include the most important agroecological processes for agriculture, which also correspond to the ecosystem services most often included in other quantitative assessments (see Literature review, Chapter 2). Our approach brings together results of the agroecological literature of the last years, as a support to an economic analysis.

Ecosystems are characterized by the interaction between all their ecological components, and this is also true of agroecosystems, where all ecosystem services are related. Agroecosystems are driven by humans to provide food (and fiber, fuel, wood...), and thus crop growth is an essential process, which depends much on nutrient availability. Nutrients needed for crop growth are supplied by soils via decomposition of soil organic matter, which is also involved in most other soil-related ecological functions. Among others, it is directly related to the capacity of soils to store carbon, and to act as a sink to reduce net emissions of greehouse gases. Soil organic matter plays a role in soil structure and then in preventing soil erosion and residues leaching or run-off into water bodies. As a consequence, soil organic matter is also related to water quality in an indirect way. Food production also relates with biodiversity: many crops (most fruits and vegetables) rely on pollination by insects. Biodiversity (insects and other wildlife and organisms) can also be crop pests, and in this case, biodiversity and trophic networks can help control them. Agricultural practices play on all theses processes, and modifying them is a big lever for managing agroecosystems. Past modifications of agricultural practices have had multiple impacts: agricultural intensification has for example contributed to soil exhaustion. Via the use of synthetic fertilizers, it has boosted primary production, but to the expense of soil organic matter and water management. Pesticides are a major element in fighting agricultural pests, but have endangered biodiversity as a whole, and pesticide residues are found in almost every water catchment in France (Commissariat Général au Développement Durable, 2013). Modifying agricultural practices via the introduction of agroecological practices is a way of achieving a better balance between agricultural production and other ecosystem services. Here agroecological practices are defined as practices achieving highly productive agroecosystems and ensuring the renewal of natural resources by using low amounts of chemical inputs.

We study ecosystem services as outputs of agriculture, and not the contribution of ecosystem services to agriculture. From the agroecological literature and the literature on ecosystem services assessments, we identified the most important ecosystem services in agriculture, the ones that constitute policy stakes:

- agricultural production, aggregating the different agricultural outputs (grain, crop residues, fodder) by means of prices.
- pollination potential, the capacity of the lanscape to offer suitable habitat and foraging resources to pollinators, so that they can deliver pollination services in crops that need it[8].
- soil fertility, the evolution of the stock of soil organic matter, to reflect the potential to provide nutrients for current and future plants.
- water quality, the presence of pollutants coming from agriculture in water bodies and causing damages to water ecosystems (pesticides, nutrients, organic particles)
- climate regulation, the contribution of agriculture to the mitigation of climate change

To be as generic as possible, we exclude services linked to the relief and hydrogeological structure of the landscape (flood control, water quantity regulation). We also exclude

[8]Given the simplicity of the model on this aspect, the indicator for pollination potential can be interpreted in a more conservative way as an indicator of the suitability of the agricultural area to provide habitat for insects and small birds.

services which need a more precise representation of crops (for example the provision of genetic biodiversity by agroecosystems, or pharmaceuticals). We consider only one provisionning service, whatever the crop use (food, fiber or energy).

Some classifications of ES consider some of them (e.g. soil fertility) not as ES, since they are intermediate agroecological processes delivering an indirect benefit to humans via an increase in the provisionning service (Haines-Young and Potschin, 2013). However, we include all these desirable agroecological processes as "agroecosystem outputs " and call them "ecosystem services" in our analysis, to acknowledge the fact that they are both desirable and under threat, and that they should be objectives of agri-environmental policies. It would be inaccurate if we would add up ecosystem services values, because of double-counting, but this is not our case. As the "intermediate" ES we include are not perfectly correlated with the provisionning service, this is also another reason to consider them separately as a desirable outcome of the agroecosystem. Similarly, we don't go into the debate over disservices: instead of considering greenhouse gas emissions from agriculture as a disservice, we consider the limitation of overall emissions as a "good output" and label it an ecosystem service. In our perspective of economists applied to agroecological issues, this issue corresponds simply to a public goods problem, and considering it this way is convenient.

4.2. Framework and notations of the model

The agroecosystem is represented as a homogeneous piece of land, which we call **field**. This field is characterised by its exogenous soil quality Q and initial stock of soil organic matter $SOM_{t=0}$. It is also the decision unit: decision variables apply on the whole field.

We adopt following indices in the mathematical equations:

- $t = [t_0 : T]$ discrete time
- $k = [1 : K]$ management option (combination of agricultural practices).

4.2.1. Control variables: Agricultural practices

We choose agricultural practices as drivers instead of land use or land cover, as land use and land cover overlook other management options for the delivery of ES (Bennett et al., 2009). For example agricultural practices such as reduced tillage or low-input agriculture (low fertilizer and pesticides) have a very important impact on ES, but cannot be assessed with only land use or land cover as drivers. Representing explicitely feedbacks between ecosystem services requires also to consider detailed drivers such as agricultural practices.

We consider several variations in the agricultural practices. Again, the choice has been made by balancing accuracy and genericity. We try to pick both the most representative practices farmers can implement in the type of agroecosystem considered (Gosme et al., 2012), and the ones that have a significant impact on the ecosystem and are often studied in ecosystem services assessments (Balbi et al., 2015), without including too many options. Therefore, each agricultural practice taken into account can correspond to a panel of actual actions. For example, the agricultural practice called "non-crop habitat" aims at representing practices supporting both biodiversity and a good water quality, like flower strips or riparian buffer strips. In reality, flower strips or grass buffers have slightly different characteristics, but in the scope of this model, they are close enough to be aggregated.

We consider agricultural practices through their combinations, which we call **management options**. We consider following agricultural practices:

- $\wedge uset = \{C; G\}$: land use (grassland / cropland). Grassland is a management option in itself and exclude any other choice listed below. Cropland can be more or less intensive, but is dedicated primarily to agricultural crop production. Grassland stands for a more extensive land use, without input of synthetic fertilizers, pesticides nor tillage. The model can account or not for the impacts of livestock in terms of nitrogen excretions, related to grasslands.

- $FTI_t = \{0; 1; 2\}$: pesticide intensity (three levels including zero pesticides). Pesticides are harmful to pests and pollinators and degrade water quality, but they are beneficial to agricultural production.

- $F_t = \{0; 1; 2; 3; 4\}$: fertilizer intensity (five levels including zero pesticides). Fertilizers bring mineral nitrogen and increase crop production, but also nitrate leaching into water bodies and greehouse gas emissions.

- $NCH_t = \{0; 1\}$: presence of non-crop habitat (yes/no). This practice represents the actions farmers can take to support biodiversity and good water quality at the margins of their fields (flower strips, buffer strips, or even hedges). They decrease the cultivated area.

- $BI_t = \{0; 1\}$: volontary biomass input (yes/no). The farmers can decide to increase fresh biomass inputs by leaving crop residues, which then increase the stock of soil organic matter.[9]

- $T_t = \{C; R\}$: 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 rate of soil organic matter, and has thus a negative short-term impact on nutrient delivery, but a positive long-term impact on soil fertility. It also reduces erosion by water.

In total, the combinations of these choices give 121 management options (grassland and 120 cropland options). An example of management option is cropland with medium fertilizer, medium pesticides, conventional tillage, without non-crop habitat nor biomass input.

The agricultural practices act as decision variables in the model and determine the level of parameters.

[9] It could also correspond to cultivating cover crops, but cover crops have an additional interest for biodiversity and water quality (they provide foraging resources and trap the nitrates), which is not included in the model.

4.2.2. Exogenous variables

The model includes two exogenous variables. Soil quality Q represents all exogenous and stable characteristics influencing potential yield of a field, and not related directly to the soil organic matter stock : soil composition (sand, loam, clay proportions), temperature, soil moisture, precipitations, slope, etc. It can not be influenced by agricultural management, and determines the potential yield, and thus the agricultural production. Soil organic matter on the contrary can be influenced by agricultural management, but its dynamics are slow. It is a state variable in the model, but its initial value SOM_0 is also an exogenous driver. It plays on agricultural production, and on ecosystem services related to nitrogen and carbon (water quality, climate regulation and soil fertility).

Figure 4.1 represents the structure of the model. Agroecological variables and represented by green ellipses, agroecological processes by green arrows, economic variables by blue arrows. The impact of agricultural practices on the agroecological processes and variables is represented through the red arrows.

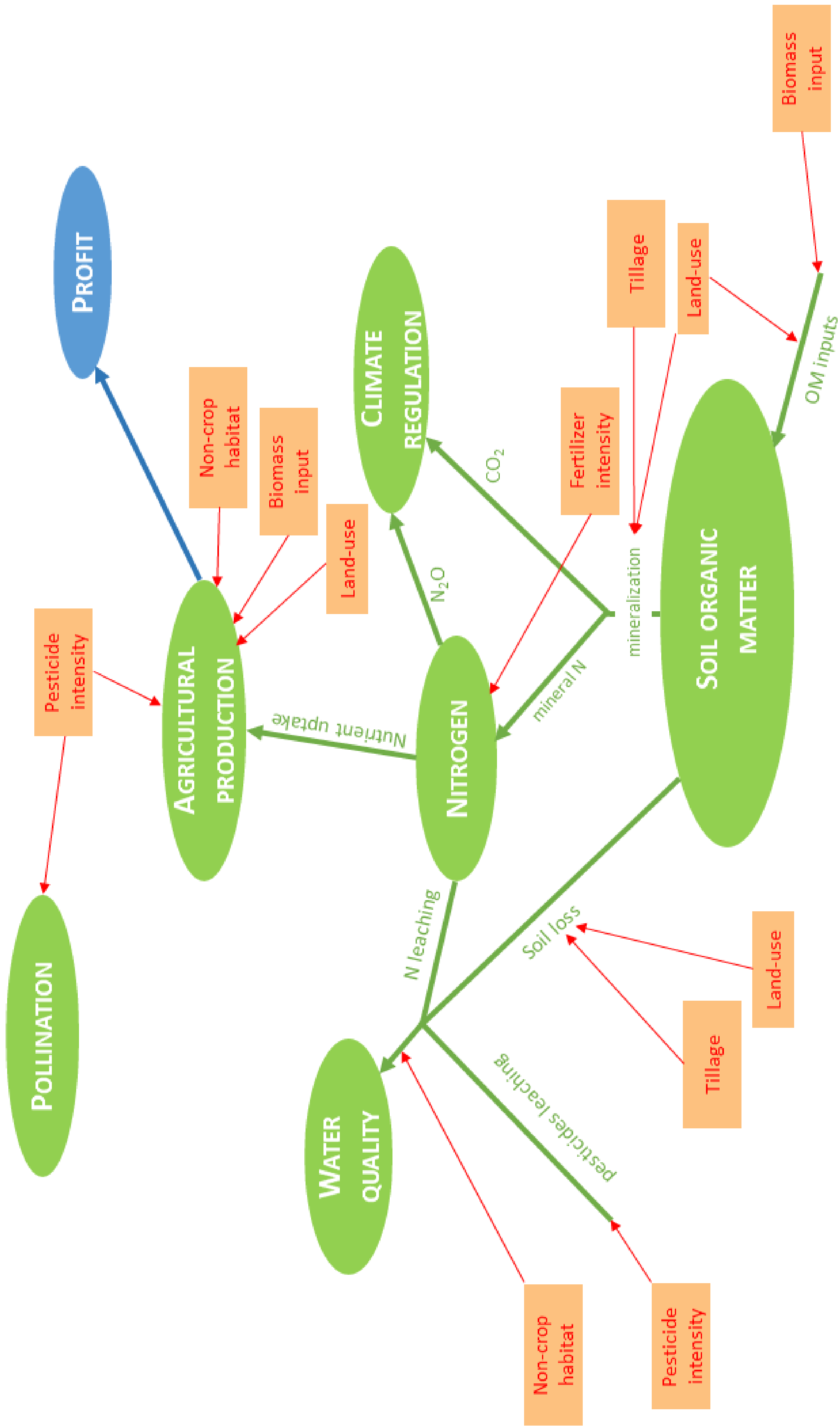

Figure 4.1.: *Schema of the agroecological and economic model*

In the following sections, for each ecosystem service included, we describe related agroecological processes, review current modelling approaches, and detail the equations of our model. Parameter values are detailed in Appendix 3.

4.3. Pollination

4.3.1. Ecological processes

Pollination is essential for the reproduction of plants, but as many crops are the fruits or seeds of plants, it is essential to crop growth itself. Pollination can be done by wind, insects, or other animals. Domesticated bees are well-known, but many wild insect species contribute to pollination, and as many plants rely on specific pollinators, they are not all substitutable. Many high-value crops (fruits and vegetables) rely on insect-pollination, so that pollinators provide a very important ecosystem service.

Pollinator populations need both foraging resources, which can be supplied by the crops, and suitable habitat for nesting and reproduction. The requirements in terms of habitat vary greatly between species, but in general, flower diversity and non-crop habitat (like flower strips, hedges) are favorable to pollinators. Pesticides, intended to kill pests, are also harmful to pollinators (Potts et al., 2010). Herbicides, because they suppress weeds which provide foraging resources also have a negative impact. Agricultural intensification, through uniformization of the landscape and pesticide use, has contributed to drive populations of pollinators down (Deguines et al., 2014).

Good practices for pollinators include non-crop habitat, and no pesticide use. Grazing has various effects according to arthropod species (Sabatier et al., 2015), and overall impact of grazing on insect abundance can be considered as globally not significant as the conclusion of Rada et al. (2014) for orthoptera. In this sense, grasslands and extenisve pastures can represent a quite suitable habitat for insect pollinators.

4.3.2. Modelling pollination in the literature

Few models of pollination exist in the literature. Pollinators comprise very diverse species, so that it would be very difficult to model each species dynamics to assess the pollination ES. Most current approaches rely on the land use and its ability to host and feed pollinators (pollination potential). This pollination potential is possibly coupled with the dependance of crops on pollination. This is the approach of Lonsdorf et al. (2009), used later in InVEST and Zulian et al. (2013). It is a species-area relationship, giving a score to each patch of a landscape in terms of nesting suitability and foraging resources. These scores are then weighted by their distance to determine the score of each field of the landscape. Similarly, Lautenbach et al. (2011) uses an estimated visitation probability, which is similar to the pollination potential.

Concerning the link between pollinator populations and pesticides, Henry et al. (2012) uses observation of homing failures due to neonicotinoids to model the impact of this class of pesticides on bee population dynamics, but there exist no generic model of the impact of pesticides on pollinator populations.

4.3.3. Pollination in our model

We treat ecosystem services as outputs of agriculture and model pollination as the ability of the field to host pollinators and provide pollination to other nereby crops that need it. Hence, we assume that the yield doesn't depend on the pollination by insects[10]. The indicator can be interpreted in a more conservative manner as an indicator for biodiversity. We model pollination with a pollinator source potential according to the methodology used by Lonsdorf et al. (2009). This approach doesn't rely on explicit modelling of pollinator populations, it gives a score for nesting suitability and foraging resources based on the land use and agricultural practices of the area. This methodology has the advantage of avoiding the explicit representation of pollinator dynamics, while including its main drivers (land use intensity, presence of non-crop habitats...) and providing a direct link between agricultural practices and ecosystem services.

Here, we capitalise on this modelling framework. We consider a "proto-pollinator", as advised in the InVEST user guide (Sharp et al., 2014). However, we depart from their model by adding the impact of pesticides on both nesting suitability and flower resources of the field, via a specific parameter capturing the increased suitability of low-input or organic crop fields. Besides, we model pollination over a unique field, and thus neglect spatial spillovers.

The final pollination source score equals

$$PS_t = FA_k \cdot NS_k \cdot PM_k \tag{4.1}$$

where FA_k and NS_k are parameters measuring the suitability of the field in terms of foraging resources and nesting, respectively. PM_k is the incresed suitability of crop fields with low pesticide inputs or none.

More precisely, FA_k and NS_k both take values between 0 to 1, and depend on land use (higher values for grasslands), non-crop habitat and pesticides. PM_k depends only on the intensity of pesticide use: it equals 1 for the maximal amount of pesticides, and is greater than one for other levels of pesticides. The pollination score PS_t ranges from 0 to 1[11]. It is an index, and has no unit. The multiplicative form implies that if either the foraging resources FA_k or the nesting suitability NS_k are equal to 0, then the index also equals zero. This reflects the fact that both conditions are necessary to host pollinators.

4.4. Soil organic matter and nitrogen

4.4.1. Ecological processes

The soil ecosystem services are tightly interrelated and still deserve extensive research efforts to bridge the knowledge gaps on ecological processes (Dominati et al., 2010). Soil is mainly composed of mineral elements (sand, clay, limestone), organic matter, and pores filled with air or water. Soil organic matter is heterogeneous, made of living microorganisms, insects and roots, fresh or decomposed organic residues and humus,

[10] Indeed, the model is calibrated on winter wheat, which doesn't depend on pollination by insects.

[11] FA_k and NS_k take low values for arable crop fields and PM_k is not high enough to make the index higher than 1.

which is the final stage of decomposition and is the most stable form of soil organic matter. Humus also represents the biggest part of soil organic matter (between 70 and 90 %). Soil organic matter plays a role in many soil functions: carbon storage, soil fertility, water quality, soil erosion (Powlson et al., 2011; Bommarco et al., 2013; Palm et al., 2014). It is an important determinant of soil structure, as organic matter particles help form stable soil agregates and favour a good water infiltration and retention, preventing soil erosion and nutrient leaching and increasing water reserve for plants. In addition, soil organic matter is made mostly of nitrogen and carbon, and it is a very important carbon pool and nutrient reserve for plants. A high content of soil organic matter also increases the cation-exchange capacity which helps fix the mineral nutrients. Last, soil organic matter provides habitat for soil biota, which in return help maintain soil functions.

Soil organic matter is a long-living stock in soil, though it is not an inert component of soil and it undergoes continuous transformations, which replenish or deplete its stock. Three main processes drive the evolution of soil organic matter: primary and secondary mineralization and humification. When fresh biomass is decomposing, first mineralisation breaks down part of it into mineral elements, releasing mineral nitrogen and carbon dioxide. This part of fresh biomass thus doesn't become organic matter. In parallel, the remaining fraction of fresh biomass is recomposed into stable humus, which is the main component of soil organic matter and a rather stable pool : this process is called humification. Humus undergoes a secondary mineralization, a slow process which degrades the organic compounds into mineral elements (mostly carbon and nitrogen) that can be taken up by plants. In short, the stock of soil organic matter is replenished by fresh biomass humified and decreases with secondary mineralization. Another cause of soil organic matter loss in a field is soil loss due to water or wind erosion (Millenium Ecosystem Assessment, 2005b).

The evolution of the stock of soil organic matter is determined by the relative importance of these flows (mineralization, humification and erosion). Multiple factors play a role in these flows: soil composition (proportion of sand, clay, lime), humidity, temperature, relief, land cover, but also agricultural practices on cultivated land. In particular, tillage increases both secondary mineralization rate by burrying fresh biomass and soil erosion by breaking soil aggregates (Attard et al., 2011; Powlson et al., 2011). Fresh organic matter left on the ground such as crop residues or cover crops increase new humus flow and contribute to replenish the stock of soil organic matter. land uses such as forests or long-term grassland have a higher soil organic matter content because of both lower mineralization rate and increased biomass inputs. More complex processes are also influenced by agricultural practices: reduced tillage increases microbial biomass which play a role in mineralization (Palm et al., 2014), fresh organic matter favours living soil organisms that help have a good soil structure (e.g. earth worms). Ground cover also helps limiting soil erosion by protecting it from direct water and wind exposure, and increases soil moisture retention as well as water infiltration (Verhulst et al., 2011).

For every agricultural practice, there exist an equilibrium stock of soil organic matter. After a change in agricultural practices, the stock of soil organic matter moves from one equilibrium to another, following what looks like an exponential-like, asymetric and slow transition path (Arrouays et al., 2002b): the evolution is slower at the begin of the transition than at the end, it is faster in the case of a soil organic matter decline than for an increase and it lasts from 20 to at least 40 years in order to reach the new equilibrium of soil organic matter.

The dynamics of soil organic matter are tightly linked with soil fertility and the nutrient cycles. The most necessary nutrients to plants are nitrogen, phosphorus and potassium. Soil organic matter particles are made of these elements, water and carbon, among others, but they are bound together under their organic form, whereas they must be under mineral (ionic) form to be taken up by plant roots. The breakdown of soil organic matter during the secondary mineralization precisely turns organic matter particles into water, carbon dioxide and ions: among others nitrates NO_3^- and ammonium NH_4^+ for nitrogen, phosphates $H_2PO_4^-$ and HPO_4^{2-} for phosphorus and ion K^+ for potassium. This way, the soil delivers nutrients to plants, which can be completed with synthetic fertilizers. Grazing livestock also deliver significant quantities of mineral nitrogen with their faeces and urine (Bristow et al., 1992; Hoogendoorn et al., 2010).

Mineral nutrients, coming either from mineralization of soil organic matter, livestock, or from synthetic fertilizers, are highly soluble into water, which makes them available for plants to be taken up, but also very easily leached to water bodies by lixiviation or run-off, causing water pollution and disrupting aquatic ecosystems (e.g. eutrophication). These unstable chemical elements undergo multiphe transformation in the soil. Nitrogen, for example, is present as nitrite ions in the soil, and according to conditions such as temperature or humidity, can become either dinitrogen (a harmless gas constituting 80% of the air we breathe) or nitrous oxide N_2O, which is a very powerful greenhouse gas. Soil organic matter cycle is therefore linked to many ecosystem services and environmental issues.

4.4.2. Modelling soil ecosystem services in the literature

Several models exist to represent the evolution of soil organic matter. The Hénin-Dupuis model (Hénin and Dupuis, 1945) considers only one pool of soil organic matter, which corresponds to humus. A fraction (k_1) of the organic residues undergo first mineralization, the rest enters the soil organic matter stock. A fixed proportion (k_2) of soil organic matter is mineralized each period. This model is still in use and has been calibrated (Wylleman et al., 2001), despite its low accuracy in some cases, especially with very high or low values of fresh biomass input (Duparque et al., 2011; Laboubée, 2007). To overcome these limitations, Andriulo et al. (1999) developed the AMG model based on the Hénin-Dupuis model, but with two compartments of soil organic matter: an active fraction and a stable fraction. Only the active fraction is subject to mineralization, the stable fraction has a very long lifetime. Other more complex models exist, among which the RothC model (Coleman and Jenkinson, 1996) which distinguishes between 5 soil organic matter compartments.

Soil erosion by wind and water is not accounted for in the models cited above, but is an important source of soil organic matter and carbon loss (Chappell et al., 2015), the more so as organic particles are leached more easily by water (Gregorich et al., 1998). Several models have been developed in Europe to assess soil loss due to erosion by water. The JRC currently uses two of them in parallel : the PESERA model and the RUSLE (Revised Universal Soil Loss Equation). The PESERA model has been developed by a working group of the JRC to map soil loss in Europe. It is a mechanisic model and has an explicit representation of rainfall, soil saturation. The first version of RUSLE, USLE has been developed by the USDA some decades ago (Wischmeier et al., 1978; Renard et al., 1997), and has been revised and recently used in the European context (Bosco et al., 2014; Panagos et al., 2015) in the same objective as PESERA. It is a simple empirical equation based on the characteristics of the soil, the slope, the crop grown and management, and other factors of erosion; it doesn't represent explicitely the physical processes underlying erosion by water. By comparing the estimations given by the two approaches, the JRC found few differences, and almost none in flat areas. We consider arable crops, which are mainly grown in flat areas, so that these differences are not relevant for our model.

4.4.3. Soil related processes in our model

We choose the simplest representation of soil organic matter (SOM) dynamics. We take the same functional form as Lifran et al. (2014), built using the Hénin-Dupuis model for soil organic carbon (Hénin and Dupuis, 1945) and consider one pool of organic matter and one pool of mineral nitrogen (whatever its real chemical form). Soil organic matter increases with the fresh organic matter being changed into humus, and is depleted by mineralization and soil erosion. We account for soil erosion via an erosion rate which depends on the agricultural practices, using the calibration of the RUSLE equation for Europe.

We make the assumption that there are no spatial interactions in the SOM dynamics, and especially that SOM leached by water is lost to water bodies off the fields, and doesn't deposit in other fields.

The dynamic of soil organic matter for field i with agricultural practices k is given by:

$$SOM_t = SOM_{t-1} - (m_k + \lambda_k)SOM_{t-1} + I_k \tag{4.2}$$

where SOM_{t+1} and SOM_t stand for the stock of soil organic matter in periods $t+1$ and t, m_k for the mineralisation rate, λ_k for the erosion rate and I_k for the fresh organic matter from biomass decomposition.

In detail, the mineralisation rate m_k describes the secondary mineralisation, which degrades stable humus into mineral nitrogen available for plants. This rate depends on the agricultural practices (tillage, land use). The erosion rate λ_k captures water erosion, which carries part of the soil away and thus decreases the stock of SOM[12]. This parameter depends on the tillage regime and land use: reduced tillage and grassland have a lower erosion rate. Biomass input $I_{i,t}$ brings new organic matter. It represent the biomass contained in non-harvested parts of plants and crop residues (straw), and it varies according to crop residue restitution. Fresh biomass first undergo a primary mineralisation, which means that 1 ton of fresh biomass gives less than 1 ton of humus (stable organic matter). However, it is expressed here directly as a humus-equivalent.

[12]We assume that soil organic matter is homogeneously distributed, and thus that the proportion of soil organic matter lost is equal to the proportion of soil eroded.

In our model, we use the evolution of SOM, ΔSOM_t as the indicator for the ecosystem service "soil fertility".

$$\Delta SOM_t = SOM_t - SOM_{t-1} \tag{4.3}$$

Nitrogen is the only mineral nutrient in the model, as it the one on which the most studies exist. Moreover, even if differences with phosphorus and potassium exist, given our very simplified framework, they would be represented in similar ways.

The second mineralization process transforms a part of soil organic matter into mineral nitrogen (and other mineral elements), which can be completed by mineral fertilizers. Other inputs of mineral nitrogen are livestock urine and faeces. Mineral nitrogen coming from these three sources is assumed to be substitutable as it is composed of the same chemical components (mostly nitrate NO_3^-, and ammonium NH_4^+). To add them up, the mass of soil organic matter mineralized has to be converted into mass of mineral nitrogen with coefficients c_2 and c_3. [13] Fertilizer quantity f_k is directly expressed as a mass of mineral nitrogen and depends on fertiliser intensity F_t. Mineral nitrogen from livestock LN_k is also directly expressed in N-equivalent. Total mineral nitrogen N_t equals

$$N_t = \frac{c_3}{c_2} \cdot m_k SOM_t + f_k + LN_k \tag{4.4}$$

The mineral nitrogen is distributed into 3 different destinations: plant, air (emissions of nitrous oxide N_2O via denitrification) and water (via leaching to groundwater and runoff to surface water). Mineral nitrogen is composed of ions, which are easily soluble in water, so it has a rather short life in the soil. We assume that all the mineral nitrogen available in one period (one year) is distributed into these 3 pools.

Some nitrogen N_{At} is emitted into the air via denitrification, transforming mineral nitrogen into N_2O, a powerful greenhouse gas which contributes to climate change. It is expressed as the mass of the nitrogen element in nitrous oxide and is computed as a fraction β of the total amount of mineral nitrogen, according tho IPCC methodology (IPCC, 2006).

$$N_{At} = \beta N_t \tag{4.5}$$

Nitrogen uptake is modelled with a linear plus hyperbolic functional form, taken from Makowski et al. (1999) [14]. The plants take up part of the nitrogen available for plants (nitrogen non emitted into the air, $N_t - N_{At}$). Up to a certain amount of external nitrogen input from synthetic fertiliser or livestock excretions N^*, nitrogen uptake by plants is proportional to the nitrogen available $(N_t - N_{At})$. Above this threshold, the nitrogen uptake slows gradually down as nitrogen available for plants increases. Nutrient uptake then influences positively the yield (see section 4.6).

$$\begin{aligned} N_{Pt} &= \gamma(N_t - N_{At}) \quad \text{for } N_t - N_{At} < N^* \\ N_{Pt} &= \gamma N^* + \frac{\gamma(N_t - N_{At} - N^*)}{1 + \epsilon(N_t - N_{At} - N^*)} \quad \text{for } N_t - N_{At} \geq N^* \end{aligned} \tag{4.6}$$

[13] For this, we use the methodology proposed by Comifer (2013) to help farmers determine their own stock of soil organic matter: we first compute the mass of carbon contained in the soil organic matter using the ratio of carbon content in organic matter c_3 (see section 4.5). Then, to get the mass of nitrogen, we use an average ratio of carbon to nitrogen in soil organic matter c_2.

[14] The original functional form features an additional fixed amount of nutrient taken up by plants, but it has been re-calibrated to 0, because we model nitrogen uptake in relation to total mineral nitrogen, not only fertiliser (see the calibration of the model in Appendix 3)

with γ the nutrient uptake coefficient, and ϵ a parameter describing the curvature of the nitrogen uptake function above the threshold N^*.

Eventually, the remaining nitrogen is leached into water bodies (N_{Wt}). This nitrogen is transported either by run-off to surface water bodies or leaching to groundwater, and contributes to impair water quality.

$$N_{Wt} = N_t - N_{At} - N_{Pt} \tag{4.7}$$

4.5. Greenhouse gases

4.5.1. Ecological processes

Agroecological processes involve many exchanges of greenhouse gases. The biggest flows are those of carbon dioxide, which is absorbed by plants during their growth (photosynthesis) and emitted by animals (respiration). Carbon contained in plants is released during their decomposition into mineral elements. Given that much of the agricultural biomass has a short lifetime (most of it will be digested or decomposed within one year), increasing carbon stored in agricultural biomass has only a very short effect. Large stocks of carbon are contained in soils as components of soil organic matter. SOM has a much slower turnover rate, this sink is much more significant: increasing carbon stored in soils by 4 ‰ each year could compensate the current net GHG emissions (Le Quéré, Corinne and Moriarty, Roisin and Andrew, Robbie M and Peters, Glen P and Ciais, Philippe and Friedlingstein, Pierre and Jones, S D and Sitch, Stephen and Tans, P and Arneth, Almut and Others, 2016). SOM management therefore is important not only to preserve soil fertility, but also to increase carbon sinks. Soil conservation (avoiding soil erosion) is a mean to reduce net greenhouse gas emissions, as soil loss is a significant source of greenhouse gases: soil organic particles leached undergo mineralisation and decompose in carbon dioxide and mineral elements (Millenium Ecosystem Assessment, 2005b). Other carbon dioxide emissions come from fossil fuel burning by farm machinery. Reduced tillage, by reducing use of machinery, represents one of the main mitigation potentials of greenhouse gas emissions in agriculture (Robertson, 2000).

When confronted to certain conditions, mineral nitrogen (nitrite ions) in the soil decompose as nitrous oxide N_2O, which is a powerful greenhouse gas, and responsible for an important part of agriculture contribution to climate change. Input of synthetic fertilizers greatly contributes to these nitrous oxide emissions, as it provides large quantities of mineral nitrogen.

Last, methane emitted by enteric fermentation in the stomach of livestock also contributes to greenhouse gas emissions. Grasslands are mainly used to feed (even indirectly) livestock, and thus it is important to include methane emissions into their greenhouse gas balance.

Greenhouse gases coming from these four sources all contribute to the greenhouse effect, and can be aggregated into a single metric such as carbon dioxide equivalent (CO_2eq).

4.5.2. Modelling

To model the flows of greenhouse gases, we follow the methodology of IPCC and InVEST. However, IPCC guidelines consider 4 carbon pools: above-ground biomass, below-ground biomass, soil carbon and wood products. We only consider the soil carbon pool, as agricultural biomass quantities are rather stable over years, and wood products are beyond the scope of this thesis. Even if some wood is harvested as a by-product of hedges, it is probably not used for a long-lasting purpose and therefore the carbon stored during the growth is quickly reemitted in the atmosphere.

In the model, we consider 4 sources of greenhouse gases: emission of nitrous oxide N_{At}, carbon dioxide from changes in soil organic carbon stock ΔSOC_t and fossil fuel burning FC_k and methane emitted by livestock $methane_t$. The sum of greenhouse gases, expressed in tons of CO_2eq per ha equal

$$GHG_t = g_1 c_4 N_{At} + FC_k + \Delta SOC_t + g_2 methane_t \tag{4.8}$$

To be accounted in the equation, nitrous oxide N_{At} emitted from mineral nitrogen needs to be first converted from mass of elemental nitrogen into mass of molecular nitrous oxide with coefficient c_4. Then it is converted to CO_2eq with its global warming power g_1. Methane emissions occur on grasslands through livestock, and also need to be converted into CO_2eq with conversion parameter g_2.

Soil organic carbon (SOC) refers to the carbon component of soil organic matter. It is approximated as a fixed fraction c_3 of the soil organic matter. (Nelson and Sommers, 1982) [15].

$$SOC_t = c_3 SOM_t \tag{4.9}$$

Therefore, soil organic carbon follows the same dynamics as soil organic matter: the same mineralization process governs the transformation of SOM into nitrogen and carbon.

$$\Delta SOC_t = SOC_t - SOC_{t-1} = c_3(SOM_t - SOM_{t-1}) \tag{4.10}$$

The emissions due to fossil fuel burning FC_k are related to agricultural practices (tillage regime and land use). Here we don't follow the IPCC methodology (Maurice et al., 2006) which advises to calculate the travelling distance and translate it into emissions with emission factors per kilometer. It would be difficult to estimate the travelling distance of farm machinery, and no references have been found for it in the literature. Rather, we rely on life-cycle analyses which give figures based on fuel consumption (see Appendix 3 for the calibration).

4.6. Agricultural production

4.6.1. Ecological processes

Biomass production depends mainly on resources available locally (water, nutrients) and on sun light. Plants also need physical support for their roots to grow properly. Soils provide physical support, as well as nutrients and water. In order to produce food and

[15] Indeed the carbon-nitrogen ratio in organic matter can vary Hassink et al. (1993), and with it the mineralization rate as well as the quantities of mineralized elements

raw materials, farmers clear land and increase cultivated area, select productive species and varieties, and add nutrients (e.g. organic manure or chemical fertilizers) or water (irrigation). They also till the soil to loosen it and facilitate the implantation of crops and increase the capacity of soils to provide nutrients. Evenually, they also fight crop pests with pesticides and suppress weed with herbicides.

4.6.2. Modelling agricultural production in the literature

Agricultural production depends in complex ways on the agroecological processes providing the ecosystem services described above (Bommarco et al., 2013). However, it has long been measured, and much data exist.

In the literature on ecosystem service assessments, two strategies are mainly used to model agricultural production. The first strategy is to use the extensive data with statistical approaches linking agricultural production to land use or other drivers such as intensity, with no mechanistic modelling of agricultural production (Lautenbach et al., 2011; Raudsepp-Hearne et al., 2010; Bateman et al., 2013; Ruijs et al., 2013; Dross et al., 2018). The second strategy is to use complex models predicting crop growth with many drivers (Kirchner et al., 2015). An original approach is the one of Balbi et al. (2015) which use bayesian network to model wheat yields, and consider 3 drivers of agricultural production: soil nitrogen availability, water availability and temperature.

4.6.3. Agricultural production in our model

We use a mechanistic model of agricultural production, where the main driver is mineral nitrogen availability.

Grassland patches produce a fixed amount of fodder and animal products Y_G.
In cultivated fields, attainable grain yield Y_{1t}, measured in tons/ha is modelled with a modified Mitcherlisch-Baule function, and depends on nitrogen taken up by the plant:

$$Y_{1t} = Q(1 - \exp^{-n_2 N_t}) \tag{4.11}$$

where Q is the exogenous soil quality (representing geophysical caracteristics of soil like depth, humidity, clay, sand and silt proportions...) and n_2 the marginal effect of nitrogen on yield.

Non-crop habitat NCH_t reduces the cultivated area and thus the yield:

$$Y_{2t} = Y_{1t}(1 - e \cdot NCH_t) \tag{4.12}$$

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

The grain yield is further reduced after crop damage due to pests D_k. Thus the grain yield eventually harvestable (in tons per ha) equals

$$Y_{3t} = Y_{2t}(1 - D_k) \tag{4.13}$$

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: in the absence of pesticide, pests are supposed to cause yield loss of a fraction h_1 of yield, and pesticide applications, according to their level, reduce this yield loss by a fraction h_{2k}.

$$D_k = h_1(1 - h_{2k}) \tag{4.14}$$

Eventually, the production of crop residues (in tons per ha) is proportioned to grain yield with factor ρ and happens only if the farmer decides to export them (i.e. $BI_t =$ 0):

$$Y_{Rt} = \rho Y_{3t}(1 - BI_t) \tag{4.15}$$

All crop products (crop, crop residue, forage) are aggregated by means of agricultural prices, and total production is used as the indicator for agricultural production.

4.7. Water quality

4.7.1. Ecological processes

Agricultural activity has an impact on water quality. Three main pollutants harm ecosystems: nutrients, pesticide residues and sediments (Ongley, 1996).

Nutrients modify the trophic equilibrium in aquatic ecosystems, favoring the growth of certain species over others, and causing for example algal bloom, which then modifies living conditions for all other species (eutrophication). A too high level of nitrates in drinking water is also not recommended for human consumption. Mineral nutrients (coming from mineralization of organic matter or from fertilizers) are present in the soil or at the surface, not all of them are used up by crops, so that water run-off or infiltration carries them to water bodies, after a more or less long journey above or below-ground. Many factors influence the transport of nutrients by water, among others the chemical properties of soils, linked to the soil organic matter. Mitigation actions aim on one hand at reducing nutrient residues (reducing their use) and on the other, at reducing their transport to water bodies (favoring a good soil structure to limit run-off water, planting riparian grass strips or hedges along rivers to catch the run-off and let nutrients deposit there instead of flowing to the rivers).

Pesticides sprayed to fight crop pests are degraded more or less rapidly by the action of sun and microbial biomass, but some residues may remain even months after the application in the soil or on the plants, and then are carried away by water in the same way as for nutrients. Pesticides may poison other living organisms including the aquatic ones, and concerns grow about their effects on human health too. Reducing pesticide residues rely mainly on reducing pesticide applications, and banning the most persistent ones (e.g. organochloride pesticides). Reducing run-off and planting riparian strips and physical protections along waterways reduce water pollution, by retaining pesticides and helping to degrade them (Dosskey, 2001).

Sediments lost due to soil erosion have several impacts on water quality: they cause excessive levels of water turbidity, affecting the ability of water organisms to photosynthese, and sediments depositing on rivers and lake beds lead to sedimentation (disruption of the flow, reduced depth of rivers...). Besides, eroded soil particles carry a high amount of nutrients, pesticides and other chemical compounds which alter aquatic ecosystems.

Again, riparian buffer strips help mitigating water pollution, as well as other actions to reduce soil loss (reduced tillage, permanent soil cover...) and facilitate water infiltration.

4.7.2. Water quality modelling in the literature

In the literature, many models predicting water quality in relation to land use and agricultural management are complex agricultural system models, including complete hydrogeological modules. One of the most used is the SWAT model (Arnold et al., 1998; Srinivasan et al., 1998), which includes many components simulating pollutant loads (nutrients and pesticide residues, sediments...), runoff, erosion, fate and transport into water bodies. This model is coupled with GIS and requires precise data about the crops and crop management, water flows, precipitations, climate, etc. Other complex models predicting water quality in response to human activities exist, for example RZWQM (Ma et al., 2001), CREAMS (Knisel, 1980).They model more or less explicitely the hydrogeological and chemical processes and require detailed input data.

Another approach is to predict pollutant loads with export coefficients which express the rate at which pollutants are exported from each landscape cell, according to its land use type. These models are mostly used for nutrient pollution. By adding all exported pollutants from each source along the water body, it is possible to predict the total pollutant load. Johnes (1996) use it in the UK to predict nitrogen and phosphorus loads in surface water, with different export coefficient corresponding to different land use types and find that predicted data is quite accurate compared to real pollutant loads. The same approach is used in the InVEST model (Sharp et al., 2014) to model nutrient concentration, coupled with GIS to associate automatically data about run-off, relief, land use etc. Export coefficients are specific to each type of land use, and other factors capture the nutrient removing efficiency of natural buffers (vegetation, wetlands...). Export coefficient approaches are only able to be applied to surface waters, not to groundwater. They avoid the effort of building and calibrating complex hydrogeological models. The approach taken by the InVEST model still requires much input data about land use, land-cover, elevation, soil depth, evapotranspiration... More specifically, for sediments and soil erosion, InVEST and Lautenbach et al. (2011) use the RUSLE equation Renard et al. (1997), with input data about the relief and buffer strips. In fact none the above cited models seems to have been used on a "virtual" landscape as we want to do, they were all applied to a real landscape, either to describe it (Balbi et al., 2015) or to provide insight on the consequences of altenative scenarios (Nelson et al., 2009; Santhi et al., 2006).

Confronted to the high input data requirement of the models above, Jones et al. (2001) tried to predict nutrient and sediment loadings from landscape metrics, and show that some well-known drivers (agriculture, riparian forests etc.) explain a high percentage of the observed nutrient and sediment loads. However, their analysis is restricted at metrics which can be assessed from land use data, and doesn't discriminate between between crops and other agricultural practices within the a given land use (like for example intensity of pesticide or nutrient use).

4.7.3. Water quality in our model

We choose to assess water quality as the contribution of one field to the water quality of neighbouring water bodies. Low fertilizer and pesticide applications, reduced tillage and non-crop habitat contribute to allow export of pollutants.

We use a index for water quality, which includes 3 important pollutants in agroecosystems: pesticide residues, nitrates, and soil organic matter particles. We derive a synthetic water quality index comprising these three dimensions. There is always a problem when aggregating several criteria into a single indicator. Rather than using an arithmetic mean which would imply that the different dimension are substitutable, we aggregate the three pollutants with a limiting factor approach to capture the fact that the worst indicator is critical. The indicator for water quality equals the worst score. Non-crop habitat trap pollutants and mitigate water pollution.

$$W_t = \min\{PL_t; NL_t; ML_t\}(1 - w\sum_i NCH_t) \tag{4.16}$$

where w is a parameter capturing the reduction of pollutants export due to non-crop habitat, and PL_t, NL_t and ML_t functions expressing pollutants loads (pesticide residues, nitrates, and organic particles) as a fraction of the maximal pollutant loads over the range of agronomic contexts considered:

$$\begin{aligned} PL_t &= \frac{FTI_t}{PL_{max}} \\ NL_t &= \frac{N_{Wt}}{NL_{max}} \\ ML_t &= \frac{ML_t}{ML_{max}} \end{aligned} \tag{4.17}$$

where $ML_t = \lambda SOM_t$ is the amount of soil organic matter leached on field i, and PL_{max}, NL_{max} and ML_{max} are the maximal loads (over all management options and agronomic contexts) of the three pollutants: pesticide residues, nitrates and organic particles.

4.8. Profit

Modelling the decisions taken by the farmer in terms of agricultural management is a way to determine endogeneously land use and management in dynamic agroecological model, or to assess the costs and constraints associated to a change in agricultural practices.

Decision-making is often based on the farmer's profit. It is often modelled in a simple way and depends much on agricultural production, even if it can encompass subsidies, taxes or costs that are not directly linked to agricultural production (Fontana et al., 2013). In simulations over long time spans, profit is often agregated into net present value: future profits are discounted (Nelson et al., 2009).

More complex representations include other drivers than profit, such as working time (Groot et al., 2012). Other more complex models consider the impacts on agricultural supply and prices, and include feedbacks at a sectoral or macroeconomic model through partial or general equilibrium models (Schönhart et al., 2011; Kirchner et al., 2015; Cramer et al., 2005).

4.8.1. Profit in our model

In our model, decision-making is based on profit, modelled as gross margin

$$\pi_t = s_G 1_{U=G} + pY_{3t} - MC_k \tag{4.18}$$

with s_G the fixed income in grassland, $1_{U=G}$ an indicator function taking value 1 if land use is grassland, p the crop prices, and MC_k the management costs associated to management option k.

$$MC_k = MC_k^B + MC_k^F + MC_k^{Pesti} + MC_k^M + MC_k^{NCH} \tag{4.19}$$

5. Simulated data

The agroecological model is used to generate a data set, which serves as the analyses of the next chapters[16]. This short chapter summarizes the simulated data set obtained.

5.1. Summary of simulated data

The model is run for one period on one homogeneous agricultural area. The output is a simulated data set, giving for each of the 121 management options (combinations of agricultural practices), a bundle of 5 ecosystem services and the associated profit.

The model has two exogenous drivers, soil quality and the initial stock of soil organic matter. The combination of these two drivers is called an **agronomic context**. We run the model for 10 different agronomic contexts (see below), and thus get simulated data sets for each of these agronomic contexts.

The model is run in two settings: including and excluding the indirect impacts of livestock excretions (methane emissions, mineral nitrogen).

Model calibration is detailed in Appendix 3. Simulations are run with Matlab. An example of a simulated data set is given in Appendix 1 .

Outputs of the model In each agronomic context, the simulated data set gives for each management option the levels of the following indicators:

- agricultural production, in euro, agregating the several marketed outputs (grain, crop residues, forage).
- pollination, as an index between 0 and 1
- water quality, as an index between 0 and 1
- contribution to the mitigation of climate change, measured as net emissions of greenhouse gases, in tons of CO_2eq.
- the evolution of soil fertility, measured as the evolution in the stock of SOM over 1 year, in tons
- the profit, measured in euro

[16]The same methodology could be done using field data, if such a detailed and comprehensive data set were available.

5.2. Exogenous drivers

Soil quality is defined in the model as potential yield, in $t.ha^{-1}$, and calibrated to represent the observed range of potential yield for winter wheat in France. The stock of soil organic matter is a state variable which is influenced by past agricultural practices, but its initial value is given.

To represent a range of agronomic conditions, we run the simulations on 10 different agronomic contexts detailed in Table 5.1.

Although the dynamics of soil organic matter don't depend directly on soil quality, the evolution of soil organic matter depends on past management decisions and hence indirectly on soil quality. Farmers choose their agricultural practices so as to maximise the profit, and this profit depends on the soil quality. Hence the most profitable management option depends on potential yield (used as a proxy for soil quality), and this is even more important given that the SOM dynamics are slow and thus depend much on past management. Therefore, to explore representative agronomic contexts and represent the fact that SOM is inherited from past management decisions, we choose the initial value of the stock of SOM depending on soil quality. More precisely, the initial stock of SOM is supposed to reflect the equilibrium stock in a situation where the prices and costs are stable over a long period, and farmers maximize the present value of their future 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 SOM and its long-term value. This is a dynamic programming problem, where the stock of soil organic matter is the state variable, and the management option the control. We solve this problem using a Bellman algorithm for a range of soil quality. The algorithm gives for each soil quality the stock of soil organic matter maximising the intertemporal profit, which we take as a representative intial value for the stock of SOM.

The algorithm works in two steps. For every soil quality, the first step determines the most profitable management option (taking into account soil organic matter dynamics) given the stock of soil organic matter. This step relies on a dynamic equation: the choice of a management option (control) modifies the stock of soil organic matter (state), which in the next period modifies the value associated to each option (control). The highest payoff possible from time t until a given time horizon T, given the state at time t equals

$$V(t, SOM(t)) = \max_{z \in Z} \Big[\pi(z, SOM(t)) + \delta V(t+1, SOM(t+1, z)) \Big] \tag{5.1}$$

with V the value function (highest intertemporal payoff), z the control, i.e. the management option, to be chosen among Z options, SOM the state (SOM stock), π the instant payoff, δ the discount factor. The value function in time t, $V(t, SOM(t))$, is the highest intertemporal payoff from time t until the last period of the simulation. This equation relates the value function in time t to the value function in time $t+1$ and transforms the dynamic problem into a static one. Given the value function in time $t+1$, it is possible to determine the control maximising the value function at time t.

This equation is solved by backwards induction for each possible stock of SOM at time T, $SOM(T)$. $V(T)$ is set to 0. First, the algorithm solves the equation 5.1 for $t = T - 1$. It calculates the optimal control $z(T-1)$ that leads to $SOM(T)$ while maximising $V(T-1)$, the payoff from $T-1$ to T, and determines the corresponding stock of soil organic matter in $T-1$ (so that $SOM(T, z(T-1)) = SOM(T)$). Then the algorithm solves the equation for $t = T-2$. It calculates the optimal control $z(T-2)$ and $V(T-2)$ (the highest intermporal payoff from $T-2$ until T), given $V(T-1)$ and $SOM(T-1)$ determined by the first iteration. It again determines $SOM(T-2)$ associated. The algorithm iterates backwards until it determines $V(T)$, the highest intertemporal payoff from period 1 to period T, the optimal control in the first period $z(1)$, and the corresponding stock of soil organic matter in the first period $SOM(1)$. All this process determines, for each stock of soil organic matter, the control maximising the intertemporal payoff: for each $SOM(1)$, $z(1)$ is the management option that maximises the intertemporal payoff over the next $T-1$ periods.

The second step makes use of the optimal controls determined in the first step to calculate optimal trajectories of the stock of soil organic matter, i.e. trajectories simulating the decisions taken by a farmer maximising its intertemporal payoff. The trajectories are determined for a given time horizon, that can be different from the one of the first step. For each possible initial value of the stock of soil organic matter, the second step retrieves the optimal control determined in the first step, calculates the new stock of soil organic matter resulting from the control. It retrieves again the optimal control associated to the new stock of soil organic matter, and so on until the last period of the simulation span. This is done for every soil quality. Finally, for each soil quality and each initial value of the stock of soil organic matter, this second step gives the optimal path of controls and the evolution of the stock of soil organic matter resulting from it. It presents all possible trajectories of the stock of soil organic matter resulting from intertemporal profit maximisation, for a given soil quality. It is then possible to see if these trajectories converge towards a common value: i.e. whether the intertemporal profit maximisation leads to a stable value of SOM in the last periods, and whether these values are equal whatever the initial values of SOM. If the trajectories converge towards a common value of SOM, this value represents the stock of SOM resulting from profit-maximisation, given a soil quality

We plot all trajectories of SOM for one soil quality, and look if they converge towards a stable value, which represents the stock of SOM which is likely to be reached in the given economic context (prices and costs).

We observe that for each soil quality, all trajectories of SOM converge towards a unique equilibrium, or at least oscillate within a restricted range[17], for a relatively long time horizon (200 periods). The predicted values for SOM reproduce a range of observ-

[17] In contexts with high potential yield, another option has a profit very close to the most profitable option (statu quo). The two options differ according to the tillage regime: the statu quo is the one with conventional tillage. Given that their profits are very close, a slight variation of the stock of SOM changes which option is most profitable. The chosen value for the stock of SOM is indeed an average value over the minimum and maximum values of stable and convergent trajectories at the simulation horizon, so that in itself, it doesn't correspond to a stable trajectory. Moreover, the simulations are sensitive to the step used to translate values into classes, so that slight variations are not always well treated by the algorithm. After choosing the agronomic contexts and running the simulations of ES on them, we see that the most profitable option is associated to a loss in SOM, which is incoherent with the assuption that this option should realise the long-term equilibrium of the stock of SOM.

able situations in agricultural landscapes in Northern Europe, and are close to estimates of SOM content mentioned in the grey literature (Ancelin et al., 2008; Antoni et al., 2011). For low soil qualities (potential yield lower than 5t/ha), grassland is the most profitable option and thus the stock of SOM converges towards its maximal value (112t/ha, 2.8%), whatever the initial stock of SOM. Soils with high potential yield (higher than 5.5t/ha) have much lower equilibrium stocks of SOM (around 75t/ha, or 1.7%). In the case of intermediate levels of soil quality (between 5t/ha and 5/5t/ha), all trajectories of SOM oscillate within a restricted range, which average value decreases with increasing soil quality.

For the following analyses, we keep 10 contexts of soil quality, and the associated stocks of SOM. The soil quality (Q) of the contexts were chosen so as to represent a range of possible situations. This variability is used to simulate heterogeneous agricultural areas in Chapter 8.

context	**Q (t/ha)**	**SOM (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

Table 5.1.: *Characteristics of the 10 agronomic contexts used in the analysis*

Although SOM decreases with soil quality, the effect of increasing soil quality on actual yield dominates the effect of diminishing SOM stock. Thus for each option, actual yield increases with potential yield, although the nitrogen supplied by the soil decreases. The only exception are options without fertiliser input, between context 3 and 4: the transition between these contexts is characterised by a low increase in potential yield and a large decrease in soil fertility, so that the effect of soil fertility dominates, in the absence of synthetic fertilisers.

Part III.

Economic incentives to maximise the provision of ES under a budget constraint

6. Maximising ES provision

The design of policies for the provision of ecosystem services comprises two steps

1. the definition of the optimal solution in terms of effort and actions to take
2. the implementation of this solution through policy instruments

The chapters follow this division: Chapter 6 deals with the definition of efficient and cost-efficient solutions to provide ecosystem services, Chapter 7 deals with the implementation of these solutions by means of economic incentives.

6.1. Introduction

Ecosystems are defined through the interactions among living organisms and with their environment, and thus ES exhibit multiple and complex linkages such as antagonisms and synergies (Bennett et al., 2009). The design of agri-environmental policies should take these interactions into account to target management options that bring maximal levels of ES.

This chapter explores how to determine the efficient and cost-efficient ways to provide ecosystem services in presence of many interacting ecosystem services. Compared to existing studies exploring the efficient provision of multiple ES (Chan et al., 2006; Naidoo et al., 2008; Ruijs et al., 2013), we follow a reverse approach. These studies first calculate (cor)relations among pairs of ES and in a second time use them to determine how to provide ES jointly. On the contrary, we first capture interactions among ecosystem services by considering **bundles** of ecosystem services and determing the efficient ones (i.e. the ones maximising ES jointly) using tools stemming from production economics. In a second time, we show that the analysis of efficient bundles in an easy and generic to explore synergies and trade-offs among ecosystem services. The interest of our approach is that determining efficient bundles is the genuine aim of many studies on multiple ES, and can directly inform public policies, contrary to correlations among pairs of ES. Our approach avoids the risk of identifying uncompatible ways to provide the ES separately (Ruijs et al., 2013).

In a second part of the chapter, we introduce the cost of providing non-marketed ecosystem services and show that given the interactions among ES, this cost can only be determined for a bundle of ecosystem services. We also argue that this cost must be calculated as an **opportunity cost**, i.e. a difference in profit with the most profitable management option. With this cost, we move from efficiency analyses to cost-efficiency analyses. Providing ecosystem services comes at a cost. It can be supported either by the farmer or the society if compensated by subsidies, but both face budget constraints, so that maximising the provision of ecosystem services requires to consider this costs (Naidoo and Ricketts, 2006). Last, we also compare the cost of two types of strategies to provide a given bundle of ecosystem services: (i) a moderate change in agricultural practices adopted on the whole agricultural area and (ii) a radical change in agricultural practices adopted on a part of the agricultural area.

The main contributions of the chapter are

- to advocate for the use of efficiency analysis tools, such as Data Envelopment Analysis, to study the interactions among several ecosystem services and to determine management options maximising multiple interacting ES (Sections 6.2 and 6.3)

- to provide a consistent definition of the cost of providing non-marketed ecosystem services, accounting for the interactions among ecosystem services; and to use it to determine cost-efficient management options (Section 6.4)

- to provide a method to compare the cost of two strategies to increase non-marketed ES: *i)* a strategy targeting a moderate increase in ES adopted over the whole agricultural area, or *ii)* a strategy of a more important increase of ES on a small part of the agricultural area (Section 6.5)

For this purpose, we use simulated data described in Chapter 5, i.e. bundles of ES and profit corresponding to 121 management options, over a range of 10 agronomic contexts.

6.2. Production possibility frontiers and interactions among multiple ecosystem services

Different methods are used in the literature to study the interactions among ES: correlations, maps, flower diagrams, etc. We showed in Chapter 2 that production possibility frontiers are an interesting tool to study these interactions. Their use in microeconomic theory makes them useful for interdisciplinary analyses.

Production possibility frontiers represent interactions among ecosystem services, their shape inform about the type of interaction and implication for the provision of ES. Besides, the bundles of ES belonging to the PPF are interesting in themselves since they represent the management options that maximise the provision of ES.

6.2.1. Different shapes of PPF and their implications

Production possibility frontiers inform about the type of interactions among ES. Three shapes of PPF can be distinguished, which correspond to different interactions among ES and have different implications: synergy, concave trade-off, or convex trade-off. These three cases are illustrated in Fig. 6.1–6.3.

While correlation coefficients or the observation of the shape of the production possibility set consider all the bundles of ES, the PPF focuses on efficient bundles and ignores the others. Thus they don't assess exactly the same interactions. The PPF represents unavoidable trade-offs that cannot be solved by reducing inefficiency (Lester et al., 2013)[18].

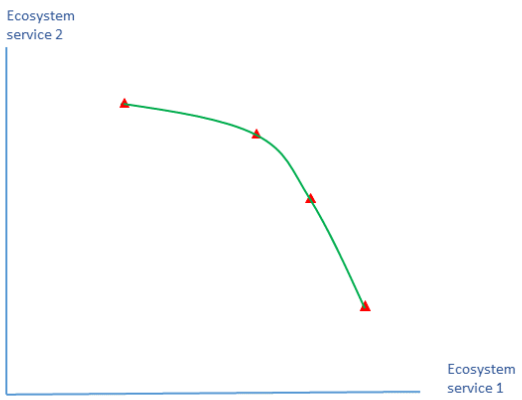

Figure 6.1.: *PPF exhibiting a **concave** trade-off between ES, where intermediate management options dominate combinations of extreme options*

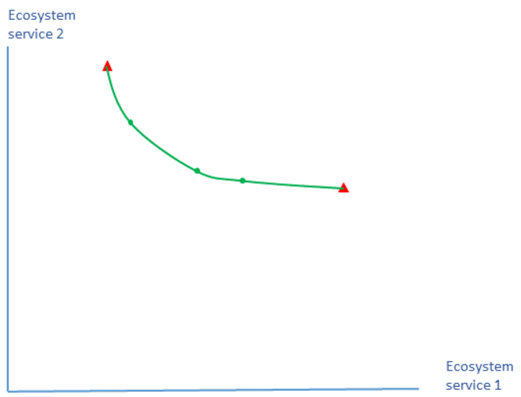

Figure 6.2.: *PPF exhibiting a **convex** trade-off between ES, implying combinations of extreme options dominate intermediate options*

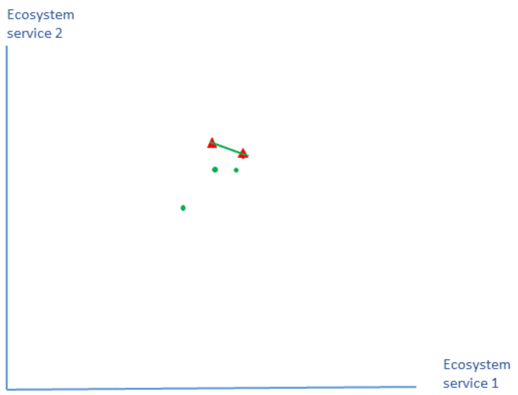

Figure 6.3.: *PPF exhibiting a synergy between ES*

Illustrative examples of the different shapes of PPF. In these figures, bundles of ES are represented with green dots and efficient bundles identified by efficiency analysis with red triangles. The PPF is the line joining the efficient bundles.

Many authors use production possibility frontiers in the literature on the optimal provision of ES. They generally estimate a functional form for the PPF and represent it explicitly, and then study its slope and curvature to assess interactions among ES. While a production possibility frontier can conceptually encompass many dimensions, most authors analyse them for 2 dimensions, and in the next paragraphs we also adopt this point of view.

Three characteristics of the PPF are useful to study trade-offs and synergies among ES: its length, its slope, and curvature.

Length of the PPF. The length of the PPF indicates whether the interaction is a synergy or a trade-off. A short PPF characterises a synergy, while a long PPF cor-

[18] Therefore, a trade-off between two ecosystem services doesn't mean that for some observations, the levels of these two ecosystem services cannot be increased simultaneously

responds to a trade-off. A synergy is a positive relationship among several ES, while a trade-off is a negative relationship. A synergy implies that maximizing one ES ensures the maximization of the other(s) and hence there is no major difficulty in defining which strategy maximizes the provision of ES. On the contrary, a trade-off among ES (downward-slopping PPF) calls for a compromise. In this case, the slope and curvature of the PPF inform respectively on the strength of the antagonism and the strategy to maximize the provision of ES. As synergies do not pose much problem in the maximization of ES, the literature tends to focus on issues related to trade-offs among ES.

Slope of the PPF. The slope of the PPF indicates how strong the trade-offs are. Wossink and Swinton (2007) conceptually relate the slope of the PPF to the strength of regulation needed to provide ES: the steeper the slope of a trade-off 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 trade-offs, information campaigns may be enough, while strong trade-offs require compensation payments. Sauer and Wossink (2013) apply this framework to a case study by estimating the production possibility frontier linking marketed outputs and ecosystem services, over fields included or not in agri-environmental schemes in England. They assess whether the commodity production and the provision of non-marketed ES stay in synergy or in a trade-off and what the marginal costs of providing non-marketed ES are (the monetary value of output lost when increasing non-marketed ES). In a rather similar approach, Ruijs et al. (2013) determine and estimate the PPF 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.

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 (on the same piece of land, by the same management option) or separately, and relates to the curvature of the PPF. Whether the PPF 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 PPF is convex as in Fig. 6.2, linear combinations of extreme bundles dominate intermediate bundles, meaning that a linear combination of extreme bundles provides more ES than bundles conciliating the provision of several ES. Such a linear combination is interpreted as the division of the landscape into different areas on which land uses performing very well with respect to the provision of one ES are adopted. 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 (i.e., the *land-sharing strategy*).

The opposite conclusion emerges when the PPF is concave (outward-bending) as in Fig. 6.1. Finding a concave PPF, 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. The smallest yield loss associated to an increase in biodiversity happens where yield is high. 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 few of the other(s)).

These interpretations of the shape of the PPF hold only when there are no interactions among neighbor 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, neighboring 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 neighboring 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 combination 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 provide them. Therefore, the use of the curvature of the PPF 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).

6.2.2. Efficient bundles to study interactions among ES

Bundles belonging to the frontier are efficient in a Pareto sense: there is no other bundle that achieves better on all ES simultaneously. They identify management options that maximise the provision of ES. They summarise complex interactions among multiple ES in a simple way.

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. This is particularly interesting since any aggregation implies assumptions over which ecosystem service should be prioritized, and impacts the results in a partly arbitrary way. Besides, it is also possible to analyze all dimensions altogether, compared to correlations which are always pairwise. Moreover, efficiency analysis is an interesting tool to select management options maximising the environmental outcomes (Ferraro, 2004).

Efficient bundles themselves can also be used to characterise synergies and trade-offs among ES. All analyses cited above rely on the slope and curvature of the production possibility frontier, either by representing it graphically or estimating it and calculating its slope and curvature. The same conclusions can be drawn only by identifying efficient bundles, without drawing the production possibility frontier itself or estimating it.

The number and relative position of efficient bundles of ES characterize the shape of the PPF and also enable to derive synergies and trade-offs among ES, as do correlation coefficients or the slope of the production possibility frontier. Let us recall that a bundle of ES is efficient if no other bundle provides more of all ES. Among efficient bundles, increasing one ecosystem service requires to decrease at least another one, and the production possibility frontier is necessarily downward-sloping. It follows that

- A large number of efficient bundles indicates a concave trade-off (see Figure 6.1).
- A small number of efficient bundles indicates either a synergy if the efficient bundles are quite similar in terms of ES provision (Figure 6.3),
- or a convex trade-off if the bundles are quite different in terms of provided ES (Figure 6.2).

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 drops the number of efficient bundles, this ES was standing on a concave trade-off with the other(s).

Relying on efficiency analysis and identifying efficient bundles as a mean to study interactions among ES has an advantage. It can encompass more than 2 dimensions, which is not the case with graphical representations or calculation of the slope of the PPF. Identifying efficient bundles enable to assess synergies and trade-offs among many ecosystem services, and summarise them. In addition, efficiency analysis directly identifies management options to maximise the provision of ES.

6.3. Efficient bundles of ecosystem services

In this section, we apply efficiency analysis to our simulated data set, and more specifically Data Envelopment Analysis (DEA) to study the interactions among multiple ES and identify efficient bundles of ES.

6.3.1. Implementation of DEA

Efficiency analysis techniques such as DEA 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 (the 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.

The rationale of our analysis is to assess which management option maximizes the provision of ES on the available land, and we focus on the provision of ES by the agroecosystem.[19] DEA is an appropriate tool to answer this question, as its principle is to find

[19]Land is considered as the only input at the landscape scale. In this section, labor, capital, pesticides or fertilizers belong to the different technologies: we don't seek to minimize them *per se*, and their detrimental consequences are already embedded in the ES provided by each management option. For example, a management option characterized by heavy use of pesticides will correspond to a

for each bundle of ES (called an *observation* in the DEA 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[20] (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 DEA which direction is the evaluated observation. This means that we examine, for each management option j and the associated bundle of ES Y_j, if a linear combination of other management options performs better in terms of ES provision, in the sense that it increases 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** β_j is interpreted as a potential proportional increase in all ES.[21]

Formally, this is done through a linear optimization problem under constraint on the production level, the right-hand side of the constraints having the proportional form $(1+\beta_j)Y_j$. For each management option (observation) $j = 1...N$, the optimization problem reads

$$\begin{aligned} \max_{\mu_i} \quad & \beta_j \\ \text{s.t.} \quad & \sum_{i=1}^{N} \mu_i Y_i \geq (1+\beta_j)Y_j \\ & \sum_{i=1}^{N} \mu_i = 1 \end{aligned} \tag{6.1}$$

where vectors Y_j and Y_i stand for the bundles of ES provided respectively by option j and each other option $i \neq j$. β_j represents the inefficiency score associated to observation j and is expressed as a percentage by which all ES could be increased at the same time with respect to the observed vector Y_j. For the efficient observations, it equals 0.

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

DEA works by comparing each bundle of ES to all possible linear combinations of all bundles of ES. To interpret it, we assume that linear combinations of bundles correspond to linear combinations of management options in the in the spirit of the land-sparing/land-sharing debate (Green et al., 2005; Phalan et al., 2011) and land use share models (Lichtenberg, 1989; Feng and Babcock, 2010; Lankoski et al., 2010; Lankoski and Ollikainen, 2011). The efficient benchmark is a linear combination of bundles of ES, and we assume that it corresponds to the adoption of the corresponding **combination of management options** on the agricultural area, i.e. several management options

bundle of ES with a higher production but a low level of water quality. In the next section, we will consider these other inputs through their influence on agricultural profit.

[20] The production possibility set is understood here as the space delimited by the linear combinations of all bundles.

[21] 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.

with different shares. These shares are given by the μ_i. This makes sense only because no interactions among management options.

Using this approach, **we perform two analyses** on the 121 simulated bundles of ES. First, we run the DEA on the five ecosystem services (agricultural output and the four non-marketed services) to find out which ones are efficient, and describe the overall interactions among our set of ES, in particular the trade-offs between provision and regulation services. Second, we run the DEA 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. The analyses are run with software R (package $Rglpk$). Detailed results are presented in the appendix 8.6.

6.3.2. Results of the efficiency analysis

Shape of the PPF and trade-offs among ES

Efficiency analysis allows us to identify the efficient bundles of ES among the 121 simulated bundles in our model. Efficient bundles maximize 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. In our data set, each bundle of ecosystem services corresponds to one unique management option. Efficient bundles correspond to management options that make an efficient use of scarce land. We refer to them as efficient management options. Their number and relative position also characterizes the shape of the PPF.

From the results of the 2 DEA run, we can state that

1. Only few bundles are efficient when considering the five ES, so that most of the management practices are not efficient: at least 100 out of 121 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. 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 maximized by grassland and by the least intensive cropland, with reduced tillage, low fertilization and pesticide use, biomass input and NCH. This shows a strong synergy among regulating ES, which are jointly produced by the same agricultural practices. It also points out the general antagonism between production and non-marketed ES. This is confirmed by the interpretation of the correlation coefficients and the shape of the PPS.

3. 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 provision and regulation services are possible. Among efficient bundles of ES, some show intermediate levels of all ES. This is a hint that the relationship between agricultural production and the other ES is probably rather concave, and

that some management options conciliating production and non-marketed ES are efficient.

These results apply for all the agronomic contexts we analysed, 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 efficient in high-yield contexts.

Efficient management options

Identifying efficient bundles of ES enable to identify management maximising the provision of ecoystem services. We analyse the agricultural practices of efficient bundles of ES. They are varied, from very extensive to very intensive ones. Among efficient bundles, intensity in pesticide is correlated with in intensity in fertilisers: no efficient management option involves very high level of fertiliser and low levels of pesticides or conversely. Almost all efficient bundles involve reduced tillage, except for very intensive ones. Concerning other agricultural practices (biomass input and non-crop habitat), no systematic link is observed.

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.

The efficiency analysis identified which options maximize 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-efficient. This is what we examine in next section.

6.4. Cost-efficient bundles of ecosystem services

In the following, we focus on the provision of non-marketed ecosystem services, given that only non-marketed ES are underprovided, and that agricultural production stands in a trade-off with them. Moreover, as explained below, agricultural production is accounted for in the cost of providing non-marketed ES.

In a context in which land is a scarce resource, such as in Western Europe, the previous analysis is useful to identify strategies that maximize the provision of ES on this fixed amount of land. However, 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 the public budget via subsidies. For example, the current European budget for agri-environmental policies is too small to cover all the land concerned by their implementation.[22] As a

[22] Over the period 2007-2012, only 25% of the agricultural area was covered by agri-environmental schemes in the EU (Duval et al., 2016), although maximizing the provision of ES probably means

consequence, to focus on realistic strategies, the opportunity cost should be considered as a criterion to minimize along with the maximization of the provision of non-marketed ES, as shown by Naidoo et al. (2006). We now explore cost-efficient strategies to provide non-marketed ES, computing the opportunity cost of bundles of ES.

6.4.1. 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 example giving an object to someone is not costly in the first sense, but induces an opportunity cost as the owner gives up the opportunity to sell it. The same holds for farmers: changing their 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, we assume that farmers behave as rational economic agents and choose the most profitable management option[23]. The **statu quo** is the most profitable option, and any change in the bundle of ES incurs 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 crucial to seek to minimize 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 ecosystem service 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. This way to define opportunity cost is, however, problematic for two reasons. 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., fertilizer use). To overcome this limit, we consider the difference in profit.[24] 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: "From the firm's point of view, the allocation of costs between joint products is essentially arbitrary" as stated by Baumgärtner et al. (2001), 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.

enrolling a greater area.

[23] This is of course an approximation, as farmers may consider other criteria than profit (working time, tediousness, ...), and in general human beings do not always behave rationally. We, however, consider the maximization of profit as a rather good approximation of farmer's behavior for our research question, and it corresponds to the logic behind common agri-environmental policies.

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

We define the **opportunity cost of a bundle of ES** as the difference between the profit of the corresponding management option and that of the most profitable option (statu quo).

With this opportunity cost, we conduct a cost-efficiency analysis over the possible bundles of ES. This analysis aims at simulatneously maximising the ecosystem services and minimising the associated cost.

6.4.2. 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. Revenues equal production times an exogenous price, for each type of production. Management costs depend on agricultural practices. 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. Table 1 in appendix 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 in Table 5.1 in the Appendix).

The statu quo depends on the agronomic context. Because of the management costs, production and profit are not perfectly correlated, and hence the statu quo is not necessarily efficient in terms of ES provision (including agricultural production):

Statu quo (most profitable management options)

- For agronomic contexts characterised by low potential yields (i.e., contexts 1 to 5 in Table 5.1, corresponding to potential yields up to $Q = 6.6\ t.ha^{-1}$), the most profitable management option corresponds to grassland (management option # 1 in Appendix 8.6). This option is efficient in terms of ES provision.
- For agronomic contexts characterised by higher potential yields (i.e., contexts 6 to 10 in Table 5.1, corresponding to potential yields above $Q = 6.6\ t.ha^{-1}$), 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 with limited fertilisation for contexts 6 to 8, and management option # 71 with higher fertilisation for contexts 9 and 10). **These options are not efficient in terms of ES provision**.

These results underline that the possible increase in ES provision differs according to the statu quo, and thus according to both agronomic conditions and relative profitability of management options. For example, where grassland is the statu quo, it is impossible to increase the provision of all non-marketed ES. Agri-environmental policies need to be differentiated, and may be of less utility in areas where the statu quo provides high levels of ecosystem services.

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-efficient ways to increase the provision of non-marketed ES. Adopting an agri-environmental 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.

6.4.3. Cost-efficiency analysis

The maximisation of non-marketed ES identifies only two efficient bundles, but possibly with high opportunity costs (according to the agronomic context). Many other bundles represent an smaller increase in non-marketed ES compared to the statu quo, but with possibly a lower opportunity cost. Cost-efficiency analysis identifies options that jointly maximise ecosystem services and minimise the opportunity cost. In presence of a budget constraint, these options are the ones that maximise the provision of ecosystem services.

We run cost-efficiency analysis: a data envelopment analysis maximising non-marketed ES with a constraint on the opportunity cost. In such a setting, the cost is treated as an input. We maximise the increase in non-marketed ecosystem services, under an additional constraint that the cost of the linear combination of ES must not exceed the one of the bundle analysed.[25]

$$
\begin{aligned}
\max_{\mu_i} \quad & \beta_j \\
\text{s.t.} \quad & \sum_{i=1}^{N} \mu_i Y_i \geq (1+\beta_j) Y_j \\
& \sum_{i=1}^{N} \mu_i C_i \leq C_j \\
& \sum_{i=1}^{N} \mu_i = 1
\end{aligned}
\tag{6.2}
$$

where the vector Y_i stands for the bundle of ES provided by the alternative options i. β_j represents the inefficiency score associated to observation j and is expressed as a percentage by which all ES could be increased at the same time with respect to the observed vector Y_j. For the efficient observations, it equals 0. C_i is the opportunity cost of bundle i, and μ_i the weight of bundle i in the efficient benchmark, i.e. the land use share of management option i in the efficient benchmark.

6.4.4. Results of the cost-efficiency analysis

The cost-efficiency analysis identifies between 2 and 13 cost-efficient options in each agronomic context. Again, all cost-efficient options in contexts with low potential yield are also cost-efficient in contexts with higher potential yield.

The results first illustrate that cost-efficiency analysis identifies much more management options than efficiency analysis: only two management options provide efficient bundles of non-marketed ecosystem services, while between 2 and 13 bundles are cost-efficient. This is a consequence of the trade-off among non-marketed ES and the op-

[25] Compared to a specification where the cost is properly minimised, this doesn't change the bundles identified as efficient, only the scores.

portunity cost. All efficient bundles are of course cost-efficient, the other cost-efficient bundles provide less ecosystem services, but at a lower cost.

The second result is that there are striking similarities between the efficient bundles identified with 5 ES (4 non-marketed ES and production) and the cost-efficient ones (replacing production by the opportunity cost), meaning there is no major contradiction between a strategy maximising all ES, and a strategy maximising non-marketed ES and minimising the opportunity cost.[26] The latter strategy identifies less efficient bundles, but which are all efficient in the first one.

The cost-efficiency analysis brings new insights. First, some options that were efficient when considering production are not efficient when considering their cost, in particular very intensive options, which bear a high management cost (because of the intensive use of fertilisers and pesticides): increasing production can be costly. In parallel, certain options that involve less production but also lower management costs are cost-efficient, while they were not efficient with the production. Besides, even if it is not efficient when considering production, the most profitable bundle is cost-efficient, which makes sense since its opportunity cost is the lowest (it equals 0). Second, as shown on Figure 6.4, these cost-efficient bundles correspond to a large range of levels of each ES. This indicates that gains in ES may be achieved at a small economic cost. If the aim is to provide intermediate levels of all non-marketed ES while minimising the cost, the solution may be to provide them together rather than separately, and favouring a shallow but widely applied increase in ecosystem services.

Efficiency analysis imposes no hierarchy among criteria, and thus a bundle may be cost-efficient and very costly if it provides slightly more ES than another. The set of cost-efficient bundles represent very diverse orientations, a cost-efficient bundle doesn't necessarily correspond to the preferences of society. In the absence of the specification of preferences, it is impossible to rank cost-efficient bundles against each other.

6.5. The cost of two strategies to provide non-marketed ecosystem services

Each bundle of ecosystem services represents a certain orientation, a certain proportion of non-marketed ES. The cost-efficiency analysis attributes a cost-efficient benchmark to each cost-inefficient bundle. This cost-efficient benchmark provides the same proportions of non-marketed ES, and determines for every orientation the cost-efficient way to provide non-marketed ES. This cost-efficient benchmark is a combination of bundles of ES, and we assume that it is possible to achieve it by adopting the corresponding combination of management options determined by the land use shares μ_i. The management options and shares of the cost-efficient benchmark are different for every bundle of ecosystem services. In theory, this is the solution to provide non-marketed ecosystem services in a cost-efficient way.

In reality, it is doubtful that policy makers are prone to consider such a diversity of cost-efficient strategies to provide non-marketed ES. Here we compare the cost of two

[26] Running a DEA with both the agricultural production and the opportunity cost confirms this result: the number of efficient bundles doesn't increase much compared to the DEA in the last section.

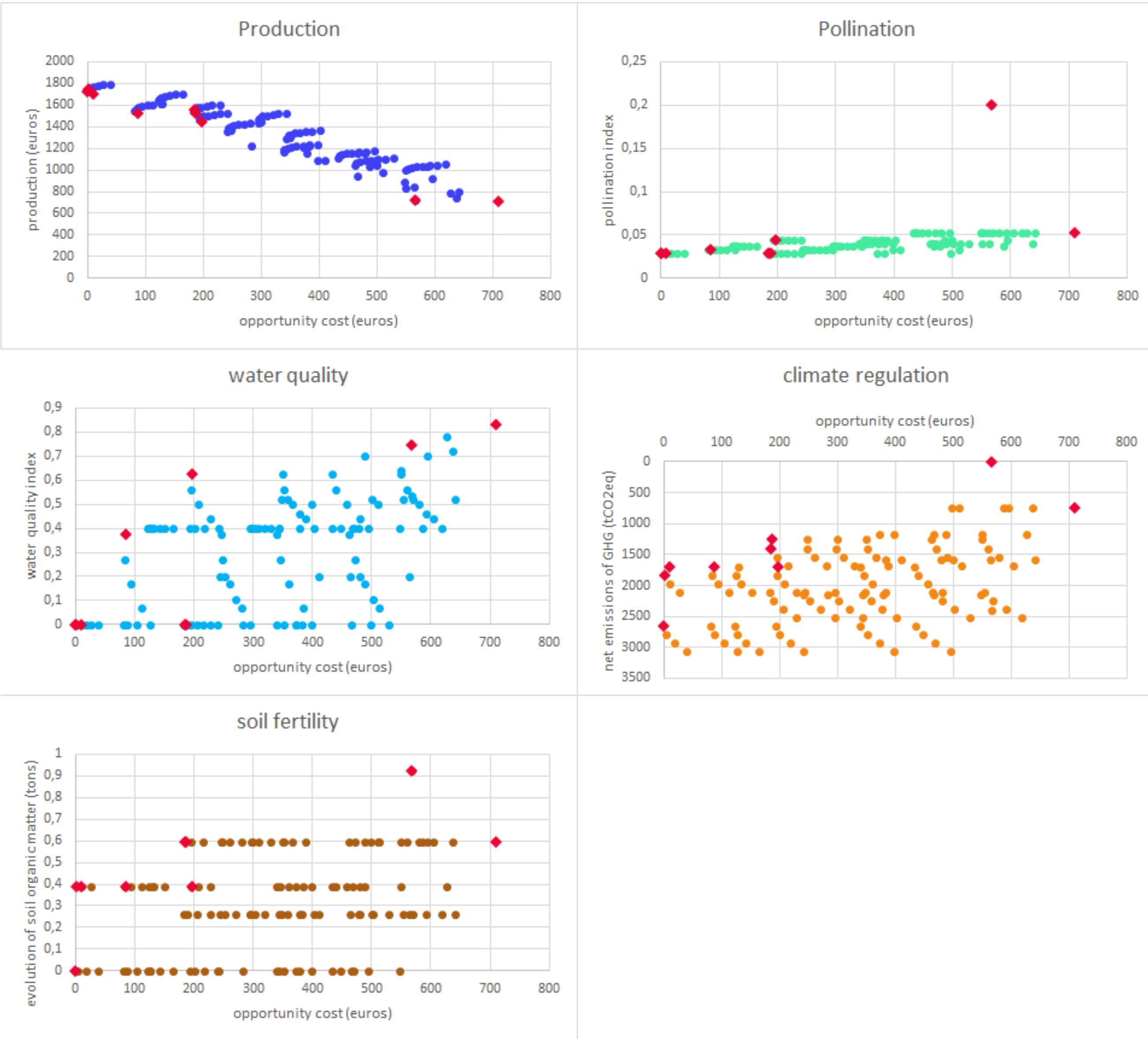

Figure 6.4.: *Cost-efficient bundles (marked in red diamonds) correspond to a large range of levels of all ES. Each points corresponds to a bundle of ES, with the level of each ES on the y-axis, and the opportunity cost on the x-axis. (context 8, potential yield = 9.9 t/ha)*

strategies to increase the provision of non-marketed ES compared to the statu quo.

6.5.1. Principle

The opportunity cost of a bundle is defined as a difference in terms of profit with respect to the statu quo (the most profitable option), and for consistency we now measure ES levels relative to the levels provided by the statu quo. Each management option thus corresponds to an alternative to the statu quo characterised by an opportunity cost and a variation in non-marketed ES provision.

The efficiency analysis of section 6.3 identified that only two management options maximise non-marketed ES provision: grassland and the least intensive cropland. Thus, all other options are inefficient in terms of non-marketed ES and the same bundle of ES could be provided on a smaller area by the **ecologically efficient benchmark** (a combination of the two efficient management options in terms of non-marketed ES). To improve the non-marketed ES provision of a given area, there are thus two strategies:

- **sprinkling strategy**: adopting a management providing more non-marketed ES on the whole area, or
- **concentrating strategy**: dedicating part of the landscape to the provision of non-marketed ES (by adopting the ecologically efficient benchmark), and leaving the rest under the statu quo.

For each bundle of ES (representing given proportions of ES), which of these strategies induces 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 benchmark is more costly to provide the same amount of ES.

To compare the cost of both strategies, we run again a DEA analysis with the four non-marketed ES, but expressing the provision of ES as a difference with the statu quo levels.[27] The DEA identifies the same bundles maximising non-marketed ES as previously (grassland and the least intensive cropland), but above all it defines the ecologically efficient benchmark for every other (inefficient) management option, as well as an efficiency score.[28] The ecologically efficient benchmark of an option j is the efficient combination of the two efficient options that increase the ES with respect to the statu quo in the same proportions than the inefficient management option j. The efficiency score β_j is the proportion by which the increase in ES can be enhanced, or equivalently, $1/(1+\beta_j)$ is the area needed to achieve the same improvement with the ecologically efficient benchmark (the rest of the area remaining under the statu quo, incurring no opportunity cost and providing no additionnal ecosystem services).

The sprinkling strategy, adopting a given management option LU_j on one unit of land is associated with an opportunity cost C_j and change in ES provision ΔES_j (second column of Table 6.1). On the contrary, the concentrating strategy, adopting the ecologically efficient benchmark LU_e on one unit of land costs C_e and provides $(1+\beta_j)\Delta\text{ES}_j$ (third column of the table). To compare both strategies for a given increase in ES, we consider the adoption of the ecologically efficient benchmark on $1/(1+\beta_j)$ units of land (fourth column).

To assess which solution is least costly to provide ES, we compare the cost of option j (C_j) and the cost of its efficient benchmark providing the same amount of ES ($C_e/(1+\beta_j)$).

[27] We thus run a directional DEA which direction is given by the variation of non-marketed ES of each option with respect to the statu quo.

[28] Efficient bundles are the same as in the previous DEA 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.

	Sprinkling strategy option j	ecologically efficient benchmark (providing more ES than option j)	**Concentrating strategy** ecologically efficient benchmark providing as much ES as option j
land use shares	LU_j on 1 unit of land	$LU_e = \sum \mu_k LU_k$ on 1 unit of land	LU_e on $1/(1+\beta_j)$ unit of land (*statu quo* on the rest)
cost	C_j	$C_e = \sum \mu_k C_k$	$C_e/(1+\beta_j)$
ecosystem services difference with the *statu quo*	ΔES_j	$(1+\beta_j)\Delta\text{ES}_j$	ΔES_j

Table 6.1.: *Opportunity cost and ecosystem services provided by option j and its ecologically efficient benchmark*

6.5.2. Results (Least costly strategy to provide ES)

The sprinkling strategy should be implemented on an agricultural area only if the opportunity cost C_j satisfies $C_j \leq C_e/(1+\beta_j)$, where C_e is the opportunity cost of its ecologically efficient benchmark and β_j is its efficiency score. Otherwise, the concentrating strategy should be implemented: the ecologically efficient benchmark should be adopted on a share $1/(1+\beta_j)$, the rest remaining under the statu quo management option.

This is illustrated on Figure 6.5, which represents the strategy of adopting the sprinkling strategy (option j) on the left, and the concentrating strategy (its ecologically efficient benchmark) on the right-hand side. The ES provided (measured as the difference compared to statu quo) are represented by the bright green area, the opportunity cost by the orange area. On the right-hand panel, the rest of the 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, this land provides no additional ES and incurs no additional cost. To equal the increase in ES provided by adopting option j on 1 hectare, the land on which its efficient benchmark has to be adopted is limited to $1/(1+\beta_j)$, as represented by the green arrow, so that both green areas are equal. Determining which strategy is cheaper to provide the ES bundle is equivalent to determining which orange area is the smallest. The solution is obviously to adopt the eologically efficient benchmark when $C_e < C_j$, but it depends on β_j when $C_e > C_j$, as represented on the figure.

This comparison shares similarities with the land-sharing/land-sparing comparison. They both compare two strategies: one conciliating marketed output and non-marketed output within the same production process, the other separating them. However, our approach differs from land-sparing/land-sharing comparisons. First, land-sparing/land-sharing comparison relies on the trade-off between biodiversity and commodity production. Our approach relies on the trade-off between non-marketed ES and opportunity cost. Non-marketed ES bear the same logic than biodiversity, but the opportunity cost is of a different nature than agricultural output: it depends on the agronomic conditions and on economic variables. Second, the land-sparing strategy assumes that concentrating biodiversity protection spares land that can then be cultivated more intensively. In

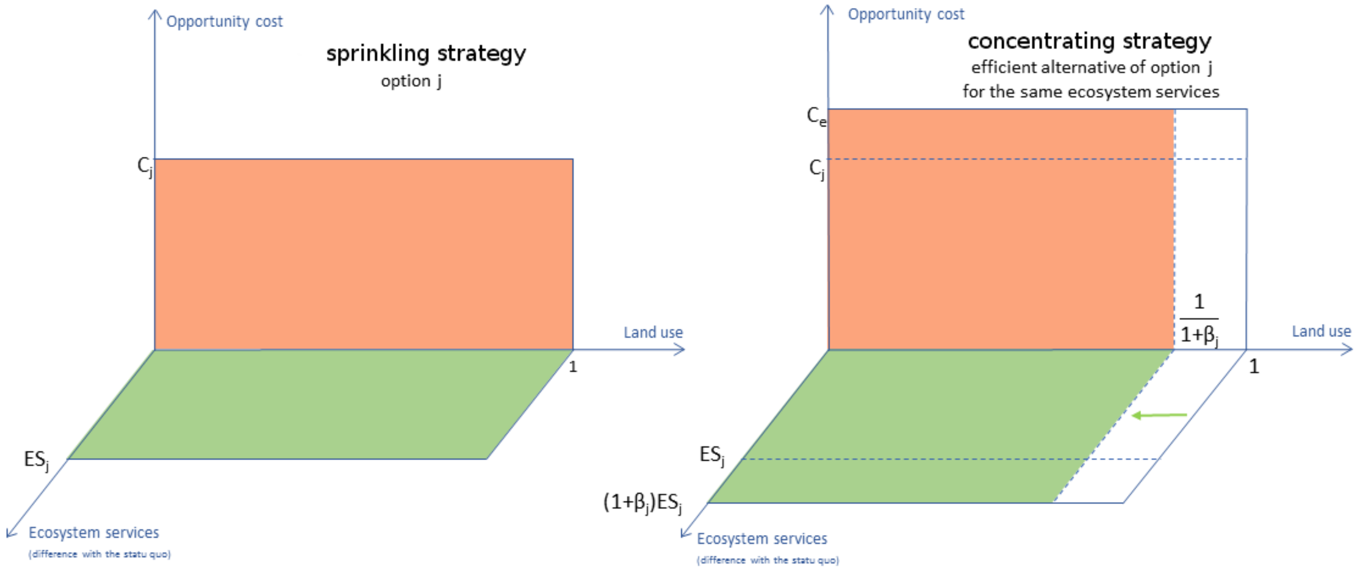

Figure 6.5.: *Illustration of two strategies to provide non-marketed ES: comparison of a given management option and its ecologically efficient benchmark.*
This example is illustrated with an ecologically efficient benchmark which is more costly for a given area than the option j, but this is no general case.

our case, the land "spared" by a more intensive provision of non-marketed ES is cultivated under the statu quo. We don't consider increase in cropping intensity on this part of the landscape, and the statu quo is not the most intensive cropland option. However, our methodology is a way to adapt the debate over the strategies to provide non-marketed ES to the multiple and complex interactions among ES, and to the importance of the opportunity cost. Namely, we rely on efficiency analysis as a way to summarise these complex interactions and select strategies maximising ES. We reason with given proportions of ES, and compare only strategies that provide the same levels of ES. This analysis also shows 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, and accounts for it in the analysis.

The results are detailed in Appendix 3 for agronomic context 8 (i.e., $Q = 9.9\ t.ha^{-1}$). They show that for about half of the 121 options, the sprinkling strategy is less costly than the concentrating strategy (green-colored lines in the table). This is also true for options that were not identified as efficient in the first efficiency analysis with all 5 ES including agricultural production. It is difficult to identify a clear pattern that could explain in which case each strategy is least costly. This depends both on the options (and thus on the proportions in which the ES are increased), and on the agronomic context. However, we notice two facts. First, in contexts with medium potential yield, the best strategy is often the concentrating strategy (very extensive management on part of the land and statu quo on the rest). Second, in contexts with high potential yield, both strategies are interesting, depending on the exact increase in non-marketed ES. In general, the sprinkling strategy is more often the least costly than in the contexts with medium potential yield. The results support the idea that a modest but broadly applied in ecosystem services might not be necessarily a bad strategy in intensive agricultural landscapes, but there is no one-size-fits-all solution.

Agricultural production is always lower for the concentrating strategy than for the sprinkling strategy. The difference is higher for intensive options and productive landscapes. This should be taken into account if the objective of maintaining yields in important.

6.6. Discussion and conclusion

This chapter develops methods to maximise the provision of multiple and interacting ES while minimising its cost. The originality of our agrecological model is to include agricultural practices as drivers, which is more realistic than analyses based on land use or land cover. We use efficiency analysis to identify management options which maximise ES provision and characterise interactions among ES. Such a tool is a way to consider multicriteria decision-making problems without aggregating the different dimensions.

Our results can be used to identify key recommendations to design agri-environmental policies aiming at compensating farmers for changes in their agricultural practices. First, we highlight the need to carefully select the management option to maximise the provision of ES. Among the important number of options, only a minority are (cost-)efficient. Second, our analysis underlines the crucial role of the agronomic context in the determination of efficient management options. Even without considering the cost of providing ecosystem services, options maximising ES differ according to the agronomic conditions. In less productive agricultural areas, a smaller number of options are efficient. This is reinforced when considering the need to minimise the cost of providing ecosystem services. We argue that this cost should be measured for a bundle of ES, and relative to the statu quo (the option chosen by farmer in the absence of agri-environmental policies). The statu quo depends on the profitability of the different options and thus on the agronomic conditions, and thus the opportunity cost of providing non-marketed ES is likey to be greater in productive agricultural areas, where the statu quo provides less ES and where the yield loss associated to the provision of non-marketed ES is higher. This calls for a differentiation of agri-environmental policies among agronomic contexts. Third, in order to maximise the provision of ES when financial resources are scarcer than land, our results show that considering the cost of providing ecosystem services is as important as considering the levels of ecosystem services. They also show that the least costly strategy to provide ES depends on the targeted increase in ES, but that fostering a modest increase in ES throughout the whole agricultural area can be a strategy maximising the provision of non-marketed ES, and limits the yield loss.

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 non-monetary hurdles related (risk aversion, role of habits etc.). Thus the opportunity costs we consider do not measure the subsidies needed to make farmers change their practices, but only part of the social cost of promoting an increase in non-marketed ecosystem services. Also, we never consider an acceptable profit for farmers and consider that agri-environmental policies only aim at increasing the provision of non-marketed ES, not at providing income support to farmers. Our conclusions should be interpreted while bearing this in mind, especially concerning grasslands.

Another limit of our work is that we consider homogeneous agricultural regions. Chapter 8 extends 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 to this chapter: Including livestock impacts

In our agroecological model, livestock impacts can be included or not. The results presented in this chapter don't include them. We present here what they change.

Livestock impacts only concern one management option: grassland. When accounted for, livestock contributes to climate change, and adds mineral nitrogen in the soil, which impairs water quality. Accounting for livestock impacts therefore decrease the levels of two ecosystem services in grasslands: climate regulation and water quality. We assume that livestock doesn't impact the pollination or the soil fertility.

Efficiency analysis Without impacts of livestock, grassland provides higher levels of most non-marketed ecosystem services, except water quality. When potential yield is low, it even provides a higher production than cropland. Including livestock impacts makes grassland less superior to other options. In the efficiency analysis with all 5 ecosystem services, this has two consequences. First, new bundles appear to be efficient. Those bundles feature no fertiliser and reduced tillage: they produce more than option 6, but still have higher levels of water quality and climate regulation compared to grassland. Second, options that were efficient only in contexts with high Spotential yield (i.e. only because of their production level) are efficient in contexts with low potential yield.

The efficiency analysis with only the non-marketed ES is also modified. One more option is efficient: option 4, identical to option 6 (least intensive cropland), but without non-crop habitat.

These changes don't disrupt the conclusions obtained without including the impacts of livestock.

Cost-efficiency analysis Similarly to the efficiency analysis, including livestock impacts in the cost-efficiency analysis has two consequences. Some extensive management options become cost-efficient, and options which were cost-efficient only with high potential yield become cost-efficient even with lower potential yield.

Comparing the cost of two strategies Including livestock impacts doesn't change much the results, and doesn't change their interpretation.

Given the small changes incurred by livestock impacts in our analyses, we don't include them in the analyses of the next chapters.

7. Economic incentives to implement solutions maximising ES provision

7.1. Introduction

Public policies are required for the provision of non-marketed ecosystem services, for example through payments such as the Agri-Environmental Schemes in the EU. These policies have to tackle the provision of multiple and interdependent ecosystem services and achieve management options maximising the provision of ES with a limited budget. To do so, policies aim at compensating the opportunity cost of providing ecosystem services through economic incentives. These incentives must comply with **participation constraints**: not only must they compensate the opportunity cost, but they also need to make the targeted management option the most attractive for farmers, as policies relying on incentives cannot force farmers to adopt a given management option[29]. However, given the interactions among ecosystem services and the complex relation between the provision of ES and the opportunity cost, the sum of incentives needed to respect the participation constraints may be higher than the opportunity cost, thus overcompensating the farmer and reducing the remaining budget for the provision of ES.

The type of incentives reflects on the participation constraints, and may have consequences on the options which can be implemented, and the associated policy budget. Current policies are mainly based on actions (agricultural practices), but recent studies advocate for a shift towards result-based incentives which shall better ensure the maximization of ecosystem services provided (Schwarz et al., 2008). On the one hand, microeconomic theory shows that result-based incentives better target (cost-)efficient bundles of outputs. On the other hand, this theoretical result holds only under the assumptions that outputs stand in a concave trade-off, and empirical studies have shown that numerous synergies among ES exist (Lee and Lautenbach, 2016), and that some interactions among result-based incentives may occur (Bryan and Crossman, 2013), which could pose problem for the calibration of incentives. Moreover, management costs are not accounted for in a realistic way by these theoretical studies, and the role of participation constraints and the policy budget are likely to be underestimated. The aim of this chapter is to explore whether a set of result-based incentives is better suited than a set of action-based incentives for the provision of cost-efficient bundles of ES.

In Chapter 3, we identified two channels through which interactions among ES could cause problems with result-based incentives:

[29] Other policy instruments like norms force farmers to adopt certain management rules (e.g. the cross-compliance in the Common Agricultural Policy), but most most policy instruments in agri-environmental policies rely on incentives.

1. result-based incentives, when calibrated incoherently, may target non-existing bundles of ES
2. interactions among result-based incentives may increase the policy budget (i.e. the sum of subsidies) needed to make farmers adopt a given management option

In this chapter, we underline the importance to consider not only the opportunity cost as in Chapter 6 but also the policy budget needed to adopt a management option, which is likely to differ from the opportunity cost. Therefore, we move from cost-efficiency analysis (as in the previous chapter) to an analysis including the policy budget, and compare how the two types of incentives behave.

This chapter relies on the theoretical economic chapter (Chapter 3). We use the simulated data to model interactions among ES and assess the impact of economic incentives on the management option chosen by a profit-maximising farmer, by means of a simple microeconomic representation and linear programming methods.

This chapter is organised as follows: in Section 7.2, we review existing literature on the regulation of multiple and interacting ecosystem services; we present the methods in Section 7.3; present some results in Section 7.4, and eventually conclude.

7.2. Literature review: incentives for the provision of multiple and interacting ES

The existence of multiple ES to regulate and interactions among them raises research questions about the provision of multiple ecosystem services and the multifunctionnality of agriculture. We reviewed results of economic theory on the regulation of public goods and joint output in Chapter 3, but focus here on more applied approaches, mainly based on modelling.

This is an interdisciplinary topic, it has been studied by researchers from both (agro)ecology and economics, with different focuses.

7.2.1. Using interactions among non-marketed ES to regulate their provision

Among the economic literature, joint production of marketed and non-marketed outputs is seen as an advantage for the regulation of the latter by Peterson et al. (2002) and Abler (2004). These theoretical economic studies take interactions among outputs as granted and use the concept of joint production to analyse the regulation of the multiple outputs of agriculture. Peterson et al. (2002) exclude a direct regulation of non-marketed outputs of agriculture because they are difficult to measure. Rather, they propose to rely on inputs which contribution can be allocated to the different outputs. For example, by setting different incentives for labour used to provide marketed output and labour used to provide non-marketed outputs. Abler (2004) states that the provision of non-marketed outputs is mainly related to land, so that using incentives based on land use may be a solution. Both authors seem to indicate that it is possible to regulate non-marketed outputs by playing on the links between inputs and outputs. Their approaches rely on simplistic representations of relations between agricultural management and ecosystem services. They don't represent explicitely the decision-making process of the farmers and how incentives play in it, so that their conclusions on the regulation of non-marketed outputs don't reflect the participation constraints.

Agroecological studies also take interactions among ES as a starting point to study the value of integrative management. Galler et al. (2015) demonstrate the value of coordination: they show that integrative agri-environmental schemes (tackling the provision of several ES in a coherent way) are more efficient and cost-efficient than maximisation of individual objectives. Such results are obtained without considering the implementation with economic incentives and the inherent participation constraints. Accounting for participation constraints, White et al. (2012) shows that cooperation for the regulation of interacting ecological objectives (they take the example of several fish species) brings an overall benefit, which may not be shared among sectors in a equitable way. Howlett and Rayner (2013) underline that when facing multiple objectives, an incentive portfolio is necessary, that takes into account the coherence of incentives.

7.2.2. Result-based incentives and unwanted side-effects

Some applied analyses compare action-based and result-based incentives but they consider specific cases, with few ecosystem services standing in a concave trade-off. Other authors have shown the possible existence of interactions among result-based incentives, without exploring further their consequences. Examples of antagonisms among result-based incentives leading to unwanted side-effects have been identified in the literature. Numerous cases of antagonisms among incentives have been identified in the literature Stavins and Jaffe (1990); Chisholm (2010); Jack et al. (2008), where generally the effect of one incentive is cancelled or decreased by the antagonistic incentive. For example Lubowski et al. (2008) reports that the effects of the conservation reserve programm in the USA in order to protect forested wetlands were diminished by the parallel measures in favor of flood control and drainage, which increased the profitability of agricultural land use and the incentive to convert land to agriculture. Jack et al. (2008) suggests that in these cases, to increase the provision of non-marketed ecosystem services or public goods, eliminating an incentive may be more effective than setting a new incentive targeting the non-marketed ecosystem service.

The interactions among ecosystem services are likely to reflect into interactions among result-based incentives and undermine the budget-efficiency of policies through two effects. In the case of a synergy among ecosystem services, a policy ignoring the synergy may miss the beneficial side-effects of one incentive on the other. On the contrary, in the case of trade-off, the policy may underestimate the level of incentives needed to trigger a change, by ignoring the negative side-effect of one incentive on the other. Calibrating smartly result-based incentives may indeed be very cost-effective to regulate several interacting ecosystem services (Crossman et al., 2011).

However, no study investigates the consequences of these interactions on the budget-efficiency of result-based incentives, compared to action-based incentives.

7.3. Methods

The applied analysis is based on the simulated agroecological and economic data set and makes use of efficiency and cost-efficiency analyses of previous chapter.

Our model makes use of discrete modalities of agricultural practices, and thus has no specified cost function linking the cost to the management options, but it considers a wide variety of management options, so that it captures the large ranges of options available for a farmer, and the difficulty for the regulator to calibrate economic incentives. We model the decision-making of a farmer by assuming that he/she will choose the management option with the highest profit[30]. The decision-making process is based on the **ranking** of options, and economic incentives aim at changing which option is the most profitable. By doing that, they change the profit of all options. Therefore, it is not enough to calibrate incentives so that their sum compensates the opportunity cost of changing management option, incentives must also ensure that all other options are less profitable. This is the **participation constraint**. This is the point where interactions among incentives become an issue, as mentioned at the end of Chapter 3.

Including many possible management options and considering participation constraints are two interesting features of our approach. For example, studies as Wünscher et al. (2008) compare incentives based on the opportunity cost of the management option they target. By doing so, they implicitely assume that the policy makers can just give the exact amount of money corresponding to the opportunity cost to each farmer in order to achieve this management option, and neglect the participation constraints. Bryan and Crossman (2013) account for the participation constraints by modelling decision-marking by the selection of the most-profitable land use, but they consider only 4 alternative management options (land uses in their cases), which probably underestimate the variety of decisions farmers are confronted to. As the opportunity cost is a relative measure, including most possible management options available to farmers is important to assess it precisely.

7.3.1. Simulating the policy budget

In the previous chapter, we defined for each management option an opportunity cost, i.e. the profit difference with the most profitable management option. As we saw, because implementing agri-environmental policies must be done through policy instruments and especially economic incentives, and the sum of the subsidies required to make a farmer adopt a given management option may exceed the opportunity cost. We call **policy budget** the minimum sum of subsidies needed. The policy budget represents the cost supported by the society to increase the provision of ecosystem services. In this section, we detail how we simulate this policy budget, for each type of incentives and each management option.

We introduce economic incentives to change the bundle of ecosystem services provided. In the absence of incentives, the bundle of ES provided is the one corresponding to the most profitable option (statu quo). Incentives modify the profit of all bundles, and

[30]Such an approach can also consider extra-profit criteria, the important is to be able to rank management options according to a unique indicator, may it be monetary or not.

thus change which option and which bundle of ES is the most profitable. With linear programming, we determine, for each option, incentives that minimise the policy budget.

We introduce **sets of incentives**

- sets of action-oriented incentives, combining 6 subsidies based on agricultural practices: subsidies for grassland, reduction in fertilizer and pesticide use, for reduced tillage, for biomass input and non-crop habitat
- sets of result-based incentives, combining 4 different subsidies based on the level of non-marketed ecosystem services provided: pollination, enhancement of water quality, limitation of GHG emisions, and increase of soil organic matter.

We choose to consider only subsidies (no taxes) which encourage the provision of non-marketed ecosystem services and more agroecological practices. For realism, we consider only "agroecological" incentives, and hence don't consider a price bonus (and no malus to avoid introducing taxes). Considering both taxes and subsidies would create a biais given that we seek to minimise the policy budget: to minimise the policy budget, our method would automatically favor taxes over subsidies. This choice also ensures that the sum of subsidies required to make a management option the most profitable is greater than its opportunity cost, which corresponds to the principle governing Agri-Environmental Measures of the CAP and other common agri-environmental measures. This choice implies to define every action and result as a positive variable, and sometimes requires to define them relative to a reference level. For example, subsidies for the reduction in fertiliser level require to set a reference level. Here, we choose the reference level as the practices and levels of ecosystem services of the most profitable management option (statu quo) in the agronomic context with the highest potential yield.

This analysis aims at providing more applied insights into the relative efficiency of result-based incentives and action-based incentives, as introduced in theory in Chapter 3. By efficiency of incentives, we refer to the simultaneous minimisation of policy budget and maximisation of ecosystem services, which would be the objective of a benevolant regulator acting in the general interest. Therefore, for every management option, we seek to minimise the sum of subsidies needed to make this option the most profitable (the necessary condition for it to be implemented by farmers) [31].

In the case of result-based incentives, we solve the following linear programm for each management option j:

$$\begin{aligned} \min_{X_j} \quad & X_j \cdot Y_j \\ \text{s.t.} \quad & X_j \cdot (Y_j - Y_i) \geq \pi_i - \pi_j \qquad \forall i \neq j \\ & X_j \geq 0 \end{aligned} \tag{7.1}$$

where X_j stands for the vector of solutions (incentives) solving the optimisation programm, and Y_j stands for the bundle (vector) of ES, so that the objective to minimise corresponds to the sum of subsidies (the policy budget).

[31] Note that the result-based incentives calibrated this don't exactly correspond to the "pigouvian" solution, as they are not calibrated at the marginal social benefit, nor at the marginal cost

π_j is the profit of option j in the absence of any subsidies, as modelled in the ecological-economic model. The first set of constraints imposes that with the set of incentives, the profit of option j be greater than equal to the profit of every other option, which is the condition for it to be adopted by a farmer as modelled in our economic framework. The last constraints impose the incentives to be non-negative, which correspond to the choice of modelling only subsidies encouraging the provision of non-marketed ecosystem services.

In the case of action-based incentives, the optimisation programm writes:

$$\begin{aligned} \min_{X_j} \quad & X_j \cdot M_j \\ \text{s.t.} \quad & X_j \cdot (M_j - M_i) \geq -\pi_j + \pi_i \qquad \forall i \neq j \\ & X_j \geq 0 \end{aligned} \tag{7.2}$$

where X_j stands for the vector of incentives solving the optimisation programm, and M_j the vector of agricultural practices. π_j represents the profit associated to management option j. The constraints correspond to the ones described above, they ensure that the given management option is the most profitable, and that the incentives are non-negative.

We run this optimisation on the simulated data set, using package $Rglpk$ of R software.

The solutions of the optimisation represent sets of incentives that "achieve" a management option (i.e. make it the most profitable one among all 121 options), for the lowest policy budget. It is possible that the optimisation doesn't find any solution within the given constraints: there exist no set of incentives which can make this management option the most profitable, and thus the regulator needs to use other regulation instruments (e.g. command-and-control, another system of incentives, etc.) if he wants to make farmers adopt it. We first analyse theoretically which options can be "achieved" with each type of incentives, in particular by comparing them to efficient and cost-efficient bundles. Some options can be “achieved” by both types of incentives, but with different policy budgets. Therefore, we also compare budgets needed to achieve management options with action-based and result-based incentives (see Section 7.4).

7.3.2. Which bundles of ES can be achieved with each type of incentives ?

With result-based incentives Let us define pure profit as the profit excluding incentives an overall profit the sum of pure profit and incentives. Result-based incentives achieve bundles that maximise jointly ecosystem services and pure profit. The farmer maximises his overall profit. In the absence of these incentives, the overall profit simply equals the pure profit, and the farmer chooses the bundle maximising his pure profit. Result-based incentives add new dimensions in the maximisation of the overall profit, so that options maximising ecosystem services but with a lower pure profit can be achieved (i.e. become the most profitable).

This can be illustrated on Figure 7.1, in the case of two ecosystem services. One axis represents the pure profit, and the other two the level of ecosystem services. All

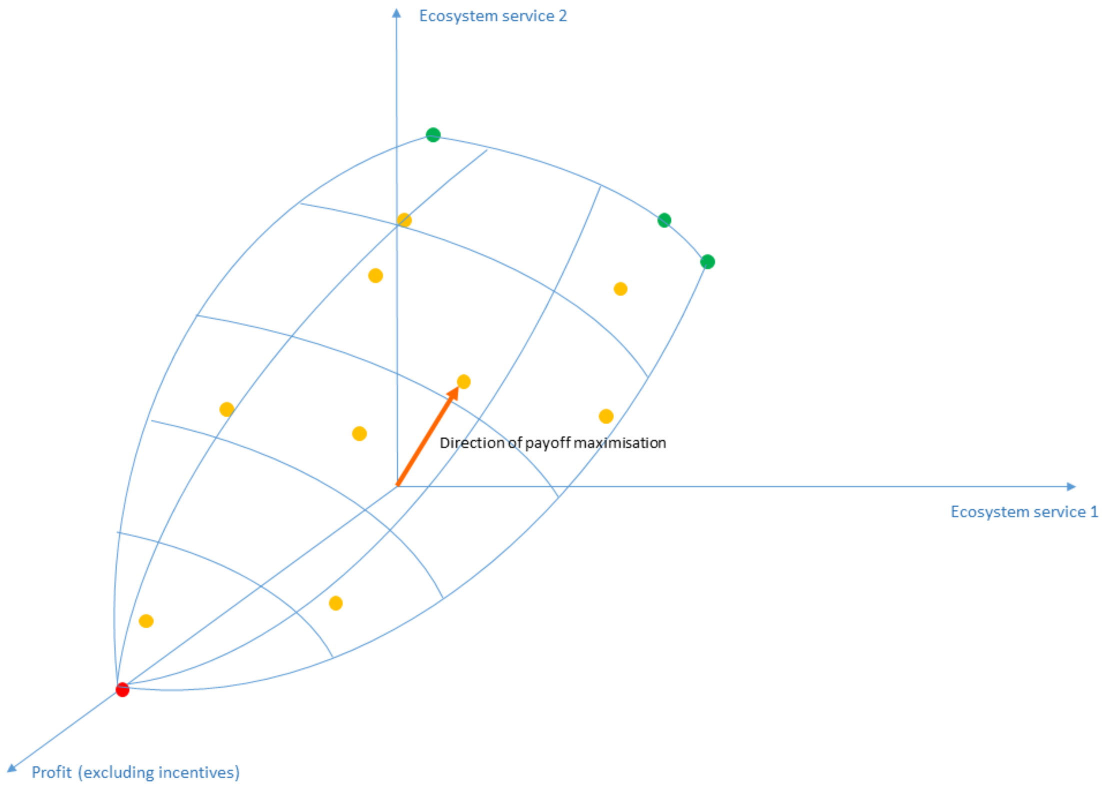

Figure 7.1.: *Schematic functioning of result-based incentives*
The envelope represents bundles maximising jointly pure profit and levels of ecosystem services. Farmers choose options that maximise their overall profit (including result-based incentives), located on the envelope (colored points). These bundles are cost-efficient. The orange arrow represents the strength of result-based incentives, which determines in which direction the farmers maximise their overall profit. Without incentives, the arrow aligns with the pure profit axis and farmers choose the bundle represented by the red point. On the contrary, strong incentives are needed to achieve the efficient bundles, maximising the provision of ES (green points).

possible options, characterised by their levels of ecosystem services and their pure profit, can be represented on the figure. They form a set, which envelope (blue "grid") is the subset of options that jointly maximises ecosystem services and pure profit (all the points represented on the figure lie on this outer surface). The farmer maximises the overall profit. Result-based incentives enter the overall profit and push the farmer to choose bundles that maximise ecosystem services. This is represented on the figure as the orange arrow, a vector coming from the origin. The relative strength of incentives (i.e. the subsidy associated to each ecosystem service) determines the direction of the vector along which the farmer maximises his overall profit. Without policy, the vector corresponds to the axis of pure profit, and thus the farmer chooses the option with is situated furthest along the corresponding axis (red point). With incentives, the direction of the vector comes closer to the axes of ecosystem services. Thus, by varying the incentives on ecosystem services, the policy maker can make farmers adopt every bundle that jointly maximises ecosystem services and pure profit (yellow points). At the extreme, if result-based incentives are infinitely high, management options maximising only ecosystem services (green points) can be achieved. For all options, the opportunity cost is calculated as the profit difference relative to a unique option, and the pure profit is a homothetic transformation of the opportunity cost (and conversely). Efficient bundles determined by directional DEA are invariant to homothetic transformations, and bundles maximising jointly pure profit and non-marketed ecosystem services are the same as bundles maximising non-marketed ecosystem services and minimising the opportunity cost.[32] Hence, result-based incentives achieve exactly cost-efficient bundles.

With action-based incentives To imagine which options can be achieved with action-based incentives, we can transpose the same representation with agricultural practices instead of ecosystem services. Options achieved by action-based incentives are those that maximise jointly pure profit and the "ecological" characteristics of agricultural practices, on which action-based incentives are calculated. All options which don't have a more "ecological" and more profitable equivalent can be achieved with action-based incentives. These options can be determined by an efficiency analysis on profit and ecological ranking of agricultural practices.

With action-based incentives, the levels of ecosystem services are not considered, and since the link between agricultural practices and ecosystem services is not linear, nothing guarantees that these options coincide with efficient or cost-efficient options. A management option may be rather extensive (and thus achievable with action-based incentives), but may provide few ecosystem services (and thus be inefficient or cost-inefficient).

7.4. Results

7.4.1. Bundles of ES achieved with each type of incentives

The linear optimisation programm cannot find a solution (set of incentives) for every option: for some options, the participation constraints cannot be respected. We call **achievable options** those for which there exist a set of incentives.

In accordance with the theoretical prediction, result-based incentives achieve exactly the cost-efficient bundles of ecosystem services determined by maximising non-marketed ES and minimising opportunity cost. Hence, between 2 options (in contexts with potential yield under 6.6t/ha) and 13 options (in contexts with high potential yield) out of 121 can be achieved with result-based incentives. In contexts with low potential yield, these options are grassland (the statu quo) and the least intensive cropland. In contexts with high potential yield, a wider variety of options can be achieved, but no options with fertiliser level higher than 2 (corresponding to 140kgN/ha) can be achieved, and apart from the least intensive cropland, all cropland options involve intermediate or high levels of fertilisers and pesticides.

[32] We also verified it by running a DEA on pure profit and non-marketed ES.

Options achieved with action-based incentives correspond to the ones being either more profitable or more extensive than the statu quo. They represent between 17 options (in contexts with potential yield under 6.6 t/ha) and 56 (with potential yield equal to 12 t/ha) out of 121. In contexts with low potential yield, intensive options are not very profitable, and only extensive options (mostly without fertiliser and pesticides) can be achieved by action-based incentives. All the options achievable in contexts with low potential yield remain achievable in the other contexts, but the higher the potential yield, the higher the number of intensive options which can be achieved. Again, due to the choice of including only incentives pushing towards more environment-friendly practices, no option with fertiliser levels higher than 2 can be achieved. As the adoption of these practices often decreases the profit, achieved options cover a large variety of options. Even if the link between agricultural practices and levels of ES is not linear, in our model all cost-efficient options can be achieved by action-based incentives.

With our dataset, more options can be achieved by action-based incentives than by result-based incentives, and all options achieved by result-based incentives are also achieved by action-based incentives.

7.4.2. Comparing the minimal policy budget with each type of incentives

Given the multiple incentives and the interactions among objectives of policies, nothing guarantees that the policy budget doesn't exceed the opportunity cost. The opportunity cost is the lower bound of the policy budget: it is the minimum amount of money that the policy maker must give to a farmer so that he chooses the given option. Consider a set of incentives compensating the opportunity cost of the targeted option; this set of incentives ensures that the overall profit of the targeted option is at least equal to the *pure* profit of the statu quo. However, this set of incentives applies to all options and possibly makes their *overall* profit change. The overall profit of the targeted option may not be higher than the overall profit of all other options (which is the participation constraint). Respecting the participation constraint may require a higher policy budget, determined by how incentives affect all options's profit. As a consequence, the opportunity cost is independant from the type of incentives, but the policy budget depends on the type of incentives.

By construction, the two types of incentives don't achieve the same options. However, some options can be achieved by the two types of incentives (all options achieved by result-based incentives). Among them, we compare the minimal policy budget needed, and three cases arise: the minimal budget is either higher with result-based incentives, higher with action-based incentives, or it is equal for both types of incentives. The last case happens when positivity constraints on incentives are binding, and hence the policy budget also equals the opportunity cost.

Table 7.1 shows results for agronomic context 8 (potential yield = 9.9 t/ha). 9 options out of 121 can be achieved with both types of incentives, and the budget with action-based incentives is smaller than the one with result-based incentives for 5 of them (options 6, 37, 43, 44 and 68), the opposite holds for one option (option 35), and the budgets are equal for 3 options (1, 47 and 67). Since option 47 is the statu quo, there is no need of incentives to achieve it, so that both policy budgets equal 0. On average, the policy budget with action-based incentives is also closer to the opportunity cost. In other contexts with a potential yield higher than 6.6 t/ha, where the statu quo is a rather intensive cropland, the results are similar: for most options, the policy budget and the gap with the opportunity cost are smaller with action-based incentives.

	opportunity cost (euro)	**minimal budget with result-based incentives** (euro)	**minimal budget with action-based incentives** (euro)
option 1 (grassland)	567	567	567
option 6	709	1953	893
option 35	86	86	91
option 37	197	254	201
option 43	8.9	60	17
option 44	188	535	192
option 47 (*statu quo*)	0	0	0
option 67	1.3	1.3	1.3
option 68	185	534	186

Table 7.1.: *Summary of management options achieved by both types of incentives (agronomic context 8, potential yield = 9.9t/ha)*

In contexts with low potential yield (under 6.6 t/ha), grassland (option 1) is the statu quo, and only two options can be achieved by both types of incentives: option 1 (grassland, the statu quo) and management option 6 (the least intensive cropland). In these contexts, the policy budget is equal with both types of incentives for option 1 (since it is the statu quo), and much greater with result-based incentives to achieve option 6. An example for context 4 (potential yield = 5.5 t/ha) is given in Table 7.2.

	opportunity cost (euro)	**minimal budget with result-based incentives** (euro)	**minimal budget with action-based incentives** (euro)
option 1 (grassland, statu quo)	0	0	0
option 6 (least intensive cropland)	459	4568	459

Table 7.2.: *Summary of management options achieved by both types of incentives (agronomic context 4, potential yield = 5.5t/ha)*

For options achieved only by action-based incentives, the policy budget is also close the opportunity cost, whatever the agronomic context.

7.4.3. Cost-efficiency and budget-efficiency

Cost-efficient options are a way to determine which options maximise ecosystem services for a given budget, but ignoring participation constraints. When accounting for these constraints, the real budget needed to provide ES is not necessary equal to the opportunity cost.

Compensating farmers at the opportunity cost constitutes a **first-best policy**. In contract theory, first-best refers to a solution maximising the principal's objective function (here minimising the policy budget) subject to all constraints except participation constraints (the constraint that the profit must exceed the other options' profit). On the contrary, **second-best** refers to a solution which maximises the principal's objective subject to all constraints, including participation constraints. This distinction between opportunity cost-efficiency (first-best) and policy budget-efficiency (second-best) is important (Drechsler, 2017).

In our case, the policy budget is often higher than the opportunity cost. As a result, some options with a low opportunity cost may require a high amount of subsidies to be adopted by a farmer, and thus some cost-efficient options (considering the opportunity cost) may not be budget-efficient (considering policy budget instead of opportunity cost). In other terms, second-best options may differ from first-best ones, and rather than considering options minimising the opportunity cost, a policy maker should indeed look at options minimising the policy budget, which can be seen as second-best options. What does it change to consider second-best options ? What can we say about the comparison of both types of incentives ?

To answer these questions, we identify budget-efficient options (maximising non-marketed ecosystem services and minimising the policy budget). For each option, we keep the lowest policy budget (either with action-based or result-based incentives), and run a DEA maximising ecosystem services for a given policy budget. We do the analysis over every option which can be achieved with any type of incentive, i.e. we include options which can be achieved only by action-based incentives. This analysis shows that second-best (budget-efficient) options are almost the same as the first-best (cost efficient) options. Only in contexts 9 and 10 (potential yield equal to 11 and 12 t/ha, resp.) are some differences [33]. These differences are marginal, and it is striking to see that the remaining results exactly coincide. Similarly to cost-efficient options, budget-efficient options are not very numerous. In contexts with low potential yield (under 6.6 t/ha), only two management options are budget-efficient: grassland and the least intensive cropland. The higher the potential yield, the higher the number of cost- and budget-efficient options (up to 12 from a total of 121 options). If cost- and budget-efficient options are almost identical, it is because the minimal policy budget is rather close to the opportunity cost, so that replacing the opportunity cost by the policy budget in the cost-efficiency analysis doesn't modify much the results. Indeed, even if the policy budget with result-based incentives is often much higher than the opportunity cost, all options achieved by result-based incentives are also achieved by action-based incentives, and often with a much lower policy budget. Figure 7.2 shows a comparison of policy budgets and opportunity cost for agronomic context 8 (9.9 t/ha).

7.4.4. Summary: virtues of each type of incentives

Result-based incentives achieve exactly the options minimising the opportunity cost and maximising non-marketed ecosystem services: they select cost-efficient, first-best op-

[33] Option 43 is cost-efficient in the first-best, and not cost-efficient in the second-best definition. On the contrary, in context 9, option 69 is not cost-efficient in the first-best, but is cost-efficient in the second-best definition.

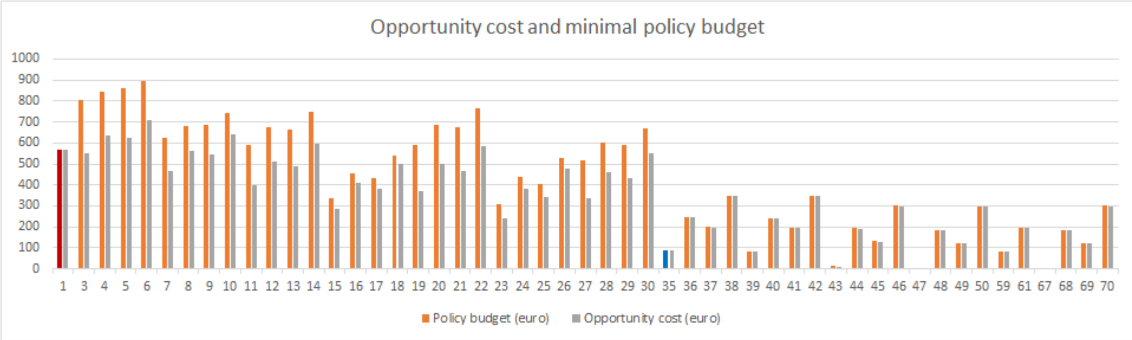

Figure 7.2.: *Comparison of policy budget (coloured bars) and opportunity cost (grey bars). (context 8, potential yield equal to 9.9 t/ha)*
The x-axis corresponds to achievable management options. Among coloured bars, orange bars show options for which the smallest policy budget is achieved with action-based incentives, blue bars with result-based incentives. Red bars show options having similar policy budget for both types of incentives.

tions. Considering the lowest policy budget among both types of incentives, we find that result-based incentives also achieve almost exactly the options minimising the policy budget and non-marketed ecosystem services: they also select second-best options. However, policy budgets with result-based incentives are often much higher than those with action-based incentives (action-based incentives are very often the ones with the lowest budget among both types of incentives). As a consequence, implementing result-based incentives make farmers choose options that are cost-efficient and budget-efficient, but in most cases, action-based incentives achieve these options for a lower policy budget. Increasing the provision of non-marketed ES in a budget-efficient way would require two steps: setting result-based incentives to choose a budget-efficient option, and paying the farmers with action-based incentives to minimise the policy budget.

7.5. Interactions among ecosystem services and the excess budget

We showed that policy budget is not necessary equal to the opportunity cost, it can be much higher. We refer to this difference as the **excess budget**, i.e. the policy budget that comes in excess of the compensation of the opportunity cost, in order to ensure participation of farmers.

The comparison of the two types of incentives reveals that this excess budget is often higher with result-based incentives than with action-based ones. However, the analysis doesn't inform about where this difference comes from. Some studies show that result-based incentives can contradict each other in terms of environmental results, and assume that this may lead to inappropriate calibration of incentives Bryan and Crossman (2013); Huber et al. (2017). Hence, we explore the drivers of this excess budget and in particular whether interactions among incentives impact it.

The policy budget is determined by the participation constraints. In particular, the policy budget allways equals the highest value of the participation constraint matrix. Let us remind that the policy budget comes from minimising the sum of incentives under the constraint that this set of incentives makes **every other bundle** less profitable. For result-based incentives (resp. action-based), the matrix of participation constraints measures the additionnality in terms of ES (resp. in terms of ecological characteristics of agricultural practices) of every bundle, relative to the bundle which policy budget is calculated. The policy budget is equal to the additionnal ecosystem services (resp. ecological characteristics of agricultural practices) multiplied by the set of incentives, and it must be greater than the difference in pure profit for every other bundle, in order to comply with participation constraints.

Choosing the levels of incentives can be rephrased in two steps: first choosing the relative weights of the different incentives (relative levels) and second their absolute level. Choosing the relative weights of incentives aims at changing the ranking of bundles, and making the one in question the most profitable. It is equivalent to change the relative price figured in Chapter 3, or to changing the direction in which the overall profit is maximised. The participation constraints first impose that there exist relative weights of incentives that make the bundle in question the most profitable, which is only possible if the bundle in question is cost-efficient (i.e. maximises either ecosystem services or environmental characteristics of agricultural practices, and minimises the cost). However, participation constraints also impose that the policy budget covers the difference existing in pure profit. This is achieved by playing with the absolute level of incentives, which was not considered on the Figure in theoretical Chapter 3. The largest profit difference is with the statu quo, and this imposes the policy budget to cover at least the opportunity cost (profit difference with the statu quo). The two steps (finding the relative levels and the absolute level) are not independant from each other. The relative weights determine the importance of the level of each ecosystem service (resp. agricultural practice) in meeting the constraint of compensating the difference in pure profit, and thus impacts the absolute level of incentives needed to meet the constraint.

The type of interactions among ecosystem services impacts the possibility of changing the ranking of options by changing the relative weights of result-based incentives. In particular, synergies among ecosystem services (the existence of bundles of outputs with close levels of ecosystem services) can require large changes in relative weights of result-based incentives in order to change the ranking of bundles. This illustrates the theoretical effect identified in Chapter 3: in the case of synergy, many relative prices can correspond to the same bundle, and a drastic change in relative price may be needed to achieve another one. In this sense, synergies among ecosystem services impose constraints on the relative weights of result-based incentives, and indirectly make the participation constraints more difficult to meet with result-based incentives. This can lead to higher policy budgets: in order to meet the constraint with a more limited range of relative weights, it is necessary to play on the absolute level of incentives.

Less interactions occur among agricultural practices: except for grassland, we allow all combinations of agricultural practices, and the modality of one agricultural practice is independant of the modalities of the other practices. As a consequence, the policy budget can be minimised by playing on all the range of relative weights of action-based incentives, without playing much on the absolute level of incentives and increasing the policy budget.

7.6. Discussion and conclusion

In this chapter, we study incentives for the regulation of multiple interacting ecosystem services. More precisely, we compare two types of economic incentives: action-based and result-based, and explore whether the multiple and interacting ecosystem services impair the theoretical efficiency of result-based incentives.

We extend similar studies by considering more than 2 ecosystem services (Gibbons et al., 2011), and explicitely accounting for interactions among them and their drivers: for example, (Bryan and Crossman, 2013; Ruijs et al., 2013) both consider more than 2 ecosystem services, but their approaches don't consider the compatibility or incompatibility among the drivers of ecosystem services (in their case, the land use).

We show that result-based incentives better target cost-efficient and budget-efficient bundles of ES, but that their policy budget is often much higher than action-based incentives. We show that interactions among ES, in particular synergies may explain this.

Our results can be related to current debate over the use of result-based incentives. While some studies clearly advocate for their use in agri-environmental policies as a way to solve efficiency problems, as well as to leave more flexibility to farmers, we underline that result-based incentives may require a higher policy budget than action-best incentives. More generally, their implementation raises other problems, in particular the measurement the ecosystem services provided, especially those which are difficult to observe or depend on the action of several farmers. Besides, these incentives would make farmers bear more uncertainty affecting the provision of ecosystem services (e.g. due to environmental conditions).

Result-based approaches enable to achieve cost- and budget-efficient bundles of ecosystem services, and hybrid solutions could be a solution to keep this property while avoiding implementation problems and unwanted interactions among incentives. Reed et al. (2014) mention approaches similar to environmental auctions, where agri-environmental projects at the farm level are selected based on predicted ecosystem services provided and the amount of compensation, but where the payment is made on the basis of the management costs, and not on the ecosystem services actually provided.

Our results are subject to some limitations. In particular, they may be driven by the fact that we explicitely considered that actions (agricultural practices) are subject to almost no interactions: the choice of one agricultural practice doesn't condition the choice of other practices. On the contrary, we insisted on interactions among ecosystem services. In reality, agricultural practices are probably more interdependant than our model considers. This may underestimate the interactions among action-based incentives as presented in the last paragraph of results. Our work generally highlights that individual incentives must be thought in coherence with each other.

Moreover, we only compared incentives on the basis of one homogeneous field. The results probably change in the presence of heterogeneity and asymetries of information: one of the main criticisms addressed to agrenvironmental measures concerning their lack of efficiency is due to the fact that homogeneous action-based incentives implicitely target areas where the opportunity cost of adopting extensive management options is low, i.e. where the potential yield is low. We already showed that our results are sensitive to the potential yield, so we can expect that doing the same analysis over a collection of heterogeneous fields would yield different results, and result-based incentives may indeed have interesting characteristics to overcome this issue. This is the focus of the next chapter.

Part IV.

What land heterogeneity changes

8. Heterogeneous areas

8.1. Introduction

In the previous chapters, we explored cost-efficient management options to provide non-marketed ecosystem services, and the economic incentives to implement these cost-efficient options. We did this analysis while assuming that the whole area had homogeneous characteristics, but showed that the cost-efficient options depend much on the agronomic context considered, and in particular on the potential yield. In reality, not all area has the same agronomic characteristics, for example potential yield varies. Given the dependance of the cost-efficient options to potential yield, in this chapter, we explore the impacts of heterogeneity on the cost-efficient solutions to provide ecosystem services.

By considering heterogeneity, this analysis considers solutions composed of combinations of management options. In addition to the likely impact of heterogeneity, the properties of efficiency analysis lead us to think that the spatial scale bears in itself an impact on the cost-efficiency. Even in the absence of heterogeneity, not all combinations of cost-efficient options are cost-efficient, so that cost-efficiency at the landscape scale differs from cost-efficiency at the field scale. In this chapter, we therefore distinguish the impact of changing the "scale"[34] of analysis, and the impact of heterogeneity.

Heterogeneity also is an issue for the implementation of policies: incentives (and in general policy instruments) always have a certain degree of uniformity, which makes it difficult to cope with the heterogeneity in cost-efficient solutions. We compare action-based and result-based incentives in presence of heterogeneity, analyse the impact of heterogeneity on the policy budget.

To assess the impact of heterogeneity on the design and implementation of cost-efficient agri-environmental policies, we use the simulated data set. We first review in Section 8.2 the literature dealing with land heterogeneity and the provision of ecosystem services. Then, in Sections 8.3 and 8.4, we present how we introduce heterogeneity in our analysis and how we analyse cost-efficency. We detail the results of the impacts of heterogeneity in Section 8.5. Last, in Section 8.6, we study the implementation of cost-efficient management options with incentives on a heterogeneous agricultural areas.

[34] Our analysis considers space in an implicit way: upscaling the analysis is modelled by allowing different management options on the agricultural area.

8.2. Literature review

Designing policies includes two steps: first defining the effort to maximise environmental benefits, and then studying the means to implement the effort targeted. We follow these steps in the literature review. We first focus on how land heterogeneity changes cost-efficient management options to provide ecosystem services, and later, we present which economic incentives are suited to implement these cost-efficient options.

With heterogeneity comes generally an important issue identified by economists, **asymetry of information**: contrary to farmers, the regulator doesn't know the heterogeneous characteristics of each decision unit (farm/field), both in terms of environmental benefits and costs.This is a problem in order to define cost-efficient actions to take: if the regulator doesn't know the precise costs and environmental benefits, he can't determine which actions are cost-efficient. Asymetry of information also impacts the implementation of policies with incentives.

8.2.1. Impact of heterogeneity on cost-efficient allocation of efforts

Cost-efficiency analysis must account for land heterogeneity. Land heterogeneity concerns the two sides of cost-efficiency: the environmental benefits (i.e. the ES provided by a given action), and the costs of adopting this action for the farmer. For exemple, a reduction in fertiliser use is associated to a greater increase in water quality if it takes place in a field subject to heavy leaching or close to a water body. And this reduction is likely to be more costly in productive areas, where a reduction in fertiliser use causes a higher yield loss.

Heterogeneity in environmental benefits is a known issue by ecologists, and has motivated an extensive literature about conservation planning: how to select actions that maximise environmental benefits (e.g. which area to conserve). This is the perspective taken by many spatial multi-ES assessments which determine areas where ecosystem services levels are high, or where a large number of ecosystem services are provided (Chan et al., 2006; Naidoo et al., 2008). Like others, these studies focus only on the maximisation of these benefits (**efficiency analysis**), and either don't account for the variability in costs of conservation, or in a too simple manner. However, considering the costs of conservation through **cost-efficiency analyses** is crucial to maximise conservation benefits when the conservation budget is limited (Ferraro, 2003; Naidoo and Ricketts, 2006). Indeed, many authors have shown theoretically (Babcock et al., 1997) and more empirically (Wünscher et al., 2008) that ignorance of variability in costs and benefits both leads to loss of cost-efficiency in the selection of environmental actions (e.g. areas to conserve). This seems rather logical that when both benefits and costs vary, cost-efficient solutions also vary.

Studying the heterogeneity of costs is present in the economic literature since a long time (Wu and Boggess, 1999), and has been integrated in the interdisciplinary literature. This is namely the objective of the analysis of Ruijs et al. (2013) or Sauer and Wossink (2013), which model the trade-off among agricultural production and non-marketed ES to assess where this trade-off is the least severe, in order to determine where the cost incurred by yield loss is the smallest, even if this cost doesn't capture the total opportunity cost incurred by conservation or management change (and in particular the cost of new management practices).

In fine, what comes out the studies on heterogeneity in costs and benefits is that targeting actions according to their costs and benefits is necessary to achieve cost-effective conservation and maximise environmental benefits for a given budget (Duke et al., 2013). Another conclusion is that the correlation between costs and benefits, and the relative variability of costs compared to benefits determines the interest of a cost-efficiency approach compared to a simple efficiency approach. The more benefits and costs are positively correlated, and the higher the variability of costs compared to the variability of benefits, the greater the interest of a cost-efficiency approach compared to a selection only through efficiency analysis (Babcock et al., 1997; Ferraro, 2003).

8.2.2. Policy instruments in presence of heterogeneity

After having determined which is the desired action to take in order to achieve environmental benefits (accross space, time, etc.), the next step is to study how to implement these actions with policy instruments. Here we focus on economic incentives (e.g. agroenvironmental payment), among all possible policy instruments.

Here, asymetry of information plays an important role, and in particular **adverse selection**: in presence of asymetry of information, uniform prices or incentives tend to select worst-quality goods or actions. For example, a subsidy for the implementation of grassland will be more profitable for farmers having a lower cost (because their soil is better adapted, because the value of the associated yield loss is lower etc.), and thus select them in priority. In the case of agroenvironmental schemes, yield loss represents an important part of the cost, and thus such grassland subsidies de facto select fields where yield loss is lower, i.e. extensive and less productive areas, which may not be the ones providing the most environmental benefits (Kuhfuss, 2013). Hence, the literature on heterogeneity in costs and asymetry of information underlines that these issues make uniform payments or incentives inefficient: the payment overcompensates the farmers that have a low cost, and fails at making farmers adopt the action if their cost is higher than the payment (Fraser, 2009; Armsworth et al., 2012). Fraser (2009) summarises the issues caused by heterogeneity when environmental benefits are supplied through agroenvironmental schemes, and shows that such a scheme leads to a misallocation of the budget, compared to a optimum where conservation effort is such that the marginal cost is equal to the marginal benefit.

Note that in reality, the issues created by heterogeneity hold even in absence of asymetry of information. Even in the ideal case where the policy maker has a precise idea of the solution that is adapted to each value of the source of variability, that is of the cost-efficient actions to adopt, the regulator cannot propose a different scheme to each farmer because of equity reasons or issues like prohibitive transaction costs (Armsworth et al., 2012).

Two types of solutions to asymetry of information and heterogeneity exist: reduce the asymetry of information, or design incentives that are robust to heterogeneity. The first type of solutions gathers proxies to estimate heterogeneous costs and benefits, or mechanisms to make farmers reveal their true costs and benefits, so that the regulator can better target the payments (Ferraro, 2008). In a similar approach, (Canton et al., 2009) proposes delegation (i.e. the calibration of incentives at a more local level) to reduce the range of heterogeneous values.

The second way to deal with heterogeneity and asymetry of information is to design incentives that enable to get closer to cost-efficient policies, even in presence of heterogeneity and asymetry of information. For example, self-screening contracts (see e.g. (Ferraro, 2008)) are contracts involving several levels of incentives and actions, designed so as to discriminate among low-cost farmers and high-cost farmers and propose them cost-efficient incentives. When the heterogeneity comes from spatial spillovers, agglomeration bonuses are also a way to target actions that bring high environmental benefits (Parkhurst et al., 2002), without the need for the policy maker to map their structure. Else, result-based incentives are supposed to better cope with heterogeneity (Antle et al., 2003; Hasund, 2013; Bureau, 2017). They equalize the marginal unitary cost of providing ES throughout heterogeneous situations, and thus are theoretically able to cope with heterogeneity both in costs and environmental benefits and select cost-efficient options despite unknown heterogeneity. Gibbons et al. (2011) explore this theoretical result. THey use a modelling approach to compare action-based and result-based incentives to provide biodiversity, and mention heterogeneity as a factor that makes result-based incentives interesting. Indeed, many empirical reviews about agroenvironmental schemes mention the hypothesis that result-based incentives may propose a solution to asymetry of information in a heterogeneous context (Schwarz et al., 2008), and result-based incentives have been experimented in agroenvironmental schemes in the last years (Musters et al., 2001).

Although papers cited above seem to have settled the advantage of result-based incentives, in particular in presence of heterogeneity, our approach is different and may provide new elements in this debate. First, the studies cited above consider only one environmental benefit, either abstract and theoretical (Schwarz et al., 2008; Hasund, 2013), or more specific like biodiversity in Gibbons et al. (2011) or carbon in Antle et al. (2003). Second, many studies advocating for result-based incentives in presence of heterogeneity are based on very theoretical modelling (Hasund, 2013). Whether they rely on very theoretical or on more precise modellling (as in Gibbons et al. (2011)), simplifications could limit their conclusions. They consider a uni-dimensional way to improve agricultural management (often called "effort", or even hidden behind the increase in environmental goods). This is coherent with the unique environmental good: even if effort can cover very diverse actions, they can all be ranked according to the unique environmental good. Another simplification is the use of monotonous relations between effort, cost and environmental benefits (the higher the effort, the higher the costs and the higher the environmental benefits).

Our approach differs because it is based on a model which is closer to agroecological modelling. First, it explicitely considers the impact of muliple ecosystem services and their interactions. Second, it considers agricultural practices and their impacts on the multiple ecosystem services, so that improvement of agricultural management may correspond diverse actions (effort is multidimensional). Last, our modelling approach doesn't specify a priori any function for the cost or ecosystem service production. Instead, these relations emerge from the simulated impact of diverse agricultural practices on ecosystem services and profit.

8.3. Heterogeneity in our framework

Heterogeneity is introduced by considering areas divided in two equal parts corresponding to different agronomic contexts. We use the term **(agricultural) area** to refer to the exogenous characteristics (the potential yield and initial stock of soil organic matter) of an heterogeneous piece of land. We use the term **landscape** to refer to the combination of management options adopted by the farmers on a given (agricultural) area. We don't model space explicitely, so that the difference between field scale and landscape scale in our framework is the possibility to have heterogeneity, and to adopt different management options. A field is an homogeneous area in its agronomic characteristics and management. A landscape starts when the agronomic characteristics of the corresponding area can vary.

Cost-efficiency on a heterogeneous area is likely to differ from cost-efficiency at the field scale, and combining options which are cost-efficient at the field scale may not result in cost-efficient landscapes. This difference in terms of cost-efficiency between field scale and landscape scale results from two effects: one is related to the landscape scale itself, and the other to the existence of heterogeneity. First, landscape scale itself changes something when using cost-efficiency analysis: even in a homogeneous area, combinations of management options that are cost-efficient at the field scale may not be cost-efficient at the landscape scale anymore. For example, if the PPF is concave (outward-bending), all linear combinations of (cost-)efficient bundles will not be (cost-)efficient. On a homogeneous area, a combination of different management options can be represented graphically by the segments linking up two different management options. This is ilustrated on Figure 8.1: options 1, 2 and 3 are all efficient, but the linear combinations of options 1 and 3 are not efficient since they are dominated by option 2. Again, this is related to the shape of the PPF, and it has nothing to do with heterogeneity in itself.

Second, heterogeneity itself is also likely to modify which combinations of options are cost-efficient. In our model, the provision of ecosystem services varies slightly with the stock of soil organic matter. More importantly, the profit varies much with potential yield, and this makes the statu quo vary too. So does the opportunity cost, and ES if measured relative to the statu quo. The opportunity cost doesn't depend linearly from potential yield, and thus the opportunity cost over a heterogeneous area will be different from the opportunity cost over a homogeneous area with identical average yield. We represent the bundles of ES and opportunity cost for different options and different agronomic contexts on Figure 8.2. We observe from this figure that ecosystem services and the opportunity cost are not clearly correlated, and that the variability of costs is higher than the variability of ecosystem services. This is therefore especially important to consider the cost in the selection of management options over heterogeneous agronomic conditions (Babcock et al., 1997; Duke et al., 2013).

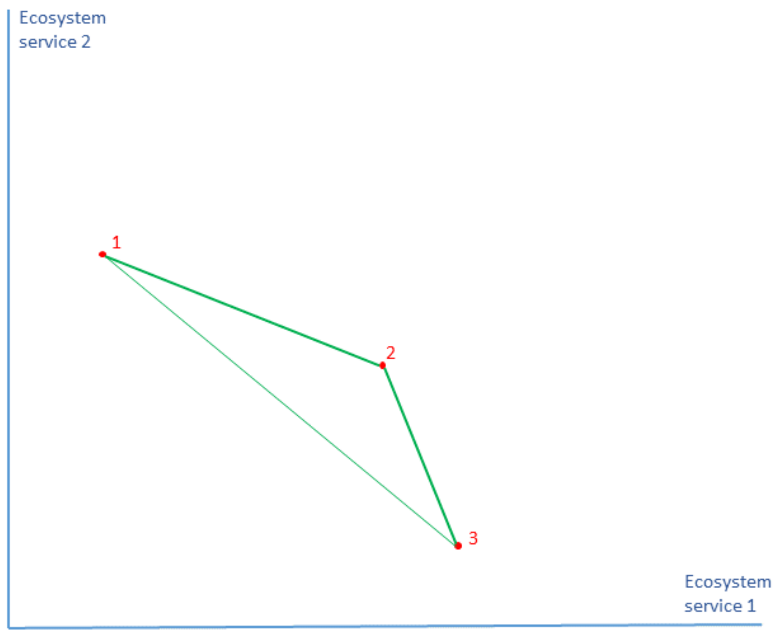

Figure 8.1.: *Effects of upscaling the efficiency analysis: not all linear combinations of efficient bundles are efficient*

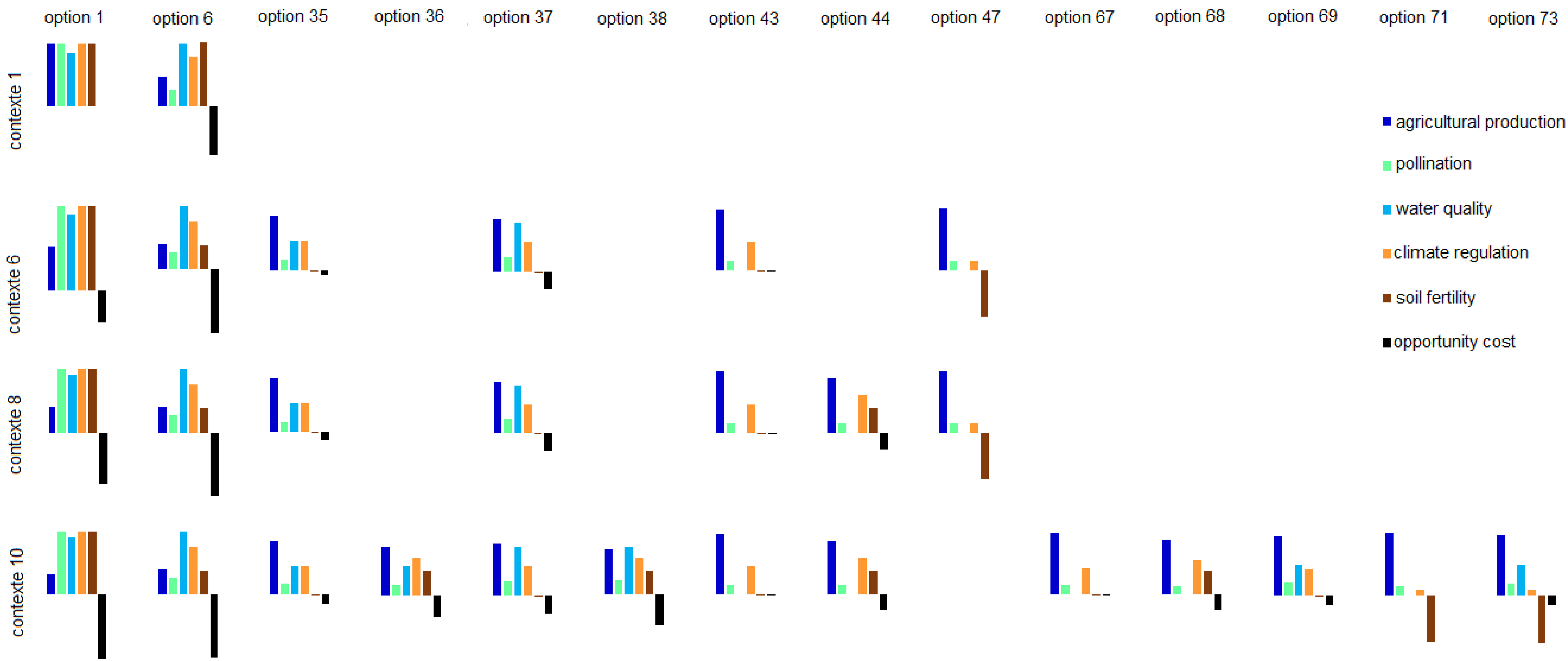

Figure 8.2.: *Variations of ES levels and opportunity cost among contexts and management options*
Levels of ES and opportunity cost are represented on a scale from 0 to 1.
This figure shows that the variability in our simulated data set comes mainly from the opportunity cost, and that the ecosystem services are not clearly correlated to the cost.

Cost-inefficiency created by either landscape scale or heterogeneity is a problem: it is likely that the design of cost-efficient management at the landscape scale starts from cost-efficient options identified at the field scale, given the very high number of possible combinations of management options at the landscape scale to be considered in a cost-efficiency analysis at the landscape scale[35].

We highlight that relying on cost-efficient options at the field scale may not be enough to design cost-efficient landscapes, and that heterogeneous areas create trouble for identifying cost-efficient management.

8.4. Cost-efficiency analysis on heterogeneous areas

8.4.1. First step: Landscape scale data

We consider agricultural areas composed of pairs of fields corresponding to different agronomic contexts. We choose pairs with various mean and variance in agronomic conditions, summarised in Table 8.1.

	Pot. yield		**SOM**		**average**	**average**	**yield**
	Field 1	**Field 2**	**Field 1**	**Field 2**	**SOM**	**pot. yield**	**difference**
area A	4	12	111.8	75.7	93.75	8	8
area B	4	8	75.7	75.7	93.75	6	4
area C	6	10	75.7	75.7	75.7	8	4
area D	8	12	75.7	75.7	75.7	10	4
area E	4	6	111.8	75.7	93.75	5	2
area F	5	7	75.7	75.7	75.7	6	2
area G	7	9	75.7	75.7	75.7	8	2
area H	9	11	75.7	75.7	75.7	10	2
area I	10	12	75.7	75.7	75.7	11	2
area J	5.5	6.5	75.7	75.7	75.7	6	1
area K	7.5	8.5	75.7	75.7	75.7	8	1
area L	9.5	10.5	75.7	75.7	75.7	10	1

Table 8.1.: *Heterogeneous areas considered. Potential yield and soil organic matter in t/ha.*

To get the simulated data on these areas, we first determine the landscapes (combinations of management options). 121 management options are available on one field, hence a large number of combinations (landscapes) are possible on a two-field area ($121^2 = 14641$). To restrict the number of landscapes and focus on cost-efficient ones, we keep only landscapes made of combinations of cost-efficient options at the field scale. Depending on the agronomic context, between 2 and 13 options are cost-efficient on a field, which gives between 4 and 117 landscapes made of cost-efficient options. We calculate the data about ecosystem services and opportunity cost at landscape level by

[35] In order to run a robust (cost-)efficiency analysis, all options must namely be included, which creates a very high number of alternatives over a landscape.

summing up data at the field level calculated for the agronomic contexts corresponding to the area. We also calculate a data set with similar landscapes on homogeneous areas with identical average agronomic characteristics. Comparing the two data sets with the same average conditions enables to isolate the two effects mentioned above: the effect of landscape scale and the effect of heterogeneity itself.

8.4.2. Second step: New cost-efficiency analyses

With these new data sets, we run again cost-efficiency analyses. In each heterogeneous area, we run a directional Data Envelopment Analysis maximising ecosystem services and minimising the opportunity cost over all landscapes made of cost-efficient options at the field scale. We do the same with the data sets of the corresponding homogeneous areas.

For each area, this cost-efficiency analysis is run similarly to the one in Chapter 6, with the sum of ecosystem services levels over the two fields, and the sum of opportunity cost over the two fields. The principle is to find the greatest increase in non-marketed ecosystem services for a given cost, among linear combinations of landscapes.

$$\begin{aligned} \max_{\mu_i} \quad & \beta_j \\ \text{s.t.} \quad & \sum_{i=1}^{N} \mu_i (Y_i^1 + Y_i^2) \geq (1+\beta_j)(Y_j^1 + Y_j^2) \\ & \sum_{i=1}^{N} \mu_i (C_i^1 + C_i^2) \leq (C_j^1 + C_j^2) \\ & \sum_{i=1}^{N} \mu_i = 1 \end{aligned} \tag{8.1}$$

Here, the index i or j stands for a two-fields-landscape. β_j represents the inefficiency score associated to landscape j and is expressed as a percentage by which all ES could be increased jointly (for the efficient landscapes, it equals 0). Y_i^1 and Y_i^2 stand respectively for the bundle of ES provided by option 1 and 2 of landscape i, C_i^1 and C_i^2 for the opportunity costs. μ_i the weight of landscape i in the efficient benchmark, i.e. the land use share of landscape i in the efficient benchmark.

8.4.3. Third step: characterising the changes

After having run the cost-efficiency analyses, we compare the results for heterogeneous and homogeneous areas. We classify each landscape into one of the following case concerning cost-efficiency:

- Case 1: the landscape is cost-efficient in both areas.
- Case 2: the landscape is cost-inefficient in both areas
- Case 3: the landscape is cost-efficient in the homogeneous area, but inefficient in the heterogeneous one

- Case 4: the landscape is cost-inefficient in the homogeneous area, but efficient in the heterogeneous one

These cases inform about the respective effects of landscape scale and heterogeneity. Analysing the proportions of landscapes corresponding to these cases informs about the two effects.

To explain what could drive the case each landscape belongs to, we classify landscapes in 4 types:

- uniform landscape: the two management options on the area are identical
- almost uniform landscape: the two management options are not identical but close (same levels of fertiliser and pesticides)
- adequate landscape: the two management options are different, but distributed in adequation with potential yield: the field with the lowest potential yield is cultivated with the least intensive option
- reverse landscape: the two management options are different, but distributed in a reverse way: the field with the lowest potential yield is cultivated with the most intensive option

By intersecting these categories with the cases detailed above, we carry an independency analysis, more precisely a Chi-square test. This statistical test enables to determine whether the appartenance to one category is correlated to the appartenance to another category. For example, are uniform landscapes more likely to belong to case 1 (landscape cost-efficient both in heterogeneous and homogeneous area) ? We detail this statistical test in Frame 8.4.3.

The Chi-square independance test

To calculate the chi-square test statistic, we compare real numbers of combinations belonging to each categories with theoretical numbers obtained if the probability to belong to different categories were independant of each other. The higher the gaps between real and theoretical numbers, the higher the statistic and the less likely is independance. This statistic follows a chi-square law, and is to compare with probability density.

Formally, this test evaluates whether hypothesis H_0 can be rejected

H_0: the two variables (landscape type and case) are independant

H_1: the two variables are not independant

Under hypothesis H_0, the following statistic follows a Chi-square distribution with $(n-1)(m-1)$ degrees of freedom:

$$\chi = \sum_i \frac{(O_i - T_i)^2}{T_i} \tag{8.2}$$

where i refers to each combination of the two variables, O_i the observed number of individuals corresponding to combination i, and T_i the theoretical number under hypothesis of independance. n and m refer to the number of modality of each variable.
If the statistic χ is greater than the probability density of the Chi-square law, then the hypothesis H_0 is rejected, and the two variables are not independant.

We also analyse how the average yield and the yield difference between the two fields of an area impact the results, and the gap between the two fields in terms of opportunity cost and profit.

8.5. Results: how heterogeneity changes cost-efficient solutions

The following table summarises the results: for each area, by comparing the results of cost-efficiency analyses in heterogeneous and homogeneous areas, we determine which case landscapes belong to.

	Heterogeneous areas											
	A	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**
average pot.yield (t/ha)	8	6	8	10	5	6	8	10	11	6	8	10
pot.yield difference (t/ha)	8	4	4	4	2	2	2	2	2	1	1	1
number of landscapes	26	12	16	78	4	10	35	84	117	4	35	90
case 1 (%)	26.9	27.3	44.4	20.5	75	30	31.4	22.6	23.9	100	37.1	28.9
case 2 (%)	42.3	36.4	38.9	48.7	0	30	31.4	52.4	49.6	0	31.4	46.7
case 3 (%)	0	0	0	16.7	25	10	28.6	17.9	20.5	0	22.9	23.3
case 4 (%)	30.8	36.4	16.7	14.1	0	30	8.6	7.1	6.0	0	8.6	1.1

Table 8.2.: *Comparison of cost-efficient landscapes in heterogeneous and homogeneous areas*
Case 1: the landscape is cost-efficient in both areas. Neither landscape scale nor heterogeneity is an issue.
Case 2: the landscape is cost-inefficient in both areas. Landscape scale itself is an issue (and we can't know if heterogeneity also is).
Case 3: the landscape is cost-efficient in the homogeneous area, but inefficient in the heterogeneous one. Heterogeneity alone is an issue.
Case 4: the landscape is cost-inefficient in the homogeneous area, but efficient in the heterogeneous one. Landscape scale is an issue, but heterogeneity solves this issue.
(Percentage values are rounded, and may not add up to 1)

We measure cost through opportunity cost so that the statu quo drives the results. Hence Results differ among areas, in particular those of areas E and J, which we interpret separately[36].

[36]Areas E and J are composed of fields with both low potential yield, on which only 2 options are

8.5.1. Impact of landscape scale

Cost-efficiency analysis in the homogeneous area captures the effect of landscape scale itself, while cost-efficiency analysis on the heterogeneous area captures the effects of both heterogeneity and landscape scale. The effects of landscape scale are thus identified by comparing cost-efficiency at the field scale and cost-efficiency on a homogeneous area. All landscapes are composed by options that are cost-efficient at the field scale, so that landscapes that are not cost-efficient on a homogeneous area (cases 2 and 4) denote an effect of landscape scale.

In all areas except E and J, as summarised in Table 8.2, cases 2 and 4 represent often more than 50% of the landscapes (between 40 and 73.1%). The proportion of landscapes concerned by landscape scale issue is especially high in areas A and B, which both have high yield difference between the two fields. It is especially low on areas G and K, which both have an medium average potential yield (8t/ha), and rather close potential yield in the two fields (difference smaller or equal to 2t/ha). On areas E and J, this proportion is equal to 0, since among the 4 landscapes considered in these areas, all of them are cost-efficient in homogeneous areas.

8.5.2. Impact of heterogeneity

The effects of heterogeneity can be identified by comparing the cost-efficiency on homogeneous and heterogeneous areas. The difference can go in two directions: either the landscape is cost-efficient in the homogeneous area, and heterogeneity makes this landscape cost-inefficient (case 3), or the landscape is cost-inefficient in the homogeneous area, and heterogeneity makes it cost-efficient (case 4).

Cases 3 and 4 gather a minority of landscapes, between 16.7 and 37.1%. More precisely, case 3, where heterogeneity itself leads to cost-inefficiency, represents between 0 and 28.6% of landscapes, higher in areas G, K and L with rather small yield difference between the two fields (less or equal to 2 t/h), and rather high average potential yield (higher than 8 t/ha). On the contrary, no landscape corresponds to case 3 in areas A, B, C and J, which all have in common a field with low potential yield (less or equal to 6 t/ha). The high yield difference between fields doesn't seem to exacerbate issues caused by heterogeneity.

Opposite case 4, where heterogeneity solves cost-inefficiency at the landscape scale, concerns between 0 and 30% of landscapes. The higher proportions of this case occur in areas A, B and F, where average potential yield is rather low (less or equal to 6 t/ha), and yield difference not too low (higher or equal to 2 t/ha), so that average potential yield is distinct from individual potential yield of each field. This high proportion in contexts with low average potential yield is probably explained by the fact that landscapes included in the cost-efficiency analysis are selected on the basis of the cost-efficiency on the individual fields. In each of landscapes A, B and F, one of the field has a potential yield higher than 6.6 t/ha, which means that more than the two least intensive options are

cost-efficient (grassland and the most extensive cropland). As a result, only 4 particular landscapes are included in the cost-efficiency analysis, and the results differ from results on other areas. It is as if the list of landscapes in these areas were truncated compared to areas with higher average potential yield: all landscapes considered in areas with low average potential yield are considered in areas with higher average potential yield, but not vice-versa.

cost-efficient. However, the average potential yield of the landscape is low, and at this potential yield, not many options are cost-efficient. As a result, landscape scale creates cost-inefficiency, but heterogeneity solves it.

While it is clear in case 1 that heterogeneity doesn't create cost-inefficiencies, it is impossible to know whether heterogeneity is also an issue in case 2 (landscapes which are cost-inefficient in both homegenous and heterogeneous areas): these landcapes are already cost-inefficient because of landscape scale, and heterogeneity may also participate to the problem, but its effect is impossible to isolate.

Last, between 20.5 and 44.4% of landscapes are cost-efficient in both homogeneous and heterogeneous areas (Case 1), i.e. neither landscape scale nor heterogeneity change cost-efficiency.

8.5.3. Determinants of the impact of landscape scale and heterogeneity

Landscapes have been divided into 4 landscape types according to the management options: uniform landscapes, almost uniform landscapes, adequate landscapes, and reverse landscapes. We intersect both categorisations, and run of Chi-square independance test. The results show that these categories are not independant for areas H, I, K and L, i.e. areas with potential yield greater or equal to 8 t/ha and low heterogeneity. Here, we interpret these relations in areas for these contexts.

First of all, some links between landscape type and case are trivial: uniform landscapes are very unlikely associated to cases 2 or 4, i.e. to be cost-inefficient in the homogeneous area. Indeed, for a uniform landscape to be included in the analysis, it means that the given management option is cost-efficient on each of the two fields. Since cost-efficient options vary almost monotonously with potential yield, if a management option is cost-efficient in two fields with different potential yield, then it is very likely to be efficient in a homogeneous area with a potential yield included inbetween, such as the mean potential yield, so that the uniform landscape is very likely to be cost-efficient on the homogeneous area. As a result, no uniform landscape is cost-inefficient in the homogeneous area, and the independence test shows that these landscapes are more often cost-efficient in both homogeneous and heterogeneous areas.

Concerning adequate landscapes, i.e. landscapes with different management options (different levels of fertilisers and/or pesticides), but which repartition is in adequation with the potential yield, the analysis shows that they are more often than predicted associated to cases 4, and less often that predicted associated to cases 3. This indicates that for adequate landscapes, heterogeneity is more likely to foster cost-efficiency.

On the contrary, reverse landscapes, where the two management options are different but organised in contrary to the ranking of potential yield, are more often associated to case 3 and less often to case 1. Hence, heterogeneity itself and landscape scale seem to be more often a problem when the intensity of agricultural management doesn't follow the potential yield.

Landscapes composed of different but similar management options are not associated to particular patterns of cost-efficiency.

8.5.4. Impact of average yield and difference in yield

The results detailed above depend on the area considered, on its average potential yield and on the difference between the two fields. Here we review how the results vary with the average yield and yield difference.

In our model, the statu quo is very important. It determines the opportunity cost of every other management option. When the statu quo is grassland, almost no increase in ecosystem services is possible, so that only two options are cost-efficient (grassland and the least-intensive cropland). In our analysis, we included only landscapes combining cost-efficient options, so that when the statu quo of one field is grassland, it restricts the number of landscapes included in the cost-efficiency analysis. Hence the effects of the average yield of the area and of the yield difference between the two fields are complex, because it mainly depends on whether grassland is the statu quo in one of the two fields.

Globally, the higher the average yield of an area, the higher the number of landscapes considered in the cost-efficient analysis, and the more landscapes with intensive options are included. The higher the yield, the stronger the link between types of landscape and cost-efficiency patterns.

The yield difference between the two fields doesn't change results in a systematic way.

8.5.5. Repartition of opportunity cost

The cost-efficiency analysis minimises the sum of opportunity costs (which amounts to maximises profit) over the landscape. But it doesn't impose any constraint on the individual opportunity costs or profits, and the opportunity cost of increasing the non-marketed ecosystem services may not be shared in an equitable way. Therefore, it is interesting to explore if certain types of landscapes are associated to a high difference in individual opportunity costs and profits. For a given area, the statu quo is fixed, the opportunity cost is a translation of the profit, and the difference between the two fields in terms of profit or opportunity cost is perfectly correlated. Though, the equality between the two fields is achieved at different levels, and the interpretation is different. The equality among opportunity costs imposes that both farmers support the same cost for providing the ecosystem services. Equal profit accross fields may hide loosers and winners of the increase in ecosystem services, but leaves both farmers with the same profit, whatever the statu quo.

Among all landscapes, given that the profit tends to increase with the potential yield, the difference in terms of profit between the two fields is often in favour of the field with highest potential yield.

In areas where grassland is the statu quo on the least productive field, the increase in ecosystem services concerns most often only the field with the highest potential yield. The profit difference between the two fields is smaller in cost-efficient landscapes than in cost-inefficient ones. The share of the opportunity cost supported by the least productive field is also generally smaller. Cost-efficient landscapes therefore seem to be more equitable in terms of profit and cost sharing than cost-inefficient landscapes.

In other areas, where grassland is not the statu quo on any field, the increase in ecosystem services can be distributed to both fields, and the results are less clear.

In terms of repartition of profit, we observe that reverse landscapes are often associated to small profit gaps, which is logical. This is true in areas where none of the field has a low potential yield. This is interesting: while we showed that such reverse landscapes are more likely to be cost-inefficient when confronted to heterogeneity and landscape scale analysis, they are those for which the cost repartition is the most balanced.

8.6. Incentives in heterogeneous agricultural areas

We explicitely distinguish between two steps in policy design: the opportunity cost of changing agricultural management, and the minimal policy budget needed to implement the desired change. In the sections above, we determined which landscapes are cost-efficient, i.e. minimise the opportunity cost and maximise the ecosystem services.

In this section, we deal with the second step and calculate the policy budget needed to make profit-maximising farmers adopt the cost-efficient landscapes. We compute the incentives needed to achieve any of the landscapes studied in the subsection above, no matter which of the four cases of cost-efficiency they belong to. Again, we consider two types of incentives: action-based and result-based incentives.

One of the issues caused by heterogeneity is linked to the impossibility to adapt the incentives to each individual field. A certain degree of uniformity in incentives is unavoidable and adds to the participation constraints, potentially increasing the gap between the opportunity cost and the policy budget. In this setting, result-based incentives are sometimes referred to as a mean to cope with heterogeneity, and ensure cost-efficient policy design in presence of heterogeneity (Schwarz et al., 2008; Gibbons et al., 2011).

In the following analysis, we explore whether it is possible to achieve the landscapes on heterogeneous areas by means of uniform incentives and whether differences exist between both types of incentives, in particular if one type of incentives leads to lower budgets.

8.6.1. Simulating the policy budget

We use linear programming to find out the set of incentives which can achieve each landscape with the smallest policy budget. The difference here lies in the constraints, which impose the incentives to be identical over two different fields, with possibly different management options and agronomic contexts, and thus different levels of ecosystem services and opportunity costs.

In the case of result-based incentives, we solve the following linear programm for each management landscape j:

$$\begin{aligned}
\min_{X_j} \quad & X_j \cdot (Y_j^1 + Y_j^2) \\
\text{s.t.} \quad & X_j \cdot \left((Y_j^1 + Y_j^2) - (Y_i^1 + Y_i^2)\right) \geq (\pi_i^1 + \pi_i^2) - (\pi_j^1 + -\pi_j^2) \qquad \forall i \neq j \\
& X_j \geq 0
\end{aligned} \tag{8.3}$$

where X_j stands for the vector of solutions (incentives) solving the optimisation programm, and Y_j^1 and Y_j^2 stand resp. for the bundle of ES on field 1 and 2 of landscape j. The objective to minimise corresponds to the sum of subsidies (the policy budget).

π_j^1 and π_j^2 are the profits of field 1 and 2 of landscape j in the absence of any subsidies, as modelled in the ecological-economic model. The subscript i denotes every other combination of management options, so that π_i^k and Y_i^k are respectively the profit and ES bundle of field $k = 1, 2$ of every other combination of two management options i.

The first set of constraints imposes that with the set of incentives, the profit of the chosen option on each field of landscape j be greater than or equal to the profit of every other possible landscape (every other combination of two management options, not only cost-efficient ones), which is the condition for it to be adopted by a farmer as modelled in our economic framework. The last constraint imposes the incentives to be non-negative, which corresponds to the choice of modelling only subsidies encouraging the provision of non-marketed ecosystem services.

In the case of action-based incentives, the optimisation programm writes:

$$\begin{aligned} \min_{X_j} \quad & X_j \cdot (M_j^1 + M_j^2) \\ \text{s.t.} \quad & X_j \cdot \left((M_j^1 + M_j^2) - (M_i^1 + M_i^2)\right) \geq (\pi_i^1 + \pi_i^2) - (\pi_j^1 + \pi_j^2) \qquad \forall i \neq j \\ & X_j \geq 0 \end{aligned} \tag{8.4}$$

where X_j stands for the vector of incentives solving the optimisation programm, and M_j^1 and M_j^2 the agricultural practices on field 1 and 2 of landscape j. π_j^1 and π_j^2 represent the profit associated to fields 1 and 2 of landscape j, and subscript i every other landscape. The constraints correspond to the ones described above for result-based incentives, they ensure that the given landscape meets the participation constraints, and that incentives are non-negative.

8.6.2. Results: which landscapes can be achieved

Uniformity of incentives over heterogeneous areas imposes new participation constraints, which add to the ones existing in absence of heterogeneity. Sets of incentives cannot meet participation constraints for all landscapes. Many landscapes cannot be achieved.

Both types of incentives achieve almost all landscapes which are cost-efficient in heterogeneous areas (landscapes corresponding to cases 1 and 4). With result-based incentives, a few exceptions appear: one cost-efficient landscape is not achieved, and the opposite is sometimes true. More exceptions appear with action-based incentives, which achieve some cost-inefficient landscapes, often with intensive options. Cost-inefficient landscapes achieved all are adequate landscapes (the ranking of intensity in management follows the ranking in potential yield), with various management options.

In the end, differences between result-based and action-based incentives seem less striking than in absence of heterogeneity: both types of incentives globally target cost-efficient landscapes, with a slight advantage for result-based incentives. Incentives select cost-efficient landscapes, and some cost-inefficient "adequate" landscapes.

8.6.3. Results: policy budget

On heterogeous areas, uniform incentives add a participation constraint, compared to homogeneous areas, or to heterogeneous areas where incentives could be differentiated so as to be adapted to each context. We assess the impact of this constraint separately from the participation constraint already existing in homogeneous areas, by comparing with differentiated incentives. For both types of incentives, we calculate the policy budget with uniform incentives as described above and reuse results from Chapter 7 to calculate the policy budget if incentives were differentiated.

Participation constraints make the policy budget diverge from the opportunity cost. To assess in which extent they are binding, we express policy budgets as a ratio to the opportunity cost. On Figure 8.3, we represented the policy budget in four settings, expressed as a ratio to the opportunity cost. Results are shown for cost-efficient landscapes for area H (potential yield 1 = 8t/ha, potential yield 2 = 12t/ha). The series A and B represent policy budget with result-based incentives, A (light blue) in the case of differentiated incentives, B (dark blue) with uniform incentives. Series C and D represent policy budgets with action-based incentives, C (light orange) with differentiated incentives, D (dark orange) with uniform incentives. The effect of heterogneity is captured by comparing B to A, and D to C. When the series is equal to 1, participation constraints don't increase the policy budget. The higher the series, the higher the excess budget - the gap between the opportunity cost and the policy budget.

For many landscapes, the policy budget doesn't differ significantly from the opportunity cost. When it differs, the ratio is almost always higher with result-based incentives than with action-based ones. The policy budget with uniform incentives (dark colors) is allways higher than (or equal to) with differentiated incentives (light colors), meaning that heterogeneity does increase the policy budget. This difference is also higher for result-based incentives, which contradicts the argument that uniform result-based incentives better deal with heterogeneity than uniform action-based incentives.

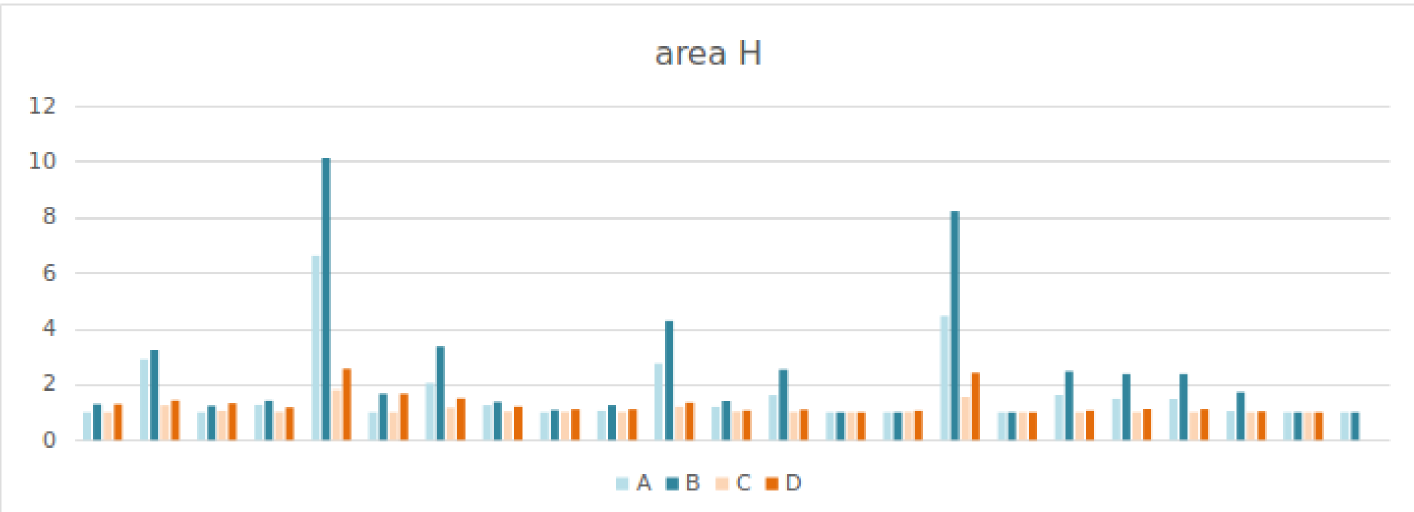

Figure 8.3.: *Comparison of result-based and action-based incentives in presence of heterogeneity*
Observations (x-axis) represent landscapes (combinations of management options over heterogeneous areas). All series are expressed relative to the opportunity cost of the landscape.
Series A: policy budget with differentiated result-based incentives
Series B: policy budget with uniform result-based incentives
Series C: policy budget with differentiated action-based incentives
Series D: policy budget with uniform action-based incentives

8.7. Discussion and conclusion

Our results show that cost-efficient policies at a higher spatial scale can't be inferred only from cost-efficient options at the field scale, and heterogeneity adds complexity to it.

The implementation of cost-efficient landscapes with incentives delivers results which are in contradiction with the existing literature. We show that result-based incentives target well the cost-efficient landscapes (which is conform to the conclusions in the literature), but that the policy budget needed is higher than with action-based incentives. We show that the ability of action-based incentives to select cost-efficient landscapes is slighly lower, but globally good. Compared to the analysis at field scale, we find the same result that result-based incentives lead to a higher policy budget, which is contrary to results in the literature. In the end, we don't find a clear advantage of result-based incentives in presence of heterogeneity.

We may find this result because among cost-efficient landscapes, the link between actions and results is rather univocal: the modalities of agricultural practices that are subsidised also increase ecosystem services.

Our model has several limits, which could modify the results if they were taken into account. We included only options that were cost-efficient at the field level, so we can only notice the cases where landscape scale and heterogeneity cancel their cost-efficiency. With heterogeneity, it could be that combinations are cost-efficient, while including options which are not cost-efficient at the field scale, and that we don't consider in the analysis.

Besides, our analysis doesn't account for spatial interactions which would call for spatial targeting of incentives and could give an advantage to result-based incentives.

Part V.

Discussion and conclusion

9. Discussion

The rationale of this thesis is to study which incentives can increase the provision of non-marketed ecosystem services. This issue directly echoes questions about the design of agri-environmental policies and has therefore received lots of attention, in particular under the form of policy analyses. Such theoretical results as ours require to be put in perspective with those analyses.

In this chapter, we review the main current policies and incentives which impact the decisions of farmers and the provision of ES in agriculture. We also review the criticisms and recommandations that apply to agri-environmental policies, and discuss our results in light of these recommandations. This also helps identify the limits of our approach.

9.1. Current context in terms of incentives

Existing incentives can be classified in two categories: incentives that push farmers to provide marketed commodities, and incentives to provide non-marketed ecosystem services. We adopt such a classification given the strong underlying tradeoff between agricultural production and non-marketed ES in intensive agricultural landscapes: fostering agricultural production generally decreases the provision of non-marketed ES. Though, we don't assume that removing agriculture would be the solution to enhance the provision of ecosystem services. In marginal lands, sustaining agricultural production can be a way to provide ecosystem services and other public goods. This is the logic behind the payment for least-favoured areas, which encourages farmers to continue production in montaineous or less fertile areas, in order to provide at the same time non-marketed ecosystem services related for example to landscape quality.

9.1.1. Two opposite types of incentives

Incentives to produce agricultural commodities Contrary to other ecosystem services, production is marketed. Agricultural prices are the main incentive to provide agricultural commodities. Everything that increase the price farmers receive for agricultural commodities has an important impact on the decisions concerning production.

In the EU, the Common Agricultural Policy (CAP) is the most important public policy concerning the agricultural sector. In the past, it aimed at increasing agricultural production while guaranteing low consumption prices, and provided a clear incentive to produce through payments proportional to agricultural output. Since 2003, subsidies have mostly been decoupled from agricultural output, which reduces the incentive to provide agricultural commodities. They now are distributed based on the agricultural area without requirement to cultivate, and are justified by their income support role. However, these payments represent a large proportion of agricultural income (Matthews, 2016), and are likely to still encourage agricultural production, especially through a wealth effect (Femenia et al., 2010). In addition, in some countries including France, the current direct payments are based on the historical level of coupled support, even if some convergence in the level of payments has occured among farms, production orientations and regions. Matthews (2016) shows that the these payments still mainly benefits to large farms (95% of direct direct payments go to farms with a higher than median income), which rely generally on more intensive management, with higher specialisation and input reliance (Mahé and Bureau, 2016).

Apart from direct payments, many incentives encourage agricultural production via investment. Subsidies for the modernisation of production processes encourage farmers to buy newer and more powerful machines, and indirectly foster a higher production by increasing productivity, decreasing the marginal production cost or requiring higher levels of inputs. In France, fiscal rules also encourage farmers to invest[37], and lead to an investment strategy based on the short-term financial returns rather than the real needs related to the production process (Delaire and Bonhommeau, 2011).

These incentives don't require many conditions or specific actions to receive the payments, and thus most farmers benefit from them. The biggest source of incentives is clearly the CAP and its direct payments.

Incentives to provide non-marketed ecosystem services On the other hand, several incentives foster the provision of non-marketed ecosystem services. They cover different types of instruments. First, some regulatory tools impose rules to which farmers must comply. The Common Agricultural Policy encompasses two layers of such norms. Direct payments are conditioned to cross-compliance (norms concerning environment, food safety and animal welfare). In addition, since 2014, 30% of direct payments are conditionned to additional requirements about agricultural practices. Environmental directives taken at the EU level (e.g. Nitrates Directive, Water Framework Directive etc.) provide other norms which encourage the provision of non-marketed ES.

Second, the CAP also contains schemes relying on incentives. In particular, farmers can apply to Agri-Environmental Measures (AEM), and receive a subsidy in exchange for implementing actions providing non-marketed ES beyond what is required by law or norms[38] (e.g. organic agriculture, implementation of hedges...). This payment compensates the corresponding income loss, it is granted for the action taken, without any requirement related to the environmental results. AEM are cofinanced by Member States, contrary to direct payments which are totally financed by the EU budget. Outside of the CAP, other existing economic incentives include taxes on pesticides and fertilisers implemented by some European countries.

These norms and economic incentives are reinforced by the creation of specific target areas judged as sensitive and more worth preserving. The Natura2000 network at the EU level, and national or regional parks and Water Catchment Areas at the national or subnational levels are a lever to foster the implementation of more ambitious regulations and incentives to provide non-marketed ES. Many other schemes exist at the national

[37] More precisely, agricultural income tax cuts are granted for investments and for capital gains when reselling machinery.

[38] Only part of Agri-Environmental Measures aim at preserving the environment. Other AEM aim at preserving rural heritage and landscapes.

or subnational levels, which focus on consultancy, information campaigns and awareness raising, and collective action.

In the end, many different incentives and norms to provide non-marketed ES cohabit, managed by diverse stakeholders. All these schemes are conditionned to respecting precise requirements, contrary to most incentives to produce.

Special case of subsidies for less-favoured areas Subsidies for less-favoured areas are a particular case of an incentive to produce that also aims at providing non-marketed ecosystem services. Less-favoured areas encompass areas where agricultural activity is endangered and provides many social co-benefits (e.g. maintenance of rural activity) and non-marketed ecosystem services (related to biodiversity and landscape quality). They gather for example montain areas, scarcely-populated and low-productivity areas. The associated payment is based on the agricultural area and conditioned to encourage extensive agricultural activities.

9.1.2. Both types of incentives contradict themselves

Although many incentives to provide non-marketed ecosystem services exist, non-marketed ecosystem services in agriculture are still unsufficient to ensure the sustainable functionning of ecosystems. The first reason is that these incentives come in contradiction with incentives to produce.

Globally, production and non-marketed ecosystem services stand in a tradeoff (Lee and Lautenbach, 2016). Indeed, agricultural practices that increase production in the short term generally endanger ecosystem's supporting processes. As a consequence, incentives to produce more, such as direct payments of the CAP, are in contradiction with incentives to provide non-marketed ecosystem services. Each farmer faces contradictory incentives: it is not the case that certain farmers receive incentives to produce and other farmers receive incentives to provide ecosystem services. The environmental conditions of direct payments and green payment are not ambitious enough to solve this contradiction (Pe'er et al., 2014; Bureau and Thoyer, 2014), and most investment support is allocated without any real environmental conditions and may have negative impacts on the environment (Allen and Hart, 2013).

9.1.3. A general disbalance in the available budget

The second reason why current incentives fail to provide sufficient levels of non-marketed ES is the disbalance in the budget allocation. Given the antagonism between both types of incentives, starting from the current situation, incentives to provide non-marketed ecosystem services should be at least as strong as incentives to produce. In practice, the budgets dedicated to both objectives differ much.

Within the CAP, the budget repartition is much in favour of incentives to produce. Over the period 2014-2020, the budget for incentives to produce represents 75% of EU's budget for the CAP. In comparison, the planned EU expenditures for Agri-environmental measures and organic farming represent slightly more than 5% of the CAP budget (own calculations, using data in European Commission (2016) and Henke et al. (2015)). Including Member States cofinancing, Duval et al. (2016) estimate that AEM represented about 8% of the total expenditures (EU + Member States) over 2007-2013. At the farm level, the same disproportion exists: direct payments make up 47% of the farmers' income on average at the UE level, other payments (including AEM and other 2nd pillar support) make up only 15% (Matthews, 2016). As a result, AEM do have a positive but generally weak impact on the environment (Barbut and Baschet, 2005; Batáry et al., 2011; Matthews, 2016)[39].

This disbalance remains even when including other incentives to provide non-marketed ES. To have an idea of the total budget dedicated to both types of incentives (including policies outside of the CAP), we compare at the Member State level on one hand the budget of the direct payment of the CAP (First Pillar) and on the other hand the total budget dedicated to the provision of non-marketed ES (including policies outside of the CAP). We assess the latter by using the expenditures reported in national accounts, dedicated to environmental issues related to agricultural areas (protection of soil and water bodies and of biodiversity and landscape). These expenditures include the national co-financing of the AEM, but not the EU's contribution[40]. Table 9.1 confirms the general disbalance in budgets. Except for Austria and the Netherlands, the budget of the First pillar of the CAP largely exceeds the expenditures for the protection of environment. These figures are only intended to provide an order of magnitude. The expenditures for the protection of the environment are probably an overestimation of public incentives to provide non-marketed ES in agriculture, since they also include expenditures applying to other economic sectors (private and public expenditures related to depollution of soils and water bodies, compensation of biodiversity or landscape disturbances related to new infrastructures)[41].

9.1.4. The available budget should be more oriented towards the provision of non-marketed ES

It is necessary to increase the provision of non-marketed ecosystem services. The two types of incentives stand in contradiction, and their respective budgets are disproportioned. In theory, the efficient way to increase the provision of non-marketed ES is to cut current incentives to produce (OECD, 2010; DG for Internal Policies (European Parliament), 2010; Matthews, 2016). Agricultural prices already are an incentive to produce, and "Public money should be allocated to public goods".

In practice, it is more realistic to envisage a lighter shift in budget and the development of AEM. The budget allocated to incentives to produce provides major support to agricultural income. Farmers depend on them for their living, and cutting these incen-

[39] More precisely, Uthes and Matzdorf (2013) find that effects depend on schemes and countries, and report success in the protection of farmland birds and pollinators populations and in extensive agricultural areas.

[40] We didn't find detailed budgets of the Second Pillar at the Member State level, but as described above, EU's budget for AEM represent around 5% of the total CAP budget, or about 6.7% of the First Pillar budget.

[41] In the case of France, according to the explanations of these statistics, 38% of the expenditures related to water and soils don't concern agriculture, and at least 17% of the expenditures for biodiversity and landscapes are compensations for the construction of new infrastructures (see `http://www.statistiques.developpement-durable.gouv.fr/lessentiel/s/depenses-protection-lenvironnement.html`)

	CAP 1st pillar budget (mio. euros)	Expenditure for env. protection - soil & water (mio. euros)	biodiversity & landscapes (mio. euros)
Austria	718	922	42
France	8332	1230	1202
Germany	5197	90[a]	1400[a]
Netherlands	840	630	789
Poland	3208	213	130

Table 9.1.: *Public expenditures to support agricultural production and the protection of environment, in several EU countries (2014)*
Expenditures for the protection of environment include national co-financing of AEM.
Sources : Eurostat, Statistisches Bundesamt, Financial report of EAGF 2014
[a] 2010 figures (no later data available)

tives would have major effects on agricultural income. In accordance with the need to maintain agricultural income, it is also unrealistic to consider taxes for pollution, and considering the provision of ES is a way to inverse the perspective.

Current transfers between both pillars of the CAP as allowed by the last reforms enable Member States to divert budget from incentives to produce towards incentives to provide non-marketed ES. However, they are not mandatory, and not all Member States decided to use this possibility. It is also possible to reduce incentives to produce while keeping income support by playing on the design of current direct payments. The introduction of green payments (conditionning 30% of direct payments to the implementation of some low-level agroecological practices) are a step in this direction, even if the agroecological practices kept don't go much beyond the current practices. Another possible solution, capping direct payments per worker, would be more compatible with the logic of income support, reduce the incentive to intensify agricultural production and ensure a more equitable repartition of direct payments accross farms of different sizes. Going further than the Green Payment, DG for Internal Policies (European Parliament) (2010) and Matthews (2016) propose to design the Common Agricultural Policy in different layers: a base payment dedicated to income support, and additional layers of subsidies targeted towards provision of low levels of ecosystem services, precise and higher levels of ecosystem services etc.

Beyond playing on the budget allocated to incentives to produce, adjusting the design of AEM is also a mean to increase the provision of non-marketed ES by getting more ES for the money spent. Many recommandations concern the enhancement of agri-environmental policies design.

This overview provides the starting point of this thesis. It pleads for the improvement of agri-environmental schemes. In this thesis, we explore questions relative to the design of agri-environmental policies: the identification of efficient actions and the necessity to account for the cost of ES provision, their implementation by means of incentives, and the consequences of heterogeneity. In the following sections, we put our results in perspective with analyses on the design of agri-environmental policies, and discuss them. We also review the limits of our approach.

9.2. Efficiency of agroecological solutions

In Chapter 6, we show that explicitely determining which solutions maximise the provision of non-marketed ES is important. The provision of the multiple ES must be thought in a coherent way: options increasing one ecosystem service don't necessarily increase the others.

This echoes criticisms and recommandations about current policies, which don't always target efficient or effective solutions. The European Court of Auditors (Cour des Comptes Européenne, 2011) estimated that in 24% of cases, the expected environmental benefits of AEM can't be proved. This doesn't mean they don't exist, but at least the AEM have been designed without being sure that they bring an environmental benefit. This underlines that the identification of environmental benefits and the link with actions should be reinforced (Uthes and Matzdorf, 2013; Duval et al., 2016), as well as the ex-post evaluation of agri-environmental schemes, which is lacking within the CAP (Epices and ADE, 2017).

Besides, current agri-environmental policies don't always enable an integrative management of the several issues in agriculture (Galler et al., 2015). EU's directives on the environment are a striking example: they are designed separateley for each environmental issue (water quality, waste, bird protection, etc.) rather than in a holistic way. This also holds for the Agri-environmental Measures. The EAFRD (the structure implementing Rural Development Measures of the CAP, of which AEM) explicitely puts up multiple ecological and social objectives, but in reality agri-environmental measures are designed to tackle mostly individual objectives.

Targeting maximal provision of ES is made difficult by spatial interactions and dynamics in the provision of ES. Our applied modelling approach doesn't include spatial interactions nor dynamics, which is a limitation but can be an interesting perspective for future work.

Spatial interactions require to meet thresholds of participating farmers, or to think the spatial arrangement of actions to provide ES, and possibly use spatially-differentiated incentives (e.g. agglomeration bonuses). Dynamics impose to define the right horizon and trajectory of actions. It is difficult to know how it could change our results, but it would make them more complex.

Our agroecological model is rather simple, especially what concerns the available management options. Therefore, we neglect interesting management options, which may change efficient solutions.

In particular, the possibility of introducing more complex crop rotations seems to be an interesting way to increase the provision of ecosystem services.

9.3. Accounting for the cost of ES provision

In Chapter 6, we show that accounting for the costs of ES provision is important. It allows to consider solutions providing lower levels of ES, at a smaller cost. Those solutions can be implemented on larger areas than more costly solutions and they may provide much ecosystem services in total. We compare the cost of two strategies to provide ES: i) favouring a modest increase of non-marketed ES over all the area, or ii) dedicating part of the area to a radical increase in non-marketed ES. We show that no strategy dominates the other, it depends on the targeted increase in ES.

The debate over which of the two strategies is the best to increase ES is present in policy analyses. For example, Member States have made different choices for the implementation of AEM. In France, the choice has been made to sprinkle them. The total budget is equivalent to this of Finland or Austria, but the area covered by AEM is the highest among all Member States, three times the area covered by Finnish or Austrian schemes. The average payment per ha is the lowest, and to make farmers participate, measures need to be weak (Duval et al., 2016). The European Court of Auditors Cour des Comptes Européenne (2011) underlines that in most cases, the repartition of budgets among measures and the level of payments are not determined so as to meet a participation threshold guaranteeing the provision of ecosystem services.

One limitation of our model is that we model decision-making by profit-maximisation, and include only variable costs in the calculation of profit. In reality, the willingness to accept (i.e. the minimal amount farmers require in order to suscribe an Agri-environmental measure) differs from the opportunity cost. We probably underestimate the real cost of providing ecosystem services and the role of other drivers of farmers' decision-making.

Our approach considers opportunity cost in a restrictive way, but indeed this is exactly the issue current agri-environmental policies face (Duval et al., 2016). The payments are calibrated on the opportunity cost and are often too low to cover the willingness to accept of most farmers, resulting in a low adoption rate. For example transaction costs are not considered. They relate to gathering information, to filling administrative forms, to monitoring, to coordination, etc. They are estimated to represent 14% of the total costs of implementing Agri-Environmental Measures (Mettepenningen et al., 2009). They impact the willingness to accept (Ruto and Garrod, 2009) and the adoption of AEM (Ducos and Dupraz, 2006). Changing agricultural practices may also need costly investments (professional training, new machines for reduced tillage or mechanical weeding etc.), which don't appear in the opportunity cost. Besides, the willingness to accept encompasses other social and psychological drivers (personal convictions about the environment, risk aversion etc), which are not accounted. This could explain that few farmers decreased pesticide inputs while the opportunity cost of reducing their use is close to zero Lechenet et al. (2017).

In order to close the gap between the agri-environmental payments and the willingness to accept, a first solution is to reduce transaction costs. For example, technical and personalised advice can decrease transaction costs, and thus the willlingness to accept of farmers (Espinosa-Goded et al., 2010). A second solution (which doesn't exclude the first) is to include all costs in the calibration of agri-environmental payments, including fixed costs. If farmers are being paid for the provision of ecosystem services, AEM should compensate the total cost, not only variable or marginal costs (Barnes et al., 2011; Duval et al., 2016). This could also be done through a better connection between agri-environmental measures and incentives financing investments. Currently, poor connections between investment support schemes and AEM exist (Duval et al., 2016). Third, it is possible to play on social and psychological drivers of willingness to accept. Insurance schemes coupled to the adoption of AEM could tackle risk aversion in front of higher risk incurred by adopting some AEM (reducing pesticides for example). Collective action can decrease the perceived risk and the path dependance (Duval et al., 2016) and help diffuse knowledge. The role of knowledge diffusion in the permanence of pro-environmental practices is underlined by Kuhfuss et al. (2016). Authors see it as an example of nudging, i.e. playing on psychological drivers of decision-making, and relate it to the emergence of new social norms. The emergence of social norms due to a larger adoption of environmental policies is also seen as a solution by Nyborg et al. (2016). Last, the valorisation of environmental efforts through the development of labels and specific value chains also facilitates the adoption of agri-environmental measures (Kuhfuss, 2013): if farmers can get a higher marketed income from their participation to AEM, the required payment is lower.

Another limitation of our approach and of current agri-environmental policies is that they neglect the variations in crop prices. An increase in crop price increases the opportunity cost of adopting AEM, and plays on the adoption rate (Uthes and Matzdorf, 2013). Adapting the payments to the opportunity costs would mean to make them vary along with the crop price, which raises problems about the budget planning, and increases income variability.

9.4. Incentives

In Chapter 7, we compare action-based and result-based incentives for the implementation of ES-increasing management options. Result-based incentives exactly select cost-efficient management options. However, we find that interactions among ES strengthen participation constraints and lead to a higher policy budget with result-based incentives than with action-based incentives.

Few research studies provide comparable results. Contrary to our results, policy analyses often cite action-based incentives as unable to select cost-efficient options, and mention result-based incentives as possible enhancements of current schemes.

Experimentations of result-based incentives remain limited, as many ecosystem services are difficult to measure precisely at the field or farm level. Result-based schemes were experimented mainly for biodiversity protection, with results being measured as the presence of abundance of some species. These schemes are not "pure" result-based incentives: the payment often doesn't varies with the result (it is rather granted for the respect of a certain result-based constraint) and these incentives are embedded in action-based schemes. The experimentations seem successful (Musters et al., 2001; Allen et al., 2014), leading to more cost-effective biodiversity conservation. Literature underlines that the choice of the indicator is very important. In particular, it must be hard to manipulate, else the risk is that farmers take actions increasing this indicator but with no real environmental benefits[42].

Reverse auctions can be cited as another example of "impure" result-based incentive. Projects financed by agri-environmental policies are selected on the basis of expected results and costs. This makes them quite different from "pure" result-based incentives, as the payment is not necessarily proportional to the real environmental result. They have been experimented for biodiversity conservation in the USA and Australia, with success (Duval et al., 2016). In France, an experimentation for enhancing water quality was less sucessful, partly due to implementation details. Studies underline that many parameters can influence their success (which advice and information farmers should receive, how many rounds the auction should count, choice of the payment amount etc.). These schemes can bear high transaction costs and introduce an improductive competition among farmers or on the contrary create collusions among farmers and increase payments required (Kuhfuss, 2013).

Reed et al. (2014) and Moxey and White (2014) provide a good summary on result-based approaches: they shouldn't be considered as the solution to all current issues encoutered by agri-environmental policies. Interesting characteristics of result-based incentives (spatial targeting and diferentiation of payments) can also be achieved with action-based schemes. Fleury et al. (2015) concludes that the success of experimentations of "flowering meadows" result-based payment relies on its inclusion in a broader scheme including training, education, advising and the development of a positive social norms toward the participation in agri-environmental measures.

9.5. Coping with heterogeneity

In Chapter 8, we introduce heterogeneity and explore its consequences. Heterogeneity itself changes cost-efficiency, and considering the landscape scale (rather than field scale) also changes the cost-efficiency results, even without heterogeneity.

The results on the importance of assessing cost-efficiency at the landscape scale are difficult to link to practical recommandations. It is difficult to assess cost-efficient solutions at the landscape scale: it would require to compare all possible combinations of management options available at the field scale. It would also be complicated to implement a solution determined at landscape scale: it would mean that the management option of a farmer are dependant on the characteristics of all fields of the landscape. However, the determination of cost-efficient solutions should at least be done at the farm scale. In this sense, part of the AEM (systemic AEM) apply to the whole farm, and can lead to more cost-effective provision of non-marketed ES.

Our results show that cost-efficient increase in non-marketed ES in heterogeneous landscapes often include a high contribution from the most productive part of the landscape.

[42] In the case of farmland bird protection, the environmental benefits lies in the respect of specific grassland management (mowing dates and heights, grazing schedules etc.). Result-based schemes where the indicator is the presence of bird nests make nests become profitable, which can lead some farmers to look for nests, fence them and manage their grassland as usual while receiving result-based payments. Such a strategy is not necessarily adapted to protecting th species, contrary to changing grassland management.

However, heterogeneity makes it impossible to tailor incentives according to the willingness to accept. It has three detrimental effects on the budget-efficiency of policies: it decreases the adoption rate (farmers having a higher willingness to accept than the payment don't participate), overcompensates those having a lower willingness to accept, and automatically select farmers having the lowest willingness to accept, which may not offer cost-efficient solutions. This is exactly the criticism current agri-environmental policies face (Uthes and Matzdorf, 2013; Duval et al., 2016): extensive agricultural areas are overrepresented, whereas the cost-efficient provision of non-marketed ecosystem services may rely on intensive areas. Targeting more intensive agricultural areas would require to set higher payments, thereby reducing potential area of application. It would also raise equity problems: more intensive farms have higher profits than extensive ones, so that granting them higher payments may seem unfair.

In presence of heterogeneity, we observe that uniform incentives are associated to a higher policy budget than differentiated ones.

This result has been identified since a long time in policy analyses and scientific literature. A solution to the issues of heterogeneity in costs and environmental benefits and to the uniformity of incentives is to design and calibrate incentives at local level, and in a participative way. Targeting a lower number of farmers who share certain characteristics can reduce the heterogeneity among farmers. Calibrating incentives at this lower level also reduces their uniformity. Blumentrath et al. (2014) conclude that policies designed at the local level and with the participation of farmers are more effective. For example, since 2013, existing Agri-Environmental Measures (at least in France) are designed and calibrated by regions (NUTS2 level). Both the targeted environmental stakes and measures, and the amount of the payment vary accross regions. Some measures are available only in certain "territories", which ensure a higher homogeneity of ES provided. This evolution enabled more efficient design (Duval et al., 2016). Similarly, Swedish payments for semi-natural grasslands vary according to their expected environmental benefit, in this case the probability that they host endangered species (Ekroos et al., 2014). The same principle stands behind demands for more subsidiarity for Member States in order to determine priorities, actions, and the calibration of payments within the CAP.

Favoring the calibration of incentives at the local level also has other advantages: it favors the participation and collaboration of multiple stakeholders (farmers, local institutions, environmental agencies, etc.), and can foster mutual trust between stakeholders and reinforce acceptation and legitimity of the policies implemented.

A more complex alternative to reduce asymetry of information is to use so-called "menus" of Agri-Environmental Measures implemented in some Member States (esp. in the UK and in Austria). A list of options with varying levels of ambition and payments is proposed and farmers can combine them according to specific rules (Duval et al., 2016).In France, a scheme to enhance water quality has been tested, with two levels of engagement: farmers can choose between a "light" version of the measure with low requirements and a low payment, and a more ambitious version of the measure, with a larger reduction in input use and a higher payment (Kuhfuss, 2013). These approaches rely on self-selection of farmers according to their own costs, and correspond to kinds of "self-screening contracts" mentioned in the theoretical economic literature Ferraro (2008). When calibrated properly, they offer differentiated incentives, without the need for the regulator to select which level of incentives each farmer should be proposes, nor to know his costs.

Concerning the implementation of cost-efficient solutions, we find that in most cases, result-based incentives need a higher policy budget than action-based incentives. Heterogeneity doesn't make result-based incentives more attractive.

This is somewhat contradictory with existing results in the literature (Gibbons et al., 2011). Our approach gives a large role to interactions among ecosystem services and especially allows for synergies among regulated ecosystem services, which probably explains this. No empirical policy analyses was found on this aspect.

9.6. Concluding remarks

Our results agree with policy analyses on several points. First, agri-environmental measures must be designed according to clear environmental objectives, and if possible in an integrative way. The link between (changes in) agricultural practices and ecosystem services provided must be established. Second, agri-environmental policies should consider the opportunity cost of ES provision in order to maximise the ecosystem services provided for a given budget. Depending on the situation, the optimal strategy can be to concentrate the provision of ES on part of the area, or to disperse it on the whole area. Third, given heterogeneity, policies should be spatially targeted, and payments differentiated. We find that result-based incentives don't provide a cure-all solution for agri-environmental policies in presence of heterogeneity, because of interactions among regulated ecosystem services. Solutions to enhance the cost-effectiveness of current policies can be implemented through action-based incentives. Last, an increase in ES provision compared to current situation requires to increase the available budget for agri-environmental measures. Much progress can probably be achieved on the efficiency of current agri-environmental schemes, but the current disbalance in the budgets should not be neglected. Despite this evident need for a stronger budget for agri-environmental policies, the last CAP negociations led to the adoption of a decline in the budget of Second Pillar (dedicated to Rural Development, including agri-environmental policies) for the Multi-Annual Framework 2014-2020. It has declined by 18%, even if the share of Second Pillar earmarked for environment rose from 25% to 30%. On the other hand, the First Pillar suffered a lower decline in its budget, and can even be considered as stable if compared to its counterfactual[43]. The next reform of the CAP is been discussed, let's hope progress will be made towards policies providing more non-marketed ES.

[43] `http://capreform.eu/the-cap-budget-in-the-mff-agreement/`

Part VI.

Appendices

Specific vocabulary used in the document

Terms defined within this thesis

- **(management) option**: combination of agricultural practices. There are 121 options available in the model, each provides a given bundle of ES.
- **(Heterogenous/homogenous) area**: set of 2 fields, with different or identical agronomic characteristics
- **landscape**: combination of management options over an area.
- **Participation constraint**: constraints imposing a management option to be the most attractive (in our mode, profitable) in order to be chosen by a farmer
- **achieve** a management option: calibrate incentives so that the given option is the most profitable option among all, i.e. respect the participation constraint.
- **opportunity cost**: cost of changing agricultural practices, including the yield loss
- **policy budget**: minimum amount of subsidies so that a management option becomes the most profitable, i.e. respect participation constraints.

Generic terms used in economics

- **willingness to accept**: minimum amount a farmer needs to receive in order to adopt a given management option. It can include the role of psychological or social drivers.

Résumé étendu en français

1. Introduction

L'agriculture fournit de nombreux biens et services aux humains. Certains de ces biens et services ont une valeur marchande. Beaucoup d'autres ne sont pas échangeables sur un marché, et n'ont pas de valeur marchande, bien qu'ils contribuent au bien-être humain: stockage de carbone qui limite le changement climatique, le plaisir d'observer un paysage... Ces biens et services sont appréhendés largement par le cadre d'analyse des services écosystémiques. Ce cadre considère tous les services rendus par les écosystèmes aux humains, qu'ils aient une valeur marchande ou non.

Au 20ème siècle, en Europe, l'intensification agricole a conduit à une augmentation de la production de denrées agricoles marchandes et à une dégradation de l'environnement et un déclin de tous les biens et services non-marchands fournis par les agroécosystèmes. L'utilisation de pesticides, la baisse de la biodiversité agricole, la suppression de nombreuses zones semi-naturelles (haies, bordures de champs, zones humides...) a eu des effets dramatiques sur la biodiversité agricole: les insectes, et particulièrement les pollinisateurs (Deguines et al., 2014), les oiseaux spécialistes agricoles (Burel et al., 1998; Donald et al., 2001; Wretenberg et al., 2006), et même quelques mammifères spécialistes des milieux agricoles (de la Peña et al., 2003; Pocock and Jennings, 2008). La teneur en matière organique des sols a diminué, ainsi que la biodiversité des sols (Matson, 1997). Les masses d'eau sont contaminées par les résidus de pesticides, des quantités excessives de nutriments qui causent leur eutrophisation, et des sédiments et particules organiques. Le secteur agricole est aussi responsable d'une part importante des gaz à effet de serre. Cette tendance n'est pas durable, à la fois pour les écosystèmes eux-mêmes, et pour la production alimentaire: le déclin des pollinisateurs et de la fertilité des sols, ainsi que le changement climatique, menacent les rendements agricoles en Europe (Deguines et al., 2014; Stoate et al., 2001; Tan et al., 2005). Les récentes évaluations des services écosystémiques concluent au déclin des services écosystémiques de régulation, qui sont non-marchands (Millenium Ecosystem Assessment, 2005b; Thérond, O.(coord) et al., 2017).

Pour comprendre et enrayer le déclin des services écosystémiques, il est crucial de prendre en compte les **interactions complexes entre les services écosystémiques**. Des processus agroécologiques communs déterminent la fourniture des différents services écosystémiques, et créent des interactions multiples et complexes entre eux. Par conséquent, il est impossible de séparer la fourniture d'un service écosystémique de la fourniture des autres services écosystémiques, et d'en faire varier un sans faire varier les autres (Bryan, 2013). Dans cette thèse, nous parlons de **bouquets de services écosystémiques** pour capturer le fait que leur fourniture est interdépendante.

D'un point de vue économique, les fortes interactions entre services écosystémiques et le déclin des services écosystémiques non-marchands sont liés. Premièrement, en l'absence de politiques agroenvironnementales, seule la production agricole génère du profit, et il n'y a pas d'incitations (au moins à court terme) à fournir des services écosystémiques non-marchands. Un agriculteur prenant en compte son seul profit individuel a intérêt à se concentrer sur la production marchande. Les services écosystémiques non-marchands sont des biens publics selon la théorie économique, ils bénéficient à plus de personnes que ceux qui les fournissent. Deuxièmement, même si les interactions entre services écosystémiques sont complexes, la production marchande entre globalement en contradiction avec la fourniture des autres services écosystémiques, ce qui a fait que l'intensification agricole a induit le déclin des services écosystémiques non-marchands. Des changements sont nécessaires pour trouver un compromis entre la production marchande et la préservation des autres services écosystémiques. En particulier, en ce qui concerne les paysages agricoles intensifs en Europe, cela signifie changer les pratiques agricoles (par exemple introduire des habitats semi-naturels, diminuer l'utilisation de fertilisants et de pesticides), plutôt que changer l'usage des sols (créer des réserves pour la conservation). Divers changements de pratiques agricoles sont envisageables, associés à des changements divers et variés en termes de services écosystémiques: réduire l'utilisation des fertilisants et pesticides n'ont pas exactement les mêmes conséquences que planter des haies, mêmes si ces différentes solutions augmentent la fourniture de services écosystémiques.

Pour inciter les agriculteurs à changer leurs pratiques agricoles, des politiques agroenvironnementales sont mises en place, par exemple à travers les Mesures Agroenvironnementales de la Politique Agricole Commune dans l'Union Européenne. Ces politiques visent à augmenter la fourniture de services écosystémiques non-marchands, en compensant les coûts associés.

Ces politiques font face à plusieurs défis.

Premièrement, l'objectif en termes de services écosystémiques est pluri-dimensionnel: ces politiques doivent combattre le déclin de plusieurs services écosystémiques. À cause des interactions complexes entre services écosystémiques, il n'est pas évident d'identifier quelles options fournissent le plus de services écosystémiques et devraient être ciblées par ces politiques. Définir des objectifs pour chaque service écosystémique séparément peut poser problème, étant donné que les solutions pour augmenter la fourniture des différents services écosystémiques peuvent ne pas être cohérentes. Par exemple, une incitation à augmenter un service écosystémique peut avoir des effets négatifs sur les autres services écosystémiques (Lindenmayer et al., 2012). Cet aspect est négligé par les solutions théoriques issues de la littérature économique, et suivre ces solutions peut causer ce problème. Plus précisément, la solution pigouvienne qui consiste à donner un prix aux biens publics en fonction du consentement à payer de l'ensemble de la société peut ne pas atteindre la solution espérée si le bouquet de service écosystémiques n'existe pas. Le type d'interactions entre services écosystémiques détermine les stratégies pour fournir des services écosystémiques. En particulier, dans le cas des services écosystémiques sont antagonistiques, il détermine s'ils doivent être fournis ensemble par une même option qui constitue un compromis, ou s'ils doivent fournis séparément par différentes options, par exemple en réservant une part du territoire à la fourniture d'un service écosystémique et l'autre part à l'autre service écosystémique. Selon les interactions entre services écosystémiques, la meilleure stratégie peut être une combinaison de plusieurs options. Le débat land-sparing/land-sharing est un exemple de cette question: la biodiversité doit-elle être fournie conjointement avec la production agricole, ou le compromis doit-il reposer sur une ségrégation spatiale avec des pratiques agricoles plus intensives d'un côté et la terre "économisée" conservée seulement pour la biodiversité de l'autre côté.

Deuxièmement, augmenter la fourniture de services écosystémiques a un coût. Ce coût comprend la perte de rendement (liée par exemple à une baisse de l'utilisation de fertilisants et de pesticides) et les coûts supplémentaires liés aux pratiques agricoles (par exemple la plantation d'une haie). C'est le coût d'opportunité, le coût de changer les pratiques agricoles pour augmenter la fourniture de services écosystémiques non-marchands. Du point de vue du bien-être agrégé, le coût total de la fourniture de services écosystémiques doit être minimisé, que les politiques agroenvironmentales le fassent porter par les agriculteurs (via des taxes) ou par la société tout entière (via des subventions). En réalité, les politiques agroenvironnementales reposent généralement sur des subventions, et les fonds publics sont limités. Dans un tel cas, le coût doit être inclus dans l'analyse pour atteindre la fourniture maximale de services écosystémiques (Naidoo et al., 2006). Négliger le coût revient à privilégier des solutions qui fournissent des niveaux élevés de services écosystémiques par hectare, mais qui peuvent être très coûteuses, alors que prendre en compte le coût permet de considérer des solutions qui fournissent moins de services écosystémiques, mais qui sont moins coûteuses. À budget donné, ces dernières solutions peuvent alors être adoptées sur une plus grande surface, et finalement fournir plus de services écosystémiques. Par conséquent, les politiques agroenvironnementales doivent trouver un compromis entre les multiples services écosystémiques et le coût. Étant donné les multiples interactions entre services écosystémiques, il est impossible de calculer le coût associé à la fourniture d'un service écosystémique individuellement. Mesurer ce coût ne peut se concevoir que pour un bouquet de services écosystémiques. Un changement dans les pratiques agricoles se répercute sur tous les services écosystémiques. Contrairement à ce qui est fait dans la littérature (Ruijs et al., 2013), tous les critères doivent être pris en compte conjointement dans la conception des politiques agroenvironnementales et la définition des solutions qu'elles doivent viser.

Le troisième défi est que les politiques sont en générales mises en place à l'aide d'incitations économiques, ce qui soulève plusieurs problèmes. Les décideurs politiques ne peuvent pas dicter aux agriculteurs les pratiques agricoles qu'ils doivent adopter, ils peuvent seulement concevoir des incitations qui poussent les agriculteurs afin qu'ils adoptent ces pratiques. Ceci introduit des contraintes de participation dans la conception des politiques publiques: pour faire adopter une option, les politiques doivent rendre cette option la plus attractive entre toutes. Tandis que les subventions agroenvironnementales cherchent à compenser le coût d'opportunité, le budget nécessaire pour respecter ces contraintes de participation peut être plus élevé que le coût d'opportunité. Si les décideurs pouvaient dicter quoi faire aux agriculteurs, alors ils pourraient les compenser exactement au niveau du coût d'opportunité, et le budget de la politique agroenvironnementales serait égal au coût d'opportunité. Étant donné le fait que le budget total est limité, un budget plus élevé réduit automatiquement la fourniture totale de services écosystémiques. Prendre en compte le processus de décision des agriculteurs et la façon dont les incitations économiques fonctionnent est donc crucial dans la conception des

politiques agroenvironnementales.
Un autre problème lié aux incitations et de choisir sur quoi elles sont basées. Par exemple, les décideurs ont le choix entre des incitations basées sur les actions et des incitations basées sur les résultats. Les incitations basées sur les actions compensent les agriculteurs sur la base des actions qu'ils mettent en œuvre, c'est-à-dire des pratiques agricoles. Les incitations basées sur les résultats payent les agriculteurs en fonction des résultats, c'est-à-dire des services écosystémiques qu'ils fournissent. La plupart des politiques agroenvironnementales à l'heure actuelle reposent sur des incitations basées sur les actions, par exemple les Mesures Agroenvironnementales de la Politique Agricole Commune. Les incitations basées sur les résultats correspondent aux solutions théoriquement efficientes pour résoudre le problème des biens publics fournis en quantités insuffisantes (Mas-Colell et al., 1995), et ont gagné en attrait ces dernières années, notamment pour la biodiversité (Musters et al., 2001; Schwarz et al., 2008). Le choix des incitations peut influencer le budget dédié à la politique agroenvironnementale, en particulier si les incitations dépendent les unes des autres. Les incitations basées sur les résultats sont susceptibles de s'influencer les unes les autres (Bryan and Crossman, 2013; Huber et al., 2017) et de créer des problème pour les calibrer. Peu de réponses existent sur cette question: les études théoriques en économie ne prennent pas en compte les interactions complexes entre services écosystémiques, tandis que les études considérant ces interactions ne concluent pas sur les conséquences de ces interactions en termes de budget des politiques agroenvironnementales. Les interactions entre les services écosystémiques sont susceptible de diminuer l'efficience théorique des incitations basées sur les résultats par rapport aux incitations basées sur les actions.

Le quatrième enjeu est celui de l'hétérogénéité. Les services écosystémiques fournis, les coûts associés et le budget de la politique agroenvironnementale varient spatialement, ou selon le moment ou les agriculteurs. Cette hétérogénéité rend la détermination des options coût-efficientes plus compliquées. Les incitations ne peuvent pas être calibrées pour chaque cas particulier, un certain degré d'uniformité est inévitable, qu'il soit dû aux asymétries d'information ou non. Deux solutions existent: réduire la variabilité ou adopter des incitations qui s'adaptent mieux à cette hétérogénéité. Réduire la variabilité passe par la définition et la calibration des incitations à une échelle spatiale plus fine, et plaide pour une conception participative ou le recours à des mécanismes qui incitent les agriculteurs à révéler les coûts et les services écosystémiques fournis. D'un autre côté, concevoir des politiques agroenvironnementales qui s'adaptent à l'hétérogénéité passe par le recours à des "self-screening contracts" ou des incitations complexes. En particulier, les incitations basées sur les résultats sont souvent citées comme étant capable de sélectionner les options coût-efficientes en présence d'hétérogénéité (Gibbons et al., 2011). Néanmoins, cet argument repose sur des analyses qui n'incluent pas de multiples services écosystémiques ni leurs interactions, qui pourraient changer ces conclusions.

Il reste un point non-traité par la litérature qui concerne les conséquences des interactions entre services écosystémiques sur la régulation de leur fourniture en agriculture.

S'attaquer à ces questions requiert l'interdisciplinarité dans la recherche, notamment en tissant des liens entre des concepts et outils économiques et la recherche en agroécologie. Les approches actuelles en économie à propos des services écosystémiques reposent souvent sur des hypothèses simplistes concernant le fonctionnenement des écosystémiques (Derissen and Quaas, 2013; Hasund, 2013; White and Hanley, 2016). D'un autre côté, les études agroécologiques passent souvent à côté des concepts économiques de base, comme l'existence de coût ou leur définition précise, ou l'importance des instruments économiques utilisés pour mettre en œuvre les politiques agroenvironnementales.

Cette thèse s'intéresse aux incitations économiques capables pour accroître la fourniture de services écosystémiques non-marchands, et se concentre plus particulièrement sur l'existence de multiple SE et leurs interactions. Elle se décline en différentes questions de recherches qui répondent aux quatre défis:

1. en présence de services écosystémiques multiples et interdépendants, comment déterminer quelles options maximisent la fourniture de services écosystémiques sous contrainte de budget

2. comment mettre en œuvre ces options avec des incitations économiques, étant donné les interactions entre services écosystémiques

3. que change l'hétérogénéité aux conclusions des questions précédentes

Cette thèse est pincipalement ancrée en économie, mais elle est aussi interdisciplinaire. Les questions de recherche au cœur de cette thèse sont donc traitées selon différentes perspectives. Premièrement, vu de la théorie microeconomic, ces questions soulèvent des problèmes liés à la production jointe et à la régulation des biens publics. Nous utilisons des outils conceptuels issus de la théorie microéconomique pour les explorer de manière théorique. Deuxièmement, nous utilisons une approche de modélisation plus appliquée pour simuler les services écosystémiques fournis par différentes options de gestion ainsi que les coûts associés, et les incitations économiques nécessaires pour faire appliquer ces options de gestion par les agriculteurs.

Les contributions de cette thèse se situent donc au carrefour de l'agroécologie et de l'économie. La première contribution est en économie théorique, elle est d'étudier comment la production jointe complique la régulation des biens publics. La production jointe désigne l'existence d'interactions entre les produits d'un processus de production, et nous appliquons ce concept aux services écosystémiques. Beaucoup d'analyses théoriques existent à propos de la fourniture de biens publics en quantité insuffisantes, et sur la manière de résoudre ce problème avec des incitations économiques et atteindre l'optimum économique. Certaines analyses s'intéressent aux conséquences de la production jointe sur l'optimum économique. Les rares analyses qui combinent les deux sujets considèrent des types de production jointe spécifiques, qui ne reflètent pas la variété des types de production jointe qui existent entre les services écosystémiques. Nous explorons les conséquences des différents types de production jointe sur la régulation des biens publics.

Deuxièmement, nous appuyons notre analyse des stratégies coût-efficientes pour fournir des services écosystémiques sur une représentation plus précise des interactions entre services écosystémiques et du coût de cette fourniture. Cela représente une contribution au regard de l'économie appliquée. La littérature existante a tendance à inclure les interactions entre services écosystémiques d'une manière très simple (Derissen and Quaas, 2013; Hasund, 2013; White and Hanley, 2016).

Troisièment, cette thèse apporte des éléments économiques solides à la littérature agroécologique sur la maximisation des services écosystémiques. Nous utilisons les outils de l'analyse d'efficience comme moyen d'identifier les options qui maximisent les services écosystémiques. Surtout, nous raffinons les analyses actuelles des politiques agroenvironnementales en intégrant des concepts économiques utiles comme le coût d'opportunité et la prise en compte explicite des contraintes de participation.

Bien qu'ancrée en économie, l'analyse présentée dans cette thèse considère les contraintes agroécologiques comme une base pour l'analyse économique. Dans ce sens, elle est apparentée à l'économie écologique: les processus agroécologiques sont vus comme des contraintes qui déterminent les processus économiques. Notre modélisation agroécologique possède plusieurs caractéristiques originales. Elle est basée sur la simulation des processus agroécologiques, et considère les pratiques agricoles (plutôt que l'usage des sols) comme déterminant clé de la fourniture de services écosystémiques. Les interactions entre services écosystémiques ne sont pas spécifiées a priori, elles émergent des processus agroécologiques simulés. De manière similaire, la relation entre les services écosystémiques fournis et le coût n'est pas spécifiée par une fonction. Nous nous gardons également d'estimer ces relations, ce qui préserve leur côté non-lisse et participe à leur réalisme. Dans l'analyse appliquée, la modélisation économique est simple, mais elle met en valeur l'importance d'identifier le statu quo (la situation en l'absence de politique agroenvironnementale) pour déterminer le coût de la fourniture des services écosystémiques, et l'importance des contraintes de participation. Une autre originalité de cette thèse est d'articuler différents niveaux d'analyse. L'analyse théorique ne permet pas de conclure, et nous faisons appel à une modélisation plus appliquée pour approfondir l'analyse dans un deuxième temps, pour enfin mettre nos résultats en regard avec les recommandations issues de l'analyse des politiques agroenvironnementales.

Cette thèse est organisée comme suit: le chapitre 2 passe en revue la littérature agroécologique sur la quantification et la représentation de multiples services écosystémiques; le chapitre 3 présente ce que la littérature et les concepts économiques théoriques nous apprennent sur la régulation des biens publics joints; le chapitre 4 et 5 présentent respectivement le modèle et les données simulées utilisée dans l'analyse appliquée. Le chapitre 6 utilise ces données pour explorer la manière de définir les options qui maximisent les services écosystémiques pour un budget total limité, tandis que le chapitre 7 questionne les moyens de mettre en œuvre une option avec des incitations économiques. Le chapitre 8 introduit l'hétérogénéité et ses effets sur la définition et la mise en œuvre des options qui maximisent les services écosystémiques sous contrainte de budget. Enfin, le chapitre 9 discute les résultats à la lumière des critiques et recommandations à propos des politiques agroenvironnementales actuelles.

Ce résumé présente un condensé de chaque chapitre dans les sections suivantes.

2. Chapitre 2: État de l'art

Les agroécosystèmes founissent de nombreux services écosystémiques, et pour concevoir des politiques agroenvironnementales il est nécessaire de quantifier ces services écosystémiques et de les représenter. Un des moyens de représenter les multiples services écosystémiques et leurs interactions est l'utilisation de frontières de possibilités de production.

De nombreuses évaluations de services écosystémiques existent, que ce soit pour les cartographier, ou pour évaluer les impacts de différents scénarios d'usage des sols ou de politiques publiques. La plupart sont basées sur des modèles, notamment en ce qui concerne l'évaluation de scénarios. Ces modèles sont plus ou moins détaillés, et les déterminants qu'ils considèrent également.

2.1. Simulation de nombreux services écosystémiques

Beaucoup de modèles considèrent comme variables d'entrée les usages ou la couverture des sols pour évaluer les services écosystémiques en agriculture. Le degré de complexité varie de simples fonctions de bénéfices (Chan et al., 2006) à des modèles statistiques plus complexes (Bateman et al., 2013) et des modèles reproduisant les processus biophysiques (Nelson et al., 2009; Goldstein et al., 2012). Cependant, ces modèles basés sur l'usage des sols sont incapables de prendre en compte des déterminants plus fins des services écosystémiques et négligent souvent les dynamiques et les interactions spatiales.

D'autres modèles considèrent des déterminants plus détaillés comme les pratiques agricoles, avec des modèles intégrés plus ou moins complets et complexes (Bekele et al., 2013; Kragt and Robertson, 2014; Balbi et al., 2015; Groot et al., 2012). Il est aussi courant de recourir à des couplages de modèles existants qui permettent une grande complexité (Schönhart et al., 2011; Kirchner et al., 2015). L'utilisation de modèles intégrés est cependant préférable (Carpenter et al., 2009).

2.2. Les interactions entre services écosystémiques

Deux types d'interactions entre services écosystémiques sont généralement distingués: les synergies et les antagonismes. Les définitions exactes varient selon le type d'analyse, mais une synergie désigne une relation positive entre deux services écosystémique, et un antagonisme une relation négative.

Les frontières de possibilité de production sont un outil qui permet de représenter les relations entre services écosystémiques dans un grand nombre d'alternatives (scenarios d'usage des sols, de politique publique etc.).

2.3. Les frontières de possibilités de production appliquées aux services écosystémiques en agriculture

L'utilisation de frontières de possibilités de production a plusieurs intérêts. Premièrement, cela permet d'identifier les impacts de plusieurs options en termes de services écosystémiques (Kragt and Robertson, 2014). Deuxièmement, les frontières permettent de représenter la forme des interactions entre services écosystémiques, et de faire le lien avec la théorie économique. Troisièmement, représenter cette frontière permet d'identifier les bouquets qui maximisent les services écosystémiques et les solutions correspondantes. Quatrièmement, ces frontières sont utiles pour calculer le coût en termes de production agricole lié à une augmentation d'un autre services écosystémique. Enfin, grâce au coût d'opportunité, il est possible de déterminer la force des incitations à mettre en œuvre pour pousser les agriculteurs à augmenter la fourniture de services écosystémiques non-marchands.

Une des limites de l'utilisation de frontières de possibilités de production est qu'elles ne permettent pas de considérer les irréversibilités ou les effets de seuil.

2.4. Faits stylisés identifiés par la littérature sur les interactions entre services écosystémiques

La littérature identifie globalement un antagonisme marqué entre la production agricole et les services écosystémiques non-marchands. Cependant, de nombreux auteurs soulignent qu'un accroissement important des services écosystémiques non-marchands est possible pour une diminution mineure de la production agricole. Au sein des services écosystémiques non-marchands, beaucoup de synergies apparaissent, mais des antagonismes existent également, si bien que ces services écosystémiques ne sont pas bien corrélés les uns aux autres.

Enfin, en comparant les différentes études, deux conclusions apparaissent: la frontière de possibilités de production doit inclure toutes les options disponibles pour être représentative, et la définition des échelles spatiales et temporelles est importante. Les interactions entre services écosystémiques varient selon l'horizon temporel et l'échelle spatiale considérés.

Néanmoins, les frontières de possibilités de production présentent beaucoup d'intérêt, notamment parce qu'elles sont faciles à relier à l'analyse économique.

3. Chapitre 3: Régulation de biens publics joints dans la théorie économique

Un agroécosystème peut être vu à la manière d'une unité de production de services écosystémiques à partir de surface agricole. La fourniture de services écosystémiques non-marchands en agriculture fait écho à deux enjeux en microéconomie théorique.

3.1. Les services écosystémiques en tant que biens publics joints

Certains services écosystémiques sont des biens publics: ils bénéficient à plus d'agents que ceux qui en supportent les coûts. Cette caractéristique fait qu'ils sont généralement fournis en quantités insuffisantes par rapport à l'optimum économique qui maximiserait le bien-être global de la société. Par conséquent, une régulation est nécessaire.

Le deuxième enjeu rencontré par les services écosystémiques est qu'ils sont interdépendants, ce qui renvoie à la notion de production jointe en économie. La quantité d'un service écosystémique dépend de la quantité des autres services écosystémiques.

Ces deux notions combinées expliquent par exemple le déclin de beaucoup de services écosystémiques de régulation: leur fourniture ne rapporte pas directement à celui qui en supporte le coût, et ils sont en antagonisme avec la production marchande.

Les hypothèses généralement faites en microéconomie conduisent à considérer les antagonismes concaves comme la règle générale de production jointe, mais d'autres interactions entre produits peuvent exister, comme des synergies, des antagonismes convexes, etc. Le type production jointe a un impact sur l'optimum économique. Elle impose des contraintes supplémentaires du côté de l'offre (le producteur ne peut pas choisir les niveaux des produits de manière indépendante), qui se répercutent sur la demande. Peu de travaux théoriques combinent ces deux enjeux. Cependant, Holmstrom (1999) note que si deux produits sont joints, et que l'un des deux n'est pas facilement mesurable, alors réguler l'autre peut servir à réguler le premier.

Si les travaux théoriques sont incomplets, nous retenons deux conclusions :

1. la production jointe, en particulier la forme de la frontière de possibilité de production, impose des contraintes supplémentaires du côté de la demande. Ce point est important: la frontière de possibilités de production résulte de processus agroécologiques et n'a aucune raison de correspondre à des hypothèses microéconomiques. Par conséquent, l'analyse économique doit considérer explicitement les processus agroécologiques comme point de départ

2. l'offre d'un produit joint dépend du prix de tous les autres produits joints. Ce point est important pour leur régulation

3.2. La régulation des biens publics joints

La littérature économique identifie généralement les incitations basées sur les résultats (ou basées sur les produits du processus de production) comme la solution optimale pour résoudre leur fourniture insuffisante. Ces incitations donnent un prix (ou le corrigent) aux biens publics, et ce faisant changent le prix relatif par rapport au bien privé. Ce résultat repose sur l'hypothèse que la frontière des possibilités de production est concave et complète. Dans ce cas, chaque prix relatif correspond à un unique bouquet de produits, qui est efficient. Cependant, la forme de la frontière des possibilités de production est déterminée par les processus écologiques et ne correspond pas forcément à cette hypothèse.

Les incitations basées sur les actions sont un autre type d'incitations, qui rémunèrent directement les processus de production qui fournissent des biens publics. En pratique, elles correspondent à la logique des Mesures AgroEnvironnementales de la PAC, ou à celle des taxes sur les pesticides ou les fertilisants. Pour cibler un bouquet efficient, le régulateur doit connaître le lien entre les actions et les résultats, ce qui n'est pas le cas avec les incitations basées sur les résultats.

Dans la pratique, les frontières de possibilités de production entre services écosystémiques sont rarement conformes aux hypothèses faites. En lien avec la premire conclusion identifiée dans la littérature, les interactions entre SE imposent des contraintes supplémentaires sur la demande, ce qui modifie le fonctionnement des incitations basées sur les résultats. Nous illustrons ce point avec deux types de production jointe autres que l'antagonisme concave. Premièrement, dans le cas d'une synergie, la frontière des possibilités de production se réduit à un arc étroit, et le changement de prix relatif créé par les incitations basées sur les résultats n'a que peu d'impact. Les incitations basées sur les résultats ciblent une option efficiente, mais ne prennent pas en compte que certains bouquets n'existent pas. Deuxièmement, la relation entre 2 services écosystémiques peut être un antagonisme convexe, la frontière de possibilités de production étant à un endroit courbée vers l'intérieur. Dans ce cas, il est possible qu'un même prix relatif corresponde à deux bouquets de services écosystémiques différents, et les incitations basées sur les résultats peuvent amener un agriculteur à choisir un bouquet différent et inférieur à celui visé. Les incitations basées sur les actions ne rencontrent pas ce problème.

Outre les différentes formes de production jointe, un autre aspect est ignoré dans la littérature théorique. Le cadre conceptuel présenté plus haut suppose que tous les bouquets sont obtenus pour un coût identique. En réalité, il est difficile de ne comparer que des bouquets obtenus au même coût, et il est utile de considérer des bouquets ayant des coûts différents. Dans ce cas, la régulation des biens publics devient plus compliquée. Nous faisons l'hypothèse que l'agriculteur cherche à maximiser son profit. La contrainte de participation impose que pour être fourni par un agriculteur, un bouquet de services écosystémiques doit être le plus rentable d'entre tous. Quand le coût de production de chacun des bouquets est identique, alors leur profit dépend uniquement de leur niveaux de services écosystémiques et du prix relatif. La somme des incitations nécessaires est égales à la différence de profit avec le bouquet initial (le coût d'opportunité). Quand le coût de production varie entre les différents bouquets, alors le profit ne dépend pas seulement des niveaux de services écosystémiques et du prix relatif, mais aussi du coût de production. Dans le cas des incitations basées sur les résultats, il n'est plus garanti que le changement de prix relatif suffise à rendre un bouquet efficient le plus rentable entre tous. La somme des incitations nécessaires à le rendre le plus rentable peut être supérieure au coût d'opportunité. La contrainte de participation amène à distinguer le coût d'opportunité du budget nécessaire au régulateur pour faire changer le bouquet de services écosystémiques fourni, et à distinguer l'optimum de premier rang (défini par le coût d'opportunité, sans prendre en compte la contrainte de participation) de l'optimum de second rang (défini par rapport au budget, en prenant en compte la contrainte de participation). Cet aspect concerne également les incitations basées sur les actions.

3.3. Contributions de cette thèse

Trois aspects sont négligés dans la littérature actuelle, et cette thèse vise à les compléter.

Premièrement, les études actuelles qui traitent des biens publics et de la production jointe reposent sur des formes fonctionnelles très simplifiées en ce qui concerne les interactions entre produits. Notre approche repose sur une modélisation agroécologique relativement réaliste, qui permet de représenter des interactions multiples et complexes entre services écosystémiques, ainsi que des relations non-linéaires avec le coût de production.

Deuxièmement, certaines études en agroécologie ignorent la contrainte de participation. Notre approche repose sur une modélisation économique simple, mais elle prend en compte explicitement la contrainte de participation.

Troisièmement, nous introduisons la présence d'hétérogénéité.

4. Chapitres 4 et 5: Modèle et données simulées

Pour l'analyse appliquée, nous utilisons des données simulées par un modèle agroécologique et économique. Le but de ce modèle est de simuler l'impact d'une large gamme de pratiques agricoles sur différents services écosystémiques, et le profit associé. Ce modèle considère les pratiques agricoles comme les déterminants de la fourniture de services écosystémiques et représente les processus agroécologiques de manière stylisée. Il n'est pas très réaliste aux yeux des agroécologues, mais constitue un essai pour mieux prendre en compte les processus agroécologiques au sein d'une analyse économique. Notamment, il laisse les interactions entre services écosystémiques émerger du fonctionnement du modèle, sans spécifier de forme fonctionnelle à priori.

Le modèle simule les services écosystémiques fournis sur un territoire homogène, pendant une période. Il représente un système en grandes cultures et prairie.

4.1. Services écosystémiques inclus

Le modèle inclus les services écosystémiques les plus importants dans les agroécosystèmes, et qui ne dépendent pas de la structure spatiale du paysage ou des caractéristiques hydrogéologiques ou du relief. Nous avons retenu les services écosystémiques suivants:

- la production agricole, agrégeant différents produits (grain, paille, fourrage etc.), exprimée en valeur monétaire
- le potentiel de pollinisation, exprimé en indice, qui représente la capacité du territoire à fournir un service de pollinisation aux cultures qui en ont besoin
- la qualité de l'eau, mesurée par un indice capturant la quantité de polluants d'origine agricole potentiellement lessivés vers les masses d'eaux (nitrates, résidus de pesticides, particules organiques)
- la régulation du climat, en tonnes d'équivalent CO_2, mesurée par les émissions nettes de gaz à effets de serre issus de la combustion de carburants, de la dénitrification, de la minéralisation de la matière organique du sol et des rejets animaux
- la fertilité du sol, exprimée en tonnes comme l'évolution du stock de matière organique du sol d'une année sur l'autre.

4.2. Pratiques agricoles inclues

Les pratiques agricoles sont considérées à travers leurs combinaisons, que nous appelons des options de gestion. Les pratiques agricoles inclues sont les suivantes:

- l'usage du sol (prairie ou culture). La prairie est une option de gestion en soi, et exclut les autres pratiques agricoles ci-dessous. La culture peut être plus ou moins intensive, mais représente des options dédiées à la production de biomasse. La prairie représente une option plus extensive, sans intrants ni travail du sol. Le modèle permet d'inclure ou non les impacts de l'élevage lié à la prairie.

- l'intensité en pesticides (3 niveaux y compris absence de pesticides). Les pesticides réduisent les ravageurs des cultures et augmentent la production agricole, mais ils impactent les pollinisateurs et la qualité de l'eau.
- l'intensité en fertilisants minéraux (5 niveaux y compris absence de fertilisation). Les fertilisants augmentent l'azote minéral et donc la production, mais ils dégradent la qualité de l'eau et augmentent les émissions de gaz à effet de serre (via la dénitrification).
- le travail du sol (conventionnel / réduit). Le travail du sol conventionnel permet un taux de minéralisation de la matière organique plus élevé, ce qui augmente la production, mais contribue à de plus fortes émissions de gaz à effet de serre et une moindre qualité de l'eau (érosion et lessivage de particules organiques).
- habitats semi-naturels (présence / absence). Les habitats semi-naturels favorisent les pollinisateurs et améliorent la qualité de l'eau, mais empiètent sur la surface cultivée et réduisent la production.
- restitution des résidus de culture (oui / non). La restitution des résidus de culture augmente la matière organique qui retourne au sol, et limite l'érosion. Elle a donc un effet positif sur la fertilité du sol, la régulation du climat et la qualité de l'eau, mais réduit la production agricole car les pailles laissées sur place ne sont pas vendues.

Ces pratiques agricoles donnent 121 combinaisons en tout. Elles font varier les paramètres du modèle.

4.3. Déterminants exogènes

Le modèle inclut 2 paramètres exogènes: la qualité du sol et le stock initial de matière organique. La qualité du sol est exprimée par un rendement potentiel et représente toutes les caractéristiques qui ne peuvent pas être influencées par les pratiques agricoles (composition minérale du sol, pente, climat etc.). Le stock de matière organique du sol dépend des pratiques agricoles passées, mais son stock initial est exogène dans nos simulations.

En faisant varier les paramètres exogènes, il est possible de considérer de nombreux contextes agronomiques. Pour en considérer un nombre réduit, et qui correspondent à des situations réalistes, nous supposons que le stock initial de matière organique du sol reflète les décisions passées d'un agriculteur qui cherche à maximiser son profit, et dépend de la qualité du sol. Sur un sol à haut rendement potentiel, l'agriculteur a intérêt à adopter des pratiques intensives, notamment un travail du sol conventionnel et l'export des résidus de culture, ce qui fait diminuer le stock de matière organique au cours du temps et abouti à un stock bas. Au contraire, sur les sols les moins productifs, l'option la plus rentable est la prairie, qui est associée à un stock de matière organique élevé.

4.4. Jeu de données

Nous faisons tourner le modèle à l'aide du logiciel Matlab. Nous sélectionnons 10 contextes agronomiques (combinaisons de qualité du sol et de stock initial de matière organique du sol). Pour chaque contexte agronomique, nous simulons les 5 services écosystémiques et le profit associé à chacune des 121 options disponibles. Ces données simulées sont celles qui servent ensuite à l'analsyse appliquée.

La conception de politiques pour la fourniture de services écosystémiques non-marchands comprend deux étapes

1. la définition de la solution optimale que la politique vise à mettre en œuvre
2. la mise en œuvre de cette solution à travers des instruments

Les deux prochains chapitres suivent ces étapes: le chapitre 5 explore les solutions qui maximisent la provision de services écosystémiques non-marchands, et le chapitre 6 traite de leur mise en œuvre par des incitations économiques.

5. Chapitre 6: Maximiser la fourniture de services écosystémiques pour un budget donné

Ce chapitre explore comment identifier les solutions efficientes et coût-efficientes pour fournir des services écosystémiques. Tout d'abord, nous étudions les solutions qui maximisent la fourniture de services écosystémiques (efficience). L'identification de ces solutions informe également sur les synergies et antagonismes entre services écosystémiques. Dans un deuxième temps, nous introduisons le coût lié à la fourniture de ces services écosystémiques et nous étudions les solutions coût-efficientes (qui maximisent les services écosystémiques et minimisent le coût).

Les principales contributions de ce chapitre sont

- de montrer l'intérêt de l'analyse d'efficience pour étudier les interactions entre services écosystémiques et déterminer les options qui maximisent de multiples services écosystémiques en interactions les uns avec les autres.
- de fournir une définition consistente du coût de la fourniture de services écosystémiques, qui prend en compte les interactions entre les services écosystémiques; et de l'utiliser pour déterminer les options coût-efficientes
- de proposer une méthode pour comparer le coût de deux stratégies pour fournir des services écosystémiques non-marchands: i) une augmentation modeste des services écosystémiques sur tout le territoire, et ii) une augmentation importante des services écosystémiques ciblée sur une partie du territoire

Ce chapitre repose sur les données agroécologiques simulées à l'aide du modèle. Le jeu de données décrit les bouquets de 5 services écosystémiques et le profit associés à 121 options de gestion différentes, sur 10 contextes agronomiques.

5.1. Frontières de possibilité de production et interactions entre services écosystémiques

Les frontières de possibilités de production (FPP) représentent les interactions entre services écosystémiques, et leur forme donne des informations sur le type d'interactions. On peut distinguer 3 formes de FPP qui correspondent à trois types d'interactions entre services écosystémiques: synergie, antagonisme concave, antagonisme convexe, illustrés sur les figures 6.1–6.3.

Pour déterminer le type d'interactions entre SE, trois caractéristiques de la frontière sont importantes: sa longueur, sa pente et sa courbure.

La longueur de la FPP indique si les services écosystémiques sont en synergie (frontière courte) ou en antagonisme (frontière longue). Dans le cas d'un antagonisme, **la pente** de la FPP indique sa force, et **la courbure** le type de stratégie qui maximise la fourniture de SE. Dans le cas d'une frontière concave (courbée vers l'extérieur), les bouquets intermédiaires maximisent la fourniture de SE, alors que dans le cas d'une frontière convexe (courbée vers l'intérieur), les combinaisons linéaires de bouquets extrêmes sont les stratégies qui maximisent la fourniture de SE. La courbure de la frontière est l'argument utilisé dans le débat land-sparing/land-sharing (Green et al., 2005; Phalan et al., 2011), mais il repose sur l'hypothèse que les combinaisons linéaires de tous les bouquets sont possibles, et donc sur l'absence d'interactions spatiales ou de dynamiques. Par conséquent, cet argument ne devrait être utilisé qu'à des échelles de temps ou d'espaces qui respectent cette hypothèse.

5.2. Utilisation des bouquets efficients pour la détermination des interactions entre SE

Les bouquets de SE qui appartiennent à la frontière sont efficients au sens de Pareto: aucun autre bouquet ne fournit plus de tous les SE à la fois. Ils identifient les options qui maximisent la fourniture de SE et résument les interactions complexes entre les SE.

Le nombre et la position des bouquets efficients permettent aussi de déterminer le type d'interactions entre SE, sans même avoir besoin de tracer ou d'estimer la frontière elle-même. Au sein des bouquets efficients, augmenter un service écosystémique requiert d'en diminuer un autre. Par conséquent:

- un grand nombre de bouquets efficients indique un antagonisme concave
- un petit nombre de bouquets efficients, dont les coordonnées sont proches indique une synergie
- un petit nombre de bouquets efficients dont les coordonnées sont éloignées indique un antagonisme convexe

5.3. Analyse d'efficience

Nous utilisons les outils de l'analyse d'efficience, et en particulier la Data Envelopment Analysis (DEA) pour analyser les bouquets de SE efficients dans notre jeu de données. Ces outils viennent de la théorie économique de la production. Le principe de cette analyse est de déterminer quelle option de gestion fournit le plus de SE pour une surface donnée.

Cette analyse se présente sous la forme d'une optimisation linéaire (voir l'équation 6.1). Cette analyse cherche la plus grande augmentation possible des services écosystémiques pour une surface agricole donnée. Elle identifie les bouquets efficients (pour lesquels l'augmentation possible est nulle) et associe à chaque bouquet inefficient un score d'inefficacité β_j qui représente l'augmentation maximale de SE en pourcentage, et la référence efficiente, c'est-à-dire la combinaison linéaire de bouquets qui permet cette augmentation.

Avec ces outils, nous exécutons deux analyses:

- une analyse avec les 5 services écosystémiques
- une analyse avec les quatre services écosystémiques non-marchands

Ces deux analyses nous permettent d'identifier les résultats suivants:

1. Seuls quelques bouquets de SE sont efficients avec les 5 services écosystémiques, ce qui souligne que beaucoup d'options sont inefficientes et que cibler des options au hasard conduit probablement à des bouquets de SE inefficients
2. En ne considérant que les 4 services écosystémiques non-marchands, seuls deux bouquets sont efficients, ce qui souligne les fortes synergies entre eux, et le fait que des solutions communes pour les fournir existent
3. Tandis que le nombre de bouquets efficients est limité, ils couvrent une large gamme de niveaux de SE, ce qui illustre que beaucoup de compromis différents sont possibles, que la relation entre les SE est plutôt concave, et que concilier la production agricole avec les services écosystémiques non-marchands, au sein d'une même parcelle, peut être une solution efficiente.

Les bouquets efficients (avec les 5 services écosystémiques) sont d'autant plus nombreux dans les contextes où le rendement potentiel est élevé.

5.4. Bouquets de SE coût-efficients

Dans un contexte où la surface agricole est limitante, l'analyse précédente fait sens: elle cherche à maximiser les services écosystémiques sur une surface donnée. Cependant, fournir les services écosystémiques non-marchands a un coût, et le facteur limitant pour la fourniture de services écosystémiques est probablement le budget dédié aux politiques agroenvironnementales.

Plus précisément, le coût de la fourniture de services écosystémiques est un coût d'opportunité, qui comprend non seulement le coût lié au changement de pratiques agricoles, mais aussi le coût lié à la perte de rendement. Nous faisons l'hypothèse que les agriculteurs choisissent l'option de gestion qui maximisent leur profit. Le bouquet de services écosystémiques fourni dans le statu quo (la situation qui prévaut en l'absence de politiques agroenvironnementales) correspond à l'option la plus rentable. N'importe quel changement dans les services écosystémiques fournis a un coût. Ce coût est soit supporté par l'agriculteur, soit par l'ensemble de la société s'il est compensé par des subventions.

Nous mesurons le coût d'opportunité d'un bouquet de services écosystémiques comme la différence de profit avec le bouquet fourni par l'option la plus rentable (le statu quo). Cette définition permet i) de tenir compte des interactions entre services écosystémiques: nous ne définissons pas le coût d'opportunité de la fourniture d'un service écosystémique pris individuellement ; et ii) de prendre en compte le coût d'opportunité dans son ensemble (coût du changement de pratiques et de la perte de rendement). Nous calculons ce coût d'opportunité pour tous les bouquets de services écosystémiques de notre jeu de données, et l'utilisons pour mener une analyse de coût-efficience.

L'analyse de coût-efficience consiste à maximiser les services écosystémiques non-marchands pour un coût d'opportunité donné. De la même manière que pour l'analyse d'efficience, c'est un problème d'optimisation linéaire (voir équation 6.2). Cette analyse identifie entre 2 et 13 bouquets coût-efficients selon les contextes agronomiques. Premièrement, les bouquets coût-efficients sont plus nombreux que les bouquets écologiquement efficients (qui maximisent les quatre services écosystémiques non-marchands): tous les bouquets écologiquement efficients sont coût-efficients, mais étant donné l'antagonisme entre coût d'opportunité et services écosystémiques non-marchands, certains bouquets sont coût-efficients même s'ils ne maximisent pas les quatre services écosystémiques non-marchands. Deuxièmement, les bouquets coût-efficients sont similaires aux bouquets efficients (avec les cinq services écosystémiques), même s'ils sont moins nombreux. Troisièmement, l'analyse de coût-efficience montre que certains bouquets efficients (avec les cinq services écosystémiques) ne sont pas coût-efficients. Il s'agit principalement de bouquets correspondant à des options très intensives. Dernièrement, les bouquets coût-efficients couvrent une large gamme de niveaux de services écosystémiques, ce qui indique qu'une augmentation des services écosystémiques non-marchands peut être obtenue à un coût faible.

Cette analyse de coût-efficience compare des bouquets qui fournissent des proportions de SE très différentes. Sans spécifier des préférences, il est impossible de trancher entre les bouquets coût-efficients.

5.5. Comparaison du coût de deux stratégies pour augmenter les SE non-marchands

L'analyse d'efficience avec quatre services écosystémiques non-marchands détermine pour chaque bouquet écologiquement inefficient le score d'inefficacité et la référence écologiquement efficiente de ce bouquet. Cette référence fournit les mêmes proportions de services écosystémiques non-marchands, mais à des niveaux supérieurs. Nous comparons le coût de deux stratégies pour fournir un bouquet donné de services écosystémiques non-marchands: i) **saupoudrage** : une option donnée qui fournit plus de SE non-marchands que le statu quo, adoptée sur toute la surface, et ii) **concentration** : sa référence écologiquement efficiente, adoptée sur une fraction de la surface.

Les résultats montrent qu'aucune des deux stratégies ne domine l'autre. Les résultats dépendent du contexte agronomique, et de l'option considérée. Ils permettent de conclure que le saupoudrage n'est pas nécessairement une mauvaise solution dans les paysages intensifs.

5.6. Conclusion

Ce chapitre permet d'identifier quelques recommandations utiles. Premièrement, les résultats soulignent l'importance de bien cibler les options: beaucoup d'entre elles fournissent des bouquets de SE inefficients. Deuxièmement, nous montrons que les résultats changent selon le rendement potentiel, ce qui appelle des politiques différenciées. Troisièmement, il est crucial de prendre en compte le coût de la fourniture de services écosystémiques non-marchands.

6. Chapitre 7: Incitations économiques

Les politiques agroenvironnementales actuelles sont basées en grande partie sur des normes et des incitations économiques, principalement des subventions. La plupart des subventions actuelles sont basées sur les actions, c'est-à-dire les pratiques agricoles, mais des études récentes plaident pour des subventions basées sur les résultats, c'est-à-dire les services écosystémiques réellement fournis. Dans ce chapitre, nous explorons quelles incitations permettent l'adoption d'options de gestion.

6.1. Méthodes

Nous utilisons le jeu de données simulées et les analyses d'efficience du chapitre précédent. Nous supposons qu'un agriculteur choisit l'option la plus rentable, et donc les subventions peuvent changer l'option choisie par un agriculteur en rendant cette option la plus rentable parmi toutes les options disponibles.

Dans le chapitre précédent, nous nous sommes intéressés au coût d'opportunité d'une option. Ici nous nous intéressons le budget minimal pour atteindre une option (c'est-à-dire la somme des subventions qu'un agriculteur doit recevoir afin que cette option soit la plus rentable et qu'il l'adopte). Le coût d'opportunité représente la borne inférieure du budget, mais le budget peut être supérieur au coût d'opportunité. La plupart du temps, le régulateur n'exige pas de l'agriculteur qu'il choisisse une option en lui donnant le montant du coût d'opportunité. Il utilise un ensemble d'incitations économiques. Le problème est que ces incitations modifient le profit de toutes les options possibles. Pour atteindre une option, le régulateur doit respecter les contraintes de participation, c'est-à-dire que les incitations doivent rendre cette option la plus rentable entre toutes, ce qui peut faire diverger le budget (la somme des incitations) du coût d'opportunité.

Nous introduisons deux types d'incitations: des incitations basées sur les actions (un ensemble de 6 subventions conditionnées à l'adoption de pratiques agricoles) et des incitations basées sur les résultats (un ensemble de 4 subventions qui varient selon les niveaux de services écosystémiques non-marchands fournis). Nous simulons ensuite le budget minimal pour atteindre chaque option avec chacun des deux types d'incitations.

Ceci est un problème d'optimisation linéaire: pour chaque option, il s'agit de minimiser la somme des subventions sous contrainte que le profit de cette option demeure supérieur au profit des autres options. La solution est un ensemble d'incitations qui permet d'atteindre une option avec le plus petit budget possible. Il n'existe pas de solution pour toutes les options: pour certaines, il est impossible de respecter les contraintes de participation.

Dans le cas des incitations basées sur les résultats, toutes les options coût-efficientes et seules celles-ci peuvent être atteintes. Dans le cas des incitations basées sur les actions, seules les options qui représentent un progrès important en termes de pratiques ou ont un coût d'opportunité relativement faible peuvent être atteintes. Dans nos résultats, les incitations basées sur les actions atteignent un plus grand nombre d'options, mais elles ciblent moins bien les options coût-efficientes.

Les options coût-efficientes sont atteintes par les deux types d'incitations, et nous comparons le budget nécessaire pour chaque type d'incitation. Globalement, les incitations basées sur les résultats ont un budget bien supérieur à celui des incitations basées sur les actions. Pour atteindre une même option, le régulateur devra donner une somme plus grande à l'agriculteur s'il utilise des incitations basées sur les résultats. Globalement, les incitations basées sur les actions nécessite un budget peu supérieur au coût d'opportunité.

Si nos résultats confirment le résultat théorique que les incitations basées sur les résultats ciblent mieux les options coût-efficientes, ils montrent aussi que cela peut entraîner un budget important pour le régulateur.

Pour expliquer celà, nous revenons au rôle des interactions entre services écosystémiques. Le choix de l'ensemble d'incitations peut être séparé en deux étapes. La première est de trouver les proportions des différentes incitations qui rend l'option visée la plus rentable. La deuxième étape est d'ajuster le montant des incitations (à proportions données) pour que le budget soit au moins égal au coût d'opportunité. Cependant, les deux étapes ne sont pas indépendantes, et les interactions entre SE jouent sur la première. En effet, lorsque deux services écosystémiques sont en synergie, cela contraint les proportions des incitations pour changer l'ordre des options en termes de profit. Par conséquent, pour respecter les contraintes de participation, la solution est de jouer sur le montant des incitations, ce qui peut conduire à des budget élevés.

6.2. Conclusion

Dans ce chapitre, nous montrons que les incitations sur les résultats permettent de mieux cibler les options coût-efficientes, mais à un budget souvent beaucoup plus élevé, à cause des interactions entre services écosystémiques.

Ces résultats sont à mettre en perspective avec les études actuelles sur l'intérêt des incitations basées sur les résultats, dont les plus récentes soulignent d'autres écueils de ces incitations et proposent des solutions hybrides, telles que des enchères inversées où les projets sont sélectionnés sur la base des résultats escomptés, et la compensation est calculée selon le coût d'opportunité.

Nos résultats présentent des limites, et notamment le fait que notre modèle inclut des interactions entre les services écosystémiques, mais presqu'aucune entre les pratiques agricoles. Dans ce sens, nos résultats soulignent plus généralement que les incitations doivent être pensées en cohérence les unes avec les autres.

Une autre limite importante est que nous considérons des territoires homogènes, alors que dans les paysages agricoles, les services écosystémiques ainsi que le coût d'opportunité des différentes options varient de manière importante. Nous traitons cet aspect dans le prochain chapitre.

7. Chapitre 8 : Le rôle de l'hétérogénéité

Dans ce chapitre, nous introduisons de l'hétérogénéité au sein des territoires utilisés dans nos simulations. Nous considérons donc des combinaisons d'options au sein de ces territoires: chaque partie homogène peut être cultivée selon une option différente.

L'hétérogénéité modifie à la fois les solutions coût-efficientes, et leur mise en œuvre par les incitations.

7.1. Paysages coût-efficients sur des territoires hétérogènes

Dans notre analyse, l'hétérogénéité est introduite en considérant des territoires composés de deux zones qui correspondent à deux contextes agronomiques différents. Sur ces territoires, nous nommons paysage la combinaison de deux options - une sur chacune des parties homogènes du territoire. Nous ne représentons pas l'espace de manière explicite, et donc dans notre analyse le passage de l'échelle du champ à l'échelle du paysage se fait en autorisant de l'hétérogénéité dans les caractéristiques du territoires.

Il est probable que les solutions coût-efficientes sur un territoire hétérogène diffère des solutions coût-efficientes au niveau de la parcelle homogène. Ici, deux effets entrent en jeu. Le premier effet est uniquement lié au changement d'échelle: la combinaison de deux solutions coût-efficientes n'est pas forcément coût-efficiente, et par conséquent, même au sein d'un territoire homogène, rien ne garantit qu'on obtienne un paysage coût-efficient en adoptant des options coût-efficientes sur chacune des parcelles. Le deuxième effet lié est à l'hétérogénéité: étant donné que les services écosystémiques et surtout le coût d'opportunité varient avec le contexte agronomique, les solutions coût-efficientes au niveau du paysage dépendront de l'hétérogénéité.

Nous considérons 12 territoires hétérogènes différents constitués de deux zones, avec des moyennes et des écart-types variés en termes de rendement potentiel.

La première analyse est une analyse de coût-efficience. Pour éviter de considérer toutes les combinaisons possibles de deux options (121=14641 combinaisons), nous considérons seulement les combinaisons d'options coût-efficientes à l'échelle de la parcelle, ce qui nous donne entre 4 et 117 paysages différents pour un même territoire hétérogène. Parmi ces paysages (combinaisons d'options), nous menons une analyse de coût-efficience pour identifier lesquels sont coût-efficients. Nous menons la même analyse sur des territoires homogènes de rendement moyen identique.

Par la suite, nous comparons les résultats des analyses sur les territoires hétérogènes et homogènes et classons les paysages en quatre cas selon qu'ils sont coût-efficients ou non dans les paysages hétérogènes ou homogènes. Ces résultats nous informent sur les rôles respectifs de l'effet d'échelle et de l'hétérogénéité elle-même. Dans la majorité des cas, l'effet d'échelle rend les paysages coût-inefficients. Dans moins de 30% des cas, l'hétérogénéité seule conduit à un paysage coût-inefficient, et dans une proportion identique, l'hétérogénéité résout la coût-inefficience qui résulte de l'effet d'échelle. Entre 20 et 45% des paysages ne sont affectés ni par l'effet d'échelle ni par l'hétérogénéité, et les combinaisons d'options coût-efficientes donnent des paysages coût-efficients. Ces pourcentages varient selon les territoires.

En étudiant les différents types de paysages, les résultats montrent que pour les paysages composés de deux options d'intensités différentes, mais dont l'agencement suit le rendement potentiel (l'option la plus extensive sur la partie la moins productive du territoire, et l'option la plus intensive sur la partie la plus productive), l'hétérogénéité est plus souvent une solution qu'un problème. Au contraire, pour les paysages composés d'options différentes, mais dont l'agencement ne suit pas le rendement potentiel, l'hétérogénéité est souvent un problème.

7.2. Mise en œuvre par des incitations économiques

Un des problèmes de la mise en œuvre de politiques en présence d'hétérogénéité est que les instruments ont un certain degré d'uniformité, ils ne peuvent pas être différenciés pour chaque parcelle. Ceci ajoute des contraintes aux contraintes de participation, et est susceptible d'accroître l'écart entre le coût d'opportunité et le budget nécessaire à la mise en œuvre d'un paysage.

Pour chacun des paysages considérés dans l'analyse de coût-efficience, nous introduisons des incitations basées sur les actions et des incitations basées sur les résultats et simulons le budget nécessaire à l'adoption d'un paysage par des agriculteurs qui maximisent leur profit individuel. De la même manière qu'au chapitre précédent, ceci est un problème d'optimisation linéaire. Pour beaucoup de paysages, aucune solution ne peut être trouvée, il n'existe aucun ensemble d'incitations qui permet de faire adopter ce paysage.

Les résultats montrent que les deux types d'incitations ciblent à peu près les paysages coût-efficients, avec un léger avantage pour les incitations basées sur les résultats. Les incitations basées sur les actions atteignent quelques autres paysages qui ne sont pas coût-efficients.

En terme de budget nécessaire, la même différence qu'en absence d'hétérogénéité apparaît: le budget nécessaire avec des incitations basées sur les résultats est souvent bien supérieur au coût d'opportunité et au budget nécessaire avec les incitations basées sur les actions.

7.3. Conclusion

Nos résultats montrent que les solutions coût-efficientes à l'échelle du paysage ne peuvent pas être inférées directement à partir des solutions coût-efficientes à l'échelle de la parcelle. L'hétérogénéité complique la situation.

Nos résultats sur la mise en œuvre de paysages par des incitations économiques montrent que les incitations basées sur les résultats nécessitent un budget élevé, alors que les incitations basées sur les actions sont aussi performantes pour cibler les paysages coût-efficients.

Une des limites de notre approche est que nous ignorons les interactions spatiales, qui nécessitent des solutions ciblées spatialement et pourraient donner un avantage aux incitations basées sur les résultats.

8. Discussion et conclusion

Le point de départ de cette thèse est d'étudier les incitations économiques à même d'accroître la fourniture de services écosystémiques en agriculture. Cet enjeu résonne avec des questionnements sur la conception des politiques agroenvironnementales et a reçu beaucoup d'attention, en particulier sous la forme d'analyses des politiques actuelles. Nos résultats théoriques doivent être mis en perspective avec ces analyses.

Dans cette discussion, nous passons en revue les principales politiques et incitations économiques qui influencent les décisions des agriculteurs et la fourniture de services écosystémiques en agriculture. Nous passons également en revue les critiques et recommandations à propos des politiques agroenvironnementales actuelles et discutons nos résultats à l'aune de ces recommandations. Cela nous permet aussi d'identifier les limites de notre approche.

8.1. Contexte actuel en termes d'incitations

Les incitations existantes peuvent être classées en deux catégories: les incitations qui poussent les agriculteurs à fournir des biens marchands et les incitations à fournir des des services écosystémiques non-marchands. Nous adoptons une telle classification étant donné l'antagonisme fort entre la production agricole et les services écosystémiques non-marchands qui existe dans les systèmes agricoles intensifs: encourager la production décroît généralement la fourniture de services écosystémiques non-marchands. Cependant, nous ne supposons pas qu'éliminer les activittés agricoles serait la solution pour accroître la fourniture de services écosystémiques. Dans les zones agricoles marginales, soutenir la production agricole est un moyen de fournir des services écosystémiquees et d'autres biens publics. C'est ce qui justifie les Idemnités compensatices de handicap naturel (ICHN) qui encourage les agriculteurs à maintenir la production dans des zones montagneuses ou peu productives, dans le but de fournir en même temps des servies écosystémiques liés à la qualité du paysage.

Deux types dincitations oppposés

Les incitations à produire Contrairement aux autres services écosystémiques, la production agricole est marchande. Les prix agricoles sont la principale incitation à produire. Tout ce qui augmente les prix agricoles (notamment par les effets de demande) à un impact important sur les décisions concernant la production agricole.

Dans l'Union Européenne, la Politique Agricole Commune (PAC) est la politique publique la plus importante dans le secteur agricole. Par le passé, elle avait pour but d'accroître la production agricole tout en garantissant des des prix bas pour les consommateurs, et constituait une incitation marquée à produire. Depuis 2003, ces aides directes ont été majoritairement découplées de la production, ce qui a réduit les incitations à produire. Elles sont maintenant accordées sur la base de la surfce cultivée, sans exigence quant à la production, et elles sont justifiées par le soutien au revenu agricole. Cependant, ces aides représentent une part importante du revenu agricole (Matthews, 2016) et il est probable qu'elles encouragent la production à travers des effets de richesse (Femenia et al., 2010). De plus, dans certains pays dont la France, les aides directes actuelles sont basées sur les niveaux historiques des aides couplées, même si une certains convergence a eu lieu entre exploitations, orientations technico-économiques et régions. Matthews (2016) montre que ces aides bénéficient toujours en grande partie aux grosses exploitations agricoles (95% des aides vont aux exploitations dont le revenu est supérieur à la médiane), qui sont généralement plus intensives, avec une spécialisation et une utilisation d'inputs plus importantes (Mahé and Bureau, 2016).

Au-delà des paiements directs, beaucoup d'incitations encouragent la production agricolevia l'investissement. Les subventions pour la modernisation de la production encouragent les agriculteurs à acheter du matériel plus récent et plus puissant, et poussent indirectement la production à la hausse en augmentant la productivité, diminuant le coût marginal de production, ou en exigeant des niveaux d'inputs plus élevés. EEN France, les règles fiscales encouragent également les agriculteurs à investir[44], et mène à des stratégies d'investissement basées sur les retours financiers à court-terme plutôt que sur les besoins réels liés à la production (Delaire and Bonhommeau, 2011).

Ces incitations ne sont pas soumises à de nombreuses conditions pour recevoir les paiements, et sont accessibles à la plupart des agriculteurs. La politique agricole commune est la principale source d'incitations à produire.

Incitations à fournir des services écosystémiques non-marchands D'un autre côté, plusieurs incitations encouragent la fourniture de services écosystémiques non-marchands. Elles recouvrent plusieurs types d'instruments. Premièrement, des instruments réglementaires (normes) imposent des règles que les agriculteurs doivent suivre. La Politique Agricole Commune comprend deux niveaux de ce type de normes. Les aides directes sont tout d'abord conditionnées au respect de la croiss-compliance (normes concernant l'nevironnement, la sécurité alimentaire et le bien-êtr animal). Depuis 2014, de plus, 30% des aides directes sont soumises au respect d'exigences supplémentaires concernant les pratiques agricoles (paiement vert). Les directives environnementales prises au niveau de L'unIon Européenne sont égalemetn une source d'autres normes qui encouragent la fourniture de services écosystémiques non-marchands.

Deuxièmement, la PAC comprend aussi des dispositifs basés sur des incitations économiques. En particulier, les agriculteurs peuvent souscrire des Mesures AgroEnvironnementales (MAE) et recevoir une aide en échange de la mise en placed'actions fournissant des services écosystémiques non-marchands au-delà du simple respect des lois et des normes[45] (par exemple l'agriculture biologique, la plantation de haies...). Ce paiement compense la perte de revenu correspondante et est accordé en fonction des actions mises en oeuvres, sans conditions sur les résultats en termes de services écosystémiques. D'autres

[44]Plus précisément, des réductions d'impôts sont acordées pour les investissements et les gains en capital lors de la revente de matériel.

[45]Seule une partie des MAE ont pour but de préserver l'environnement. Les autres MAE visent à préserverle patrimoine rural et les paysages.

incitations économiques ont été mises en place par ertains États européens en dehors de la PAC, comme par exemple les taxes sur les pesticides et les engrais.

Les normes et les incitations économiques sont renforcées par la création de zones spécifiques, jugées particulièrement sensibles et devant être préservées. À l'échelle de l'UE, le réseau Natura2000 et les parcs naturels régionaux et nationaux à l'échelle nationale sont des leviers pour envourager l'adoption de régulations et d'incitations plus ambitieuses pour fournir des services écosystémiques non-marchands. Beaucoup d'autres dispositifs existent au niveau national et infra-national, basés sur l'accompagnement, la diffusion d'information etc.

Au final, beaucoup dincitations différentes pour la fourniture de services écosystémiques non-marchands cohabitent, qui sont gérées par des acteurs différents. Tous ces dispositifs sont conditionnés au respect d'exigences particulières, contrairement à la plupart des incitations à produire.

Cas particulier des subventions pour les zones défavorisées Les subventions pour les zones défavorisées (indemnités compensatrices de handicap naturel, ICHN) sont un cas particulier d'incitation à produire qui vise aussi à fournir des services écosystémiques non-marchands. Les zones défavorisées comprennent les zones où les activités agricoles sont menacées et fournissent des co-bénéfices sociaux (par ex. la mintien d'activité économique en zone rurale) et des services écosystémiques (liés à la biodiversité la qualité du paysage). Elles rassemblent notamment les zones montagneuses ou de piémont, les zones peu peuplées ou peu productives. Le paiement associé est basé sur la surface agricole et est conditionné de manière à encourager des activités agricoles extensives.

Les deux types d'incitations se contredisent

Bien que que de nombreuses incitations à fournir des services écosystémiques non-marchands existent, ces services écosystémiques ne sont pas fournis en quantités suffisantes pour assurer le fonctionnement durable des écosystèmes. La première raison est que ces incitations entrent en contradiction avec les incitations à produire.

De manière générale, la production et les services écosystémiques non-marchands sont en antagonisme (Lee and Lautenbach, 2016). En effet, les pratiques agricoles qui accroissent la production à court terme menacent généralement les processus écologiques de support. Par conséquent, les incitations à produire, telles que les aides directes de la PAC sont en contradiction avec les incitations à fournir des services écosystémiques non-marchands. Chaque agriculteur fait face à ces incitations contradictoires: les deux types incitations ne sont pas attribuées à deux groupes distincts d'agriculteurs. Les exigences environnementales nécessaires pour bénéficier des aides directes ou du paiement vert ne sont pas assez ambitieuses pour résoudre cette contradiction (Pe'er et al., 2014; Bureau and Thoyer, 2014), et la plupart du soutien à l'investissement est alloué sans réelles conditions environnementales et est susceptible d'avoir des impacts négatifs sur l'environnement (Allen and Hart, 2013).

Un déséquilibre dans le budget disponible

Étant donné l'antagonisme entre les deux types d'incitations, partant de la situation actuelle, les incitations à fournir des services écosystémiques non-marchands devraient être au moins aussi importantes que les incitations à la production. En pratique, il existe un important déséquilibre en termes de budget. C'est la deuxième raison pour laquelle les incitations à fournir des services des services écosystémiques échouent.

Au sein de la PAC, la répartition du budget est largement en faveur des incitations à la production. Les aides à la production représentent 74% du budget communautaire de la PAC pour la période 2014-2020. En comparaison, pour la même période, les dépenses communautaires prévisionnelles pour les Mesures AgroEnvironnementales et l'agriculture biologique représentent à peine plus de 5% du budget (d'après nos propres calculs avec les données de European Commission (2016) et Henke et al. (2015)). En incluant le co-financement de ces mesures par les États membres, Duval et al. (2016) estiment qu'elles représentent 8% des dépenses totales sur la période 2007-2013. Au niveau des exploitations, la même disproportion existe: les aides directes constituent 47% du revenu agricole en moyenne en UE, tandis que les autres paiements (incluant les MAE et les autres mesures du 2nd pillier de la PAC), en constituent seulement 15% (Matthews, 2016). Par conséquent, les MAE ont un impact positif, mais limité sur l'environnement (Barbut and Baschet, 2005; Batáry et al., 2011; Matthews, 2016)[46].

Ce déséquilibre demeure dans la plupart des cas en incluant les autres incitations à fournir des services écosystémiques. Pour avoir une idée budget total dédié aux deux types d'incitations (y compris en dehors de la PAC), nous comparons au niveau des États membres d'une part le budget des aides directes de la PAC (budget du 1er pillier), et d'autre part le budget total dédié à la protection de l'environnement (y compris en dehors de la PAC). Nous évaluons ce dernier par les dépenses inscrites dans les comptes nationaux pour les enjeux environnementaux liés aux zones agricoles (protection des sols et des masses d'eaux, et de la biodiversité et des paysages). Ces dépenses incluent le co-financement national des MAE, mais pas le financement de l'UE[47]. Le tableau 1 montre ce déséquilibre en termes de budget. Exceptés l'Autriche et les Pays-Bas, le budget du Premier pillier de la PAC est largement supérieur aux dépenses pour la protection de l'environnement. Ces chiffres n'ont pour but que de fournir un ordre de grandeur. Les dépenses pour la protection de l'environnement surestiment probablement les incitations à fournir des services écosystémiques non-marchands en agriculture, puisqu'elles incluent également des dépenses relatives à d'autres secteurs économiques (dépenses privées et publiques liées à la dépollution des sols et des masses d'eau, compensation des pertes de biodiversité liées aux nouvelles infrastructures)[48]

[46]Plus précisément, (Uthes and Matzdorf, 2013) trouve des effets contrastés selon les mesures et les pays, et rapporte des effets positifs notamment pour la protection des oiseaux spécialistes agricoles et des pollinisateurs et dans les zones extensives.

[47]Nous n'avons pas trouvé les chiffres détaillés du Second Pillier pour chaque État-membre, mais comme indiqué plus haut, ils représentent en moyenne 5% du budget total de la PAC, soit 6,7% du budget du Premier pillier.

[48]Dans le cas de la France, selon les explications de ces statistiques, 38% des dépenses liées aux masses d'eau et aux sols ne concernent pas l'agriculture, et au moins 17% des dépenses au titre de la biodiversité et des paysages sont des compensations pour la construction d'infrastructures (voir `http://www.statistiques.developpement-durable.gouv.fr/lessentiel/s/depenses-protection-lenvironnement.html`)

	budget des aides directes de la PAC (mio. euros)	Dépenses pour la protection de l'envrionnement - sols & masses d'eau (mio. euros)	biodiversité & paysages (mio. euros)
Autriche	718	922	42
France	8332	1230	1202
Allemagne	5197	90[a]	1400[a]
Pays-Bas	840	630	789
Pologne	3208	213	130

Table 1.: *Dépenses publiques pour la production agricole et la protection de l'environnement, dans plusieurs pays de l'UE (2014)*
Les dépenses pour la protection de l'environnement incluent le co-financement national des MAE.
Sources : Eurostat, Statistisches Bundesamt, Financial report of EAGF 2014
[a] ces chiffres datent de 2010 (données plus récentes non disponibles)

Le budget devrait être plus orienté vers la fourniture de services écosystémiques non-marchands

Il est nécessaire d'accroître la fourniture des services écosystémiques non-marchands en agriculture. Les deux types d'incitations sont contradictoires, et leur budgets sont déséquilibrés. En théorie, accroître les services écosystémiques non-marchands de manière efficace passe par une coupe dans les incitations à la production (OECD, 2010; DG for Internal Policies (European Parliament), 2010; Matthews, 2016). Les prix agricoles sont déjà une incitation à produire, et l'argent public devrait servir à fournir des biens publics.

En pratique, il est plus réaliste d'envisager une modification moins radicale des budgets, et le développement des Mesures AgroEnvironnementales. Le budget alloué aux incitations à la production sont un soutien majeur au revenu agricole. Les agriculteurs en dépendent pour vivre et une coupe dans ces aides directes aurait des effets important sur le revenu agricole. Les dernières réformes de la PAC ont autorisé les États membres à transférer du budget d'un pillier à l'autre et ainsi des incitations à la production vers les incitations à fournir des services écosystémiques. Cependant, ces possibilités sont facultatives, et tous les États membres n'ont pas fait le choix de les utiliser. Une autre possibilité pour réduire les incitations à la production tout en maintenant le soutien au revenu est de jouer sur le design des aides directes. L'introduction des paiements verts (qui conditionnent 30% des aides directes à la mise en œuvre de pratiques plus respectueuses de l'environnement) sont un pas dans cette direction, même si les pratiques retenues ne vont pas beaucoup au-delà des pratiques actuelles. Une autre solution possible, plafonner les aides directes par travailleur, serait tout à fait compatible avec le principe d'un soutien au revenu, tout en diminuant l'incitation à la production et en assurant une répartition plus équitable des aides directes.

DG for Internal Policies (European Parliament) (2010) et Matthews (2016) poussent la logique des paiements verts plus loin et proposent une modification de la construction de la PAC, en imaginant plusieurs "couches" de paiements: un paiement de base pour soutenir le revenu, puis différentes couches pour inciter à fournir des services écosystémiques, avec plusieurs degrés d'exigence et donc de paiement.

Au-delà de ces pistes qui touchent le budget alloué aux incitations à la production, le design des Mesures AgroEnvironnementales elles-mêmes est un levier important pour la fourniture de services écosystémiques non-marchands. Il s'agit de fournir plus de services écosystémiques à budget donné, et beaucoup de recommandations issues des analyses de politiques vont dans ce sens.

Ces constats sont le point de départ de cette thèse, qui plaide pour l'amélioration des Mesures AgroEnvironnementales. Dans cette thèse, nous explorons des questions liées à la conception des politiques agroenvironnementales: l'identification des actions efficientes et la nécessité de prendre en compte le coût de la fourniture de services écosystémiques, les incitations utilisées pour mettre en œuvre ces solutions, et les conséquences de l'hétérogénéité. Dans les sections suivantes, nous mettons nos résultats en perspective avec des analyses sur les politiques agroenvironnementales. Nous passons également en revue les limites de notre approche.

8.2. Efficience des solutions agroécologiques

Dans le chapitre 6, nous montrons qu'identifier les solutions qui maximisent les services écosystémiques est important. La fourniture des différents services écosystémiques doit être pensée de manière cohérente: les options qui augmentent un service écosystémique n'augmente pas forcément les autres services.

Cette conclusion fait écho à certaines critiques et recommandations à propos des politiques actuelles, qui ne ciblent pas toujours des solutions efficientes ou efficaces. La Cour des Comptes Européenne (Cour des Comptes Européenne, 2011) estime que dans 24% des cas, les bénéfices environnementaux escomptés par la mise en œuvre de MAE ne peuvent pas être démontrés. Cela ne veut pas dire qu'ils n'existent pas, mais tout du moins ces mesures agroenvironnentales ont été conçues sans être sûr qu'elles apportent un bénéfice environnemental. L'identification des bénéfices environnementaux devrait être renforcée (Duval et al., 2016), ainsi que l'évaluation ex-post des mesures agroenvironnementales, qui fait aujourd'hui défaut (Epices and ADE, 2017).

De plus les politiques agroenvironnementales actuelles ne s'attaquent pas toujours aux différents enjeux de manière intégrée (Galler et al., 2015). Les directives de l'UE sur l'environnement sont un exemple frappant: elles sont conçues séparément pour chaque enjeu (qualité de l'eau, déchets, protection des oiseaux, etc.) et non pas de manière holistique. Cela vaut également pour le FEADER (la structure qui met en œuvre les Programmes de Développement Rural, dont les Mesures AgroEnvironnementales). Le FEADER affiche explicitement des objctifs environnementaux et sociaux multiples, mais en réalité les MAE sont conçues pour s'attaquer à un objectif précis.

Les interactions spatiales et les dynamiques agroécologiques rendent difficile l'identification des solutions qui fournissent les niveaux de services écosystémiques maximaux. Notre modélisation appliquée n'inclut pas d'interactions spatiales ni de dynamiques, ce qui est à la fois une des limites de notre approche, et ouvre des perspectives de recherche.

Les interactions spatiales exigent un seuil de participation de la part des agriculteurs, ou de penser l'arrangement spatial des actions à mettre en œuvre, et éventuellement de recourir à des incitations différenciées spatialement (par ex. des bonus d'agglomération). Les dynamiques imposent de définir l'horizon temporel et les actions au cours du temps. Il est difficile d'imaginer comment cela changerait nos résultats, dans tous les cas cela les rendrait plus complexes.

Notre modèle agroécologique est relativement simple, notamment en ce qui concerne les options de gestion inclues. Par conséquent, nous négligeons des pratiques agricoles intéressantes, qui pourraient modifier les solutions efficientes.

En particulier, l'introduction de rotations plus complexes semble une piste intéressante pour accroître les services écosystémiques non-marchands.

8.3. Prise en compte du coût de la fourniture de services écosystémiques

Dans le chapitre 6, nous montrons l'importnace de prendre en compte les coûts liés à la fourniture des services écosystémiques non-marchands. Cela permet de considérer des solutions qui fournissent moins de services écosystémiques, mais à un coût plus faible. Elles peuvent être mises en places sur des surfaces plus étendues, et fournir des niveaux totaux de services écosystémiques élevés. Nous comparons le coût de deux stratégies pour fournir les mêmes services écosystémiques non-marchands: i) une augmentation modeste des services écosystémiques non-marchands, mais sur une grande surface, ou ii) dédier une partie de la surface agricole à une augmentation importante des services écosystémiques. Nous montrons qu'aucune stratégie ne domine l'autre, les résultats dépendent de services écosystémiques ciblés.

Le débat à propos de laquelle des stratégies est la meilleure façon d'augmenter les services écosystémiques est présent dans les analyses des politiques actuelles. Par exemple, les États membres ont fait différents choix dans la mise en œuvre des MAE. En France, le choix a été fait de les saupoudrer. Le budget totale est équivalent à celui de la Finlande ou de l'Autriche, mais est réparti sur une surface trois fois supérieure. Le paiement moyen par hectar est le plus faible de toute l'UE, et pour assurer la participation des agriculteurs, ces mesures agroenvironnementales sont forcément peu exigentes (Duval et al., 2016). Il n'existe pas d'analyses permettant de comparer les résultats de ces différentes stratégies. La Cour des Comptes Européennes (Cour des Comptes Européenne, 2011) souligne que dans la plupart des cas, la répartition des budgets entre les mesures, et le niveau des paiements ne sont pas réfléchis en fonction d'un seuil de participation qui garantirait la fourniture effective de services écosystémiques.

Une des limites de notre modèle est la représentation de la prise de décision des agriculteurs, qui repose uniquement sur la maximisation du profit, et n'inclut que les coûts variables dans le calcul du profit. En réalité, le consentement à recevoir (le montant minimal que les agriculteurs exigent pour souscrire une MAE) diffère du coût d'opportunité. Nous sous-estimons probablement le coût réel lié à la fourniture des services écosystémiques non-marchands et le rôle des autres déterminants dans la prise de décision.

Notre approche considère le coût d'opportunité de manière restrictive, ce qui est de fait le problème que rencontrent les politiques agroenvironnementales actuelles (Duval et al., 2016). Les paiements sont calibrés selon le coût d'opportunité et sont souvent trop faibles pour couvrir le consentement à recevoir de la plupart des agriculteurs, ce qui résulte en un faible taux de participation. Par exemple, les coûts de transaction ne sont pas inclus dans le calcul du coût d'opportunité. Ces coûts sont ceux liés à l'information, aux démarches administratives, au contrôle, à la coordination etc. Ils sont estimés à 14% des coûts totaux liés à l'adoption des MAE (Mettepenningen et al., 2009). Ils jouent sur le consentement à payer (Ruto and Garrod, 2009) et l'adoption de MAE (Ducos and Dupraz, 2006). Changer les pratiques agricoles peut exiger des investissements coûteux (formation, matériel pour le travail du sol réduit ou le désherbage mécanique etc.), qui ne sont pas comptabilisées dans le coût d'opportunité. En outre, le consentement à recevoir comprend d'autres déterminants d'ordre social ou psychologique (convictions personnelles sur l'environnement, aversion au risque, etc.), qui ne sont pas prises en compte dans le coût d'opportunité. Cela pourait expliquer que peu d'agriculteurs réduisent leur utilisation de pesticides, alors que le coût d'opportunité est très faible (Lechenet et al., 2017).

Afin de combler l'écart entre les paiements agroenvironnementaux et le consenement à recevoir une première solution est de réduire les coûts de transaction. Par exemple, les conseils techniques personnalisés et l'accompagnement peut diminuer ces coûts de transaction et par là le consentement à recevoir (Espinosa-Goded et al., 2010). Une deuxième solution (qui n'exclut pas la première), est d'inclure tous les coûts dans la calibration des paiements, y compris les coûts fixes (liés à l'investissement entre autres). Si les agriculteurs sont payés pour fournir des services écosystémiques, les paiements devraient compenser le coût total, pas seulement le coût variable ou marginal (Barnes et al., 2011; Duval et al., 2016). Cela peut passer par une meilleure articulation des MAE avec les aides à l'investissement qui existent notamment dans le 1[er] pillier. Actullement, cette articulation n'est pas très bien aménagée (Duval et al., 2016). Troisièmement, il est possible de jouer sur les déterminants sociaux et psychologiques du consentement à recevoir. Les dispositifs d'assurance-récolte peuvent compenser le risque perçu comme supérieur suite à l'adoption de certaines MAE (par exemple de réduction des pesticides). L'action collective peut aussi diminuer le risque perçu et la dépendance au sentier (Duval et al., 2016), tout en encourageant la diffusion de connaissances. Le rôle de cette diffusion de connaissances dans la pérennité des changements de pratiques induits par les MAE est souligné par Kuhfuss et al. (2016). Les auteurs la voient comme un exemple de *nudge*, c'est-à-dire le fait de jouer sur les déterminants psychologiques de la prise de décision, et la relient à l'émergence de nouvelles normes sociales. L'émergence de ces nouvelles normes sociales liée à l'adoption plus répandue de politiques environnementales est vue comme une solution par Nyborg et al. (2016). Dernièrement, la valorisation des efforts environnementaux par le développement de labels et de filières spécifiques favorise aussi l'adoption des mesures agroenvironnementales (Kuhfuss, 2013): si les agriculteurs peuvent obtenir un revenu marchand plus important de par la souscription d'une MAE, le paiement exigé est plus faible.

Une autre limite de notre approche et des politiques agroenvironnementales actuelles est qu'elles négligent les variations des prix agricoles. Une augmentation des prix augmente le coût d'opportunité lié à l'adoption de MAE et diminue le taux de participation. Adapter les paiements au coût d'opportunité impliquerait de les indexer sur les prix agricoles, ce qui pose des problèmes en termes de planification du budget et augmente la variabilité du revenu.

8.4. Incitations économiques

Dans le chapitre 7, nous comparons les incitations basées sur les actions et celles basées sur les résultats pour la mise en œuvre de solutions pour augmenter la fourniture de services écosystémiques. Les incitations basées sur les résultats ciblent exactement les options coût-efficientes. Cependant, nos résultats montrent que les interactions entre services écosystémiques renforcent les contraintes de participation et conduisent à un budget plus élevé avec les incitations basées sur les résultats.

Peu de recherches offrent des résultats comparables. Contrairement à nos résultats, les analyses des politiques actuelles citent souvent les incitations basées sur les résultats comme étant incapables de sélectionner les actions coût-efficientes et mentionnent souvent les incitations basées sur les résultats comme une amélioration possible des politiques actuelles.

Les expérimentations d'incitations basées sur les résultats sont assez limitées, notamment parce que beaucoup de services écosystémiques sont difficiles à mesurer à l'échelle du champ ou de la ferme. Des subventions basées sur les résultats ont été expérimentés en grande partie pour la protection de la biodiversité, les résultats étant mesurés par la présence ou l'abondance de certaines espèces. Ces subventions ne sont pas de "pures" incitations basées sur les résultats: souvent, le paiement ne varie pas vraiment selon le résultat (il est accordé sous condition de respecter une contrainte basée sur le résultat, par exemple la présence d'une ou plusieurs espèces), et ces subventions sont intégrées dans des politiques basées sur les actions. Ces expérimentations semblent réussies (Musters et al., 2001; Allen et al., 2014), elles conduisent à une manière plus coût-efficace de protéger la biodiversité. La littérature souligne l'importance du choix de l'indicateur pour mesurer le résultat. Notamment, cet indicateur doit être difficile à manipuler, ou les agriculteurs risquent de choisir leurs actions de façon à jouer sur cet indicateur sans fournir véritablement de service écosystémique[49].

Les enchères inversées sont également souvent citées comme un autre exemple "impur" d'incitation basée sur les résultats. Les actions financées par les politiques agroenvironnementales sont sélectionnées sur la base des résultats environnementaux et des coûts escomptés. Cela les différencient des incitations basées sur les résultats "pures", puisque le paiement n'est pas proportionnel au résultat environnemental réel. Ce mécanisme d'enchères a été expérimenté avec succès pour la protection de la biodiversité aux États-Unis et en Australie (Duval et al., 2016). En France, une expérimentation pour améliorer la qualité de l'eau a été moins réussie, en partie à cause de détails dans leur mise en œuvre. Les études soulignent que de nombreux paramètres influencent leur succès (informations données aux agriculteurs, accompagnement, nombre de tours de l'enchère, montant du paiement, etc.). Ces dispositifs peuvent aussi être à l'origine de coûts de transaction élevés, et introduire de la compétition inutilement entre les agriculteurs, ou au contraire créer des collusions qui font augmenter les paiements demandés (Kuhfuss,

[49] Dans le cas de la protection des oiseaux spécialistes agricoles, les bénéfices environnementaux reposent sur une gestion précise des prairies (dates et hauteur de fauche, phases de pâturage etc.). Des incitations basées sur les résultats, pour lesquelles le résultat est mesuré par la présence d'un nid rendent les nids "rentables" économiquement. Cela peut conduire certains agriculteurs à rechercher les nids, à les enclore et gérer le reste de leur prairie sans faire plus d'effort, tout en recevant les paiements. Une telle stratégie ne permet pas forcément de protéger l'espèce, contrairement aux pratiques de gestion des prairies.

2013).

Reed et al. (2014) et Moxey and White (2014) offrent une bonne synthèse de l'intérêt des approches basées sur les résultats: elles ne doivent pas être considérées comme la solution à tous les problèmes des incitations basées sur les actions. Les caractéristiques intéressantes des incitations basées sur les résultats (le ciblage spatial, la différentiation des paiements, etc.) peuvent aussi être appliquées avec des incitations basées sur les actions. Fleury et al. (2015) concluent que le succès des incitations basées sur les résultats dans le cas des prairies fleuries repose sur son inclusion au sein d'un programme plus vaste qui comprend des formations, du conseil, le développement d'une norme sociale positive envers la participation aux mesures agroenvironnementales.

8.5. Adaptation à l'hétérogénéité

Dans le chapitre 8, nous introduisons de l'hétérogénéité et nous explorons ses conséquences. En elle-même, l'hétéogénéité change la coût-efficience, et considérer l'échelle du paysage (au lieu de l'échelle de la parcelle) change également les résultats en termes de coût-efficience, même en l'absence d'hétérogénéité.

Le résultat sur l'importance d'analyser la coût-efficience à l'échelle du paysage peut difficilement être relié à des recommandations pratiques. Ce serait compliqué d'évaluer les solutions coût-efficientes à l'échelle d'un paysage: il faudrait inclure toutes les combinaisons possibles des options disponibles sur chaque parcelle. Ce serait également compliqué d'appliquer une solution déterminée à l'échelle du paysage: cela supposerait que les pratiques d'un agriculteur soient dépendantes des caractéristiques de toutes les fermes du paysage. Néanmoins, l'analyse de coût-efficience devrait au moins être pensée à l'échelle de la ferme. Dans cette optique, une part des MAE (dites MAE systèmes) s'appliquent à toute la ferme et peuvent contribuer à la mise en œuvre de solutions plus coût-efficaces.

Nos résultats montrent que les solutions coût-efficientes pour fournir des services écosystémiques non-marchands incluent souvent une contribution importante de la partie la plus productive.

Cependant, l'hétérogénéité rend impossible la calibration exacte des incitations en fonction du consentement à recevoir. Elle a trois effets négatifs sur l'efficacité en termes de budget des politiques: elle diminue le taux d'adoption des MAE (les agriculteurs qui ont un consentement à recevoir supérieur au paiement ne participent pas), surcompense les agriculteurs qui ont un consentement à recevoir plus faible et sélectionne implicitement les agriculteurs qui ont le consentement à recevoir le plus faible mais qui n'offrent pas forcément des solutions coût-efficentes. C'est exactement les critiques auxquelles les politiques agroenvironnementales font face actuellement (Uthes and Matzdorf, 2013; Duval et al., 2016): ce sont surtout des agriculteurs en zones moyennement productives qui participent, tandis que les solutions coût-efficientes reposent potentiellement sur les zones plus productives. Cibler les zones agricoles plus intensives nécessite des paiements plus élevés, ce qui réduit la surface d'application à budget égal, et pose des problèmes d'équité: les fermes des zones productives ont un profit supérieur, et leur donner des aides plus élevées peut sembler injuste.

En présence d'hétérogénéité, nous observons que les incitations uniformes sont associées à un budget plus élevé que les incitations différenciées.

Ce constat a été fait depuis longtemps par les analyses de politiques publiques et la littérature scientifique. Une solution à l'hétérogénéité dans les coûts et les bénéfices environnementaux et à l'uniformité des incitations est de concevoir et de calibrer les incitations au niveau local, et de manière participative. Viser un nombre plus réduit d'agriculteurs, qui partagent certaines caractéristiques peut réduire l'hétérogénéité. Calibrer les incitations à ce niveau réduit leur uniformité. Blumentrath et al. (2014) concluent que les politiques conçues au niveau local et avec la participation des agriculteurs sont plus efficaces. Par exemple, depuis 2013, les Mesurs AgroEnvironnementales (au moins en France) sont conçues et calibrées au niveau des régions. Les enjeux environnementaux visés, les mesures et le montant des paiements varient selon les régions. Certaines mesures ne sont disponibles que dans des territoires spécifiques, ce qui assure une plus grande homogénéité en termes de bénéfices environnementaux. Cette évolution a permis une meilleur efficacité (Duval et al., 2016). De la même manière, les paiements pour les prairies semi-naturelles en Suède varient selon la probabilité qu'ils abritent des espèces protégées (Ekroos et al., 2014). Le même principe justifie les demandes vers plus de subsidiarité laissée aux États membres de l'UE pour définir les priorités, les actions et la calibration des paiements au sein de la PAC.

La calibration des incitations au niveau local a également d'autres avantages: cela favorise la participation et la collaboration d'acteurs multiples (agriculteurs, institutions locales, agences environnementales, etc.), et cela peut favoriser la confiance mutuelle entre les acteurs et renforcer l'acceptation et la légitimité des politiques agroenvironnementales.

Une alternative plus complexe pour réduire l'asymétrie d'information est l'utilisation de "menus" de mesures agroenvironnementales, qui sont mis en œuvre dans certains États membres (en particulier au Royaume-Uni et en Autriche). Une liste de mesures avec différents niveaux d'engagement et de paiement est proposée, et les agriculteurs peuvent combiner ces mesures selon des règles fixées (Duval et al., 2016). En France, des mesures pour la qualité de l'eau ont été testées selon ce principe: deux versions de la même mesure étaient disponibles, une version "légère" avec des exigences et un paiement faibles et une version "lourde" (Kuhfuss, 2013). Ces approches reposent sur l'auto-sélection des agriculteurs en fonction de leurs coûts et correspondent aux "contrats auto-sélectifs" (self-screening contracts) mentionnés dans la littérature économique (Ferraro, 2008). Calibrées de manière appropriées, elles offrent des incitations différenciées sans que le régulateur ait besoin de définir quel niveau d'incitation offrir à chaque agriculteur ni de connaître ses coûts.

En ce qui concerne la mise en œuvre des solutions coût-efficientes avec des incitations économiques, nos résultats montrent que les incitations basées sur les résultats requièrent souvent un budget plus important que celles basées sur les actions. L'hétérogénéité ne les rend pas plus avantageuses.

Ces résultats sont quelque peu contradictoires par rapport aux études comparables (Gibbons et al., 2011). Notre approche donne un rôle important aux interactions entre services écosystémiques et notamment inclut des synergies, ce qui explique probablement cette divergence. À notre connaissance, aucune étude empirique ne propose de résultats sur ce point.

8.6. Pour conclure

Nos résultats et les analyses des politiques publiques se rejoignent sur plusieurs points. Premièrement, les mesures agroenvironnementales doivent être conçues selon des objectifs environnementaux clairs, et si possible de manière intégrée. Le lien entre les (changements de) pratiques agricoles et les services écosystémiques fournis doit être établi. Deuxièmement, les politiques agroenvironnementales doivent prendre en compte le coût d'opportunité de la fourniture de SE afin de maximiser les services écosystémiques pour un budget donné. Selon la situation, la stratégie optimale peut être de concentrer les efforts sur une petite partie du territoire, ou au contraire de saupoudrer les efforts sur tout le territoire. Troisièmement, étant donné l'hétérogénéité, les politiques doivent être ciblées spatialement et les paiements différenciés. Nous trouvons que les incitations basées sur les résultats ne constituent pas une solution miracle pour les politiques agroenvironnementales en présence d'hétérogénéité, notamment en raison des interactions entre les services écosystémiques régulés. Les solutions pour améliorer la coût-efficacité des politiques actuelles peuvent être mises en œuvre via des incitations basées sur les actions. Enfin, augmenter les services écosystémiques non-marchands par rapport à la sitation actuelle exige une augmentation du budget dédié aux politiques agroenvironnementales. Beaucoup de progrès peut être fait en ce qui concerne la coût-efficacité des mesures agroenvironnementales, mais il n'en reste pas moins que le déséquilibre en termes de budget ne doit pas être négligé. Malgré ce point évident, la dernière réforme de la PAC a abouti à une diminution du budget du Second Pillier (dédié au Développement Rural, comprenant les mesures agroenvironnementales) de 18% pour la période 2014-2020, même si au sein de ce budget la part réservée aux mesures agroenvironnementales a augmenté de 25 à 30%. Parallèlement, la baisse de budget du Premier pillier n'était que de 13%, et est même stable comparée à la situation contrefactuelle (si aucune réforme n'avait eu lieu)[50]. Les négociations pour la prochaine réforme de la PAC ont lieu actuellement, espérons qu'elles marquent un progrès pour la fourniture de services écosystémiques non-marchands.

[50] http://capreform.eu/the-cap-budget-in-the-mff-agreement/

Detailed outputs of simulations and analyses

1. Output of the simulations

(agronomic context 8, potential yield = 9.9 t/ha)

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)	Pollination (index)	Water quality (index)	Net emissions of GHG (t CO_2eq)	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)	Pollination (index)	Water quality (index)	Net emissions of GHG (t CO_2eq)	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)	Pollination (index)	Water quality (index)	Net emissions of GHG (t CO_2eq)	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

2. Efficient bundles of ecosystem services in all agronomic contexts

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

	Contexts									
	1	2	3	4	5	6	7	8	9	10
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	Contexts									
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3. Output of the cost-efficiency analysis (context 8)

Bright green lines (options 1 and 6) maximise non-marketed ecosystem services. Orange-colored line (option 47) is the statu quo (most profitable option). All ecosystem service levels and the opportunity cost are expressed relative to it. Grey-colored lines are options that provide less non-marketed ES than the statu quo and are excluded from the analysis.
Light blue lines are options for which the **concentrating strategy** is less costly. Light green lines are options for which the **sprinkling strategy** is less costly.

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

Model calibration

This appendix provides first a summary table of parameter values used for the simulations, and a detailed description of the sources for these values. The values are given for a 1ha agricultural area.

1. Summary table

parameter	meaning	value
FA_k	Foraging resources for pollinators (index)	0.5 in grasslands 0.23 in croplands with non-crop habitat 0.2 in cropland without NCH
NS_k	Habitat suitability for pollinators (index)	0.4 in grasslands 0.23 in cropland with NCH 0.2 in cropland without NCH
PM_k	pesticide impact factor on pollinators (index)	1.4 for no pesticides 1.2 for medium pesticide intensity 1 for maximal pesticide intensity
m_k	mineralisation rate of organic nitrogen (fraction)	0.015 for reduced tillage and grassland 0.019 for conventional tillage
λ_k	erosion rate (fraction)	0.00006 for grassland 0.00007 for cropland, reduced tillage, and with crop residue restitution 0.00009 for cropland, reduced tillage and without crop residue restitution 0.0002 for cropland, conventional tillage and with crop residue restitution 0.00025 for cropland, conventional tillage and without crop residue restitution
I_k	fresh organic matter inputs (t/ha) (expressed in humus equialent)	1.684 for grassland 1.348 for cropland, reduced tillage, and with crop residue restitution 1.142 for cropland, reduced tillage and without crop residue restitution 1.321 for cropland, conventional tillage and with crop residue restitution 1.059 for cropland, conventional tillage and without crop residue restitution
f_k	Mineral nitrogen input from synthetic fertilisers (kg/ha)	0 for no fertilisers 110 for fertiliser intensity 1 140 for fertiliser intensity 2 170 for fertiliser intensity 3 200 for fertiliser intensity 4
LN_k	Mineral nitrogen input from livestock on grassland (kg/ha)	62 if livestock impacts are accounted for
c_3	Carbon content of soil organic matter (fraction)	0.58

c_2	Carbon to nitrogen ratio in soil organic matter (no unit)	9
β	denitrification rate (fraction)	1%
γ	nutrient uptake coefficient (fraction)	0.87
ϵ	nitrogen uptake parameter (no unit)	0.0032
N^*	nitrogen uptake threshold (kg/ha)	206
c_4	conversion of elemental nitrogen into nitrous oxide (no unit)	1.57
c_5	conversion of elemental carbon into carbon dioxide (no unit)	3.66
g_1	global warming potential of nitrous oxide (no unit)	298
g_2	global warming potential of methane (no unit)	34
FC_k	carbon dioxide due to fuel burning (kg CO_2eq/ha)	8 for grassland 120 for reduced tillage 150 for conventional tillage
-	pesticide doses (standard doses)	3 for medium pesticide intensity 6 for maximal pesticide intensity
w	reduction in pollutants due to non-crop habitats (no unit)	40%
n_2	marginal effect of uptaken nitrogen on yield	0.015 (no unit)
e	area covered by non-crop habitat, if any (no unit)	5%
h_1	fraction of crops damaged by pest in absence of pesticides (no unit)	0.3
h_{2k}	Fraction of crop damage avoided by pesticides (no unit)	0 for no pesticides 0.6 for medium pesticide intensity 0.9 for maximal pesticide intensity
ρ	crop - residue ratio (no unit)	1
-	crop price (euro/t)	170
s_G	revenues of grassland (euro/ha)	720
MC_k^B	base management costs (euro/ha)	197 for any management option
MC_k^F	fertiliser costs (euro/kg N)	1.15
MC_k^{Pesti}	pesticide costs (euro/standard dose)	33
MC_k^M	mechanisation costs (euro/ha)	150 for grasslands 225 for reduced tillage 300 for conventional tillage
MC_k^{NCH}	costs of implementing non-crop habitat (euro/ha)	35

Table 1.: *Summary of parameter values*

2. Pollination

In line with the formalism adopted to model pollination, we chose reference values for parameters of the pollination model given by the appendix of Zulian et al. (2013).

2.1. Flower resources for pollinators

FA_k describes the foraging resources for the pollinators provided by agricultural practice k. Reference values are given for many land uses, we keep:

- value for non-irrigated arable land: 0.2.
- value for fodder on arable land: 0.5
- riparian areas : 0.8

Concerning the values, we identify grassland in our model with the land use class "fodder on arable land", semi-natural elements with the clas "riparian areas", and we take the generic value for arable land to represent the foraging resources of cropland.

To adapt these values to our model, we assume that tillage and fertiliser intensity have no impact upon pollination. Biomass input may have an impact but the lack of consistent data drives us to ignore it. Semi-natural elements present in a crop field have an impact calculated with the percentage of the area they cover: the parameter for foraging resources in a cropfield with semi-natural elements is an area-weighted mean of parameters for "arable crop" and "riparian areas", with semi-natural elements representing 5% of the field when existing. The impact of pesticide is considered separately, as explained in section 2.3 below.

In the end, we keep following values :

	grassland	cropland	cropland with SNE
Foraging resources	0.5	0.2	0.23

2.2. Habitat suitability for nesting

NS_k the habitat suitability for nesting for agricultural practice k. The values given in the appendix of Zulian et al. (2013) are as follow:

- general value for arable land: 0.2.
- fodder on arable land: 0.4
- riparian areas: 0.8

Following the same logic as for the parameter for foraging resources above, we take following values:

	grassland	cropland	cropland with SNE
Foraging resources	0.4	0.2	0.23

2.3. Multiplicator to account for the impact of pesticides on pollinator abundance

Pesticides kill or have direct sublethal effects on pollinators, and herbicides also reduce the foraging resources available. No methodolgy is envisionned by Zulian et al. (2013) to account for the impact of pesticides on the pollination service. To avoid double counting the impact of pesticide, we multiply the pollination source score (the product of the two last parameters) by a coefficient measuring the increase in pollinators due to a reduction in pesticide (and herbicide and fungicide) use. We assume that given values represent current average European situation, and encompass the use of pesticides.

To assess the effect of pesticides, we use studies comparing pollination success in conventional versus organic farming systems, rather than studies estimating the impact of pesticides on flower resources and nesting suitability [51]. Measuring effective pollination through fuit set, Andersson et al. (2012) find that the proportion of fully pollinated strawberries is 2.6 times higher in organic fields, while Andersson et al. (2014) show that bean pods contain from 30% to 40% more beans in average, in organic fields. Klein et al. (2012) also find significant higher pollinator visitation rates in organic almond orchards, but the magnitude varies among species and landscape composition, with orders of magnitude for additional visits 0 to 100% in organic plots compared to conventional ones.

Eventually, considering the figures above, we assume that organic farming bears an increase in pollinator of 40% compared to the maximum intensity of pesticide use, while medium intensity bears an increase of 20%, which gives following values as multiplicators of the pollination source score:

	high intensity	medium intensity	no pesticides
multiplicator of pollinator source score	1	1.2	1.4

3. Pests

In the absence of pesticides, pests are supposed to destroy 30% of the effective yield. This figure stems from a study on the crop yield gap between organic and conventional agriculture De Ponti et al. (2012).

Pesticides are supposed to kill a fraction of pests, and save a proportional fraction of the yield gap. The marginal return of pesticides is assumed to be decreasing, i.e. the first applications of pesticides are more effective than the last ones. Hossard et al. (2014) identify that reducing pesticide use by 50% compared to current levels would reduce production by 5 to 13%. We assume that the medium pesticide intensity suppresses 60% of pests, and thus saves 60% of the yield loss due to pests; and that applying pesticides at maximum intensity kills 90% of pests, and save 90% of the yield loss. In the end, this means that pest damage are calibrated at 30% of yield in the absence of pesticides, 12% with medium intensity and 3% with high pesticide intensity.

[51] Indeed, papers studying the reduction in plant diversity due to herbicides (Krauss et al., 2011; Geiger et al., 2010) and the toxicity of pesticides for pollinators (Brittain et al., 2010; Pisa et al., 2014; Henry et al., 2012; Kevan et al., 1997) are difficult to generalise and interprete in terms of pollinator abundance

4. Soil organic matter and nitrogen

4.1. Parameters of the equation for soil dynamics

The SOM follows a dynamic characterised by the existence of equilibrium stocks. If the same management option is chosen over a long period of time, the stock of soil organic matter converges towards a value which depends on the management option. Thus, the flows must be coherent with each other and the equilibrium stock: $I_k = \overline{SOM}_k(m_k+\lambda_k)$.

The values for the equilibrium stocks $\overline{SOM}_k$ are determined with the stocks of soil organic carbon estimated by Arrouays et al. (2002a) in French winter wheat fields, under different management options (conventional or reduced tillage, grassland, restitution of crop residues). These estimates of soil organic carbon are converted in soil organic matter using a coefficient describing the carbon content of soil organic matter: from experimental data, Nelson and Sommers (1982) estimated that SOM content 58% carbon, and thus the conversion parameter from soil organic carbon to soil organic matter is $c_3 = 1.72$.

	Soil organic matter at equilibrium (adapted from Arrouays et al. (2002a), in ton/ha)
Cropland, conventional tillage, without crop residue restitution	55
Cropland, reduced tillage, without crop residue restitution	76
Cropland, conventional tillage, with crop residue restitution	69
Cropland, reduced tillage, with crop residue restitution	89
Grassland	112

We assume that only tillage regime and the land use affect mineralisation rates m_k. Mineralisation rates for cropland are taken from Mary and Guérif (1994), who give generic values for conventional and reduced tillage (resp. 1.5% and 1.9% per year), based on experiments in arable crop systems in Nothern France. For grassland, contradictory values were found, and we take the same value as cropland with reduced tillage.

Soil loss due to water erosion depends on land use (cropland vs. grassland), tillage regime and the restitution of crop residues. To assess the the soil organic matter lost due to erosion, we use the erosion rate, i.e. the proportion of soil eroded by water. We assume that the proportion of SOM lost due to erosion equals the erosion rate. Erosion rates λ_k are taken the work of the European Soil Data Center and its Pan-European Soil Erosion risk assessment (PESERA). In the largest arable crop areas in France, soil loss is estimated about 1t per ha per year (Eurostat, 2015). Considering a deep soil (30cm, in coherence with the type of soil from which Arrouays et al. (2002b) took values, as well as the measurement depth), with a 1.34t/m^3 density, which amounts to 4002t/ha, 1t/ha soil loss makes an erosion rate of 0.0249%. Considering than in these areas, intensive arable crop systems are the main land use, this value is associated in our model to cropland with conventional tillage. The same working group estimated the soil loss reduction due to soil conservation practices (Panagos et al., 2015), and according to their figures, conservation tillage decreases soil loss by 65%, and crop residues by another 20%. The same study estimates that the erosion rate in pastures is between 33% and 4 times lower than in a conventional wheat field, and from 2,5 up to 20 times lower for grasslands on poor soils. In the end, we keep an erosion rate on grassland which is 4 times lower than the reference value for conventional wheat field.

Organic matter inputs are calibrated with values calculated with parameters above and the soil dynamics equation to be coherent. They vary according to the tillage regime, which is coherent differentiated first mineralisation rates (Ancelin et al., 2008).

	mineralisation rate	erosion rate	organic matter inputs (t/ha)
Cropland, conventional tillage, without crop residue restitution	1.9%	0.025%	1.059
Cropland, conventional tillage, with crop residue restitution	1.9%	0.020%	1.321
Cropland, reduced tillage, without crop residue restitution	1.5%	0.009%	1.142
Cropland, reduced tillage, with crop residue restitution	1.5%	0.007%	1.348
Grassland	1.5%	0.006%	1.684

4.2. Nitrogen inputs

The average amount of synthetic nitrogen fertiliser applied in France is about 162kgN/ha, when no organic nitrogen is applied (Agreste, 2014). In our model, we consider among the most fertiliser intensive crops (wheat and rapeseed), and thus we take values fertiliser intensity ranging from 110 to 200 kgN/ha.

Nitrogen excreted by livestock is calibrated with the figures of Smith et al. (2000), who estimate the yearly nitrogen excretion of a dairy cow between 76 and 116 kg. Taking the mean value and assuming a livestock density of 0.65 units per ha (which corresponds to a rather extensive pasture), this amounts to a nitrogen input of 62kg per ha and per year.

	external nitrogen inputs
Cropland, no fertiliser	0
Cropland, fertiliser 1	110
Cropland, fertiliser 2	140
Cropland, fertiliser 3	170
Cropland, fertiliser 4	200
Grassland (without livestock impacts)	0
Grassland (with livestock impacts)	62

4.3. Nitrogen content of soil organic matter

We use parameters values and methodology from (Comifer, 2013) to convert the mass of mineralised soil organic matter into mass of mineral nitrogen. This parameter is calculated in two steps. First, the carbon content in soil organic matter $c_3 = 0.58$ (Nelson and Sommers, 1982) is used to convert mass of soil organic matter into mass of carbon. Second, the mass of carbon is converted into mass of nitrogen by means of the ratio of carbon to nitrogen $c_2 = 9$. Thus a mass unit of soil organic matter gives $\frac{c_3}{c_2} = 0.0646$ mass units of nitrogen.

4.4. Denitrification

The percentage of mineral nitrogen emitted as nitrous oxide through denitification, β, is equal to 1%, according to the default values for Tier 1 method in IPCC (2006).

4.5. Nitrogen uptake

In the nutrient uptake function, the recovery coefficient γ is calibrated at 0.87, and the curvature parameter ϵ equals 0.0032. These values are taken from Makowski et al. (1999), from which we took the functional form used.

The threshold of mineral nitrogen above which the function for nutrient uptake is not linear anymore (N^*) is calibrated at 206kgN/ha. This value has been derived from the estimates for parameters of the nutrient uptake function (Makowski et al., 1999) and the yield response function Monod et al. (2002). The original functional form from Makowski et al. (1999) links nutrient uptake with applied fertiliser, while our function links nutrient uptake with total mineral nitrogen available. The original function form has a fix amount of nitrogen taken up in the absence of applied fertiliser. Theoretically, this amount should correspond to the nitrogen provided by the soil (mineralisation of the soil organic matter), but it is much higher than the mineralised nitrogen predicted by our equation for soil organic matter, probably because our equation overlooks some additional sources of nitrogen (atmosphere). Nevertheless, we assume that in the absence of mineral nitrogen, the nutrient uptake would be 0, and keep this functional form, the same nitrogen recovery coefficient (which measures the part of the first units of nitrogen that are taken up by plants) and the curvature parameter. The threshold above which the nitrogen uptake function changes form is recalibrated: it equals the original amount of applied fertiliser plus the average amount of nitrogen supplied by the soil in our model.

5. Greenhouse gases

Methane emitted by livestock is calibrated from the French case in Soussana et al. (2007), corresponding to an extensive and non fertilised grassland with 0,65 livestock unit per ha per year. Methane emissions from cattle on a pasture are thus calibrated at 62kg/ha.

The diverse forms of nitrogen have different molar masses, so that parameters are needed to convert mass of the different forms. Mineral nitrogen present in the soil is expressed as mass of elemental nitrogen, and a factor $c_4 = 1.57$ is used to convert it into mass of N_2O, following the methodology detailed IPCC (2006). Similarly, one mass unit of carbon gives $c_5 = 3.66$ mass units of carbon dioxide.

Besides, to be able to derive one synthetic indicator for the global warming power of greenhouse gases, they have to be expressed in the same unit. Following the IPCC methodology (Myhre et al., 2013), we attribute a global warming potential $g_1 = 298 CO_2 eq$ to N_2O and a $g_2 = 34 CO_2 eq$ to methane.

We assume that the only practices affecting fossil fuel burning are the tillage regime and the land use. Reduced tillage incurs less field work, and thus lower carbon emissions dur to machinery. We calibrate this parameter with the values found by Robertson (2000), and cited by Arrouays et al. (2002a) in their extensive assessment. Emissions due to fossil fuel burning are thus assumed to equal $FC_{T=1} = 160$ kgCO2eq/ha for conventional tillage and $FC_{T=0} = 120$kgCO2eq/ha for reduced tillage. Fossil fuel burning for alfalfa cultivation, which represents temporary grassland, is assumed to be $FC_{G,P} = 8kgCO2eq/ha$.

6. Water quality

Pesticide intensity is expressed in an aggregated index used in France, and which measures the applications of all pesticides compared to specific allowed application. Values for this parameter are taken from previous work (Barraquand and Martinet, 2011), and correspond to the observed values. In our model, high and medium pesticide intensities are represented through an index of 6 and 3 respectively, while the average intensity in France for winter wheat was between 4.9 and 5.6 in intensive production areas (Agreste, 2013), and the technical institute for pesticide reduction gives as examples a conventional scenario with an intensity of 5.06 and a pesticide reduction scenario with an intensity of 1.93 (ONEMA, 2011).

The filtering efficiency of semi-natural elements, w, is calibrated to be 40%, based on specific studies: Dosskey (2001) carry out an extensive literature review, and find that semi-natural elements lead to reduced pollutant concentration in water bodies between 40 and 100% of sediments and generally more than 40% of nitrates and pesticides.

7. Agricultural production

Parameters for the yield function are calibrated from Makowski et al. (1999), adapted to account for the nitrogen mineralised from the SOM. The marginal effect of nitrogen is calibrated at $n_2 = 0.015$.

e, the proportion of the field cover by semi-natural elements, is assumed to be 5% if they exist. This percentage corresponds for example to a strip of 5m on one side of the field (length = 100m), which is in line with recommendations by the chambers of agriculture (Chambre d'agriculture des Hauts-de-France, 2008), and to the minimal amount of zones of ecological interest required by the new common agricultural policy for receiving direct green payments.

The ratio of crop residues (straw) to grain yield, ρ, is between 0.9 and 1.4 for conventional crop systems, with modern wheat varieties (Chambre d'agriculture Midi-Pyrénées, 2007). We keep a conservative value of 1.

Grassland in our model represents an environment-friendly alternative, and corresponds to extensive grasslands, that can be found even on poor soils. We calibrate the production on grasslands at 6 ton dry matter per ha. We assume that its production doesn't depend on soil quality: even on rather bad soils, choosing appropriate species enables to produce a quite similar amount of hay. An extensive grassland (no pesticides, no fertiliser, no tillage, low grazing intensity of grazed) can produce around 6 or 7t dry matter per ha, according to (FRAB Midi-Pyrénées, 2011; Dephy Ecophyto, 2014a,b).

8. Profit

As an indicator for profit, we choose gross margin, which is the difference between receipts (from the sale of crop, grain and fodder) and the operating costs.

8.1. Prices

The price for grain is first assumed to be around 170euro per ton, which is the average price over mid-2010 to mid-2015 for wheat in France, statistics given by FranceAgrimer (FranceAgrimer, 2016).

Crop residues (straw), if exported, are valued at 20euro per ton, a price corresponding to straw to be collected on the field. [52]

The price of fodder (hay) is calibrated at 120 euro per ton, price recommended in 2016 by the Chamber of agriculture of Picardie in France[53].

8.2. Management costs

Management costs are calibrated with values for operating costs for different practices. Operating costs cover only costs linked to mechanisation, fertilisers, pesticides, and generic variable costs. For example, the wages or rent paid for the use of capital are not considered as operating costs. They can be considered as variable costs, meaning that they can be directly associated to the cultivationof one hectare of land.

Mechanisation costs vary with the tillage regime and land use. Mechanisation costs (including fuel and machinery use) is calibrated at 270 euro/ha for conventional cropland, and at 250euro/ha for reduced tillage, on the basis of case studies from the network for pesticide reduction in France[54]. Based on the same case studies, the mechanisation cost for grassland is calibrated at 150euro/ha. These values don't reflect exactly the reduction in carbon dioxide emissions between the different management options, given that the mechanisation costs account mainly for the costs of machinery use, which may not be proportional to fossil fuel burning.

Based on previous work (Barraquand and Martinet, 2011), the price of fertiliser is calibrated at 1,15 euro/kgN. Although this value dates back to some years, there is no clear upward trend of price of fertilisers, this price rather fluctuates with the price of major crops (INSEE, 2016). The cost of fertilisers is obtained by multiplying this price by the applications.

Costs for pesticides are calibrated on the basis of previous work: each dose of pesticide (corresponding to the French treatment index) is assumed to cost 33 euro according to calculations of (Butault et al., 2011), while medium and high pesticide-intensive cropland

[52] http://www.hautsdefrance.chambres-agriculture.fr/exploitation-agricole/gerer-son-exploitation/fermages-baremes/bareme-fourrages/

[53] ibid.

[54] http://grandes-cultures.ecophytopic.fr/gc/innovation-en-marche/fermes-dephy/r%C3%A9seau-de-fermes-dephy-ecophyto-des-syst%C3%A8mes-de-cultures

are associated resp. to 3 and 6 doses of pesticides per ha, which makes up resp. 99 and 198 euro/ha.

We assume no difference in costs (else than opportunity costs of selling the straw) between leaving crop residues on the field and exporting them, since the price for crop residues is calibrated on the basis of straw to be collected on the field.

Cost of implementing semi-natural elements are calibrated to 35 euros for a 5m by 100m strip, on the basis of the estimated cost for flower strips by agricultural or environmental institutions (Chambre d'agriculture des Hauts-de-France, 2008; Agrifaune, 2012).

Other operating costs included in our analysis, which apply to any management option, are taken from the EU report on farms (European Commission - Agriculture and Rural Development, 2013), based on wheat production in France, and amount to 197 euro per ha (66 euro for seeds, 7 for energy (other than machinery fuels) and 124 for other costs).

type of cost	vary with	values
fertilisation costs	fertiliser intensity	1.15 euro/kgN
pesticide costs	pesticide intensity	33 euro/FTI (dose)
mechanisation costs	tillage regime, land use	270 euro/ha for conventional tillage, 250 euro/ha for reduced tillage, 150 euro/ha for grasslands
semi-natural elements	semi-natural elements	35euro/ha if semi-natural elements are implemented
other operating costs	/	197 euro/ha

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ÉCOLE DOCTORALE
Agriculture, alimentation, biologie, environnement, santé (ABIES)

Titre : Incitations économiques pour la régulation de la fourniture de bouquets de services écosystémiques dans les agroécosystèmes

Mots clés : services écosystémiques, agriculture, incitations économiques, politiques agroenvironnementales, biens publics, production jointe

Résumé : Les agroécosystèmes font face à un déclin des services écosystémiques (SE) de régulation, non-marchands. Nous l'interprétons via deux concepts économiques : les biens publics qui appellent une régulation, et la production jointe qui souligne les conséquences des interactions entre SE dans leur régulation.
Cette thèse étudie comment accroître la fourniture de SE non-marchands par des incitations économiques, en prenant en compte la multiplicité des SE et les interactions entre eux.
Nous étudions d'abord la régulation des biens publics joints à l'aide de microéconomie théorique. Ensuite, nous menons une analyse appliquée avec des données agroécologiques simulées et des méthodes numériques pour définir les solutions coût-efficientes et les incitations pour leur mise en œuvre. Nous comparons plus particulièrement les incitations basées sur les actions et sur les résultats.
Nous montrons théoriquement que les interactions entre SE rendent leur régulation plus complexe, notamment avec des incitations basées sur les résultats, et quand le coût varie selon les bouquets de SE. Dans l'analyse appliquée, nous montrons que prendre en compte le coût de la fourniture des SE est crucial pour maximiser leur fourniture avec un budget limité. Nous montrons que les incitations basées sur les résultats sélectionnent les solutions coût-efficientes mais induisent un budget plus élevé que les incitations basées sur les actions, à cause des interactions entre SE. Enfin, nous montrons que l'analyse à l'échelle du paysage et l'hétérogénéité modifient les solutions qui maximisent les SE, mais pas les propriétés des deux types d'incitations.
Nos résultats soulignent que les politiques agro-environnementales doivent cibler les services écosystémiques de manière intégrée, si possible à l'échelle de la ferme ou du paysage et considérer le coût de leur fourniture. Les incitations basées sur les résultats ne sont pas la solution à tous les problèmes des politiques agroenvironnementales.

Title : Economic incentives for the regulation of the provision of bundles of ecosystem services in agroecosystems

Keywords : ecosystem services, agriculture, economic incentives, agri-environmental policies, public goods, joint production

Abstract : Agroecosystems show a decline in regulating, non-marketed ecosystem services (ES). We interpret this decline through two economic concepts: public goods, which call for regulation, and joint production, which underlines the role of interactions among ecosystem services in their regulation.
This thesis studies how to increase the provision of non-marketed ES through the implementation of economic incentives, while accounting for their multiplicity and the complex interactions among them.
We first study the regulation of joint public goods with microeconomic theory. We then carry an applied analysis with simulated agroecological data and numerical methods to define cost-efficient solutions and simulate the implementation these solutions with economic incentives. We especially compare result-based and action-based incentives.
We show theoretically that interactions among ES make their regulation more complex, especially with result-based incentives and when the production cost varies among bundles of ES. In the applied analysis, we show that accounting for the cost is crucial to maximise ES with a limited budget. We show that result-based incentives select cost-efficient bundles of ES but lead to higher policy budgets than action-based ones, due to interactions among ES. Eventually, we show that considering the landscape scale and heterogeneity plays on the solutions maximising ES, but not on the comparison between result-based and action-based incentives.
Our results underline that agri-environmental policies need to target ES in a integrative way, at the farm or landscape scale, and consider the cost of providing non-marketed ES. Result-based incentives don't solve all issues of agri-environmental policies.

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