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Climate policy in the agricultural sector: effectiveness, monitoring costs, carbon sequestration and risk

Roxane Photinodellis

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Politique climatique dans le secteur agricole : efficacité, coûts de surveillance, séquestration du carbone et risque

Climate policy in the agricultural sector: effectiveness, monitoring costs, carbon sequestration and risk

Thèse de doctorat de l'université Paris-Saclay

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Titre : Politique climatique dans le secteur agricole : efficacité, coûts de surveillance, séquestration du carbone et risque

Mots clés : Changement climatique, Atténuation, Economie de l'environnement, Economie agricole

Résumé : L'objectif de cette thèse est d'explorer l'arbitrage entre l'efficacité des politiques d'atténuation des émissions de GES nettes et les coûts associés au suivi des émissions de GES et à la séquestration du carbone. Dans la première contribution, notre objectif est d'estimer le coût marginal et l'efficacité en termes d'atténuation des émissions de GES nettes de plusieurs politiques couvrant partiellement les sources et les puits d'émissions. Nous constatons que la subvention pour la séquestration totale du carbone et celle pour le carbone contenu uniquement dans la biomasse aérienne se révèlent plus coût-efficaces qu'une taxe sur les GES. Dans la deuxième contribution, à l'échelle de la parcelle, nous comparons les coûts de participation de deux types de contrats carbone dont le suivi est basé pour l'un sur les prédictions d'un modèle et pour l'autre sur les résultats mesurés. Pour ce dernier, nous prenons en considération l'aversion au risque des exploitants face à l'incertitude entourant les résultats de séquestration du carbone. L'objectif est de mettre en évidence les arbitrages concernant le partage du risque entre le régulateur et l'exploitant, tout en définissant des conditions théoriques de participation aux contrats pour ce dernier. Dans la troisième contribution, à l'échelle d'une région française, nous comparons la coût-efficacité de contrats carbone dont le paiement est basé sur les résultats mesurés, sur les prédictions d'un modèle, ou sur la mise en œuvre de la pratique. Cette analyse prend en compte une partie de l'hétérogénéité entre les agriculteurs. Nos résultats indiquent que la forme de contrat la plus coût-efficace consiste à rémunérer les exploitants à l'hectare pour la mise en place d'agroforesterie intra-parcellaire.

Title : Climate policy in the agricultural sector: effectiveness, monitoring costs, carbon sequestration and risk

Keywords : Climate change, Mitigation, Environmental economics, Agricultural economics

Abstract : The aim of this thesis is to explore the trade-off between the effectiveness of net GHG mitigation policies and the costs associated with monitoring emissions and carbon sequestration. In the first contribution, our objective is to estimate the marginal cost and effectiveness in terms of mitigating net GHG emissions for several policies partially covering emission sources and sinks. We find that subsidies for total carbon sequestration and for carbon contained only in above-ground biomass are more cost-effective than a GHG tax. In the second contribution, at the scale of the plot, we examine the costs of participation in two types of carbon contracts, one model-based and the other results-based. For the latter type, we take into account the risk aversion of farmers regarding the uncertainty surrounding carbon sequestration results. The goal is to highlight the trade-offs regarding risk-sharing between the regulator and farmers, while defining theoretical conditions for farmer participation in these contracts. In the third contribution, at the scale of a French region, we compare the cost-effectiveness of different carbon contracts, whether results-based, model-based, or practice-based. This analysis takes into account some of the heterogeneity among farmers. Our results indicate that the most cost-effective contract form is to remunerate farmers per hectare for the implementation of intra-parcel agroforestry.

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Chapter 1

General introduction

In 1979, in Geneva, the first world conference on climate took place, a decisive event that led to crucial conclusions for our understanding of climate change. This conference highlighted that abnormal concentrations of carbon dioxide (CO_2) in the atmosphere, attributed to human activities, are responsible for a foreseeable long-term increase in temperatures, a phenomenon capable of profoundly disrupting our climate.

In 1990, the Intergovernmental Panel on Climate Change (IPCC) published its first report, confirming the initial concerns and shedding light on the severe consequences of global warming. This report anticipates an increase in natural disasters, such as violent storms, cyclones, and rising sea levels, with devastating impacts on ecosystems and human communities. One of the most concerning aspects emphasized by this report is the "irreversible" nature of climate change [Stanhill, 1992].

Faced with these findings, the Kyoto Protocol, in 1990, marked the beginning of an international commitment to reduce greenhouse gas emissions (GHGs). This initiative was followed in 2015 by the Paris Agreement (during COP21) where several nations, including France, committed to limiting the increase in global temperatures to 2°C

above pre-industrial levels. The Paris Agreement was the first to introduce the concept of "carbon neutrality" on a global scale, in accordance with the recommendations of the IPCC and the United Nations Framework Convention on Climate Change (UNFCCC). Carbon neutrality entails a balance between GHG emissions from human activities and those removed from the atmosphere, through natural carbon sinks. To contribute to the GHG mitigation effort, France is committed to achieving carbon neutrality by 2050 through all sources and sinks of GHG emissions in all sectors of activity [SNBC, 2020].

The agricultural sector plays a pivotal role in national commitments to address climate change. Indeed, it stands as the second-largest GHG emitter in France, contributing to 16% of national emissions [Citepa, 2023]. It is the main source of GHG emissions other than carbon dioxide (CO_2): it emits methane (CH_4) and nitrous oxide (N_2O), which have a Global Warming Potential (GWP) over 100 years approximately 28 times and 264 times higher than CO_2 [IPCC, 2013]. In 2021, the agricultural sector accounted for 23% of total CO_2 national emissions, 69% of CH_4 emissions, and 84% of N_2O emissions (excluding Land use, land-use change, and forestry (LULUCF)) [Citepa, 2023]. In parallel, the agricultural sector holds the potential to sequester carbon in its soils, thus serving as a natural carbon sink. It is an ideal opportunity to achieve the goal of carbon neutrality, as it offers a low-cost solution compared to mitigating greenhouse gases in other sectors, while serving as a means of offsetting unavoidable emissions. This has given rise to the concept of carbon farming, which involves compensating farmers for their accountable carbon sequestration efforts. The GHG mitigation produced is then sold on the voluntary carbon market in the form of a credit. This perspective has led to the emergence of carbon certification frameworks and labels such as, for example, the 'Low Carbon Label' in France. This system enables companies and institutions to purchase carbon credits and voluntarily offset their emissions. This has sparked the

European Commission's interest in establishing a European certification framework for carbon farming, whose design is currently in progress.

Thus even though several projects are underway, there are currently no policies in France or Europe ambitious enough to achieve the mitigation objectives in the agricultural sector. How, then, can farmers be encouraged to mitigate net greenhouse gas emissions in this sector? Furthermore, considering the sector's multitude of emission sources and sinks, and the significant heterogeneity among farmers in terms of costs and potential for net GHG mitigation, how to monitor these GHG emissions and carbon sequestration cost-effectively ?

This introductory chapter aims to provide an overview of the economic aspects of incentive policies to encourage the implementation of practices aimed at mitigating net GHG emissions in the agricultural sector. First, I examine the technical levers for mitigation in this sector, describing the methods for estimating their mitigation potential and their implementation costs. Next, I discuss the methods for regulating net GHG and the challenges associated with their implementation, with a focus on monitoring. Finally, I explore the issues associated with the development of carbon farming schemes, which, due to the voluntary nature of carbon farming, raise questions about participation in the contract and the risks to farmers.

1.1 Technical levers for GHG mitigation

Biofuels produced from agricultural matter can help mitigate GHG emissions by substituting for the use of fossil fuels and sequestering carbon in soils and biomass. However, there is already an extensive literature on biofuels [OECD, 2006, Searchinger et al., 2008, Kim et al., 2009, Gabrielle et al., 2014, Fradj et al., 2016], and this solution is contro-

versial due to undesirable effects such as competition with food production and the risk of CO_2 emissions leakage.

Thus, in this thesis, I focus on management practices applicable at the farm scale that lead to a reduction in GHG emissions and/or carbon sequestration in the soil and above-ground biomass, which can be termed mitigation practices.

The primary sources of GHG emissions in the french agricultural sector are CH_4 produced by enteric fermentation by livestock and N_2O emissions from managed soils due to fertilizer application [Citepa, 2023]. Consequently, emission reduction practices primarily target these two sources.

Mitigation practices involve more or less significant changes in agricultural production. Indeed, GHG reductions can be achieved by reducing the size of cattle herds or converting cropland to grassland, which has a significant impact on agricultural supply. Other practices require less substantial adjustments in activities and in volumes produced, and have been identified in the literature. Some works consider a set of options deemed capable of reducing GHG emissions and/or sequestering carbon, as in the case of [Smith et al., 2008a] on a global scale and [Pellerin et al., 2017] which identifies a portfolio of 10 mitigation practices on a French scale. Some others focus on livestock practices [Clemens and Ahlgrimm, 2001, Dollé et al., 2011, Doreau et al., 2014, Lombardi et al., 2021], manure management practices [Chadwick et al., 2011], land management practices [Eagle and Olander, 2012], nitrogen fertilization [Bouwman, 2001] or on carbon sequestration [Lal, 2013, Chenu et al., 2014, Singh et al., 2018, Bamière et al., 2023]. This work shows that the costs of implementation and the mitigation potential of mitigation practices depend on the pedoclimatic and economic conditions. Thus, there is no exhaustive list of mitigating practices to implement. We give a few examples of practices identified at the scale of metropolitan France by [Pellerin et al., 2017,

[Bamière et al., 2023] according to several criteria (a priori positive mitigation potential, possible large-scale implementation, social acceptability, etc.). Some practices involve adjustments in input use, such as optimizing or reducing nitrogen fertilizer application, or altering livestock feed rations to favor systems with less cereals and more grass, along with changes in livestock management by extending grazing periods. Emissions reduction is also achievable through manure management, particularly through methanization to produce renewable energy. Lastly, other options for carbon sequestration in soils and biomass include practices like no-tillage, the implementation of cover crops, the introduction of grassland into crop rotation, hedgerow planting, and the adoption of agroforestry systems.

When assessing the mitigation potential of practices, some studies focus solely on direct emissions, which are defined as emissions occurring in the field, emitted directly by the farm's agricultural activities, such as CH_4 from livestock enteric fermentation or N_2O from soil management practices. Other studies, take into account both direct emissions and indirect emissions upstream of agricultural production. Indirect emissions include those associated with the production and the transport of inputs such as animal feed and fertilisers, which contribute to the overall carbon footprint of the agricultural system. [Pellerin et al., 2017] and [Bamière et al., 2023] take into account both direct and indirect emissions.

Due to changes in input consumption resulting from its implementation, a practice may sequester carbon but increase GHG emissions, and conversely, it may sequester carbon and also mitigating GHG emissions. As an example, [Alemu et al., 2017] revealed that practices aimed at enhancing grazing management by extending the grazing period not only reduce N_2O emissions by decreasing the need for fertilizers but also result in lower emissions by reducing the use of concentrates in animal feed ration in favor of

grazing. Conversely, agroforestry, while contributing to the sequestration of carbon in the soil and biomass, increases fuel consumption for tree planting and maintenance, and therefore CO_2 emissions. Furthermore, a moderate intensification of permanent pasture can enhance soil carbon sequestration but also increases N_2O emissions due to the increased fertilizer application [Bamière et al., 2023]. Moreover, mitigation practices can exhibit synergistic or antagonistic interactions when implemented simultaneously on the same plot. For instance, [Launay et al., 2021a] demonstrate that the combination of cultivating cover crops and applying new organic resources can lead to higher mitigation in net GHG emissions compared to the sum of their individual potentials.

It is therefore essential to take into account all the sources and sinks of GHGs when assessing the mitigation potential of a given practice.

The potential for mitigating net GHG emissions from a change in practice depends on characteristics of the farm and its initial situation, such as the animal species present, their numbers, and current management practices, including feed rations, fertilization intensity, grazing duration, etc. Since it is unthinkable to measure GHG emissions in the field ¹, typically, these management data are cross-referenced with emission factors derived from national inventories using IPCC formulas [Eggleston et al., 2006], in order to estimate the variations in GHG emissions resulting from the implementation of a practice. Uncertainties are associated with these emission factors depending on the data used and their accuracy. For example, in the case of emissions linked to a cow's enteric fermentation, the accuracy with which the breed, exact feed rations, etc. are taken into account will have a direct impact on the accuracy of the associated emissions factor.

For carbon sequestration, the potential of a practice is highly dependent on soil

¹Even if it is possible to measure emissions directly using techniques such as flow chambers, this is very costly and unthinkable to implement for large-scale monitoring of GHG emissions.

and climatic conditions and has some specific characteristics. Firstly, the sequestration potential of a practice can be defined as the difference between the storage linked to the implementation of the practice and the storage that would have occurred without the practice. Secondly, for a given practice carbon sequestration has a limited potential. Indeed, each sequestration practice has an equilibrium state that depends on the soil on which it is applied. In other words, once this equilibrium is reached, it is not possible to store additional carbon with this practice. Thirdly, carbon sequestration is not permanent. This means that when the practice is stopped, the stored carbon is released into the atmosphere. The rate at which it is de-sequestered is even faster than the rate at which it is sequestered. Finally, carbon sequestration is a dynamic process with an exponential trajectory at the beginning of the adoption of a practice, gradually approaching its limit. The mitigation potential of a carbon-sequestering practice is thus realized progressively. As a result, there is a time lag between the moment a practice is adopted and the moment it reaches its full carbon sequestration potential. These characteristics make carbon sequestration a complex biological process, with significant spatial and temporal variability, making it difficult to estimate and monitor stocks.

Soil carbon sequestration can be assessed by field measurements, but these results are site-specific and this method is expensive. Another, less costly approach to estimating carbon sequestration is to use model predictions. There are two main types of models to assess carbon sequestration: statistical models and dynamic models, also known as mechanistic models. Statistical models involve an equation in which SOC is estimated on the base of parameters related to various factors such as temperature, precipitation, current soil management, etc. [Barracough et al., 2015, Gray and Bishop, 2016, Meersmans et al., 2016, Yigini and Panagos, 2016, Sanderman et al., 2017]. As for dynamic models, they have existed for a long time. The first one was created in 1945

by Hénin and Dupuis, who approximated carbon storage with an exponential function [Hénin and Dupuis, 1945]. These dynamic models differ based on the mechanisms considered, such as the number of organic matter reservoirs. Additionally, they vary in terms of spatial and temporal scales, ranging from a few micrometers to thousands of kilometers and from a few hours to centuries [Manzoni and Porporato, 2009]. Among the most well-known models are the carbon stock evolution models RothC [Andrén and Kätterer, 2001], AMG [Andriulo et al., 1999] and ICBM [Coleman and Jenkinson, 1996], as well as crop models like MONICA [Nendel et al., 2011], STICS [Brisson et al., 1998], DayCent [Ogle et al., 2010], DSSAT [Jones et al., 2003], Century [Parton et al., 1987], and specific grassland models such as PaSim [Calanca et al., 2007]. These dynamic models have the ability to account for pedoclimatic heterogeneity by using parameters related to soil texture, irrigation practices, and temperature. However, these models estimate carbon stocks with a range of uncertainty that depends not only on the parameters used but also on the formulation of the equations, since despite ongoing developments [Campbell and Paustian, 2015], our understanding of the biochemical processes involved in carbon sequestration remains incomplete.

Thus, even if it is associated with uncertainties, assessing the mitigation potential of practices makes it possible to quantify the technical mitigation potential in the agricultural sector. Technical potential refers to the intrinsic capacity of practices to reduce GHG emissions and/or sequester carbon, regardless of economic, social or political constraints. [Smith et al., 2008b] estimate that the global technical GHG mitigation potential achievable in 2030 by agriculture is around 5500 to 6600 $MtCO_2e.yr^{-1}$. [Murray, 2004] reports that the technical potential for carbon sequestration in US cropland ranges from 300 to 550 $MtCO_2e.yr^{-1}$. On the French scale, [Pellerin et al., 2017, Bamière et al., 2023] estimate the technical mitigation potential of each practice by estimating a maximum

technical base for each practice (which is defined by the areas and numbers of animals on which a practice can be applied, taking into account any technical constraints such as soil and crop characteristics, or the age of the animals). For example, for the introduction of legumes, which reduce nitrogen requirements, the technical potential is estimated at depending on the estimation method, 1.30/1.75 $MtCO_2e.yr^{-1}$ for France as a whole, if it is introduced on arable crops, and 1.24/0.92 if introduced into temporary grassland.

The implementation of a mitigation practice is associated with costs entailed by management changes. They can be variable (due to variations in the use of inputs, yield losses or gains, sale of new products such as wood for agroforestry or electricity for methanisation), or fixed (such as the cost of planting trees and hedges, the cost of purchasing dedicated equipment, etc.). Consequently, like GHG emissions, the estimation of these costs depends on the characteristics and initial situation of the farms. Previous studies show that implementing mitigation practices is generally costly for farmers. For example, according to [Bamière et al., 2023], in mainland France, implementing agroforestry requires a fixed cost of 1007 $€\cdot ha^{-1}$ associated with soil preparation, protection and mulching and sowing under the tree, and an annual cost of around 50 $€\cdot ha^{-1}$, which corresponds to the difference between the cost of maintaining the trees and the profit from the sale of pruned wood. But for some practices, the implementation costs can be negative, resulting in savings for the farmers, often attributed to reduced input use. Nevertheless, despite these potential advantages, these practices are not widely adopted by farmers. This can be attributed to various barriers that increase the adoption costs of these practices for farmers. These barriers, as studied in the literature, take on different forms, including financial constraints like cash flow or investment capacity when fixed costs are significant, risks associated with yield variability due to the practice implementation [De Pinto et al., 2013, Acosta et al., 2019], limited resources

such as labor, and private transaction costs (costs related to information acquisition, time dedicated to training, and administrative procedures necessary for implementation). [Bamière et al., 2014] estimate the private transaction costs on average in France for each of the practices identified by [Pellerin et al., 2017], using a formula defined by [Mettepenningen et al., 2007], which takes into account the farm's standard gross output, its size (in hectares for crop practices and by herd size otherwise), and the farmer's level of education. The estimated private transaction costs range from 9 to 72 €/per hectare per year for crop practices and from 3.8 to 6 €/per head per year for animal practices. For example, the introduction of legumes in temporary grassland has an estimated technical implementation cost of $-31.4 \text{ €} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ per hectare per year, with a private transaction cost estimated at $38.9 \text{ €} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$, resulting in a total implementation cost of $7.5 \text{ €} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$.

Therefore, farmers do not adopt mitigation practices, or do so insufficiently, due to various associated implementation costs, whether they are technical, economic, social or cultural. However, since mitigating GHG emissions is necessary for public welfare, and since public interests do not match farmers' private interests, public intervention is then necessary.

1.2 Public regulation of agricultural GHG emissions mitigation

In the agricultural sector, public policies can be designed to influence either consumption [Wirsenius et al., 2011, Camilleri et al., 2019, Muller et al., 2019, Bazoche et al., 2023], (i.e. the food choices of individuals and households that indirectly impact production

choices through an effect on demand), or production directly, (i.e. the adoption of agricultural practices by farmers). In this thesis, my focus is on agricultural policies related to agricultural production.

Public policies aimed at promoting the adoption of net GHG mitigation practices in the agricultural sector can be of different types. They can be informative in nature, involving training or information campaigns to raise awareness of the importance of adopting these practices. Or these policies can be regulatory in nature, taking the form of economic instruments or norms. The economic instruments can be based on prices [Pigou, 1932], such as subsidies and taxes, or on quantities [Dales and Pollution, 1968] with emission permit markets. This thesis concentrates on economic instruments. Indeed, the literature suggests that this form of regulation is more effective because of its ability to offer greater flexibility, which is particularly appropriate given the heterogeneity of the farmers involved (economic, behavioural, and linked to pedoclimatic conditions) [Newell and Stavins, 2003]. In this way, each farmer is assumed to be rational and to choose the means of mitigation that minimises her/his total mitigation costs.

When designing economic instruments, it is necessary to find the most cost-effective way of internalising the externality represented by GHG emissions. The aim is to achieve a mitigation target at the lowest cost. The cost-effectiveness of a policy depends in particular on the type of instrument and its level. Pigou's theory states that the optimum level of a tax in the Pareto sense is equivalent to the cost of the social damage caused by the externality [Pigou, 1932]. Such an optimal carbon price can only be estimated with perfect knowledge of the marginal cost of the damage to society and the marginal cost of depolluting for farmers, as inadequate estimates can lead to the implementation of an instrument that does not provide sufficient incentives or leads to distortions. However, these informations are difficult and costly to obtain in practice,

because of the multitude of sources and heterogeneity of the agents.

Thus, instead of estimating a Pareto-optimal instrument level, the economic literature has rather focused on estimating the marginal cost of mitigation for various levels of instruments/types of regulations. Indeed, there is a vast literature on the analysis of the marginal costs of mitigating GHG emissions, based on different assumptions about the mitigation practices considered and estimates of their costs and mitigation potential. There are four main approaches to estimating them, each with its own characteristics that lead to differences in estimates of marginal abatement costs [Vermont and De Cara, 2010].

First, the engineering or bottom-up approach consists in estimating the technical cost and the mitigation potential of each practice taken individually [DeAngelo et al., 2006, Beach et al., 2008, Smith et al., 2008a, Schulte et al., 2012, Beach et al., 2015, Herrero et al., 2016], then in ranking practices by increasing abatement cost. That is how [Henderson et al., 2017] estimate that on a global scale, with the adoption of practices concerning only ruminant breeding, it is possible to achieve an abatement of $379 \text{ MtCO}_2e.an^{-1}$, two-thirds of which can be obtained at a carbon price of $20\$.tCO_2e^{-1}$. At the scale of the United Kingdom, [Moran et al., 2011], estimate that around 6.36 MtCO_2e can be avoided in 2022 for a carbon price equal to or less than $0 \text{ €}.tCO_2e^{-1}$. The engineering modelling approach is an ideal basis for comparing the technical cost and mitigation potential of various practices. But it does not take into account the opportunity costs associated with adopting practices.

The general and partial equilibrium approach, [Schneider et al., 2007, Van Doorslaer et al., 2015], accounts for respectively one and more sectors, and considers prices as endogenous. It is thus possible to capture market effects. But the partial equilibrium approach does not take into account the indirect repercussions of a mitigation policy on other sectors. The

general equilibrium approach takes account of these repercussions, but requires more data, linked to different sectors, which is likely to impact on the uncertainty of the results obtained.

The last approach is the micro-economic modeling of the supply side [De Cara et al., 2005, Ramilan et al., 2011, Lengers and Britz, 2012]. This approach looks at how production activities and farming practices can be modified to mitigate GHG emissions. Prices are considered to be exogenous. However, this approach does not take into account the effects of changes in supply on production prices. But it is the most appropriate for modelling the adoption of mitigation practices in agriculture, because it captures the individual behaviour of farmers, and therefore takes into account the opportunity costs of adopting different practices. It generally requires less data than the general and partial equilibrium approaches, and can be based on previously estimated detailed technical mitigation costs and potential.

Despite this abundant literature, which aims to guide policy-makers in implementing cost-effective policies to mitigate net GHG emissions in the agricultural sector, the measures currently in place in France, and more generally around the world, are limited and insufficient. Within OECD nations, around 60% of emissions are presently not covered by any carbon pricing mechanism [OECD, 2021].

Indeed, several measures at the European level are likely to indirectly impact GHG mitigation, often through the restriction of organic fertilizer or total fertilizer application. Notable examples include the Nitrate Directive, the European Commission's Biodiversity Strategy for 2030, and, at the French level, the Agricultural Recovery Plan, Climate and Resilience Law, and the national program for the management of vinicultural effluents. However, only the agri-environmental and climate measures (AECM) under Pillar II of the Common Agricultural Policy (CAP) are specifically aimed at directly mitigating

GHG emissions and sequestering carbon in the agricultural sector. These measures aim to reward farmers for implementing "environmentally friendly" practices. Originally designed to preserve biodiversity by limiting pesticide and water use while also protecting soils, a climate component was added during the 2014-2020 reform. AECMs were continued for the subsequent period, 2021-2027. In practical terms, AECMs involve voluntary five-year commitment contracts, for which farmers are remunerated based on the enrolled hectares or herd size. Practices vary between countries; in France, they include reducing the use of nitrogen fertilizer, minimizing soil tillage, implementing cover crops, adopting diets that reduce methane production, etc.

But these AECMs are not intended to achieve the sector's mitigation objectives on their own. In addition, their effectiveness in mitigating GHG emissions is being called into question. According to a special report from the European Court of Auditors ² published in 2021, while during the 2014-2020 period, around 100 billion euros of CAP funding was devoted to climate change mitigation, the sector's GHG emissions show no significant change since 2010. In addition, [Fanchone et al., 2022] carried out an ex-ante analysis at farm level of the new AECMs for the period 2021-2027 in Guadeloupe and show that it is not easy to articulate various environmental objectives in addition to local constraints. This article shows that the new generation of AECMs focuses more on reducing pesticide use than on mitigating emissions. Thus, the implementation of these AECMs could lead to an increase in GHG emissions due to the increased use of diesel for mechanical weeding and imports of inputs (imports of diesel and brush cutters for mechanical weeding).

There are several barriers to the introduction of more ambitious and effective cli-

²Special Report, Common Agricultural Policy and climate : Half of EU climate spending but farm emissions are not decreasing, 2021

mate policies in the agricultural sector. Among these barriers is the risk that climate policy will lead to an increase in income inequality [Ohlendorf et al., 2021]. Moreover, although the burden of reducing emissions falls on the current generation, the benefits in terms of reducing GHG emissions will be perceived mainly by future generations, creating a gap between current efforts and the long-term benefits associated with avoiding or slowing climate change [Tebaldi and Friedlingstein, 2013]. Another complication is in the measurement of GHG emissions. Since GHG emissions are by definition diffuse pollution, like all types of pollution, measuring them accurately is very costly, and unthinkable. Because GHG emissions and the costs of reducing them on a farm come from many sources, and as described above are dependent on the farm's own characteristics (UAA, herd size, soil and climate conditions, etc.) and the initial situation (history of practices, practices currently in place such as fertiliser doses, livestock feed rations, etc.). However, these informations are expensive to obtain.

It is on this challenge, which consists of the trade-off between the costs of monitoring and the effectiveness of policies in terms of mitigating net greenhouse gas emissions, that the problematic of this thesis focuses.

The first contribution of this thesis concerns policy coverage. Indeed, monitoring costs partly depend on policy coverage. The economic literature explores alternatives to an instrument that taxes/subsidises all GHG sources and sinks. [Garnache et al., 2017] studies, on the scale of California, the marginal mitigation costs and mitigation level of several second best policies, such as taxes covering different sources of emissions, CH_4 , N_2O and CO_2 , and subsidies for the implementation of different mitigation practices. [De Cara et al., 2018] investigates the optimal coverage of a tax, covering the largest emitters as a priority, and taking MRV costs into account, on a European scale. They study the trade-off between increasing efficiency in terms of mitigating greenhouse gas

emissions by considering a broader coverage and the rise in MRV costs when the tax is applied to a larger number of farmers. In northern France, [Durandeau et al., 2010] compares the results of a first best policy with a tax both on livestock size and fertiliser inputs. They show that the second best policy achieves 8 % mitigation at a carbon price of 68 €. tCO_2e^{-1} .

The first contribution of this thesis contribute to this literature. It compares the effectiveness and marginal mitigation costs of policies associated with instruments targeting GHG emissions and carbon sequestration.

Others GHG mitigation initiatives are being developed as part of the European Green Deal, which aims to promote climate neutrality by 2050 across all sectors and, more generally, to strengthen environmental protection. The Farm to Fork strategy is part of the European Green Deal, and aims to make EU food production and consumption more sustainable, by focusing on "environmentally-friendly" farming practices, reducing the use of pesticides, promoting healthy and sustainable food, and reducing food waste. Carbon farming is being developed as part of this strategy, and this ongoing development is the context for the other two contributions of this thesis. Although carbon farming encourages farmers to adopt carbon sequestration practices, its objective is not specifically to mitigate emissions from the agricultural sector, but to offset emissions from all sectors of the economy.

1.3 Carbon farming schemes

The European Commission's ³ definition of carbon farming is as follows : "[It is a] green business model that rewards land managers for taking up practices leading to carbon

³European Commission : [carbon farming website page](#)

sequestration, combined with strong benefits on biodiversity." The estimated carbon sequestration is then sold as a credit to a company or institution to voluntarily offset its emissions.

Today, this system already exists. In fact, there are many voluntary carbon certification frameworks around the world that certify projects in the agricultural sector. For example, there are international standards such as the Verified Carbon Standard (Verra), Gold Standard and Plan Vivo as well as the American Carbon Registry in the USA and the Label Bas Carbone in France. These certification standards aim to ensure transparency in the carbon credit generation process and to protect stakeholders by ensuring that the project adheres to social and ethical standards. They also seek to guarantee the reliability, credibility, and environmental integrity of carbon credits, ensuring that the carbon sequestration generating the credits is real and verifiable. However, these certification standards are not currently regulated, and their role in terms of reliability, credibility, and environmental integrity is questioned. Indeed, the multitude of rules in the various frameworks creates confusion and a risk of green washing.

This is why the European Commission wants to develop a standardised regulatory framework through the implementation of a European certification framework. The European Commission presented a proposal in March 2023, in which it set a carbon sequestration target of $310MtCO_2e$ by 2030 via carbon farming. Public policies relating to the introduction of a European certification framework are still being developed, and their design faces a number of challenges, particularly in relation to the quantification of carbon sequestration due to the difficulties associated with monitoring carbon stocks and assessing the additionality of sequestration. Feedback from past mechanisms such as the Clean Development Mechanism (CDM) and the REDD+ (Reducing Emissions from Deforestation and Forest Degradation) program should be used to implement carbon

farming effectively.

Carbon farming, as defined by the European Commission, shares certain similarities with the CDM established under the Kyoto Protocol. This mechanism allowed companies whose countries had committed to mitigating their emissions to purchase carbon credits from offset projects in countries without a quantified commitment to mitigation. Suspended since the Paris Agreement in 2016, its effectiveness has been the subject of debate. Indeed, according to some authors, this mechanism lacks credibility due to an overestimation of its effects in terms of GHG mitigation [Repetto, 2001, Alexeew et al., 2010]. However, overestimating carbon sequestration represents a major danger, as it could lead society to unknowingly miss its climate targets. This potential overestimation is partly explained by the problem of assessing additionality, i.e. the effectiveness of the project in terms of mitigation compared with what would have been the case if it had not been implemented. [Cames et al., 2016] show that 85% of the projects covered by the CDM, and 73% of the certified emissions reductions had a low probability of being additional and not over-estimated, whereas only 2% of the projects covered by the CDM and 7% of the certified emissions reductions had a high probability of being additional and not over-estimated. By studying 1350 wind farms set up in India under the CDM, [Calel et al., 2021] estimate that 52% of the credits generated come from projects that would have gone ahead anyway without the CDM. In the context of REDD+ program, a recent paper [West et al., 2023] shows that for 18 projects in several different countries, the credits sold on the voluntary carbon market were used to offset around three times more GHG emissions than they actually avoided. This is explained by an overestimation of deforestation if the projects had not been implement. The difficulty of measuring additionality also arises in the design of carbon farming. It is not efficient to pay for actions that would have taken place in all cases, and windfall effects must be avoided

as far as possible to have an efficient instrument. As shown by [Pautsch et al., 2001], an instrument rewarding only new adopters is more cost-effective than an instrument rewarding all voluntary farms. However, this article does not take into account the moral hazard of farmers, who could temporarily stop practices that have already been implemented only to adopt them again later, being considered as new adopters, which would have a negative effect on net GHG emissions. The difficulty in measuring the additionality of carbon sequestration also stems from the fact that it is difficult to estimate a counterfactual trajectory for soil carbon stock without policy. Even if this is possible thanks to process modelling, by making assumptions, it is necessarily associated with uncertainty.

The potential overestimation of mitigation associated with credits sold may also be due to an overestimation of mitigation due to insufficiently accurate monitoring. This uncertainty about the monitoring method has not been taken into account consistently in the CDM, where nothing about it is clearly addressed in the general regulatory framework [Shishlov and Bellassen, 2016]. The monitoring method has an impact on the form of payment received by the farmer, which can be either for specific measures or for results. In the first scenario, the economic agent receives a payment for implementing specific practices, which isn't precise, but isn't expensive either, while in the second, the payment depends on the results measured in terms of carbon sequestration, which is more precise but also more expensive. The literature has compared the cost-effectiveness of these two forms of payment. [Parks and Hardie, 1995] study the introduction of subsidies for converting marginal agricultural land into forest and show that a subsidy per ton of carbon sequestered is more cost-effective than a subsidy per hectare. In fact, a subsidy per hectare leads to more hectares enrolled for less carbon sequestered. [Pautsch et al., 2001] compare the cost-effectiveness of several systems for subsidising carbon sequestration

resulting from the introduction of a conservation tillage practice in the state of Iowa. They show that for the same results in terms of carbon sequestered, a subsidy per ton is up to 4 times less expensive than a subsidy per hectare. [Bamière et al., 2021] show in the French context and for 3 carbon sequestration practices (extension of temporary grassland, planting of hedges at the edge of fields and no-till), that subsidising carbon sequestration results is more cost-effective than a command and control policy. These studies therefore show, by considering different practices in different geographical locations, that result-based payments are more cost-effective than practice-based payments because of the heterogeneity among farmers in terms of mitigation potential and costs of implementing practices. But they do not consider the monitoring costs that can increase the implementation cost of a results-based policy. In the context of Montana, [Antle et al., 2003] take soil monitoring costs into account and show that despite the weight of these costs, practice-based contracts can be up to 5 times more expensive than results-based contracts for the same level of carbon sequestration. Other methods, such as model-based monitoring, make it possible to save on monitoring costs while paying the farmer according to the predictions of carbon sequestration, [Sidemo-Holm et al., 2018] demonstrate that this method is more cost-effective than action-based. Logically, this would seem to be a good compromise between cost and uncertainty, but in reality, there can be considerable uncertainty over the model's prediction, which means that the payment received may not correspond at all to what is actually sequestered. In this case, it would not necessarily be more cost-effective than result-based.

However, this existing literature does not consider certain aspects which could call into question the consensus that results-based contracts are more cost-effective than practices-based contracts, such as moral hazard, information asymmetry (concerning who holds the best information about the practices to be implemented) and the risk of par-

ticipation in the contract for farmers. Indeed, on this last point, although the 2021 manual on the implementation of the European carbon farming by the Commission leans towards result-based contracts, this report nonetheless emphasizes the need to account for the risk to farmers in the design of carbon farming schemes [European Commission, 2021].

Indeed, the choice between result-based payments and practice-based or model-based payments involves a trade-off in terms of risk-sharing. On the one hand, in the case of practice-based and model-based payments, the farmer knows the amount of her/his remuneration, but the regulator lacks precise information about the amount of carbon sequestration resulting from the implementation of the practice. Thus, society faces the risk of overestimating sequestered carbon, leading to inefficiency and an increased likelihood of failing to meet commitments, thereby exposing itself to the damages caused by climate change. On the other hand, with a result-based contract, the farmer bears the risk, as (s)he does not know the amount of the payment they will receive while supporting the costs of implementing the practice, which increases the cost of participating in the contract. The last two contributions of this thesis focus on incorporating the farmer's risk into the design of cost-effective carbon farming schemes.

Finally, the non-permanence of carbon sequestration is also a barrier to the implementation of carbon sequestration regulations. In practice, this means that when the maximum carbon stock is reached, no further carbon is sequestered. However, in order to preserve what has been sequestered, the farmer must continue to bear the costs associated with implementing the practice. While the maximum stock is generally considered to be reached around 20-25 years after the practice is first implemented [West et al., 2004], maintaining the carbon stock for around 100 years would be necessary for the sequestered carbon to be considered as permanent mitigation of GHG emissions. This would involve excessive maintenance costs for the farmer. This is why some studies propose to take

into account the non-permanence of the carbon sequestration by discounting the payment according to the maintenance duration [Gulati and Vercammen, 2005, Kim et al., 2008]. Nevertheless, according to [Rosenberg et al., 2000] and other authors, carbon sequestration should be seen as a temporary solution, awaiting the development of less costly mitigation technologies, for example. With this in mind, [Lal, 2008] suggests that the carbon sequestration option should be used as a "bridge to the future".

* * *

In summary, the literature began by assessing practices aimed at reducing GHG emissions in the agricultural sector. The potential and costs of implementing these practices vary from farm to farm. In addition, carbon sequestration is often estimated using models, whose assumptions influence the results. The literature has then examined the economic potential of agricultural GHG mitigation through various modelling methods, assumptions, and contexts. GHG regulation policies in the agricultural sector face challenges, including the costs of monitoring GHG emissions and carbon sequestration. Moreover, since mitigation in agriculture can be less costly than in other sectors, it is seen as an attractive option for offsetting emissions in other sectors. With this in mind, the EU wants to create a regulatory framework for carbon farming. The design of this framework, including the form of contracts, presents a number of challenges. Although the literature largely addresses the question of the cost-effectiveness of carbon sequestration incentives depending on the form of the contracts, it neglects certain behavioural obstacles, such as the risk perceived by farmers who enter into such contracts. Furthermore, the non-permanence of carbon sequestration is also a major challenge in the implementation of carbon farming.

1.4 Contributions

I address questions related to the cost-effective implementation of net GHG mitigation practices in the agricultural sector. More specifically, this thesis focuses on the trade-off between monitoring costs and the effectiveness of greenhouse gas mitigation and carbon sequestration policies in the agricultural sector.

Indeed, ideally, if all the information were available, at no cost, we could perfectly consider the heterogeneity among agents in terms of costs of implementing mitigation practices (taking into account all technical costs as well as other costs such as risk, etc.) and mitigation potential, to design an instrument that provides an incentive to mitigate net GHG emissions. In addition, a farmer's mitigation results would be perfectly known and also without cost. Thus, the cost-effective allocation of mitigation efforts would be enabled by implementing an instrument that covers all sources and sinks of emissions in the agricultural sector, and which is results-based.

However, in reality, perfect information is not available, implementing a policy requires a trade-off between the costs of monitoring greenhouse gas emissions and carbon sequestration, and the effectiveness in terms of net GHG mitigation. The costs of monitoring depend in particular on the coverage of the policy and the precision of the estimation of GHG mitigation. On the one hand, reduced coverage of sources and sinks allows for cost savings in monitoring, but it decreases the effectiveness in terms of net GHG mitigation. On the other hand, more precise monitoring reduces the risk of undue payments and overestimation of GHG mitigation but can increase the cost of participating in the program by making payments uncertain for farmers. This has implications for the implementation of mitigation practices and, consequently, net GHG mitigation effectiveness. Thus, saving on monitoring costs is likely to impact the

effectiveness of the policy. This thesis aims to explore the trade-off between saving monitoring costs and the associated variation in GHG mitigation policy effectiveness in the agricultural sector through three contributions.

In Chapter 2, entitled "Towards carbon neutrality: Cost-effectiveness of targeting GHG emissions vs. carbon sequestration in agricultural sector", I study the effectiveness, in terms of net GHG emissions mitigation and marginal mitigation cost, of several instruments partially covering emission sources and sinks, compared with one instrument covering all sources and sinks. Specifically, due to possible interactions between sources and sinks, I investigate whether an instrument encouraging only carbon sequestration or only GHG reduction can have a positive effect on net GHG emissions mitigation. In addition, I examine which of these two types of partial coverage policy is the most cost-effective.

To address this, I developed a static inter-temporal model on the supply side, considering both crop and livestock activities, GHG emissions, and carbon sequestration. I apply this model at the scale of the Grand Est region, with 40 farm groups. I consider several mitigation practices and taking into account the heterogeneity in terms of cost and mitigation potential among each farm group whenever is possible, and at least among each sub-region of the Grand Est (Champagne, Lorraine, Alsace). In this first contribution, I compare the cost-effectiveness of alternative policies to the first best, assuming that monitoring costs are lower for these alternatives, but I do not explicitly take these costs into account.

In the other two chapters, I explicitly take monitoring costs into account. In addition, I compare the cost-effectiveness of results-based, model-based, and practice-based incentives. I consider the risk associated with payment uncertainty in the case of results-based contracts. This uncertainty impacts the costs of farmers' participation in these

types of contracts, which in turn affects the GHG mitigation effectiveness of results-based contracts.

Indeed, Chapter 3, entitled "Adoption of carbon farming schemes: Risk matters", we study a voluntary carbon farming scheme, and compare the cost of participation for the farmer between a results-based scheme and a model-based scheme. In the case of the results-based contract, the farmer bears the risk of not receiving sufficient payment to offset the costs of implementing the carbon sequestration practice. In the case of model-based scheme, a discount on the level of payment is applied according to the level of uncertainty of the carbon sequestration, but the payment at the end of the contract is known with certainty to the farmer. Our aim is to explore analytically and find theoretical conditions for farmer decisions regarding contract participation and monitoring methods, using a stylised plot-level approach. This also allows us to highlight the trade-offs between the regulator and the farmer in terms of risk sharing and to discuss the consequences of the choice of monitoring. We apply our stylised approach. We use data from an experimental site in Ultuna (Sweden), and from a multi-model (AMG, RothC, ICBM, Daycent) ensemble of soil organic carbon stock. We make various assumptions about the risk aversion parameter, the level of payment per tonne of CO_2e and the time horizon of the scheme. We use a CARA-type (Constant Absolute Risk Aversion) utility function to take into account the farmer's risk aversion. The simplicity of its form makes it easier to explore the trade-offs between the different monitoring methods. We assume that the uncertainty in carbon sequestration is estimated as the standard deviation of the predictions of each of the models in the ensemble. This second chapter is co-written with Stéphane De Cara, Laure Bamière, and agronomists and soil scientists Elisa Bruni, Bertrand Guenet and Claire Chenu.

Considering a single farmer allows us to discuss the trade-offs involved the cost of

participating in a contract according to the monitoring method, but this does not allow us to conclude on which is the most cost-effective scheme, since we do not take into account the heterogeneity in terms of costs and mitigation potential of practices and risk aversion among farmers. In the Chapter 4, entitled "Cost-effectiveness of carbon farming schemes taking farmers' risk aversion into account", I take these heterogeneities into account. The aim is to compare the cost-effectiveness of practice-based, model-based, and results-based contracts for different time horizons, and different levels of carbon sequestration variability among farms. To do this, I propose a generalization of the analytical framework of Chapter 3 to several farms. But, I take into account the farmers' risk aversion using a CRRA (Constant Risk Relative Aversion) function. Although less practical to manipulate in a stylised framework than a CARA function, there are more data available from the experimental literature estimating farmers' risk aversion coefficients based on a CRRA function, which makes it easier and more robust to make assumptions about the data for the empirical application. I then apply this stylised framework to the static inter-temporal model developed in the Chapter 2, at the scale of the Grand Est region. This also allows to consider the opportunity costs of entering into the contract, linked to other farm activities. In addition, I estimate the standard deviation of the carbon sequestration at the scale of each of the 3 sub-regions of the Grand Est region, using crop yield as a proxy. In Chapter 3, I use a set of 4 models to estimate the mean and standard deviation of in-situ carbon sequestration. Whereas in this last contribution, I use the predictions of a single model (STICS) for the mean of sequestered carbon, and therefore crop yields as a proxy for the standard deviation for each sub-region.

Chapter 2

Towards carbon neutrality:

**Cost-effectiveness of targeting GHG
emissions vs. carbon sequestration in
agricultural sector**

Abstract

France has set a target of achieving carbon neutrality by 2050, as part of its commitment to the Paris Agreement. The agricultural sector stands as the second-largest emitter, accounting for approximately 16 % of France's greenhouse gas emissions. It is crucial to determine effective mitigation strategies to ensure the realization of these goals. I seek to estimate, at the same level of regulatory instrument, the loss of efficiency in terms of net GHG mitigation between a policy covering all sources and sinks of GHG emissions, and partial coverage policies, i.e. a GHG tax, a subsidy for carbon sequestration in soil and biomass and a subsidy for carbon sequestration in biomass. The aim is also to see whether any of the partial coverage policies would achieve a higher net GHG emissions mitigation result than the others at a lower marginal mitigation cost. To address this, I develop an inter-temporal static model of agricultural supply that maximizes farmers' income under each of these mitigation policies targeting different emission sources and/or sinks. An originality of this approach lies in the simultaneous consideration of greenhouse gas emissions, and carbon sequestration, and of crop and livestock activities. Another aspect is the dynamic of soil carbon sequestration. I apply this model to 40 farm groups in the Grand Est region (France). The results indicate that given my hypotheses, my results suggest that, all of the partial coverage policy scenarios show net GHG mitigation. But the difference in terms of net GHG mitigation is significant between policies covering all sources and sinks and partial coverage policies: depending on the carbon price trajectory, compared with a tax on net GHG emissions, a tax on GHG emissions leads to net GHG emissions that are 185 to 340% higher, and subsidies for total carbon and biomass lead to net GHG emissions that are 80 to 95% higher. Moreover, a subsidies for total and only biomass carbon sequestration are more cost-effective than a tax on

GHG emissions. Indeed, according to the carbon price trajectory scenarios, compared with the carbon sequestration subsidy scenarios (total and biomass), the marginal costs of the GHG tax scenario are higher by a factor of between 2 and 5, and mitigation is lower by a factor of between 2 and 3.

Keywords: *GHG mitigation, climate policy, carbon sequestration, monitoring, agricultural supply model*

2.1 Introduction

Under the Paris Agreement, 195 countries around the world have committed to limiting global temperature rise to 2 ° C. In France, this goal is reflected in the National Low-Carbon Strategy (SNBC), which aims to achieve carbon neutrality at the national level by 2050. Carbon neutrality is defined as the balance between anthropogenic greenhouse gas (GHG) emissions and carbon sequestration in soils and biomass.

Agriculture is one of the few sectors having the capacity to store carbon, operating as a carbon sink. Despite this potential, it is the second highest emitting sector in France with circa 16% of the national GHG emissions [Citepa, 2023]. In this sector, the reduction of GHG emissions (mainly N_2O and CH_4) and the increase in carbon sequestration can result from changes in management practices, such as changes in inputs use (e.g. less mineral fertilizers, more grass fodder in feed rations) and adoption of practices such as intra-plot agroforestry and the use of nitrification inhibitors, or from changes in activities, e.g. reduction of herd size, change in land allocation, etc.

The National Low-Carbon Strategy sets mitigation targets for agricultural GHG emissions of 18% by 2030 and 46% by 2050, as compared to 2015 emissions [SNBC, 2020]. However, most mitigation levers are costly for farmers to implement [Doreau et al., 2014, Pellerin et al., 2017, Bamière et al., 2023]. Although there are already incentives to implement some mitigation levers through the introduction of AECM (agri-environmental and climate measures) in the 2015-2022 Common Agricultural Policy (CAP), these are deemed insufficient by the SNBC to achieve the targets set. ¹. It is then necessary to develop cost-effective mitigation strategies.

¹The "Low Carbon Label" also contributes to mitigating net greenhouse gas emissions in the agricultural sector. However, this carbon certification scheme is not solely aimed at offsetting emissions within agriculture, but rather offsetting emissions in the sector of the firm which buys the carbon credits.

Even though the most effective mitigation policy to reduce greenhouse gas (GHG) emissions in this sector involves regulating all sources and sinks of emissions, it presents significant economic challenges. This is primarily due to the diversity of sources and sinks and the spatial heterogeneity of emission factors and carbon sequestration potential. The costs associated with monitoring and measuring such a policy would be excessively high, making its implementation difficult. Due to these difficulties, some studies have explored a regulation that partially covers emissions sources and sinks: for example, [Mérel et al., 2014] on N_2O emissions, [De Cara et al., 2005] on N_2O and CH_4 , [De Cara and Jayet, 2000, Schneider et al., 2007] on the three GHGs, and [Pautsch et al., 2001, Antle et al., 2003, Kurkalova et al., 2004, Feng et al., 2006, Antle et al., 2007, Ragot and Schubert, 2008, Haim et al., 2014] focused on soil carbon sequestration.

However, regulating only part of the emissions sources and sinks can have both positive or negative impacts on emissions from other sources and sinks. Indeed, an agricultural practice that mitigates one GHG or sequesters carbon may also be emitting (or mitigating) another GHG or dесеquestering (or sequestering) carbon, as illustrated by the example of reduced tillage, which sequesters carbon but emits N_2O due to the non-burial of organic effluents, or the planting of hedges, which sequesters carbon while also reducing N_2O emissions through a reduction of fertilized areas [Pellerin et al., 2017]. Moreover, an instrument regulating GHG emissions can impact carbon sequestration, for example by increasing the opportunity cost of maintaining grasslands following a reduction in the bovine herd, which is a significant CH_4 emitter [Alemu et al., 2017, Liang et al., 2020].

Therefore, partial regulation of emission sources and sinks can lead to interactions that result in varying degrees of efficiency loss compared to full coverage regulation and

should be studied to support the cost-effective implementation of mitigation policies in the agricultural sector.

[Garnache et al., 2017] compare the difference in social cost, for several levels of abatement, between a tax on net greenhouse gas emissions, a tax on N_2O emissions and a subsidy for carbon sequestration, using a crop supply model applied to California. They show that at $20\$.tCO_2e^{-1}$, regulating N_2O and CH_4 emissions respectively results in a 16% and 21% loss of abatement compared with the first best solution (regulation of all greenhouse gas sources and sinks). However, the sequestration options considered in this study are limited and consist only of different soil tillage intensities. Additionally, their model is static and does not account for the non-linear dynamics of carbon sequestration over time. Finally, their model does not include livestock, and therefore does not account for the opportunity costs associated with the potential adjustments between the crop and livestock activities, which limits its ability to capture the interactions between GHG emissions and carbon sequestration.

Our objective here is to evaluate, for the same level of cost, the effectiveness in terms of net GHG mitigation of several partial coverage policies, namely a tax on GHG emissions, a subsidy for total carbon sequestration (biomass + soil) and a subsidy for sequestration from biomass, compared with a policy covering all sources and sinks. I take into account trade-offs between livestock and crop production as well as a variety of soils and biomass carbon sequestration practices. This would make it possible to determine whether it is relevant to introduce regulations covering part of emission sources and sinks, and which one between GHG emissions and total and only biomass carbon sequestration, in order to save on transaction and monitoring, reporting, and verification costs without losing too much efficiency.

In order to address these questions, I develop an agricultural supply model. I consider

this approach to be the most appropriate for modeling the adoption of mitigation practices in agriculture, since it captures the individual behavior of farmers, takes into account the opportunity costs of adopting different practices, without requiring as many data and hypotheses as a partial equilibrium model. The model I develop is an inter-temporal static model consisting of 31 periods. I consider both crop and livestock activities. An original aspect of this work is the consideration of the dynamics of the soil carbon sequestration process, its non-permanence, and its finite nature for a given land management practice. I also account for carbon sequestration in biomass, that I model as a constant increase. The mitigation practices I incorporate are part of those identified for the French agricultural sector by [Pellerin et al., 2017, Bamière et al., 2023] : development of intra-plot agroforestry, planting of hedgerows along field edges, introduction of temporary grasslands into fodder maize monocropping, introduction of legumes into crop rotations, use of nitrification inhibitors, and addition of nitrate to ruminant feed rations. The model maximizes the sum of profits for all farmers over the specified time horizon. I consider various carbon price trajectories and several mitigation policy scenarios, namely a tax on net GHG emissions, a tax on GHG emissions, a subsidy for carbon sequestration in soil and biomass, and a subsidy for carbon sequestration in biomass. I explore the latter policy scenario because practices such as the development of agroforestry and hedgerows demonstrate a significant potential for carbon sequestration in biomass and may require less costly monitoring methods than soil carbon sequestration (e.g. remote sensing). Therefore, in each period and in response to the mitigation regulation, each farmer decides on land use allocation, herd size, feed rations, and whether or not to implement various mitigation practices. I assume that the regulator has perfect knowledge of the emissions and carbon sequestration production process and I disregard transaction costs and administration costs. I focus specifically on the opportunity costs borne directly by

farmers in terms of reduced agricultural profits and on the results of the different policies in terms of their effectiveness on net GHG mitigation. I apply this model to the French region of Grand-Est, since it has a diversity of agricultural activities, including cattle, dairy and crop farming. I consider 40 farm groups divided between the 3 sub-regions that make up the Grand-Est region : Champagne-Ardenne, Lorraine, and Alsace. I take into account the economic heterogeneity between farm groups and the spatial heterogeneity arising from soil and climate conditions at the farm-group and sub-region level.

This paper contributes to the literature on cost-effective implementation of GHG mitigation policies in the agricultural sector [Schneider et al., 2007, Van Doorslaer et al., 2015, De Cara et al., 2005, Lengers and Britz, 2012, Ramilan et al., 2011, Durandeu et al., 2010], and specifically on the effectiveness of second-best policies [Garnache et al., 2017, De Cara et al., 2018]. More generally, this work contributes to the literature on monitoring in the case of non-point source pollution regulation [Xepapadeas, 2011].

In section 2.2, I present the model. Section 2.3 is devoted to the description of the selected farm groups and mitigation practices, as well as data and model validation. Section 2.4 presents the results. Section 2.5 provides elements for discussion and conclusion.

2.2 Model :

The model is a supply-side mathematical programming model. Its objective is to maximize farms' gross margins while considering multiple constraints. I adopt an intertemporal static approach in which each farmer makes all strategic decisions at the initial time for the entire time horizon, having access to information covering the entire period. Each farmer is assumed to behave rationally, seeking to maximize her/his profit. There

are four main types of constraints: (i) the objective function, (ii) technical constraints related to agricultural production (e.g. satisfying feed requirements of livestock and crop needs in terms of nitrogen fertilisation), (iii) policy constraints related to existing regulations, such as the Nitrates Directive and the maintenance of permanent grassland, and (iv) equations for accounting for GHG emissions, carbon sequestration.

The objective function consists in maximizing the discounted sum of profits of all individual farms over a specified time horizon :

$$\max_{X_{l,p,t}, A_{l,i,h,t}, QB_{l,d,t}} \sum_t \sum_l \Pi_{l,t} \cdot \rho^t \quad (2.1)$$

where $\Pi_{l,t}$ is the profit of farm l at time t and ρ is the discount factor. As the farms are independent and do not buy or sell products from or to each other, it is the same as maximizing the total profit of each farm. The objective depends on decisions of each farmer at each period t on :

- The area allocated to each crop, $X_{l,p,t}$, where p represents the combination of a cropping activity with one or more GHG mitigation or carbon sequestration practice.
- The number of "composite" animals of type i to which practice h is applied, $A_{l,i,h,t}$. A "composite" animal refers here to the combination of the main animal (e.g. a dairy or a suckler cow) and its associated herd (e.g. calves, heifers, etc.). The composition of the associated herd is assumed to be fixed per dairy or suckler cow in each farm and only the number of composite animals can vary. This allows to avoid complex herd demographic constraints in the model. The h represents the combination of a feed ration (with varying amounts of fodder maize and grass

forage) and possibly a feed-related GHG mitigation practice (e.g., adding nitrates to the animal's ration).

- $QB_{l,d,t}$ is the quantity of fertilizer or feed purchased (d is the index that encompasses crop outputs, feed inputs and the mineral fertilizer input. $g \in d$ specifically denotes the subset of crop outputs within this index).

$$\Pi_{l,t} = \sum_g (QS_{l,g,t} \cdot ps_g) - \sum_p (X_{l,p,t} \cdot c_{l,p}) + \sum_{i,h} (A_{l,i,h,t} \cdot gma_{l,i,h}) - \sum_d (QB_{l,d,t} \cdot pb_d) - FC_{l,t} - \tau_t \cdot MP_{l,t} \quad (2.2)$$

where $QS_{l,g,t}$ is the quantity of crop product g sold, knowing that part of what is produced may be consumed on the farm to meet the animals' feed requirements. ps_d and pb_d are respectively the selling and buying price of d , where $g \in d$. Prices are considered exogenous because farmers are assumed to be price takers, in line with a supply-side model. $c_{l,p}$ is the production cost of the crop-practice combination p , including seeds, pesticides, and fuel costs. $gma_{l,i,h}$ is the gross margin per animal i under practice h . $FC_{l,t}$ is the sum of fixed costs linked to the implementation of some practices (most practices have only annual costs, but some, such as agroforestry and hedges, also have high installation costs). $FC_{l,t}$ is calculated by multiplying the installation cost parameter per ha by the difference between the areas of p between this period and the previous one. Depending on the mitigation policy scenario, $MP_{l,t}$ represents net greenhouse gas emissions, non-net greenhouse gas emissions, or carbon sequestration. The parameter τ is the carbon price, which can act either as a subsidy on carbon sequestration or a tax on emissions, depending on the policy scenario.

If the mitigation policy concerns non-net GHG emissions :

$$MP_{l,t} = E_{l,t}$$

where

$$E_{l,t} = \sum_p (X_{l,p,t} \cdot ef_p) + \sum_{i,h} (A_{l,i,h,t} \cdot efa_{i,h}) + QM_{l,t} \cdot efm + \sum_d (QB_{l,d,t} \cdot efd) \quad (2.3)$$

where ef_p is the emission factor of p (in $tCO_2e.ha^{-1}$) and includes CO_2 emissions from fuel combustion as well as upstream emissions linked to fuel use and manufacture of machinery. $efa_{i,h}$ (in $tCO_2e.head^{-1}$) includes CH_4 and N_2O emissions associated to enteric fermentation and effluent management (in the barn or when grazing, during storage). $QM_{l,t}$ is the quantity of manure spread on farm and efm includes N_2O emissions related organic nitrogen fertilization. Finally, efd in tCO_2e per ton of input d , includes upstream emissions linked to the production of purchased inputs (animal feed and mineral fertilizer), as well as N_2O emissions linked to the use of mineral nitrogen fertilizer.

If the mitigation policy concerns only carbon sequestration in soils and above-ground biomass :

$$MP_{l,t} = \Delta C_{l,t}$$

$$\Delta C_{l,t} = \sum_p X_{l,p,t} \cdot \left[(C_{l,t}^m - C_{l,t-1}^m) - (C_{l,t}^b - C_{l,t-1}^b) + biom_{l,p} \right] \quad (2.4)$$

where $\Delta C_{l,t}$ is the carbon sequestration in soil and above-ground biomass between t-1 and t, $C_{l,t}^s$ is the soil carbon stock per hectare at t, for the scenario s , where $s \in \{m, b\}$, with m for mitigation policy scenario and b for baseline scenario. Soil carbon

sequestration is calculated as the difference between the average carbon storage per hectare resulting from production choices in the fixed mitigation policy scenario and the average carbon storage per hectare resulting from production choices when there is no mitigation policy. $biom_{l,p}$ is the annual carbon sequestration coefficient in above-ground biomass due to the implementation of p .

While, for a matter of simplicity and lack of data, the sequestration from above-ground biomass is modeled as a constant annual increase, the soil carbon storage is modeled as follows :

$$C_{l,t}^s = \frac{\sum_p X_{l,p,t} \cdot (\overline{C}_{l,p} + (C_{l,t-1}^s - \overline{C}_{l,p}) \cdot e^{-v_p})}{uaa_l} \quad (2.5)$$

where uaa_l is the farmer's UAA. $C_{l,t}^s$ is estimated by averaging carbon accumulation per ha over the entire farm area, through a weighting based on the area of soil carbon stock per hectare of each p relative to the previous stock. This soil carbon stock changes is derived from the Misterlich Baule function, as in the one-carbon pool model of Hénin Dupuis [Hénin and Dupuis, 1945], considering key parameters $\overline{C}_{l,p}$ ², and v_p ³ are the parameters of the "maximum" stock achievable by practice p in l and the carbon sequestration "rate" of practice p in farm l respectively.

²The "maximum" stock achievable by a practice is defined according to the IPCC as the carbon stock reached after 20 years when the practice and the pedoclimatic conditions are unchanged. By agronomists, it is defined as the product of the quantity of carbon brought to the soil in $tC.ha^{-1}$, noted m , by the isohumic coefficient $K1$ (it represents the proportion of m "entering" the soil, the rest being supposed to be immediately mineralized.), on $K2$ which is the coefficient of destruction (or mineralization) of the organic matter , per year: $\overline{C}_p = \frac{m \cdot K1}{K2}$.

³ v_p represents the $K2$ coefficient. It depends on the climatic conditions, the type of soil (clay content, limestone, pH of the water) and on soil tillage intensity (ploughing accelerates mineralization).

If the mitigation policy concerns net GHG emissions, $MP_{l,t}$ is defined as :

$$MP_{l,t} = E_{l,t} - \Delta C_{l,t} \quad (2.6)$$

Net GHG emissions are the difference between GHG emissions and carbon sequestration in soil and above-ground biomass between t-1 and t.

Our model is subject to the following constraints:

$$\sum_p X_{l,p,t} \leq uaa_l \quad \forall l, t \quad (2.7)$$

$$\sum_h A_{l,i,h,t} \leq \sum_h A_{l,i,h,t-1} \quad \forall t, l, i \quad (2.8)$$

$$\sum_h A_{l,i,h,t} \geq (1 - ev) \cdot \sum_h A_{l,i,h,t-1} \quad \forall t, l, i \quad (2.9)$$

$$\sum_{i,h} A_{l,i,h,t} \leq \sum_p X_{l,p,t} \cdot srl_p \quad \forall l, t \quad (2.10)$$

$$QB_{l,g,t} + QI_{l,g,t} \geq \sum_{i,h} A_{l,i,h,t} \cdot fr_{i,h,g} \quad \forall l, g, t \quad (2.11)$$

$$QB_{l,fo,t} = 0 \quad (2.12)$$

$$QI_{l,g,t} + QS_{l,g,t} \leq \sum_p X_{l,p,t} \cdot yl_{g,p} \quad \forall l, g, t \quad (2.13)$$

$$QB_{l,min,t} + QM_{l,t} \geq NR_{l,t}(A_{l,i,h,t}, X_{l,p,t}) \forall l, t \quad (2.14)$$

$$QBMG_{l,t} + \sum_{i,h} A_{l,i,h,t} \cdot nxgl_{i,h} \geq \sum_p X_{l,p,t} \cdot cg_p \cdot nrgl_{l,t} \forall l, t \quad (2.15)$$

$$QBMCl_{l,t} + QM_{l,t} \geq \sum_p X_{l,p,t} \cdot (1 - cg_p) \cdot nrc_{l,p,t} \forall l, t \quad (2.16)$$

$$QBMG_{l,t} + QBMCl_{l,t} = QB_{l,min,t} \forall l, t \quad (2.17)$$

$$QM_{l,t} \leq \sum_{i,h} A_{l,i,h,t} \cdot nxm_{l,i,h} \forall l, t \quad (2.18)$$

$$\sum_{af} X_{l,af,t} \geq \sum_{af} X_{l,af,t-1} \forall l, t \quad (2.19)$$

$$\sum_{he} X_{l,he,t} \geq \sum_{he} X_{l,he,t-1} \forall l, t \quad (2.20)$$

$$QM_{l,t} \leq lnd \cdot \sum_p X_{l,p,t} \cdot (1 - cg_p) \quad (2.21)$$

$$X_{l,pg,t} \geq sgl \forall l, t \quad (2.22)$$

Equation 2.7, means that the sum of the areas used for p activities cannot exceed the UAA (Utilised Agricultural Area) allocated to each farm.

The purpose of constraints 2.9 and 2.8 is to limit the variation of the herd. I assume

that the herd of each species i cannot increase (equation 2.9), and cannot decrease by more than ev percent (here 15%) per year compared to the previous year (equation 2.8), regardless of practice h . This constraint prevents a sudden collapse or increase in the number of animals, which is incompatible with the needed for capital investment in the case of herd growth, and with the adjustment of consumption in the event of a herd decrease.

Constraint 2.10 states that the number of composite animals must not exceed a certain stocking rate per hectare of main fodder area, where $sr_{l,p}$ is the stocking rate.

Constraint 2.11 states that the animals' needs must be met either through intra-consumption $QI_{l,g,t}$ or through external purchases $QB_{l,g,t}$. I add constraint 2.12, where $f \in g$, represents grass and maize fodder products. This constraint indicates that it is not possible to buy grass and fodder maize in any form (silage, hay or pasture) from outside the farm to meet the needs of the animals, thus farms must be self-sufficient in grass and maize fodder.

Constraint 2.13 implies that the crop products sold and intra-consumed must not exceed what is produced.

Equation 2.15 states that the nitrogen requirements of grasslands (permanent and temporary) can be covered by the purchase of mineral nitrogen $QBMG_{l,t}$ and/or the nitrogen released by animals during grazing, where cg_p is the proportion of grassland in the activity p , $nrg_{l,t}$ the parameter of nitrogen requirement of grassland per hectare, and $nng_{l,i,h}$ is the parameter of nitrogen produced by animals during grazing. Equation 2.16 states that the nitrogen requirements of other crops must be fulfilled through manure $QM_{l,t}$ and/or the purchase of mineral nitrogen $QBMC_{l,t}$. $nrc_{l,p,t}$ is the parameter of nitrogen requirement per ha of crops other than grassland in the activity p .

Equation 2.17 is a budget equation for the amount of mineral nitrogen to be purchased,

where the total purchase of mineral nitrogen denoted $QB_{l,min,t}$ is the sum of the quantity of mineral nitrogen to be spread on grassland $QBMG_{l,t}$ and on other crops $QBMC_{l,t}$. Equation 2.18 restricts the amount of manure applied to crops to that produced by the animals, where $nxm_{l,i,h}$ is the parameter of nitrogen produced in the form of manure.

Constraints 2.19 and 2.20 state that the areas dedicated to practices in which agroforestry and hedges are implemented, respectively, can only increase or remain constant from one year to another, where af and he refer to subsets of practice combinations p .

Constraint 2.21 corresponds to the Nitrate Directive and limits the amount of manure spread on crops at 170 kgN.ha^{-1} (denoted lnd).

Equation 2.22 constrains the maintenance of already existing permanent grasslands, where sg_l is the initial area of permanent grassland on farm l at t_0 and pg standing for the subset of permanent grassland activities (where $pg \in p$). Indeed, ploughing grasslands has consequences for carbon stocks, biodiversity, and nitrate leaching. That is why we've decided to account for this EU CAP regulation and also to investigate what mitigation strategies can be efficient and compatible with this constraint.

We initialize crop activity areas for which the fixed installation costs are positive. This precaution is taken to prevent the model from assuming their establishment prior to the optimization period, thereby avoiding paying for their installation costs.

2.3 Data and model validation

We propose an application of the model to the Grand-Est region since it is a diversified region in terms of agricultural activities with both livestock and crop farms. This region is divided into 3 sub-regions : Champagne-Ardennes, Lorraine, and Alsace. Regarding data on mitigation practices, I have two detailed sources of information for

France: [Pellerin et al., 2017, Bamière et al., 2023]. These sources use respectively the year 2010 and the average of the years 2009-2013 as a reference period. Therefore, for a matter of consistency, I use the average of the years 2009-2013 as reference period to parameterize the model, whenever possible. Using average values over a five-year period also allows to smooth inter-annual variations (in yields, prices, production costs, etc.).

The model's input data and the details of their construction are available in the supplementary materials document of this Chapter, available at this address <https://eng-psae.versailles-grignon.hub.inrae.fr/personalpages/photinodellis>.

2.3.1 Farm groups

To construct farm groups, I aggregate individual data from the French Farm Accountancy Data Network (FADN). First, I group farms by sub-region, type of farming excluding wine and fruit production (Cereals-Oilseeds-Protein crops, General cropping, Dairy cattle, Rearing and fattening cattle, Mixed livestock, and Mixed crop-livestock), and economic size for each year of the reference period. Next, I eliminate groups that are not present in at least one of the five years and groups that do not meet statistical confidentiality requirements (i.e., less than two surveyed farms). Finally, I calculate the average characteristics (UAA, number of farms represented, yields, herd size, etc...) of each remaining farm groups over the period 2009 to 2013. We obtain 40 farm groups, representing 63% and 82% of the Grand-Est region farms and UAA, respectively (see Table 1 in supplementary materials for details).

Each farm group is considered a modeling unit representing the economic behavior

of all similar farms. To aggregate the results at the scale of the Grand Est region and its sub-regions, the outputs of each group are multiplied by the number of farms they represent.

2.3.2 Crop and livestock activities

We focus on the region's six main cropping activities, i.e. soft wheat, rapeseed, barley, and grain maize — which have the highest area share in the farm groups of the three sub-regions (see table 2.1)— and on fodder maize and permanent grassland —which are essential for feeding the herd. The UAA of the sample farm groups, including all the surfaces of the crop activities considered, represents 76% of the UAA of Champagne, 91% of Lorraine and 78% of Alsace of the FADN database of all Grand Est farms. Soft wheat, barley, rapeseed, and grain maize are combined as a single crop rotation activity. Parameters related to this rotation (such as costs, GHG emissions, and nitrogen needs), are calculated using a surface weighting (see Table 3 in supplementary materials). Furthermore, I assume that the farmers can set aside land and convert it to bare fallow, fallow being associated with no costs and no income. Farmers can thus allocate land between 4 distinct options (before considering mitigation practices): the wheat-barley-rapeseed-grain maize rotation, fodder maize, permanent grassland, and bare fallow.

	Champagne	Lorraine	Alsace
Total number of farms represented	9839.91	7364.70	3870.06
Average UAA of farm groups (ha)	159.86	161.53	70.34
of which :			
Wheat (%)	25.79	18.32	15.59
Rapeseed (%)	13.40	10.93	1.40
Barley (%)	21.05	13.43	1.23
Grain maize (%)	2.07	1.28	47.17
Permanent grassland (%)	15.67	43.16	20.59
Fodder maize (%)	2.20	6.22	3.76
Total (%)	80.18	93.33	89.74

Table 2.1: Share of the selected crop and fodder activities in the average UAA of farms groups ⁴ in each sub-region. Data source : FADN database 2009-2013

Yields of wheat, barley, rapeseed, and grain maize are obtained from the FADN 2009-2013 data by farm group (see Tables 5, 6, 7, 8 in supplementary materials). We use fodder yields by sub-region (permanent grassland, fodder maize, and straw, see tables 9, 10, 11 in supplementary materials), crop production costs per ha by sub-region (including expenses for pesticides, seeds, and fuel⁵, see tables 15, and 16 in supplementary materials), and national crop and straw prices (Table 13 in supplementary materials) estimated by [Bamière et al., 2023]. These estimates are based on FADN 2009-2013 data, with the exception of straw prices. In the model, I assume that farmers are self-sufficient in maize silage and grass fodder. Straw is the only forage they can sell or buy, with two different prices. The selling price of straw comes from data in IDELE's 2015 case study on dairy cattle in the Grand Est region ⁶ and a transportation cost is added to obtain the purchase price⁷.

⁵Although the crop production costs are not heterogeneous across farm groups, for rotation, since the share of each activity differs across farms, the production costs per hectare of rotation also vary.

⁶Idele, Dairy cattle, 2015.

⁷The transportation cost is calculated using a formula from the French Centre National Routier website <https://www.cnr.fr/formule-trinome>. This formula is based on 2022 economic data, assuming a 20 km distance from the farm to the delivery point and 1 hour for the delivery.

Nitrogen requirements per crop and sub-region come from the ADEME "Base Carbone"⁸ based on the 2006 french cultivation practice surveys⁹, which provides data by sub-region and crop (see tables 17 and 18 in supplementary materials).

For livestock, I consider only bovines, which represent around 90% of the animals in LUs in the RICA Grand Est database for the period 2009-2013. Dairy and suckler cows are also the most represented animals in the three sub-regions (see table 2.2). I consider that only farm groups specialized in livestock farming can have a herd, i.e. all farm groups except those specialized in "cereal, oilseeds, and protein crops" and "general cropping". In addition, I assume that "dairy farms" do not have a suckler cow herd and vice versa for farms specialized in "rearing-fattening cattle". There are two types of composite animals: dairy cattle and suckler cattle. The proportion of each category of cattle in the composite animal (calf, heifer, young male, etc.) which forms its herd, is specific to the number of heads of each category in each farm group, and is parameterized based on the RICA 2009-2013 data of the farm groups (see Table 27 in supplementary materials)¹⁰

⁸Base Carbone is a French public database of emissions factors, [Base Carbone V23.0](#). In this database, crop nitrogen requirements are estimated based on "Enquête pratiques culturales, Agreste, 2006.

⁹Enquête pratiques culturales, Agreste, 2006

¹⁰The FADN provide information on the number of animals in the various categories in each farm. For farms specialising in mixed cattle farming and farms specialising in crop and livestock farming, I use an assumption according to which the proportion of animals per dairy cow corresponds to the average observed on farms specialising in dairy production in the same sub-region. The distribution of the other animal categories, apart from dairy cows, is estimated on the basis of the remaining part of the total number of animals in each animal category on each farm.

	Champagne	Lorraine	Alsace
Total number of farms represented	9839.91	7364.70	3870.06
Average number of animals (cattle, sheep, pigs, etc.) in farm groups (LU)	45.64	107.95	31.07
of which :			
Dairy cows (%)	30.31	26.45	42.45
Suckler cows (%)	11.85	16.65	6.15
Other cattle (%)	52.51	56.75	36.30
Total (%)	94.67	99.85	84.90

Table 2.2: Share of each bovine category in the average herd of farm groups in each sub-region. Data source : FADN database 2009-2013.

The following parameters for each composite animal thus depend on the composition of its herd : stocking density limit, income, production cost, feed requirements, purchase and sale price per composite cow, and organic nitrogen produced. We choose to use individual FADN data from 2009-2013 per farm group (see Table 30 in the supplementary materials) as the stocking density limit (i.e., the maximum number of Livestock Units (LU) per hectare of forage area) because there is no established universal standard for stocking density. Indeed, stocking density is an individual decision that depends on production factors and data such as erosion, soil conditions, grass quality, water availability, and the soil's capacity to support animals.

Cattle feed requirements are calculated on the basis of standard cases provided by IDELE for the Grand Est region (reference year 2015 for dairy cattle and 2011 for suckler cattle ¹¹, see Tables 32 and 34 in supplementary materials). There are three different feed rations for each composite cow, named according to the main feedstock: "maize fodder", "maize fodder-grass", and "grass". The production cost per composite cow is calculated using the IDELE standard cases (see Tables 35 and 36 in supplementary materials)

¹¹Idele, Dairy cattle, 2015, Idele, Suckler cattle, 2011

and include veterinary expenses, milk controls, insemination costs, and various other expenses. In addition, I also take into account the cost of processing grass into silage and baling, based on processing cost data provided by [Bamière et al., 2023] (see Table 38 in supplementary materials).

The revenue per composite animal per feed ration (see Tables 39 and 40 in supplementary materials) is estimated by using revenue data per animal type from the FADN 2009-2013 farm groups data¹². For suckler cows this revenue includes the sale of livestock and by-products (hides, meat) and for dairy cows it also includes revenue from milk. The initial feed ration is determined based on the proportion of maize fodder in the main forage area, using the criteria from IDELE standard cases. To estimate the income from milk and from the sale of cattle and other products (meat, hides) of the alternative feed rations, I apply variation coefficients (provided by IDELE) to the income associated with the initial feed ration, which allows to take into account differences in milk and meat yields.

The organic nitrogen excreted per composite cow in each feed ration and available for spreading as well as that excreted during grazing, are calculated based on the OMINEA database, [OMINEA, 2023]¹³ (reference year 2013), and time spent grazing in the field according to IDELE standard cases (see Table 41 in supplementary materials).

¹²In the same way as for the proportions of each animal category per composite dairy cow, for farms specialising in mixed livestock and mixed crop-livestock, income from the sale of cattle and other products (meat, hides) per dairy cow is the average for farms specialising in dairy in the corresponding sub-region. The remaining income from the sale of cattle and other products (meat, hides) per farm group is allocated to suckler cows.

¹³Ominea database from Citepa

2.3.3 GHG emissions, carbon sequestration, and mitigation practices

The GHG scope of this work includes CO_2 emissions related to fuel combustion and production as well as upstream emissions associated to agricultural machinery production, per hectare of crop activity. These data are sourced from the ADEME Base Carbone and are estimated at the national level (see Tables 19 and 20 in supplementary materials). I consider the 0-30 cm soil horizon for soil organic carbon (SOC) sequestration, which is the reference horizon used in IPCC reports and national inventories. For initial carbon stocks in soils, I use average data per sub-region and land use from [Martin et al., 2019], derived from a national measurement campaign (see Table 21 in supplementary materials). The soil carbon sequestration rates are derived from [Arrouays et al., 2002], based on a literature review (see Table 24 in supplementary materials). The maximum carbon stocks for baseline crop activities (without mitigation practices) are from [Bamière et al., 2023] and are simulated using the STICS crop model over a 30-year time horizon, whereas those for bare fallow are from [Arrouays et al., 2002] (see table 23 in supplementary materials). Regarding animal activities, I take into account CH_4 associated with enteric fermentation, as well as CH_4 and N_2O related to effluent management per composite animal (including storage and production of effluent in building without spreading), estimated from the OMINEA database (reference year 2013) (see Table 42 in supplementary materials). Additionally, N_2O direct emissions, volatilization and lixivation related to effluent discharge during grazing and from the spreading of manure and mineral fertilizers are also considered, sourced from IPCC [Eggleston et al., 2006]. Furthermore, upstream CO_2 emissions related to mineral fertilizer production are taken into account and also sourced from the Base Carbone (see Table 44 in supplementary

materials). Finally, I consider GHG emissions related to the purchase of animal feed, using data from the ADEME Base Carbon for milk powder and bovine concentrate. We calculate the emission factor linked to straw purchase on the basis of several assumptions, using data from the ADEME Base Carbone, FADN data on typical farm groups, and IPCC emission factors (see Table 44 in supplementary materials).

The mitigation practices I consider are among those identified for the French agricultural sector by [Pellerin et al., 2017] and [Bamière et al., 2023], who provide a consistent and detailed assessment of mitigation costs and potential, as well as detailed information on agricultural practices in the French context. Furthermore, their marginal abatement cost curves and results align with similar European studies conducted by [Moran et al., 2011, Schulte et al., 2012]. I selected the following practices : the use of nitrification inhibitors to fertilize crops, the introduction of legumes in the crop rotation, the introduction of temporary grassland at the expense of fodder maize mono-cropping, the introduction of intra-plot agroforestry, the plantation of hedges along plot edges, and the addition of 1% nitrate in ruminants ration. These practices offer a significant mitigation potential, despite the costs involved, and are not already widely adopted. In addition, they make it possible to cover the various levers, i.e., N_2O and CH_4 emissions mitigation as well as carbon sequestration).

Thus, I have data on variations in costs, yields, and nitrogen requirements and on mitigation potential, compared to the baseline, from [Pellerin et al., 2017, Bamière et al., 2023] (table 2.8 in appendix for details on each practice). Using these information, I estimate costs, yields, nitrogen requirements, GHG emissions, and carbon sequestration for each combination of crop and mitigation practice(s). The maximum carbon stock achievable by each carbon-sequestering practice is obtained by adding its total additional carbon sequestration potential over 30 years from [Bamière et al., 2023] to the initial carbon stock.

We briefly describe below the features of the mitigation measures of interest to us, but for more details refer to [Pellerin et al., 2017, Bamière et al., 2023, Bamière et al., 2021b, Bamière et al., 2021a].

Nitrification inhibitors substitute part of mineral fertilizer, 1 year out of 5 to avoid ecotoxicity issues, and decrease by 10% the amount of nitrogen that is substituted, thus reducing the spreading frequency. The practice decreases not only N_2O emissions, but also CO_2 emissions associated to fuel consumption and fertilizer manufacturing. It incurs costs for purchasing inhibitors as well as fuel and machinery savings. .

The **introduction of legumes on 16,5% of the crop rotation area** is done at the expense of soft wheat (1/6), rapeseed (1/6), and barley (2/3) areas and gross margins. Legumes are not fertilized, hence this practice reduces N_2O and CO_2 emissions by avoiding emissions from substitute crops and reducing mineral fertilizer and fuel consumption (direct and upstream emissions) . Estimates of the gross margin associated with peas and faba bean from [Bamière et al., 2023] turn out to be statistically insignificant for the three sub-regions of the Grand Est, and are even considered zero for Alsace. This can be explained by the fact that legumes yields are highly variable in general and the presence of legumes is extremely marginal in the Grand-Est region according to the 2009-2013 FADN data. Due to this, I assume that the legume yields for the Lorraine and Champagne sub-regions are those of the annual agricultural statistics estimated at the Grand Est level for the period 2009-2013, and for Alsace, I assume a zero yield (see Table 12 in supplementary materials). Prices and production costs are taken from [Bamière et al., 2023].

The **introduction of temporary grassland into fodder maize mono-cropping** allows not only soil carbon sequestration by the grassland but also a decrease in N_2O and CO_2 emissions resulting from input savings such as fertilizers, fuel, etc. It also

decreases nitrogen requirements for the following crop (fodder maize). Note that this practice impacts the type and quantity of forage available for livestock.

The **introduction of intra-plot agroforestry** sequesters carbon both in soil and above-ground biomass and has the highest potential according to [Bamière et al., 2023]. This practice consists in planting tree rows on 12.5% of the arable area, which then changes production costs, yields and nitrogen requirements per hectare. This practice incurs fixed costs corresponding to the purchase and planting of the trees as well as annual tree rows maintenance costs. Unlike [Bamière et al., 2023], I only consider incomes from the sale of pruning wood, but not that from the sale of trees. Indeed, according to [Bamière et al., 2023], trees are harvested after 50 years, while the time horizon is 30 years. Moreover, when the practice ends, I assume that the carbon sequestered in biomass is released.

The **plantation of hedges along the fields edges** corresponds to the plantation of 49 to 68 linear meters of hedge per hectare, depending on the average plot size in a given region. It decreases the cultivated area by 2.9%, 3% and 3.4% for Champagne, Lorraine, and Alsace, respectively. Like agroforestry, this practice sequesters carbon both in soil and biomass, and reduces N_2O and CO_2 emissions as a result of lower inputs linked to crop substitution with hedgerows. As with agroforestry, I take into account fixed installation costs, annual maintenance costs, and the sale of pruning wood, but not the sale of whole trees. In addition, the substitution of crop area with hedges results in a loss of gross margin from cash crops.

We consider that agroforestry and hedgerow areas are initialized at 0, given their low initial adoption in this region [Bamière et al., 2023].

All crop practices can be applied to fodder maize and the crop rotation, with the exception of the introduction of temporary grassland, which can only be implemented with

fodder maize, and the introduction of legumes, which can only be implemented with the crop rotation. Beyond that, the crop mitigation practices are all cumulative, and the cumulative assumptions are those derived from [Pellerin et al., 2017, Bamière et al., 2023] and are available in table 2.7 in the appendix. In addition, farmers can convert cropland to lower-emission crops or even permanent grassland, which sequesters carbon. Although this is not defined as a mitigation practice as such, it is a means of mitigating GHGs.

The **addition of 1 % nitrate in the ration of ruminants** decreases the emissions of CH_4 from enteric fermentation by 10 %. The implementation of this practice requires the purchase of nitrates but entails a reduction in urea usage in the feed ration, which, however, does not fully offset the costs, making this practice costly. This mitigation practice can be combined with any of the feed menus. Note that farmers can also choose the feed menu they give their dairy and suckler cows, which also has an impact on cow and farm emissions. For instance, the more grass in the feed ration, the lower the milk and meat yield but also the lower the need for concentrates and the production cost per cow. This has an impact both on upstream emissions linked to the production of concentrates and on carbon sequestration linked to the farm area allocated to grassland.

2.3.4 Carbon price trajectories

We propose to parameterize the model for 3 more or less ambitious carbon price trajectories (see figure 2.1). The first one is a social cost of carbon trajectory and is derived from a macroeconomic model at the scale of France, from [Germain et al., 2020], using

the "accounting" approach ¹⁴. The second trajectory is based on the 'cost' approach and corresponds to the social cost of carbon for France from [Quinet et al., 2019]. Note that [Germain et al., 2020] also provide an estimated trajectory using the "cost" approach which is closely aligned with the Quinet trajectory. Finally, to obtain a trajectory more in line with current carbon prices on the EU ETS market, reaching around 50 €. tCO_2e^{-1} in 2020, I consider a "low" trajectory corresponding to half the prices estimated by [Quinet et al., 2019].

These prices are estimated for the years 2020, 2030, 2040, and 2050. To obtain a 30-year trajectory, I assume a linear increase in the carbon price between these years.

¹⁴The "cost" approach and the "accounting" approach are the two main approaches used to estimate the social price of carbon. The "cost" approach is defined by the monetary value given by the community to actions enabling the emission of one ton of CO_2e to be avoided. The "accounting" approach consists in measuring the price at which a ton of CO_2e must be invoiced in order to achieve carbon neutrality.

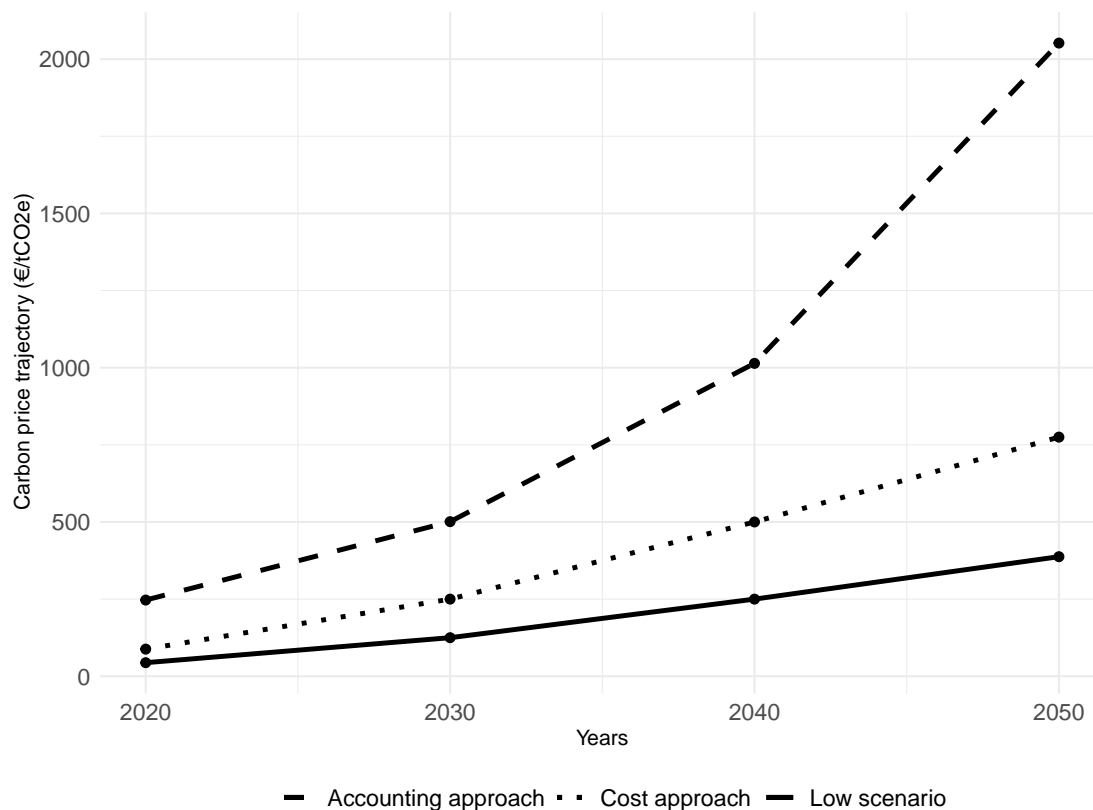


Figure 2.1: Carbon price trajectories

2.3.5 Model Validation

The model I developed is used in this article to simulate the impact of various GHG mitigation policies on farmers' decisions, regarding the implementation of GHG and/or carbon sequestration mitigation practices. Therefore, it is important to assess the model's ability to replicate farmers' choices in terms of crop allocation and herd size.

First, I determine the extent to which the model simulations match the observed data in terms of crop allocation and herd size for the reference period. To do so, I compare the share of each crop in the UAA of the crop activities considered as well as the herd size between, on the one hand, the model outputs for the first year of the baseline scenario (i.e.,

without mitigation policy) and, on the other hand, the average 2009-2013 FADN data for the farm groups. Table 2.3 indicates shows that despite a slightly larger area of grassland in the simulated data than in the observed data, and slightly lower areas for other crops, the crop allocation remains very similar. The differences with the grassland are due to the fact that, in the simulations, farmers implement more temporary grassland- fodder maize rotation and permanent grassland than in observed data, to meet the equations for animal feed requirements, aiming to balance the constraints. However, this difference compared to observed data remains limited.

Regarding livestock, the size of the suckler cattle herd is slightly lower between simulated and observed data and the size of the dairy cattle herd is the same. This is because I imposed a constraint preventing the cattle population from increasing, and this constraint is saturated in the case of the dairy herd.

		Grand-Est		Champagne		Lorraine		Alsace	
		Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.
Share in UAA (%)	Wheat	26,12	25,32	31,70	31,13	21,65	20,79	17,27	15,63
	Grain maize	6,47	6,30	2,52	2,44	1,54	1,49	49,74	48,42
	Barley	18,91	18,52	25,74	25,41	14,89	14,39	1,57	1,33
	Rapeseed	14,03	13,79	17,15	16,93	13,14	12,83	1,76	1,67
	Fodder maize	5,13	4,02	3,30	2,67	7,07	5,38	5,82	4,88
	Grasslands	29,34	32,05	19,59	21,41	41,72	45,12	23,83	28,05
	Total	100	100	100	100	100	100	100	100
Number of animals (k heads)	Dairy cows	391,9	391,9	131,18	131,18	209,98	209,98	50,74	50,74
	Suckler cows	175,2	171,11	49,0	45,36	118,3	118,31	7,8	7,43

Table 2.3: Simulated and observed data for the share of each crop in the total UAA of the cropping activities considered and for the number of dairy and suckler cows in thousands of head, at the level of the Grand Est and its 3 sub-regions. *Sim = Simulated, Obs= Observed. Simulated data are the results of the first year of simulation of the model in the baseline scenario (without mitigation policy) with the application's input parameters. Observed data are from FADN 2009-2013 data for the farm groups.*

In a second step, I compare the magnitude of GHG emissions between, on the one hand, the baseline scenario (with no mitigation policy) with the model parameterized with application data and, on the other hand, the regional inventory from ATMO Grand Est¹⁵. For the latter I average reported agricultural emissions for the years 2010 and 2012, at the scale of the Grand Est region. It's important to note that I can only compare the magnitude and not the precision of the results with observed data because, unlike ATMO, my results include upstream GHG emissions¹⁶. In addition, the simulated herd size is slightly lower than the observed one for 2009-2013. The baseline simulated GHG emissions are of $8.537MtCO_2e$, whereas observed emissions are $8.202MtCO_2e$. These

¹⁵Invent'Air V2023, ATMO Grand Est

¹⁶In other words, ATMO only accounts for on-site emissions, while I also consider emissions related to the production of purchased inputs (fertilizers, fuel, etc.).

figures show a relative consistency in the magnitudes.

Next, I run the model parameterized with the application data for the baseline scenario. Without mitigation regulation and with constant parameters for yields, prices, production costs, nitrogen requirements, etc., crop allocation and changes in herd size should remain stable over time, and I ensure this is the case. By observing the results of this simulation over the 31 periods in figure 2.2, I find that crop allocation remains constant. Regarding livestock, at the very beginning of the temporal horizon, there is a slight decrease in the suckler cattle herd and a decrease in the share of the dairy herd fed with the "maize" feed ration, in favor of an increase in the share of the dairy herd fed with the "grass" feed ration. Afterwards, the dairy and suckler herds as well as the feed rations remain constant.

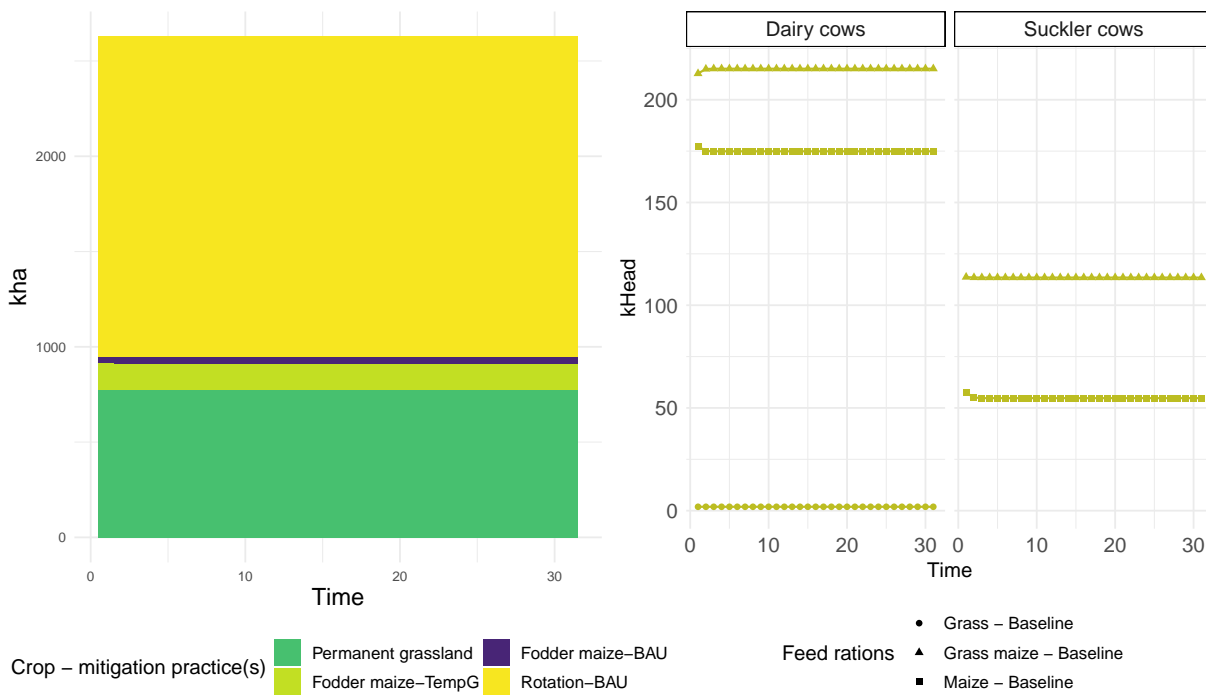


Figure 2.2: Simulated changes in herd size and in the share of each crop in the total UAA of considered cropping activities for farm groups in the Grand Est region. *Data: model outputs parameterized on application data, for baseline scenario (without mitigation policy). TempG= Temporary grassland.*

Finally, I want to assess the model's ability to simulate changes in crop allocation and herd size when crop prices change. For that purpose, I run the model (without mitigation policy), parameterized with crop prices (wheat, barley, rapeseed, and grain maize) from 2009-2013 and 2015, respectively (FADN data, see Table 14 in supplementary materials). Prices show an increase of 3% for legumes and a decrease of between 3% and 14% for other crops, between the period 2009-2013 and 2015. I then calculate the variation in simulated crop share (percentage point) and in simulated herd size (percentage) between these two periods¹⁷, and compare these simulated variations to the ones observed.

¹⁷ Assuming that farmers base their production decisions on the previous year's prices, I use 2016 areas and herd size.

		Grand-Est		Champagne		Lorraine		Alsace	
		Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.
Difference in share of UAA (in % point)	Wheat	1.72	-0.05	2.04	-0.09	1.96	-0.00	2.89	-0.05
	Grain maize	-0.38	-0.02	-0.93	-0.02	-0.52	-0.00	-4.29	-0.10
	Barley	-1.82	-0.03	-1.58	-0.05	-1.39	-0.00	0.40	-0.01
	Rapeseed	-1.09	-0.02	-0.59	-0.04	-1.12	-0.00	-0.24	-0.00
	Fodder maize	0.69	0.02	0.24	0.03	0.91	0.00	0.83	0.01
	Grassland	0.88	0.10	0.82	0.18	0.16	-0.00	0.40	0.15
Changes in herd size (in %)	Dairy cows	1.45	0.00	-17.05	0.00	9.43	0.00	17.88	0.00
	Suckler cattle	6.34	1.69	14.01	5.58	-0.64	0.00	65.99	4.94

Table 2.4: Simulated and observed variation in the share of each crop in the total area of the cropping activities considered (in percentage points) and variation in dairy and suckler cow herds size (in %), at the level of the Grand Est and the 3 sub-regions. *The observed data are based on FADN data from the reference period (the aggregated data for the period 2009-2013) and year 2016 (see tables 2.10 and 2.11 in appendix). As for the simulated data, they correspond to the difference between model outputs at t=1 for two crop prices scenarios: reference period (i.e., baseline scenario) and 2015 price data for wheat, barley, rapeseed, and grain maize (see table 2.9 in appendix).*

Table 2.4 shows that the model generally succeeds in simulating changes in crop allocation between the two periods, with an increase in areas dedicated to fodder maize and permanent grassland and a decrease in most other crops. However, the model is not able to reproduce the slight increase in wheat areas due to the grouping wheat, barley, rapeseed, and grain maize into the same crop rotation. Thus, if a farmer reduces the share of this rotation, the areas dedicated to all these crops decrease accordingly. Regarding livestock, the simulations show an increase in the number of suckler cattle, which aligns with the observed data. However, the modest increase in the dairy herd observed is not reflected in the simulation results. This disparity can be attributed to the constraints I imposed on livestock evolution.

Overall, these results indicate that the model appears capable of simulating consistent decisions by the farmers regarding crop allocation, herd size, and feed rations.

2.4 Results

For a given carbon price trajectory, the amount of the tax or subsidy follows the carbon price each year. The various policy scenarios - namely a tax on net GHG emissions, a GHG tax, a subsidy for carbon sequestration in soil and above-ground biomass and a subsidy for carbon sequestration in above-ground biomass - are then evaluated and compared against each other for each carbon price trajectory. Comparison criteria include net GHG mitigation and the marginal net GHG mitigation cost to farmers of implementing mitigation practices, both over the entire time horizon. In other words, I examine the difference in efficiency in terms of net GHG emissions and costs to farmers between a tax covering all sources and sinks of GHG emissions in the agricultural sector and several partial coverage policies. To understand these differences in efficiency, I examine the mitigation strategies employed by farmers in response to the different policies, i.e. changes in livestock numbers, animal feeding systems, combination of crop-mitigation practice(s) allocation. Finally, I end the analysis of the results by exploring the consequences of each policy scenario on crop production and on the cost of maintaining permanent grassland. It is important to consider these aspects because a significant loss of production could lead to a leakage of GHG emissions, and therefore be ineffective overall, and a high cost of maintaining permanent grassland could lead to reversals despite the ban, and therefore to carbon de-sequestration.

The results, displayed in Table 2.5, illustrate that, regardless of the carbon price trajectory, all mitigation policies considered lead to a reduction in net greenhouse gas

emissions.

The most effective scenario in terms of net GHG mitigation is, as according the theory, the one that covers all sources and all sinks, i.e. the tax on net GHG emissions. For this scenario, the mitigation achieved varies between approximately $370MtCO_2e$ and $430MtCO_2e$ over the 30 periods compared to the baseline GHG emissions. The difference in terms of net GHG mitigation is significant between policies covering all sources and sinks and partial coverage policies: depending on the carbon price trajectory, compared with a tax on net GHG emissions, a tax on GHG emissions leads to net GHG emissions that are 185 to 340% higher, and subsidies for total carbon and biomass lead to net GHG emissions that are 80 to 95% higher.

The results in terms of net mitigation and marginal cost of mitigation are very close between both total carbon sequestration subsidy and carbon sequestration only in biomass subsidy scenarios. This is interesting because monitoring the carbon stored in soils is very costly. Moreover, subsidies for total and biomass carbon sequestration prove to be more cost-effective than a tax on GHG emissions. These subsidies result in higher net mitigation at a lower marginal cost, irrespective of the carbon price trajectory scenario. For instance, in a Medium carbon price scenario, the net mitigation achieved by the subsidy scenarios for total carbon sequestration and biomass alone is 309.23 and 308.25, respectively, compared to 129.96 for the GHG tax scenario. The marginal cost of net mitigation is approximately 27 euros/tCO₂e for the both carbon sequestration subsidy scenarios and 111.81 euros for the GHG tax scenario. In this Medium scenario, the net mitigation of the GHG tax is thus half as much as in the two sequestration subsidy scenarios, with a marginal cost more than four times higher. For the low and high scenarios, the marginal costs of the GHG tax scenario are respectively higher by a factor of more than 2 and 5, and net GHG mitigation is lower by a factor of more than 3

and 2 than for the two (total and biomass) carbon sequestration subsidy scenarios.

Furthermore, it should be noted that the marginal net GHG mitigation costs over the time horizon for implementing practices in total and biomass carbon sequestration subsidy scenarios are lower than for the net GHG tax scenario.

In order to understand the differences in effectiveness in terms of net GHG mitigation between the policy scenarios, consider Figure 2.3, which illustrates the evolution of net GHG emissions over the time horizon for each mitigation policy scenario and carbon price trajectory, at the scale of the Grand Est region. For each policy scenario, at each time period, net emissions are determined by subtracting the amount of sequestered carbon from the total greenhouse gas emissions. For carbon sequestration, the total carbon storage (soil+biomass) from the baseline scenario (see figure 2.4) is subtracted from the total carbon storage resulting from the implementation of the policy. In figure 2.3, I can see that the subsidy for total carbon sequestration (in soil and biomass) and the subsidy for above-ground biomass carbon sequestration have the advantage of rapidly reducing net GHG emissions, even if they increase slightly thereafter, while remaining lower than in the GHG tax scenarios.

Carbon price scenario		Low				Medium				High			
Policy scenario	BAU	Net GHG	GHG	Carbon soil+ biomass	Carbon biomass	Net GHG	GHG	Carbon soil+ biomass	Carbon biomass	Net GHG	GHG	Carbon soil+ biomass	Carbon biomass
Average GHG mitigation comp. to BAU ($MtCO_2e. yr^{-1}$)	0.00	0.17	0.17	0.03	0.03	0.18	0.26	0.03	0.03	0.18	0.26	0.04	0.03
Average SOC sequestration comp. to BAU ($MtCO_2e. yr^{-1}$)	0.00	1.19	-0.15	1.17	1.16	1.23	-0.84	1.22	1.22	1.23	-1.25	1.23	1.22
Average Biomass sequestration comp. to BAU ($MtCO_2e. yr^{-1}$)	0.00	7.42	0.00	7.22	7.17	7.76	0.00	7.76	7.76	7.76	0.00	7.76	7.76
Net GHG emissions ($MtCO_2e$)	282.25	-87.72	209.72	-5.85	-3.93	-135.83	152.86	-26.98	-26.44	-145.89	123.22	-28.07	-26.49
Net GHG Change comp.to Net GHG scenario (%)		0.00	339.08	93.33	95.52	0.00	212.54	80.14	80.53	0.00	184.46	80.76	81.84
Marginal net GHG mitigation cost ($€tCO_2e^{-1}$)		31.55	59.31	25.68	25.58	38.87	111.81	26.99	27.02	42.14	150.47	27.00	27.04

Table 2.5: Summary of results for each mitigation policy and carbon price trajectory scenarios, over the entire time horizon, at the scale of the Grand Est region. *comp. to* = compared to, *BAU* = baseline scenario. *Net GHG Change vs. Net GHG Scenario* = Change in net GHG mitigation compared to net GHG scenario for the same carbon price trajectory scenario. The marginal cost of net GHG mitigation is calculated by dividing the sum of discounted opportunity cost of farmers over the time horizon of the selected policy scenario compared to the baseline with net GHG mitigation. Negative values for net GHG emissions = carbon sequestration is higher than GHG emissions.

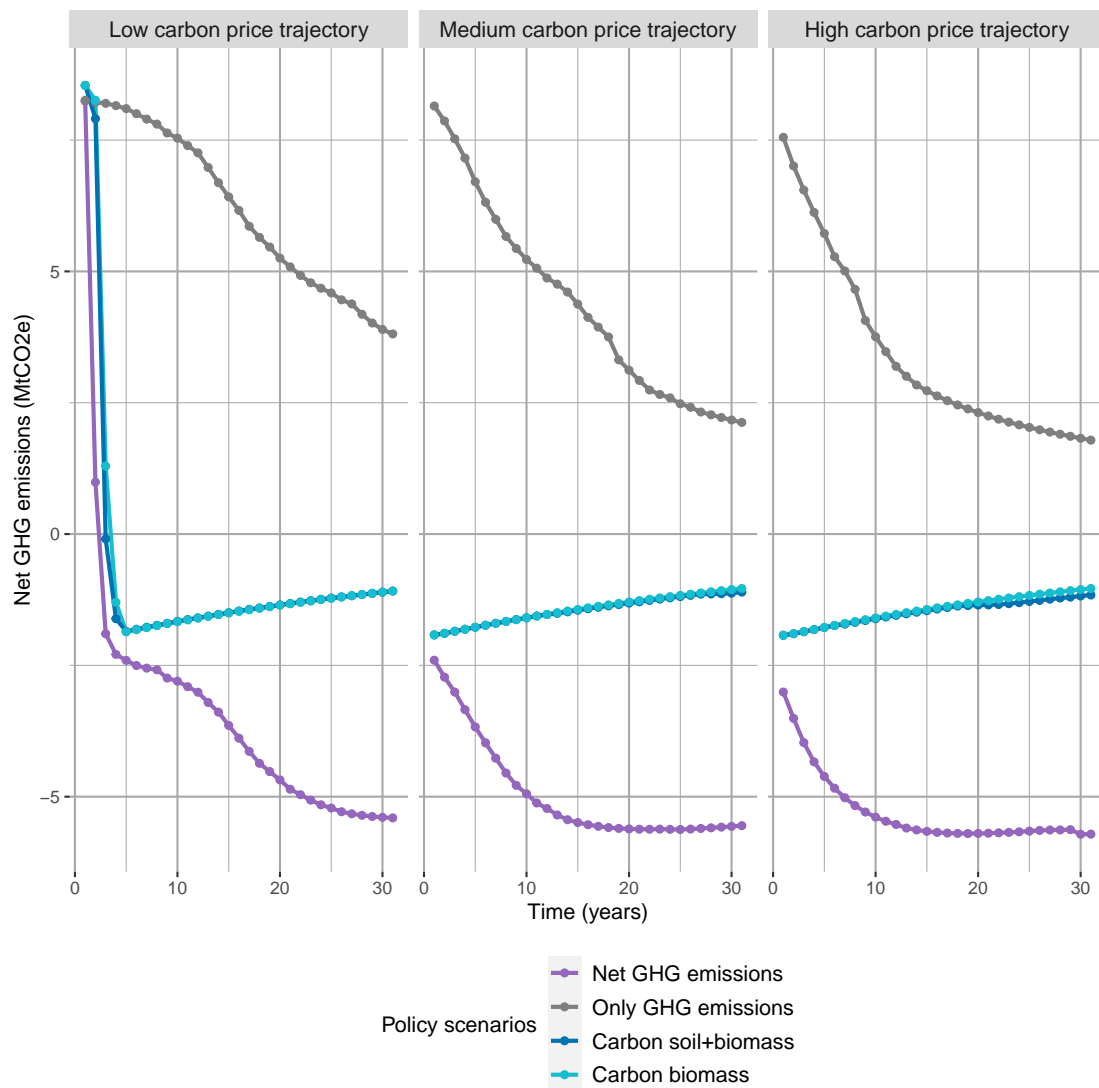


Figure 2.3: Trajectories of net GHG emissions over time for each mitigation policy and carbon price scenario at the scale of the Grand Est region.

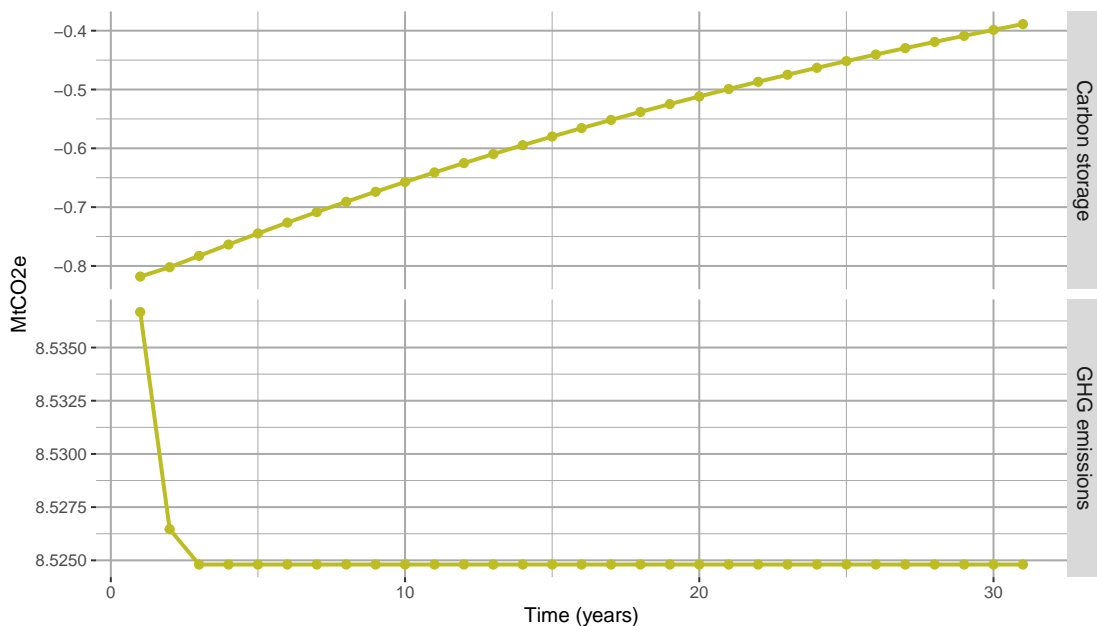


Figure 2.4: Trajectories of GHG emissions and carbon storage over time for baseline scenario, at the scale of the Grand Est.

In order to understand the differences in net emissions trajectories between policy scenarios, I can observe Figure 2.5 which shows the breakdown of net emissions arising from the various policies between GHG emissions and total carbon sequestration in soil and biomass. The GHG emissions tax scenario shows a de-sequestration of carbon over time, the higher the carbon price trajectory, the earlier this sequestration takes place in the time horizon. Note that the net emissions tax scenario as well as the total and the biomass carbon sequestration subsidy scenarios achieve the same amount carbon sequestration over the time horizon for the medium and high carbon trajectories. In addition, the two scenarios subsidising carbon sequestration have the same GHG emissions levels. In these policy scenarios, GHG emissions remain constant (and close to that of the baseline, see Figure 2.4), while sequestration rapidly reaches a high level but then slightly declines, which explains why the net GHG emissions of these scenarios increase slightly over

time.

The slight decrease in total carbon sequestration in the net GHG tax and soil and biomass sequestration subsidy, and biomass sequestration subsidy scenarios, is explained by the kinetics of the evolution of the stock in the soil. In fact, the carbon stock in the soil increases over time, but less and less. Moreover, carbon storage in the baseline is less and less negative (the stock decreases from one period to the next, but less and less, see Figure 2.4). This can be explained by the fact that permanent grassland (except in Alsace) has a tendency to store carbon naturally over time.

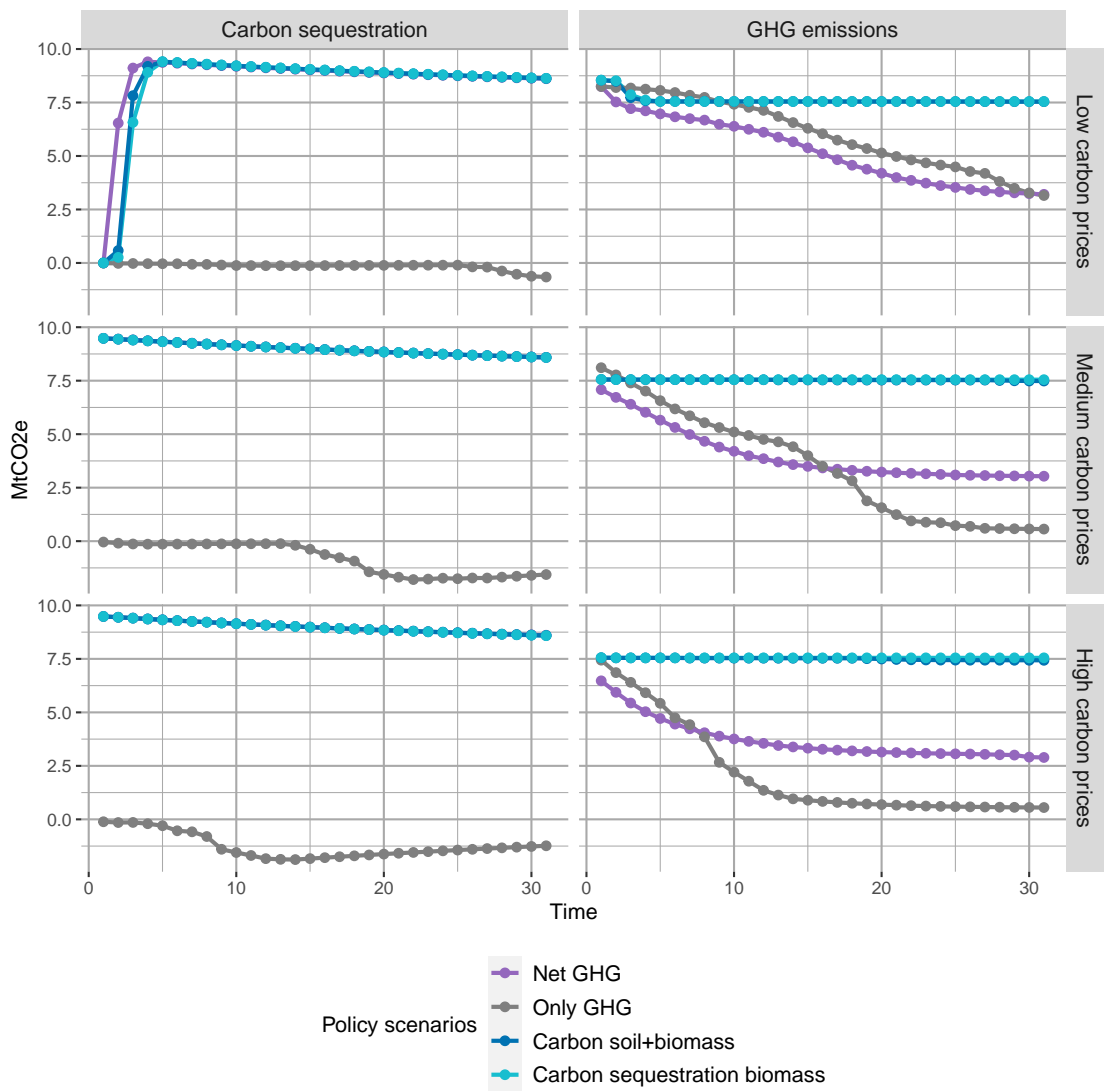


Figure 2.5: Trajectories of GHG emissions and carbon sequestration over time for each mitigation policy and carbon price trajectory at the scale of the Grand Est region. *In this graph, when carbon sequestration is negative, this means that there is a de-sequestration of carbon.*

These variations in GHG emissions and carbon sequestration are explained by the strategies adopted by farmers to maximize their profit under each of these different mitigation policies in terms of herd size, animal feeding (Figure 2.6), and choices of

crops and agricultural practices (Figure 2.7) over time. For the next two graphs, I focus solely on the implementation of mitigation practices in the medium carbon price trajectory, as the mechanisms are similar for low and high price trajectories (see appendix for details : figures 2.8 and 2.9 for herd evolution and figures 2.10 and 2.11 for crop practices evolution).

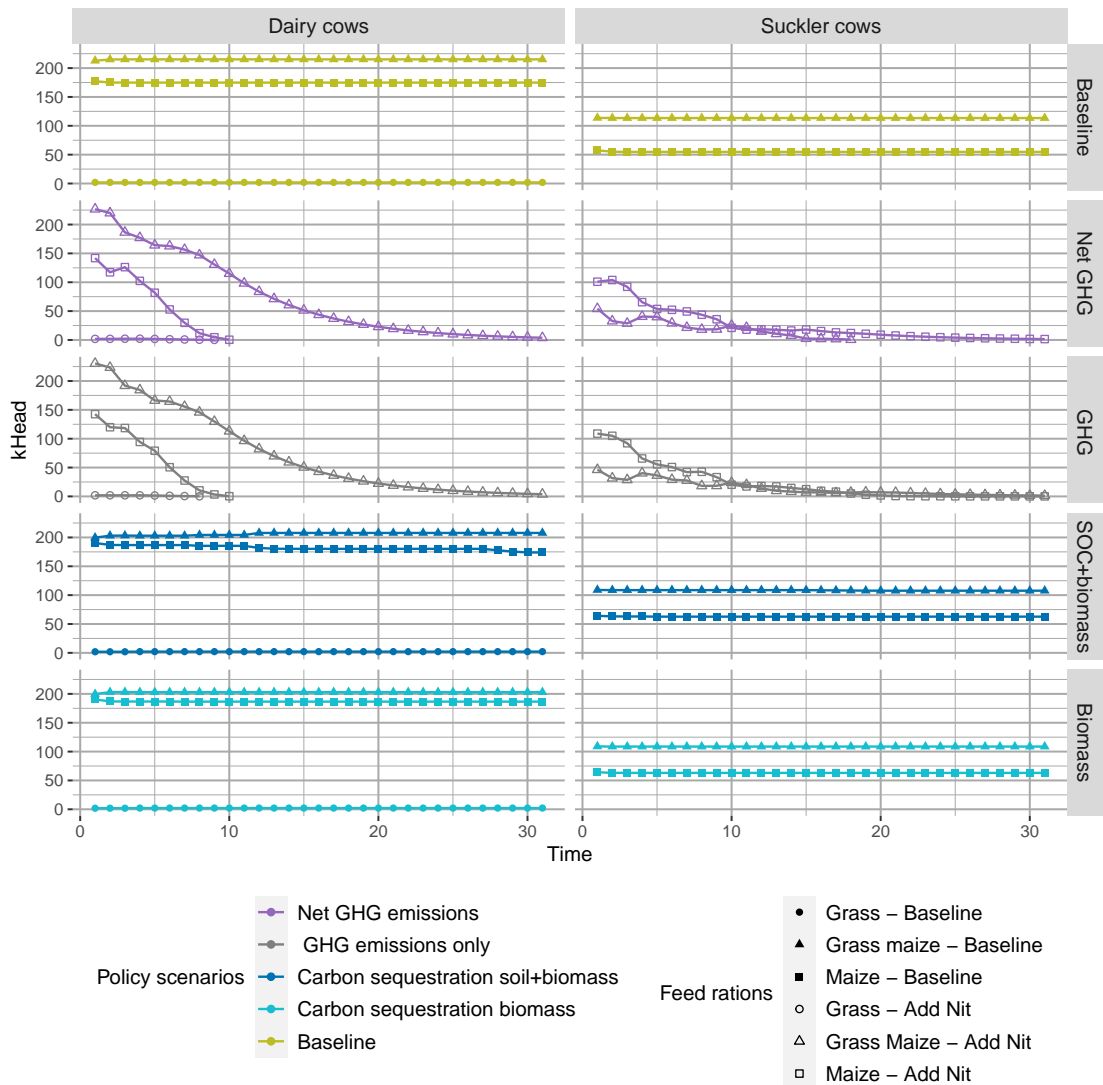


Figure 2.6: Herd size evolution over time for each mitigation policy in the medium carbon price trajectory at the scale of the Grand Est region. *The values on the y-axis represent the number of composite cow heads, expressed in thousands. Add Nit = addition of 1 % nitrate in the ration of ruminants.*

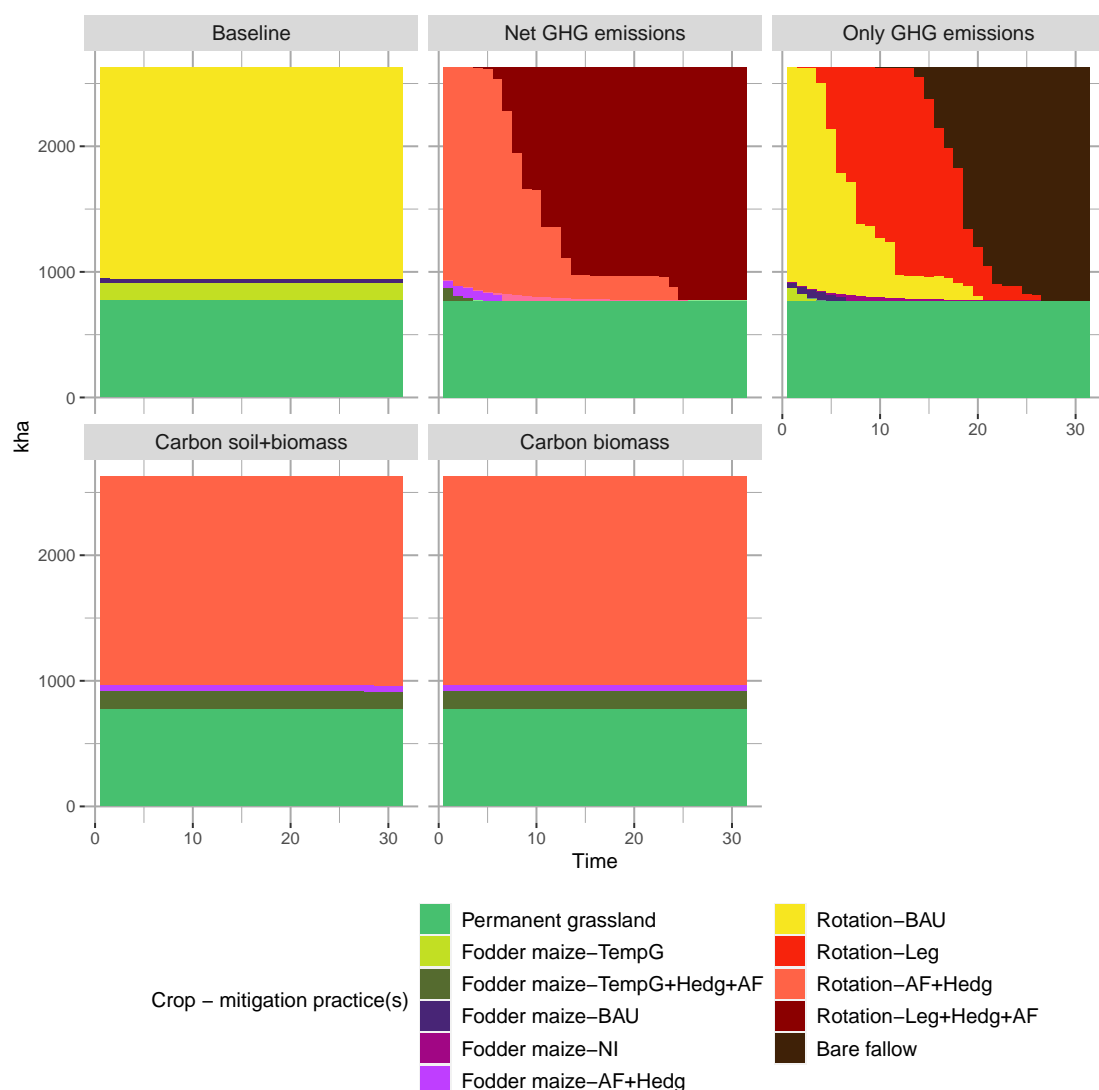


Figure 2.7: Evolution of crops allocation over time for each mitigation policy in the medium carbon price scenario, at the scale of the Grand Est region. *Each vertical bar represents the composition of the UAA in the Grand Est region for a specific policy scenario and time period.*

By relating the results in Figures 2.6 and 2.7, I can understand the trajectories of carbon sequestration and GHG emissions for the different policy scenarios. Firstly, for the net GHG tax scenario, I observe a significant decline in the dairy and suckler

cows over time, explaining the substantial reduction in GHG emissions. This explains also why the marginal mitigation cost of the tax on net greenhouse gases is higher than that of subsidies for total carbon and biomass sequestration, since reducing livestock farming reduces production and is costly, but it is a very high emitter and in this scenario emissions are taxed. Additionally, the crop allocation composition results for this scenario indicate the implementation of practices over the temporal horizon that both reduce GHG emissions (such as the introduction of legumes) and enhance carbon sequestration (agroforestry and hedges). This explains why this scenario also significantly reduces GHG emissions and sequesters carbon at the same time.

Similarly, the GHG tax scenario also demonstrates a decrease in dairy and suckler cows, equivalent to the net GHG tax scenario. Furthermore, the expansion of bare fallow land over time explains the de-sequestration in this scenario.

As for the scenarios of subsidies for total carbon sequestration and only biomass, the dairy and cattle livestock remains consistently high over time, resulting in elevated and constant GHG emissions in these scenarios. In the total carbon sequestration subsidy scenario, there is a shift in the number of dairy cows fed with the corn system to those fed with the maize-grass system. While not visible in the figure, there is a slight difference in crop allocation between the subsidy scenarios for total carbon sequestration and biomass carbon sequestration (approximately 1000 ha). Indeed, the area of temporary grassland - fodder maize rotation is slightly higher in the total carbon sequestration subsidy scenario (refer to Table 2.4 in the appendix for the difference in fodder maize produced between these scenarios), since this practice sequesters carbon in the soil (which is remunerated in this scenario) and not in the above-ground biomass. This explains the difference in dairy cow feed rations between total carbon and biomass subsidies scenarios. Apart from that, the implementation of cropping practices is similar between these two scenarios

which explains their close results in terms of net GHG mitigation and net marginal GHG mitigation cost. In addition, as in the tax on net GHG emissions scenario, these both scenarios show wide adoption early in the time horizon of practices that sequester a lot of carbon in above-ground biomass, which explains the rapid carbon sequestration over time in these three scenarios.

Let us now examine, for each scenario, the results in terms of crop production and cultivated area variation compared to the baseline, as well as the average cost of maintaining permanent grassland per hectare.

In terms of crop production variation compared to the baseline, the scenarios of subsidy for total carbon sequestration and biomass exhibit a maintained silage production and a decrease in wheat, barley, and grain rapeseed production, nearly identical across the three carbon price scenarios, at around 11%. This can be explained by the yield loss induced by the implementation of agroforestry and hedges in these scenarios.

For the carbon tax scenario, the loss in wheat, grain, barley, and rapeseed production is lower compared to the two sequestration subsidy scenarios (total and biomass), except for barley, which shows a stronger loss compared to the baseline. Similar to the carbon tax scenario, maize fodder production decreases significantly compared to the baseline, approximately 50% to 80% depending on the carbon price scenarios. This can be explained by the reduction in livestock in these two scenarios.

For the Medium and High carbon price scenarios, the carbon tax scenario shows a decrease in wheat, barley, grain maize and rapeseed production, explained by fallowing cultivated land. In the Low-carbon price scenario, there is both an increase in the production of wheat, barley, rapeseed and grain maize rotation and an increase in bare fallow areas of around 2%, compared to the baseline.

Carbon price scenario	Policy scenario	Wheat (%)	Grain (%)	Barley (%)	Rapeseed (%)	SilageM (%)	Legumes (k tons)	Variation crop areas comp. to BAU (%)	Cost maintenance of Perm Grass (€. <i>ha</i> ⁻¹)
	Baseline	0	0	0	0	0	0	0	377.15
Low	Net GHG	-6.16	-5.68	-11.41	-8.43	-52.05	6882.65	0	474.99
	Only GHG emissions	2.26	5.28	-4.04	-0.60	-51.53	9759.48	-2.39	342.91
	Carbon soil+biomass	-11.76	-11.56	-11.49	-11.50	1.13	0	0	458.01
	Carbon biomass	-11.67	-11.54	-11.38	-11.42	1.69	0	0	577.08
Medium	Net GHG	-5.34	-4.74	-14.02	-8.38	-77.71	12150.17	0	677.26
	Only GHG emissions	-39.12	-23.69	-44.31	-42.58	-78.01	7193.15	-30.25	315.17
	Carbon soil+biomass	-12.50	-12.42	-12.27	-12.29	0.63	0	0	682.30
	Carbon biomass	-12.63	-12.95	-12.28	-12.28	1.96	0	0	924.32
High	Net GHG	-6.87	-5.80	-17.64	-9.86	-71.29	16267.01	0	1122.53
	Only GHG emissions	-72.83	-62.74	-76.21	-75.27	-83.31	4606.66	-52.43	347.08
	Carbon soil+biomass	-12.27	-11.40	-12.22	-12.26	-0.64	0	0	1257.46
	Carbon biomass	-12.63	-12.97	-12.28	-12.28	2.11	0	0	1793.51

Table 2.6: Variation in crop production compared with baseline in percentage of tonnes produced, variation in number of hectares under cultivation (converted to bare fallow) and cost of maintaining permanent grassland, for each mitigation policy and carbon price trajectory at the scale of the Grand Est. *comp. to BAU = compared to baseline scenario. Perm Grass = permanent grassland. Cost maintenance of PermGrass is the average cost of maintaining the initial permanent grassland. It is the average per ha per year for all farm groups (weighted by their representativeness) of the dual associated with the constraint of maintaining the initial permanent grassland.*

Regarding the cost of maintaining the initial permanent grassland area, it can be interpreted as an average opportunity cost over the time horizon to maintain one hectare of initial permanent grassland. It is surprising to note that this cost is higher for the scenarios of net GHG tax and subsidy for total carbon sequestration and in biomass than in the GHG tax scenario, which does not remunerate carbon sequestration or tax de-sequestration. This explained by the fact that in scenarios remunerating carbon sequestration, other practices sequester more carbon than permanent grassland (such as hedges and agroforestry) and could therefore yield more if implemented on these areas. Whereas, in the case of the GHG tax scenario, the solutions to reduce GHGs from the land have a more limited potential, and the grassland does not generate revenue but incurs low costs and has low emissions.

2.5 Discussion and Conclusion

This article examines, for the same carbon price levels, the difference in efficiency between a tax covering all sources and sinks of GHG emissions in the agricultural sector and several partial coverage policies, namely a tax on GHG emissions, a subsidy for carbon sequestration in soil and biomass, and a subsidy for above-ground biomass carbon sequestration. The aim is also to see whether one of the partial coverage policies could achieve higher or equal net GHG emissions mitigation for a lower marginal mitigation cost than the others.

In the context of the Grand Est region and given my hypotheses, the results suggest that, all of the partial coverage policy scenarios show net GHG mitigation, although there is carbon de-sequestration in the GHG tax scenario and high GHG emissions in the total and biomass carbon sequestration subsidy scenarios. But the difference in terms of

net GHG mitigation is significant between policies covering all sources and sinks and partial coverage policies: depending on the carbon price trajectory, compared with a tax on net GHG emissions, a tax on GHG emissions leads to net GHG emissions that are 185 to 340% higher, and subsidies for total carbon and biomass lead to net GHG emissions that are 80 to 95% higher.

Moreover, a subsidy for total or biomass carbon sequestration is more cost-effective than a tax on GHG emissions. Indeed, according to the carbon price trajectory scenarios, compared with the carbon sequestration subsidy scenarios (total and biomass), the marginal costs of the GHG tax scenario are higher by a factor of between 2 and 5, and mitigation is lower by a factor of between 2 and 3.

In addition, it is interesting to note with these carbon sequestration subsidy scenarios (total and biomass), the implementation of carbon sequestration practices is rapid and significant, which is a crucial point given the IPCC's call for "rapid decarbonisation" to counter climate change and its impacts.

Total carbon sequestration and biomass carbon sequestration scenarios have very similar results, in terms of net GHG mitigation, marginal mitigation cost, and implementation of mitigation practices. This is interesting because controlling only above-ground biomass would save on control costs while being almost as effective as if the subsidy also covered soil carbon. Thus, taking into account monitoring costs, partial coverage of sequestration with a subsidy for above-ground biomass would a priori be more cost-effective than coverage of all carbon sinks (soil + biomass).

As for the implementation of a tax on GHG emissions, it induces a decrease in GHG emissions mainly due to an increase in bare fallow - supposedly de-stocking soil carbon - and a decrease in dairy and suckler livestock, which leads to a decrease in agricultural production.

Our work could be extended in several ways. Firstly, the economic model could be applied to other regions, and other mitigation practices could be considered, to see if this changes the results. By integrating fertilizer requirements as an endogenous parameter and considering a yield-nitrogen response function, it would be possible to reduce greenhouse gas emissions by reducing nitrogen inputs. Nitrous oxide emissions are one of the main sources of emissions in the agricultural sector, and the model currently only incorporates a few practices to reduce nitrogen inputs. Moreover, the results are significantly influenced by the constraints I have imposed on livestock variations. Since the supply model does not take into account, by definition, the effects of supply variation on prices, it is important to note that without these constraints, livestock numbers increase excessively. However, in reality, there are other constraints such as limitations in terms of buildings, human resources, and equipment, which prevent unlimited livestock growth. Ultimately, this article can be seen as an exercise in which I have set the maximum livestock capacity and in which current livestock can only decrease.

Although I have considered the possibility of introducing fallow lands, I have not examined the possibility that these lands may be either completely removed from the agricultural market and urbanized, which would have a much more significant impact on the carbon de-sequestration of these lands, or converted to forest, which would sequester a lot of carbon.

Finally, it might be interesting to consider costs other than the technical costs of implementing practices, such as information and education costs, which could have an impact on the cost of adopting practices. Additionally, I do not account for the costs of monitoring greenhouse gas emissions and carbon sequestration, which affect the cost-effectiveness of policies. I model regulation as results-based, without considering the technical costs of monitoring or behavioral aspects such as risk aversion regarding

received/paid amounts, which impact the adoption of practices and thus the marginal cost of mitigation. In Chapters 3 and 4, I account for the impact of farmers' risk aversion in monitoring costs and highlight trade-offs in terms of costs and uncertainties among results-based, model-based, and also practice-based monitoring methods in Chapter 4.

2.6 Appendices

2.6.1 Appendix of Data

Mitigation practice(s)	Area share on which (combined) practices are applicable	
	Fodder maize baseline	Rotation baseline
Nitrification inhibitors	Applicable on 100%	Applicable on 100%
Including legumes in the rotation	Not applicable	$\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively
Including legumes in the rotation +Nitrification inhibitors	Not applicable	$\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively and inhibitors on 100% of surface area
Agroforestry	Reduces cultivated area by 12.5%	reduces cultivated area by 12.5%
Hedges	Reduces cultivated area by approx. 3% (depends of regions)	reduces cultivated area by approx. 3% (depends of regions)
Inserting temporary grassland into fodder maize	50% of maize fodder and 50% of temporary grassland	Not applicable
Agroforestry + Hedges	Cumulative reduction in surface area for both practices (Intra-plot tree lines + hedges) 12.5% + approx. 3% less cultivated surface area.	Cumulative reduction in surface area for both practices (Intra-plot tree lines + hedges) : 12.5% + approx. 3% less cultivated surface area.
Hedges + Nitrification inhibitors	Reduce the cultivated area by approx. 3%, apply nitrification inhibitors to the remaining.	Reduce the cultivated area by approx. 3%, apply nitrification inhibitors to the remaining.
Agroforestry + Nitrification inhibitors	Reduce the cultivated area by 12.5%, apply nitrification inhibitors to the remaining.	Reduce the cultivated area by 12.5%, apply nitrification inhibitors to the remaining.
Agroforestry + Hedges + Nitrification inhibitors	Reduce the cultivated area by 12.5% + approx 3%, apply nitrification inhibitors to the remaining	Reduce the cultivated area by 12.5% + approx 3%, apply nitrification inhibitors to the remaining

Including legumes in the rotation + Agroforestry	Not applicable	Reduce the cultivated area by 12.5% and $\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively to the remaining
Including legumes in the rotation + Hedges	Not applicable	Reduce the cultivated area by approx 3% and $\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively to the remaining
Including legumes in the rotation + Hedges + Agroforestry	Not applicable	Reduce the cultivated area by 12.5% and on the remaining : $\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively
Including legumes in the rotation + Nitrification inhibitors +Hedges	Not applicable	Reduce the cultivated area by approx 3%, and on the remaining : $\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively + inhibitors on 100%
Including legumes in the rotation + Nitrification inhibitors + Agroforestry	Not applicable	Reduce the cultivated area by 12.5%, and on the remaining : $\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively + inhibitors on 100%
Including legumes in the rotation + Nitrification inhibitors +Hedges + Agroforestry	Not applicable	Reduce the cultivated area by 12.5%+approx 3%, and on the remaining : $\frac{1}{6}^{th}$ of legumes in the rotation area, divided into $\frac{1}{6}^{th}$, $\frac{2}{3}^{rd}$, and $\frac{1}{6}^{th}$ on wheat, barley, and rapeseed, respectively + inhibitors on 100%

Inserting temporary grassland into fodder maize + Nitrification inhibitors	Inhibitor only on the 50% share of fodder maize	Not applicable
Inserting temporary grassland into fodder maize + Nitrification inhibitors + Hedges	Reduce the cultivated area by approx 3% and in the remaining : Inhibitor only on the 50% share of fodder maize	Not applicable
Inserting temporary grassland into fodder maize + Nitrification inhibitors + Agroforestry	Reduce the cultivated area by 12.5% and in the remaining : Inhibitor only on the 50% share of fodder maize	Not applicable
Inserting temporary grassland into fodder maize + Nitrification inhibitors + Agroforestry + Hedges	Reduce the cultivated area by 12.5% + approx. 3% and in the remaining : Inhibitor only on the 50% share of fodder maize	Not applicable
Inserting temporary grassland into fodder maize + Agroforestry	Reduce the cultivated area by 12.5% and in the remaining : 50% of fodder maize and 50% of temporary grassland	Not applicable
Inserting temporary grassland into fodder maize + Hedges	Reduce the cultivated area by approx. 3% and in the remaining : 50% of fodder maize and 50% of temporary grassland	Not applicable
Inserting temporary grassland into fodder maize + Hedges + Agroforestry	Reduce the cultivated area by 12.5% + approx. 3% and in the remaining : 50% of fodder maize and 50% of temporary grassland	Not applicable

Table 2.7: Interactions between mitigation practices and applicability to crop activities. Describes how (combined) mitigation practices can be implemented in terms of surface area on fodder maize and wheat-barley-rapeseed-grain maize rotation, according to the assumptions made in the [Pellerin et al., 2017] for nitrification inhibitors and introduction of legumes in the rotation, and in [Pellerin et al., 2020] and [Bamière et al., 2021] for the other mitigation practices. The area taken up by the introduction of hedges around plots is estimated in [Bamière et al., 2023] according to sub-region: 2.9% in Champagne, 3% in Lorraine and 3.4% in Alsace.

Practices	Effects on GHGs	Parameters variation (yields, production costs, ..)	Variation in GHG emissions	SOC sequestration (0-30 cm) ($tCO_2e.ha^{-1}.yr^{-1}$)	Above-ground biomass sequestration ($tCO_2e.ha^{-1}.yr^{-1}$)	Scale of costs and mitigation potential	Source
Use of nitrification inhibitors	N_2O , CO_2	Application of inhibitors 1 year in 5, -6,1 $€ha^{-1}.yr^{-1}$ fuel savings during spreading, +0,34 $€kgN^{-1}$ purchasing inhibitors, 10% reduction in the reducible nitrogen dose per crop the year of application	-0.003 $tCO_2e.ha^{-1}.yr^{-1}$ due to fuel combustion, -0.0001584 $tCO_2e.ha^{-1}.yr^{-1}$ of upstream emissions linked to fuel use	0	0	Farm group for crop rotation (different share of crop activities), Sub-region for fodder maize	[Pellerin et al., 2017, Bamière et al., 2021b]
Introduction of legumes in the crop rotation	N_2O , CO_2	Yield reductions on areas where legumes are introduced, crop production costs on 1/6 of the ha replaced by those of legumes, suppression of nitrogen input : - 0.033 $tN.ha^{-1}.yr^{-1}$ per ha of legumes introduced	+0.077 $tCO_2e.ha^{-1}.yr^{-1}$ per ha of legumes due to symbiotic nitrogen fixation, -0.021 $tCO_2e.ha^{-1}.yr^{-1}$ and - 0.00465 per ha of legume for respectively direct and indirect CO_2 emissions linked to the use of fuel (reduction in nitrogen spreading)	0	0	Farm group (different share of crop activities)	[Pellerin et al., 2017, Bamière et al., 2021b]

Temporary grassland insertion (in a fodder maize mono culture)	<i>SOC</i> , <i>N₂O</i> , <i>CO₂</i>	Variation in production (yields, nitrogen and production costs) due to reallocation of surface : 50% fodder maize, 50% temporary grassland, reduced nitrogen dosage on the crop following the grassland (fodder maize) (in <i>tN.ha⁻¹.yr⁻¹</i>): Champagne 0.017, Lorraine 0.012, and Alsace 0	Variation in GHG emissions due to reallocation of surface	Champagne : 1.168, Lorraine 1.148, Alsace 0.160	0	Sub-region	[Pellerin et al., 2020, Bamière et al., 2021a, Bamière et al., 2021b]
Intra-plot agroforestry 83	SOC, biomass, <i>N₂O</i> , <i>CO₂</i>	Variation in production (yield loss, reduction of production costs and nitrogen) on 12.5% of the surface due to tree planting, yield loss on the inter-row after 16 years of implementation, assuming annualization of this loss over 30 years: additional annual loss of 3.97%, Tree maintenance costs : 95.33 <i>€.ha⁻¹.yr⁻¹</i> , Sale of pruning wood: 14.7 <i>€.ha⁻¹.yr⁻¹</i> , Fixed installation costs (due to tree planting): 1007 <i>€.ha⁻¹</i>	12.5% reduction in GHG emissions per ha of crops, -0.12 <i>tCO₂e.ha⁻¹.yr⁻¹</i> substitution effect due to the use of thinning wood (energy)	Champagne 0.917, Lorraine 0.917, Alsace 0.88 (In the case of combination with the insertion of temporary grassland in fodder maize : Champagne and Lorraine 0.5692, Alsace 0.550)	3.3	Farm group for rotation (different share of crop activities) and Sub-regional for fodder maize	[Pellerin et al., 2020, Bamière et al., 2021a, Bamière et al., 2021b]

Planting hedges along plot edges	SOC, biomass, N_2O , CO_2	Decrease in yields, production costs and nitrogen inputs due to variation in crop area per ha (For Champagne -2.9%, Lorraine -3%, Alsace -3.4%), Maintenance costs ($€\cdot ha^{-1}\cdot yr^{-1}$): Champagne 62, Lorraine 61.83, Alsace 70.1, Wood sales ($€\cdot ha^{-1}\cdot yr^{-1}$): Champagne -24.9, Lorraine -25.5, Alsace -28.93, Fixes implementation cost ($€\cdot ha^{-1}$): Champagne 631, Lorraine 646, Alsace 732	Reduction in GHG emissions due to variation in crop area, -0.12 $tCO_2e\cdot ha^{-1}\cdot yr^{-1}$ substitution effect due to the use of thinning wood (energy)	On crop : Champagne 0.07, Lorraine 0.071, Alsace 0.078; On temporary grassland : Champagne 0.023, Lorraine 0.024, Alsace 0.027	Champagne : 0.858 Lorraine 0.877, Alsace 0.995	Farm group for rotation and sub-regional for fodder maize	[Pellerin et al., 2020, Bamière et al., 2021a, Bamière et al., 2021b]
Addition of 1 % nitrate in ruminants ration	CH_4	-11.6 $€\cdot head^{-1}\cdot yr^{-1}$ for dairy cow and 6.8 for young dairy cattle, 5.7 for young suckler cattle from nitrate purchases and urea savings	-0.289 $tCO_2e\cdot head^{-1}\cdot yr^{-1}$ for dairy cow and -0.203 for young cattle	0	0	Farm group (different composition of composite cattle)	[Pellerin et al., 2017, Bamière et al., 2021b]

Table 2.8: Additional mitigation potential and variation in parameters (yields, costs, nitrogen application, etc.) per mitigation practice. *The SOC is provided here as a yearly constant parameter, but it is only indicative since we consider carbon sequestration in the soil dynamically. The parameters associated with this dynamic process, such as the sequestration rate, initial carbon stock, and maximum stock, are represented in Tables 21, 23, and 24 in supplementary materials. Moreover, these data are additional to the baseline practice, but total costs, total yields, GHG emissions and other parameters are available for each combination of mitigation practice(s) - crop activity and livestock activity- mitigation practice(s), in supplementary materials (Tables 5, 6, 7, 8, 9, 10, 13, 15, 16, 17, 18, 19, and 20 for crop activities and 35, 36 and 42 for livestock activities).*

2.6.2 Appendix of the validation

		Champagne	Lorraine	Alsace	Grand Est
Share in UAA (%)	Wheat	31.04	20.79	15.58	25.27
	Grain	2.42	1.49	48.33	6.28
	Barley	25.36	14.39	1.32	18.50
	Rapeseed	16.89	12.83	1.67	13.77
	Fodder maize	2.70	5.38	4.90	4.04
	Grassland	21.59	45.12	28.20	32.15
	Total	100	100	100	100
Thousand Head Count	Dairy cows	131.18	209.98	50.74	391.90
	Suckler cows	47.89	118.31	7.80	174.01

Table 2.9: Share of each crop in the total UAA of the cropping activities considered and the number of dairy and suckler cows in k heads at the level of the Grand Est and the 3 sub-regions, from the results of the first year of simulation of the model in the baseline scenario (without mitigation policy) using 2015 FADN prices for crops (wheat, barley, rape, grain maize, legumes).

		Champagne	Lorraine	Alsace	Grand Est
Share in UAA (%)	Wheat	33.66	23.28	20.31	27.69
	Grain	1.85	0.98	46.14	6.42
	Barley	23.86	13.25	1.99	16.84
	Rapeseed	16.34	11.74	1.55	12.70
	Fodder	3.41	7.80	6.40	5.64
	Grassland	20.88	42.95	23.62	30.72
	Total	100	100	100	100
Thousand Head Count	Dairy cows	114.07	233.26	60.40	407.73
	Suckler cows	86.42	166.83	16.41	269.66

Table 2.10: Share of each crop in the total UAA of the cropping activities considered and the number of dairy and suckler cows in k head at the level of the Grand Est and the 3 sub-regions, from the aggregated public database of the FADN for the year 2016.

		Champagne	Lorraine	Alsace	Grand Est
Share in UAA (%)	Wheat	31.62	21.32	17.42	25.97
	Grain	2.77	1.50	50.42	6.80
	Barley	25.44	14.64	1.58	18.66
	Rapeseed	16.93	12.86	1.79	13.79
	Fodder maize	3.17	6.89	5.57	4.95
	Grassland	20.06	42.79	23.22	29.84
	Total	100	100	100	100
Thousand Head Count	Dairy cows	137.51	213.17	51.24	401.92
	Suckler cows	75.80	167.90	9.89	253.59

Table 2.11: Share of each crop in the total UAA of the cropping activities considered and the number of dairy and suckler cows in k head at the level of the Grand Est and the 3 sub-regions, from the average for the years 2009 to 2013 of the aggregated public database of the FADN.

2.6.3 Appendix of results

Herd size evolution over time for low and high carbon price trajectory

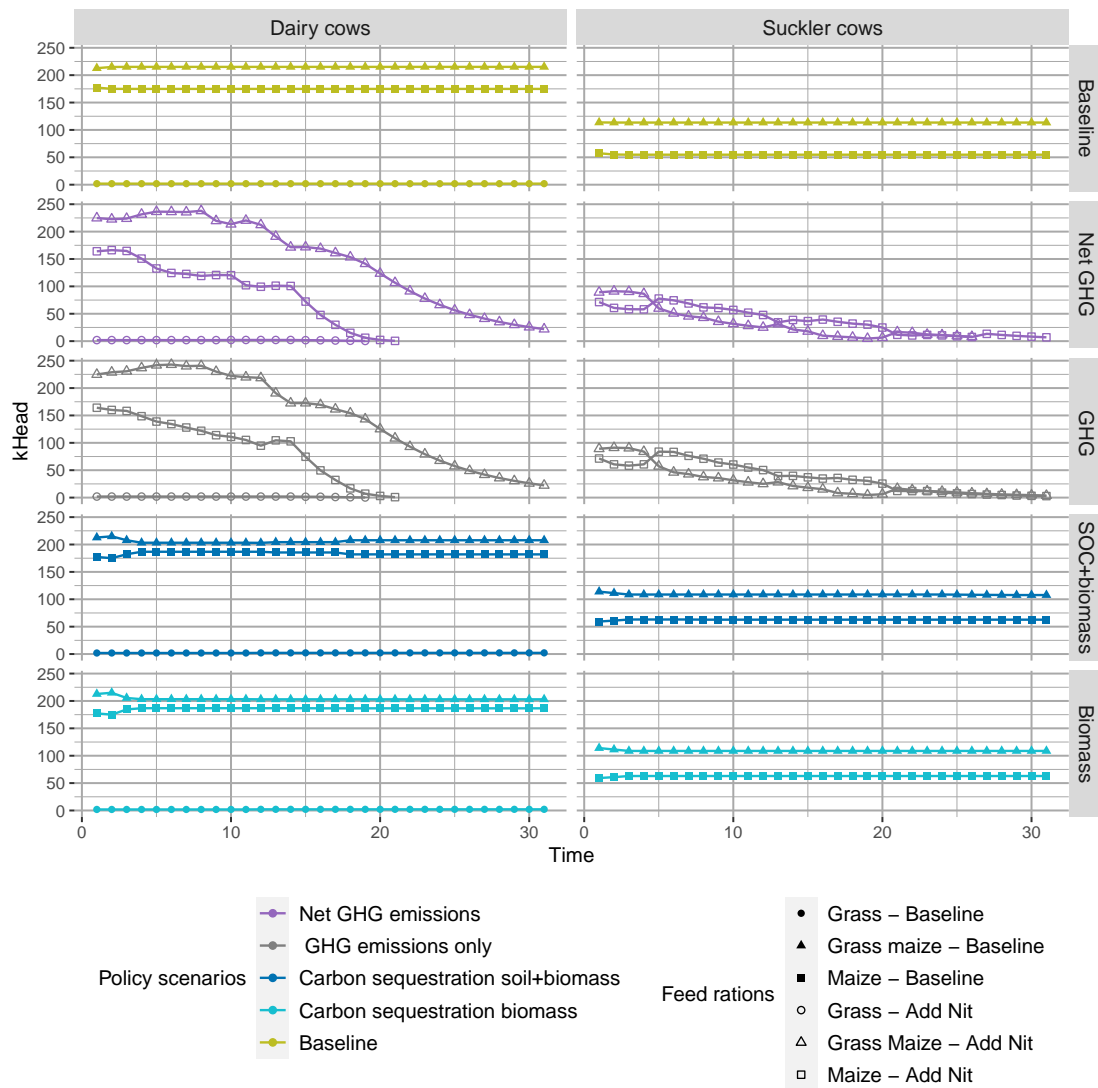


Figure 2.8: Herd size evolution over time for each mitigation policy and low carbon price scenario at the scale of the Grand Est. *Add Nit = practice addition of 1 % nitrate in the ration of ruminants.*

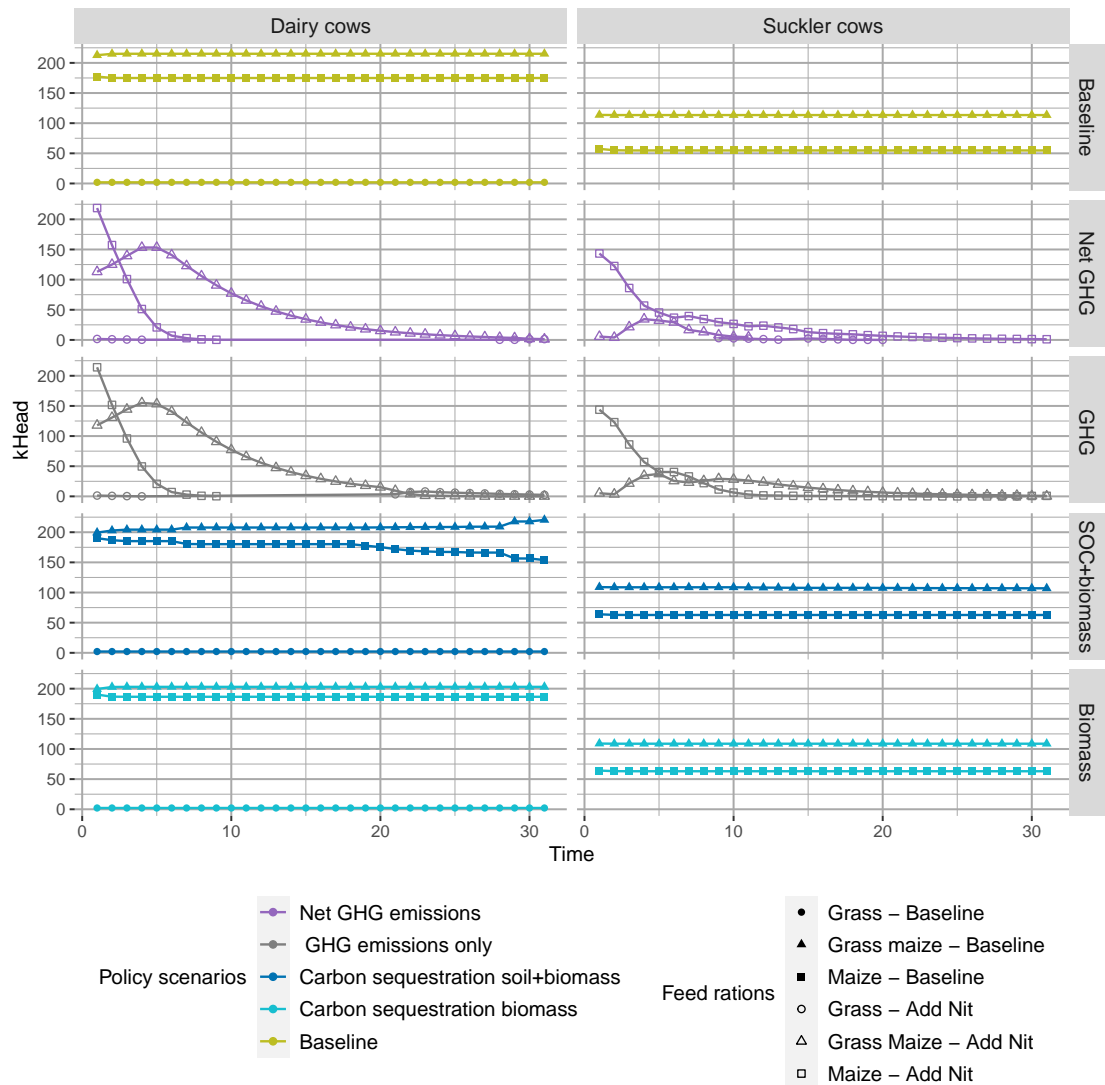


Figure 2.9: Herd size evolution over time for each mitigation policy and high carbon price scenario at the scale of the Grand Est. *Add Nit = practice addition of 1 % nitrate in the ration of ruminants.*

Evolution of the surface area of combinations crop - mitigation practice(s) over time for low and high carbon price trajectories

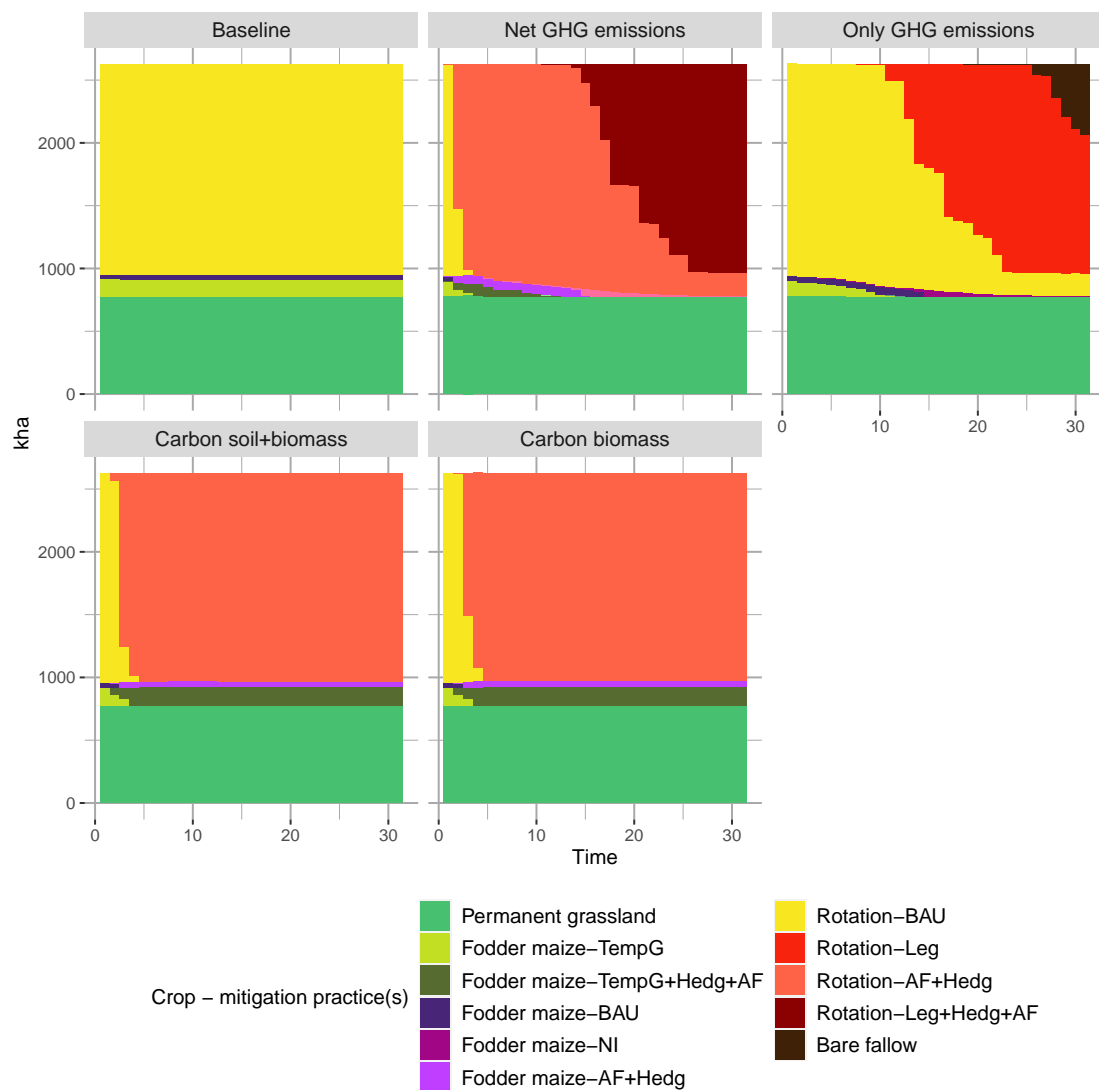


Figure 2.10: Evolution of the surface area of combinations crop - mitigation practice(s) over time for each mitigation policy and low carbon price scenario at the scale of the Grand Est

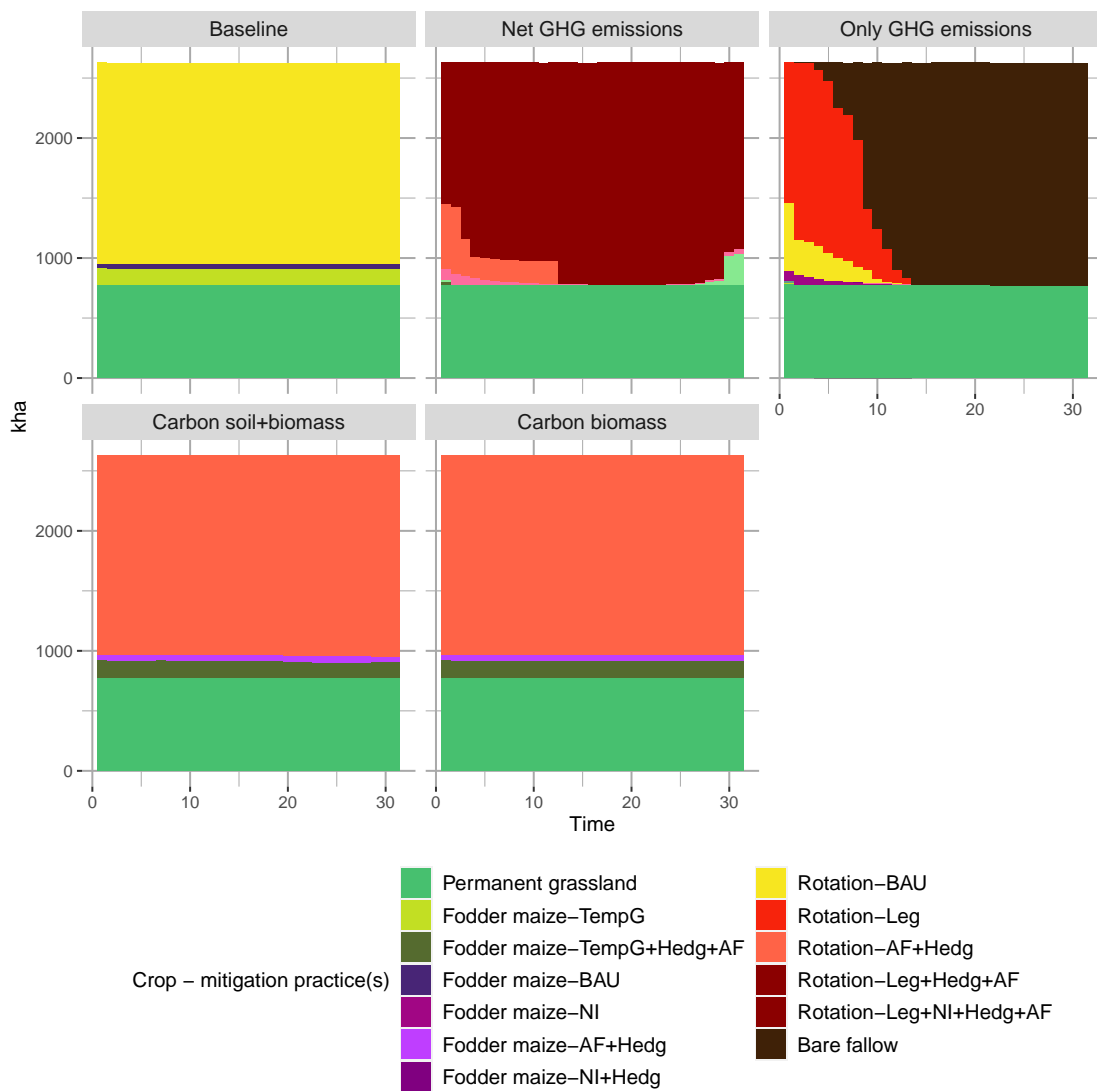


Figure 2.11: Evolution of the surface area of combinations crop - mitigation practice(s) over time for each mitigation policy and high carbon price scenario at the scale of the Grand Est

Chapter 3

Adoption of carbon farming schemes:

Risk matters

Abstract

Ensuring that carbon farming credits accurately reflect actual carbon sequestration is a major challenge. Although activity-based schemes are the most common design in existing voluntary carbon farming schemes, the economic literature has shown that result-based schemes are more cost-effective. However, this result overlooks the risk induced by result-based schemes when carbon sequestration is not known with certainty *ex ante*. We propose a stylized economic framework that integrates soil science data and models to investigate the effect of risk on the adoption of carbon farming scheme by farmers. This framework is illustrated based on a data set combining observations from a long-term experiment site (Ultuna, Sweden), multi-model ensemble simulations of soil organic carbon dynamics, and a wide range of economic assumptions. We investigate both analytically and quantitatively the conditions under which a farmer chooses to participate to a voluntary carbon farming scheme when the quantity of carbon eventually achieved is uncertain. We explore various carbon farming schemes designs (result-based, hybrid) that differ in terms of monitoring costs and accuracy, with contrasted risk implications for the farmer. We show that the incentives to participate to a result-based scheme may be decreasing for high carbon prices.

Keywords: *carbon farming, monitoring schemes, soil carbon sequestration, schemes design*

3.1 Introduction

Carbon removals will be required to keep total net greenhouse gas emissions in line with ambitious climate targets. This is particularly important in the EU, where achieving climate neutrality by 2050 will require offsetting the residual emissions that will inevitably remain even if the planned mitigation measures are fully implemented [ESABCC, 2023]. In this context, agricultural practices that have the potential to increase soil organic carbon (SOC) sequestration have received renewed attention in recent years [Radley et al., 2021].

One way to encourage farmers to adopt such practices is through the development of voluntary carbon farming schemes, which allow farmers to sell the amount of sequestered SOC as credits to other economic agents willing to offset their own emissions. The development of carbon farming schemes is an increasingly important component of the EU climate strategy for the agricultural sector [Paleari, 2024].

An important challenge with such credits is to ensure that they accurately reflect actual sequestration. The design of carbon farming schemes is key in this regard. The majority of carbon farming schemes currently in place are activity-based¹, in the sense that the issuance of credits is only conditioned by the adoption of practices deemed to increase SOC sequestration. The quantity of issued credits is often extrapolated based on empirical relationships derived from controlled experiments and/or process-based model simulations calibrated to average conditions.

Although relatively easy to implement and monitor, activity-based schemes raise two main issues. The first one relates to allocative efficiency. SOC sequestration results from biological processes that depend on conditions that are heterogeneous in time and space. Issuing a uniform per-hectare quantity of credits for a given practice may not incentivize

¹See <http://reports.crea.gov.it/powerbi/CarbonSchemesInventory.html>

a cost-effective allocation of the overall mitigation effort among heterogeneous farmers. The second issue relates to uncertainty. As the amount of carbon sequestration is not known with certainty *ex ante*, there is a probability that the practice actually fails to fully deliver the expected carbon sequestration, thus leading to undue carbon payments and undermining the credibility of the mechanism.

This issue is all the more important that, contrary to what prevails for mandatory instruments such as the EU Emissions Trading System, the incentives to reduce (or even account for) monitoring uncertainty in voluntary carbon markets are limited [Bellassen et al., 2015]. The lack of harmonization in this regard is likely to lead to a classic 'market-for-lemons' situation, where 'high-quality' credits (characterized by low uncertainty in the amount of sequestered carbon) are crowded out by 'low-quality' ones.

There is thus a strong need to design a credible and harmonized monitoring and certification framework for carbon farming schemes [Wilkki and Reeve, 2021]. This led policymakers to promote the development of result-based schemes, where the quantity of carbon credits is based directly on the actual amount of carbon sequestered [Radley et al., 2021]. However, implementing such schemes requires costly monitoring, reporting, and verification (MRV) methods such as direct measurements, remote sensing, satellite imagery, sensor networks, possibly combined with modeling [Bellassen et al., 2015, Smith et al., 2020]. Moreover, result-based schemes may involve financial risk for farmers, as the impacts on soil carbon take years to materialize and depend on factors beyond the farmer's control. This may discourage farmers to participate to the scheme [Radley et al., 2021].

What is the effect of the design of a carbon farming scheme on the decision of farmers to participate? We address this question taking into account not only MRV costs and uncertainty, but also the risk implications for farmers under various designs.

A large part of the related literature has focused on quantifying the cost-effectiveness gains permitted by result-based (per-ton of sequestered carbon payments) relative to activity-based instruments (uniform per-hectare payments). An illustration of these gains can be found for example in [Parks and Hardie, 1995], who study a policy designed to encourage carbon sequestration through the conversion of marginal agricultural land to forest. In a similar vein, [Pautsch et al., 2001] explore several instruments aimed to incentivize soil carbon sequestration by adopting conservation tillage practices in the Midwest (US). They find that per-ton subsidies reflecting the heterogeneity in the ability of soils to sequester carbon (depending on the locality and the crop type) dominate uniform per-hectare subsidies. [Antle et al., 2003] obtain similar results and find that, when considering the spatial variability of the carbon sequestration potential in Montana (US), per-hectare carbon sequestration contracts are five times more expensive than per-ton contracts. Although per-ton contracts are more costly to monitor, MRV costs remain lower than the efficiency loss associated with per-hectare payments. This literature, however, overlooks the risk implications for farmers. In addition, the monitoring methods, as well as their costs and level of precision, are considered exogenous.

The conclusion that the cost-effectiveness gains permitted by result-based schemes more than offset monitoring costs is supported by many works in this strand of literature. It depends however on the monitoring methods used. [Antle et al., 2003] consider a combination of soil samples analyses and model simulations. As suggested by many other authors [Mooney et al., 2002, Conant et al., 2011, Smith et al., 2020, Yogo et al., 2021], this helps containing monitoring costs, but also has implications on monitoring uncertainty. The use of multi-model ensemble simulations allows us to account for such an uncertainty.

Our study is also related to the literature in environmental economics that has investi-

gated the endogenous choice of monitoring [Stranlund and Dhanda, 1999, Millock et al., 2002, Millock et al., 2012]. [Millock et al., 2002], for example, consider a situation characterized by heterogeneous agents and asymmetric information (i.e., mitigation potential and costs are known to the agents, but not to the regulator). The authors study a mechanism whereby each agent is offered the choice between two types of environmental payments based either on the measured environmental outcome (which implies monitoring costs), or on a fixed amount (no monitoring required). Such a mechanism permits to reveal the agent's private information and determine endogenously the optimal degree of monitoring. Although we envisage a similar choice of payments, we consider a different form of uncertainty in the sense that farmers themselves do not know *ex ante* with certainty the quantity of carbon sequestered *ex post*. Our focus is thus on the risk implications for farmers rather than on asymmetric information.

To deal with uncertainty, [Kim and McCarl, 2009] propose a discount on the carbon payment. This approach can be rationalized by a Reliable Minimum Estimate (RME) approach, which ensures that the practice delivers at least the expected carbon sequestration with a given probability. It has been extensively used in the literature examining the implications of the REDD+ program (Reducing Emissions from Deforestation and Forest Degradation, [Grassi et al., 2008, Köhl et al., 2009, Plugge et al., 2013]). We build on [Kim and McCarl, 2009] to examine hybrid schemes as an alternative to both activity- and result-based schemes. This leaves some flexibility in setting the required level of precision attached to carbon credits.

To study how the design of a carbon farming scheme affects adoption, we propose a stylized framework that combines economic modeling of the farmer's behaviour and models and data from soil sciences. The model highlights the essential features of the trade-offs of interest. We leave aside the well-documented cost-effectiveness gains

permitted by result-based payment when farmers are heterogeneous to focus on the risk implications on the farmer's decision to participate to the scheme. This analytical framework is illustrated with data from a long-term experimental site in Ultuna, Sweden that provides longitudinal observations on SOC stocks under various carbon-enhancing farming practices [Kätterer et al., 2011]. These data are combined with multi-model ensemble simulations of the dynamics of soil carbon stock [Bruni et al., 2021], which allows us to estimate the *ex ante* expected value of additional SOC sequestration, as well as its variance. The farmer's optimal decision is examined under a wide range of economic assumptions regarding implementation and monitoring costs, risk-aversion, unit carbon payment, and scheme duration.

The findings offer new insights into the impact of the design of carbon farming schemes on adoption. In particular, they show that, even if (and especially if) the unit carbon payment is large, risk-averse farmers may not adopt result-based schemes. The intuition is that result-based schemes, besides requiring costly monitoring, induce a risk for the farmer, which is amplified by the carbon price. It may thus be the case that result-based schemes, although cost-effective, may be adopted only by a small number of farmers even if the carbon price appears to be attractive and monitoring is based on state-of-the-art MRV techniques. To some extent, hybrid schemes may mitigate this issue. However, for plausible values of the variability in SOC sequestration, imposing a high level of precision implies to set a large uncertainty discount, which may also discourage adoption.

The remainder of the paper is structured as follows. Section 3.2 introduces the analytical framework. The data used for the empirical application are described in Section 3.3. The results of the empirical application are presented in Section 3.4. Section 3.5 discusses the implications of our findings. Section 3.6 concludes.

3.2 Analytical framework

We first propose a stylized analytical framework to study the incentives to engage into a voluntary carbon farming scheme when the effective carbon sequestration is not known with certainty *ex ante*. The model is kept as simple as possible to highlight the main trade-offs of interest.

Consider a farmer who has the possibility to engage voluntarily into a carbon farming scheme. The scheme requires the adoption of a new practice, which is deemed to increase soil organic carbon (SOC) sequestration relative to a predefined baseline. Once adopted, the practice must be maintained for the next T years. Adopting the new practice involves additional costs and/or foregone revenue relative to maintaining the baseline practice for the same period of time. The flow of future net costs over T years is assumed to be known to the farmer at the time of the decision $t = 0$. The total value of these costs, expressed in present €/ha (discount factor: ρ), is positive and denoted C_T .

Assume for simplicity that the baseline is defined by an existing stock of SOC (in tCO₂e/ha), which is perfectly known at $t = 0$. In contrast, the amount of additional carbon that will be effectively sequestered after T years of maintaining the practice cannot be known with certainty at $t = 0$. One reason is that future SOC developments partly depend on stochastic drivers (e.g. temperature, rainfall, occurrence of droughts, etc.) that are beyond the farmer's control. Another reason may be the incomplete knowledge of the underlying biological processes. Therefore, only the distribution of the total additional amount of SOC sequestered over the full duration of the scheme (denoted by Y_T) is known at $t = 0$. Following [Kim and McCarl, 2009], we make the

simplifying assumption that Y_T is normally distributed:

$$Y_T \sim \mathcal{N}(\bar{y}_T, \sigma_T^2), \quad (3.1)$$

where $\bar{y}_T > 0$ and $\sigma_T^2 > 0$ denote the expected value and variance of additional SOC sequestration over the full duration of the contract, respectively. Note that the normality assumption in (3.1) does not rule out the possibility that the practice may in fact result in de-sequestration (negative realization of Y_T) after T years of maintaining the practice.

In (3.1), \bar{y}_T and σ_T are indexed by T to signal that they both depend on the number of years during which the practice is maintained. Carbon accumulation takes time to materialize, and the longer the practice is maintained, the larger the expected additional SOC. Thus, \bar{y}_T is increasing with respect to T (possibly at a declining rate to reflect saturation). Moreover, the further away the time horizon, the less precise the *ex ante* estimate of the additional sequestration to be expected between 0 and T , and therefore the larger σ_T .

In exchange of the participation to the scheme, the farmer receives a carbon payment. We consider two contrasted scheme designs in this regard, which differ in the way they address the uncertainty affecting SOC sequestration and its monitoring.

The first scheme is akin to a result-based scheme (*R*). The farmer receives a unit payment for each additional ton of carbon sequestered (τ , in €/tCO₂e). The total amount of additional carbon sequestration effectively achieved is measured at $t = T$, for example by means of a soil analysis. The measurement cost, denoted by m (in €/ha), is incurred by the farmer and known with certainty at $t = 0$. As the carbon measurements and the payment occur at $t = T$, τ and m are discounted to be expressed in present value.² The

²Other arrangements regarding the timing of the payment are possible. For example, a payment can be made upfront at $t = 0$ or in equal annual installments based on the expected value \bar{y}_T , the total

effective payment thus depends on the realization of Y_T . However, at the time of the farmer's decision ($t = 0$), the only knowledge about the discounted per-hectare net gain under this design ($G_T^R = \tau Y_T - m - C_T$) is that of its distribution, which follows directly from (3.1):

$$G_T^R \sim \mathcal{N}(\tau \bar{y}_T - C_T - m, \tau^2 \sigma_T^2) \quad (3.2)$$

The level of the unit payment τ impacts not only the expected net gain, but also its variance. Therefore, the larger τ , the larger the risk faced by the farmer in terms of net gain under a result-based scheme. This will play an important role in our results.

Under the second scheme design, the payment is based on the *ex ante* expected environmental outcome \bar{y}_T , rather than on its *ex post* realization. This makes this design closer to that of an activity-based scheme. This design does nevertheless account for the fact that the additional SOC sequestration is not known with certainty *ex ante* through an uncertainty discount d [Kim and McCarl, 2009]. d is unitless and such that $0 \leq d \leq 1$. Only a share $(1 - d)$ of the expected SOC sequestration is eligible for the carbon payment. Under such a hybrid scheme (H), the monitoring costs are assumed to be zero (the implementation of the practice is assumed to be observable at no cost and Y_T is common knowledge). The net gain is therefore known with certainty at $t = 0$:

$$G_T^H = \tau(1 - d)\bar{y}_T - C_T. \quad (3.3)$$

The limit case $d = 1$ corresponds to the absence of any carbon payment. The case $d = 0$ corresponds to a pure activity-based scheme in the sense that the carbon payment reflects the full expected additional sequestration, regardless of the probability that the

payment being adjusted downward on upwards at $t = T$ depending on the carbon sequestration effectively achieved. This does not affect the results as long as the payments are discounted accordingly.

practice may fall short of delivering the expected environmental outcome.

The Reliable Minimum Estimate (RME) approach is useful to rationalize the use of an uncertainty discount and relate it to the probability that a hybrid scheme may result in undue payments. The RME approach is a prudent and robust approach that consists of setting a probability α that the quantity of carbon eligible to the payment exceeds the quantity of carbon effectively sequestered. The lower α , the more demanding the scheme, and the higher the confidence that the practice will deliver at least the quantity of carbon covered by the payment. This approach is similar to the one recommended by the IPCC [Penman et al., 2003] when accounting for uncertainties in carbon stocks assessments. To some extent, it is also similar to what can be found in some existing carbon farming schemes (e.g., the French carbon certification framework, *Label Bas Carbone*³).

With our notations, the RME approach translates into:

$$P(Y_T \leq (1 - d)\bar{y}_T) = \alpha. \quad (3.4)$$

Using simple algebraic manipulations and the symmetry of the normal distribution, the uncertainty discount can be expressed as

$$d = \frac{\sigma_T}{\bar{y}_T} z_{1-\alpha}, \quad (3.5)$$

where $z_{1-\alpha}$ represents the $(1 - \alpha)$ quantile of the standard normal distribution $\mathcal{N}(0, 1)$.

Equation (3.5) highlights the close relationship between the uncertainty discount and the probability of undue payments. The more demanding the scheme, the lower α ,

³See for example https://www.legifrance.gouv.fr/jorf/article_jo/JORFARTI000037657978.

the higher $z_{1-\alpha}$, and, therefore, the higher the uncertainty discount. As an illustration, suppose that the scheme demands a 90%-probability that the sequestration covered by the payment be greater than that effectively achieved ($\alpha = 0.1$). In this case, the uncertainty discount should be $z_{0.9} \approx 1.282$ times larger than the coefficient of variation of the additional sequestration. Note that, for a given value of the coefficient of variation, some values of α are incompatible with the constraint that d must remain between 0 and 1. The minimum attainable value of α is obtained when $d = 1$. Because of the symmetry of the normal distribution, this value also corresponds to the probability that the practice leads to de-sequestration (see Eq. (3.4)).

In order to deal with both certain (as in Eq. (3.3)) and uncertain (as in Eq. (3.2)) payments, additional assumptions regarding the farmer's attitude toward risk are needed. Our analysis relies on a standard expected utility approach, whereby the farmer's preferences are thus represented by a monotone increasing von Neumann-Morgenstern utility function $U(\cdot)$. We assume that the farmer is risk-averse ($U(\cdot)$ is concave). The farmer prefers to adopt the result-based scheme R rather than not to adopt it if and only if

$$E[U(G_T^R)] \geq U(0), \quad (3.6)$$

where $E[\cdot]$ denotes the expectation operator and the net gain under the baseline practice is certain and normalized to 0.

To keep the analysis as simple as possible, we consider a constant absolute risk-aversion (CARA) specification $U(\cdot)$ (e.g., $U(G) = 1 - e^{-\phi G}$), which summarizes the farmer's attitude toward risk into one parameter $\phi > 0$, which is the constant risk aversion coefficient. Together with the assumption of a normal distribution (3.2), this considerably simplifies the expressions of certainty equivalents. Under these assumptions, the

inequality (3.6) is equivalent to

$$\tau \bar{y}_T - C_T - m - \frac{\phi}{2} \tau^2 \sigma_T^2 \geq 0, \quad (3.7)$$

where the left-hand side is the certainty equivalent of the net gain under a result-based scheme, that is the certain net gain that provides a utility level equal to $E[U(G_T^R)]$. The last term of this expression corresponds to the risk premium associated with the stochastic nature of the result-based payment.

The quadratic expression in (3.7) leads to a simple interpretation. For values of the unit payment τ close to zero, the certainty equivalent is negative and the farmer prefers to stay out of the result-based scheme because it is costly to implement the new practice and monitor its environmental outcome. As τ increases, the incentives to adopt the scheme first increase until they reach a maximum at $\tau = \hat{\tau} = \bar{y}_T / (\phi \sigma_T^2)$. For values of τ larger than $\hat{\tau}$, the incentives to engage into the result-based decrease as the risk premium increases more rapidly with respect to τ than the expected payment. It may thus be the case that inequality (3.7) is not verified for any positive value of τ . This occurs if and only if the maximum value of the certainty equivalent is negative, that is, \bar{y}_T such that $(\bar{y}_T / \sigma_T)^2 < 2\phi(C_T + m)$. Whenever this condition does not hold, there exists an interval of non-negative values of τ within which inequality (3.7) is verified. For the farmer to voluntarily engage into the result-based scheme, τ has to be large enough to secure a sufficiently large expected payment, but not too large to contain the risk premium.

We now turn to the condition under which the farmer prefers to engage into a hybrid scheme rather than not to participate. As the payment is known with certainty under

such a scheme, the condition simply reduces to (for any given $\tau > 0$):

$$(i) \tau(1 - d)\bar{y}_T - C_T \geq 0 \text{ or, equivalently, } (ii) d \leq 1 - \frac{C_T}{\tau\bar{y}_T}. \quad (3.8)$$

Holding everything else constant, the greater the uncertainty discount d , the smaller the incentives to engage into the hybrid scheme. The lower bound for d given in part (ii) of (3.8) ensures that the carbon payment covers the costs of implementing the practice.

Consider now that the farmer is offered the choice between a result-based scheme (R) and a hybrid scheme (H).⁴ For a given unit payment $\tau > 0$, the farmer prefers the result-based scheme if and only if the certainty equivalent of the net gain obtained under the scheme R is greater than that obtained under scheme H , that is:

$$(i) d\tau\bar{y}_T - m - \frac{\phi}{2}\tau^2\sigma_T^2 \geq 0 \text{ or, equivalently, } (ii) d \geq \frac{m + \frac{\phi}{2}\tau^2\sigma_T^2}{\tau\bar{y}_T}. \quad (3.9)$$

The right-hand side of part (ii) in (3.9) provides an upper bound for d , which is increasing with respect to monitoring costs (m), the absolute risk aversion coefficient (ϕ), and the variance of the additional SOC sequestration (σ_T^2). Note that a risk-averse farmer always prefers a pure activity-based scheme ($d = 0$) to a result-based scheme. The interpretation of the quadratic expression in part (i) of (3.9) is similar to that of inequality (3.7). For small values of τ , the inequality is not satisfied and the hybrid scheme is preferred to the result-based scheme. When $d > 0$, the left-hand side increases as τ increases until $\tau = \tilde{\tau} = d\bar{y}_T/(\phi\sigma_T^2)$, and then decreases as the risk premium is increasing with respect to the (square of the) unit payment. Note that, since $\tilde{\tau} = d\hat{\tau}$ with $0 < d \leq 1$, $\tilde{\tau}$ is positive

⁴This echoes some provisions of the carbon farming methodologies certified by the French *Label Bas Carbone*, which leave the choice to farmers to either use a default carbon accounting method (based on average SOC sequestration in average conditions) with a discount on the payment, or a higher-tier method based on measurements of the relevant characteristics of the farm in exchange of a full payment.

and smaller than $\hat{\tau}$. There exists an interval of non-negative values of τ within which the result-based scheme is preferred to the hybrid scheme if and only if $(d\bar{y}_T/\sigma_T)^2 \geq 2\phi m$. Using Eq. (3.5), this condition simplifies to

$$z_{1-\alpha}^2 \geq 2\phi m. \quad (3.10)$$

Inequality (3.10) provides a simple condition under which a result-based scheme may emerge as the preferred option for the farmer for all least some values of τ . If this condition is not satisfied, the farmer will always prefer either the hybrid scheme or not to participate at all, regardless of the value of τ . Interestingly, this condition does not depend on the absolute level of the expected sequestration, but on the level of requirement imposed by the hybrid scheme as reflected by $z_{1-\alpha}$. It underscores the trade-off between the level of requirement of the hybrid scheme, and the absolute risk aversion and monitoring cost involved by the result-based scheme.

The results are synthesized in Figure 3.1, which presents the farmer's preferred design as a function of the uncertainty discount for a fixed unit payment $\tau > 0$.⁵ There are three cases to consider. In the first case, the carbon payment does not cover the cost of implementing the practice even under a pure activity-based scheme ($d = 0$). Therefore, the farmer prefers to stay out of any carbon farming scheme, regardless of its design and the value of d in $[0, 1]$. For the two remaining cases, the implementation costs are covered by $\tau\bar{y}_T$.

The second case corresponds to a situation where the farmer prefers not to participate

⁵It would be possible to carry out a similar analysis of the farmer's preferred design as a function of the unit payment τ for a given value of the uncertainty discount d . However, the number of cases to consider would be much larger because of the quadratic nature of the inequalities in (3.7) and (3.9). In particular, this would require to combine the (possibly overlapping) ranges of values of τ where each of these inequalities are satisfied, and the conditions under which these ranges are non-empty. Rather than exploring all possible configurations in general, we keep such a discussion for the empirical application.

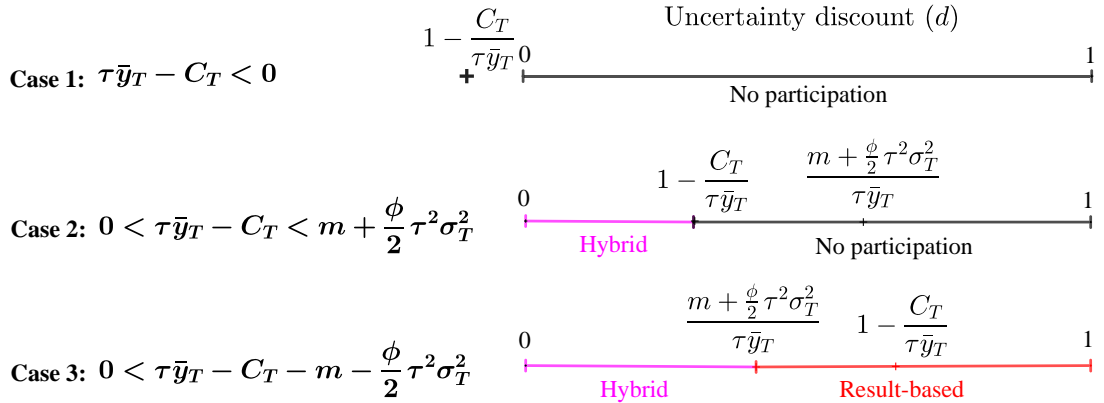


Figure 3.1: Preferred design as a function of the uncertainty discount d in $[0, 1]$ for a given value of the unit carbon payment $\tau > 0$.

rather than to engage into a result-based scheme (the inequality (3.7) is not satisfied). In this case, the farmer prefers the hybrid scheme as long as the uncertainty discount remains sufficiently low for the carbon payment to cover the costs (inequality (3.8)), and prefers not to participate otherwise.

In the third and last case, the result-based scheme is preferred to non-participation (inequality (3.7)). Yet, it is dominated by the hybrid scheme if the monitoring costs and the risk premium are high relative to the uncertainty discount (inequality (3.9)).

The duration of the carbon scheme is admittedly an important aspect of any carbon farming scheme. One reason is that SOC sequestration may be subject to the issue of non-permanence [Kim et al., 2008]. In the event that the carbon-enhancing practice is reversed after T years, some (or all) of the carbon accumulated may be released to the atmosphere. We thus examine how T affects the different components of the farmer's net gain under various design.

As the duration of the scheme increases, the total quantity of expected additional SOC sequestration increases, possibly at a declining rate as SOC stock approaches its

equilibrium level. The expected (nominal) carbon payment is therefore increasing with respect to T . Total implementation costs, as they are incurred on an annual basis, are also increasing with respect to T . These two effects impact the farmer's net gain under both hybrid and result-based schemes, although differently according to the value of the uncertainty discount d and the (time) discount rate ρ . Monitoring costs, as long as they are incurred only once at the end of the scheme, are decreasing with respect to T when expressed in present value due to (time) discounting. The larger the time horizon, the lower the precision of the prediction of future SOC sequestration, the larger σ_T , and therefore the larger the nominal risk premium under a result-based scheme. For a given of the probability of undue payment, this implies a larger uncertainty discount. As the chosen value of T impacts differently the net gains under hybrid and result-based schemes, it may also impact the farmer's preferred design.

The trade-offs described in this section depend on the characteristics of the SOC sequestration (expected value, variance) permitted by the carbon-enhancing practice, time horizon of the scheme, unit carbon payment, implementation and monitoring costs, and behavioral parameters determining risk-aversion. In the next two sections, we examine quantitatively the terms of these trade-offs.

3.3 Empirical application: Data

In this section, we describe the data used as input parameters for the application of the analytical model presented in Section 3.2.

3.3.1 Experimental data on SOC sequestration

The evolution and distribution of SOC stock under the baseline and the alternative practice are based on data from a long-term agricultural experiment and on simulations of a set of models.

The experimental site is located in Ultuna, Sweden [Kätterer et al., 2011]. The following crops are grown: oats, Swedish turnips, mustard, spring barley, fodder beet, winter wheat, and fodder rape. Ultuna's climate is cold temperate and sub-humid, with an annual mean temperature of 5.7°C and annual mean precipitation of 613 mm. The soil is classified as Eutric Cambisol [FAO, 1989] and is a clay loam with 36.5% clay, 41% silt, and 22.5% sand [Kätterer et al., 2011]. Other soil characteristics are shown in appendix 3.7.1.

Data from this site, which date back to 1956, have been chosen because of the long period of time they cover (over 30 years with the same practice) and their relatively high temporal frequency. Soil samples were collected and analyzed in 1956, 1967, 1974, 1975, and then biannually until 2008 (except in 1981 and 2003). For years without a measurement campaign, we linearly interpolated SOC stocks with the previous and following SOC measurements. This approach is valid in the time scale of a few years to a few decades. We consider two treatments: 1) a control treatment (unfertilized) taken as the baseline, and 2) an exogenous organic matter (EOM) treatment where sewage sludges were spread on the soil⁶. Figure 3.2 illustrates a progressive increase in the

⁶Note that applying sewage sludge does not actually lead to direct carbon sequestration. In fact, EOM contains carbon that was already produced and is simply redistributed somewhere else. Hence, it does not actively fix additional carbon from the atmosphere. However, this experiment allows us to simulate a situation where SOC stocks are increased after the implementation of an agricultural practice and it is a convenient approximation (especially in terms of data availability) of other SOC-improving practices, such as cover cropping, agroforestry systems, or improved crop rotations [Chenu et al., 2019, Bruni et al., 2022]. For simplicity, in the text we will refer to carbon sequestration associated to sewage sludge, but the reader should be aware of this difference.

observed carbon stock difference over time between the area where the sewage sludge practice is implemented and the control area. After 30 years, the gap between these two stocks reaches approximately $100 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$, representing the additional carbon storage resulting from the continued implementation of the sewage sludge practice for 30 years. Note that in the 21st year (year 1978) after the implementation of the sewage sludge practice, there is a spike in the observed carbon stock. This impacts the observed additional carbon sequestration for the sewage sludge practice (see figure 3.3).

3.3.2 Multi-model ensemble simulations

SOC stocks are simulated using a multi-model ensemble consisting of four SOC models: AMG [Andriulo et al., 1999], Century ([Parton, 1988], ICBM [Andrén and Kätterer, 1997] and RothC [Jenkinson et al., 1990] (see figure 3.2).

All models are based on conceptualized pools of carbon. The carbon from plant material and EOM enters the system and is decomposed following a first-order kinetics. AMG has one active compartment, where the carbon is decomposed at an annual rate, and one compartment that is considered stable throughout the simulation length. The decomposition of SOC in the active pool depends on pedoclimatic variables that are taken into account via environmental functions [Andriulo et al., 1999, Clivot et al., 2019]. ICBM has two pools of SOC that vary annually, a young and an old SOC pool. In this model, environmental factors are represented by a coefficient that affects the decomposition of SOC in both soil compartments and is estimated via soil humidity and temperature environmental functions [Andrén and Kätterer, 1997]. RothC has four dynamic pools (decomposable and resistant plant material, microbial biomass, and humified organic matter) and one inert SOC pool that is constant through time. Soil temperature, precip-

itation, and potential evapotranspiration affect the decomposition of SOC at a monthly time step [Jenkinson et al., 1990]. Century has 4 pools of litter C and 3 pools of SOC (an active, a slow and a passive pool that differentiate by their decay rates). In this model, soil temperature and soil relative humidity affect the decomposition of C in the different pools, at a daily time step [Parton, 1988].

Because of the different structures of the models, the initial value of SOC stock is estimated with different techniques for each of them. For AMG, the observed initial SOC stock is considered and separated in the active and stable compartment according to a fixed parameter (i.e. 0.65 for soils that were under arable use for a long period previous to the onset of the experiment). The other models consider that SOC stocks are at equilibrium before the beginning of the experiment. Hence, an equilibrium solution to the differential equations is calculated. For ICBM the equilibrium solution is solved analytically. For Century and RothC, the differential equations at equilibrium are solved semi-analytically [Xia et al., 2012, Huang et al., 2018, Bruni et al., 2021].

Daily soil surface temperature and moisture, precipitation, and potential evapotranspiration are simulated using the land-atmosphere model ORCHIDEE [Krinner et al., 2005]. Simulations are run using a 3-hourly global climate dataset at 0.5 °C (GSWP3).⁷

Each model is forced to begin at the initial measurement of the experimental data. The SOC stock evolution is simulated for each model separately. The median across model simulations is calculated for each year for sewage sludge practice.

Figure 3.2 shows that RothC and ICBM predict a carbon stock trajectory significantly deviating from the observed data for the area where the sewage sludge practice is implemented. The predictions of these two models "offset" each other, as the median of the multi-model ensemble is quite close to the observed data for sewage sludge practice.

⁷See <http://hydro.iis.u-tokyo.ac.jp/GSWP3/>

However, the level of uncertainty in carbon sequestration, estimated as the standard deviation of predictions from different models, is substantial, as indicated by the error intervals of carbon sequestration shown in Figure 3.3, since the predictions of ICBM and RothC pull the standard deviation upwards.

Figures 3.2 and 3.4 show that the model predictions, and therefore the multi-model median, are unable to capture the spike observed in the 21st year in the observed data for the area where sewage sludge is implemented.

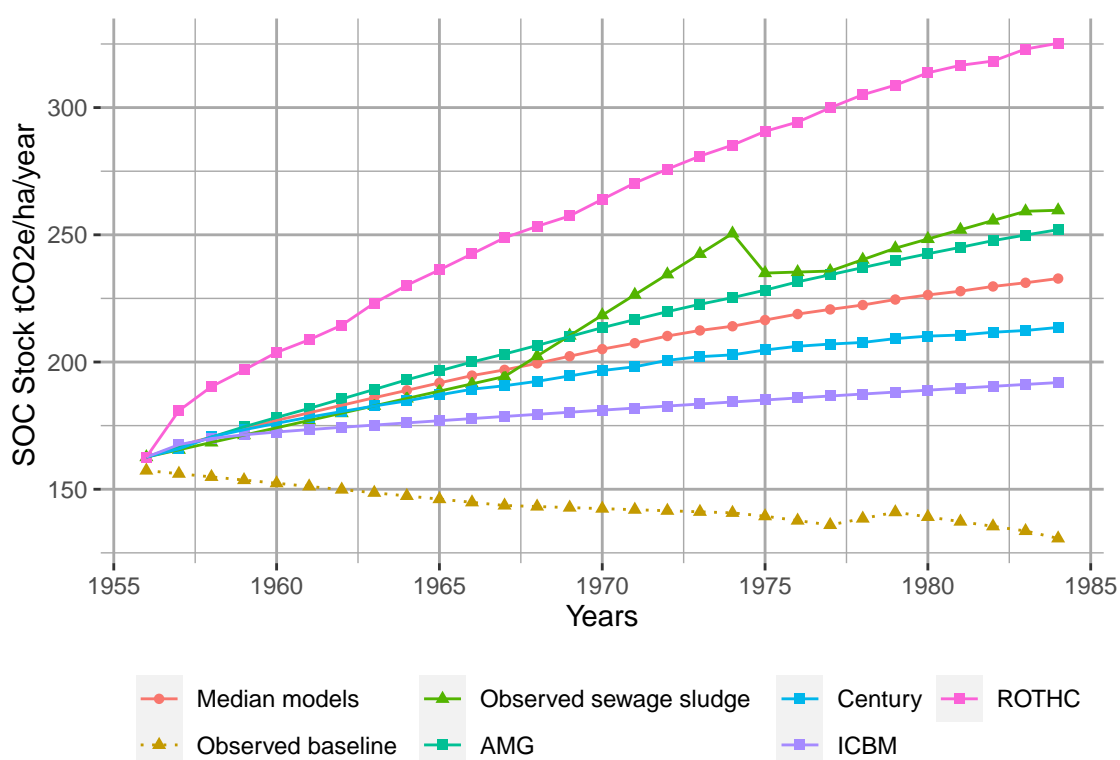


Figure 3.2: Changes in soil organic carbon stocks from the observed baseline, the observed sewage sludge practice, the simulations of each model for the sewage sludge practice, and the median of these simulations.

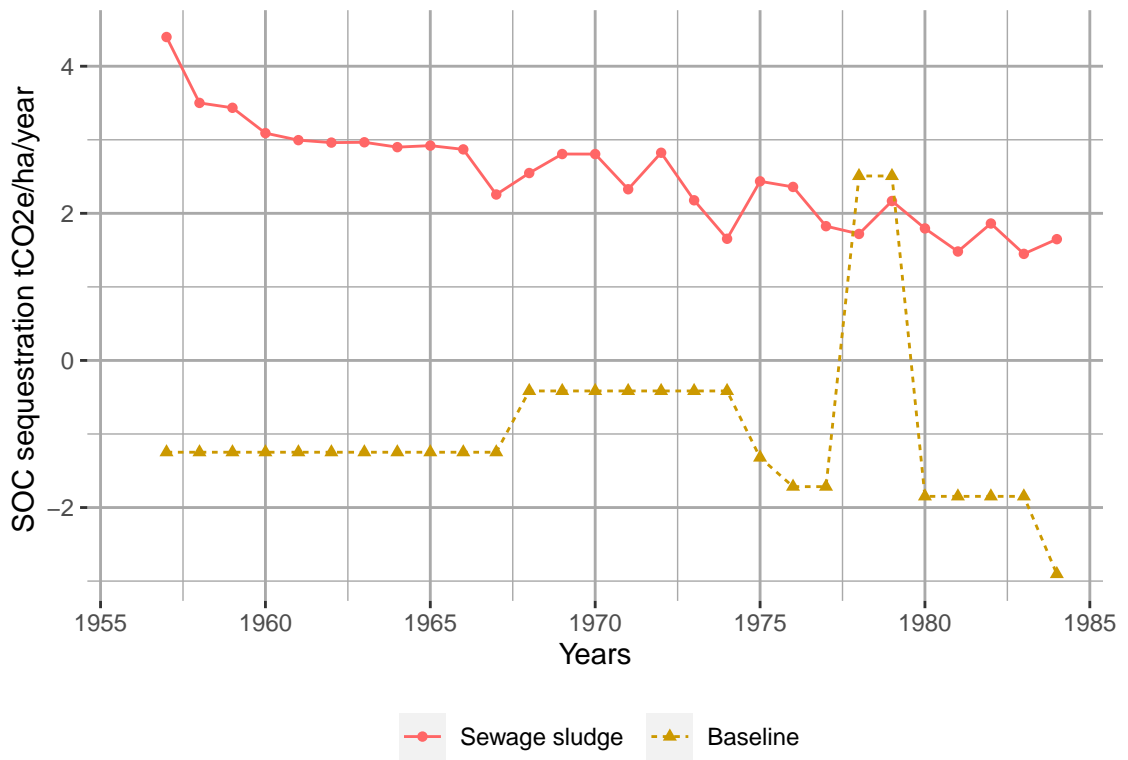


Figure 3.3: Evolution of the annual carbon sequestration for the median of models for sewage sludge spreading and for observed baseline (difference of the SOC stock between two years).

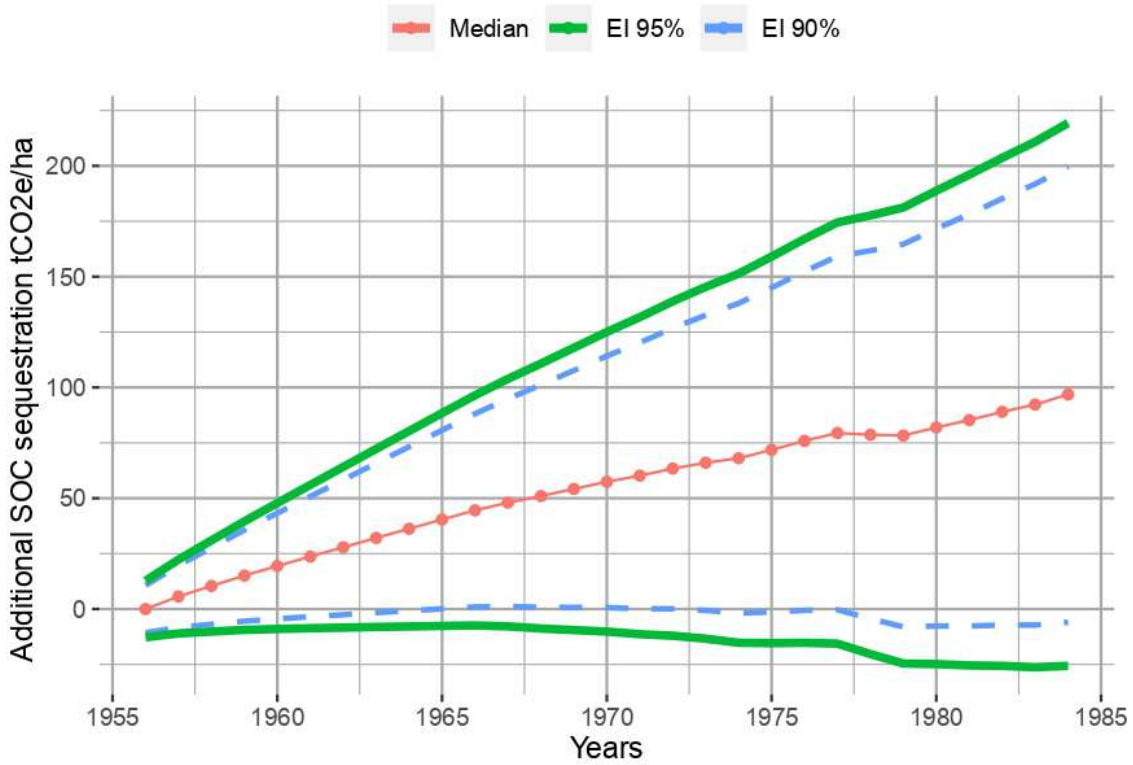


Figure 3.4: Median of additional annual cumulative carbon sequestration for sewage sludge spreading for each contract duration in red. This is the difference between the sequestration of the sewage sludge practice derived from the median of predictions of the model ensemble and the baseline for each contract duration. Error interval (EI) related to the model predictions: $EI_{1-\alpha,T} = \bar{y}_T \pm z_{1-\alpha} \cdot \sigma_T$. In green, $\alpha = 5\%$, in blue $\alpha = 10\%$

The value of σ_T is the standard deviation of the carbon stock predicted by the multi-model is calibrated to the average deviation between the predictions of the four models from the median. Figure 3.5 shows that it increases almost linearly with the time horizon considered for the prediction.

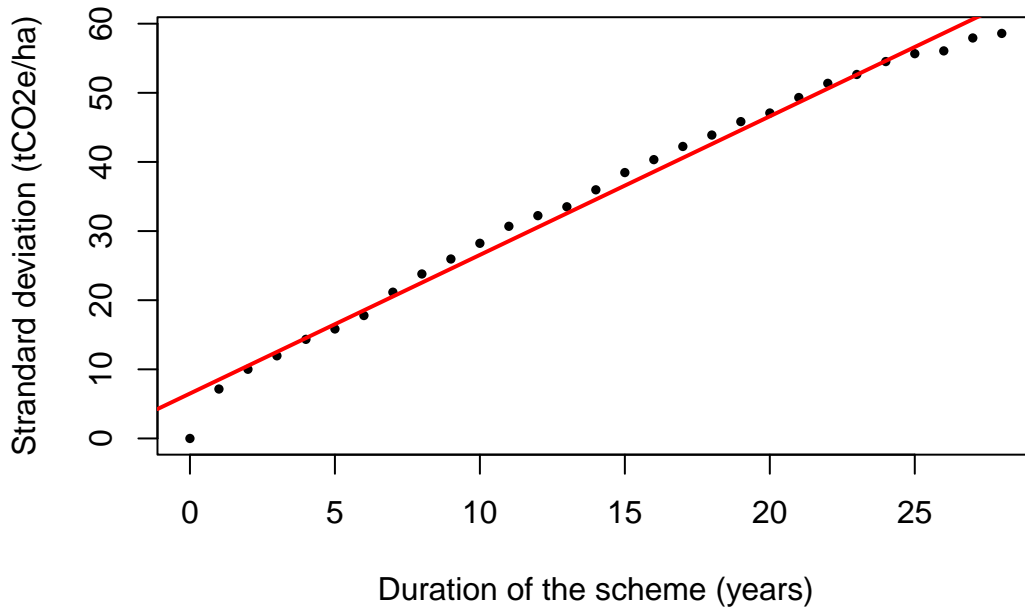


Figure 3.5: Standard deviation of the multi-model carbon stock prediction as a function of the chosen time horizon (sewage sludge spreading). *The red line corresponds to regression with respect to the time horizon (intercept: $5.15 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$, slope: $2.6 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$, $R^2 = 0.99$).*

3.3.3 Economic parameters

Because of the lack of economic data pertaining to the experimental site, we use estimates from [Bamière et al., 2023] that were obtained in the French context. Our central assumption regarding implementation costs is based on the average additional cost per hectare for the application of new organic resources. [Bamière et al., 2023] estimate these costs at $22.6 \text{ €}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$. This amount includes yield variations and mineral fertilizer savings relative to the baseline practice, as well as EOM transportation, and spreading costs (see the breakdown in Appendix 3.7.1). To account for

sources of variability and uncertainties, two other values for C_T are explored: 11.3 and 45.2 $\text{€}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$.

Result-based schemes require to measure SOC stocks. We assume that this monitoring involves direct measurements through analyses of soil samples. Available estimates of sampling and testing cost data vary greatly from one source to the other (literature, national sampling campaigns, or private laboratories, see appendix 3.7.1). We assume that soil bulk density, which is necessary to estimate SOC stock, is known at the time of the decision, and is assumed to be constant throughout the duration of the scheme. To parameterize monitoring costs, we retain information obtained from a French private laboratory, which charges 64 € per tested sample for sampling and carbon content dosage.

Monitoring costs and precision depend strongly on the number of samples tested [Arrouays et al., 2018]. We base our central assumption regarding the sampling rate (5 per hectare) on the minimum number of samples considered by [Arrouays et al., 2018] as necessary to account for variability. Admittedly, this is a conservative assumption that tends to favor result-based schemes. By comparison, the number of samples collected on the experimental site of Ultuna is 4 for each plot of 16 m^2 . Using the same sampling rate would have led to prohibitive monitoring costs. Our central assumption leads to monitoring costs amounting to 320 $\text{€}\cdot\text{ha}^{-1}$. In the empirical application, two alternative assumptions for monitoring costs of 160 and 640 $\text{€}\cdot\text{ha}^{-1}$ are also explored.

Parameterizing the attitude toward risk is challenging. In a recent systematic review of available estimates of European farmers' risk preferences, [Iyer et al., 2020] show that the absolute risk aversion coefficient (ARC) differs greatly from one study to another depending on the locality, estimation methods, and decision-making context. As indicated by [Raskin and Cochran, 1986] and [Antón et al., 2002], the variety of contexts

and assumptions (type of risk, prices, revenues, quantities) makes the comparison of ARC estimates difficult. To calibrate the ARC used in the model, we first eliminate the results reported in [Iyer et al., 2020] that correspond to risks very different from the one we consider (natural disaster, changing policy environment, organic farming versus conventional). Our 'high risk aversion' scenario is based on the maximum of the remaining estimates reported in [Iyer et al., 2020]. It corresponds to an ARC of $2.23 \text{ ha.k}\text{€}^{-1}$ estimated by [Vollenweider et al., 2011] in the context of voluntary implementation of agri-environmental measures. Two lower additional values of the ARC are examined: 0.5575 ('low risk aversion') and 1.115 ('medium risk aversion').

We consider (nominal) unit carbon payments ranging from 0 to 100 €/tCO_{2e}, which is in line with the range of variation of the carbon price on the EU Emission Trading Scheme to date. By comparison, the French 'climate-energy contribution' was equal to 55 €/tCO_{2e} in 2019, and most of carbon farming schemes report a price between 15 to 50 €/tCO_{2e}. The annual discount rate is 5%.

3.4 Empirical application: Results

The certainty equivalent of the discounted annual net gain under a 5-year result-based scheme is depicted in Figure 3.6 for values of the unit carbon payment up to 100 €/tCO_{2e}, three levels of the absolute risk aversion coefficient (ϕ), and nine combinations of implementation and monitoring costs. For unit carbon payments below 10 €/tCO_{2e}, the certainty equivalent is negative in all the configurations explored.

The quadratic nature of the certainty equivalent with respect to the unit carbon payment appears clearly in Figure 3.6. As discussed in Section 3.2, the higher the risk aversion, the lower the level of the unit payment at which the certainty equivalent

reaches a maximum. If risk aversion is high (right panel), the maximum is attained for a unit payment of approximately 55 €/tCO₂e. Above this value, the certainty equivalent decreases as the increase in the risk premium more than offsets the increase in the expected carbon payment. The higher the risk aversion, the narrower the range of unit payment values for which the result-based scheme is preferred to no participation. Under low risk aversion and our central assumptions regarding monitoring and implementation costs, this condition holds true for all values of the nominal unit carbon payment between 20 €/tCO₂e and (well above) 100 €/tCO₂e. By contrast, this range is 24-90 €/tCO₂e under high risk aversion. Note that, in the case of high risk-aversion and high monitoring costs, no participation is always preferred to the result-based scheme, regardless of the value of the unit carbon payment.

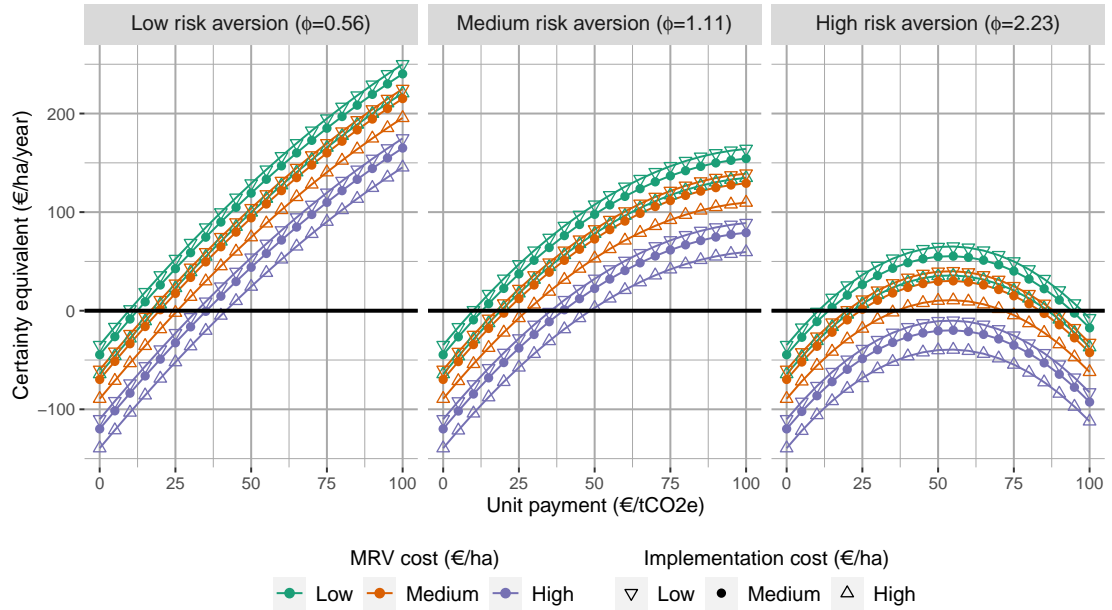


Figure 3.6: Result-based scheme: Certainty equivalent of the discounted annual net gain as a function of the nominal unit carbon payment for three levels of implementation cost, MRV costs, and risk aversion. *Parameter values: Expected additional sequestration over the whole duration of the scheme ($T = 5$ years): $\bar{y}_T = 23.7 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$ with a standard deviation of $\sigma_T = 15.84 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium) and 640 (high) $\text{€}\cdot\text{ha}^{-1}$. Implementation costs are based on nominal annual costs of 11.3 (low), 22.6 (medium), and 45.3 (high) $\text{€}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$. The annual discount rate is 5%.*

By construction, the net gain under a hybrid scheme is linear with respect to the unit carbon payment. The intercept of this relationship is determined by the implementation costs, and its slope depends negatively on the uncertainty payment. This relationship is illustrated for three values of the implementation costs and three values of the uncertainty discount d . Each value of d corresponds to a probability of undue payment α as per Equation (3.5). Given the coefficient of variation of additional SOC sequestration ($15.84/23.7 \approx 0.67$), a hybrid scheme that only covers half of the expected sequestration leads to a probability of undue payment of approximately 0.23. In such a case and even under high implementation costs, the hybrid scheme is preferred to no participation for

values of the unit carbon payment above 25 €/tCO₂e. Note that even if the uncertainty discount is very high ($d = 0.95$), the probability of undue payment remains relatively large (approximately 0.08). In that case, even if the unit carbon payment reaches 100 €/tCO₂e, the net gain from a hybrid scheme is at best very small, and negative under our central assumption regarding implementation costs.

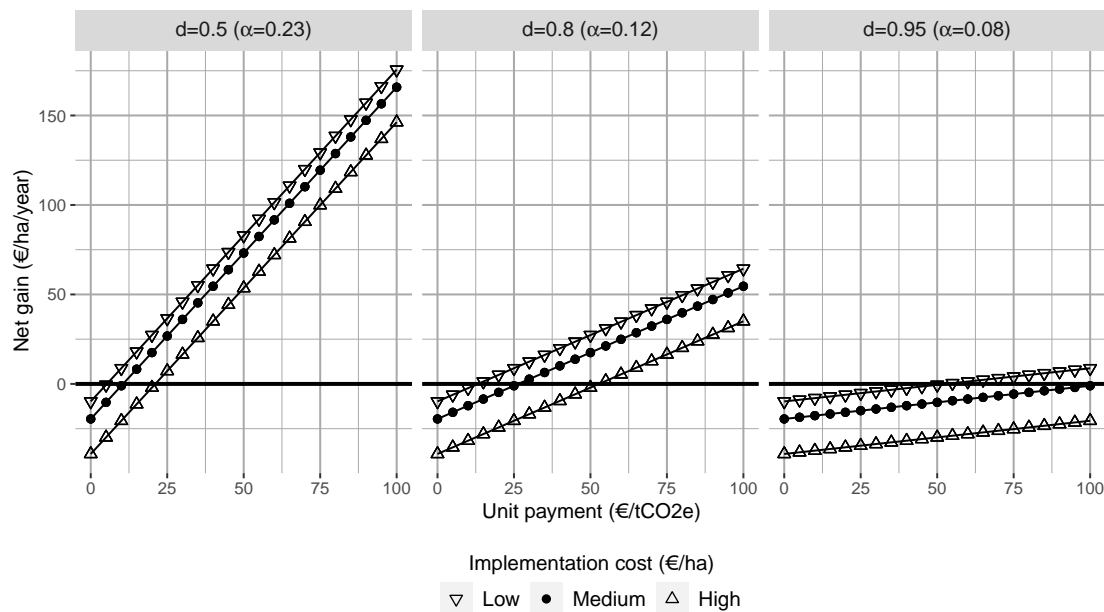


Figure 3.7: Hybrid scheme: Discounted annual net gain as a function of the nominal unit carbon payment for three levels of implementation cost and uncertainty discount. *Parameter values: Expected additional sequestration over the whole duration of the scheme ($T = 5$ years): $\bar{y}_T = 23.7 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$ with a standard deviation of $\sigma_T = 15.84 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$. Implementation costs are based on nominal annual costs of 11.3 (low), 22.6 (medium), and 45.3 (high) $\text{€}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$. The annual discount rate is 5%.*

The above results are synthesised in Figure 3.8. It depicts the combinations of unit carbon payment and uncertainty discount under which each design is preferred by the farmer. The results are presented for our central assumption in terms of implementation costs, and for nine combinations of risk-aversion and monitoring costs.

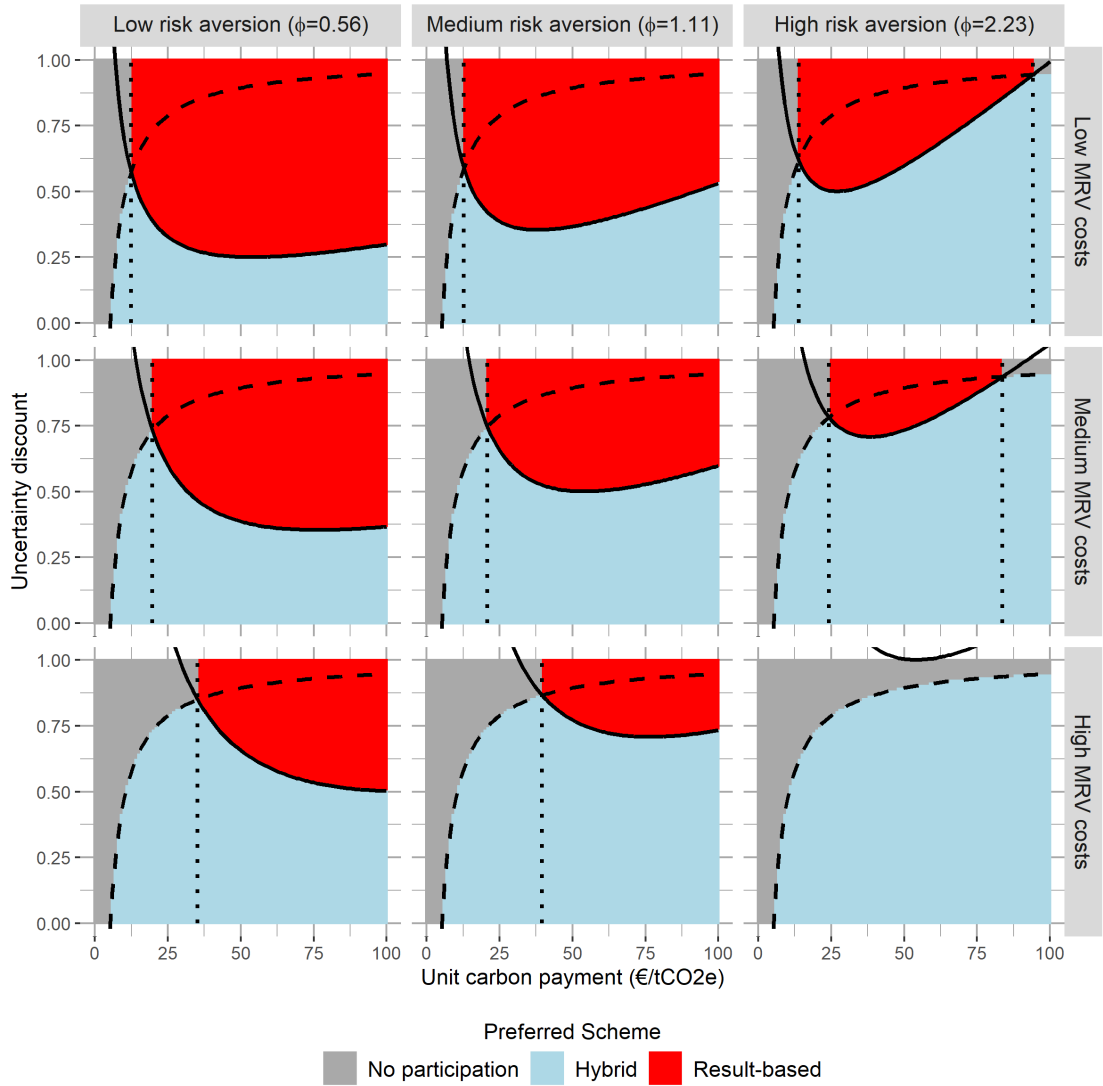


Figure 3.8: Preferred scheme as a function of the nominal unit carbon payment and the uncertainty discount for various degrees of risk aversion and different MRV costs. *Note: The curves represent the critical values of the nominal unit carbon payment based on inequalities (3.7) (dotted), (3.8) (dashed), and (3.9) (solid). Parameter values: Expected additional sequestration over the whole duration of the scheme ($T = 5$ years): $\bar{y}_T = 23.7 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$ with a standard deviation of $\sigma_T = 15.84 \text{ tCO}_2\text{e}\cdot\text{ha}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium) and 640 (high) $\text{€}\cdot\text{ha}^{-1}$. Implementation costs are based on nominal annual costs of $22.6 \text{ €}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ (medium). The annual discount rate is 5%.*

In all the configurations explored in Figure 3.8, for values of the unit carbon payment below 5 €/tCO₂e, the farmer prefers not to participate at all even if offered a pure activity-based scheme ($d = 0$). In this case, the carbon payment does not cover the implementation costs. For higher values of the unit carbon payment, the hybrid scheme may become profitable provided that the uncertainty discount is sufficiently small. For the result-based scheme to be preferred, the uncertainty discount must be sufficiently large and the value of carbon payment must lie within a range that secures a sufficient expected payment without involving a too large risk premium. The higher the risk aversion and/or the monitoring costs, the smaller the set of combinations of unit carbon payment and uncertainty discount under which the result-based scheme is preferred. Under our assumptions and when risk aversion and monitoring costs are high, the result-based never emerges as the preferred design regardless of the value of the unit carbon payment.

Figure 3.9 depicts the same results as in Figure 3.8, but as a function of the probability of undue payment (α). It underscores the importance of the level of requirement under a hybrid scheme. If too demanding (i.e., if α is too low), a hybrid scheme may not be adopted, and the farmer prefers either not to participate or the result-based scheme. Given the coefficient of variation of SOC sequestration ($15.84/23.7 \approx 0.67$), this is the case for all values of α below 0.066, which corresponds to the limit probability when the uncertainty discount approaches 100%. Note that it implies that the level of requirement recommended by the IPCC in carbon assessments (5%) is not attainable with a hybrid scheme in that situation.

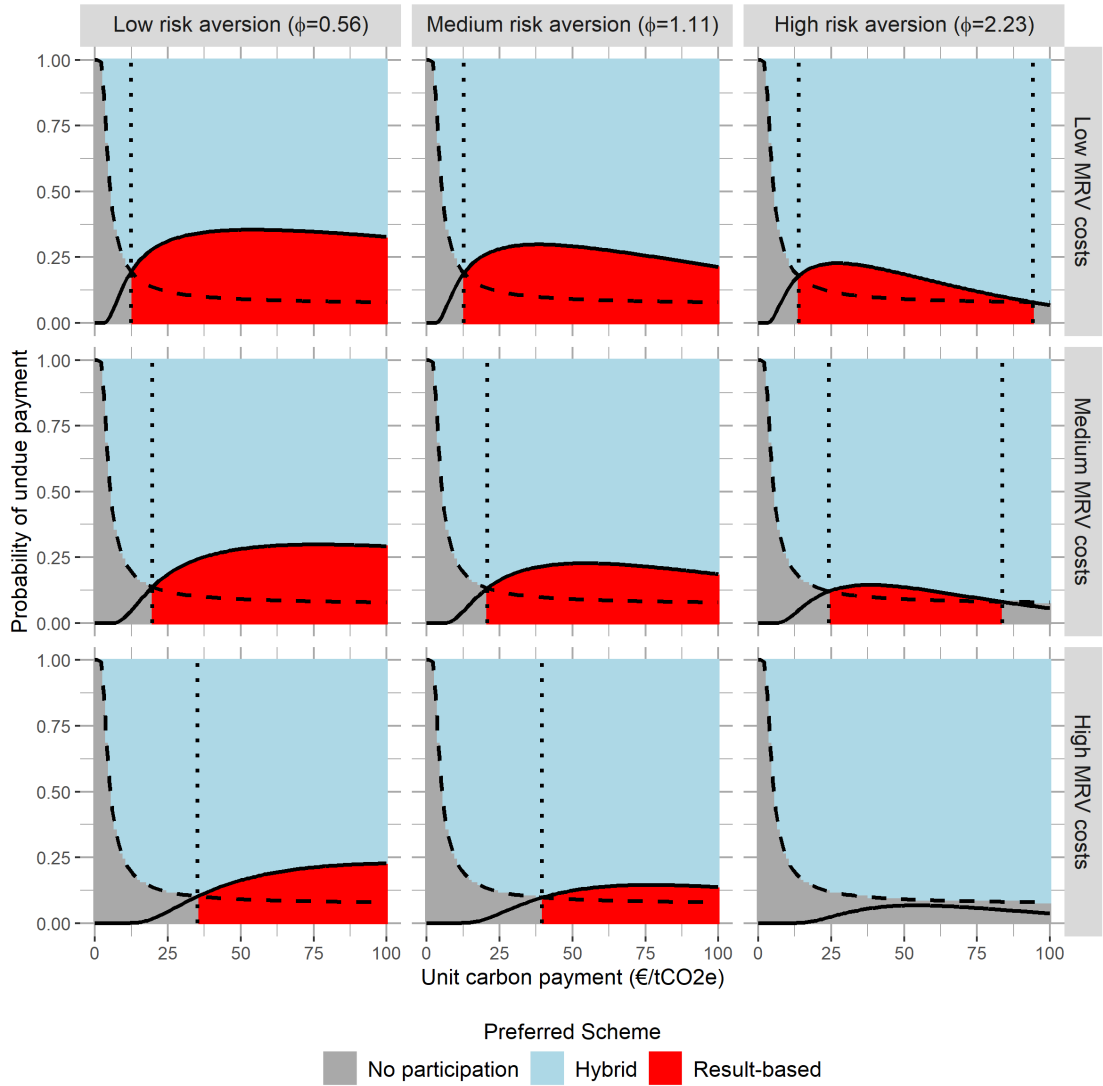


Figure 3.9: Preferred scheme as a function of the nominal unit carbon payment and the probability of undue payment for various degrees of risk aversion and different MRV costs. *Note: The curves represent the critical values of the nominal unit carbon payment based on inequalities (3.7) (dotted), (3.8) (dashed), and (3.9) (solid). Parameter values: Expected additional sequestration over the whole duration of the scheme ($T = 5$ years): $\bar{y}_T = 23.7 \text{ tCO}_2\text{e.ha}^{-1}$ with a standard deviation of $\sigma_T = 15.84 \text{ tCO}_2\text{e.ha}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium) and 640 (high) €/ha⁻¹. Implementation costs are based on nominal annual costs of 22.6 €/ha⁻¹.year⁻¹ (medium). The annual discount rate is 5%.*

Figure 3.10 depicts the certainty equivalent of the discounted annual net gain under a year result-based scheme as a function of the scheme duration for a carbon payment set at $50 \text{ €/tCO}_2\text{e}^{-1}$, three levels of the ARC (ϕ), and three levels of monitoring costs. As seen in Figure 3.6, a five-year result-based scheme yields a positive certainty equivalent under all combinations except if the ARC and monitoring cost are high. The incentives to participate to a result-based scheme are maximum for a scheme duration from 3 to 7 years depending on the assumptions on monitoring costs and risk aversion. For longer schemes, the certainty equivalent of the discounted annual net gain decreases with respect to the scheme duration due to the combined increase in the risk premium and total implementation costs.

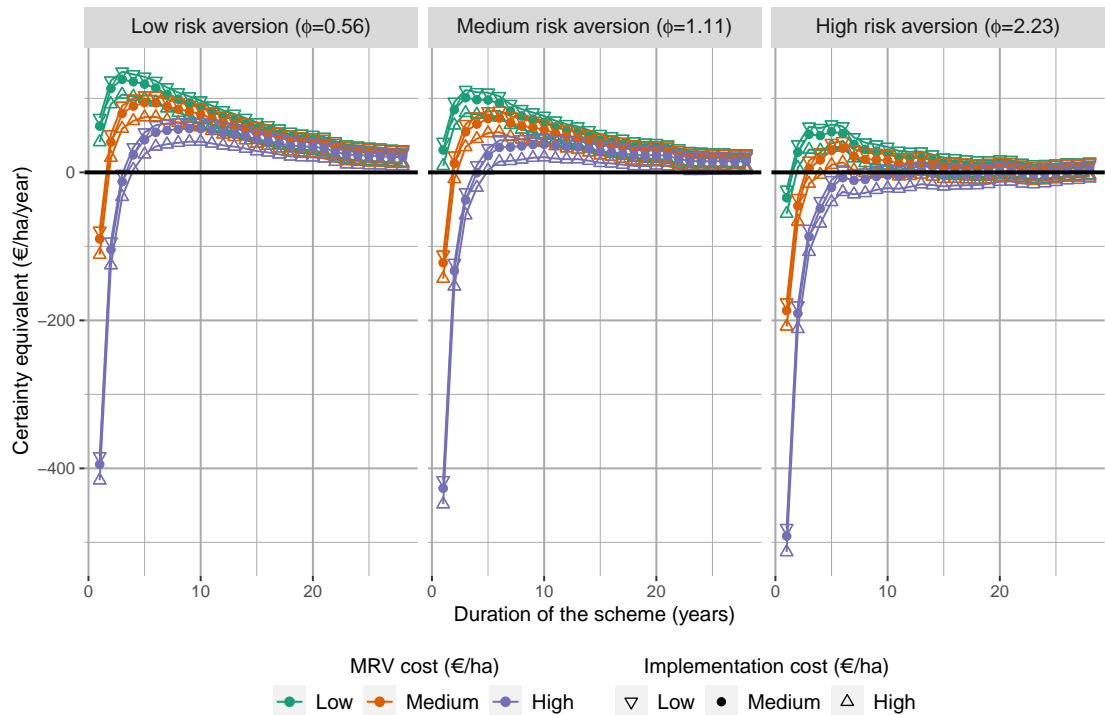


Figure 3.10: Result-based scheme: Certainty equivalent of the discounted annual net gain as a function of the duration of the scheme for three levels of implementation cost, MRV costs, and risk aversion. If the certainty equivalent is positive the farmer chooses to participate in a results-based scheme rather than not to participate. *Parameter values: Nominal unit carbon payment: $50 \text{ €} \cdot \text{tCO}_2\text{e}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium) and 640 (high) $\text{€} \cdot \text{ha}^{-1}$. Implementation costs are based on nominal annual costs of 11.3 (low), 22.6 (medium), and 45.3 (high) $\text{€} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. The annual discount rate is 5%.*

In Figure 3.11, we compare the net gains between a result-based scheme and a hybrid scheme as a function of the scheme duration for a nominal unit carbon payment level set at $50 \text{ €} \cdot \text{tCO}_2\text{e}^{-1}$, a probability of undue payments of 10%, three levels of ARC, and three levels of monitoring costs. The results indicate that, under these assumptions and despite decreasing incentives to participate to the scheme that tend to 0 as T increases, the preferred design remains the result-based scheme. This is partly due to the level of

requirement ($\alpha = 0.1$), which imposes a high uncertainty discount. We also test for higher carbon payments levels of 25 and 75 €/tCO₂e⁻¹ (see appendix figures 3.12 and 3.13). Note that the level of payment shows a higher influence on the shape of the certainty equivalent curve than the parameters of ARC, implementation costs, and monitoring costs.

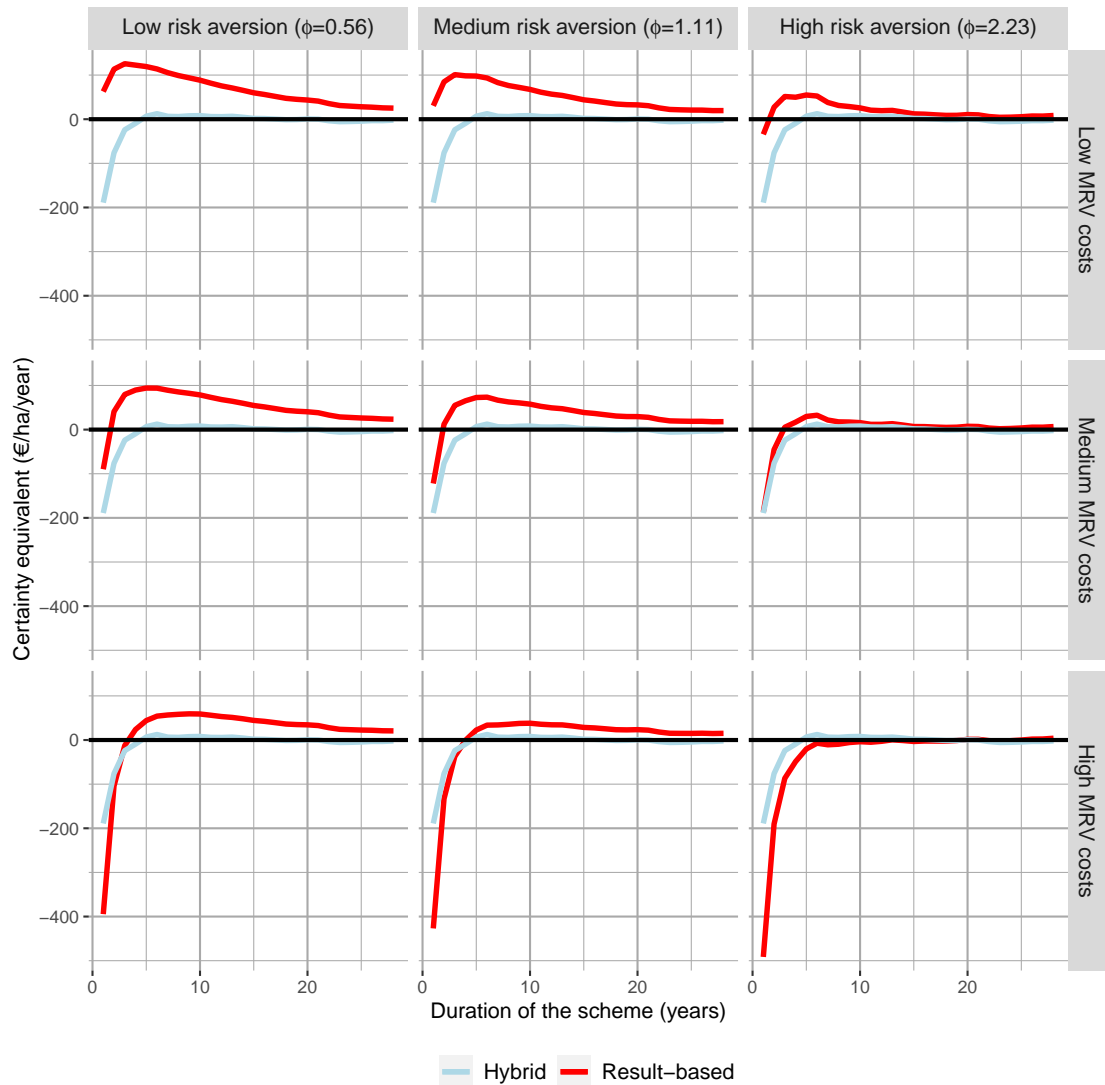


Figure 3.11: Farmer's net grain from Hybrid scheme and Certainty equivalent from Result-based scheme, as a function of several contract durations for three levels of risk aversion and three levels of MRV costs. *Parameter values: The level of carbon payment is $50 \text{ €} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium), and 640 (high) $\text{€} \cdot \text{ha}^{-1}$. Implementation cost is based on nominal annual cost of 22.6 (medium). The probability of undue payment α is based on nominal value of 10%. The annual discount rate is 5%.*

3.5 Discussion

The main mechanism at play stems from the fact that an increase in the unit carbon payment has two opposite effects on the incentives to engage into a result-based scheme. On the one hand, it increases the expected carbon payment. On the other hand, it magnifies the risk faced by the farmer in a more-than-proportional manner (quadratic under our assumptions). If the farmer is risk-averse—i.e., characterized by a concave utility function as is the case under our assumptions—, the latter effect dominates for large values of the unit carbon payment. Therefore, the result-based scheme can only be adopted if the unit payment is not too low (otherwise the implementation and monitoring costs would not be covered), but also not too large (otherwise the risk premium would discourage the farmer from contracting).

Resorting to hybrid schemes may help overcome this issue. In addition to being simpler to implement and monitor, a hybrid scheme could be adopted in configurations where a result-based scheme would not. By reducing monitoring costs and providing farmers with a less risky source of income, hybrid schemes may thus foster the overall adoption of carbon farming schemes. This however implies that, with a given probability, the carbon eventually sequestered may fall short of that covered by the payment.

To some extent, this probability can be controlled through the choice of the uncertainty discount. But, again this involves a trade-off between adoption and environmental effectiveness. The larger the discount, the more demanding the scheme, but also the lower the incentives to adopt the scheme. Given the uncertainty in SOC sequestration estimated with the multi-model ensemble simulations for a five-year carbon scheme, imposing a reliable minimum estimate of 90% of the expected SOC sequestration requires an uncertainty discount of approximately 86%. This contrasts with the results obtained

by [Kim et al., 2009] who find an uncertainty discount ranging from 15 to 20% for a similar level of requirement. This difference is due to a larger uncertainty affecting SOC sequestration in our empirical application, partly because the multi-model ensemble simulations integrate the uncertainty in the representation of the underlying mechanisms by various models.

Another important aspect of the design of a carbon farming scheme is its duration. The longer the farmer commits to adopt the carbon-enhancing practice, the larger the environmental benefit, and the less acute the issue of non-permanence, but also the lower the incentives to participate to a carbon farming scheme. Our empirical findings indicate, that, even with a unit carbon payment of 50 €/tCO₂e, incentives to participate to a carbon farming scheme decrease rapidly when the scheme duration exceeds 4 to 7 years.

Our results have been obtained with a simple model. We discuss its potential limitations and their implications on the results.

The only source of uncertainty considered in the model is that affecting additional SOC sequestration. Other components of the net gain associated with a scheme may also be uncertain such as baseline levels of SOC, net gain under the baseline practice, or implementation and monitoring costs of the carbon-enhancing practice. Considering also the uncertainty in these components would reinforce the importance of taking into account risk aversion.

The representation of the farmer's attitude toward risk could be considerably enriched by taking a more general specification of the farmer's attitude toward risk (expected utility with a non-constant absolute risk aversion, loss aversion based on prospect theory). One difficulty in this case to obtain credible values for the behavioral parameters that determine attitude toward risk.

In this work, we focus on the impact of risk-aversion. One aspect that has been overlooked is the fact that committing to a carbon farming scheme for a long period of time limits the farmer's flexibility in future economic decisions. Long-term carbon farming schemes thus imply a (negative) option value, that reflects the opportunity cost of not having the possibility to opt out of in the event that returning to the conventional practice becomes profitable (for instance because of a context of high agricultural prices). The longer the duration of the scheme, the larger this effect, and the lower the incentives to participate to a carbon farming scheme.

Another important aspect related to the duration of the scheme is the issue of non-permanence. In our simple model, we do not account for what happens to SOC stocks after the end of the scheme. Therefore, nothing prevents farmers to reverse their practice after the end of the scheme and release (at least some of) the carbon sequestered. Taking the non-permanence into account issue requires additional provisions, for example through of a non-permanence discount [Kim et al., 2008]. This would lower the incentives to participate to a carbon farming scheme.

3.6 Concluding remarks

In the context of carbon farming schemes deployment, there is a need to develop a cost-effective SOC monitoring policy. In this respect, result-based schemes have a clear advantage over activity-based schemes. However, as carbon farming schemes are voluntary instruments, their adoption by farmers will be crucial for such an instrument to have a substantial environmental impact. The level of carbon payment, the costs of implementing the carbon-enhancing practice, and the monitoring costs play an obvious role in the decision to adopt a carbon farming scheme. Our main contribution is to show

that the design of the scheme and its risk implications for farmers are also determinant.

In this work, we have proposed a simple framework to study the conditions under which a farmer chooses to participate to a voluntary carbon farming scheme. It combines elements from economics and soil science. We explicitly consider situations where the quantity of carbon eventually achieved is uncertain. We compare various scheme designs (result-based, hybrid) that differ in terms of monitoring costs and accuracy and have contrasted risk implications for the farmer. The empirical application combines data from a long-term experimental site (Ultuna, Sweden), a multi-model ensemble of simulations of soil organic carbon dynamics, and a wide range of economic assumptions.

Our findings underscore some important trade-offs between monitoring costs and accuracy and risk, which affect the incentives to participate to a carbon farming scheme. Our empirical results further show that carbon farming schemes may offer a profitable option for farmers. However, this result depends critically on the combination of unit carbon payment, expected value and uncertainty of SOC sequestration, scheme duration, risk aversion, monitoring costs, and implementation costs. Although our results have been obtained under a set of admittedly simplifying assumptions, they offer some general insights into the economic and environmental implications of carbon farming schemes' design.

In particular, we show that, under a plausible combination of assumptions, the incentives to participate to a result-based scheme may be decreasing with respect to the unit carbon payment as the risk premium increases more rapidly than the expected net gain. This has potentially important policy implications. Even if (and especially if) the unit carbon payment is large, result-based schemes may end up being adopted by only a low proportion of farmers, with therefore little environmental impact. This notably occurs when the risk aversion and/or monitoring costs are high.

The choice between result-based and hybrid schemes is, to a large extent, an issue of risk sharing between farmers and the society as a whole. Under a pure activity-based scheme (no uncertainty discount), the risk that the practice under-performs in terms of SOC sequestration is fully borne by the society. This risk is entirely transferred to the farmer under a result-based scheme. Hybrid schemes based on an uncertainty discount can be seen as a way to compensate the society for (some of) the risk that the practice does not fully deliver the expected environmental outcome. At the scale of the farm (the one examined in our framework), this comes inevitably with a probability that the carbon eventually sequestered falls short of that covered by the payment.

If the hybrid scheme is adopted by a sufficiently large population, the risks at the individual level are pooled together, thus mitigating the risk at the aggregate level. This does not rule out the possibility of undue payments for some farmers, but it may be compensated by an environmental over-performance for other farmers. Again, the overall effective environmental outcome depends critically on the uncertainty discount, which determines the 'quality' of the certified carbon credits, but also the overall adoption rate.

Leaving the choice between result-based and hybrid scheme may be interesting when designing the scheme, especially if the farmer's attitude toward risk is unknown. The minimum requirement that the carbon payment covers at least the costs of implementing the carbon-enhancing practice remains. If the monitoring costs are not too large, and for given values of the uncertainty discount and unit payment, farmers with low risk-aversion are likely to choose the result-based scheme, while more risk-averse farmers will turn to the hybrid scheme. Compared to the situation where one only one type of scheme is offered to farmers, this may foster both adoption and the overall quality of the resulting carbon credits by reducing the the gap between the carbon covered by the payment and that effectively sequestered.

3.7 Appendices

3.7.1 Appendix of the empirical application

Soil characteristics of the experimental site

Units		
Reference		[Kätterer et al., 2011]
Longitude	$^{\circ}E$	17.65
Latitude	$^{\circ}N$	59.82
Experiment duration		1956-still ongoing (1956-2008 period in our application)
Mean annual temperature	$^{\circ}C$	5.72
Mean annual precipitation	mm	613.35
Bulk density at t=0	$g.cm^{-3}$	1.4
Clay	%	36.5
Silt	%	41
Sand	%	22
pH		6.23
Calcium carbonate	$g_{CaCO_3}.kg^{-1}$ soil	0
(Average) C input from crop in the control treatment	$Mg\ C\ ha^{-1}\ yr^{-1}$	1.03
(Average) C input from crop in the sewage sludge treatment	$Mg\ C\ ha^{-1}\ yr^{-1}$	2.59
(Average) C input from EOM in the sewage sludge treatment	$Mg\ C\ ha^{-1}\ yr^{-1}$	1.84

Table 3.1: Pedoclimatic information about the experimental site in Ultuna (Sweden). Soil data is referred to the topsoil (0-20 cm) and to the beginning of the experiment

Cost of practices implementation

For more details on the calculation assumptions, see [Bamière et al., 2023, Bamière et al., 2021a, Bamière and Schiavo, 2022, Launay et al., 2021b].

Items	Cost (€. <i>ha</i> ⁻¹)
Variations in yields	-90.00
Mineral fertilization	-7.60
Purchase of NOR	73.00
Transportation	42.40
Spreading	4.80
Total cost to the farmer	22.60

Table 3.2: Variations of net costs. *Negative costs are gains.*

Soil carbon sampling costs

Source	Location	Total cost	Sampling	Carbon dosage	Bulk density (BD)	Other cost (legal costs, insurance etc..)
[Gruijter et al., 2018]	Australia	110 A\$/sample	100 A\$/sample	10 A\$/sample		
Australian Farm Institute ⁸	Australia	110 A\$/sample	30 A\$/sample + 1000 A\$/year	40 A\$/sample		(> 1500 A\$/year)
[Mooney et al., 2002]	Montana (US)	16 US\$/sample		10 US\$		
[Smith, 2016]	Oregon (US)	25 US\$/sample				
Auréa - private laboratory (from personal communication)	France	64 €/sample (180 with BD)	40€/sample (150€/sample with BD)	24€/sample	30€/sample	
RMQS ⁹ site (from personal communication)	France	7000 €/site (≈ 280 €/sample)	3000 €/site			

Table 3.3: Soil carbon sampling costs. Logistics costs (salaries, transports etc..) are taken into consideration in the cost estimate presented below.

Symbol	Definition	Unit	Value(s)
T	Duration of the scheme	Years	5 in figures 3.6, 3.7, 3.8, 3.9 and [1, 30] in figures 3.10, 3.11, 3.12, 3.13
ρ	Discount factor	Unitless	0.05
C_T	Technical costs of practice implementation over the full duration of the contract	$\text{€}.ha^{-1}$	Discount factor ρ and annual costs of {11.3, 22.6, 45.2} $\text{€}.(ha.yr)^{-1}$ in figures 3.6, 3.7, 3.10 and 22.6 $\text{€}.(ha.yr)^{-1}$ in figures 3.8, 3.9, 3.11, 3.12 and 3.13
\hat{y}_T	Expected value of SOC accrual over the full duration of the contract	$tCO_2e.ha^{-1}$	23.7 in figures 3.6, 3.7, 3.8, 3.9 and [5.65, 96.84] in figures 3.10, 3.11, 3.12, 3.13
σ_T	Standard error of SOC accrual over the full duration of the contract	$tCO_2e.ha^{-1}$	15.84 in figures 3.6, 3.7, 3.8, 3.9 and [7.21, 64.89] in figures 3.10, 3.11, 3.12, 3.13
τ	Unit payment for each additional ton of carbon sequestered	$\text{€}.tCO_2e^{-1}$	0 to 100 in figures 3.6, 3.7, 3.8, 3.9 and 50 in figures 3.10, 3.11, 3.12, 3.13
m	Measurement cost	$\text{€}.ha^{-1}$	Discount factor and {160, 320, 640} $\text{€}.ha^{-1}$
d	Uncertainty discount	Unitless	{0.5, 0.8, 0.95} in figure 3.7, [0, 1] in figure 3.9
α	Probability of undue payment	Unitless	{0.08, 0.12, 0.23} in figure 3.7, [0, 1] in figure 3.8 and 0.1 in figures 3.11, 3.12, 3.13
$z_{1-\alpha}$	$(1 - \alpha)$ quantile of the standard normal distribution	Unitless	{0.688, 1.175, 1.405} in figure 3.7, \approx [0, 2.48] in figure 3.8 and 1.2816 in figures 3.11, 3.12, 3.13
ϕ	Constant absolute risk-aversion	$ha.k\text{€}^{-1}$	{0.5575, 1.115, 2.23}

Table 3.4: Summary of parameters

Summary of parameters

3.7.2 Appendix of results

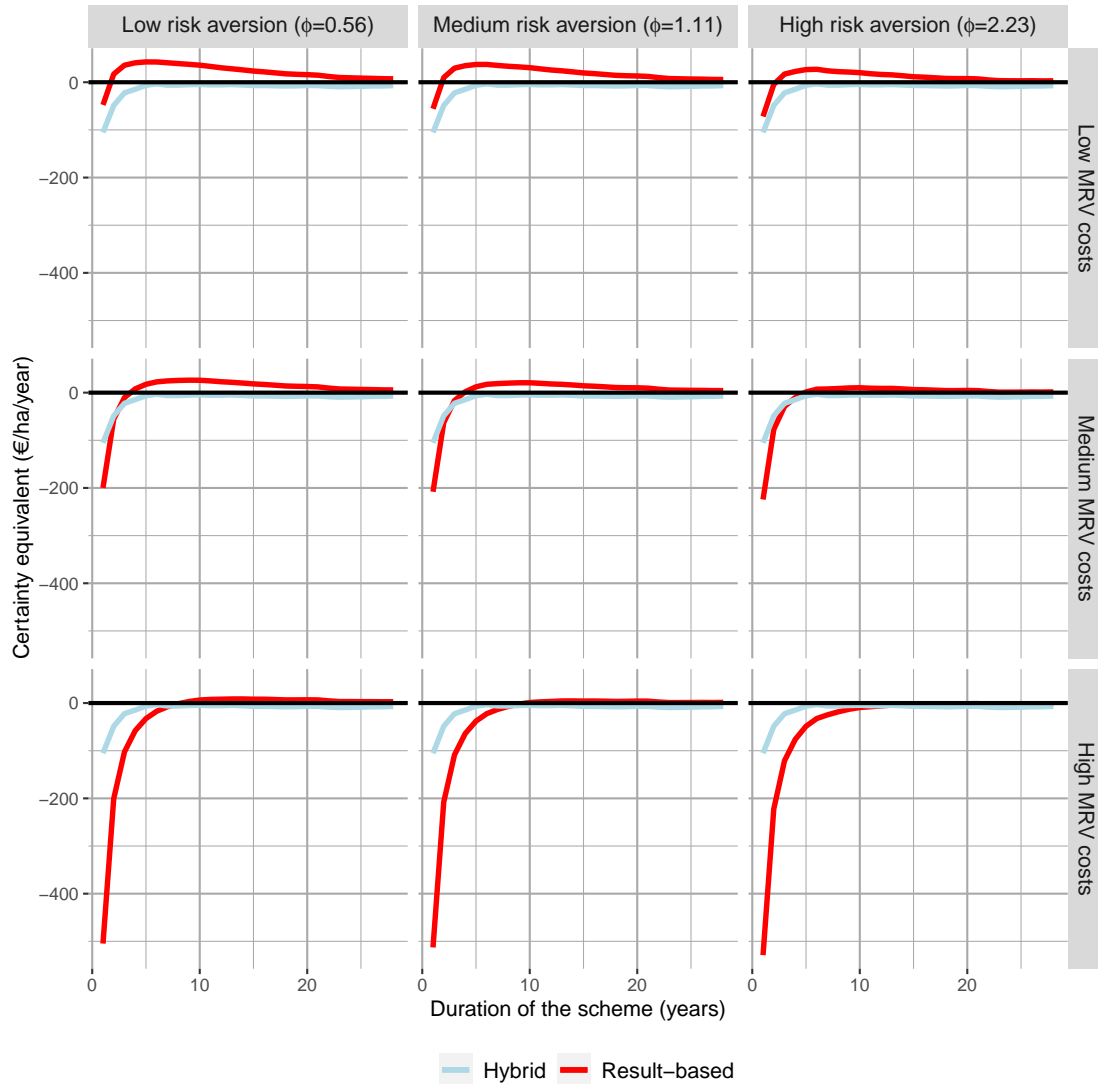


Figure 3.12: Farmer's net gain from Hybrid scheme and Certainty equivalent from Result-based scheme, as a function of several contract durations for three levels of risk aversion and three levels of MRV costs. *Parameter values: The level of carbon payment is $25 \text{ €} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium), and 640 (high) $\text{€} \cdot \text{ha}^{-1}$. Implementation cost is based on nominal annual cost of 22.6 (medium). The probability of undue payment α is based on nominal value of 10%. The annual discount rate is 5%.*

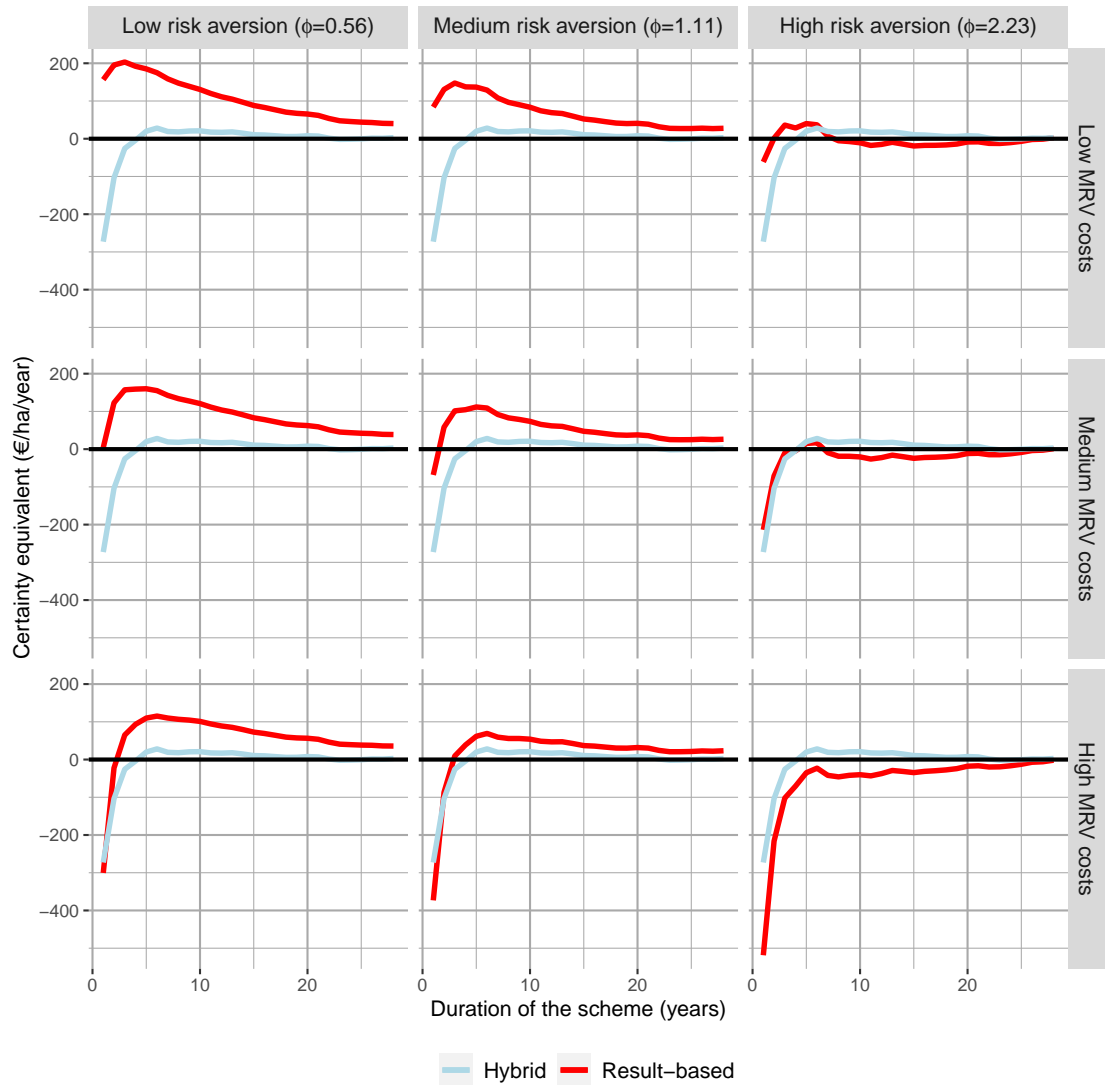


Figure 3.13: Farmer’s net grain from Hybrid scheme and Certainty equivalent from Result-based scheme, as a function of several contract durations for three levels of risk aversion and three levels of MRV costs. *Parameter values: The level of carbon payment is $75 \text{ €} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$. MRV costs are based on a nominal value of soil analysis of 160 (low), 320 (medium), and 640 (high) $\text{€} \cdot \text{ha}^{-1}$. Implementation cost is based on nominal annual cost of 22.6 (medium). The probability of undue payment α is based on nominal value of 10%. The annual discount rate is 5%.*

Chapter 4

Cost-effectiveness of carbon farming schemes taking farmers' risk aversion into account

Abstract

The European Union plans to establish a standardized certification framework for "carbon farming" to incentive farmers to maintain and increase carbon stocks. However, to ensure cost-effectiveness, careful consideration must be given to the design of carbon farming schemes. Existing literature suggests that results-based contracts are more cost-effective than practice-based contracts, but without considering farmers' risk aversion towards payment. This paper assesses the cost-effectiveness of various contract types (result-based, practice-based, and hybrid) while taking into account farmers' risk aversion in relation to the uncertainty of measuring carbon sequestration. I show that for relatively low values of carbon sequestration variability, the model-based, results-based, and practice-based contracts rewarding the implementation of intra-parcel agroforestry have nearly the same results in terms of marginal cost and carbon sequestration. For higher values of carbon sequestration variability, the participation costs increase for farmers in model-based and results-based contracts, thus the most cost-effective contract is the one remunerating the implementation of agroforestry. Since accurately estimating carbon sequestration variability is challenging, as it depends on numerous factors, and impacts farmers' participation in results-based and model-based contracts, to encourage the adoption of sequestration practices, it seems more relevant to implement payments per hectare for the implementation of agroforestry.

Keywords: *carbon farming, farmer's risk aversion, monitoring schemes, soil carbon sequestration, carbon contract design*

4.1 Introduction

Europe has set the ambitious goal of achieving carbon neutrality by 2050, by mitigating greenhouse gas emissions and offsetting residual emissions. This offset can be achieved by storing carbon in both the above-ground biomass and soils of the agricultural sector. To encourage farmers to maintain and increase carbon stocks level, the European Union plans to establish a harmonized certification framework for "carbon farming". For this strategy to be cost-effective, particular attention must be paid to the design of carbon farming schemes.

There are three types of carbon farming contracts: results-based, practice-based, and hybrid. In a practice-based contract ¹, the farmer is paid based on the implementation of an agricultural practice that has been identified as sequestering carbon. The actual amount of carbon sequestered is not known. Results-based ² contracts consist in paying the farmer according to the measured results. Entering into a results-based contract is therefore considered a risky decision for farmers, since they do not know the amount of the payment they will receive, but they still pay the costs of entering into the contract (opportunity cost and cost of implementing the carbon sequestration practice(s)). Finally, the hybrid approach can be defined as all the alternatives between practice-based and results-based schemes. It can be a model-based contract, i.e. a payment based on the predictions of a model calibrated on in-situ carbon stocks measurements. This is the definition I use from now on for hybrid contract. These three schemes differ regarding monitoring costs, uncertainty about the amount of carbon actually sequestered, and risk sharing between the farmer and the regulator, see figure 4.1.

¹Also called activity-based, per-hectare contract.

²Also known as per-ton contract, payment-by-results, performance-based, output-based, output-oriented in the literature.

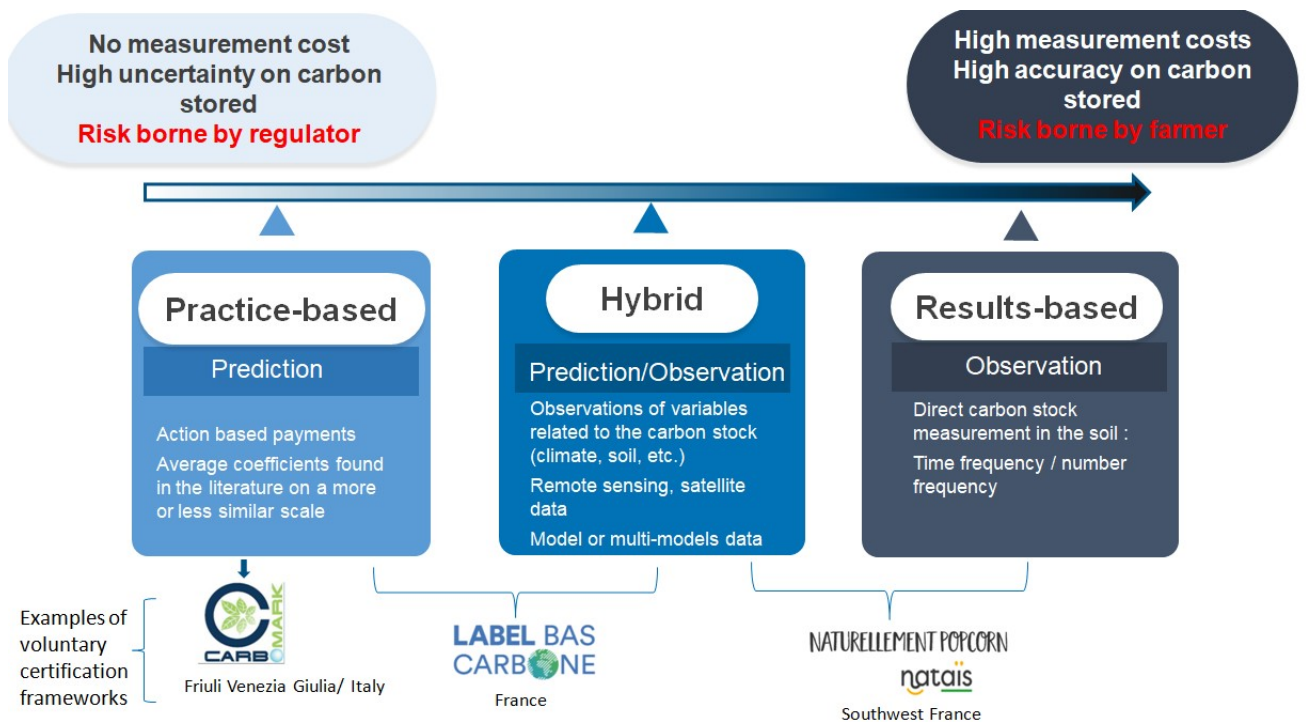


Figure 4.1: Continuity in the different forms of monitoring, the choice of monitoring consists of a trade-off between costs (for farmers and regulator), uncertainties about the carbon sequestered, and the risk for the regulator or farmer.

Some of the literature on incentives to adopt carbon sequestration practices studies and compares cost-effectiveness of these schemes. [Parks and Hardie, 1995, Pautsch et al., 2001, Bamière et al., 2021] show with applied theoretical multi-farm modeling that results-based contracts are more cost-effective than practiced-based contracts because of the spatial heterogeneity among farmers in terms of carbon sequestration potential. But these works do not consider the monitoring costs that can increase the cost of implementing results-based contracts. [Antle et al., 2003] take soil monitoring costs into account and show, in the context of Montana, that despite their weight, a results-based contract is five times less expensive than a practice-based contract for the same level of carbon seques-

tration. Nonetheless, the existing literature examining the cost-effectiveness of carbon farming schemes typically assumes that farmers are risk-neutral. Yet, a significant body of experimental studies suggests that farmers' risk aversion is a key factor in understanding potential barriers to participating in results-based schemes within the context of payment for ecosystem services (PES) [Matzdorf and Lorenz, 2010, Birge et al., 2017, Wezel et al., 2018, Šumrada et al., 2021]. In Chapter 3, we consider the risk aversion of a farmer related to the results of carbon sequestration for a result-based contract. We analytically explore the trade-offs between the adoption of results-based contract and model-based (hybrid) contract at the plot level. However, we only take into account a single farmer and therefore do not consider the heterogeneity among farmers, which is necessary to determine the most cost-effective scheme. Finally, this previous literature adopts a fixed time horizon perspective, without considering the potential impact of contract/subsidy duration on cost-effectiveness. This is a crucial issue, as the potential for carbon sequestration is limited. In the case of results or model-based contracts, if this limit is likely to be reached before the end of the defined time period, the farmer will have to bear the costs of implementing the practice without receiving any payment at the end of that period. This situation may lead to an increase in the costs associated with these contracts compared to practice-based contracts as the time horizon lengthens. It should be noted that carbon sequestration is not permanent, and the longer the time horizon of the contract, the more it pushes back the date at which the carbon is potentially released, leaving more time to consider permanent sequestration solutions.

The main objective is to determine which type of contract is the most cost-effective among three options: a practice-based contract, a model-based contract, and a results-based contract. This work contributes to the existing literature in four ways. Firstly, it takes account of the heterogeneity among farmers both in terms of the potential

and cost of carbon sequestration practices and of their risk aversion. Secondly, it considers the dynamic aspect of carbon sequestration as well as several time horizons. Thirdly, in addition to the practice-based and result-based contracts widely studied in the literature, a model-based contract is also studied, to see which is the most cost-effective of the three types. A model-based contract is not risky for the farmer and saves on monitoring costs. Finally, it includes monitoring costs associated to results-based and model-based contract. The goal is to explore whether taking farmers' risk aversion into account when implementing results-based contracts can affect participation costs to such an extent as to call into question the consensus established in the literature that these contracts are the most cost-effective. In addition, it is also interesting to examine whether extending the duration of contracts increases the costs (commitment costs for farmers and monitoring costs) of results-based and model-based contracts compared with practice-based contracts.

Since the uncertainty surrounding carbon sequestration results can arise from a number of sources - spatial heterogeneity, but also measurement errors, model prediction errors, variations in temperatures and other biophysical factors, etc. - it is difficult to have an accurate measure of the uncertainty to which a farmer exposes her/himself by committing to a results-based contract. Thus, first, as [Kim and McCarl, 2009], I consider that the uncertainty in carbon sequestration results from the spatial heterogeneity of carbon sequestration potentials among farms. Thus, I estimate the variability of carbon sequestration like [Kim and McCarl, 2009], using crop yields as a proxy. Then, as the data for carbon sequestration spatial variability values are fairly low, and spatial variability is only part of the carbon sequestration variability, I examine which contract is the most cost-effective for a more extreme carbon sequestration variability value. The aim is to see what effect a more extreme value of variability would have on the relative

marginal carbon sequestration costs of the different contracts for the same result in terms of carbon sequestration.

To do this, I develop a simplified analytical framework in which farmers make decisions regarding the area enrolled under the carbon farming scheme proposed to them. I assume that there are three scenarios, one for each type of scheme proposed to them : results-based, hybrid, and practice-based scheme. In the case of results-based and hybrid contracts, farmers also choose the carbon sequestration practice. In the hybrid contract, payments depend on the evolution of the carbon stock predicted by a model, and a discount for uncertainty in this prediction is applied to the payment. It is assumed that monitoring costs are borne by the regulator. Farmers' behavior towards risk is considered using expected utility theory.

To empirically apply this analytical framework, I use the static inter-temporal supply model developed in Chapter 2, applied to the Grand Est region, modeling the decisions of 40 representative farm groups. To assign parameter of risk aversion to these farm groups, I perform a Monte Carlo based on experimental data from [Rommel et al., 2023]. The carbon sequestration practices I consider involve sequestration in soils or in both soils and above-ground biomass.

The remainder of the paper is structured as follows. Section 4.2 defines our assumptions about the stylized analytical framework I develop and describes its structure. Section 4.3 presents the method and data used for the empirical application. Section 4.4 presents the results. Section 4.5 discusses our contribution and gives perspectives for further work.

4.2 Analytical framework

I consider 3 types of contracts for carbon sequestration: practice-based, model-based, and results-based contracts. Farmers' behaviors are modeled in an ex-ante manner. I assume that farmers make their commitment decisions based on their current knowledge of the entire time horizon. Farmers choose the area enrolled in the contract. While the mitigation practice is imposed by the regulator under the practice-based contract, farmers can individually choose the practice, but only one, under the other two types of contract. For the model-based contract, an uncertainty discount is applied to the payment as in Chapter 2. The results-based contract payment is assumed to be estimated based on measurement campaigns at regular time intervals on a precise scale and within a population of farmers committed to the contract. In the case of results-based contract, the farmer is exposed to the risk that the carbon sequestration measured does not correspond to that initially estimated. There is therefore uncertainty about the result of the measurement of carbon sequestration. I assume that this uncertainty is clearly indicated to the farmer before the contract is concluded. I assume that farmers are risk averse and model this using a constant relative risk aversion (CRRA) utility function, which is a commonly used hypothesis in the economics literature to assess farmers' risk aversion [Bougherara et al., 2011, Reynaud and Couture, 2012, Bocquého et al., 2014, Rommel et al., 2023]. I also assume that there is no moral hazard. Therefore, payment for a results-based contract is uncertain, whereas payment for a model-based and a practice-based contract is certain. The three contracts are not proposed simultaneously, i.e. farmers do not choose which type of contract to commit to, and therefore constitute 3 different scenarios.

Thus, for each type of contract, farmers maximize their individual profit by choosing

the area enrolled and the duration of the contract as a function of the carbon price, the cost of implementing the practices, the uncertainty of carbon sequestration potential of the practices in the sampled area, and their own parameters of risk aversion. Monitoring costs are assumed to be borne by the regulator, not by farmers. In this section, I present the farmer's utility function, then describe the objective function according to the type of contract. Finally, I describe the method used to estimate the monitoring costs associated with results-based and model-based contract.

4.2.1 Farmer's individual utility function

Each individual farm is denoted l . Each farmer has a baseline profit from her/his activities at each t , denoted $\Pi_{l,t}$. Farmers are free to take part in the contract, but if they commit, it is for a fixed duration denoted T . Each farmer chooses the area enrolled in the contract, denoted $A_{l,p}$. $uaa_l \geq A_{l,p} \geq 0$, where uaa_l is the total area of the farm l on which the carbon sequestration practice can be applied. The income at each t associated with the adoption of practice p on $A_{l,p}$ hectares is :

$$\Pi_{l,t} + A_{l,p} \cdot [g_{l,t}^p - c_{l,t}^p] \quad (4.1)$$

$c_{l,t}^p$ is the opportunity cost of committing to the contract. It includes the cost of giving up other agricultural activities that are not compatible with p on $A_{l,p}$ during the T periods and the costs of implementing practice p , i.e variable costs - variations in yields and production costs, fuel, nitrogen, etc - and also fixed costs - planting hedges and trees.

If a practice-based contract, $g_{l,t}^p = g$, where g is the payment for adopting a practice p set by the regulator, in $\text{€} \cdot tCO_2e^{-1}$.

If a model-based contract :

$$g_{l,t}^p = \tau \cdot \bar{s}_{r,t}^p \cdot [1 - d_T]$$

where τ is the carbon price, $\bar{s}_{r,t}^p$ is the average additional carbon sequestration³ estimated by the model for the geographic area r to which farm l belongs. d_T is the uncertainty discount based on the concept of reliable minimum estimate (RME), the expression of which is developed here in the same context of carbon farming as in Chapter 2 and in [Kim and McCarl, 2009] :

$$d_T = \frac{z_{1-\alpha} \cdot \sigma_r^p}{\bar{s}_{r,t}^p}$$

where $z_{1-\alpha}$ is the quantile of the normal distribution, $1 - \alpha$ is the confidence level of the sequestered carbon estimate, and σ_r^p is the standard error of the average additional carbon sequestration estimation in r . The greater the standard deviation, the more uncertain the result of measuring carbon sequestration. Note that σ_r^p depends on the size of the geographic area r at which the measurement campaign is carried out. This standard deviation is assumed to be time-independent, even though in reality the carbon sequestered could change under the influence of various factors which may be time-dependent (temperatures, accidents such as fires in one part of the region, etc.).

If a results-based contract :

$$g_{l,t}^p = \tau \cdot \Delta \hat{S}_{r,t}^p$$

where $\Delta \hat{S}_{r,t}^p$ is the random variable modeling the difference of the carbon stock mea-

3

surement between t and $t - 1$:

$$\Delta \hat{S}_{r,t}^p \sim \mathcal{L}(\bar{s}_{r,t}^p, \sigma_r^p)$$

I assume that farmers make their decision whether or not to commit based on the knowledge they now have over the entire time horizon, which is why the results of the measurement are modelled with a random variable.

I assume that the farmer's attitude towards risk is characterized by a Constant Relative Risk Aversion (CRRA) utility function (see [Samuelson, 1961]) :

$$U(\Pi_{l,t}) = \frac{(\Pi_{l,t})^{1-\gamma}}{1-\gamma} \quad (4.2)$$

where γ is the parameter of relative risk aversion, and I assume that $\gamma \neq 1$.

4.2.2 Maximizing the farmer's utility function

For all contracts, the farmer maximizes her/his profit by choosing the area $A_{l,p}$ on which (s)he implements the sequestration practice. For results-based and model-based contracts, the farmer also chooses the practice p (s)he implements. I assume that the farmers make their decisions independently and that there are no constraints binding them together⁴. I use a static inter-temporal approach. The objective function consists in maximizing the sum of farmers discounted expected utility over the time horizon T :

⁴They exchange nothing, and don't have a collective goal in terms of mitigation or production.

For practice-based :

$$\max_{A_{l,p}} \sum_l \sum_{t=1}^T \rho^t \cdot E \left[U(\Pi_{l,t} + A_{l,p} \cdot [g_{l,t}^p - c_{l,t}^p]) \right] \quad (4.3)$$

$$\text{s.t. } 0 \leq A_{l,p} \leq uaa_l \forall l \quad (4.4)$$

For model-based and results-based contracts :

$$\max_{A_{l,p}} \sum_l \sum_{t=1}^T \rho^t \cdot E \left[U(\Pi_{l,t} + \sum_p A_{l,p} \cdot [g_{l,t}^p - c_{l,t}^p]) \right] \quad (4.5)$$

$$\text{s.t. } 0 \leq A_{l,p} \leq uaa_l \cdot x_{l,p} \quad \forall l, p \quad (4.6)$$

$$\sum_p x_{l,p} = 1 \quad \forall l \quad (4.7)$$

$$x_{l,p} \in \{0, 1\} \quad \forall l, p \quad (4.8)$$

$$(4.9)$$

The binary variable $x_{l,p}$ takes the value of 1 to indicate the exclusive selection of practice p .

I discuss the optimal decision of the farmer at farm level, concerning $A_{l,p}$ for each type of contract.

If practice-based scheme:

$$\max_{A_{l,p}} \sum_{t=1}^T \rho^t \cdot E \left[U[\Pi_{l,t} + A_{l,p} \cdot (g - c_{l,t}^p)] \right] \quad (4.10)$$

$$\text{s.t. } 0 \leq A_{l,p} \leq uaa_l \quad \forall l \quad (4.11)$$

Since the payment is certain, the program of the farmer can be written :

$$\max_{A_{l,p}} \sum_{t=1}^T \rho^t \cdot \Pi_{l,t} + \sum_{t=1}^T \rho^t \cdot A_{l,p} \cdot (g - c_{l,t}^p)$$

According to the first-order conditions the solutions of $A_{l,p}$ is a corner equilibrium : if $\sum_{t=1}^T \rho^t \cdot g \geq \sum_{t=1}^T \rho^t \cdot c_{l,t}^p$ the farmer implements the practice on the entire possible area and $A_{l,p} = uaa_l$. If $\sum_{t=1}^T \rho^t \cdot g \leq \sum_{t=1}^T \rho^t \cdot c_{l,t}^p$, then $A_{l,p} = 0$.

If model-based scheme, since the payment is certain, it can be written :

$$\max_{A_{l,p}} \sum_{t=1}^T \rho^t \cdot \Pi_{l,t} + \sum_{t=1}^T \rho^t \cdot \sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p \cdot [1 - d_T] - c_{l,t}^p) \quad (4.12)$$

$$\text{s.c. } 0 \leq A_{l,p} \leq uaa_l \cdot x_{l,p} \quad \forall l, p \quad (4.13)$$

$$\sum_p x_{l,p} = 1 \quad \forall l \quad (4.14)$$

$$x_{l,p} \in \{0, 1\} \quad \forall l, p \quad (4.15)$$

According first-order conditions :

$$\sum_{t=1}^T \rho^t \cdot (\tau \cdot \bar{s}_{r,t}^p \cdot [1 - d_T]) = \sum_{t=1}^T \rho^t \cdot c_{l,t}^p \quad (4.16)$$

For a given p , the solution for $A_{l,p}$ is a corner equilibrium : the farmer chooses $A_{l,p} = uaa_l$ if the carbon farming contract payment per hectare is superior to the opportunity cost per hectare of committing to the contract, otherwise $A_{l,p} = 0$.

p being a discrete choice between a selection of practices that the farmer can implement, the practice p chosen is the one such that the difference between the contract payment and the total opportunity cost is maximum :

$$\arg \max_{A_{l,p}} \sum_{t=1}^T \rho^t \cdot \sum_p A_{l,p} \cdot \left[\left(\tau \cdot \bar{s}_{r,t}^p \cdot [1 - d_T] \right) - \sum_{t=1}^T \rho^t \cdot c_{l,t}^p \right]$$

If results-based scheme, using the certainty equivalent of the CRRA utility function (see the development of the formula in appendix 4.6.1), it can be written :

$$\max_{\{A_{l,p}\}} \sum_{t=1}^T \rho^t \cdot \frac{1}{1 - \gamma} \cdot \left[\Pi_{l,t} + A_{l,p} \cdot \left(\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p \right) - \frac{\gamma}{2} \cdot \frac{A_{l,p}^2 \cdot \tau^2 \cdot (\sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} \cdot \left(\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p \right)} \right]^{1-\gamma} \quad (4.17)$$

$$\text{s.c. } 0 \leq A_{l,p} \leq uaa_l \cdot x_{l,p} \quad \forall l, p \quad (4.18)$$

$$\sum_p x_{l,p} = 1 \quad \forall l \quad (4.19)$$

$$x_{l,p} \in \{0, 1\} \quad \forall l, p \quad (4.20)$$

If $\gamma = 0$, the choice of practice p is the same as in the case of the model-based contract. If $\gamma \neq 0$, the farmer chooses the practice p that shows the greatest difference between the marginal gain from the contract (payment minus opportunity cost) and the marginal cost linked to the risk:

$$\arg \max_{A_{l,p}} \sum_{t=1}^T \rho^t \cdot \sum_p A_{l,p} \left(\tau \bar{s}_{r,t}^p (1 - d_T) - c_{l,t}^p \right) - \sum_{t=1}^T \rho^t \left[\left(\frac{\gamma}{2} \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} \left(\tau \bar{s}_{r,t}^p (1 - d_T) - c_{l,t}^p \right)} \right) \right] \quad (4.21)$$

Calculations of first order conditions for $A_{l,p}$ are detailed in appendix 4.6.2. I show that with a results-based contract, if $\gamma = 0$ (i.e., the farmer is risk-neutral), the solution for $A_{l,p}$, for a given p , is the same as for the model-based contract.

If $\gamma \neq 0$, $A_{l,p}$, for a given p , is such that :

$$\sum_{t=1}^T \rho^t \cdot (\tau \bar{s}_{r,t}^p - c_{l,t}^p) = \sum_{t=1}^T \rho^t \left[\frac{\partial \left(\frac{\gamma}{2} \frac{(A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + A_{l,p} \cdot (\tau \bar{s}_{r,t}^p - c_{l,t}^p)} \right)}{\partial A_{l,p}} \right] \quad (4.22)$$

The optimal area in practice p , $A_{l,p}$, for each farm l , is such that the expected gain from the contract (payment minus opportunity cost) per unit of area equals the marginal cost of risk. More precisely, this implies that the optimal surface is reached when an increase in $A_{l,p}$ results in a change in the risk premium equal to the expected carbon farming contract gain per unit of area.

Thus, this theoretical framework highlights the fact that for model-based and practice-based contracts, the choice of the area enrolled in the contract is a corner equilibrium, where all the area is enrolled if the costs per hectare associated to the contract are lower than its benefits per hectare. Whereas when risk is taken into account, in a result-based contract, the choice of the area to be enrolled also depends on the level of uncertainty, modelled here by σ_r^p , and the farmer's aversion to risk γ . Thus, in this case, to achieve maximum profit, the farmer can "diversify" by enrolling only part of her/his area. In this way, (s)he obtains a part of the surface area through which (s)he does not obtain any payment from carbon farming, but (s)he does not take any risk, and another part, through which (s)he takes risks but can obtain additional income. This is in line with the portfolio theory developed by [Markowitz, 1991].

4.2.3 Measurement costs

For the sake of simplification, I assume that the monitoring costs are borne by the regulator. Alternatively, it is also possible to charge these monitoring expenses to a

private project owner. Thus, this monitoring cost is not part of the objective function, but depends on the results of farmers' decisions ⁵.

The results-based and model-based contracts require initial measurements of carbon stocks. The results-based contract requires also measurements during the contract at regular time intervals. I consider that the other transaction costs of implementing the contracts (verification costs, administrative costs, negotiation costs, etc.) are the same for all three types of contracts, which is why I do not consider them here. I follow [Antle et al., 2003] and assume that the sampling design is supposed to be a stratified sampling approach. For the number of samples, I take the formula expressed in [Mooney et al., 2002]. The number of samples n_r^p to be carry out from the population of farms adopting practice p in sub-region r is such that :

$$n_r^p = \frac{z_{1-\alpha}^2 \cdot N_r^p \cdot V_r^p}{N_r^p \cdot \epsilon^2 + z_{1-\alpha}^2 \cdot V_r^p} \quad (4.23)$$

where $z_{1-\alpha}$ is the quantile of the normal standardized distribution for the confidence level desired $1 - \alpha$, N_r^p total number of hectares of p in the sub-region r , V_r^p is the variance of the estimation of carbon sequestration ⁶ and ϵ is defined as the acceptable margin of error for the estimation of carbon sequestration. The smaller it is, the closer the estimate is to the "true value", but the greater the number of samples required to achieve this precision. The number of farms adopting the same practice influences the number of measurements required and, consequently, the monitoring expenses. The

⁵For each payment level and contract type, monitoring costs are added to farmers' opportunity costs for implementing the contract, derived from simulations. This total cost to society is then divided by the carbon sequestration (or net abatement) achieved, to obtain the marginal cost of carbon sequestration (mitigation) for each contract type and payment level.

⁶It is not the square of standard deviation of the mean as $\sigma_r^p \cdot \frac{\sigma_r^p}{\sqrt{n}} = \sqrt{V_r^p}$, where n is the sample size from which the annual mean of carbon sequestration over the time horizon is calculated.

cost of soil analysis is assumed to be the same whatever the soil sequestration practice. Thus, for model-based and result-based contracts, the total cost of monitoring for a given type of payment is the discounted product of the total number of samples by the cost per sample. In the case of the model-based contract, the measurement campaign only takes place once at the beginning of the time horizon, and in the case of the result-based contract, it takes place at regular fixed intervals during the contract. Note that the total number of samples depends on the carbon price, τ , as the number of hectares enrolled in a contract is contingent on this price and therefore varies with it.

4.3 Application to the Grand Est region

4.3.1 Model

To model farmers' decisions, I use the static inter-temporal supply model developed in Chapter 1, which takes into account both crop and livestock activities, with the same 40 farm groups. Each farm group serves as a modeling unit representing the economic behavior of similar farms. I use the same dataset and apply the model for the Grand Est region, considering the heterogeneity across its three sub-regions: Champagne-Ardenne, Lorraine, and Alsace. In this model, the following activities are considered: dairy cattle, beef cattle, and crop-related activities (i.e. forage crops, wheat-barley-rapeseed-grain maize rotation, and permanent grassland). However, I introduce some modifications to the assumptions.

I assume that two carbon sequestration practices cannot be simultaneously applied to the same hectare due to a lack of data regarding carbon sequestration variance within sub-regions under such circumstances.

The carbon price is still exogenous but, contrary to Chapter 1, is assumed to be constant over time. I run model-based and results-based contracts scenario for payments ranging from 0 to $250\text{€}.tCO_2e^{-1}$, by steps of $10\text{€}.tCO_2e^{-1}$. For practice-based contracts simulations are run for payments ranging from 0 to $1750\text{€}.ha^{-1}$ by steps of $70\text{€}.ha^{-1}$.

For the model-based payment, I set α at 5%, so the quantile of the normal distribution is 1.64. To calculate the number of samples required in model-based and results-based contracts, I assume α and ϵ are both equal to 5%.

I choose 3 time horizons: 10, 20, and 30 years.

As [Antle et al., 2003] I assume a measurement frequency of 5 years. Indeed, it is the minimum average time interval after which a change in soil carbon stock can be observed and measured [Conant et al., 2003]. For the sake of simplicity, in the case of the result-based contract, I assume that the payment from carbon sequestration, which is in reality received every 5 years at the end of a measures campaign, is divided by 5, and this value is annually discounted and then added to the farmer's annual profit.

4.3.2 Carbon sequestration practices

The carbon sequestration practices are the same as those in Chapter 1. One involves converting cultivated land into permanent grassland and the three others are among those identified by [Pellerin et al., 2020]. These practices include the introduction of temporary grassland into fodder maize mono-cropping, the implementation of intra-plot agroforestry, and the plantation of hedges along fields. The introduction of temporary grassland can only be implemented on fodder maize, while agroforestry and hedges can be applied to both fodder maize and the crop rotation. Note that, in addition to

the annual implementation costs, both agroforestry and hedges involve fixed installation costs (such as purchasing and planting trees). For more details about the assumptions on the implementation of these three practices please refer to Chapter 1 and, for even more in-depth information, consult [Pellerin et al., 2020].

For each practice, the carbon stocks achieved after 30 years of implementation (used as data on equilibrium carbon stocks) and the costs of implementation are taken from [Pellerin et al., 2020]. However, [Pellerin et al., 2020] do not provide an estimate of the standard deviation of the average carbon sequestration resulting from the implementation of these practices on a sub-regional scale. In order to obtain an estimate of these standard deviations in each sub-region, I apply a method used by [Kim and McCarl, 2009], which involves assuming that the variation of crop yield is a "proxy" of the variation of carbon sequestration. Indeed, agricultural productivity is directly related to SOC contents [Lal, 2006, Kopittke et al., 2019]. [Kim and McCarl, 2009] use the coefficient of variation (CV), defined as the ratio of the standard deviation of the carbon sequestration estimation to the carbon sequestration estimation. [Kim and McCarl, 2009] conduct a regression of the CV of carbon sequestration against the CV of crop yield, using yield data simulated by a crop model at the scale of Eastern Texas and carbon stock inventories, and find a coefficient of 2.138. They then use this estimate, along with the CV of crop yields, to estimate the variation in carbon sequestration within Texas and at finer scales (county and district). Here, I calculate the CV of the crop yields over the space for the 2009 - 2013 period among the 40 farm groups for the Grand Est region and its three sub-regions (see table 4.1) ⁷.

⁷CVs of crop yields over time have also been estimated (see table 4.5 in appendix), and are lower than CVs over space, as in [Kim and McCarl, 2009].

Region	Grain maize	Rapeseed	Wheat	Barley	Average
Grand Est	2.74	1.82	2.05	2.26	2.13
Champagne	1.94	2.99	3.06	3.40	3.12
Lorraine	3.55	2.50	2.25	2.50	2.43
Alsace	3.04	1.39	1.91	3.67	2.73

Table 4.1: Coefficient of variation for crop yields over space (2009-2013) (Unit :%). For the average CV of crops it is weighted by the surface area of each crop in each sub-region in the 2009-2013 data for the 40 farm groups from the FADN.

The results in Table 4.1 show slightly lower CVs compared to those reported in [Kim and McCarl, 2009]. However, there are many reasons for these differences. Indeed, the region studied in [Kim and McCarl, 2009], Texas, has other pedoclimatic conditions, different crops, and distinct geographical scales. All these factors can influence CV values.

It is therefore difficult to determine whether the coefficient linking the CV of yields and the CV of carbon sequestration of 2.138 found by [Kim and McCarl, 2009] would be applicable here. To the best of my knowledge, there are no French studies that have assessed the relationship between agricultural yields and carbon sequestration. But while I lack data on the variability of carbon sequestration in the three sub-regions of the Grand Est, I have information regarding the CV of 2019 carbon stocks based on land use from [Martin et al., 2019], see table 4.2. Although this stock variability cannot directly give an estimate of carbon flux variability, it can provide an order of magnitude. Indeed, carbon sequestration partly depends on soil carbon stock levels at equilibrium.

Sub region	Land use	CV (%)	Number of samples
Champagne	Crop	3.70	57.00
Lorraine	Crop	7.46	22.00
Alsace	Crop	13.69	13.00
Champagne	Permanent grassland	5.97	20.00
Lorraine	Permanent grassland	5.46	30.00
Alsace	Permanent grassland	19.08	2.00

Table 4.2: Coefficient of variation of carbon stocks in the three sub-regions for different land use from 2019 experimental data [Martin et al., 2019].

Carbon stocks CV also appear relatively low, roughly corresponding to a ratio of 2 (or slightly higher) compared to crop yields CV at the scale of the three sub-regions. Except in the case of Alsace, where I observe a higher CV which may be explained by the limited number of soil samples collected in this region. Given the lack of information and data surrounding the estimation of the parameter linking the yield CV and the carbon sequestration CV, I run the model for three values of coefficient 1, 2, and 4, i.e. half, the equivalent, and twice the estimate of [Kim and McCarl, 2009]. The aim is to comment on the results in terms of cost-effectiveness between the types of contract according to this value. For each contract, there are therefore three standard deviation (σ_r^p) scenarios: low, medium, and high. Table 4.3 shows the standard deviation values and the associated carbon sequestration averages that I use to infer the standard deviation (σ_r^p) from the CV.

Practice	Region	Soil CS	Above-ground biomass CS	Total CS	Standard deviation (σ_r^p)		
					Low	Medium	High
Agroforestry	Champagne	4.58	16.50	21.09	0.66	1.32	2.63
	Lorraine	4.58	16.50	21.09	0.51	1.02	2.05
	Alsace	4.40	16.50	20.90	0.57	1.14	2.28
Hedges	Champagne	0.35	4.29	4.64	0.14	0.29	0.58
	Lorraine	0.36	4.38	4.74	0.12	0.23	0.46
	Alsace	0.39	4.97	5.37	0.15	0.29	0.59
Temporary grass-land	Champagne	5.84	0.00	5.84	0.18	0.36	0.73
	Lorraine	5.74	0.00	5.74	0.14	0.28	0.56
	Alsace	5.80	0.00	5.80	0.16	0.32	0.63
Permanent Grass-land	Champagne	9.85	0.00	9.85	0.31	0.61	1.23
	Lorraine	9.85	0.00	9.85	0.24	0.48	0.96
	Alsace	7.30	0.00	7.30	0.20	0.40	0.80

Table 4.3: Mean of carbon sequestration for each practice and each region and the standard deviation of this mean according to the different value scenarios of the coefficient linking crop yield CVs with sequestered carbon CVs, in $tCO_2e.ha^{-1}$ over 5 years. CS = carbon sequestration. The means come from [Pellerin et al., 2020], except for conversion to permanent grassland. The average carbon sequestration per year and per ha over 30 years is calculated using the carbon stock evolution function (Misterlich Baule) of the model developed in Chapter 1.

The values of the variance of the carbon sequestration per practice and per sub-region used to calculate the number of samples required in the case of a results-based and model-based contract are estimated from the values of the standard deviation in table 4.3.

Since the data for the spatial variability values of carbon sequestration are quite low, and spatial variability is only part of the variability of carbon sequestration, the simulations are also run for a higher coefficient, of 10, linking the CV of crop yields to the CV of carbon sequestration. The aim is to see how the results in terms of the cost-effectiveness of the different contracts change for a much higher variability of carbon sequestration. The data relating to carbon sequestration CV and the resulting standard deviations for each practice in each sub-region are available in Table 4.8 in the appendix.

4.3.3 Farmers' risk aversion parameters

Assuming farmers' utility function follows a CRRA function, I use estimates of the relative risk aversion parameter from [Rommel et al., 2023] because their publication is recent, and one of the samples in their study is composed of potato producers located in regions geographically close to our area of application. Risk aversion parameters were estimated based on lottery choice experiments. The average coefficient of relative risk aversion in this sample is estimated at 0.183 with a 95% confidence interval of [0.137,

0.229] and the standard deviation is 0.23 ⁸.

These values are consistent with other estimates of relative risk aversion of French agricultural producers from the literature. In the same article, [Rommel et al., 2023] also estimate the average coefficient of relative risk aversion in a sample of 28 organic farmers with a more diversified production (vegetable growers, livestock and crops) in the North-West part of France. This estimation is 0.187 with a 95% confidence interval of [0.119, 0.256] and a standard deviation of 0.184. [Bocqu ho et al., 2014], for a sample of 107 farmers producing various types of products in Bourgogne, estimate an average relative risk aversion of 0.21 with a 95% confidence interval of [0.173, 0.251], and standard deviation of the sample equal to 0.195. [Bougherara et al., 2011] relied on a sample of 30 farmers in central France and obtained average estimates of risk aversion of 0.125 with standard errors of the mean estimation of 0.098.

To implement the Monte Carlo method on the risk aversion parameter, I carry out 1000 random draws, where each draw corresponds to a combination of risk aversion parameters for each farm group. Simulations for the result-based contract are then run for each draw.

4.3.4 Measurement costs

For the carbon stock measurement costs, I use the same assumptions as in Chapter 2, i.e 64 €/sample. This data is from a French private laboratory and has been obtained through personal communication. It includes the costs of measurement and the soil analyses as well as labour cost (travel and laboratory work). For practices involving the implementation of agroforestry and hedgerows, it is also necessary to measure the carbon sequestered in biomass. Similar to [Cardinael et al., 2017], I assume that above-ground

⁸Deduced from the standard error of the estimate of the mean

biomass is estimated by multiplying trunk and branch volume by wood density, using the global wood density database. Thus I assume the wood volume to be measured when taking soil samples, thus incurring no additional measurement costs. Note that, I do not consider data uncertainty from the wood density database.

4.3.5 Additionality of practice implementation

I assume that the contracts remunerate only the surface areas – or the carbon sequestration on the surface areas– newly converted to practices in order to avoid windfall effects. To establish the baseline I run the model without contract nor economic incentive (as in the model validation in Chapter 1). The areas under permanent grassland and fodder maize-temporary grassland rotation implemented in each farm group in the baseline are not eligible to the contracts (see tables 4.6.3 and 4.6.3 in the appendix).

4.4 Results

Figure 4.2⁹ shows the marginal cost curve of carbon sequestration for each type of contract for the 3 time horizons and the 3 levels of carbon sequestration spatial variability scenarios. For the results-based contract, the curve shown is the average of the certain equivalent values obtained from the various Monte Carlo runs on the risk aversion parameter. We can see that, for equivalent carbon sequestration and a given type of contract, the marginal cost of carbon sequestration decreases as the time horizon increases. This observation is mainly explained by the fact that, for identical implementation costs, the practices generate additional carbon sequestration over a longer period, and so carbon sequestration is higher.

⁹The uncut figure is shown in figure 4.6 in Appendix .

The different carbon sequestration variability scenarios do not appear to have any real influence on the carbon sequestration cost curves for the different contracts, even though the practices adopted differ slightly depending on the variability scenario (see figures 4.9 and 4.8 in appendix). This can be explained by the low values of carbon sequestration spatial variability data, which were used as input parameters.

It should also be noted that the curves for the practice-based for agroforestry, model-based and results-based contracts are the same for all time horizons, which can be explained by the fact that for most of the carbon sequestration spatial variability and time horizon scenarios, farmers largely adopt agroforestry for the results-based and model-based contracts (see figures 4.9 and 4.8 in appendix). Indeed, although agroforestry is expensive to implement, with fixed costs of around $1000 \text{ €} \cdot \text{ha}^{-1}$, it has a very high carbon sequestration potential. Moreover, the practice-based for agroforestry, model-based and results-based contracts have almost similar results, as the data of carbon sequestration spatial variability have low values. Indeed, the entry costs for contracts, which vary from those of practice-based contracts, such as the risk premium for farmers in results-based contracts, the discount in uncertainty level for model-based contracts, and the monitoring costs (the number of samples depends on the variability of carbon sequestration) for both results-based and model-based contracts, are relatively low.

The practice-based contracts for the implementation of hedges and the conversion of cropland into permanent grassland have a higher marginal cost than the practice-based contract for agroforestry, result-based and model-based contracts for the same carbon sequestration, due to the low sequestration potential of these practices compared with agroforestry.

This figure also shows that the most expensive and least efficient contract is the practice-based one for introducing temporary grassland into a mono-culture of fodder

maize. This can be explained by the fact that this practice is costly because the farmer cannot directly generate income (in the model grass cannot be bought nor sold by the farmers), so its implementation is associated with the cost of giving up a profitable crop activity. Nevertheless, grass fodder is used to feed cattle, generating income but emitting a lot.

Furthermore, Figure 4.3¹⁰, which represents the marginal cost curves of the net GHG mitigation, indicates a slightly negative net abatement (resulting in additional emissions compared to the baseline) for all levels of payment per hectare for practice-based contract for temporary grassland and for certain levels of payments for model-based and results-based contracts (except for a time horizon of 30 years with a low and medium variability scenario). The increase in net emissions compared with the baseline associated with the practice-based for temporary grassland contract is a maximum of $0.5MtCO_2e$ for a 20-year time horizon and $0.8MtCO_2e$ for a 30-year time horizon. For the model-based and result-based contracts, it is a maximum of $0.1MtCO_2e$, whatever the time horizon and the sequestered carbon variability scenario. The practice is implemented for payments ranging from 30 to 70 $\text{€} \cdot tCO_2e^{-1}$ for model-based contracts and from 20 to 60 $\text{€} \cdot tCO_2e^{-1}$ for result-based contracts, for a time horizon of 20 years and a medium sequestration variability scenario (see figures 4.9 and 4.8 in appendix). The remuneration for these practices leads to an increase in GHG emissions compared to the baseline. Indeed, at these payment levels, this remuneration increases the opportunity cost of sustaining a larger herd compared to the baseline, since these practices involve the production of grass, which serves as feed for the animals. But the increase in the number of livestock is limited (according to the model's assumptions), which is why practice-based for temporary grassland contracts have a higher cost than other contracts.

¹⁰The uncut figure is shown in figure 4.7 in Appendix.

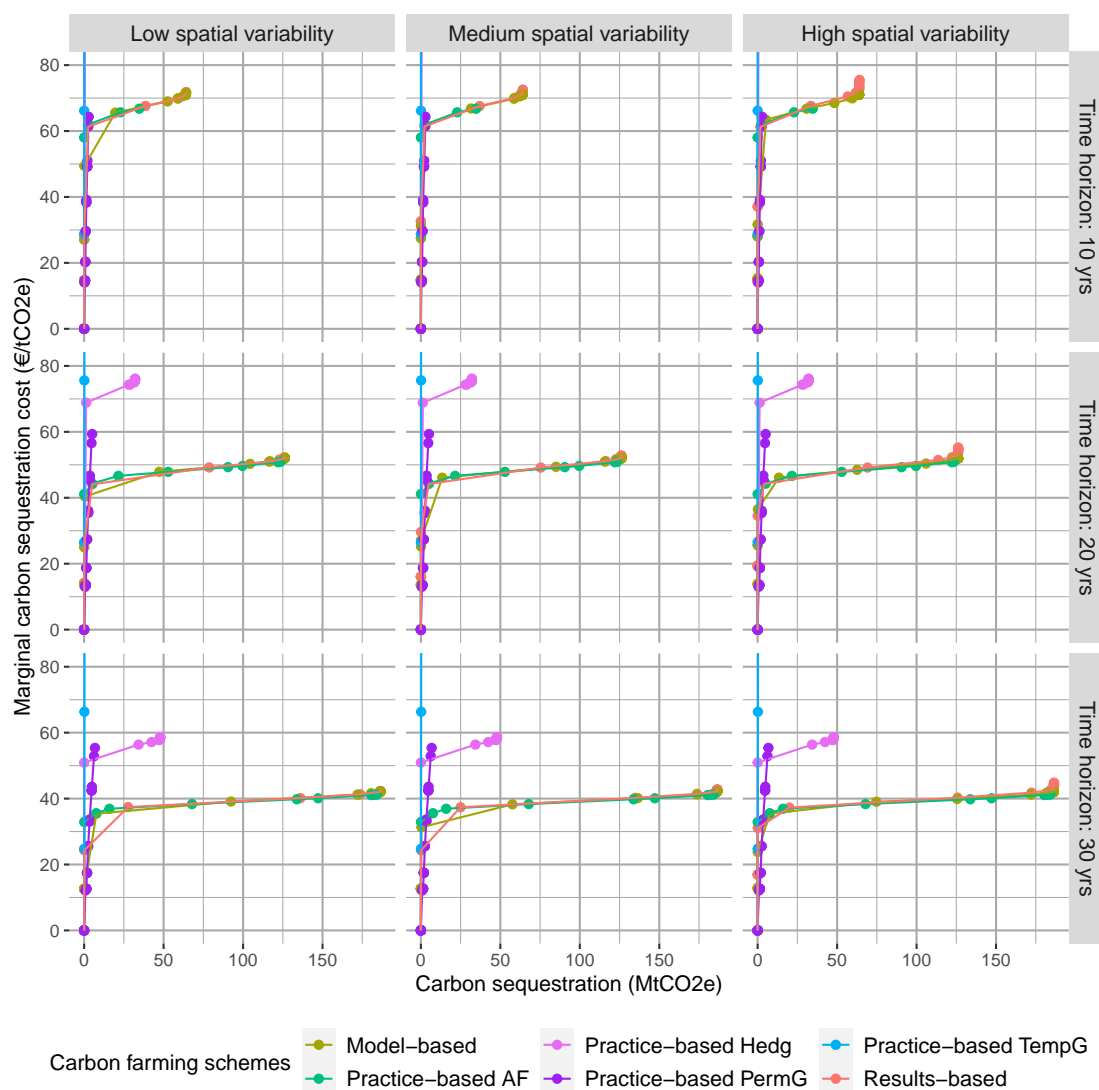


Figure 4.2: Curve of marginal carbon sequestration costs for each type of carbon farming scheme, as a function of the duration of the time horizon and the scenario of spatial variability of carbon sequestration. *For the results-based scenario, the results shown in this figure are from the average of the certain equivalent values from the Monte Carlo draws for a fixed payment level in $\text{€} \cdot \text{tCO}_2\text{e}^{-1}$. Carbon sequestration is estimated as additional sequestration over the entire time horizon relative to the baseline. Marginal costs are, for each level of payment (per ha or per ton depending on the nature of the contract), all the certain equivalent of the opportunity costs incurred by farmers in setting up contracts (difference between the sum of discounted profits in baseline and with the contract) plus the monitoring costs borne by the regulator, both over the entire time horizon, divided by carbon sequestration. AF= agroforestry, TempG= temporary grassland, PermG= permanent grassland, Hedg= hedgerows*

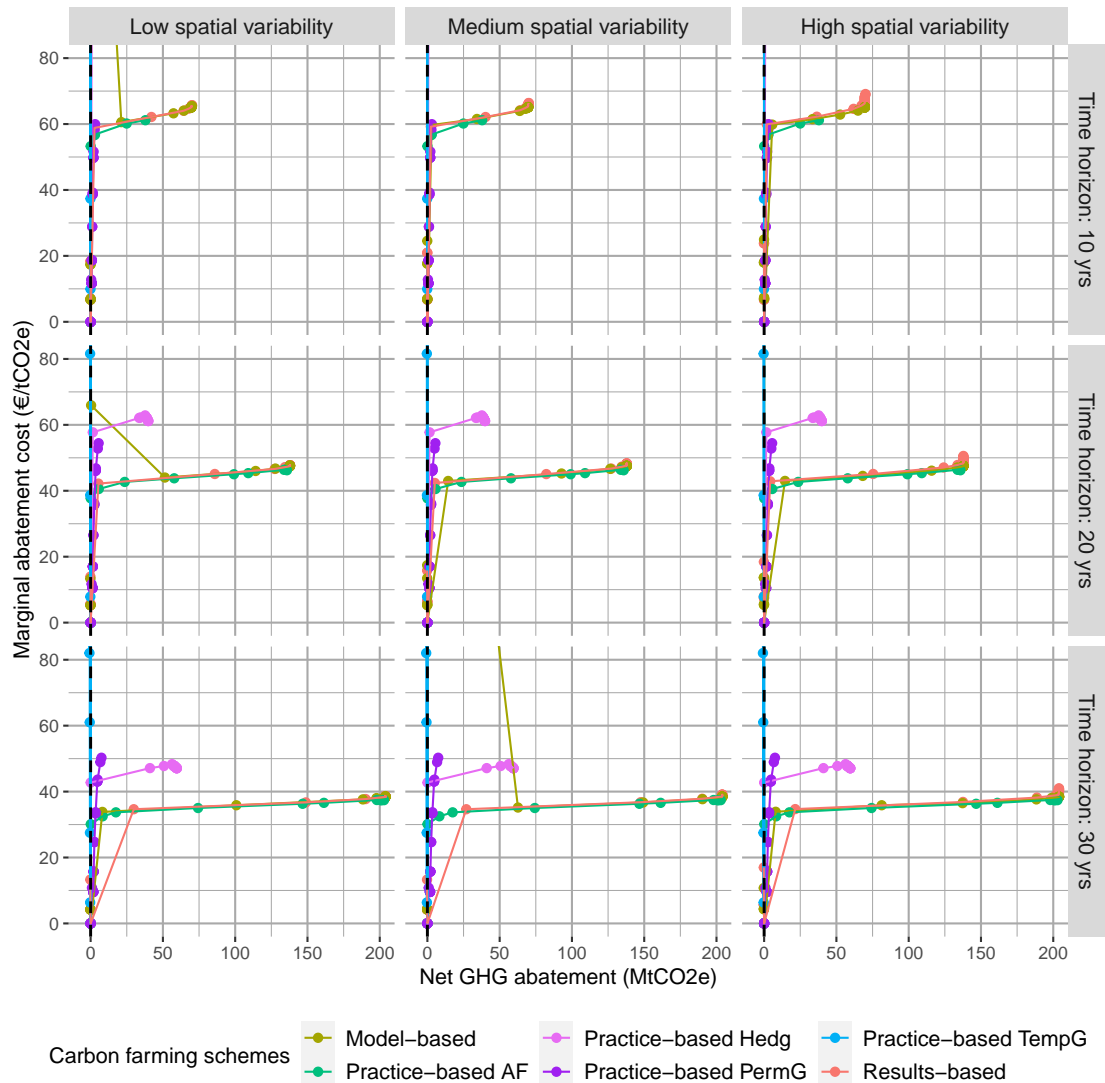


Figure 4.3: Curve of marginal Net GHG abatement costs for each type of carbon farming scheme, as a function of the duration of the time horizon and the scenario of spatial variability of carbon sequestration. For the results-based scenario, the results shown in this figure are from the average of the Monte Carlo draws for a fixed payment level in $\text{€} \cdot \text{tCO}_2\text{e}^{-1}$. For each contract scenario, net GHG abatement is estimated as the difference between GHG emissions from the baseline scenario with GHG emissions less carbon sequestration from the considered scenario, over the entire time horizon. Marginal costs are, for each level of payment (per ha or per ton depending on the nature of the contract), all the certain equivalent of the opportunity costs incurred by farmers in setting up contracts (difference between the sum of discounted profits in baseline and with the contract) plus the monitoring costs borne by the regulator, both over the entire time horizon divided by net GHG abatement. AF= agroforestry, TempG= temporary grassland, PermG= permanent grassland, Hedg= hedgerows

Figure 4.4 shows the effect of the risk to farmers of entering into a result-based contract on the marginal costs of carbon sequestration for the 3 scenarios of carbon sequestration variability and the 3 time horizons. There is little or no difference between the results-based scenario where no risk is taken into account and the average results-based scenario. Nor is there really any difference between the average results-based scenario and the values of the 95% confidence interval of results-based scenario. This can be explained by the fact that the values of the carbon sequestration spatial variability data in the 3 scenarios are low. Thus, in order to test the evolution of the relative costs of the different contract scenarios as the variability of carbon sequestration increases, I run the model for the different contracts assuming that the coefficient relating the CVs of crop yields to the CVs of carbon sequestration is 10 (see figure 4.5), i.e. respectively 10 times, 5 times and 2.5 times higher than the Low, Medium and High scenarios explored previously.

Figure 4.5 shows the marginal carbon sequestration cost curves for the model-based, result-based (mean, 95% confidence intervals and risk-free) and practice-based for agroforestry scenarios, for a time horizon of 20 years, and a higher spatial variability scenario. The practice-based for agroforestry scenario is the most cost-effective, since the increase in carbon sequestration variability increases the marginal cost of carbon sequestration for results-based and model-based contracts in two ways. Firstly, it affects the cost of farmers participation in the contracts via the discount on uncertainty for the model-based contract and via the risk premium on the payment for carbon sequestration for the result-based contract. Secondly, the increase in carbon sequestration variability increases the monitoring costs for this two types of contracts since according to the formula used to estimate the number of samples required when the variability of carbon sequestration increases, the number of samples increases.

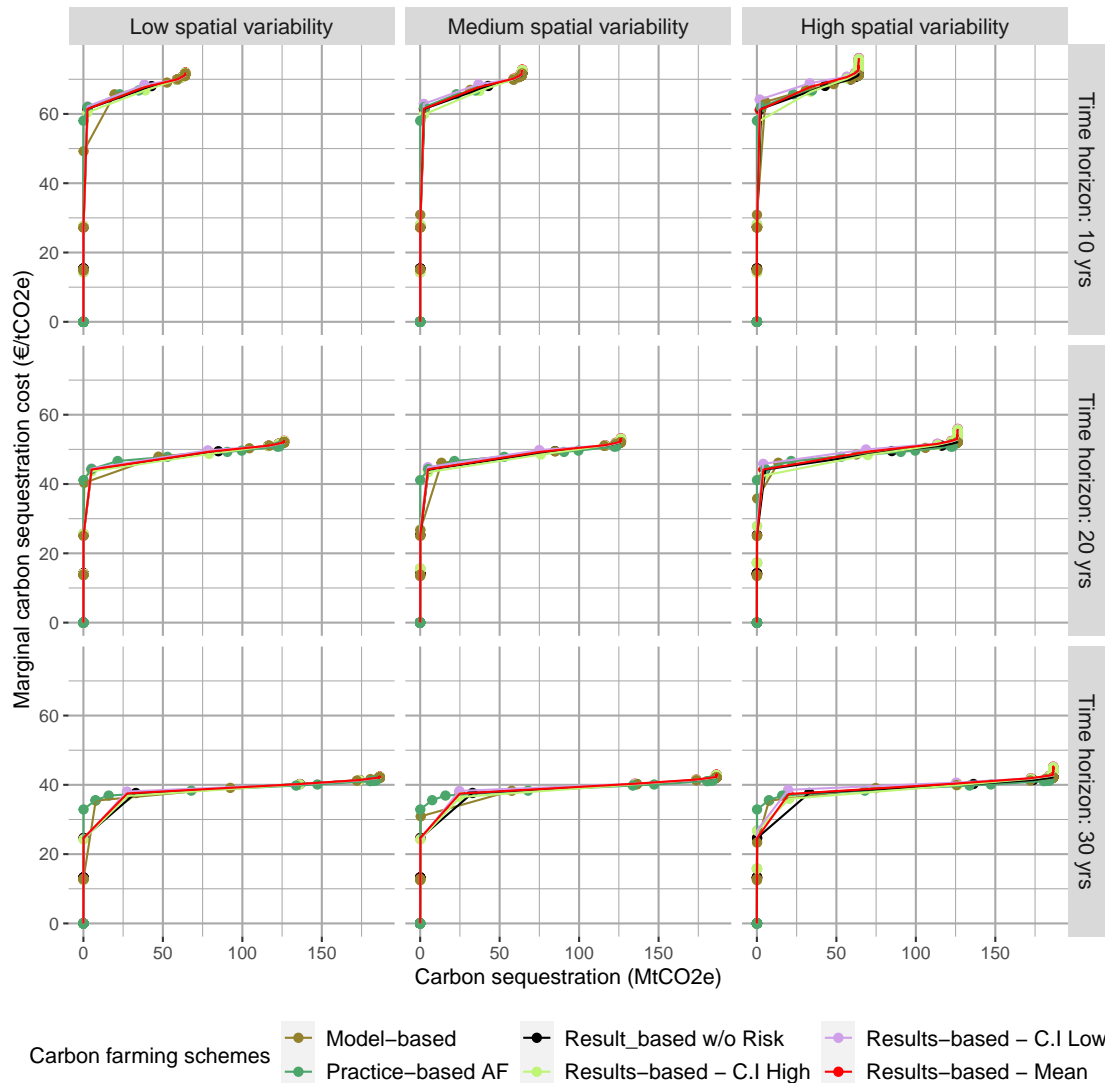


Figure 4.4: Marginal carbon sequestration cost curve as a function of spatial variability scenario and time horizon, and not, for the Grand Est region. *Results-based - Mean* = Mean value of the results of the results-based contract for a fixed payment level. *Results-based w/o risk* = Results-based contract without taking into account the risk for the farmers, relating to the uncertainty of the payment received from the carbon farming contract. *Results-based IC-Low/High* = Low/high value of the 95% confidence interval of the results of the results-based contract for a fixed payment level. Carbon sequestration is estimated as additional sequestration over the entire time horizon relative to the baseline. Marginal costs are, for each level of payment (per ha or per ton depending on the nature of the contract), all the certain equivalent of the opportunity costs incurred by farmers in setting up contracts (difference between the sum of discounted profits in baseline and with the contract) plus the monitoring costs borne by the regulator, both over the entire time horizon, divided by carbon sequestration.

The model-based contract has a limited carbon sequestration potential compared with the other contracts, since the discount on the level of uncertainty is more affected by the increase in the variability of carbon sequestration than the risk premium, for equal payments for carbon sequestration. Despite this, the difference between result-based and practice-based for agroforestry contracts is a maximum of $10 \text{ €} \cdot tCO_2e^{-1}$ for the same carbon sequestration achieved. But the result-based contract (mean and confidence interval) is more expensive than the model-based contract for the same mitigation, due to higher monitoring costs. The result-based contract that takes account of the risk shows a marginal cost that increases rapidly from a sequestration of $100MtCO_2e$, which can be explained by the fact that the risk premium is a quadratic function of the level of payment. Finally, the result-based contract without taking risk into account, for sequestration values between 0 and around $80MtCO_2e$, has a higher marginal cost than the result-based contract that takes risk into account. Indeed, without considering risk, results-based contract participation remains consistent regardless of carbon sequestration variability. However, as carbon sequestration variability increases, so do monitoring cost. This monitoring cost does not increase linearly with the number of ha enrolled, which is why after $80MtCO_2e$, the marginal cost of the result-based contract without taking the risk into account decreases.

Table 4.4 shows the marginal costs of carbon sequestration, the costs of measures, and the marginal costs of carbon sequestration for several type of contracts. We can clearly see the increase in monitoring costs as the variability of carbon sequestration increases. In comparison with these values, with a practice-based for agroforestry contract, and the same time horizon of 20 years, an additional carbon sequestration of around 100 (or 120) $MtCO_2e$ is achieved at a marginal cost of 49.64 (50.78) $\text{€} \cdot tCO_2e^{-1}$ for a payment per ha of 1610 (1680) $\text{€} \cdot ha^{-1}$.

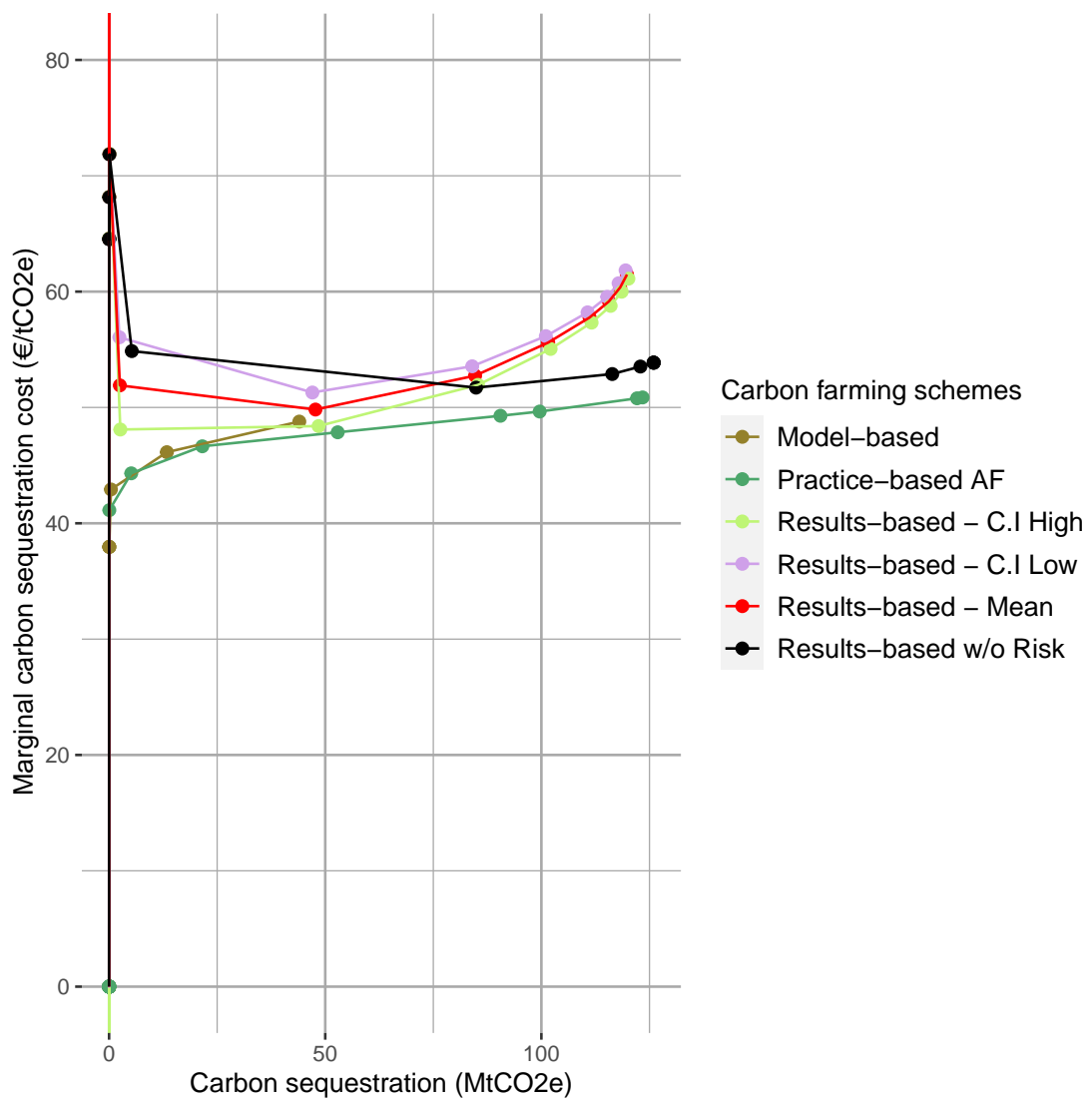


Figure 4.5: Marginal carbon sequestration cost curve for a 20 years time horizon, for the Grand Est region, with a very high carbon sequestration variability scenario. *The CV of the yields is multiplied by 10 (i.e. respectively 10 times, 5 times and 2.5 times the values taken for the Low, Medium and High scenarios previously studied). Results-based - Mean = Mean value of the results of the results-based contract for a fixed payment level. Results-based w/o risk = Results-based contract without taking into account the risk for the farmers, relating to the uncertainty of the payment received from the carbon farming contract. Results-based IC-Low/High = Low/high value of the 95% confidence interval of the results of the results-based contract for a fixed payment level. Payment per ton ranges from 0 to 140 €. tCO_2e^{-1} for results and model, and payment per hectare still ranges from 0 to 1750 €. ha^{-1} for practice based. Carbon sequestration is estimated as additional sequestration over the entire time horizon relative to the baseline. Marginal costs are, for each level of payment (per ha or per ton depending on the nature of the contract), all the certain equivalent of the opportunity costs incurred by farmers in setting up contracts (difference between the sum of discounted profits in baseline and with the contract) plus the monitoring costs borne by the regulator, both over the entire time horizon, divided by carbon sequestration.*

Type contract	CS variability scenario	Area enrolled (kha)	Number of sample (k)	Cost of sampling (k€)	Cost of sampling per ton of CS ($\text{€} \cdot tCO_2e^{-1}$)	Marginal cost of CS - farmers ($\text{€} \cdot tCO_2e^{-1}$)	Total marginal cost of CS ($\text{€} \cdot tCO_2e^{-1}$)	CS ($MtCO_2e$)	Net GHG mitigation ($MtCO_2e$)
Results-based Mean	2.00	306	0.2	186	0.00	51.74	51.74	122.88	134.40
	10.00	32	358	280520	2.76	53.20	55.96	101.58	111.00
Results-based C.I High	2.00	306	0.2	186	0.00	51.73	51.73	122.89	134.40
	10.00	32	361	283162	2.77	52.64	55.41	102.13	111.49
Results-based C.I Low	2.00	306	0.2	186	0.00	51.75	51.75	122.87	134.39
	10.00	32	354	277879	2.75	53.77	56.52	101.03	110.50
Results-based w/o Risk	2.00	208	19	15116	0.12	51.53	51.66	122.91	134.43
	10.00	313	2947	245678	1.99	51.53	53.53	122.91	134.43
Model-based	2.00	302	0.2	190	0.00	50.96	50.96	115.77	126.51
	10.00	3	2.5	152	3.38	34.58	37.96	0.04	-0.06

Table 4.4: Area enrolled, marginal costs and measurement costs, for each contract, for 20 years time horizon, $100 \text{€} \cdot tCO_2e^{-1}$ payment, and two carbon sequestration variability scenarios, one where variability is 2 times (Medium scenario) that of crop yields and the other 10 times. CS = carbon sequestration.

4.5 Discussion and conclusion

The question is whether the risk for farmers associated with uncertainty regarding the amount of carbon sequestration under a contract can impact the consensus in economic literature that result-based contracts are more cost-effective than practice-based ones. Additionally, the aim was to examine whether the model-based contract, being a hybrid contract with no risk but with a discount on uncertainty on the carbon sequestration and lower monitoring costs compared to the result-based contract, could be more cost-effective than the result-based contract.

With the carbon sequestration spatial variability values used at the beginning of the results, representing 1, 2 and 4 times the average coefficient of variation (CV) of crop yields in the sub-regions, the result-based, model-based and practice-based contracts remunerating the implementation of agroforestry are equivalent and are the most cost-effective. This is because agroforestry is the practice mainly implemented in the result-based and model-based contracts above a certain level of payment, and as the variability values are very low, the monitoring costs as well as the risk premium for results-based contracts and uncertainty discount for model-based contracts, are almost zero. Thus, taking risk into account does not affect the cost-effectiveness of results-based contracts. However, these results are based on the fact that the chosen carbon sequestration variability data are low and there is still uncertainty as to the robustness of these values. For this reason, an important prospect for improving this work lies in the selection of the values used to represent the variability of carbon sequestration. An interesting approach could be to explore other choices of carbon sequestration variability values from predictions of carbon sequestration models. Applying a Monte Carlo approach to carbon sequestration variability parameter could also be an appropriate

solution.

In order to see what effect a more extreme value of spatial variability would have on the relative marginal costs of contracts for the same carbon sequestration, I run the simulations for a carbon sequestration variability scenario representing 10 times the average CV of crop yields in the sub-regions, i.e. a carbon sequestration CV of around 30%. With this assumption, the practice-based agroforestry contract becomes the most cost-effective, as the increase in variability increases the risk premium for the result-based contract, the discount on the level of uncertainty for the model-based contract, and the monitoring costs for these both types of contract. Even so, the values of the marginal sequestration costs between these contracts remain close, since at most the marginal costs of sequestration for the result-based contract are 10 $\text{€}.\text{tCO}_2\text{e}$ more than those of the practice-based agroforestry contract for the same carbon sequestration, i.e. there is a maximum difference of 1.2 times the costs of the practice-based contract.

Thus, although work needs to be done on the values of carbon sequestration variability, risk for farmers about carbon sequestration results does not appear to have as much impact on the marginal costs of carbon sequestration. In addition, although not addressed in this work, the method of estimating uncertainty about carbon sequestration and the confidence placed in these data by farmers can also impact participation in results-based and model-based schemes. Indeed, the estimation of carbon sequestration uncertainty data can be costly and inaccurate, since it depends on numerous sources. Although in this work I focus on the spatial sequestration variability, other factors, like weather conditions [Birge et al., 2017, Šumrada et al., 2021], the risk of extreme events such as fire and storms, or uncertainty relating to the formulation of the equations of the carbon sequestration prediction model, also impact on the variability of carbon sequestration. In addition, I assume here that the farmer is aware of the spatial variability of carbon

sequestration before deciding to participate in the contract. However, in reality, this data can be provided by the project owner, or regulator for example, but it is not necessarily in their interest to deliver an accurate information to the farmers, at the risk of them not participating. Thus, the farmer may not trust this information, and decide not to commit in the case of results-based or model-based contract.

Given this brief discussion of the uncertainty of carbon sequestration data, and the results of simulations for a scenario of very high carbon sequestration variability (representing 10 times the average CV of crop yields), which show that the practice-based for agroforestry contract gives as equal or even better results than model-based and results-based contracts : it seems more appropriate to implement practice-based contracts to remunerate the implementation of agroforestry. Although expensive to implement, agroforestry has a very high carbon sequestration potential, of which around 75% per hectare is in the above-ground biomass according to [Bamière et al., 2023]. Furthermore, in the context of carbon farming, an estimate of the credits per ha of agroforestry planted could be made on the basis of observation of the average above-ground biomass using remote-sensing [West et al., 2008, Zaki and Latif, 2017]. This option is less costly than measuring soil carbon in-situ via soil analysis [Gehl and Rice, 2007]. But, I only take into account the technical costs of implementing agroforestry, and there are a certain number of factors that could act as a barrier and increase the costs of implementing this practice. Indeed, this practice has an impact on yields, and here I take into account [Bamière et al., 2023]'s hypothesis of a loss of around 11% of yields per ha, but this loss may be uncertain and constitutes another source of risk in implementing the practice for farmers. The risk of fire, which could release the carbon stored, could also have an impact on the implementation of this practice (even if it depends on who bears the risk of release). On the other hand, the costs of information and learning may

be significant given that this practice is not widely adopted in mainland France at the moment. Farmers' preference for the present could also be a barrier on investment in the implementation of agroforestry (purchase of trees, plantation, etc.). Lastly, the fear of a change in policy and an end to payment per hectare of agroforestry implemented could also be a barrier. Thus, the implementation of practice-based for agroforestry contracts could be promising, since it could save on monitoring costs and avoid the difficulties associated with estimating farmers' risk aversion and variability in carbon sequestration, but other costs could affect its implementation and need to be studied.

This work could be extended in various ways. In this study, I focus solely on few soil carbon sequestration practices. It might be possible to consider other practices identified as having an interesting sequestration potential on a French scale by [Bamière et al., 2023]. Finally, I don't consider moral hazard. However, in the case of results-based contracts, depending on the probability of carbon sequestration measurement, farmers may have more or less incentive to implement the carbon sequestration practice correctly, which could have consequences for farmers' participation.

4.6 Appendices

4.6.1 Certainty equivalent and risk premium of the CRRA utility function

The following formulas are adapted from concepts and teachings taken from E. Duguet online course¹¹.

We start with the CRRA utility function:

$$U(x) = \frac{x^{1-\gamma}}{1-\gamma}$$

where I make the assumption that $\gamma \neq 1$.

The absolute risk aversion index, denoted as $A_a(x)$, is given by:

$$A_a(x) = -\frac{U''(x)}{U'(x)} = \frac{1-\alpha}{x}$$

We can calculate the absolute risk premium denoted RP as follows:

$$RP = \frac{\sigma^2(X)}{2} \cdot A_a(E(W)) = \frac{\sigma^2(X)}{2} \cdot \frac{1-\alpha}{w + E(X)}$$

where X is the random part of a wealth W , and w is the certain part, with $W = X + w$.

The relative risk aversion index, denoted as γ , is given by:

$$\gamma = x \cdot A_a(x) = 1 - \alpha$$

Now, for the Certainty Equivalent (CE) in the case of Constant Relative Risk Aversion

¹¹http://emmanuel.duguet.free.fr/poly_miv1e.pdf

(CRRA), the formula is :

$$CE = E(X) + w - \frac{\gamma}{2} \cdot \frac{\sigma^2(X)}{w + E(X)}$$

$$= E(W) - \gamma \frac{\sigma^2(X)}{2 \cdot E(W)}$$

4.6.2 Details of analytical framework calculations

If results-based scheme the maximization problem at the farm level is :

$$\begin{aligned} & \max_{A_{l,p}} \sum_l \sum_{t=1}^T \rho^t \cdot E \left[U \left(\Pi_{l,t} + \sum_p A_{l,p} \cdot [g_{l,t}^p - c_{l,t}^p] \right) \right] \\ \text{s.c. } & 0 \leq A_{l,p} \leq uaa_l \cdot x_{l,p} \quad \forall l, p \\ & \sum_p x_{l,p} = 1 \quad \forall l \\ & x_{l,p} \in \{0, 1\} \quad \forall l, p \end{aligned}$$

Using the certainty equivalent of the CRRA utility function it can be written :

$$\begin{aligned} & \max_{A_{l,p}} \sum_{t=1}^T \rho^t \cdot \frac{1}{1-\gamma} \cdot \left[\Pi_{l,t} + \sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p \cdot (1-d_T) - c_{l,t}^p) - \frac{\gamma}{2} \cdot \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p)} \right]^{1-\gamma} \\ \text{s.c. } & 0 \leq A_{l,p} \leq uaa_l \cdot x_{l,p} \quad \forall l, p \\ & \sum_p x_{l,p} = 1 \quad \forall l \\ & x_{l,p} \in \{0, 1\} \quad \forall l, p \end{aligned}$$

For the first-order conditions for $A_{l,p}$, there are two cases: $\gamma = 0$ and $\gamma \neq 0$. In the first case, $\gamma = 0$, this means that the farmer is completely risk-neutral, and in this case the solution for $A_{l,p}$ is the same as for the model-based contract.

For $\gamma \neq 0$, first order conditions for $A_{l,p}$ are :

$$\sum_{t=1}^T \rho^t \cdot \left[\frac{\partial \left(\sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p) \right)}{\partial A_{l,p}} - \frac{\partial \left(\frac{\gamma}{2} \cdot \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p)} \right)}{\partial A_{l,p}} \right] \cdot \left[\Pi_{l,t} + \sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p) - \frac{\gamma}{2} \cdot \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} \cdot (\tau \cdot \bar{s}_{r,t}^p - c_{l,t}^p)} \right]^{-\gamma} = 0 \quad (4.24)$$

$$\sum_{t=1}^T \rho^t \left[\frac{\partial \left(\sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p) \right)}{\partial A_{l,p}} - \frac{\partial \left(\frac{\gamma}{2} \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p)} \right)}{\partial A_{l,p}} \right] \cdot \left[\Pi_{l,t} + \sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p) - \frac{\gamma}{2} \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p)} \right]^{-\gamma} = 0 \quad (4.25)$$

I know that the second product of the equation cannot be equal to 0 because a priori the certainty equivalent cannot be equal to 0 (which would mean that the farmer is willing to accept the risk without financial compensation, which is not possible because in this case $\gamma \neq 0$). Thus :

$$\sum_{t=1}^T \rho^t \left[\frac{\partial \left(\sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p) \right)}{\partial A_{l,p}} \right] = \sum_{t=1}^T \rho^t \left[\frac{\partial \left(\frac{\gamma}{2} \frac{(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p)^2}{\Pi_{l,t} + \sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p)} \right)}{\partial A_{l,p}} \right] \quad (4.26)$$

$$\sum_{t=1}^T \rho^t \cdot (\tau \bar{s}_{r,t}^p - c_{l,t}^p) = \sum_{t=1}^T \rho^t \left[\frac{\gamma}{2} \cdot \left(\sum_p A_{l,p} \cdot \tau \cdot \sigma_r^p \right)^2 \cdot \frac{\tau \bar{s}_{r,t}^p - c_{l,t}^p}{\left(\Pi_{l,t} + \sum_p A_{l,p} (\tau \bar{s}_{r,t}^p - c_{l,t}^p) \right)^2} \right] \quad (4.27)$$

4.6.3 Appendix of Data

Region	Grain maize	Rapeseed	Wheat	Barley	Average
Grand Est	1.94	2.27	1.16	1.34	1.67
Champagne	1.78	1.87	0.94	1.60	1.55
Lorraine	3.23	3.38	1.96	1.06	2.4
Alsace	1.62	1.73	0.42	1.85	1.4

Table 4.5: Coefficient of variation for crop yield over time for the period 2009-2013 (Unit: %)

Farm group	Surface area (ha)
1	0.99
2	4.53
3	29.89
4	0.09
5	0.99
6	2.46
7	46.52
8	79.24
9	128.44
10	90.28
11	70.14
12	94.05
13	146.14
14	47.79
15	79.78
16	18.65
17	34.62
18	56.89
19	43.60
20	72.95
21	114.69
22	65.41
23	142.80
24	92.90
25	134.72
26	50.99
27	58.10
28	96.67
29	4.56
30	2.80
31	1.04
32	3.71
33	2.33
34	50.38
35	33.96
36	59.32
37	65.83
38	14.89
39	20.59
40	58.47

Table 4.6: Area of permanent grassland in ha in baseline (without carbon sequestration contract) by farm group. This area is the same for all periods in the time horizon.

Farm group	Time period	Surface area in fodder maize - temporary grassland (ha)
7	All	3.12
8	All	11.91
9	All	36.01
10	All	3.65
12	Period 1	5.81
12	Period 2	0.78
13	All	30.11
14	Period 1	3.14
14	Period 2	0.39
15	All	38.59
19	All	3.82
20	All	14.53
21	All	24.93
22	All	1.63
23	All	21.51
24	All	17.54
25	All	33.21
26	All	2.88
27	All	11.13
28	All	30.19
35	All	25.32
36	All	48.23
38	Period 1	0.13
39	All	12.32
40	All	20.07

Table 4.7: Area of the rotation fodder maize- temporary grassland in ha in baseline scenario (without carbon sequestration contract) by farm group. *This area is the same for all periods in the time horizon, except for farm groups 12, 14 and 38 where the periods are specified. In these farm groups, the surface area of this rotation is not the same every year because it is used to adjust to the needs of the livestock, which then decreases (since in these farms the livestock decreases in the baseline, see the validation of the model in Chapter 1).*

Practice ($tCO_2e.ha^{-1}$)	Region (%)	Mean total CS ($tCO_2e.ha^{-1}$) (%)	SD 10 ($tCO_2e.ha^{-1}$) (%)	CV 10 ($tCO_2e.ha^{-1}$)
Agroforestry	Champagne	21.09	6.60	31.29
	Lorraine	21.09	6.60	31.29
	Alsace	20.90	5.70	27.27
Hedges	Champagne	4.64	1.40	30.17
	Lorraine	4.74	1.20	25.32
	Alsace	5.37	1.50	27.93
Temporary grassland	Champagne	5.84	1.80	30.82
	Lorraine	5.74	1.40	24.39
	Alsace	5.80	1.60	27.59
Permanent grassland	Champagne	9.85	3.10	31.47
	Lorraine	9.85	2.40	24.37
	Alsace	7.30	2.00	27.40

Table 4.8: Mean of carbon sequestration for each practice and each region and the standard deviation of this mean for a coefficient of 10 linking crop yield CVs with sequestered carbon CVs, in $tCO_2e.ha^{-1}$, over 5 years. CS = carbon sequestration.

4.6.4 Appendix of results

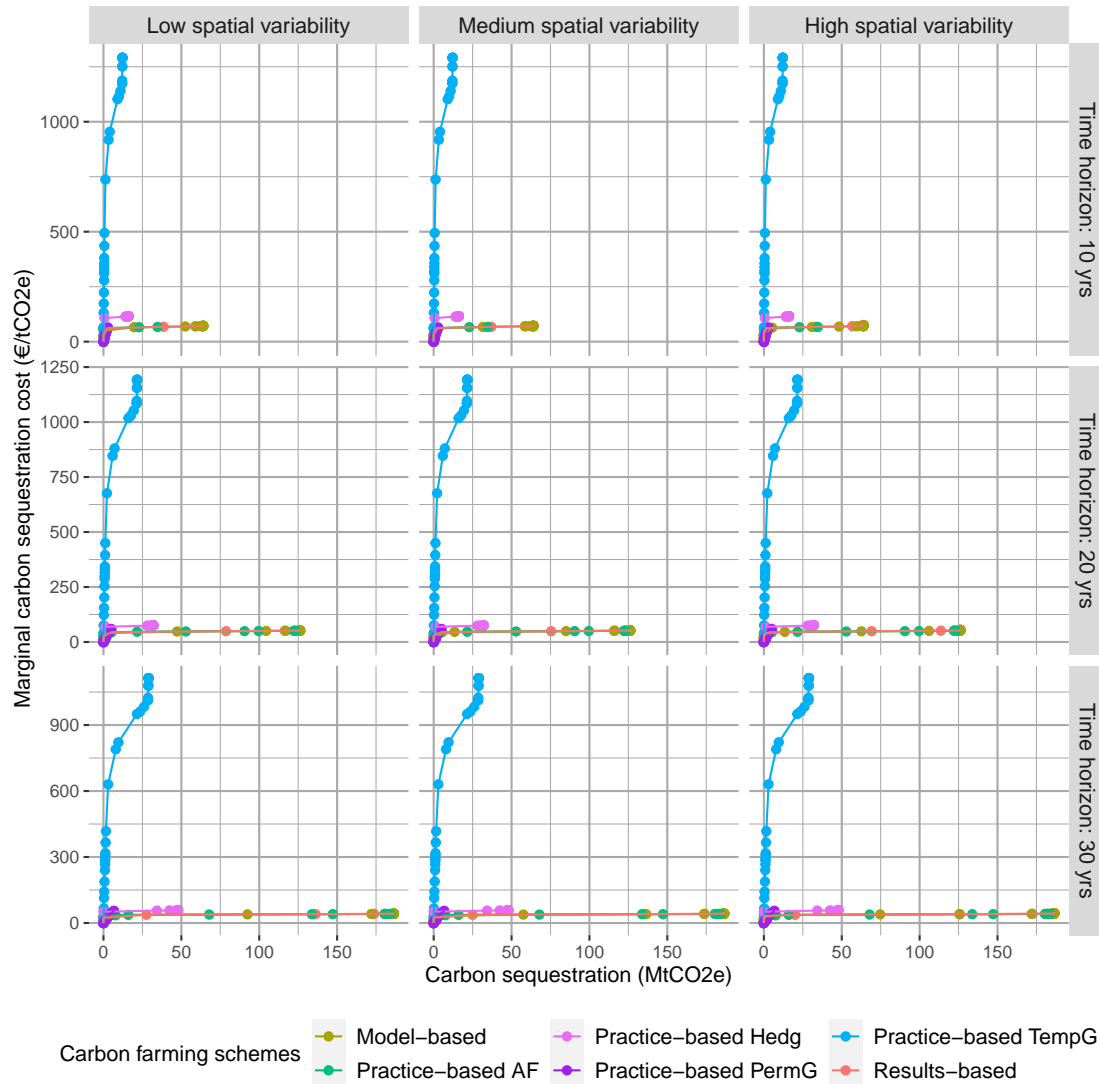


Figure 4.6: Curve of marginal carbon sequestration costs for each type of carbon farming scheme, as a function of the duration of the time horizon and the scenario of spatial variability of carbon sequestration. *For the results-based scenario, the results shown in this figure are from the average of the Monte Carlo draws for a fixed payment level in $\text{€} \cdot \text{tCO}_2\text{e}^{-1}$.*

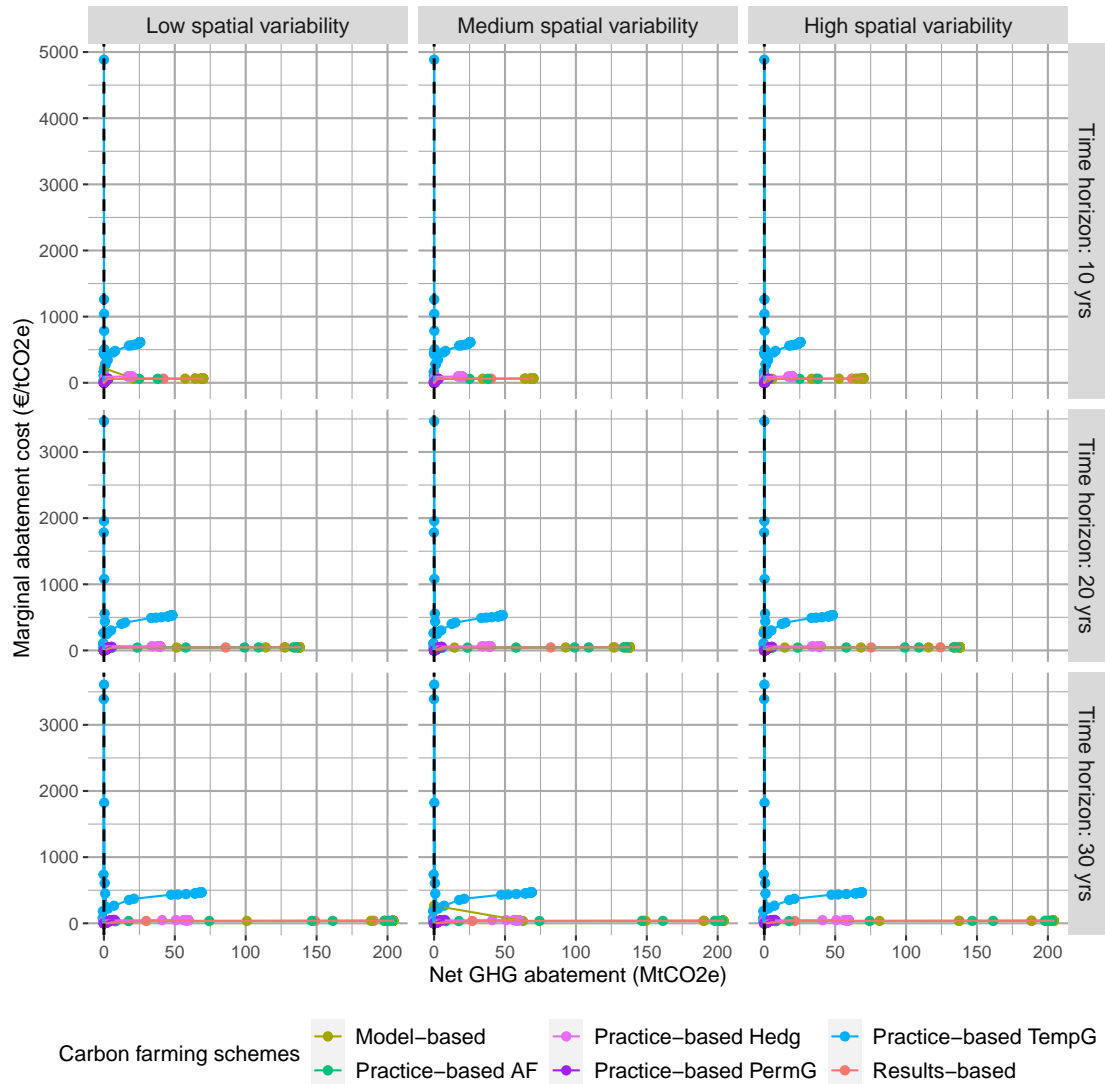


Figure 4.7: Curve of marginal Net GHG abatement costs for each type of carbon farming scheme, as a function of the duration of the time horizon and the scenario of spatial variability of carbon sequestration, for the Grand Est region. *For the results-based scenario, the results shown in this figure are from the average of the Monte Carlo draws for a fixed payment level in $\text{€} \cdot \text{tCO}_2\text{e}^{-1}$.*

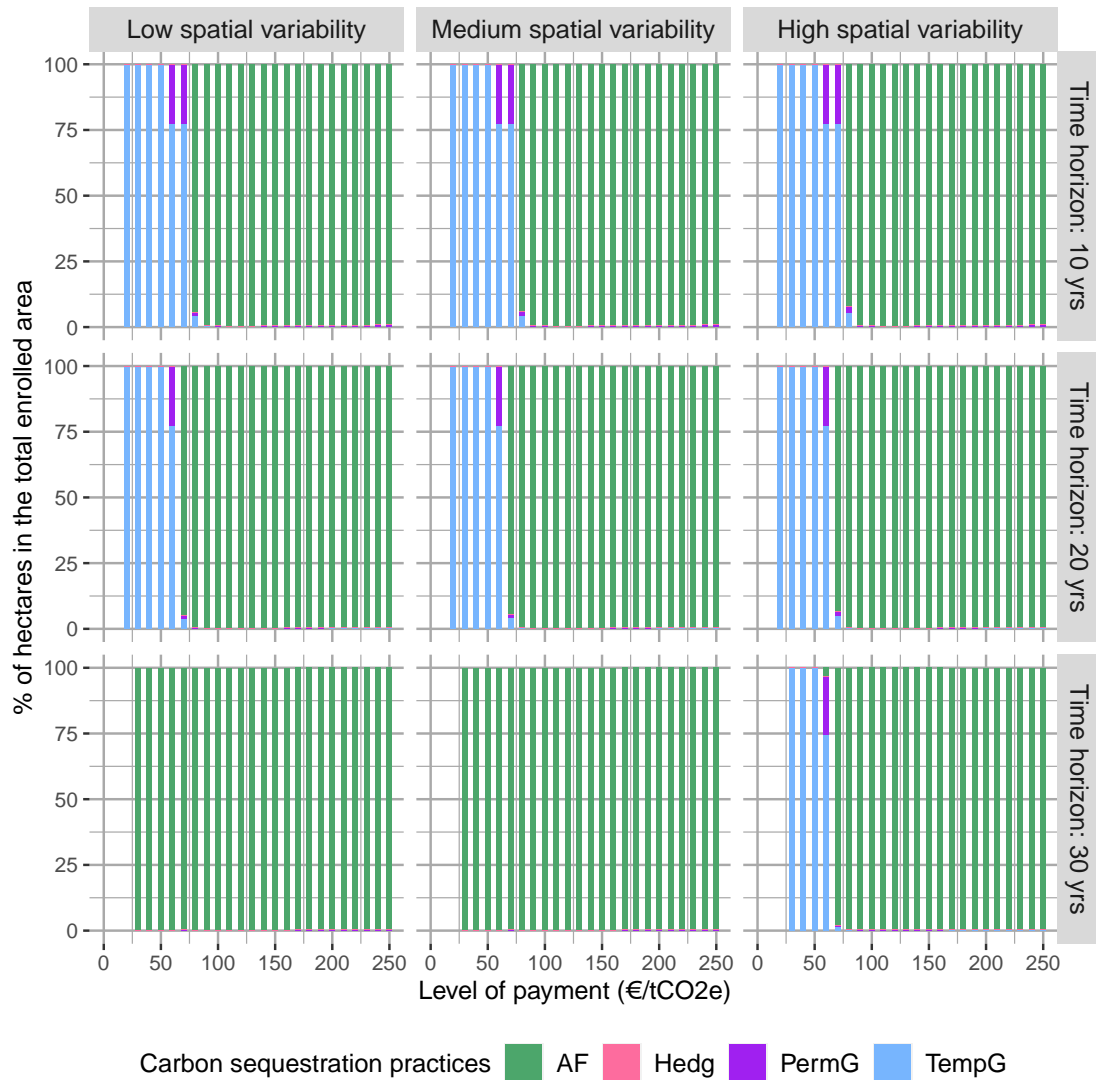


Figure 4.8: Percentage of hectares of each carbon sequestration practice in the total area enrolled as a function of the level of payment for the sequestration in $\text{€} \cdot \text{tCO}_2\text{e}^{-1}$, for the Result-based contract scenario, for the Grand Est region. *The results shown in this figure are from the average of the Monte Carlo draws for a fixed payment level in $\text{€} \cdot \text{tCO}_2\text{e}^{-1}$.*

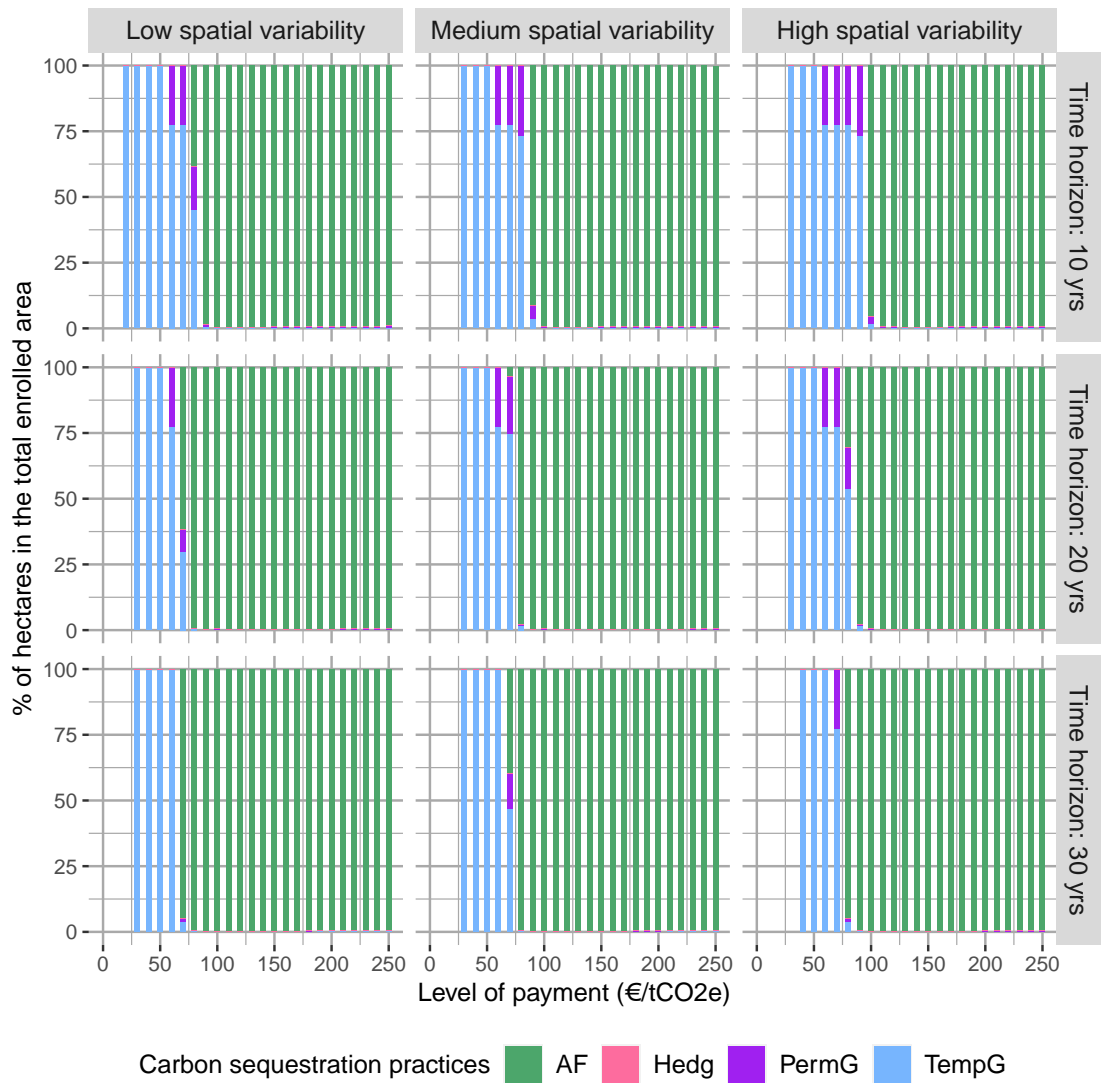


Figure 4.9: Percentage of hectares of each carbon sequestration practice in the total area enrolled as a function of the level of payment for the sequestration in $\text{€} \cdot t\text{CO}_2e^{-1}$, for the Model-based contract scenario, for the Grand Est region.

General conclusion

The agricultural sector is the second largest contributor of GHGs in France and it offers the opportunity for less costly mitigation than other sectors. However, since there are many sources and sinks of greenhouse gases and the potential and cost of mitigating agricultural practices vary from one farm to another, it is costly to monitor changes in net GHG emissions in this sector. But saving on monitoring costs can have an impact on the effectiveness of mitigation policies. Indeed, implementing an instrument with partial coverage allows for savings in monitoring costs, but its effectiveness is necessarily lower than that of an instrument with full coverage. Furthermore, results-based monitoring minimizes the risk of undue payments due to the heterogeneity in the mitigation potential of agents, compared, for example, to practice-based monitoring. However, this introduces a risk for farmers who do not know the amount of remuneration but bear the costs of implementing a practice, thereby increasing their participation costs and impacting the effectiveness of the policy. Therefore, the challenge of this thesis is to explore the trade-offs between the effectiveness of policies aimed at mitigating net greenhouse gases emissions and the costs of monitoring emissions and carbon sequestration.

This general conclusion is divided into three parts. The first part summarises the contributions of each chapter of the thesis. The second section presents the main limits

of the contributions. The third section suggests several ideas for future research.

Summary of contributions

Chapter 2 examines the implementation of policies that partially cover the sources and sinks of net GHG emissions. Using simulations from a supply-side model I have developed, I compare the effectiveness in terms of net GHG mitigation and the marginal mitigation cost of a tax on GHG emissions, a subsidy for total (soil+ biomass) and biomass carbon sequestration, with a tax covering all GHG sources and sinks. I take into account crop and livestock activities, GHG emissions, and carbon sequestration. I apply this modelling framework to the Grand Est region. The results suggest that the instruments partially covering GHG emissions or sinks all show net GHG mitigation. However, the difference in terms of net GHG mitigation is significant between policies covering all sources and sinks and partial coverage policies: depending on the carbon price trajectory, compared with a tax on net GHG emissions, a tax on GHG emissions leads to net GHG emissions that are 185 to 340% higher, and subsidies for total and biomass carbon sequestration lead to net GHG emissions that are 80 to 95% higher. Moreover, a subsidies for total and only biomass carbon sequestration are more cost-effective than a tax on GHG emissions. Indeed, according to the carbon price trajectory scenarios, compared with the carbon sequestration subsidy scenarios (total and biomass), the marginal costs of the GHG tax scenario are higher by a factor of between 2 and 5, and mitigation is lower by a factor of between 2 and 3. Moreover, total carbon sequestration and biomass carbon sequestration scenarios have very similar results, in terms of net GHG mitigation, marginal mitigation cost, and implementation of mitigation practices. This is interesting because monitoring only above-ground biomass would save

on monitoring costs while being almost as effective as if the subsidy also covered soil carbon. I can therefore assume that, taking monitoring costs into account, partial coverage of carbon sequestration by a subsidy for carbon sequestration in above-ground biomass would be more cost-effective than covering all carbon sinks (soil + biomass).

Finally, in this Chapter I do not include the cost of monitoring greenhouse gas emissions and carbon sequestration, influencing the cost-effectiveness of policies. I model regulation as results-based, without considering the technical costs of monitoring or behavioral aspects such as risk aversion, which impact the adoption of practices and thus the effectiveness and the marginal cost of mitigation. In the context of carbon farming, in Chapters 3 and 4, I take into account the risk for farmers related to the uncertainty of the gain from a contract and focus on cost-effective monitoring design for carbon farming contracts.

In Chapter 3, we compare the participation costs for a farmer between two types of carbon farming contracts: one based on model predictions and the other on measured results. We take into account the costs of implementing a carbon sequestration practice. For the results-based contract, we consider measurement costs, uncertainty surrounding the amount of gain, and the risk aversion of the farmer. For the model-based contract, we take into account a discount on payment that depends on the level of carbon sequestration uncertainty. The objective is, on the one hand, to present the theoretical conditions for participation in each contract based on parameters specific to the farmer, and on the other hand, to highlight the trade-offs in terms of risk-sharing among the farmer, the regulator, and society. To illustrate this, we empirically apply the analytical framework. An interesting aspect is that we observe that the certainty equivalent of the results-based scheme's gain, as a function of carbon payment level, takes the form of inverted U-shaped curve, initially increasing and subsequently decreasing. Thus, incentives to participate

in a results-based scheme may decrease beyond a certain carbon price.

With a few differences, Chapter 4 uses the same analytical framework as Chapter 3, and also explicitly takes into account the trade-offs between costs and uncertainties in the context of participation in the carbon farming scheme. In Chapter 4, unlike Chapter 3, I consider multiple farmers and their heterogeneity in terms of risk aversion, costs of implementing practices, and mitigation potential. The goal is to determine which contract is the most cost-effective among a results-based contract, a model-based contract, and several contracts each remunerating the implementation of a different carbon sequestration practice. The heterogeneity of farmers' risk aversion is taken into account by means of a Monte Carlo simulation on this parameter. I show that for relatively low values of carbon sequestration variability, the model-based, results-based, and practice-based contracts remunerating the implementation of intra-parcel agroforestry have nearly the same results in terms of marginal cost and carbon sequestration. For higher values of carbon sequestration variability, the participation costs increase for farmers in model-based and results-based contracts, thus the most cost-effective contract is the one remunerating the implementation of agroforestry. Since accurately estimating carbon sequestration variability is challenging, as it depends on numerous factors (pedoclimatic conditions, heterogeneity in the history of practices, risk of extreme events such as fire), to encourage the adoption of sequestration practices, it seems more relevant to implement payments per hectare for the implementation of agroforestry. Indeed, the results show that the implementation of intra-parcel agroforestry is the most adopted in the case of results-based and model-based contracts, since despite its high fixed costs of implementation, it has a very high carbon sequestration potential in soils and biomass. Moreover, while the increased variability of carbon sequestration affects monitoring costs and participation costs for farmers in model-based and results-based contracts, the practice-based contract

remains the most cost-effective.

Limits of contributions

One of the main limitations of Chapter 2 lies in the constraints I imposed on variations in livestock, which strongly influenced the results. The supply model, by its very nature, does not account for the effects of changes in supply on prices, and in the absence of these constraints, the number of heads of cattle increases excessively. However, in reality, other constraints, such as limitations related to buildings, human resources, and equipment, prevent unlimited growth of the herd. It would be necessary to rework the model constraints to consider these elements. In conclusion, the results obtained can be considered as relative to an exercise in which I set the maximum capacity of the herd, and where the current herd can only decrease.

For Chapter 3, the analytical framework is simple in that it takes into account the risk for the farmer participating in a results-based contract, ignoring, for example, the impact of income on risk aversion, as the difference between risk aversion to loss and risk aversion to gain. Nevertheless, this framework easily facilitates understanding trade-offs in terms of risk, participation in the contract, and uncertainties regarding carbon sequestration, which is the aim of this work. Additionally, we have set a fixed level of precision for soil analysis, but this can vary. Hence, there is a trade-off between the precision level of measurement and the associated monitoring costs in the case of the results-based contract, which we have not taken into account in this analysis.

Regarding Chapter 4, the values of carbon sequestration spatial variability used in the application are very low, and for the moment I have not been able to conclude on their representativeness. Nevertheless, I conducted simulations with a much higher level

of variability to assess the impact on the results, and they showed limited variation compared to those obtained with lower variability values.

Finally, in all three chapters, I consider the technical costs of implementing practices, and for Chapters 3 and 4, the costs related to the risk associated with the carbon contract's gain. However, other types of costs could impact the implementation costs of practices for farmers. For instance, there are costs related to the risk of yield loss linking to the implementation of the practice, as well as the cost of risk of fluctuations in agricultural product prices and transaction costs (information and education), etc that could impact the cost of committing into a contract and/or implementing a mitigation practice. Taking these costs into account could affect the results obtained in these three chapters. Particularly for Chapters 2 and 4, where agroforestry is widely adopted by farmers in simulations, as despite having significant implementation costs, it has a high carbon sequestration potential in both soil and biomass. However, the implementation of this practice, currently less widely adopted, could ,for example, require significant training costs and costs related to the risk of yield loss at the tree level and in the inter-rows.

Research perspectives

Certain barriers to the implementation of GHG mitigation incentives in the agricultural sector have not been explored in this thesis.

One example is the problem of assessing additionality in the estimation of carbon sequestration, relative to the estimation of the baseline. In the three chapters of this thesis I make the assumption that the trajectory of the carbon stock when the farmer does not adopt the practice is known with certainty and without cost. Indeed, for

Chapter 2 and Chapter 4, I make the assumption that in the static inter-temporal model that the baseline trajectory of SOC stock depends on input parameters resulting from the STICS simulations made by [Pellerin et al., 2020]. In Chapter 3, we assume that the baseline trajectory of SOC stock corresponds to that measured on the control plot of the experimental site in Ultuna. However, in the context of implementing carbon sequestration incentives, to estimate the carbon stock resulting from a mitigation practice, the determination of the baseline carbon stock is necessary and requires a trade-off between accuracy and associated costs (measurements, modeling, maintenance of a plot or part under baseline practices, etc.). The accuracy of the baseline estimate has an impact on the cost-effectiveness of the policy and can lead to a windfall effect. This windfall effect can be linked to farmers who would have implemented the practice in the coming years without intervention, but also to those known as the first movers, who have already adopted the practice. Not remunerating them could be socially unacceptable. The cost/uncertainty trade-off for estimating the baseline trajectory, as well as the trade-off between social acceptability and coverage of first movers, should be studied as part of the design of the European carbon farming certification framework.

Furthermore, net GHG mitigation policies can lead to carbon leakage, which I do not consider. In other words, changes in market conditions can lead to an increase in GHG emissions inside or outside the agricultural sector and inside or outside borders. Indeed, the implementation of net GHG mitigation policy can lead to a reduction in agricultural production through the introduction of practices that impact crop yields, or the herd size, or the removal of cultivated land from the market. This reduction in production can increase agricultural prices. On the one hand, this may provide sufficient incentive for the removal of other cultivated land from voluntary GHG mitigation projects. On the other hand, it may drive production out of the country, where agricultural production

costs are lower and product selling prices more affordable, thus boosting production elsewhere at the risk of also increasing GHG emissions locally. Since the implementation of carbon sequestration practices, such as intra-parcel agroforestry, could reduce agricultural yields, and it could be interesting to estimate the effect on the supply and price of agricultural products and the risk of leakage associated with the introduction of the carbon farming.

In addition to the uncertainty of carbon sequestration results discussed in Chapters 3 and 4, other behavioral factors, such as moral hazard, can impact monitoring costs in the context of implementing mitigation policies. Indeed, we assumed in Chapters 3 and 4 that whatever the monitoring method, mitigation practices were perfectly implemented by farmers. However, in reality some farmers may not perfectly implement the practice in order to save on implementation costs. Even with visual monitoring of practice implementation (e.g. satellite or in-situ monitoring), it is not always possible to determine whether a practice is being implemented perfectly, or to attribute observed defects (such as trees or hedges not growing sufficiently compared to what is expected) to a deliberately 'bad' implementation of the practice. With practice-based monitoring, these behaviours go unnoticed. On the other hand, with results-based monitoring, there is a probability that during the measurement campaign, the 'cheating' farmer's plots will not be sampled, enabling her/him to receive payment. This dimension should be taken into account when comparing the cost-effectiveness of monitoring methods. To study this, a stylised analytical framework could be constructed, initially at plot level, modelling the farmer's decision as to the effort deployed for the implementation of the practice, as a function of her/his aversion to risk and the probability that her/his payment will be impacted by the different monitoring and verification methods. The aim would be to assess whether this could affect the cost-effectiveness of the contracts and, if so,

to adjust the accuracy of the measurement campaigns (and thus increase the probability that a farmer's plots will be sampled) for results-based contracts.

In this thesis, the co-benefits associated with GHG mitigation practices have not been considered. These co-benefits can be ecosystem services, for example, the protection of biodiversity by creating new habitats in the case of agroforestry and hedgerows [Jose, 2009, Lal et al., 2015]. Mitigation practices can also help to make soils more resilient in the face of climate change [Lal et al., 2011]. For example, the use of cover crops can help to protect soils from erosion. Finally, the implementation of mitigation practices can generate an increase in crop yields or/and additional demand for labour, thereby contributing to the local economy. Several authors in the literature suggest that the co-benefits associated with the implementation of a practice should be taken into account, thereby increasing the level of incentive to implement the carbon sequestration practices. This is particularly important in the context of the European carbon farming certification framework, where the value of co-benefits can influence the sale of carbon credits on the voluntary market. Indeed, while in this thesis I have focused on the incentive for farmers to participate by taking into account an exogenous level of payment, in reality, the way in which a credit is sold on the voluntary carbon market has an impact on the level of payment made to farmers, the continuation of projects and therefore the environmental benefits of the policy. Thus, highlighting the co-benefits associated with a project would enable it to be sold more quickly or at a higher price on the voluntary market. But above all, it would be interesting to determine which co-benefits are most valuable to credit buyers. From a research perspective, it would be relevant to assess the willingness to pay of buyers of carbon credits for the co-benefits associated with the implementation of GHG mitigation practices. This approach would enable project developers and farmers to select practices on the basis of their co-benefits,

reducing the risk that carbon credits resulting from mitigation would be sold at too low a price, or would take time to sell. To answer this research question, data from existing voluntary carbon certification frameworks that highlight the co-benefits associated with their credits could be used. These data, associating the types of co-benefits (or absence of co-benefits) with the credits, as well as the prices at which the credits are sold, could be used to estimate willingness to pay using a hedonic pricing method.

Other elements relating to the sale of credits on the voluntary market may be problematic and need to be studied. One example is the fact that we do not yet know precisely whether the credits produced by the carbon farming will be used to voluntarily offset the emissions of companies without them being required to make any GHG reduction efforts, or whether they will be used to offset the emissions of companies within the framework of a market like the ETS (Emissions Trading Schemes). Another point is the voluntary framework of European certification, meaning that existing private voluntary frameworks (Low Carbon Label, Gold Standard, etc.) can voluntarily have their credits certified by the European framework. These credits will likely be more expensive to purchase than credits not certified by the European framework, which could be a barrier to the spread of more credible and transparent carbon credits.

Finally, an aspect that has received little attention in the literature concerns the time horizon of carbon sequestration policies. As carbon sequestration is not permanent, the cessation of a policy to encourage the implementation of sequestration practices could lead to the release of carbon stored in the form of CO_2 into the atmosphere. Furthermore, maintaining such policies on the long run could be costly for farmers and discourage them from committing to carbon farming contract. Currently, in the context of European certification of carbon farming, no fixed time horizon is defined, although sequestration is considered by several authors as a temporary solution pending the development of less

costly and permanent mitigation solutions. The sale of credits with different durations could be authorised, resulting in credits with varying monetary value. However, it is legitimate to ask what interest the buyer would have in acquiring a credit that (s)he would have to buy back a few years later, especially in anticipation of a potential rise in the price of carbon. In addition, there would be costs associated with monitoring the purchase of credits. It would therefore be interesting to know whether it would be more cost-effective to propose credit with different terms or the same term for all.

Résumé long

Le secteur agricole est l'un des plus grands émetteurs de GES (Gaz à Effet de Serre) en France, se classant au deuxième rang. En même temps, il a la capacité à séquestrer du carbone dans les sols et la biomasse aérienne, ce qui constitue un moyen de compenser les émissions incompressibles. De plus, ce secteur permet une atténuation des émissions de GES moins coûteuse que dans d'autres secteurs. Cependant, il n'existe actuellement aucune politique en France, ou au sein de l'Union Européenne, spécifiquement destinée à atteindre l'objectif d'atténuation fixé pour le secteur agricole. Plusieurs raisons expliquent cela, l'une d'entre elles étant le coût associé à la surveillance des émissions de gaz à effet de serre et de la séquestration du carbone. La multiplicité des sources et des puits d'émissions, ainsi que l'hétérogénéité du potentiel d'atténuation des exploitations, expliquent les coûts exorbitants et inenvisageables à payer par le régulateur pour une surveillance régulière, complète et précise de ces sources et puits.

Ainsi, cette thèse se concentre sur le compromis entre les coûts de surveillance et l'efficacité des politiques d'atténuation des émissions de GES et de la séquestration du carbone dans le secteur agricole.

En effet, idéalement, si toute l'information était disponible, sans coût, nous pourrions parfaitement prendre en compte l'hétérogénéité en termes de coûts (en tenant compte de tous les coûts techniques ainsi que d'autres coûts tels que le risque sur les rendements

,etc.) et de potentiel d'atténuation des exploitations, afin de mettre en place un instrument incitant à atténuer les émissions de GES nettes. L'allocation coût-efficace des efforts d'atténuation serait permise par la mise en place d'un instrument couvrant l'ensemble des sources et des puits d'émissions du secteur.

Cependant, dans la réalité, l'information parfaite n'étant pas disponible, la mise en œuvre d'une politique nécessite un compromis entre les coûts de surveillance des émissions de GES et de la séquestration du carbone, et l'efficacité en termes d'atténuation nette des GES. Les coûts de surveillance dépendent en particulier de la couverture de la politique et de la précision de l'estimation de l'atténuation des GES. D'une part, une couverture réduite des sources et des puits permet de réduire les coûts de surveillance, mais diminue l'efficacité en termes d'atténuation nette des GES. D'autre part, une surveillance plus précise réduit le risque de paiements indus et de surestimation de l'atténuation, mais peut augmenter le coût de participation au programme, en rendant le paiement incertain pour les agriculteurs. Cela a une incidence sur la mise en œuvre des pratiques d'atténuation, et donc, sur l'efficacité de l'atténuation des émissions de GES nettes. Par conséquent, les économies réalisées sur les coûts de suivi sont susceptibles d'avoir un impact sur l'efficacité de la politique. Cette thèse vise à explorer le compromis entre l'économie des coûts de surveillance et la variation associée de l'efficacité de politique d'atténuation des GES dans le secteur agricole à travers trois contributions.

Dans le Chapitre 2, intitulé "Towards carbon neutrality: Cost-effectiveness of targeting GHG emissions vs. carbon sequestration in agricultural sector", j'étudie l'efficacité, en termes d'atténuation nette des émissions de GES et de coût marginal d'atténuation, de plusieurs instruments couvrant partiellement les sources et les puits d'émission, par rapport à un instrument couvrant l'ensemble des sources et des puits. Plus précisément, en raison des interactions possibles entre les sources et les puits, je cherche à savoir si

un instrument encourageant uniquement la séquestration du carbone ou uniquement la réduction des GES peut avoir un effet positif sur l'atténuation nette des émissions de GES. En outre, j'examine lequel de ces deux types de politique de couverture partielle est la plus coût-efficace.

Pour ce faire, j'ai développé un modèle statique intertemporel du côté de l'offre, prenant en compte à la fois les activités de culture et d'élevage, ainsi que les émissions de GES et la séquestration du carbone. J'applique ce modèle à l'échelle de la région Grand Est, à 40 groupes types d'exploitations agricoles. Je considère plusieurs pratiques d'atténuation et prends en compte leur hétérogénéité en termes de coût et de potentiel d'atténuation au sein de chaque groupe type lorsque cela est possible, et au moins au sein de chaque sous-région du Grand Est (Champagne, Lorraine, Alsace).

Les résultats suggèrent que les instruments couvrant partiellement les émissions ou les puits de GES présentent tous une atténuation nette des GES. Toutefois, la différence en termes d'atténuation nette des GES est importante entre les politiques couvrant toutes les sources et tous les puits et les politiques de couverture partielle. En outre, les subventions pour la séquestration du carbone total et de la biomasse sont plus coût-efficaces qu'une taxe sur les émissions de GES. En effet, selon les scénarios de trajectoire du prix du carbone, par rapport aux scénarios de subvention du séquestration du carbone (total et biomasse), les coûts marginaux du scénario de taxe sur les GES sont plus élevés d'un facteur compris entre 2 et 5, et l'atténuation nette est plus faible d'un facteur compris entre 2 et 3. Enfin, les scénarios de séquestration total du carbone et de séquestration du carbone dans la biomasse donnent des résultats très similaires en termes d'atténuation nette des GES, de coût marginal d'atténuation et de mise en œuvre de pratiques d'atténuation. Ce résultat est intéressant, car le fait de ne surveiller que la biomasse aérienne permettrait d'économiser sur les coûts de surveillance tout en étant

presque aussi efficace que si la subvention couvrait également le carbone du sol. On peut alors supposé que, compte tenu des coûts de surveillance, la couverture partielle de la séquestration du carbone par une subvention pour la séquestration du carbone dans la biomasse aérienne serait plus coût-efficace que la couverture de tous les puits de carbone (sol + biomasse).

Dans cette première contribution, je compare le rapport coût-efficacité des politiques alternatives à la première meilleure, en supposant que les coûts de surveillance sont inférieurs pour ces alternatives, mais je ne tiens pas compte de ces coûts de manière explicite.

Dans les deux autres chapitres, nous prenons explicitement en compte les coûts de surveillance. En outre, nous comparons le rapport coût-efficacité des incitations basées sur les résultats, sur les prédictions de modèles et sur la mise en place des pratiques. Nous prenons en compte le risque associé à l'incertitude des paiements dans le cas des contrats basés sur les résultats. Cette incertitude a une incidence sur les coûts de participation des agriculteurs à ces types de contrats, ce qui, à son tour, affecte l'efficacité des contrats axés sur les résultats en matière d'atténuation des émissions de GES nettes.

En effet, dans le Chapitre 3, intitulé "Adoption of carbon farming schemes: Risk matters", nous étudions un programme volontaire de carbon farming et comparons le coût de la participation de l'agriculteur entre un contrat basé sur les résultats et un contrat basé sur les prédictions de modèle. Dans le cas du contrat basé sur les résultats, l'agriculteur supporte le risque de ne pas recevoir un paiement suffisamment important pour compenser les coûts de mise en œuvre de la pratique de séquestration du carbone. Dans le cas d'un suivi basé sur les prédictions d'un modèle, une remise sur le niveau de paiement est appliquée en fonction du niveau d'incertitude du carbone séquestré, mais le paiement à la fin du contrat est connu avec certitude par l'agriculteur. Notre

objectif est d'explorer analytiquement et de trouver des conditions théoriques pour les décisions des agriculteurs concernant la participation au contrat et les méthodes de suivi, en utilisant une approche stylisée à l'échelle de la parcelle. Cela nous permet également de mettre en évidence les compromis entre le régulateur et l'agriculteur et de discuter des conséquences du choix de la surveillance. Pour illustrer notre approche stylisée, nous utilisons des données provenant d'un site expérimental à Ultuna (Suède) et d'un ensemble multi-modèle (AMG, RothC, ICBM, Daycent) de stock de carbone organique du sol. Nous faisons diverses hypothèses sur le paramètre d'aversion au risque, le niveau de paiement par tonne de CO₂e et l'horizon temporel du contrat. Nous utilisons une fonction d'utilité de type CARA (Constant Absolute Risk Aversion) pour prendre en compte l'aversion au risque de l'agriculteur. La simplicité de sa forme permet d'explorer plus facilement les compromis entre les différentes méthodes de suivi. Nous supposons que l'incertitude de la séquestration du carbone est mesurée par l'écart-type des prédictions de chacun des modèles de l'ensemble. Ce deuxième chapitre est co-écrit avec Stéphane De Cara, Laure Bamière, et les agronomes et pédologues Elisa Bruni, Bertrand Guenet et Claire Chenu.

La prise en compte d'un seul agriculteur permet de discuter des arbitrages sur le coût de participation à un contrat selon la méthode de suivi, mais ne permet pas de conclure sur celle qui est la plus coût-efficace, puisque nous ne prenons pas en compte l'hétérogénéité entre les agriculteurs en termes de coûts, de potentiel d'atténuation des pratiques et d'aversion au risque. Dans le Chapitre 4, intitulé "Cost-effectiveness of carbon farming schemes taking farmers' risk aversion into account", je prends en compte ces hétérogénéités. L'objectif est de comparer le rapport coût-efficacité des contrats basés sur les pratiques, sur les prédictions de modèle et sur les résultats pour différents horizons temporels et différents niveaux de variabilité de séquestration du carbone issu

de la mise en place des pratiques, entre les exploitations. Pour ce faire, je développe un cadre analytique dans lequel le paiement dépend du type de contrat proposé, similaire à celui du Chapitre 3, à la différence que chaque agriculteur peut choisir la part de la surface agricole utilisée incluse dans le contrat. En outre, je prends en compte l'aversion au risque des agriculteurs en utilisant une fonction CRRA (Constant Risk Relative Aversion). La littérature expérimentale estimant les coefficients d'aversion au risque des agriculteurs privilégie la forme CRRA, ce qui rend plus claire et plus robuste la formulation d'hypothèses sur les données pour l'application. J'applique ensuite ce cadre stylisé au modèle intertemporel statique développé dans le Chapitre 2, à l'échelle de la région Grand Est. Cela me permet également de prendre en compte les coûts d'opportunité lié à la participation au contrat. Dans cette dernière contribution, j'utilise les prédictions d'un seul modèle (STICS) pour la moyenne du carbone séquestré, et donc l'écart-type des rendements des cultures comme proxy de l'écart-type de la séquestration du carbone pour chacune des pratiques dans chaque sous-région. Les résultats montrent que pour des valeurs relativement faibles de la variabilité de la séquestration du carbone, les contrats basés sur les prédictions d'un modèle, basés sur les résultats et basés sur la pratique récompensant la mise en œuvre de l'agroforesterie intra-parcellaire ont pratiquement les mêmes résultats en termes de coût marginal de la séquestration du carbone et de séquestration du carbone, et sont les plus cout-efficaces. Pour des valeurs plus élevées de la variabilité de la séquestration du carbone, les coûts de participation augmentent pour les agriculteurs dans les contrats basés sur les prédictions d'un modèle et sur les résultats, de sorte que le contrat le plus coût-efficace est celui qui rémunère la mise en œuvre de l'agroforesterie. Comme il est difficile d'estimer précisément la variabilité de la séquestration du carbone, car elle dépend de nombreux facteurs (conditions pédoclimatiques, hétérogénéité de l'historique des pratiques, risque d'événements extrêmes tels

que les incendies), et que cette donnée est susceptible d'impacter les couts de participation des agriculteurs et donc l'efficacité en terme de séquestration du carbone, il semble plus approprié, pour encourager l'adoption de pratiques de séquestration du carbone, de mettre en place des paiements à l'hectare pour la mise en œuvre de l'agroforesterie.

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