Agricultural management of ecosystem services: insights from production and environmental economics

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AGRICULTURAL MANAGEMENT OF ECOSYSTEM SERVICES:

INSIGHTS FROM PRODUCTION AND ENVIRONMENTAL ECONOMICS

THESE POUR LE DOCTORAT EN SCIENCES ECONOMIQUES

présentée par

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« J'préfère te dire, j'ai failli perdre mon sang froid. »

Etienne Daho - Weekend à Rome

Abstract

The thesis aims to study both theoretically and empirically the management of ecosystem services by the farmers from the perspective of the economic theory. The concept of ecosystem services is an interdisciplinary concept that refers to "the services that nature offers to human for free". The economic literature has mainly investigated this concept in measuring the value of these services, with few attention to the behavior of agents modifying these services. The thesis is divided into two parts.

In the first part, I study both the supply and the demand for the productive ecosystem services (for example, pollination or biological control) by analyzing the behavior of farmers, considered as potential agroecosystem managers. Inspired by the literature on landscape ecology, I introduce biodiversity indicators that are function of land-use into existing models from agricultural production economics literature. This reunion provides a unified theoretical model for analyzing farmers' choices regarding the management of productive ecosystem services. The empirical works consists in estimating all or parts of this theoretical model. My main contribution to the literature is to prove, based on the farmers' observed behavior, that farmers do manage productive ecosystem services. I bring other elements to the literature, notably by providing new insights on the agricultural technology when productive ecosystem services are considered, or by showing that collective management of productive ecosystem services can only rarely arise spontaneously in real landscapes where farmers are heterogeneous.

In the second part, I study the demand for the jointly provided public goods by the farmers' modification of ecosystem service flows, i.e. I study the specificities of the demand for environmental services provided by farmers (in the sense of Engel et al., 2008). In particular, I study the role of the geographic scale of the demand for the design of agri-environmental policy. Indeed, if local public goods influence the welfare of the agents within a defined geographical area (e.g., the improvement of water quality by maintaining a wetland upstream of a treatment plant), global public goods can influence the welfare of all agents (e.g., the carbon sequestration into the soil of a wetland). In this part, I apply the framework of several literatures developed in environmental economics (for example, the literature on environmental federalism or on the "distance-decay") to the specificities of the environmental services provided by farmers; in particular, I integrate that the environmental service provided by a farmer affects the supply of multiple public goods in most cases, the demand for these public goods arising at different geographical scales. I contribute to the literature by showing that, although most of the demand for environmental services provided by farmers is captured locally (at the municipal level), some of the demand is captured by larger and farer areas. This has implications for the governance and the design of agri-environmental policies, which I explore through two examples: the reduction of pesticide application and the maintenance of agricultural wetlands.

Keywords: supply analysis; demand analysis; ecosystem services; environmental services; public goods; agriculture; agro-environmental policy

Résumé

La thèse étudie théoriquement et empiriquement la gestion des services écosystémiques par les agriculteurs sous l'angle de la théorie économique. Le concept de services écosystémiques est un concept interdisciplinaire désignant « les services qu'offrent gratuitement la nature à l'homme ». La littérature économique s'est principalement emparée de la question de la mesure de la valeur de ces services, en s'intéressant peu ou prou aux comportements des agents modifiant ces services. La thèse se divise en deux parties.

Dans la première partie, je m'intéresse à l'offre et à la demande de service écosystémique productifs (par exemple, la pollinisation ou le contrôle biologique) en analysant le comportement des agriculteurs, considérés comme de potentiels gestionnaires des agroécosystèmes. Inspiré par la littérature en écologie du paysage, j'introduis des indicateurs de biodiversité dépendant des assolements dans des modèles existants issus de la littérature en économie de la production appliquée à l'agriculture. Ce rapprochement fournit un modèle théorique unifié où l'on peut analyser les choix des agriculteurs vis-à-vis des services écosystémiques productifs. Les travaux empiriques développés par la suite consistent à estimer toute ou partie de ce modèle théorique. Ma principale contribution à la littérature est de prouver, à partir de l'analyse des comportements observés des agriculteurs, que les agriculteurs gèrent consciemment les services écosystémiques productifs. J'apporte d'autres éléments à la littérature, en fournissant notamment des éléments importants sur la technologie agricole lorsque les services écosystémiques productifs sont considérés, ou en montrant que la gestion collective des services écosystémiques productifs ne peut que rarement émergée spontanément dans des paysages réels où les agriculteurs sont hétérogènes.

Dans la deuxième partie, je m'intéresse à la demande pour les biens publics fournis conjointement par les agriculteurs via la modification des flux de services écosystémiques, i.e. je m'intéresse à la demande pour les services environnementaux fournis par les agriculteurs (au sens de Engel et al., 2008). En particulier, j'étudie le rôle de l'échelle géographique de la demande sur la conception de politique agro-environnementale. En effet, si les biens publics locaux vont influencer le bien-être des agents au sein d'une zone géographique délimitée (e.g. amélioration de la qualité de l'eau en maintenant une zone humide en amont d'une station de traitement), les biens publics globaux peuvent influencer le bien-être de l'ensemble des agents (e.g. séquestration du carbone dans une zone humide). Dans cette partie, j'applique les cadres d'analyse de plusieurs littératures développées en économie de l'environnement (par exemple, la littérature sur le fédéralisme environnemental ou sur le « distance-decay ») aux spécificités des services environnementaux fournis par l'agriculture ; en particulier, le service environnemental fourni par un agriculteur influe le plus souvent sur la fourniture de multiple biens publics, biens publics dont l'échelle de la demande diffèrent. Je contribue à la littérature en montrant que, bien que la plupart de la demande pour les services environnementaux fournis par les agriculteurs soit capturée localement (à l'échelle de la municipalité), une partie de la demande s'exprime à des échelles plus importantes. Cela a des implications pour la gouvernance et la conception des politiques agroenvironnementales, que j'explore à travers deux exemples : la réduction de l'application des pesticides et le maintien des zones humides agricoles.

Mots-clés : analyse de l'offre ; analyse de la demande ; services écosystémiques ; services environnementaux ; biens publics ; agriculture ; politique agroenvironnementale

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Synthèse

L'agriculture reste l'activité économique la plus intensive pour l'utilisation des terres, occupant 37,5% de la surface terrestre mondiale et 54,7% des terres européennes. L'agriculture est par conséquent en charge de la gestion de la majorité des écosystèmes terrestres de la planète. Par essence, le travail de l'agriculteur a toujours été de gérer les multiples composantes des écosystèmes pour bénéficier au maximum de ces fonctionnalités. L'exemple de la rotation triennale introduite au court du Moyen-Age en est une illustration. Utilisant le vocable du Millenium Ecosystem Assessment (2005), les agriculteurs gèrent les services écosystémiques, c'est-à-dire les « bénéfices obtenus par les agents issus des écosystèmes ». La modification des niveaux de services écosystémiques par les agriculteurs peut leur être directement profitable mais elle peut aussi influencer l'utilité d'autres agents, notamment les résidents. Inspirée par Zhang et al. (2007), la figure 1.1. résume les liens entre les services écosystémiques et la production agricole. Dans ma thèse, je considère que les flux allant « depuis » les écosystèmes agricoles vers les biens agricoles sont des services écosystémiques productifs, tandis que les flux allant « depuis » les systèmes agricoles vers les services non-marchands sont les services écosystémiques contribuant aux biens publics. La modification des niveaux de services écosystémiques non-productifs est définie par Engel et al. (2008) comme des services environnementaux. La ligne en pointillé représente l'influence des pratiques agricoles sur les flux écologiques au sein des écosystèmes agricoles. Si les services environnementaux présentent par définition des caractéristiques de bien public, les services écosystémiques productifs peuvent présenter des caractéristiques de bien public ou de bien privé. Les services écosystémiques productifs présentent des caractéristiques de bien public si les flux générés par un agriculteur influent sur la rentabilité d'autres agriculteurs (par exemple, par le biais de la lutte biologique ou de la pollinisation), alors qu'ils présentent des caractéristiques de bien privé sinon (par exemple, la fertilité du sol).

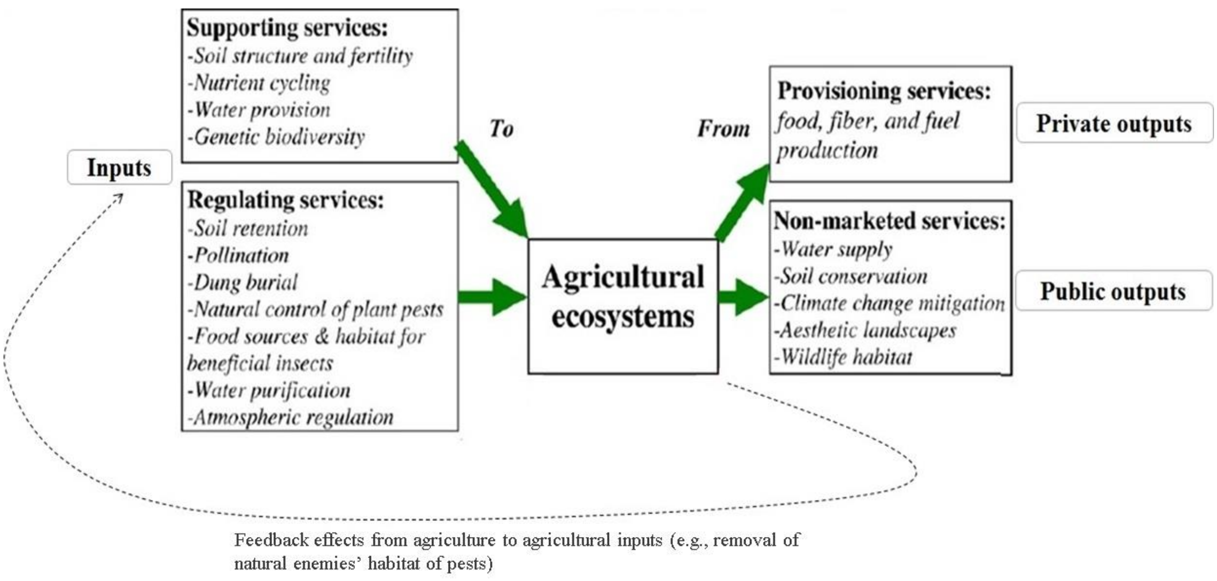

Figure 1.1. Services écosystémiques et écosystèmes agricoles (Source: inspiré de Zhang et al., 2007)

Depuis la fin de la seconde guerre mondiale, les activités agricoles se sont modernisés afin d'atteindre des objectifs économiques et de sécurité alimentaire. Ces objectifs ont été atteints dès les années 60 à la faveur d'une utilisation accrue du capital et des intrants chimiques et d'une utilisation moindre du travail et des services écosystémiques productifs, devenus moins rentables (Manuelli et Seshadri, 2014). Ces modifications ont généré une baisse de la qualité environnementale et ont incité les politiques à modifier leurs politiques de soutiens agricoles. Les réformes successives de la politique agricole commune (PAC) ont en effet abouti à un abandon des subventions couplées à la production et à l'introduction de subventions conditionnées à des pratiques agricoles respectueuses de l'environnement. Ces réformes sont toutefois critiquées par une partie des preneurs d'enjeux, comme par exemple les lobbys écologistes, pour la faible additionalité de plusieurs de ces mesures. En France, ces débats ont abouti vers une forme de consensus sur la nouvelle forme d'agriculture socialement désirable : l'agroécologie. L'agroécologie vise à réconcilier objectifs environnementaux et économiques en incitant à un changement de pratiques vers l'utilisation plus intensive des services écosystémiques productifs comme le non-labour, la lutte biologique ou la fertilisation organique. Les résidents devraient bénéficier de cette nouvelle d'agriculture via l'augmentation de la fourniture de services écosystémiques non-productifs mais aussi via la diminution de l'utilisation d'intrants chimiques générant des externalités négatives sur l'environnement et la santé.

Si cette nouvelle forme d'agriculture n'est soutenue publiquement par les gouvernements français que depuis 2012, elle a fait l'objet de nombreuses études scientifiques, notamment en économie. L'économie agricole s'est en effet inspirée de la littérature en économie de l'environnement et l'économie écologique pour étudier ces nouveaux enjeux. Une grande partie de cette littérature s'est attachée à déterminer la valeur des services écosystémiques en utilisant les principes de l'économie écologique. De nombreuses zones d'incertitude persistent toutefois, empêchant les politiques de promouvoir efficacement l'agroécologie ou, plus généralement, n'importe quel objectif agroenvironnemental.

En effet, quand le bien-être dépend de biens publics comme c'est le cas dans le contexte agroenvironnemental, une politique efficace (au sens de Pareto) doit idéalement implémenter les conditions de Bowen Lindahl Samuelson. Ces conditions explicitent que la fourniture efficace de biens publics est satisfaite lorsque la somme pour tous les consommateurs du taux marginal de substitution (TMS) entre chaque bien public et un bien privé choisi arbitrairement est égale au taux marginal de transformation (TMT) entre le bien public et les biens privés choisis. Le gouvernement doit implémenter des instruments afin d'atteindre cet équilibre. Toutefois, de nombreuses zones d'incertitudes persistent concernant les TMS et TMT dans le contexte agroenvironnemental. En d'autres termes, la connaissance des bénéfices et les coûts marginaux de la fourniture de biens et services agroenvironnementaux restent encore largement perfectible, en particulier à cause d'un manque de compréhension des éléments constitutifs de l'offre et de la demande pour les services écosystémiques. Le principe général de la thèse est de considérer, au contraire de la littérature sur l'évaluation monétaire des services écosystémiques, que la valeur des services écosystémiques est par nature fluctuante et qu'elle dépend essentiellement du comportement des producteurs et des consommateurs vis-à-vis des services écosystémiques. En considérant explicitement l'offre et la demande pour les services écosystémiques productifs et non-productifs, la thèse a pour but de fournir de nouvelles informations sur la gestion des services écosystémiques par les agriculteurs afin d'améliorer l'efficacité des politiques agro-environnementales.

La thèse comprend deux parties. La première partie examine la gestion des services écosystémiques productifs par les agriculteurs; i.e. les services écosystémiques sont considérés comme des intrants en technologie agricole. Dans cette première partie, on considère que les agriculteurs gèrent eux-mêmes la fourniture de services écosystémiques, c'est-à-dire qu'ils présentent eux-mêmes une demande pour maintenir un agro-écosystème de bonne qualité. Cette première partie fournit de nouvelles informations sur le coût marginal que doivent supporter les agriculteurs pour gérer les services écosystémiques (voir les conditions de Bowen Lindahl Samuelson). Dans la deuxième partie, je postule que la consommation des services environnementaux fournis par les agriculteurs présente des caractéristiques de bien public. En effet, les services environnementaux fournis par les agriculteurs contribuent à la modification de plusieurs flux de services écosystémiques (par exemple, la séquestration du carbone) impliqués dans la fourniture de biens publics divers (par exemple, la stabilité du climat). L'objectif de cette deuxième partie est d'examiner l'impact de la distribution spatiale de la demande pour les différents services écosystémiques sur l'efficacité des instruments publics. Je porte une attention particulière aux caractéristiques des biens publics fournis conjointement par les agriculteurs, notamment (i) s'il s'agit d'un bien public local ou global et (ii) de la distance entre le consommateur et la source du bien public s'il s'agit d'un bien public local. Cette deuxième partie examine certaines spécificités de la demande pour les services écosystémiques (voir les conditions de Bowen Lindahl Samuelson).

Objectifs de la première partie :

Le premier objectif de la thèse est d'examiner comment les agriculteurs gèrent les services écosystémiques pour leurs propres intérêts, en accordant une attention particulière aux aspects temporels et spatiaux de leur gestion. En effet, si nous connaissons certaines spécificités des technologies agricoles en ce qui concerne les services écosystémiques productifs, par exemple, qu'elles augmentent les rendements des cultures, aucune ne permet d'attester que les agriculteurs gèrent ces services écosystémiques consciemment et efficacement. En d'autres termes, nous ne savons pas si ces effets sont (au moins en partie) intériorisés par les agriculteurs ou s'il s'agit d'externalités pures. Ceci est en partie dû aux choix méthodologiques de la littérature existante: la productivité de la biodiversité et des services écosystémiques associés a été estimée en utilisant des équations de forme réduite, empêchant de tirer des conclusions sur le comportement des agriculteurs vis-à-vis de ces actifs. En particulier, personne ne peut conclure à une causalité entre les flux de services écosystémiques productifs et le comportement des agriculteurs. Le comportement des agriculteurs qui gèrent des services écosystémiques productifs devrait toutefois afficher des indices de cette gestion, notamment en ce qui concerne les variables de choix habituelles, telles que les applications en intrants ou les choix d'assolement. La mesure de la gestion productive des SE nécessite de spécifier ces mécanismes sous-jacents. La principale question de recherche de cette partie est donc:

(1) Les agriculteurs gèrent-ils la provision de services écosystémiques productifs afin de maximiser leur profit?

La réponse à cette question nécessite d'intégrer le comportement des agriculteurs dans la littérature sur la productivité de la biodiversité (e.g. Heisey et al., 1997 ; Di Falco & Perrings, 2003, 2005 ; Di Falco & Chavas, 2008 ; Mastushita et al., 2016, 2018). L'analyse des choix des agriculteurs en fonction des incitations économiques a été l'objet d'une grande partie de la littérature de l'économie de la production appliquée à l'agriculture. J'utilise les cadres d'analyse développés dans cette littérature pour modéliser et évaluer les choix des agriculteurs en ce qui concerne les services écosystémiques productifs. L'un des avantages de cette littérature est qu'elle décompose les choix des agriculteurs en une séquence de choix (Chambers et Just, 1989): 1) les agriculteurs optimisent le niveau d'intrants quasi-fixes à moyen terme (sur plusieurs campagnes agricoles), 2) les agriculteurs optimisent l'allocation des intrants allouables à court terme (pour une campagne agricole) en considérant les intrants quasi fixes comme fixes (et exogènes) et 3) les agriculteurs optimisent les intrants variables à très court terme (pendant une partie d'une campagne agricole) en considérant les intrants quasi fixes et ceux allouables comme fixes (et exogènes). Cette série de choix dépend des propriétés de la technologie agricole et de l'anticipation du contexte économique par les agriculteurs, c'est-à-dire l'ensemble des prix, des réglementations et des incitations publiques. Les agriculteurs optimisent différentes variables de choix à différents horizons temporels, et toutes les variables de choix sont liées par les anticipations des agriculteurs. Les nouvelles informations obtenues par l'agriculteur entre un choix et les suivantes permettent aux agriculteurs de réviser leurs prévisions et d'adapter leurs choix.

Nous profitons de cette décomposition en « très court », « court » et « moyen » termes pour examiner les choix des agriculteurs en ce qui concerne les services écosystémiques productifs sur les différentes variables de choix. À très court terme, les agriculteurs appliquent des intrants variables différemment selon les niveaux de services écosystémiques productifs si les services écosystémiques productifs et les intrants variables interagissent dans la technologie agricole. À ma connaissance, il existe peu de preuves de telles interactions sur les rendements moyens, même si Di Falco et Chavas (2006) ont souligné que les pesticides et les ES productifs, évalués à l'aide d'un indicateur de diversité biologique des cultures, interagissaient négativement sur la variance des rendements. À court terme, les agriculteurs choisissent leur allocation de culture à l'échelle de la ferme. Cette sous-littérature sur les choix d'utilisation des terres fournit une base intéressante pour l'élaboration d'un cadre unifié de la gestion des services écosystémiques productifs car la diversité culturale est considérée comme un indicateur pertinent des services écosystémiques productives. Les agriculteurs peuvent ainsi choisir leur superficie pour modifier les flux de services écosystémiques à l'échelle de leur exploitation. À ma connaissance, aucune étude existante n'a examiné un tel lien entre les choix de superficies à court terme et la productivité de la biodiversité et des services écosystémiques productifs connexes. En fait, la seule littérature empirique qui étudie le lien entre les choix de superficie et la productivité des services écosystémiques productifs est la littérature la rotation des cultures (Hendricks et al., 2014a, 2014b ; Hennessy, 2006 ; Thomas, 2003). Cependant, les services écosystémiques productifs associés à la rotation des cultures sont intrinsèquement dynamiques et n'apparaissent qu'à moyen terme. À ma connaissance, seuls Di Falco et Chavas (2008) ont expliqué que la biodiversité présente des effets productifs à court et à moyen termes. Néanmoins, Di Falco et Chavas (2008) se concentrent sur la productivité et ignorent le comportement des agriculteurs, de sorte que la gestion à moyen terme des services écosystémiques productifs à l'échelle de la ferme reste largement méconnue. Les propriétés dynamiques des services écosystémiques productifs suggèrent toutefois que la gestion des SE pourrait être similaire à la gestion du capital, un sujet qui a été intensément étudié par les économistes agricoles (Thijssen, 1996). La thèse étudie de manière théorique et empirique la gestion des services écosystémiques productifs à l'échelle de la ferme au cours des trois périodes identifiées.

Même si une telle gestion effective des services écosystémiques productifs à l'échelle de la ferme a rarement été mesurée, plusieurs travaux théoriques et de simulation l'ont supposé afin d'étudier l'impact d'instruments de politiques publiques (Baumgärtner et Quaas, 2010; Brunetti et al., 2018). En plus d'assumer une gestion à l'échelle de la ferme, certains de ces travaux théoriques ont aussi considéré que la gestion à l'échelle du paysage des services écosystémiques productifs était possible. En effet, une critique évidente de la gestion des services écosystémiques productifs à l'échelle de la ferme est que les agriculteurs ne sont pas indépendants les uns des autres. Les services écosystémiques productifs étudiés sont des biens publics qui s'étendent sur un paysage continu partagé par plusieurs agriculteurs (Zhang et al., 2007). On peut donc considérer que les agriculteurs qui gèrent des services écosystémiques productifs à l'échelle de la ferme génèrent des externalités de production pour les autres agriculteurs. La thèse étudie de manière empirique les avantages potentiels de la gestion collective des services écosystémiques productifs, en utilisant les résultats des chapitres sur la gestion à l'échelle de la ferme des services écosystémiques productifs.

La thèse apporte des réponses complémentaires à la question de recherche (1), en apportant des éléments de preuve sur la gestion agricole (chapitre 2) de différents types de SE productifs (chapitre 3), en tenant compte de leurs spécificités temporelles (chapitre 4) et spatiales (chapitre 5). Ces informations peuvent être utiles du point de vue de la mise en place d'instruments de politiques publiques.

Objectifs de la seconde partie :

La deuxième partie de la thèse a pour objectif d'étudier le rôle de la distribution spatiale de la demande pour les différents services écosystémiques jointement fournis par un service environnemental dans la conception de politiques agro-environnementales. En effet, la consommation de services environnementaux présente des caractéristiques de biens publics. Chaque service environnemental influe un ensemble particulier d'agents, allant d'agents voisins (par exemple, pollution de l'eau) à des agents du monde entier (par exemple, émission de carbone). La distribution spatiale des agents affectés dépend des propriétés des biens publics affectés par le service environnemental, c'est-à-dire s'il s'agit d'un bien public local ou mondial global et, s'il s'agit d'un bien public local, de la forme de ses impacts dans l'espace (c'est-à-dire de son effet « distance-decay »). La distribution spatiale de la demande pour les services environnementaux et les services écosystémiques non-productifs affectés devraient influencer la conception de la politique agroenvironnementale, comme le suggère la littérature sur le fédéralisme environnemental (Oates, 2001).

En effet, la littérature sur le fédéralisme environnemental considère que les différents gouvernements hiérarchiques ne sont pas tous aussi efficaces dans la conception et la mise en œuvre d'instruments de politiques publiques environnementales. La conclusion principale de cette littérature peut être résumée par le théorème de la décentralisation d'Oates (Oates, 1972): en l'absence d'externalités entre juridictions et de coûts de transaction différenciés entre les gouvernements hiérarchiques, les responsabilités fiscales devraient être décentralisées. Dans ce cas, chaque pays/région bénéficie de ses avantages informationnels (Deacon et Schläpfer, 2010; Oates, 2001) pour mieux intégrer l'hétérogénéité des goûts (Bougherara et Gaigné, 2008; Tiebout, 1956) et les conditions de production locales (Maes et al. 2012; Wolff et al., 2017). Toutefois, s'il existe des externalités intergouvernementales, comme dans le cas des biens publics globaux, les responsabilités budgétaires doivent être centralisées, chaque gouvernement générant des externalités autrement.

Si ces considérations sur la demande pour les services environnementaux sont relativement courantes en économie de l'environnement, elles sont rares en économie agricole, qui s'est principalement concentrée sur l'offre de services environnementaux. Les services environnementaux produits par l'agriculture présentent toutefois la spécificité de contribuer à plusieurs biens publics en même temps en raison des propriétés de production jointe des technologies agricoles. Cette fourniture jointe de produits agricoles et de services environnementaux est reconnue dans la littérature en économie agricole sous le concept de « multifonctionnalité » (OCDE, 2001). Ces biens publics fournis conjointement affectent les agents différemment dans l'espace, de sorte que l'application du théorème de Oates n'est pas immédiate. Cependant, dans la pratique, les incitations publiques modifiant le service environnemental des agriculteurs visent à assurer la fourniture d'un bien public particulier dans la plupart des cas. Par exemple, la France interdit régulièrement l'utilisation de pesticides spécifiques afin de réduire la pollution par les pesticides. Cependant, étant donné que plusieurs biens publics sont fournis conjointement par le service environnemental, ces incitations publiques modifient la fourniture d'autres biens publics non ciblés. Par exemple, une interdiction des pesticides pourrait entraîner une augmentation de l'application d'engrais, plusieurs études suggérant que les engrais et les pesticides sont des substituts (par exemple, Femenia et Letort, 2016). Dans cet exemple spécifique, la modification de l'application d'engrais par les agriculteurs français devrait accroître la pollution de l'eau en France, mais le gouvernement français pourrait anticiper et internaliser cet effet avec un autre instrument. En tout état de cause, si le gouvernement français maximise le bien-être social de ses citoyens, il est incité à le faire car la pollution liée à l'utilisation de pesticides et d'engrais est un bien public local. Plusieurs lobbys soulignent par ailleurs qu'une interdiction des pesticides induirait également plus d'émissions de carbone (Générations futures, 2018). Même si le gouvernement français peut anticiper ces émissions, il n'est pas incité à intérioriser l'effet dans son ensemble, car la stabilité du climat est un bien public global.

En outre, les gouvernements accordent généralement une plus grande attention aux instruments encourageant la fourniture de biens publics locaux, car dans la pratique, les biens publics locaux ont généralement une valeur marginale supérieure à celle des biens publics globaux (par exemple, Johnston et Ramachandran, 2014; Lanz et Provins, 2013; Logar et Brouwer, 2018; Schaafsma et al., 2012). Cependant, les biens publics globaux peuvent avoir une incidence sur l'efficacité de nombreux instruments de politiques environnementales visant des biens publics locaux, car ces instruments modifient la production localement mais aussi indirectement le commerce mondial. En effet, une réglementation locale plus stricte en matière de pesticides pourrait réduire la production alimentaire locale, ce qui pourrait être partiellement compensé par une augmentation des importations, ce qui inciterait d'autres localités à augmenter leur production agricole. Cette augmentation induit en particulier des changements dans l'utilisation des terres, mettant les terres telles que les forêts en production agricole et entraînant une augmentation des émissions de carbone dans ces régions/pays (Searchinger et al., 2008). Ces effets influencent l'efficacité de l’intervention des différents gouvernements hiérarchiques (Harstad et Mideksa, 2017). Bien que ces effets induits soient bien connus de la littérature sur le changement climatique, les études empiriques mesurant ces effets induits par les réglementations locales font défaut. Ces effets induits pourraient toutefois être particulièrement prononcés dans le secteur agricole, car les produits agricoles ont également la particularité de se substituer relativement bien les uns aux autres sur les marchés mondiaux (Hertel, 2002; Peeters et Surry, 1997).

Ces caractéristiques sont susceptibles de modifier les conditions d’application habituelles des conditions de Bowen Lindahl Samuelson, qui ne sont pas vérifiées dans le cadre de l'équilibre général impliqué par les échanges commerciaux et si certaines externalités jointes sont ignorées. La principale question de recherche de cette partie peut être formulée comme suit:

(2) Quelle est l'influence de la distribution spatiale de la demande pour les services environnementaux sur la conception des politiques agro-environnementales?

J'étudie cette question à l'aide de plusieurs méthodes et étudie différents types de services environnementaux, allant du changement d'affectation des sols (chapitres 7, 8 et 9) à la réduction des applications chimiques (chapitre 7) et à la réduction de la densité animale (chapitre 8). En particulier, je simule deux types d'incitations financières: une taxe *ad valorem* sur les pesticides (chapitre 7) et des subventions basées sur l'utilisation des terres (chapitre 9). J'utilise des fonctions de demande explicites pour des biens publics globaux (chapitre 9) et locaux (chapitres 8 et 9) lorsque cela est possible. En cas de manque d'informations, je quantifie la fourniture de biens publics locaux et globaux en se référant uniquement implicitement aux fonctions de demande (chapitre 7). Ces trois chapitres apportent des réponses complémentaires à la question de recherche (2). Ces informations peuvent être utiles du point de vue de la conception de politiques agroenvironnementales.

Plan de la thèse:

La thèse est organisée en deux parties principales avec un total de onze chapitres, y compris cette introduction générale, deux discussions et un chapitre de conclusion. Il s'appuie sur une revue de littérature (qui peut devenir un article d'opinion) et six articles de recherche préparés au cours de la thèse. La première partie, composée des chapitres 2 à 6, traite de la gestion des services écosystémiques productifs par les agriculteurs eux-mêmes et peut être considérée comme une étude détaillée de l'offre agro-environnementale. La deuxième partie, composée des chapitres 7 à 10, traite de la distribution spatiale pour la demande des biens publics fournis par les agriculteurs qui gèrent les services écosystémiques non-productifs et peut être considérée comme une étude détaillée de la demande agro-environnementale. Ces enquêtes détaillées permettent de mieux comprendre les spécificités des conditions de Bowen Lindahl Samuelson dans le cas de la gestion agricole des services écosystémiques.

PREMIÈRE PARTIE: LES SERVICES ÉCOSYSTÉMIQUES, UNE APPROCHE PAR L'ÉCONOMIE DE LA PRODUCTION

Le chapitre 2, intitulé « Microéconomie, biodiversité et services écosystémiques: revue de la littérature et cadre général », propose un modèle structurel pour comprendre la gestion par les agriculteurs des services écosystémiques productifs. Ce modèle théorique repose sur les connaissances et les principes de modélisation de trois littératures complémentaires: la littérature sur la productivité de la biodiversité et des services écosystémiques productifs associés, la littérature sur les indicateurs de biodiversité et la littérature sur les choix de production des agriculteurs (notamment la sous-littérature sur les choix d'assolement). Je discute de l'implication d'un tel rapprochement pour la littérature utilisant les modèles habituels de choix de production des agriculteurs, à savoir les applications d'intrants variables et les choix de superficie. J'explique que le modèle proposé correspond à une généralisation des modèles de choix d'assolement existants. Je donne les implications d'un changement marginal de surface d'une culture pour l'ensemble des rendements, d'applications d'intrants et de coûts de gestion des intrants fixes (travail et capital) en fonction de l'assolement initial. Le modèle proposé contribue à la littérature sur la productivité de la biodiversité en distinguant les avantages (rendements supplémentaires et économies d'intrants variables) et les coûts (réorganisation du capital et du travail) associés à une gestion productive des services écosystémiques productifs. Je présente les trois chapitres suivants (chapitres 3 à 5) comme des extensions du chapitre 2.

Le chapitre 3, coécrit avec Pierre Dupraz (SMART-LERECO, INRA) et intitulé « Capacité productive de la biodiversité dans les exploitations mixtes du nord-ouest de la France: un système primal à plusieurs productions », examine les choix à très court terme des agriculteurs pour fournir des estimations détaillées du rôle des services écosystémiques productifs dans la multi-technologie agricole. Ce travail se démarque des travaux existants sur la productivité de la biodiversité pour deux raisons. Premièrement, nous examinons deux indicateurs d'habitats de biodiversité, à savoir la diversité des cultures et les prairies permanentes, et examinons leurs impacts sur les rendements en céréales et en lait. Les informations sur la productivité des prairies permanentes sont cruciales dans la mesure où peu d'études s'y intéressent alors que ces habitats sont considérés comme les plus riches de la planète en terme de diversité d'espèces (Wilson et al., 2012). Deuxièmement, nous accordons une attention particulière aux interactions productives entre les habitats de la biodiversité et les intrants chimiques. Nous estimons trois modèles complémentaires détaillants les différentes manières de modéliser les intrants variables en fonction qu'ils soient publics ou privés et en fonction qu'ils interagissent avec les services écosystémiques productifs ou non. Les résultats contribuent à la littérature sur la productivité de la biodiversité en fournissant des productivités marginales détaillées et non constantes de la biodiversité. Nous estimons nos différents modèles en utilisant la méthode des moments généralisés (GMM) avec effets fixes individuels sur un panel non-cylindré d'agriculteurs de 999 agriculteurs du Nord-Ouest de la France (Basse Normandie, Bretagne et Pays de la Loire) de 2002 à 2015.

Nos résultats montrent que les deux types de composantes de la biodiversité augmentent les rendements céréaliers et laitiers mais qu'ils présentent des propriétés d'intrants non-coopératifs. Nous montrons aussi que ces deux composantes sont non-coopératifs avec les fertiliseurs et les pesticides. Les effets les plus importants concernent la diversité culturale, mettant en lumière le peu d'incitations qu'ont les agriculteurs à maintenir les prairies permanentes. Nous montrons aussi qu'il est nécessaire de prendre en compte le comportement d'optimisation des agriculteurs dans le très court terme, l'absence d'instrumentation des applications d'intrants variables conduisant à une surestimation de 100% des paramètres associés à la productivité des services écosystémiques productifs. Ce chapitre a conduit à la publication d'un document de travail présenté lors de quatre congrès nationaux et internationaux: (i) le 149ème séminaire de l'Association européenne d'économie agricole (EAAE) à Rennes (France) en 2016, (ii) la 10ème

conférence annuelle des journées de la recherche en sciences sociales INRA-SFER-CIRAD à Paris (France) en 2016, (iii) les 34èmes conférences des Journées de microéconomie appliquée (JMA) au Mans (France) en 2017 et (iv) le 15ème congrès de l'EAAE à Parme (Italie) en 2017.

Le chapitre 4, coécrit avec Elodie Letort (SMART-LERECO, INRA) et intitulé « Comment les agriculteurs gèrent-ils la biodiversité culturale ? Un modèle de choix d'assolement dynamique avec rétroaction productive » aborde l'estimation du modèle structurel proposé au chapitre 2 dans un cadre dynamique. Ce travail contribue à la littérature sur la productivité de la biodiversité en fournissant la preuve que les agriculteurs gèrent la biodiversité culturale en optimisant de manière dynamique les choix d'assolement. En effet, l'identification d'une gestion productive des services écosystémiques productifs à court terme est un défi car tous les choix de production observés au cours de la même année peuvent être considérés comme simultanés. La prise en compte des effets productifs dynamiques à moyen terme facilite l'identification économétrique de cette gestion dès lors qu'ils sont anticipés. Nous estimons alors un modèle inspiré de la théorie de l'investissement où nous considérons la biodiversité comme un capital naturel qui se déprécie et où les agriculteurs investissent chaque année via leurs choix d'assolement. Pour ce faire, nous utilisons la méthode des GMM sur un échantillon d'agriculteurs de la Meuse. La biodiversité est approximée en utilisant la diversité culturale, en particulier l'indice de Shannon. Nous montrons ainsi que les agriculteurs anticipent les futurs effets productifs de la biodiversité. Si la dynamique des écosystèmes est la même que dans l'étude italienne de Di Falco et Chavas (2008), alors les effets anticipés semblent complètement intégrés. Près de deux tiers des effets productifs de l'écosystème sont toutefois liés au niveau courant de l'indicateur de biodiversité. Contrairement au reste de la littérature qui a examiné ces effets de manière agrégée, nous sommes capables de décomposer ces effets sur plusieurs produits (blé, orge et colza) et plusieurs éléments constitutifs des marges brutes (applications d'engrais, applications de pesticides et rendements) dans le court comme dans le long terme. Ces effets sont résumés dans la figure 4.3.

Table 4.3. Elasticités des rendements, des demandes d'intrants variables et des marges brutes relativement à la biodiversité

	Short-term				Long-term			
	Mean	**SD.**	**Min**	**Max**	**Mean**	**SD.**	**Min**	**Max**
yield_wheat_biodiversity	0.03	0.01	0.02	0.04	0.05	0.01	0.03	0.06
yield_barley_biodiversity	0.07	0.02	0.04	0.11	0.11	0.02	0.06	0.16
yield_rapeseed_biodiversity	0.01	0.001	0.005	0.02	0.01	0.002	0.007	0.02
pesticides_wheat_iodiversity	-0.28	0.05	-0.49	-0.17	-0.40	0.07	-0.70	-0.24
pesticides_barley_biodiversity	-0.14	0.02	-0.21	-0.08	-0.20	0.03	-0.30	-0.12
pesticides_rapeseed_biodiversity	-0.41	0.07	-0.65	-0.21	-0.58	0.10	-0.93	-0.30
fertilizer_wheat_biodiversity	-0.08	0.02	-0.14	-0.04	-0.11	0.02	-0.20	-0.05
fertilizer_barley_biodiversity	-0.06	0.01	-0.10	-0.03	-0.08	0.02	-0.14	-0.04
fertilizer_rapeseed_biodiversity	-0.05	0.01	-0.09	-0.02	-0.07	0.01	-0.13	-0.03
gross_margins_wheat_biodiversity	0.10	0.08	0.04	2.16	0.14	0.11	0.06	3.09
gross_margins_barley_ biodiversity	0.12	0.03	0.07	0.29	0.18	0.04	0.10	0.41
gross_margins_rapeseed_ biodiversity	0.13	0.03	0.06	0.29	0.18	0.04	0.08	0.41

Ce chapitre illustre également l'intérêt de notre cadre d'analyse par rapport aux modèles de choix d'assolement habituels, dans lesquels les services écosystémiques productifs ne sont pas modélisés, lors de la simulation d'une taxe *ad valorem* sur les pesticides. La prise en compte des services écosystémiques productifs dans les technologies agricoles multi-produits montre que les agriculteurs peuvent en fait plus facilement s'adapter à la taxe que ce qui avait été précédemment estimé. Ce chapitre a été publié dans l'*European Review of Agricultural Economics* (HCERES rang A, CNRS rang 2) et a été présenté lors de quatre congrès et workshops nationaux et internationaux: (i) le 34ème congrès annuel de la JMA au Mans (France) en 2017, (ii) le 23ème congrès de la Association européenne d'économie de l'environnement et des ressources (EAERE) à Athènes (Grèce) en 2017, (iii) du 15ème congrès de l'EAAE à Parme (Italie) en 2017 et (iv) de la première école de printemps de GREEN-ECON à Marseille (France) en 2018.

Le chapitre 5, coécrit avec Hugues Boussard (BAGAP, INRA) et Claudine Thenail (BAGAP, INRA) et intitulé « Services écosystémiques productifs et gestion collective: enseignements tirés d'un modèle de paysage réaliste », développe l'idée que les choix d'assolement des agriculteurs ne sont pas indépendants des choix des agriculteurs voisins. Nous examinons ici l'intérêt qu'auraient les agriculteurs à coordonner leurs choix d'assolements à l'échelle du paysage. Ce chapitre utilise la fonction de densité de carabes spatialement explicite déterminée dans Martel et al. (2017) et les résultats obtenus dans les deux chapitres précédents pour mesurer la rentabilité de la lutte biologique fournie par les carabes. Nous comparons les bénéfices individuels issus de la gestion individuelle et coordonnée des services écosystémiques productifs générés par les carabes, en considérant à la fois des agriculteurs hétérogènes et des paysages réalistes sur une zone d'un kilomètre de diamètre. Nous montrons, comme le reste de la littérature considérant des agriculteurs homogènes, que les gains collectifs sont d'en moyenne 4%. Toutefois, nos résultats montrent que les bénéfices individuels de la coordination sont très hétérogènes et dépendent des conditions initiales. En particulier, l'introduction de l'hétérogénéité des agents remet en cause la propriété de stabilité interne dans 85% des cas alors qu'elle était toujours respectée lorsque les agents étaient considérés homogènes. Nous concluons donc que l'apparition d'une gestion coordonnée des services écosystémiques productifs dans des paysages réels est peu probable. Ce chapitre a été présenté lors du congrès national organisé par le réseau PAYOTE à Paris (France) en 2017.

Le chapitre 6 présente les résultats des chapitres 2 à 5 en relation avec la question de recherche de la première partie de la thèse. Il souligne également les limites des analyses proposées et fournit quelques suggestions pour des recherches futures. J'ai combiné la littérature sur la productivité des services écosystémiques productifs avec la littérature sur le comportement microéconomique des agriculteurs pour prouver que les agriculteurs géraient effectivement la fourniture de services écosystémiques productifs. La littérature sur le comportement microéconomique des agriculteurs utilise les choix observés des agriculteurs, notamment l'application d'intrants variables et les choix d'assolement, pour déterminer les réactions des agriculteurs vis-à-vis des incitations économiques. S'appuyant sur les avantages de la première littérature, qui approxime la biodiversité à des indicateurs basés sur l'utilisation des terres, le chapitre 2 propose un modèle théorique unifié qui spécifie le comportement des agriculteurs en ce qui concerne les services écosystémiques productifs. Les chapitres 3 à 5 sont des travaux empiriques dans lesquels j'estime différentes versions de ce modèle théorique. À l'aide d'un cadre dynamique, le chapitre 4 montre que les agriculteurs gèrent la biodiversité et les services écosystémiques productifs connexes pour tirer parti de leurs effets productifs. Sur la base du comportement observé des agriculteurs, les chapitres 3 et 4 fournissent également de nouvelles informations sur la productivité de différents types de composantes de la biodiversité pour une série de résultats désagrégés, y compris des interactions détaillées avec des intrants chimiques. Ces résultats suggèrent que les services écosystémiques productifs soutenus par la biodiversité des cultures en exploitation et les prairies permanentes (i) bénéficient différemment aux différents produits agricoles, (ii) sont des substituts aux pesticides et aux engrais et (iii) ont des effets productifs dynamiques. Enfin, au chapitre 5, j'ai étudié les effets de la gestion coordonnée des services d’écosystèmes productifs, ce qui est une stratégie prometteuse pour augmenter les profits individuels selon la littérature. A partir d’un modèle de paysage réaliste avec des agriculteurs hétérogènes, nos résultats indiquent que, si une gestion coordonnée conduit en moyenne à des gains collectifs et individuels, ces gains sont relativement limités et inégalement répartis entre les agriculteurs coordonnés, ce qui limite l'émergence de stratégies de gestion coordonnées dans des paysages réels. J'espère que la décomposition et la formulation des modèles proposés inspirerons d'autres recherches dans ce domaine, notamment sur la gestion des risques, question que je n'ai pas prise en compte en dépit des nombreuses discussions sur la valeur d'assurance de la biodiversité et des services écosystémiques productifs connexes.

DEUXIÈME PARTIE: LES SERVICES ENVIRONNEMENTAUX, UNE APPROCHE PAR L'ÉCONOMIE DE L'ENVIRONNEMENT

Le chapitre 7, coécrit avec Alexandre Gohin (SMART-LERECO, INRA) et intitulé « Simulation des impacts sur le marché et sur l'environnement des politiques françaises en matière de réduction de l’utilisation des pesticides: une évaluation macroéconomique », mesure les effets induits de l'introduction d'un système de taxation des pesticides en France sur les changements d'affectation des sols dans le monde et sur les émissions de carbone associées. Ces effets induits affectent négativement l'utilité de la localité où l'instrument initial est mis en œuvre car les émissions de carbone contribuent au changement climatique mondial. Cette évaluation, la première sur les pesticides à ma connaissance, illustre les impacts environnementaux induits de la politique environnementale locale lorsqu'un bien public mondial est conjointement fourni par l'agriculture. Le document est construit en deux étapes: (1) nous estimons le modèle microéconométrique des choix de production de Carpentier et Letort (2014) sur les régions françaises en utilisant une entropie maximale généralisée, et (2) nous utilisons les élasticités estimées pour simuler les effets de marché et la GCU avec modèle d'équilibre général GTAP-Agr pour l’année 2011.

Nous apportons trois contributions. La première est d’estimer les comportements des agriculteurs sur l’ensemble du territoire français alors que la majorité de la littérature n’utilise qu’une unique région. Nous estimons le modèle pour chacune des régions françaises sur les données des comptes économiques de l’agriculture de 1991 à 2015. La deuxième est de s’intéresser à l’ensemble des activités agricoles et non seulement aux fermes céréalières comme c’est le cas dans la majorité de la littérature (Böcker & Finger, 2017). Nous trouvons que l'élasticité prix agrégée française de la demande pour les pesticides est de -0,82, en accord avec Fadhuile et al. (2017). En particulier, l'élasticité prix de la demande est de -0,30 pour les céréales, ce qui correspond à l'élasticité médiane estimée dans la littérature sur les échantillons de fermes céréalières (Böcker & Finger, 2017). Notre troisième contribution est de simuler les effets induits de deux politiques publiques visant à réduire l'utilisation de pesticides en France. Nous trouvons par exemple qu'une taxe de 50% sur le prix des pesticides réduirait la consommation française de pesticides de 37% mais engendrerait des émissions de gaz à effet de serre équivalents à 10% des émissions actuelles de l'agriculture française, principalement à cause de la déforestation dite « importée ». Elle engendrerait par contre une augmentation de l'utilisation de fertiliseurs de 5% due à la substitution à la marge intensive des cultures intensives en pesticides vers les cultures moins intensives en pesticides mais plus intensives en fertiliseurs. Nous trouvons aussi que cette taxe n'aurait qu'un effet de -5% sur les revenus des agriculteurs. Les pertes affectent bien sûr les fermes céréalières mais un tiers de ces pertes affectent les élevages français. Cet exercice souligne les différents arbitrages que doit résoudre le décideur public. Le scénario dit technique simulant une innovation induite par la politique (comme c'est espéré pour les plans Ecophyto) résout tous ces arbitrages mais ne peut apparaitre que dans le long terme. Ce document a été présenté à la réunion annuelle de l'Association de l'agriculture et de l'économie appliquées (AAEA) à Washington D.C. (États-Unis) en 2018.

Le chapitre 8, coécrit avec Abdel Osseni (SMART-LERECO, INRA) et Pierre Dupraz (SMART-LERECO, INRA) et intitulé « Découpler les valeurs des externalités agricoles selon l'échelle: une approche hédonique spatiale en Bretagne », développe une analyse de la valorisation hédonique spatiale permettant d'évaluer de manière monétaire les impacts des activités agricoles sur la fourniture de différents biens publics via l'étude du prix des maisons. Nous contribuons à la littérature sur la valorisation hédonique des externalités agricoles en distinguant la valeur à deux échelles différentes: l'échelle infra-municipale (où les habitants et les activités agricoles sont situées dans la même commune) et à l'échelle l'extra-municipale (où les résidents et les activités agricoles sont situées dans différentes municipalités). En effet, des études antérieures avaient généralement estimé la fonction hédonique à une seule échelle spatiale, soit à l'échelle infra-municipale (Bontemps et al., 2008; Le Goffe, 2000), soit à une échelle inférieure (Cavailhès et al., 2009; Ready et Abdalla, 2005), ignorant que les biens publics locaux pourraient avoir un impact sur les résidents à une plus grande échelle, comme le suggère des études soulignant l'effet « distance-decay ».

Notre analyse théorique suggère que, comme chaque activité fournit conjointement plusieurs biens publics, certaines activités peuvent générer une externalité négative (positive) à l'échelle infra-municipale, mais une externalité positive (négative) à une plus grande échelle. A partir des données sur la vente de 3000 maisons bretonnes entre 2010 et 2012, nous montrons que les principales interactions spatiales dans notre jeu de données sur les maisons bretonnes sont dues à des effets de diffusion des externalités et non à de l'hétérogénéité spatiale ou à la diffusion de prix. Pour cela, nous réalisons les tests du multiplicateur de Lagrange sur les modèles linéaires et spatiaux (SAR, SEM, SARAR, SLX, SDM, SDEM et GNS). Nous montrons que les activités liées à l'élevage de bovins génèrent des effets négatifs sur le prix des maisons à l'échelle infra-municipale mais positifs à l'échelle extra-municipale. Nous montrons que les activités liées à l'élevage de porcins et de volailles génèrent des effets négatifs à toutes les échelles. Les résultats de ce chapitre ont contribué au projet PROVIDE H2020 sur la fourniture de biens publics par l'agriculture et la foresterie européennes. Ce chapitre a contribué à l'élaboration d'un document de travail présenté lors de trois conférences nationales et internationales: (i) la 35ème conférence annuelle des JMA à Bordeaux (France) en 2018, (ii) le 6ème Congrès international d'économie agricole (ICAE) à Vancouver (Canada) en 2018 et (iii) la réunion annuelle de l'AAEA à Washington DC (États-Unis) en 2018.

Le chapitre 9, coécrit avec Matteo Zavalloni (Université de Bologne) et intitulé « Décentralisation de la conception des politiques agroenvironnementales: le cas des zones humides abandonnées en Bretagne », analyse l'intérêt de la décentralisation de la conception des politiques agroenvironnementales suggérée par la Commission européenne pour la prochaine réforme de la PAC. Inspirés par la littérature sur le fédéralisme environnemental, nous examinons les avantages d'une telle réforme en utilisant un modèle dans lequel (i) une économie est composée de régions homogènes, (ii) l'agriculture produit conjointement des biens publics locaux et globaux sur les mêmes terres, à un coût marginal croissant, et (iii) les gouvernements locaux intègrent l'hétérogénéité des valeurs des biens publics locaux à l'intérieur de leurs frontières, mais négligent l'impact des biens publics globaux sur le bien-être d'autres régions. Alors que les chapitres 7 et 8 informent sur les externalités générées par une juridiction sur les autres lors de la prise en compte des biens publics locaux et mondiaux, le chapitre 9 examine directement la politique optimale pour améliorer le bien-être de la société dans son ensemble. Nous analysons d'abord théoriquement le problème et, dans un deuxième temps, nous paramétrons le modèle théorique au cas de la gestion des zones humides en Bretagne (France) sur la base des résultats de PROVIDE WP4. Nous montrons que les gains de la décentralisation sont principalement dus à une réorientation des subventions vers les terres ayant le plus de valeurs et à une diminution du budget agro-environnemental. Les gains d'une telle réforme pourraient atteindre 67% dans le cas particulier des zones humides agricoles. Les résultats de ce chapitre ont contribué au projet PROVIDE H2020. Ce document a été présenté lors de trois conférences nationales et internationales: (i) la 11ème conférence des journées de recherche en sciences sociales INRA-SFER-CIRAD à Lyon (France) en 2017, (ii) le 6ème Congrès mondial de l'économie de l'environnement et des ressources (WCERE) à Göteborg (Suède) en 2018 et (iii) au 6ème congrès de l'ICAE à Vancouver (Canada) en 2018.

Le chapitre 10 discute les résultats des chapitres 7 à 9 en relation avec la question de recherche de la deuxième partie de la thèse, à savoir si la distribution spatiale de la demande de services environnementaux affectait la conception optimale des instruments (agro)environnementaux. En d'autres termes, la deuxième partie a analysé les spécificités de la demande de services environnementaux fournis par les agriculteurs. Pour cela, j'ai introduit le principe selon lequel les agriculteurs qui gèrent des agro-écosystèmes influencent conjointement la provision de plusieurs biens publics locaux et globaux dans trois littératures couramment utilisées en économie de l'environnement: la littérature sur le lien entre le commerce et la qualité environnementale, la littérature sur l'évaluation par la méthode des prix hédoniques et la littérature sur le fédéralisme environnemental. Au chapitre 7, j'introduis cette propriété dans un modèle général d'équilibre général calculable afin d'étudier les effets induits de la réglementation d'un bien public local sur la fourniture de biens publics globaux. Appliqué au cas de la réduction des applications de pesticides, qui génère de nombreux types de pollution locale (touchant non seulement la santé mais également les biens publics environnementaux), je souligne que, si une taxe de 50% sur le prix des pesticides pouvait réduire les applications nationales de 37%, les effets de marché ont entraîné l'émission de 9 millions de tonnes d'équivalent CO_2 dans d'autres régions du monde. Ces émissions, dues principalement aux changements d'affectation des sols dans d'autres pays et en particulier à la déforestation dans certains pays d'Amérique latine, sont égales à 10% des émissions de carbone effectives du secteur agricole français. Au chapitre 8, j'ai introduit le principe selon lequel les agriculteurs qui gèrent des agroécosystèmes produisent conjointement des biens publics locaux avec des distributions spatiales différentes selon les modèles de prix hédoniques habituels. En utilisant les connaissances tirées de la littérature sur le « distance-decay », j'explique que les agriculteurs qui gèrent des agro-écosystèmes génèrent des externalités de formes complexes dans l'espace. En utilisant des méthodes économétriques spatiales, les résultats montrent que même si l'essentiel de la valeur des externalités est capturé à l'intérieur de la municipalité où la production a lieu, les effets de « distance-decay » liés aux biens publics locaux fournis conjointement affectent également le bien-être des municipalités voisines. Par exemple, si les activités porcines présentent des effets négatifs à toutes les échelles, les activités liées aux bovins laitiers, y compris la gestion des pâturages, présentent des effets négatifs dans la municipalité où la production a lieu mais des effets positifs dans les municipalités voisines. Enfin, le chapitre 9 s'inspire de la littérature sur le fédéralisme environnemental. J'y ai introduit le principe selon lequel les agriculteurs qui gèrent des agroécosystèmes produisent conjointement des biens publics locaux et globaux afin d'étudier l'efficacité de la décentralisation de la conception de la politique agroenvironnementale. J'ai considéré en particulier que les gouvernements hiérarchiques présentent différents niveaux d'informations sur la demande de biens publics globaux et locaux fournis conjointement, ce qui influe sur le degré optimal de décentralisation. Ces trois chapitres soulignent que même si l'essentiel de la valeur d'un service environnemental est capturé localement, la demande de services environnementaux émanant de zones plus vastes et plus éloignées influence le bien-être social et donc la conception de la politique optimale. Le chapitre 10 souligne également les limites des analyses proposées et fournit quelques suggestions pour des recherches futures.

CONCLUSION

Le manuscrit se termine par le chapitre 11, qui contient des remarques finales et un résumé des principaux résultats de la thèse. Il fournit également des recommandations politiques basées sur les deux parties de la thèse. En particulier, j'y explique que la politique agroenvironnementale optimale consisterait à appliquer le principe de Tinbergen, avec autant d'instruments que de biens publics visés. Pour cette raison, je soutiens que le soutien à une forme spécifique d'agriculture n'est pas optimal. Par exemple, le soutien public aux transitions agroécologiques présenté au chapitre 7 souligne que les bénéfices environnementaux dus à la réduction des applications de pesticides sont réduits par les émissions mondiales induites. Cette caractéristique est commune à l'agriculture biologique: elle améliore la fourniture de certains biens publics mais réduit celle d'autres biens publics. Débattre du type d'agriculture à soutenir est un non-sens pour les économistes, qui préfèrent le débat sur le type de biens publics que souhaite la société. Par exemple, au lieu de subventionner directement l'agroécologie dans le but de réduire les applications de pesticides, le régulateur devrait encourager les agriculteurs à réduire les applications de pesticides en ciblant directement les applications de pesticides (par exemple, grâce à un système de taxation des pesticides): comme souligné aux chapitres 2 à 5, un agriculteur qui maximise ses profits se tournera de plus en plus vers des pratiques plus agro-écologiques. Les chapitres 7 à 9 montrent toutefois à quel point la question de savoir quels biens publics la société veut est déjà complexe. En effet, chaque gouvernement hiérarchique dispose d'informations différentes sur la demande de biens publics fournis conjointement par le service environnemental. Les administrations locales disposent de meilleures informations sur la demande de biens publics locaux, ce qui encourage la décentralisation mais génère également des externalités vers d'autres juridictions, provenant soit du commerce (chapitre 7), soit de la production conjointe de biens publics locaux et mondiaux (chapitres 7 à 9), ce qui encourage la centralisation. J'ai conclu au chapitre 9 que même si cela ne conduisait pas à une situation Pareto optimale, les gouvernements nationaux sont les gouvernements les plus aptes à concevoir des politiques agroenvironnementales. Le chapitre 7 montre toutefois que, si le commerce et la production jointe ne sont pas pris en compte, une intervention nationale pourrait quand même avoir des effets inattendus dans la localité où l'instrument est initialement mis en œuvre. J'espère que ces travaux, ainsi que la littérature croissante sur le rôle de la distance dans l'évaluation de la valeur des biens publics locaux, encourageront les chercheurs à intégrer davantage la distribution spatiale de la demande pour les différents biens publics lors de l'analyse des multiples dimensions de l'efficacité des instruments agro-environnementaux.

La promulgation récente d'incitations économiques visant à soutenir des pratiques agricoles plus respectueuses de l'environnement a entraîné un nombre croissant de discussions au sein de la société. C'est le cas en France, où le développement de l'agroécologie et les incitations proposées ont donné lieu à de nombreux débats entre divers acteurs: agriculteurs et lobbys industriels, lobbys environnementalistes et décideurs, pour n'en nommer que quelques-uns. En tant que doctorant en économie, je souhaitais contribuer à ces débats en explorant les concepts de services écosystémiques et de services environnementaux du point de vue économique, en mettant l'accent sur l'offre et la demande de biens publics influencés par les flux de services écosystémiques. En approfondissant les conditions de Samuelson dans ce cas particulier, j'espère que ces travaux contribueront à améliorer l'efficacité des politiques agroenvironnementales. Je suis déjà ravi que certaines conclusions de cette thèse aient été intégrées dans la note de synthèse du projet PROVIDE H2020 adressée à la Commission européenne. Je suis encore plus heureux que mes travaux aient contribué à la création d'un paiement pour services environnementaux en Bretagne afin de soutenir la préservation des caractéristiques du paysage agricole breton traditionnel telles que les zones humides agricoles et les haies. Ces deux exemples illustrent les raisons pour lesquelles un économiste devrait davantage effectuer des recherches participatives. Quoiqu'il en soit, si j'ai été capable de formuler des recommandations politiques, la complexité du fonctionnement de l'agroécosystème nécessite des recherches supplémentaires sur les nombreuses spécificités sous-jacentes de l'offre et la demande de services environnementaux avant que quelqu'un puisse réellement atteindre la condition de Samuelson correspondante.

Contents

List of Tables

List of Figures

CHAPTER 1: GENERAL INTRODUCTION

1.1 Background: The farmer, the agroecosystem and the society

1.1.1 Ecosystem services and environmental services: definitions

Agriculture remains the main land-intensive economic activity, occupying 37.5% of world lands and 54.7% of European Union lands in 2014 (63.1% in France).[1][2] These levels make agriculture in charge of the management of a large part of the Earth's ecosystems. In essence, the job of the farmer has always been to deal with all the components of the ecosystem to benefit from its best potentialities. Using the vocabulary popularized by the Millennium Ecosystem Assessment (MEA, 2005), farmers have to manage ecosystem services (ES), i.e., "the benefits people obtain from ecosystems" (MEA, 2005).[3] Using the MEA classification, farmers have developed practices to benefit from supporting and regulating services, which, for example, reduce pest damages, increase pollination or improve soil fertility (Zhang et al., 2007). These productive ES, which can be considered as flows of ecological functionalities (Haines-Young and Potschin, 2010), have always been part of the agricultural technology. A famous example of agricultural practice relying on productive ES is the triennial rotation. This practice, implanted in West Europe during the Middle Ages, introduces a spring cereal or legume crop into the two-year winter cereal-fallow rotation, creating a three-year rotation pattern that modifies nutrient cycles to enhance soil fertility (Federico, 2005).

In addition to managing ES for their own interest, farmers have contributed to the expression of other ES, notably, using the MEA classification, provisioning services (e.g. production of energy), regulating services (e.g. carbon sequestration) and cultural services (e.g. participation to recreational activities). These goods and services are jointly produced with agricultural goods but are consumed by non-farming agents. For example, the agriculture from the beginning of the 20th century managed plots and associated semi-natural elements (hedgerows in Brittany – Malassis, 2001), ensuring the provision of woods and provided habitats for wild game that contributed to the energy, food and leisure of the population. Most of these benefits are non-

[1] https://donnees.banquemondiale.org/indicateur/NV.AGR.TOTL.ZS?locations=1W
[2] https://data.worldbank.org/indicator/AG.LND.AGRI.ZS
[3] More accurately, the management of ecosystem potentialities (which can be defined by a vector of ecosystem functionalities) by economic agents leads to benefits for people (including the manager). I will use the term "ecosystem services" to refer, though inaccurately, to the ecosystem functionalities and processes. This etymological use is realized by most studies referring to specific processes of the ecosystem influencing agents' welfare, even without explicit references to the benefits for people. I provide a deeper discussion on that point in Chapter 2.

marketable, as the consumption of these goods presents public good characteristics. These public goods can be considered as specific stocks that are produced by an ecological production function depending on the ES flows over time. These non-marketable benefits have led stakeholders to refer to agriculture as a "multifunctional" economic activity (Cooper et al., 2009; OECD, 2001). Although the "multifunctional agriculture" framework has been developed in parallel to the "ecosystem service" framework, these two concepts are intrinsically linked, notably through the payment for environmental services (PES) literature (Engel et al., 2008; Wunder, 2005). This literature expressly recognizes that farmers can improve the utility of other agents through the modification of the ES provision that is implicated in the production of public goods. In this framework, the modification of ES flows by an agent is an "environmental service" (Engel et al., 2008). The value of the environmental service depends on the variation of the stocks of public goods induced by the modification of the ES flows. The PES literature studies the effectiveness of economic incentives in internalizing these environmental services. Thus, the recent literature recognizes that farmers use ES to produce agricultural goods and environmental services. In economic terms, the productive ES are inputs of agricultural technology, whereas environmental services are outputs of agricultural technology. Inspired by Zhang et al. (2007), Figure 1.1. summarizes the links between ES and agricultural production. In the thesis, I consider that the flows going "to" agricultural ecosystems are the productive ES, while the flows going "from" agricultural systems are the environmental services contributing to public goods (the public outputs) and private agricultural goods. The dotted line represents the influence of farmers' practices on the flows inside the agricultural ecosystems so that they modify the productive ES flows. While environmental services by definition present public good characteristics, productive ES can present public or private good characteristics. ES present public good characteristics if the flows of productive ES generated by one farmer influence the profitability of other farmers (e.g., through biological control or pollination), while they present private good characteristics if not (e.g., soil fertility).

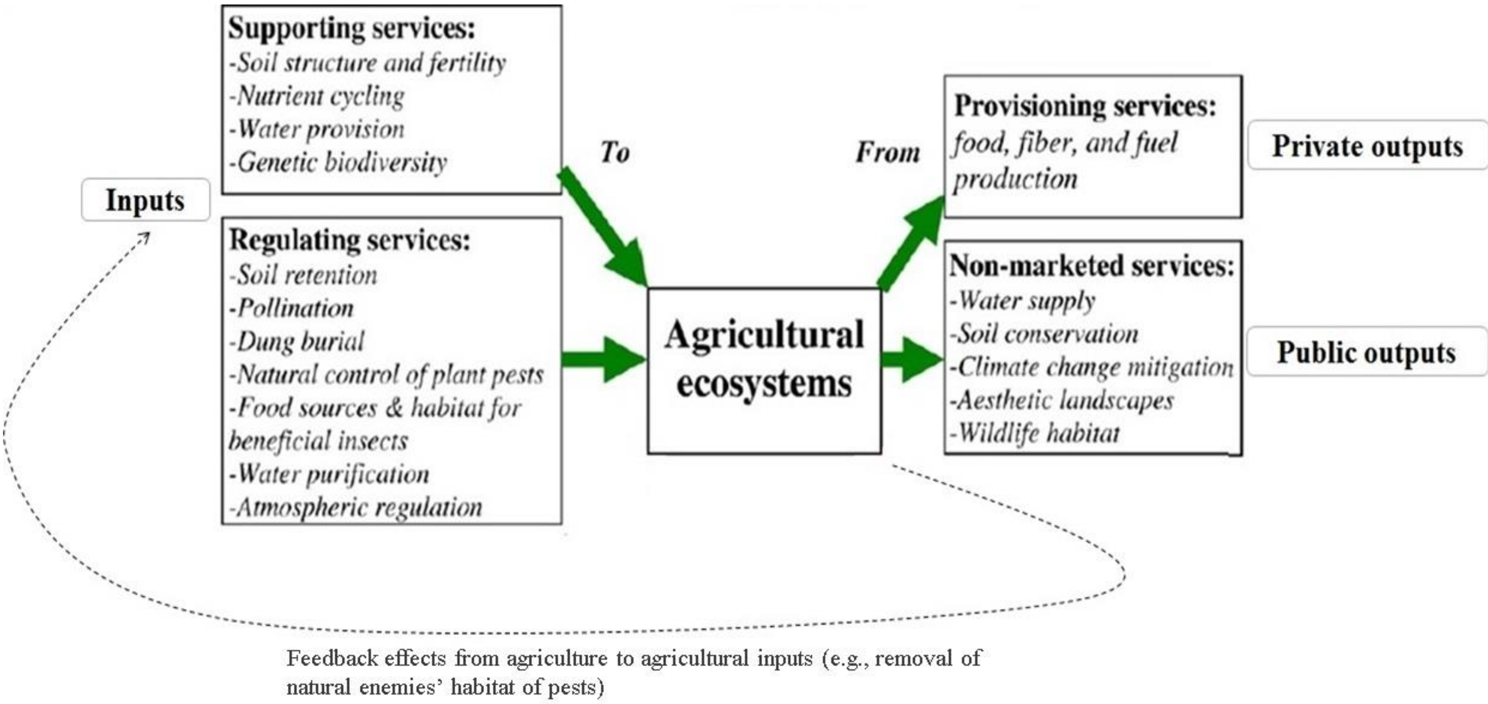

Figure 1.1. Ecosystem services in agricultural ecosystems (*Source: inspired by Zhang et al., 2007*)

1.1.2 Ecosystem services, environmental services and public incentives

Since the end of the Second World War, agriculture has modernized towards objectives of food safety and economic growth. In France, these goals were reached in the sixties, when the country became the first food exporter of the world. The increase in food production has required a deep transformation in the production process, notably with the intensification of the use of chemical and capital inputs, to the detriment of the use of productive ES. As evidence of these modifications, the worldwide sale of pesticides multiplied by 30 between 1960 and 2000 (Agrios, 2005). These evolutions have been encouraged by agricultural commodity price support and storage policies as well as by favourable market conditions with relatively low prices of chemical and capital inputs compared to the constantly increasing labour price (Manuelli and Seshadri, 2014). These market and policy contexts have benefitted from the profitability of the agriculture of the mid-20th century.

However, modern agricultural practices have degraded the provision of several regulating and cultural ES that are valued by our societies (Sutton et al., 2011; TEEB, 2010). For example, estimations on biodiversity evolution worldwide have yielded a reduction of 20% between 1970 and 2000 (Butchart et al., 2010). In Europe, the common farmland bird indicator shows a decline of 55% between 1980 and 2015 (PECBMS, 2017).[4] Although the objectives were not guided by the reduction of the negative externalities generated by the modernization of agriculture but rather by budget concerns, policymakers have shifted agricultural support from price support towards payments that are decoupled from yields and production choices. In addition, the successive reforms of the Common Agricultural Policy (CAP) in 1992, 1999, 2003 and 2008 have progressively abandoned production quotas and generalized agricultural and environmental conditions to obtain direct farm support. These policies have relinked agricultural production with world market prices, leading European farmers to be more subject to low output prices. The progressive increase in energy prices has also increased variable input prices. At the same time, policymakers have offered subsidies to finance the agricultural provision of environmental services through the development of large-scale policies (e.g., the generalization of agro-environmental schemes since the early 1990s) or more local ones (e.g., the "water plan" of Munich to improve water quality – Grolleau and McCann, 2012). These policy programmes were optional from the farmers' perspective. In 2003, the CAP reform introduced compulsory cross-compliance with agricultural and environmental conditions to obtain direct farm support. The CAP greening reform of 2014 also introduced green payments with additional conditions, for example, regarding minimum acreage diversity. Other policies have focused more directly on input prices, notably to reduce the consumption of polluting inputs by farmers (e.g., pesticide taxes). These market and policy contexts have decreased the profitability of the agricultural systems and practices inherited from the 20th century.

The decline in agricultural profitability and the increase in societal concerns about agricultural externalities define a new context for agriculture in the 21st century. This context influences the policy and professional debates, notably in France, where numerous debates on the future of agriculture have been held since the Grenelle Environment Forum in 2007 and even more deeply during the COP 21 that led to the Paris agreement in 2016. These debates have defined a new challenge for agriculture: reconciling agricultural production with its ecosystems to improve environmental quality and, if possible, agricultural profitability. This new path for agriculture is known as "agroecology" in France, which promotes agricultural production systems that use productive ES more intensively.[5] Agroecology received deep institutional

[4] Data available on the website of the Pan-European Common Bird Monitoring Scheme at the following address: http://www.ebcc.info/index.php?ID=639
[5] This definition is broader than the scientific definition of "agroecology", which is rather a discipline popularized by agronomists and ecologists (e.g. Altieri, 2018). Dalgaard et al. (2003) defined this discipline as the study of the interactions between plants, animals, humans and the environment within agricultural systems. The political and

support during Stephane Le Foll's mandate as the French agricultural minister between 2012 and 2017.

A growing number of agricultural stakeholders have supported this institutional objective. For example, several farmer associations have developed action programmes to improve the utilization of productive ES (e.g., the BASE or the "Bleu-Blanc-Coeur" associations).[6] Several cooperatives have designed their development strategies around the promulgation of agroecology (e.g., Terrena, Triskalia, "Fermiers de Loué"), and even some food industries have tried to guarantee their provision of raw material from "agroecological" products (e.g., Nestlé, LU). More recently, the major French farmers' union (FNSEA) accepted the objective to reduce pesticide use by 50% in 2025, in agreement with the French government programme Ecophyto II.

1.1.3 Agroecology and science: trends in the economic literature

The institutional promulgation of the agroecology objectives of environmental quality and agricultural profitability have also been the subject of much research in agronomy and economics. In particular, the growing attention paid by economists to environmental and agro-environmental issues has required the transformation of previously existing approaches. Indeed, contrary to the physiocrats or the classical economists, the neoclassical economists had not explicitly recognized the link between economic activity and environment (or nature) before the environmental crisis at the beginning of the seventies (Gómez-Baggethun et al., 2010). Based on the report of the "Club of Rome" initially published in 1968 (Meadows et al., 1972), the new environmental economics sub-field of economics emerged: it initially aimed to study the effects of environmental externalities on the welfare of our societies and to identify the most efficient instruments to internalize them. In parallel, a part of the initial environmental economics has focused more on the coevolution of ecosystems and societies, constituting a separate sub-field of economics: ecological economics. This sub-field has developed interdisciplinary works to analyse the dependence of agents on nature (attesting the use of the concept of "natural capital" – Missemer, 2018), notably based on the works of Nicholas Georgescu-Roegen (Missemer, 2015), Robert Costanza (e.g., Costanza et al., 1997) or Herman

societal sense of "agroecology" stands on the scientific principles of agroecology as well as on other forms of agriculture, such as conservation farming.

[6] See https://asso-base.fr/ and https://www.bleu-blanc-coeur.org/#!

Daly (e.g., Daly, 1991). Environmental and ecological economists have participated in the renewing of agricultural economics.

Since the 1980s, agricultural economists have shown interest both in the costs and benefits of externalities generated by agriculture and in the dependence of agriculture on natural processes. This evolution is notably visible in the top-review journals of the field (e.g., American Journal of Agricultural Economics), where a growing number of studies on the environmental impacts of agriculture (e.g., Caswell et al., 1990), agro-environmental policies (e.g., Plantinga, 1996; Wu and Segerson, 1995), the links between agriculture and ecology (Bockstael, 1996; Lazarus and Dixon, 1984; Orazem and Miranowski, 1994) or, more generally, environmental economic debates, such as environmental monetary valuation (e.g., Bockstael and Kling, 1988; Espinosa and Smith, 1995; Smith et al., 1986), have been published.

In particular, a growing research agenda has measured the effects of productive ES on agricultural yields and profits (Chavas and Di Falco, 2012; Di Falco and Chavas, 2006, 2009; Di Falco et al., 2010; Donfouet et al., 2017; Heisey et al., 1997; Matsushita et al., 2017; Smale et al., 1998). As some ES are unobservable, this literature has measured ES using biodiversity habitat indicators developed by landscape ecologists. Most of the literature has focused on the productivity and profitability of on-farm crop diversity, considering that these indicators are proxies of productive ES bundles (i.e., of the jointly provided productive ES). It appears that on-farm crop diversity increases the means of crop yields and reduces their variances, contributing to the idea that crop diversity has both productive and insurance values (Baumgärtner, 2007; Chavas, 2009). Some refinements have been developed following these results; for example, Di Falco and Chavas (2008) found that these effects are dynamic and depend on weather conditions. More recently, the literature has found similar productive effects from semi-natural elements. For example, Klemick (2011) found that upstream forest fallows increase mean crop yields, and Finger and Buchmann (2015) found that grasslands decrease yield variance. Whereas these papers empirically measured the productivity or profitability of on-farm biodiversity, implicitly considering productive ES as private inputs, more recent works have also examined the potential benefits of coordinated management of ES at the landscape scale using simulation techniques and considering productive ES as inputs with public good characteristics (Atallah et al., 2017; Cong et al., 2014; Epanchin-Niell and Wilen, 2014).

The integration of the ES concept into scientific and political debates also corresponds to reflections on the role of agricultural policies, particularly the CAP, for the provision of environmental services (Lefebvre et al., 2015; Van Zanten et al., 2014). The PES literature has notably questioned the effectiveness of CAP agro-environmental measures (AEM) because AEM design focuses on opportunity costs but does not integrate environmental benefits (Engel et al., 2008). The integration of environmental benefits into the AEM design leads to the question of additionality, which appears to be rather limited in AEMs (Chabé-Ferret and Subervie, 2013). The lack of additionality has led agricultural economists to investigate alternatives, such as spatial targeting (Fooks et al., 2016; Wünscher et al., 2008), minimum participation rules (Dupraz et al., 2009; Wu and Skelton-Groth, 2002) and agglomeration bonus (Bamière et al., 2013 ; Wätzold and Dreschler, 2014), the two last alternatives being motivated by scale issues in the provision of environmental services. The principle of this scale issue is that, as uncoordinated efforts of dispersed contractors over space lead to low additionality, the regulator should encourage farmers to group their efforts over space (Lewis et al., 2009).

However, scale issues also arise for the consumption of the jointly provided public goods by the environmental service, with public goods being either local or global (Atkinson and Stiglitz, 1980). Local public goods affect the agents' utility around the locality of provision, whereas global public goods affect the agents' utility all around the world. The demand for a specific environmental service thus depends on the geographical scale of the demand for each of additional stock of public goods provided by the ES flows from the considered environmental service. In addition, the distance to the source of the local public good matters for its beneficiaries. This question has been investigated in the "distance-decay willingness-to-pay" literature with the valuation of agricultural externalities over space (Ay et al., 2016; Jørgensen et al., 2013; León et al., 2016). The issue of the scale of the demand for environmental services questions the design of existing agro-environmental policies (Beckmann et al., 2009; Ogawa and Wildasin, 2009; Sigman, 2005).

Despite the growing number of studies on the interactions between agriculture, its ecosystem and society, many challenges remain to understand the development of "agroecology", which appears to be multiform and complex (Therond et al., 2017). Such challenges are crucial for the optimal design of agro-environmental policy.

1.2 Efficient agro-environmental policy in theory

Indeed, as previously underlined, the agricultural management of ecosystems modifies the provision of goods and services with public good characteristics, i.e., non-rivalry and non-excludability. Non-rivalry is the possibility for several agents to consume the same good. Non-excludability refers to the situation where no one can be excluded from consuming the good. Actually, this means that the cost to prevent somebody from benefiting from the public good is higher than the social value of the good itself. In the case of a Pareto-optimal economy with only private goods, the social benefit of the last unit of a private good is equal to the change in the welfare of the person who receives this last unit. With the usual assumptions of perfectly informed price-takers, complete markets with no transaction costs and local nonsatiation of preferences, the first theorem of welfare states that competitive markets tend towards an efficient allocation of resources in the Pareto sense. However, in the case of public goods, the market is not able to lead to a Pareto-optimal allocation because several agents consume the same goods, which leads to incomplete markets (Greenwald and Stiglitz, 1986). This market failure opens the door for government intervention, which implements suitable instruments to reach Pareto equilibrium.

The optimal provision of public good is based on the work of Samuelson (1954). The Samuelson condition, also known under the name of Bowen Lindahl Samuelson, states that the efficient provision of public goods is satisfied when the sum of the marginal rate of substitution between each public and an arbitrarily chosen private good for all consumers is equal to the marginal rate of transformation between the public and the chosen private good. In the specific case where the private good is a numeraire, the Samuelson condition becomes:

$$\sum_{i=1}^{I} U_m^i = C_m \tag{1.1}$$

where U_m^i is the marginal utility of consumer *i* and C_m is the marginal cost to provide the public good. Relation (1.1) states that the optimal provision of a public good is reached when the sum of the marginal benefits of I consumers equals the marginal cost of the public good provision. Relation (1.1) represents the usual optimal condition in the case of public goods, where the marginal social benefit of providing the public good is equal to the sum of the marginal benefits received by all people. The purpose of the government is to implement instruments to encourage the public good producers to provide (more) public goods (or less public bads in the case of pollution, for example) and thus to tend towards relation (1.1).

Relation (1.1) offers a simple framework to optimize policy intervention. However, the optimal conditions depend on the providers' cost and consumers' utility functions, which could be rather complex depending on the considered public good. This is the case for agroecosystems, whose quality can be considered as an input of a vector of public goods, while the properties of the cost and benefit functions remain largely unknown and uncertain.

Regarding the supply side of agro-environmental goods, the cost function in relation (1.1) can be considered dependent on several components of agro-environmental conditions. Indeed, as explained earlier, the ecosystem provides productive ES that can be profitable for the farmers so that farmers constitute a part of the demand for the ES bundles. The profit-maximizing farmers will thus manage the ES bundles differently according to the different conditions of prices and regulations, suggesting that the corresponding cost functions in (1.1) are non-linear, with potentially many local minima. There remain many uncertainties regarding the profitable properties of agroecosystems, for instance, with regard to the interactions of productive ES with chemical inputs. The productive interactions between the ES bundles and chemical inputs can lead, for example, to a complex marginal cost function depending on the relative prices of the different chemical inputs. These uncertainties imply that the right-hand side of relation (1.1) is subject to many doubts. The first part of the thesis focuses on the specificities of the profitable properties of productive ES and their potential consequences on the marginal cost to maintain an agro-ecosystem of good quality in relation (1.1).

Regarding the demand side of agro-environmental goods, the utility function in relation (1.1) is subject to many uncertainties. First, the issue of identifying U_i per se is difficult, even if the literature on valuation has developed for more than sixty years (Smith, 2004). This feature is notably due to the joint production of public goods because the management of agroecosystem quality contributes to the joint provision of several goods and services with public good characteristics. The utility function U_i thus depends on the value for the different public goods and on the relative individual preferences for the different public goods. Last but not least, it can be complex to identify the *I* consumers affected by the improvement of the agroecosystem quality, notably with regard to the geographical scale of the affected agents (Bateman et al., 2006). The second part of the thesis focuses on the specificities of the geographical scales of the demand for agro-environmental management of agroecosystems and their potential consequences on the utility function to maintain an agro-ecosystem of good quality in relation (1.1).

Improving the knowledge about these marginal cost and utilities functions is crucial to improve public intervention regarding the agro-environment.

1.3 Objectives and research questions

The PhD thesis aims to provide new insights into farmers' management of ES to improve the efficiency of agro-environmental policies. It is assumed that agriculture is an economic activity that uses productive ES with other inputs to jointly produce agricultural commodities and environmental services. The thesis consists of two parts. The first part examines the farmers' management of productive ES; i.e., the ES are considered inputs in agricultural technology. In this first part, farmers are considered to manage the provision of ES for themselves; i.e., the farmers themselves present a demand to maintain an agro-ecosystem of good quality. This first part provides new insights into the marginal cost that farmers face when managing ES (see relation (1.1)). In the second part, I postulate that the consumption of the environmental services provided by the farmers has public good characteristics. Indeed, the environmental services provided by farmers contribute to the modification of several ES flows (e.g., carbon sequestration) that are involved in the provision of diverse public goods (e.g., climate stability). The research objective of this second part is to examine the impact of the geographical scale of the demand for environmental services from agriculture on the agents' utility and on the efficiency of public instruments. In particular, I pay attention to the characteristics of the jointly provided public goods, with regard to the (i) local or global public good nature and (ii) the distance to the source of ES for the local public goods. This second part examines some specificities of the demand for agro-environmental goods and services (see relation (1.1)).

1.3.1 First part: ecosystem services, a production economics approach

The first objective of the thesis is to examine how farmers manage ES for their own interests, with specific attention to the temporal and spatial aspects of their management. Indeed, if we know some specificities of agriculture technologies with regard to productive ES, for instance, that they increase crop yields, there is a lack of evidence that farmers effectively manage productive ES. In other words, we do not know whether these effects are (at least partly) internalized by the farmers or whether they are pure externalities. This is partly due to the methodological choices of the existing literature: the productivity or the profitability of biodiversity and related productive ES has been estimated using reduced-form equations (yields or profit), preventing the derivation of any conclusions on the farmers' behaviour with regard to these inputs/assets. In particular, this prevents us from concluding any causality between productive ES flows and farmers' behaviour. The behaviour of the farmers managing productive ES should, however, display indices of such management, notably with regard to usual choice variables, such as input applications or acreage allocation. In particular, farmers managing productive ES make decisions according to both the benefits and the cost of productive ES, i.e., the demand and the supply for productive ES. The measurement of productive ES management requires specifying these underlying mechanisms. The main research question of this part is thus:

(1) Do farmers manage productive ES?

The answer to this question requires integrating the farmers' behaviour into the literature on the productivity of biodiversity. The modelling of farm scale operations and farmers' choices has been the purpose of the production economics literature applied to agriculture. I take advantage of this literature to model and assess farmers' choices with regard to productive ES at the farm scale. One advantage of this literature is that it decomposes the farmers' choices into a sequence of choices (Chambers and Just, 1989): 1) farmers optimize the level of quasi-fixed inputs in the medium term (on several agricultural campaigns), 2) farmers optimize the allocation of allocable input in the short term (for one agricultural campaign) considering quasi-fixed inputs as fixed (and exogenous) and 3) farmers optimize variable input in the very short term (during a part of one agricultural campaign) considering quasi-fixed and allocable inputs as fixed (and exogenous). This series of choices depends on the properties of the agricultural technology and on the farmers' anticipation of the economic context, i.e., the set of prices, regulations and public incentives. Farmers optimize different choice variables at different time horizons, and all choice variables are linked by the farmers' anticipations. The new information obtained by the farmer between one choice and the following allows farmers to revise their anticipation and adapt their choices.

We take advantage of this decomposition in the "very short", "short" and "medium" terms to examine farmers' choices with regard to productive ES on the different choice variables. In the very short term, farmers apply variable inputs differently according to the levels of productive ES if productive ES and variable inputs present productive interactions. To my knowledge, there is a lack of evidence of such interactions on mean yields, even if Di Falco and Chavas (2006) have emphasized that pesticides and productive ES, assessed with a crop biodiversity indicator, interact negatively on variance yields. In the short term, farmers perform acreage allocation by crop at the farm scale. This sub-literature on land-use choices provides an interesting basis for the development of a unified framework on productive ES management because land-use diversity is considered a relevant indicator of productive ES. Farmers can thus choose their acreage to modify the productive ES flows at their farm scale. To my knowledge, no existing study examines such a link between short-term acreage choices and the productivity of biodiversity and related productive ES in the short term. In fact, the single empirical research stream that studies the link between acreage choices and productivity of productive ES is the scarce literature on crop rotation (Hennessy, 2006; Thomas, 2003). However, the productive ES associated with crop rotation are inherently dynamic and appear only in the medium term. To my knowledge, only Di Falco and Chavas (2008) have explained that biodiversity presents productive effects in both the short and the medium terms. Nevertheless, Di Falco and Chavas (2008) focus on productivity and ignore farmers' behaviour, such that the management of productive ES at the farm scale in multiple campaigns remains overlooked. The dynamic properties of productive ES suggest, however, that ES management may be similar to the management of capital, a topic that has been intensely studied by agricultural economists (Thijssen, 1996). The present PhD thesis theoretically and empirically investigates the management of diverse productive ES at the farm scale in the three identified periods.

Even if such farm-scale management has rarely been measured,[7] several theoretical and simulation works have assumed such farm-scale management to investigate the impact of public incentives or alternative management on ecosystem and biodiversity evolution (Baumgärtner and Quaas, 2010; Brunetti et al., 2018). In addition to assuming farm-scale management, some of these theoretical works have considered that the landscape-scale management of productive ES is possible. Indeed, one obvious criticism of the farm-scale management of productive ES is that farmers are not independent from each other. The studied productive ES are public goods that spread over a continuous landscape, which is shared by several farmers (Zhang et al., 2007). One can thus consider that farmers managing productive ES at the farm scale generate productive externalities for other farmers. The present PhD thesis empirically investigates the potential advantage of collective management of productive ES, using the results obtained from the chapters on the farm-scale management of productive ES.

[7] To my knowledge, only Di Falco et al. (2014) has provided evidence of such management, although in the case of risk management, i.e. based on variance yields. I am not aware of any work providing evidence of productive ES management based on mean yields.

The PhD thesis will provide complementary answers to the research question (1), providing evidence on the agricultural management (Chapter 2) of different types of productive ES (Chapter 3) considering their temporal (Chapter 4) and spatial (Chapter 5) specificities. These insights may be valuable from a policy perspective.

1.3.2 Second part: ecosystem services, an environmental economics approach

The second objective of the PhD thesis is to investigate the role of the geographical scale of the demand for environmental services in the design of agro-environmental policies. Indeed, the consumption of environmental services presents characteristics of public goods. Each environmental service influences a particular set of agents, ranging from neighbouring agents (e.g., water pollution) to agents around the world (e.g., carbon emission). The scales of the affected agents depend on the properties of the produced public good, i.e., whether it is a local or a global public good and, if it is a local one, on its range of impacts over space (i.e., on its "distance decay"). The scale issues in the demand for environmental services should influence the agro-environmental policy design, as suggested by the environmental federalism literature (Oates, 2001).

Indeed, the environmental federalism literature considers that hierarchical governments are not equally efficient in implementing environmental instruments. This literature studies the effectiveness of the decentralization vs. the centralization of environmental policy design. Its main conclusion can be summarized in Oates' decentralization theorem (Oates, 1972): in the absence of interjurisdictional externalities and differentiated transaction costs between hierarchical governments, fiscal responsibilities should be decentralized. In this case, each jurisdiction benefits from its informational advantages (Deacon and Schläpfer, 2010; Oates, 2001) to better integrate the heterogeneity of tastes (Bougherara and Gaigné, 2008; Tiebout, 1956) and local production conditions (Maes et al., 2012; Wolff et al., 2017). However, if there are interjurisdictional externalities, such as in the case of global public goods, the fiscal responsibilities should be centralized, each government generating externalities otherwise.

If these considerations are relatively common in environmental economics, they are scarce in the agricultural economics literature, which has primarily focused on the supply of environmental services. The environmental services produced by agriculture present, however, the specificity to contribute to several public goods at the same time due to the property of the joint production of agricultural technologies. This joint provision of agricultural goods and environmental services is recognized in the agricultural economics literature under the concept of “multifunctionality” (OECD, 2001). These jointly provided public goods affect agents differently over space, such that the application of Oates’ decentralization theorem is not straightforward. However, in practice, the public incentives modifying the environmental service of the farmers are targeted to insure the provision of one public good, in most cases.[8] For example, France regularly introduces bans on specific pesticides to reduce pesticide pollution. However, because several public goods are jointly provided by the environmental service, these public incentives modify the provision of other non-targeted public goods. For example, a ban on pesticides could lead to an increase in fertilizer application, as several studies suggest that fertilizer and pesticides are substitutes (e.g., Femenia and Letort, 2016). In this specific example, the modification of French farmers’ fertilizer application should increase water pollution in France, but the French government could anticipate and internalize this effect with another instrument. In any case, if the French government maximizes the social welfare of its citizens, it has the incentive to do this because pollution linked to pesticide and fertilizer use is a local public good. Several lobbies emphasize that a ban on pesticide would also induce more carbon emissions (Generation futures, 2018). Even if the French government can anticipate these emissions, it does not have the incentive to internalize the entire effect because carbon emission is a global public good.

In addition, governments usually pay deeper attention to instruments encouraging local public good provision, as in practice, local public goods usually have a higher marginal value than global public goods (e.g., Johnston and Ramachandran, 2014; Lanz and Provins, 2013; Logar and Brouwer, 2018; Schaafsma et al., 2012). However, global public goods can impact the effectiveness of numerous environmental policies, even in the locality where they are implemented, as such policies modify production locally but trade worldwide. Indeed, stricter local regulation of pesticides may reduce local food production, which may be partially compensated by increased imports, providing incentives for other localities to increase agricultural production. This increase especially induces land-use changes, putting lands such as forests into agricultural production and leading to increased carbon emissions in these localities (Searchinger et al., 2008). These effects influence the effectiveness of the hierarchical governments (Harstad and Mideksa, 2017). While these “leakage” effects are well known in the literature on climate change, empirical studies measuring such effects induced by local

[8] In practice, due to informational issues, the public incentives are designed based on the farmers’ efforts rather than on the public good outcomes (White and Hanley, 2016).

regulation are missing. These leakage effects could, however, be particularly pronounced in agriculture, as agricultural goods also have the specificity of being relatively good substitutes for each other on global markets (Hertel, 2002; Peeters and Surry, 1997).

These features are susceptible to modifying the usual Samuelson conditions, which are not verified under the general equilibrium implied by trade and if some joint externalities are ignored. The main research question of this part can be formulated as follows:

(2) What is the influence of the geographical scale of the demand for environmental services on the design of agro-environmental policies?

I investigate this question using several methods and consider different types of environmental services, ranging from land-use change (Chapters 7, 8 and 9) to the reduction of chemical applications (Chapter 7) and the reduction in animal density (Chapter 8). In particular, I simulate two types of financial incentives: a pesticide *ad valorem* tax (Chapter 7) and subsidies based on land use (Chapter 9).[9] I use explicit demand functions on global (Chapter 9) and local public goods (Chapters 8 and 9) when possible. In the case of a lack of information, I quantify the provision of local and global public goods with only implicit reference to demand functions (Chapter 7). These three chapters provide complementary answers to research question (2). These insights may be valuable from a policy perspective.

1.4 Outline of the PhD thesis

The thesis is organized into two main parts with a total of eleven chapters, including this general introduction, two discussions and a concluding chapter. It relies on one literature review (which may become an opinion paper) and six research articles that have been prepared during the course of the PhD. The first part, composed of Chapters 2 to 6, addresses the management of productive ES by the farmers themselves and can be considered a detailed investigation of the agro-environmental supply. The second part, composed of Chapters 7 to 10, addresses the geographical scale of the demand for the public goods that are provided by the farmers managing ES and can be considered a detailed investigation of agro-environmental demand.

[9] These two instruments are not equivalent, taxes being usually considered more efficient than subsidies because there is no moral hazard and/or adverse selection (Laffont and Tirole, 1993; White and Hanley, 2016). However, taxes are more difficult to implement in practice due to societal discontent, which explains why much of environmental policy relies on subsidies.

These detailed investigations provide some insights into the specificities of the Samuelson conditions in the case of the agricultural management of ES.

PART ONE: ECOSYSTEM SERVICES, A PRODUCTION ECONOMICS APPROACH

Chapter 2, entitled "Microeconomics, biodiversity and ecosystem services: literature review and general framework", offers a structural model to understand farmers' management of productive ES. This theoretical model is based on three complementary research streams: the literature on the productivity of ES, the literature on biodiversity indicators and the literature on farmers' production choices (notably the sub-literature on acreage choices). I discuss the implication of such a framework regarding usual models of farmers' production choices, namely, the variable input applications and the acreage choices. I explain that the proposed model corresponds to a generalization of existing acreage models. The proposed model contributes to the literature on the productivity of biodiversity by distinguishing the benefits (additional yields and variable input savings) and the costs (reorganization of capital and labour) associated with productive ES management. I present the three following chapters (Chapters 3 to 5) as extensions of Chapter 2.

Chapter 3, entitled "Biodiversity productive capacity in mixed farms of northwest France: a multi-output primal system", considers the farmers' very short-term choices to provide detailed estimations of the role of productive ES in agricultural technology. This work stands out among the other papers on the productivity of biodiversity for two reasons. First, we consider two indicators of biodiversity habitats, namely, crop diversity and permanent grasslands, and examine their impacts on cereals and milk yields. Second, we pay special attention to the productive interactions between biodiversity habitats and chemical inputs. The results contribute to the literature on the productivity of biodiversity by providing detailed non-constant marginal productivities of biodiversity. This chapter has conducted to the publication of a working paper, which has been presented in four national and international congresses: (i) the 149th seminar of the European Association of Agricultural Economics (EAAE) in Rennes (France) in 2016, (ii) the 10th annual conference of INRA-SFER-CIRAD social science research days in Paris (France) in 2016, (iii) the 34th Journées de Microéconomie Appliquée (JMA) conference in Le Mans (France) in 2017 and (iv) the 15th congress of the EAAE in Parma (Italy) in 2017.

Chapter 4, entitled “How do farmers manage crop biodiversity? A dynamic acreage model with productive feedbacks” addresses the estimation in a dynamic framework of the structural model proposed in Chapter 2. This work contributes to the literature on the productivity of biodiversity by providing evidence that farmers manage crop biodiversity by dynamically optimizing acreage choices. Indeed, the identification of productive ES management is challenging in the short term because the production choices in a given year can be considered simultaneous. The consideration of the dynamic productive effects in the medium term eases the econometric identification of such management. This chapter also illustrates the interest of our framework compared to usual farmers’ choice models, where productive ES are ignored, when simulating a pesticide *ad valorem* tax. The consideration of productive ES in agricultural technology highlights that farmers have more potential to adapt to the tax. This chapter has been published in the European Review of Agricultural Economics and has been presented in four national and international congresses and workshops: (i) the 34th JMA annual congress in Le Mans (France) in 2017, (ii) the 23rd congress of the European Association of Environmental and Resources Economics (EAERE) in Athens (Greece) in 2017, (iii) the 15th congress of the EAAE in Parma (Italy) in 2017 and (iv) the 1st spring school of GREEN-ECON in Marseille (France) in 2018.

Chapter 5, entitled “Productive ecosystem services and collective management: lessons from a realistic landscape model”, develops the idea that farmers’ acreage choices are not independent from their neighbouring farmers’ choices. It examines farmers’ interests in the coordinated management of their acreage choices at the landscape scale. The chapter utilizes the spatially explicit carabid beetle density function determined in Martel et al. (2017) and the results obtained in the two previous chapters to measure the profitability of biological control provided by carabid beetles over space. We compare the individual profits emerging from the individual and coordinated management of productive ES generated by carabid beetles considering both heterogeneous farmers and realistic landscapes. This chapter contributes to the literature on the collective management of productive ES by considering heterogeneous agents and more realistic ecological functioning. Contrary to existing empirical studies that focus on homogenous agents, our results question the respect of the internal stability and shed doubts on the possible emergence of coordinated management in real landscapes. This chapter was presented in the national workshop organized by the PAYOTE network in Paris (France) in 2017.

Chapter 6 discusses the results obtained from Chapters 2 to 5 in relation to the research question of the first part of the PhD thesis. It also underlines the limits of the proposed analyses and provides some suggestions for future research.

PART TWO: ENVIRONMENTAL SERVICES, AN ENVIRONMENTAL ECONOMICS APPROACH

Chapter 7, entitled "Simulating the market and environmental impacts of French pesticide policies: a macroeconomic assessment", measures the induced effects of the introduction of a pesticide taxation scheme in France on land-use change (LUC) worldwide and on related carbon emissions. These induced effects negatively affect the utility of the locality where the initial instrument is implemented because carbon emissions contribute to global climate change. This assessment, the first on pesticide to my knowledge, illustrates the induced environmental impacts of local environmental policy when a global public good is jointly provided by agriculture. The paper is constructed in two steps: (1) we estimate Carpentier and Letort's (2014) microeconometric model of production choices on French regions using generalized maximum entropy, and (2) we use the estimated elasticities to simulate the market effects and LUC with the general equilibrium model GTAP-Agr. This paper was presented in the Agricultural and Applied Economic Association (AAEA) annual meeting in Washington D.C. (USA) in 2018.

Chapter 8, entitled "Decoupling values of agricultural externalities according to scale: a spatial hedonic approach in Brittany", develops a spatial hedonic pricing analysis to monetarily assess the impacts of agricultural activities on residents' utilities. We contribute to the literature on the hedonic valuation of agricultural externalities by distinguishing the value at two different scales: the infra-municipal scale (where the residents and the agricultural activities are located in the same municipality) and the extra-municipal scale (where the residents and the agricultural activities are located in different municipalities). Indeed, previous studies have usually estimated the hedonic function at a single spatial scale, either the municipal scale (Bontemps et al., 2008; Le Goffe, 2000) or a lower scale (Cavailhès et al., 2009; Ready and Abdalla, 2005), ignoring that local public goods could impact residents at a broader scale, as suggested by the literature on distance-decay willingness-to-pay. Our results suggest that because each activity jointly provides several public goods, some activities could generate negative (positive) externality at the infra-municipal scale but positive (negative) externality at a larger scale. The results of this chapter contributed to the PROVIDE H2020 project on the provision of public goods by European agriculture and forestry.[10] This chapter has contributed to one working paper, which has been presented at three national and international conferences: (i) the 35th JMA annual conference in Bordeaux (France) in 2018, (ii) the 6th International Congress of Agricultural Economics (ICAE) in Vancouver (Canada) in 2018 and (iii) the AAEA annual meeting in Washington D.C. (USA) in 2018.

Chapter 9, entitled "Decentralization of agri-environmental policy design: the case of abandoned wetlands in Brittany", analyses the interest in the decentralization of the design of Agri-Environmental Policies as suggested by the European Commission for the next CAP reform. Inspired by environmental federalism literature, we examine the gains of such reform using a model in which (i) an economy is composed of homogeneous regions, (ii) agriculture produces local and global public goods jointly on the same lands at an increasing marginal cost and (iii) local governments integrate the heterogeneity of local public good values inside their boundaries but neglect the impact of global public goods on the welfare of other regions. While Chapters 7 and 8 inform on the externalities generated by one jurisdiction on the others when considering local and global public goods, Chapter 9 directly examines the optimal policy to improve the welfare of the whole society. We first theoretically analyse the problem, and in a second step, we parameterize the theoretical model to the case of wetland management in Brittany (France) based on PROVIDE WP4 results. The results of this chapter contributed to the PROVIDE H2020 project. This paper has been presented in three national and international conferences: (i) the 11th conference of INRA-SFER-CIRAD social science research days in Lyon (France) in 2017, (ii) the 6th World Congress of the Environmental and Resources Economics (WCERE) in Gothenburg (Sweden) in 2018 and (iii) the 6th congress of ICAE in Vancouver (Canada) in 2018.

Chapter 10 discusses the results obtained from Chapters 7 to 9 in relation to the research question of the second part of the PhD thesis. It also underlines the limits of the proposed analyses and provides some suggestions for future research.

[10] PROVIDE is the acronym of "PROVIding smart DElivery of public goods by EU agriculture and forestry". It is financed by the European Commission under the grant agreement n°633838. Additional information is available at http://www.provide-project.eu/.

The manuscript ends with Chapter 11, which draws some concluding comments along with a summary of the main findings of the thesis. It also provides policy recommendations based on these two parts of the PhD thesis.

1.5 References

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PART ONE:

ECOSYSTEM SERVICES, A PRODUCTION ECONOMICS APPROACH

The aim of this part is to investigate the farmers' management of productive ecosystem services (ES). In Chapter 2, I develop a theoretical framework inspired by two economic literatures to model the microeconomic behaviour of farmers effectively managing productive ecosystem services. In Chapter 3, I assume that the productive ecosystem services are exogenous in the very short term to examine their properties in the agricultural technology. In particular, I examine the productive interactions of two types of biodiversity components, which are assumed to support the provision of productive ES, with conventional chemical inputs. In Chapter 4, I estimate the theoretical model of Chapter 2 in a dynamic framework and provide evidence that farmers manage, at least partly, productive ES. In Chapter 5, I investigate the benefits of the coordinated management of productive ES using a simulation framework with heterogeneous farmers and a realistic landscape. Finally, Chapter 6 presents the general discussion of the results of Chapters 2 to 5.

CHAPTER 2. MICROECONOMICS, BIODIVERSITY AND ECOSYSTEM SERVICES: LITERATURE REVIEW AND GENERAL FRAMEWORK

The aim of this chapter is to translate agroecological principles into the usual mircroeconomic models of farmers' behaviour. This chapter presents a general framework in which farmers manage the provision of productive ecosystem services (ES) based on land-use allocation. I review the literature and discuss the interest in combining literature streams to theoretically link farmers' behaviour and productive ES. In particular, I consider that the introduction of biodiversity indicators based on acreage choices into agricultural production functions allows us to measure the potential productive ES provided, a notion that I summarize in the term "productive capacity of biodiversity". Based on known results from the combined literature, I present the implications of this theoretical model by analysing the properties of the production set and the optimal conditions of the farmers. I explain that this can decompose the benefits and the costs of biodiversity, capturing the benefits through additional yields and variable input savings and the costs through the management of labour and capital at the farm scale. I explain that the developed model is a generalization of the usual production economics models applied to agriculture where land is an allocable input.

2.1 Introduction

While measures of environmental quality over the 20th and 21st centuries have indicated a steady decline in biodiversity, notably due to landscape simplification and deforestation (Barnosky et al., 2011; Butchart et al., 2010; Kleijn et al., 2009), several strategies have been developed to highlight the dependence of human welfare on biodiversity and nature. Among these efforts, the Millennium Ecosystem Assessment (MEA, 2005) has undoubtedly had the greatest influence among policymakers and researchers in recent years. This influence is explained by the popularization of the notion of ecosystem services (ES), defined as “the benefits people obtain from ecosystems" for “free” (MEA, 2005). The concept of ES provides a link between economics and ecological functioning, thus making the concept as a whole valuable for researchers in different disciplines who specialize in different aspects of these systems. The MEA (2005) has distinguished four categories of ES: supporting, provisioning, regulating and cultural ES, where supporting services ensure the provision of the other ES categories. The “supporting service” category has long suffered from a lack of clear conceptualization because other ES also influence each other through complex relations of complementarity or substitution (Bennett et al., 2009; Müller et al., 2016). Several alternative typologies have been proposed to overcome this issue (EFESE, 2017; Potschin-Young et al., 2017; TEEB, 2010). Among the different proposals, the French evaluation of ecosystems and ecosystem services (EFESE) suggests distinguishing “input ES” from the others, making it possible to isolate the contribution of the ecosystem from the efforts of farmers (EFESE, 2017). In this framework, input ES includes supporting services and some other regulating services, such as biological control, that contribute to provisioning and cultural services, particularly, those services supported by agricultural landscapes (e.g., agricultural outputs). Such a distinction is relatively similar to that of White and Hanley (2016), where the input ES corresponds to the modification, stemming from farmers’ efforts, of the flow of productive ecological functionalities.

Given the sensitivity of agricultural yields to ecological processes, farmers are considered to be some of the largest beneficiaries of input ES (Zhang et al., 2007). Relying on the long economic tradition of monetarization, multiple works have valued input ES based on the sensitivity of agricultural yields to specific input ES such as pollination or biological control (Daniels et al., 2017; Gallai et al., 2009; Matsushita et al., 2017). However, because input ES may interact with each other, some economists have privileged the productivity measure of biodiversity indicators (e.g., Di Falco and Chavas, 2008; Donfouet et al., 2017; Matsushita et al., 2016), given that the productivity of biodiversity is closely related to the value of input ES (Chavas, 2009; Tilman et al., 2005). Biodiversity entails rich biotic interactions that ensure well-functioning ecosystems and ES provision (Haines-Young and Potschin, 2010; TEEB, 2010). Because measures of species abundance are costly, scientists usually quantify biodiversity by using indicators. Due to difficulties in measuring "direct" biodiversity indicators, such as the common bird index (Levrel, 2007), indirect indicators based on landscape structure have been privileged. These indirect indicators provide information on the habitat quality of a landscape by measuring its easily observable characteristics. These indicators have been computed in agricultural landscapes, which provide large habitats for agrobiodiversity in many parts of the world, including in Europe, where biodiversity and agriculture have evolved conjointly. In particular, biodiversity indicators are correlated with the provision of several input ES, which explains how the productivity measure of biodiversity indicators allows the valuation of bundles of input ES.[11] Given that the causalities between landscape patterns and biodiversity are variable and well known, and hence uncertain, these input ES should be considered as "potentially" provided. In other words, biodiversity indicators support a vector of input ES that are only potentially provided to the farmers, with the effective provision depending on unobservable characteristics of the ecosystems. This idea is a common feature of landscape ecology, where indicators of indicators are often used (Feld et al., 2009): in my precise case, biodiversity indicators are indicators of biodiversity, biodiversity being the indicator for productive ecosystem services.

The literature on the productivity of biodiversity has provided valuable information on the role of landscapes and ecosystems in the productivity and profitability of agriculture. Scientists working in this field have produced diverse results, including that diversified ecosystems were positively correlated with agricultural yields, confirming the ecologists' hypothesis of an "over-yielding" effect (Hooper et al., 2005), i.e., the additional amount of biomass produced in an ecosystem compared to any of its species/crops alone. Researchers have also highlighted that biodiversity reduces the variance of agricultural yields (Matsushita et al., 2016), confirming the ecologists' "diversity-stability" hypothesis (MacArthur, 1955). These results have led some authors to conclude that biodiversity and attached input ES have both a productive value (Chavas, 2009) and an insurance value (Baumgärtner, 2007).

[11] For example, the usual crop diversity indicators are positively correlated to soil structure (Mäder et al., 2002), pollination (Garibaldi et al., 2016) and biological control (Gardiner et al., 2009; Letourneau et al., 2011).

The notion of input ES stems directly from an economic perspective and, in particular, from a production economic one. The field of production economics analyses the behaviour of firms and is a cornerstone of microeconomics. Production economics aims to explain the principles by which a firm decides the quantity of output to produce (i.e., the firm's supply side) and the quantities of inputs to use (i.e., the firm's demand side). The developed theories regarding the choices of output and input quantities rely on the properties of the technologies and on the input and output prices. In particular, given the same input prices, producers use inputs with the highest productivities. However, even if the literature on the productivity of biodiversity indicators has used the properties of agricultural technologies, it has rarely integrated the behaviour of producers and usually ignores the role of prices. In particular, farmers may use inputs that are more profitable than biodiversity, i.e., farmers may substitute biodiversity with other inputs (fertilizers and pesticides). Overall, we do not know if biodiversity indicators and related input ES are managed by the farmers, i.e., if they are pure externalities or if they are, at least partly, internalized. This may explain the surprising dichotomy between, on the one hand, the previous conclusions that biodiversity and input ES increase yields and, on the other hand, the common feeling, supported by many ecological measures, that biodiversity is declining (Butchart et al., 2010; Waters et al., 2016) and that agriculture is increasingly specialized (Evenson and Gollin, 2003).

In particular, one may consider that farmers are both consumers and suppliers of input ES. On the one hand, because they exploit the results of the productivity of biodiversity indicators, farmers would be considered by production economists to be consumers of input ES. On the other hand, because the provision of input ES depends partly on ecosystem functioning and thus indirectly on landscape structure, which depends in turn on farm acreage, farmers' acreage choices influence the provision of input ES. In other words, farmers may also be suppliers of input ES. There is very little economics literature about this feedback link, so we ignore whether farmers actually integrate it into their decisions. The study of this feedback link is the cornerstone of the first part of my PhD. I want to provide evidence of such management (or absence of management) using the principles of production economics. Indeed, if farmers do integrate these feedbacks, then even if input ES are not exchanged on real markets, their management by farmers should involve mechanisms similar to those of usual markets for goods and services: the demand and the supply of input ES are codetermined and reach an equilibrium. As usual, the demand and the supply of input ES depend on the prices and the productive properties of other inputs. The specificity of supply and demand for input ES is that a farmer is in charge of both, implying that the levels of input ES depend on the internal equilibrium of his farm. In other words, farmers manage the input ES levels on their farms, or, as highlighted by Chavas et al. (2010), "*farmers are in the business of managing their local ecosystem*". I argue that a deeper examination of the management of input ES could illuminate the dichotomy between biodiversity losses and input ES.

Here, I propose a theoretical framework to analyse farmers' management of input ES. For that purpose, I bring together the literature on the productivity of biodiversity indicators with the literature on farmers' production choices, more particularly the sub-literature on farmers' acreage choices. The model developed here considers that farmers manage their acreage by taking into account that acreage influences input ES provision and, indirectly, the profitability of outputs thanks to additional yields and/or input savings. The proposed approach allows us to specify both the supply and the demand for input ES or, in other words, to examine both the benefits and the costs linked to the management of input ES. I think that both literatures could benefit from such a rapprochement. The "biodiversity productivity" literature could benefit from the literature on acreage choices, not only to improve the former's comprehension of the costs and benefits of input ES but also to limit endogenous biases about the empirical applications of those ES. The literature on acreage choices could benefit the literature on the productivity of biodiversity indicators to improve the representation of agricultural technology and to improve the evaluations of agricultural and environmental policies. Similar to previous studies, I am not able to observe the flows of input ES, and my model relies on biodiversity indicators. I refer to the capacity of an ecosystem to provide input ES based on its observable characteristics as the "biodiversity productive capacity". This concept expresses that biodiversity indicators support a vector of input ES that are only potentially provided to the farmers. The theoretical model that I develop considers a risk-neutral framework, informing only on the "productive value" of biodiversity (Chavas, 2009). The theoretical framework is built at the farm scale, and I assume that farmers manage their acreage independently from each other. I also consider that farmers have access to all information and, in particular, that farmers know the technical properties of input ES. Some of the obvious limitations of these assumptions will be discussed later.

The chapter is organized as follows. I first present a review of the literature on (i) the productivity assessment of biodiversity indicators, (ii) the biodiversity indicators from landscape ecology and (iii) land-use choices. The second section presents the theoretical framework of the first part of the PhD thesis. I present the optimal conditions of the management of biodiversity productive capacity and discuss them with regard to microeconomic results. The third section discusses the results and links them with chapters 3, 4 and 5. I conclude the chapter in the last section.

2.2 Literature review

The literature review succinctly presents three literatures: the economic literature on the productivity of biodiversity, the ecological literature on biodiversity indicators and the economic literature on land-use choices, with special attention to acreage models.

2.2.1 Literature review on the productivity of biodiversity

In the early years of ecological economics, ES were sometimes considered as inputs (Westman, 1977).[12] Yet, prior to the 1990s, the study of input ES in agricultural economics did not have a focus on biodiversity-friendly practices, such as soil conservation practices (Barbier, 1990) or crop diversification (Heisey et al., 1997), with the exception of the attention paid to the management of pest pressure (an "ecosystem dis-service" - Lichtenberg and Zilberman, 1986). Since this period, one of the most dynamic research fields has focused on the beneficial effects of (agro)biodiversity on agriculture, and this field has made both theoretical and empirical contributions. Most empirical works have been concerned with the productive properties of biodiversity indicators, with significant contributions from both agricultural and development economists.[13] The seminal works of Heisey et al. (1997) and Smale et al. (1998) on the productivity of the intra-specific diversity of wheat have paved the way for a rich empirical literature on the productivity of intra- and inter-specific diversities of crops (Bangwayo-Skeete et al., 2012; Bellora et al., 2017; Chavas and Di Falco, 2012a; Di Falco and Chavas, 2006, 2008, 2009; Di Falco and Perrings, 2003, 2005; Di Falco and Zoupanidou, 2017; Di Falco et al., 2007, 2010; Donfouet et al., 2017; Finger and Buchmann, 2015; Matsushita et al., 2016; Ofori-Bah and Asafu-Adjaye, 2011; Omer et al., 2007; van Rensburg and Mulugeta, 2016).[14] Whereas Smale et al. (1998) and Meng et al. (1998) have distinguished different components of diversity (i.e., spatial, temporal, apparent and latent diversities), the large majority of papers have focused

[12] See Gómez-Baggethun et al. (2010) for an interesting historical perspective on the introduction of nature and ES into economics, from François Quesnay (1758) to The Economics of Ecosystems and Biodiversity (2010).
[13] The two sub-disciplines are interested in using the productivity of biodiversity indicators when considering the possible reduction of other agricultural inputs. However, agricultural economists' motivation stems mainly from the polluting nature of most chemical inputs, while development economists are more motivated by food security issues in a context of the difficulty of accessing chemical inputs in developing countries.
[14] Intra-specific diversity refers to genetic diversity for the same crop (i.e. the different varieties), whereas inter-specific diversity refers to the diversity among crops.

on “spatial diversity”, i.e., what Meng et al. (1998) called the “*the amount of diversity in a given geographical area*”. In practice, the range of the studied geographical area differs, ranging from the farm-scale case, which is the most common (Di Falco and Chavas, 2009), to the landscape (Bellora et al., 2017), the cantonal (Donfouet et al., 2017) or the regional scales (Di Falco and Chavas, 2008).

The methodology of the cited studies consists of directly estimating the marginal effects of the selected indicators on reduced-form equations, typically mean and/or variance yields or profit, to assess their productivity and/or profitability. The results are clear: biodiversity productive capacity is a productive input that enhances mean agricultural yields. Studies based on profit analysis have also concluded a profitable effect of biodiversity (Di Falco and Perrings, 2005). Chavas and Di Falco (2012b) find, for example, that crop diversity contributes to 17% of average farmers’ revenues. Studies using a data envelopment analysis approach have also concluded that crop diversity increases technical efficiency (Barnes, 2006; Fontes and Groom, 2018; Karunarathna and Wilson, 2017; Ofori-Bah and Asafu-Adjaye, 2011). It also appears that crop diversity (i) has decreasing marginal returns on yield and profit (Di Falco and Chavas, 2006), (ii) enhances future yields (Di Falco and Chavas, 2008), (iii) is a substitute for soil fertility (Di Falco and Zoupanidou, 2017) and (iv) is a risk-reducing input that may be suitable for risk production management (Di Falco and Chavas, 2009; Di Falco and Perrings, 2005). In particular, it appears that crop diversity reduces the variance of yields more when pesticide applications are low (Di Falco and Chavas, 2006), when the weather is dry (Di Falco and Chavas, 2008) and when farmers have poor access to financial insurance schemes (Falco et al., 2014). These evidences contribute to the idea that biodiversity has both productive and insurance values (Baumgärtner, 2007; Chavas, 2009). In particular, because biodiversity provides natural insurance to risk-averse farmers, its levels increase with uncertainty (Quaas and Baumgärtner, 2008). Other works have studied the productivity of semi-natural areas such as grasslands (van Rensburg and Mulugeta, 2016) and forests (Klemick, 2011; Matsushita et al., 2017), considering these elements as proxies of specific components of biodiversity within agricultural ecosystems. These works have identified similar properties: biodiversity indicators have positive agricultural productivity (Klemick, 2011), constitute a risk-reducing input (Matsushita et al., 2017) and are profitable overall (van Rensburg and Mulugeta, 2016). A smaller body of research has also confirmed that biodiversity increases yields when the areas under agro-environmental measures are considered as an original biodiversity indicator (e.g., Omer et al., 2007). In contrast, Omer et al. (2007) found that biodiversity is associated with lower technical efficiency.

Although Chavas (2009) tries to decompose the productive effects of crop diversity using the economic properties of agricultural technology, most of the results are explained by ecological and agronomic literature. These studies provide several explanations for the ecological processes inside the “black box” of biodiversity productivity. From an ecological perspective, biodiversity may increase crop production in three ways. The first is due to the sampling effect, which implies that an increase in the number of species/crops increases the probability that key species/crops with the strongest effects on performance are present in the ecosystem (Di Falco, 2012; Fontes and Groom, 2018). The second explanation is due to complementarity effects (Hooper et al., 2005). Complementarity effects stem from the heterogeneity of needs across species/crops over time, which increases the efficiency of resource use over time. Therefore, when resources are the limiting factor to growth, increasing species/crop diversity increases ecosystem/crop productivity. The third explanation relates to facilitation effects. Facilitation effects express the positive interactions between species/crops (Hooper et al., 2005), e.g., one crop is able to provide a critical resource (e.g., nitrogen) to other crops. According to Hooper et al. (2005), the facilitation and complementary effects are the main reasons for the over-yielding effect, i.e., the additional amount of biomass produced in an ecosystem compared to any of its species/crops alone. This effect has been measured empirically, notably by Costanza et al. (2007), who proved that species diversity increases net primary production. From an agronomic perspective, the over-yielding effect refers to the additional yield of a crop when it is grown with other crops compared to its yields in a monoculture. The agricultural practices used to manage the underlying mechanisms have also been investigated by the agronomic literature. Some of them are well known from farmers. Among them, crop rotations are applied by most farmers. Indeed, suitable crop rotation enhances the yield of subsequent productions through its beneficial role in (i) biological protection against pests, disease and weeds; (ii) the nutrient stock available for subsequent production and (iii) the soil structure, which allows better root penetration in the subsequent production (Hennessy, 2006). More recent works have analysed the effects of agricultural practices at scales larger than the plot thanks to “landscape agronomy” approaches (Benoît et al., 2012). These works have found beneficial effects of diversified arable landscapes on crop yields, notably thanks to pollination (Garibaldi et al., 2016) or biological control (Gardiner et al., 2009).

Although the literature on biodiversity productive capacity gives useful results about the sensitivity of agricultural yields to input ES, this literature still suffers from several limitations (most of them are detailed in Chapter 2). From my point of view, the main bias is due to the lack of consideration of the optimizing role of the farmer. Except for the work of Di Falco et al. (2014) on the impact of crop diversity on financial insurance subscriptions, no paper has explicitly linked biodiversity indicators and farmers' choices. However, as biodiversity indicators depend on land use and as land use results from the choices of utility-maximizing agents, biodiversity indicators depend on farmers' choices. The aim of this chapter is to present a theoretical framework that explicitly represents the choices that enable farmers to benefit from biodiversity productive capacity.

2.2.2 Landscape ecology and biodiversity indicators

We refer to the notion of biodiversity as it applies to different hierarchical levels: gene, species, family, ecosystem, etc. Overall, these elements are difficult to observe, justifying the use of biodiversity indicators. Like all indicators, biodiversity indicators are instruments that distinguish between the object to be measured (i.e., the biodiversity levels and/or ES) and the measure itself, which can be connected by several terms thanks to implicit models (Desrosières, 2003). Indicators rely on observable characteristics of the object to be measured and are thus inherently imperfect. Among the diversity of biodiversity indicators, two groups are currently distinguished: (i) direct indicators (or taxonomic indicators) that measure the abundance or presence of species or indicator species in point maps (Gregory et al., 2005) and (ii) indirect indicators (or structural indicators) based on land use. Examples of direct indicators are the common bird index (Gregory et al., 2005). However, direct indicators are subjected to criticisms from ecologists, not only because they do not provide information on ecosystem dynamics but also because they require costly counting of species.[15] The consequence is that the direct indicator approach is not used much in economics, even if successful examples have recently been achieved (Mouysset et al., 2014). Most of the literature in economics, and especially the "biodiversity productivity" literature, has used indirect indicators.[16] Notably, their utilization is

[15] In France, the common bird index is measured thanks to a participatory research approach where amateur ornithologists send their measures to scientists at the French Natural History Museum.

[16] Baumgärtner (2006) reviews the diversity indicators used by ecologists and economists to measure biodiversity. While the former rely on relative abundance of species or habitats, the latter usually rely on the divergence of species into phylogenetic trees, i.e. on the diversity of features. These last indicators have been developed following Weitzman (1992) and are known as "distance-dissimilarity" indicators. In practice, these last indicators have been used in the perspective of the "Noah's ark problem", i.e. the most efficient allocation of conservation efforts between species (Weitzman, 1998). To my knowledge, these indicators have rarely been used for alternative objectives and never for productivity assessment.

favoured because information on landscape composition is easier to access compared with other data. This is particularly true with regard to information on acreage choices, which is directly available in most economic datasets such as the Farm Accountancy Data Network (FADN), which is the database used by the European Commission to assess the impact of CAP reform on farmers' choices.

Indirect indicators benefit from a higher consensus of support from the scientific community, thanks to landscape ecology. Landscape ecology is the science studying relationships between ecological processes and landscape structure (Turner, 1989); its main difference from other approaches in ecology is the consideration of the heterogeneity of spatial patterns at the landscape scale (Burel and Baudry, 2003). Indeed, if some ecosystem processes operate at the plot scale or at lower scales, others are expressed at the landscape scale. The basic idea is that heterogeneous landscapes are better able to support ecological functionalities than are homogeneous landscapes. However, like other ecologists, landscape ecologists focus on the dynamics of ecological processes, notably in response to disturbances.

In practice, landscape ecology studies the abundance and dynamics of species according to the two dimensions of landscape structure: landscape composition and landscape configuration (Burel and Baudry, 2003). Landscape composition refers to the number of patch types in the landscape and their relative abundance. For example, the amount of cropland or grassland or the density of roads can be aspects of landscape composition. Landscape configuration represents the spatial arrangements of the patches. As a consequence, indirect indicators can rely on (i) the single configuration dimension (Kindlmann and Burel, 2008), e.g., patch isolation (Bender et al., 2003), (ii) the single composition dimension, e.g., Shannon or Simpson indexes (Nagendra, 2002) or (iii) both single configuration and dimension, e.g., the distance to a specific area such as hedgerows (Morandin et al., 2014) or more complex functions that mix different indicators (Martel et al., 2017). Landscape ecologists also study the impact of the scale at which the indicator is calculated on underlying ecological processes (Turner, 1989). Due to data availability, the literature on the productivity of biodiversity relies on biodiversity indicators that are based on landscape composition (see appendix 2.B. for a discussion on the selection of these indicators).

It would be a mistake to reduce landscape ecology to the computation of ecological indicators or, more generally, to the study of ecological processes at the landscape scale. Indeed, some works from the European school of landscape ecology analyse landscape as a holistic object

(Wu, 2006), specifically considering that landscape structure is the result of agents' choices. In other words, they expressly recognize that ES are the results of economic agents, these links being roughly represented in the "cascade" framework (Haines-Young and Potschin, 2010). European landscape ecologists increasingly couple land-use models that include agents' choices with ecological models to study the influence of agents' land-use choices on landscape structure and related ES provision (e.g., Valbuena et al., 2010). If these approaches are often based on decision rules (Martel et al., 2017), some collaborations between economists and landscape ecologists have explicitly focused on the indirect impact of policies on landscape structure and related ES (Brady et al., 2009, 2012; Turpin et al., 2009). Overall, land-use and ES concepts allow landscape ecologists and economists to work together. I present the literature on land-use choices in the next section.

2.2.3 Land use choices

Land-use choice models have increasingly been developed since Plantinga (1996) and now constitute an increasing trend in environmental economics (Bateman et al., 2013). These models explain land-use evolutions of aggregated land categories (i.e., urban, forest, croplands) in the long term, notably to explore the issues related to biodiversity losses such as deforestation (Chakir and Parent, 2009; Watson et al., 2000). These models are notably used to explore the links between land managers' choices and the provision of some "non-input" ES (Ay et al., 2014; Bateman, 2014; Bateman et al., 2013; Chakir et al., 2012; Laukkanen and Nauges, 2014; Polasky et al., 2011). Even if these studies do not consider the landowner who managed the ES, they constitute a large literature with an explicit link between ES and land-use, as highlighted by the bibliometric analysis in appendix 2.A.

Prior to this well-known literature, agricultural economists had already studied farmers' land-use choices. Even if agricultural economists' approaches are not adapted to the study of similar long-term mechanisms, they present several advantages. First, these models explicitly consider that agents produce different outputs. If both literatures examine how the landowners' utility maximization explains the landowners' decision to devote one piece of land to one usage rather than to others, environmental economists assume that the landowner's decision about one piece of land is independent from her decision about another piece of land. Relaxing this assumption, acreage models examine the interactions between choices about different pieces of land managed by the same agent. In other words, acreage models analyse some joint production processes at stake on farms (Chambers and Just, 1989). Second, acreage models rely on the specific representation of agricultural technology, which is almost absent in the land-use literature. This explicit representation is sufficiently flexible to represent the effects of specific inputs, which have been ignored by the acreage literature. Third, acreage models pay deeper attention to economic incentives, especially prices, policies and regulations. The last two features allow us to examine farmers' production choices on each piece of land (i.e., the choices at the intensive margins) in addition to the land-use choices (i.e., the choices at the extensive margin). I present here some of the economic models developed to examine farmers' choices in the short term and in a certain framework. I first present models from linear mathematical programming. I then present the models based on the duality theory (developed in the 1980s) and especially dual models where land is an allocable fixed input (developed in the 1990s). I finish the presentation with the recent acreage models that mix primal and dual approaches (developed since the mid-2000s).

2.2.3.1 Linear programming models

Historically, agricultural economists first developed models based on linear programming to simulate policy instruments based on acreage choices (Carpentier et al., 2015). These models have often assumed fixed margins for each output, and the diversification motives were introduced thanks to linear constraints on land use, representing either labour and machinery constraints attached to each crop, or thanks to agronomical constraints. Given the high flexibility of constraint implementation and technological representation, these linear programming models easily provide ex-ante evaluations of different policy instruments. However, their relatively complex structure prevents the determination of easily interpretable analytical solutions, which are usually discontinuous in their parameters. This prevents econometric estimations of the solutions and requires a sensitive model calibration. The interacting constraints also lead to known "bang-bang"-type responses in which agents' optimal choices switch from one constraint to another in response to policy changes, a scenario that is usually unrealistic. These limits have encouraged economists, especially econometricians, to develop new approaches.

2.2.3.2 Dual models

Historically, econometricians' initial works applied to agriculture focused on primal models, directly estimating production functions to measure the marginal productivity of inputs using output and input quantities. However, since the development of the duality theory (Fuss and McFadden, 1978), economists have analysed farmers' choices according to the evolution of prices to investigate the properties of agricultural technology. In practice, these models rely on generic properties of multi-output production technologies and generic objective functions (usually indirect profit functions) to explain farmers' responses to economic incentives. The objective functions are usually modelled with flexible functional forms that enable the estimation of farmers' choices without overly restrictive assumptions on the form of the technology and without constraints. These approaches provide more easily determinable and smoother responses than do linear programming models, and the optimal conditions are usually determined using Hotelling's Lemma (Weaver, 1983). Dual models are also interesting because the required data on agricultural prices and quantities are easily available. If they are well adapted to evaluate the effects of taxes or subventions on farmers' decisions, their structure complicates the evaluation of decoupled instruments (e.g., area-based decoupled payments). This has led some authors to propose an extension of these models.

In particular, the dual models with specific acreage choices are an extension of "pure" dual models. Based on the seminal article of Chambers and Just (1989), these models also account for special constraints on fixed factors, for instance, on farms' total land constraints. Because farmers face constraints, they must allocate the constrained fixed input between the production of the different outputs. In the case of land, farmers must decide how to allocate outputs given a constrained total amount of land. Allocating fixed input into different outputs, the profit function defined by Chambers and Just (1989) overcomes the "apparently input-joint technology" issue developed by Shumway et al. (1984). Similar to "pure" dual models, these models benefit from the utilization of reduced form models to prevent the need to represent any specific technology. The estimation of these models is developed in three steps. Authors first derive the optimal yields and input utilization conditional on acreage; then, they determine the optimal acreage based on these optimal margins and finally compute input demand and output supply functions based on the optimal acreage (i.e., intensive margin choices). Here, acreage choices are the results of a profit maximization problem with land as an allocable fixed input. Thus, these models enable the examination of both farmers' choices at the intensive margin (i.e., input utilization and optimal yields for each crop) and the extensive margin (farmers' acreage choices). The popularity of these models is explained by their tractability and their adaptation to available data, which are suitable characteristics to evaluate area-based instruments (Guyomard et al., 1996; Lacroix and Thomas, 2011; Sckokai and Moro, 2006).

One crucial assumption of this literature is that it has to consider a crop diversification motive, which otherwise would provide optimal solutions selecting the monoculture of the most rentable crop as a rational choice, which is usually not observed in reality. Four types of motives have been distinguished to explain diversification. They are related to (i) decreasing marginal return to crop acreages (Just et al., 1983), (ii) constraints associated with management of quasi-fixed inputs (Carpentier and Letort, 2014; Sckokai and Moro, 2006), (iii) crop rotation benefits (Orazem and Miranowski, 1994; Thomas, 2003) or (iv) market risk-spreading (Chavas and Holt, 1990; Sckokai and Moro, 2006). First, papers focusing on decreasing marginal return to crop acreages are usually motivated by the Ricardian idea that land quality is heterogeneous, which implies that the best lands are cropped first. Second, papers focusing on constraints associated with the management of quasi-fixed inputs aim to account for machinery constraints (particularly output-specific machinery) or peak loads due to labour constraints. This idea was first developed by works using positive mathematical programming (PMP) (Howitt, 1995). Relying on linear programming models, the PMP approach adds a quadratic term in the objective function, which eases the calibration on real data and provides smoother responses than the “bang-bang”-type ones. This quadratic term is usually interpreted as the “implicit” management cost of acreage (Heckelei and Wolff, 2003). This management cost function has also been used by Carpentier and Letort (2012, 2014) or Chavas and Holt (1990) for econometric estimations. In this case, the parameters of the management cost function are interpreted as all the constraints and benefits associated with crop diversification, including its productive effects (i.e., its “negative” costs). Third, papers motivated by crop rotation management have developed dynamic acreage models where farmers maximize the sum of their discounted anticipated profits over their carrier and where the yields or the variable input applications depend on a fertility indicator, which is a function of past acreages (Orazem and Miranowski, 1994; Thomas, 2003). In this literature, the diversification motive is due to price anticipations and the characteristics of agricultural technology that imply trade-offs between current and future benefits. Finally, crop diversification for risk management motives relies on the commonly known portfolio strategy, which I do not develop here as I consider a risk-neutral framework.

Acreage models with land as an allocable fixed input are particularly useful to examine the indirect effects of one output price change, or of one output-specific area subsidy, on the lands devoted to other outputs (Lansink and Peerlings, 1996). These indirect effects are interpreted by Chambers and Just (1989) as the reorganization of fixed inputs at the farm scale due to the modification of the economic context. For example, an increase in wheat prices leads to an increase in the wheat supply at the intensive and extensive margins, but it also impacts oilseed supply due to reorganization of the vector of the farm's fixed inputs, including total farmland area, which makes it possible to explain acreage choices. This decomposition of effects at the farm scale is a real gain from these models compared to "pure" dual models. However, like "pure" dual models, dual models with land as an allocable fixed input suffer from some limits. First, the utilization of reduced-form equations provides some parameters that are difficult to interpret. Second, these models are not well adapted to integrate non-marketed inputs (such as agronomical techniques). Third, the obtained results rely heavily on behaviour assumptions regarding price anticipations, especially (Nerlove and Bessler, 2001).

2.2.3.3 The recent revival of more primal models

The difficulty of interpreting parameters and the mechanisms at stake in dual models are partly explained by the implicit representation of agricultural technology, which, according to Just and Pope (2002), two of the fathers of dual models, may have been excessively simplified. Abstract representation of agricultural technology enables the estimation of flexible models, but it loses relevance when the technology is complex or when the input has no explicit price, as is the case for biodiversity productive capacity.

Recent acreage models have combined dual and primal approaches to derive original structural models and overcome these issues. In particular, Carpentier and Letort (2012, 2014) proposed a multioutput acreage model with the same yield function as that used by Pope and Just (2003) for each output. This approach enables an explicit representation of the multioutput agricultural technologies with relevant agronomical interpretations, which is particularly suited when considering non-marketed inputs such as climatic variables. The irrelevant interactions between outputs are captured implicitly by the flexible cost function proposed by the PMP literature, enabling the assumption of constant return to acreage that eases the econometric estimation. Carpentier and Letort (2014) prove that when an entropic cost function is specified, their model is similar to the multinomial logit land-use model developed by Lichtenberg (1989) or Plantinga (1996). However, Carpentier et al. (2015) stressed that both types of studies can benefit from each other, and the primal approaches developed in land-use models could offer new ideas to agricultural economists. One illustration is the use of common land-use models to examine farmers' choices regarding the structure of crop rotations at the plot level according to prices (Hendricks et al., 2014; Livingston et al., 2015).

As biodiversity productive capacity is a non-marketed input, this mix between primal and dual approaches is well suited to link the "biodiversity productivity" literature and the acreage models. In the next section, I develop an acreage model that takes into account the productivity gains and the variable input saving gains from biodiversity. For this purpose, I introduce a biodiversity indicator based on acreage shares at the farm scale in the production functions. To separate the effects of diversified acreage on gross margins (yields and variable inputs) from those due to the management of quasi-fixed inputs, I also introduce the management cost function developed in the PMP literature. In other words, I consider two motives for crop diversification: the productive effects of biodiversity and the management costs of fixed input management. This link between biodiversity indicators and land-use choices provides a new framework for the analysis of the management of biodiversity productive capacity. This framework considers both the benefits (productivity gains and input savings) and the costs (the management of fixed inputs) of biodiversity at the farm scale.

2.3 Theoretical framework

In this section, I present the mathematical formalization from the acreage model literature and from the biodiversity productivity literature. Then, I propose a unified framework from which I derive the optimal conditions for acreage choices and variable input applications. I then discuss these conditions and compare them to the results of both literatures.

2.3.1 Acreage models: general assumptions and results

Here, I present the farmer's choice model in the short term in a static and certain framework. The present model aims to explain the acreage choices, the choices of variable input utilization and the supply choices for each output for one year. We consider that capital, labour and total land are fixed inputs and are considered exogenous. The endogenous variables are acreage choices and variable input choices. I assume that farmers produce K crops for which they allocate S_k units of land to each crop k. On each unit of land k, the farmers produce y_k (i.e., y_k is the yields of output k) and use x_{ik} of variable input i, with $i \in [1;I]$. We denote $\mathbf{x}_k = (x_{ik} : i \in [1;I])$ the vector of variable inputs applied on one unit of area k and $\mathbf{X}_k = (S_k \times \mathbf{x}_k^{'})$ the vector of inputs applied to k at the farm scale. We note $\mathbf{S} = (S_k : k \in [1;K])$ the vector of allocated area at the farm scale (i.e., the acreage), $\mathbf{Y} = (S_k \times y_k : k \in [1;K])$ the vector of production at the farm scale and $\mathbf{X} = \left(\sum_{k=1}^{K} \mathbf{X}_k ; i \in [1;I]\right)$ the vector of applied inputs at the farm scale. $\mathbf{Z}$ is the vector of fixed inputs and includes labour, machinery and total area

$\bar{S}$ (with $\bar{S}=\sum_{k=1}^{K} S_k$) available at the farm scale. In addition, farmers face the market price p_k for each k, with $\mathbf{p}=\left(p_k : k\in[1;K]\right)$ being the vector of the output prices. Similarly, farmers face $\mathbf{w}=\left(w_i : i\in[1;I]\right)$ where w_i is the price of input i and $\mathbf{a}=\left(a_k : k\in[1;K]\right)$ where a_k is the sum of the area-specific subsidies for k. We assume that farmers are price-takers for $\mathbf{p}$, $\mathbf{w}$ and $\mathbf{a}$.[17]

We assume that the farmers maximize their profit. According to Fuss and McFadden (1978), under specific conditions on the technology, the profit function $\Pi(\mathbf{p},\mathbf{w},\mathbf{a},\mathbf{Z})$ exists and is convex, linearly homogenous and continuous in $\mathbf{p}$, $\mathbf{w}$ and $\mathbf{a}$. $\Pi(\mathbf{p},\mathbf{w},\mathbf{a},\mathbf{Z})$ is also non-decreasing in $\mathbf{p}$ and $\mathbf{a}$ and non-increasing in $\mathbf{w}$. We denote the restricted profit function $\Pi^r(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z})$ conditional on $\mathbf{Z}$. Π^r relies on a constrained production set noted T. Π^r has the same properties as Π and is non-decreasing in $\mathbf{Z}$. According to Fuss and McFadden (1978) and assuming no subsidies $\mathbf{a}$ for the moment, we can define Π^r as:

$$\Pi^r = \underset{\mathbf{Y},\mathbf{X}}{\mathrm{Max}}\left\{\mathbf{p}'\,\mathbf{Y}-\mathbf{w}'\,\mathbf{X};\left(\mathbf{Y},\mathbf{X},\mathbf{Z}\right)\in\mathrm{T}\right\} \tag{2.1}$$

as long as T is bounded compact and quasi-convex in $\mathbf{Y}$ and $\mathbf{X}$. In addition, T is assumed to be closed (i.e., T contains its frontiers), non-empty, characterized by impossibility of free production (i.e., $\forall(\mathbf{Y},\mathbf{X},\mathbf{Z})\in\mathrm{T}, \mathbf{Y}\neq 0 \Rightarrow \mathbf{X}\neq 0$), free disposal (i.e., $\mathbf{Y}\in\mathrm{T}$ implies that $\mathbf{Y}_0\in\mathrm{T}$ for any $\mathbf{Y}_0\leq\mathbf{Y}$) and shut-down properties (i.e., $(0,0)\in\mathrm{T}$). Under these assumptions, we can define the set $\mathbf{Y}(\mathbf{X},\mathbf{Z})=\left\{\mathbf{Y};\ (\mathbf{Y},\mathbf{X},\mathbf{Z})\in\mathrm{T}\right\}$ constitute the sets of $\mathbf{Y}$ which can be produced by $\mathbf{X}$. $\mathbf{Y}(\mathbf{X})$ is non-empty, closed and admits an upper bound. The properties of T and $\mathbf{Y}(\mathbf{X},\mathbf{Z})$ explain the homogeneity and continuous properties of Π. The dual models presented in the previous part have thus relied on this framework to determine optimal input demand and output supply. Indeed, Hotelling's Lemma applied to the restricted profit function stipulates that Marshallian demand and supply are defined by:

[17] This is probably correct for $\mathbf{w}$ and $\mathbf{a}$ but less debatable for $\mathbf{p}$ as output prices depend on output quality, which depends (at least partly) on $\mathbf{X}$.

$$\begin{cases} \mathbf{X}^*(\mathbf{p},\mathbf{w};\mathbf{Z}) = -\dfrac{\partial \Pi^r(\mathbf{p},\mathbf{w};\mathbf{Z})}{\partial \mathbf{w}} \\ \mathbf{Y}^*(\mathbf{p},\mathbf{w};\mathbf{Z}) = \dfrac{\partial \Pi^r(\mathbf{p},\mathbf{w};\mathbf{Z})}{\partial \mathbf{p}} \end{cases} \tag{2.2}$$

These solutions were investigated by "pure" dual models (e.g., Weaver, 1983). Now, if we integrate that total land is exogenous and thus that land is an allocable input, we can define the profit as a function of indirect profit functions conditional to $\mathbf{S}$. Here, we explicitly introduce acreage considering that the production set T' includes not only $\mathbf{Y}$, $\mathbf{X}$ and $\mathbf{Z}$ but also $\mathbf{S}$. Assuming that subsidies $\mathbf{a}$ are available, we can define the restricted profit function as:

$$\Pi^r = \underset{\mathbf{Y},\mathbf{X},\mathbf{S}}{\mathrm{Max}} \left\{ \mathbf{p}'\mathbf{Y} - \mathbf{w}'\mathbf{X} + \mathbf{a}'\mathbf{S} \ ; \ (\mathbf{Y},\mathbf{X},\ \mathbf{S},\mathbf{Z}) \in \mathrm{T}' \ \text{and} \ \sum_{k=1}^{K} S_k = \bar{S} \right\} \tag{2.3}$$

where $(\mathbf{Y},\mathbf{X},\mathbf{S},\mathbf{Z}) \in \mathrm{T}'$ is the production set constraint. T' has the same properties as T but also depends on $\mathbf{S}$ (Guyomard et al., 1996). We can thus write (2.3) as a two-step maximization, first with the maximization of the indirect profit functions Π^{rs} conditional to $\mathbf{S}$ such that:

$$\Pi^{rs} = \underset{\mathbf{Y},\mathbf{X}}{\mathrm{Max}} \{ \mathbf{p}'\mathbf{Y} - \mathbf{w}'\mathbf{X}; (\mathbf{Y},\mathbf{X},\mathbf{S},\mathbf{Z}) \in \mathrm{T}' \} \tag{2.4}$$

Then, the maximization of

$$\Pi^r = \underset{\mathbf{S}}{\mathrm{Max}} \{ \Pi^{rs}(\mathbf{p},\mathbf{w};\mathbf{S},\mathbf{Z}) + \mathbf{a}'\mathbf{S} ; \mathbf{1}'\mathbf{S} = \bar{S} \} \tag{2.5}$$

with $\mathbf{S}^* = \underset{\mathbf{S}}{\mathrm{argmax}} \{ \Pi^{rs}(\mathbf{p},\mathbf{w};\mathbf{S},\mathbf{Z}) + \mathbf{a}'\mathbf{S} \ ; \ \mathbf{1}'\mathbf{S} = \bar{S} \}$ being the optimal solution of (2.5). Because Marshallian demand and supply depend on acreage choices, the determination of $\mathbf{S}^*$ enables their specification:

$$\begin{cases} \mathbf{X}^*(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z},\bar{\mathbf{S}}) = -\dfrac{\partial \Pi^{rs}(\mathbf{p},\mathbf{w};\mathbf{S}^*(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z}),\mathbf{Z})}{\partial \mathbf{w}} \\ \mathbf{Y}^*(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z},\bar{\mathbf{S}}) = \dfrac{\partial \Pi^{rs}(\mathbf{p},\mathbf{w};\mathbf{S}^*(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z}),\mathbf{Z})}{\partial \mathbf{p}} \end{cases} \tag{2.6}$$

The solution for $\mathbf{S}^*$ depends on specific T'. In practice, two types of restricted profit functions have been proposed:

$$\Pi^r\left(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{S},\mathbf{Z}\right)=\sum_{k=1}^{K}\Pi_k^{r*}\left(p_k,\mathbf{w},a_k;S_k,\mathbf{Z}\right) \tag{2.7}$$

And:

$$\Pi^r\left(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{S},\mathbf{Z}\right)=\sum_{k=1}^{K}\Pi_k^{r*}\left(p_k,\mathbf{w},a_k;S_k,\mathbf{Z}\right)-C\left(\mathbf{S};\mathbf{Z}\right) \tag{2.8}$$

With Π_k^r being the crop-specific profit: $\Pi_k^{r*}=\underset{Y_k,\mathbf{X}_k}{\text{argmax}}\left\{p_kY_k-\mathbf{w}'\mathbf{X}_k\ ;\ \left(Y_k,\mathbf{X}_k;S_k,\mathbf{Z}\right)\in \mathrm{T}_k\right\}$ with T_k the feasible input set for crop k. The production function $Y_k\left(\mathbf{X}_k;S_k,\mathbf{Z}\right)$ is the frontier of T_k for crop k. Y_k is a quasi-concave function of $\mathbf{X}_k$ and is non-decreasing in $\mathbf{Z}$. The results from the acreage literature also specify that Y_k is quasi-concave in S_k. We assume that there is no joint production between each crop based on the levels of other productions $\mathbf{Y}_{-k}$.

Both (2.7) and (2.8) respect the profit function properties defined previously. In (2.7), the only diversification motive is the decreasing return to acreage, which insures the convexity of the profit and enables the identification of the optimal acreage without any explicit representation of jointness in $\mathbf{S}$. Conversely, the profit function in (2.8) is the sum of the crop profit minus a function of the acreage $C(\mathbf{S})$, which is alternatively called the implicit cost function (Heckelei and Wolff, 2003) or the "diversification cost" function (Carpentier and Letort, 2012, 2014). These costs are also interpreted in the literature as dynamic or static adjustment costs (Chambers and Just, 1989; Lansink and Stefanou, 2001; Orazem and Miranowski, 1994), or as the underemployment of fixed input (Dupraz, 1996). The addition of $C(\mathbf{S})$ enables us to consider the fixed input jointness between the K crops due to the allocation of fixed inputs among several outputs (Chambers and Just, 1989) and is thus independent from the crop-specific technologies $Y_k\left(\mathbf{X}_k;S_k,\mathbf{Z}\right)$.

Problems (2.7) and (2.8) are resolved in two stages. The first stage determines the optimal gross margins conditional on S_k, defined as $\pi_{k|S_k}^{r*}=\text{argmax}_{\mathbf{x}_k}\,p_ky_k-\mathbf{w}'\mathbf{x}_k;\ \left(y_k,\mathbf{x}_k;S_k,\mathbf{Z}\right)\in\mathrm{T}_k'$ with π_k^r being twice continuously differentiable, linearly homogeneous and convex in prices, increasing in fixed quantities and output prices, increasing in fixed quantities and output prices and decreasing in input prices (Chambers and Just, 1989; Lansink and Peerlings, 1996). This first stage specifies farmers' optimization of variable input $\mathbf{x}^{*}_{k|S_k}$. The second stage is the optimal allocation of $\mathbf{S}$. Considering the total land constraint $\sum_{k=1}^{K} S_k = \bar{S}$, Hotelling's lemma leads to:

$$\pi^{r*}_{k|S_k}\left(p_k,\mathbf{w};S^*_k,\mathbf{Z}\right)+a_k+S^*_k p_k \frac{\partial y_k\left(\mathbf{x}_k;S_k,\mathbf{Z}\right)}{\partial S_k} = \lambda\left(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z}\right) \quad \forall k\in\left[1;K\right] \tag{2.9}$$

where λ is the shadow value of the total land constraint and $y_k\left(\mathbf{x}_k;S_k,\mathbf{Z}\right)$ is the frontier of T'_k, which is a quasi-concave function of $\mathbf{x}_k$ and non-decreasing in $\mathbf{Z}$. Because λ is the same for each output, the optimal land allocation is obtained when the marginal profits of land in each use are equal, i.e., when the first-order conditions are satisfied. The augmentation of a_k relative to $\mathbf{a}_{-k}$ (i.e., the vector of $\mathbf{a}$ without its k[th] element) or p_k relative to $\mathbf{p}_{-k}$ increases S^*_k. However, the negative value of $\partial y_k\left(\mathbf{x}_k;S_k,\mathbf{Z}\right)/\partial S_k$, due to the negative marginal return to acreage, limits the incentives towards specialization.

In (2.8), the diversification motive is also to the implicit management cost function. With the consideration of the land constraint, Hotelling's lemma leads to:

$$\pi^{r*}_{k}\left(p_k,\mathbf{w};S^*_k,\mathbf{Z}\right)+a_k+S^*_k p_k \frac{\partial y_k\left(\mathbf{x}_k;S_k,\mathbf{Z}\right)}{\partial S_k} = \lambda'\left(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z}\right)+\frac{\partial C\left(\mathbf{S}\right)}{\partial S_k} \quad \forall k\in\left[1;K\right] \tag{2.10}$$

where λ' is the shadow price value due to land constraints. Carpentier and Letort (2012) assumed that $\partial C(\mathbf{S})/\partial S_c$ is positive such that the addition of the implicit management cost function limits diversification. The optimal land allocation is also obtained when the marginal crop-specific profits are equal, i.e., when the first-order conditions are satisfied.[18] We will see that this property is, however, questionable as management costs due to labour and/or capital can present scope economies (Dupraz, 1996).

The analyses of (2.7) and (2.8) consider the optimal acreage choices of the two main diversification motives in the acreage literature in a short-term and risk-neutral framework. The

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[18] Note that Carpentier and Letort (2012, 2014) have assumed $\partial y_k\left(\mathbf{x}_k;S_k,\mathbf{Z}\right)/\partial S_k = 0$ to ease the estimation. In this case, $\pi^{r}_{k}\left(p_k,\mathbf{w};S_k,\mathbf{Z}\right)=\pi^{r}_{k}\left(p_k,\mathbf{w};\mathbf{Z}\right)$ and the resolution of (2.8) leads to easier interpretable and estimable conditions.

crop rotation models have a quite similar structure but can only be interpreted in a dynamic framework. Similarly, I do not present the acreage models with portfolio strategy as they exist in a risk-averse framework. In the next section, I discuss the properties of the results from the "biodiversity productivity" literature.

2.3.2 Results from the "biodiversity productivity" literature and implications

Rather than focusing on the effect of S_k on π_k^r, the "biodiversity productivity" literature has more generally examined the impact of $\mathbf{S}$ on diverse economic indicators, including y_k ($\forall k \in [1;K]$), Y_k ($\forall k \in [1;K]$), Y (with $Y = \sum_k Y_k$) or Π. These effects are due to the biodiversity productive capacity, which depends on the biodiversity indicator $B(\mathbf{S})$. If the choice of the indicator is an issue in itself, the indicators used usually respect the same properties. First, other things being equal, biodiversity indicators increase with the number of crops in the farm, i.e., $\partial B(\mathbf{S})/\partial K > 0$. Second, biodiversity indicators usually respect $\partial B(\mathbf{S})/\partial S_k > 0$ and $\partial^2 B(\mathbf{S})/\partial S_k^2 < 0$, i.e., the augmentation of the area devoted to crop k increases biodiversity productive capacity in the first stage until it reaches a threshold where $B(\mathbf{S})$ decreases. Usually, the highest level of $B(\mathbf{S})$ is reached for an equally distributed acreage, meaning that the value of the threshold is $\bar{S}_k = \bar{S}/K \ \forall k \in [1;K]$. When S_k exceeds $\bar{S}_k$, k becomes one of the main crops at the farm scale, and acreage diversity is reduced. These properties express the basic idea from landscape ecology that heterogeneous landscapes are more able to support ecological functionalities than are homogeneous landscapes. Obviously, these generic properties do not necessarily apply to biodiversity in real landscapes but do not conflict with common sense when thinking of landscapes in agricultural regions specialized towards crop production, which some call "agrobiodiversity" (Baumgärtner and Quaas, 2010). These properties may be more questionable in landscapes with abundant semi-natural elements such as trees, hedgerows, permanent grasslands or extensive forests (Duflot et al., 2015; Holland et al., 2017; Martel et al., 2017). In the following, I assume that these properties are verified in agricultural landscapes.

Results from the "biodiversity productivity" literature have all determined that biodiversity indicators increase yields, whether they are crops (e.g., Di Falco et al., 2010; Donfouet et al., 2017), fodder (Finger and Buchmann, 2015) or animal productions (van Rensburg and Mulugeta, 2016). Thus, $\forall k \in [1;K]$, we have:

$$\frac{\partial y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial B(\mathbf{S})} > 0 \tag{2.11}$$

These results suggest that an increase of $B(\mathbf{S})$ leads to an increase of the vector of input ES (or a decrease of potential ecosystem disservices such as pest pressure). I now present the implications of biodiversity productive capacity results for acreage and variable input choices.

2.3.2.1 Production functions and biodiversity productive capacities

The properties of the biodiversity indicator imply a more complex relation between acreage for a specific crop and the yields of other crops. First, the number of crops in the farm increases the yields of each crop. This means that introducing an additional crop while maintaining an equally distributed acreage increases the yields of all farms' outputs.[19] The influence of the number of crops on acreage choices has not been examined in the acreage choice literature until recently. Indeed, the problem of the optimal number of crops at the farm level is characterized by a corner solution. Some recent papers have proposed Tobit-like approaches to address these censored observations (e.g., Lacroix and Thomas, 2011). However, most of them address this issue to adequately account for corner solutions in order to produce unbiased and consistent parameters. The exception is Koutchade et al. (2015), where the aim is to determine the parameters of the implicit management cost function for each possible crop combination.

Second, the distribution of the crops among the acreage influences the yields of all crops. Indeed, the derivative of $y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)$ relative to S_l ($\forall (k;l) \in [1;K] \times [1;K]$) is given by:

$$\frac{\partial y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_l} = \frac{\partial y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial B(\mathbf{S})} \frac{\partial B(\mathbf{S})}{\partial S_l} \tag{2.12}$$

[19] Note that this is not necessarily true when considering the crop production at the farm scale Y_k ($\forall k \in [1;K]$) as the new crop might substitute the previous area devoted to k. This relation is only verified at the margin.

where $y_k\left(\mathbf{x}_k;B(\mathbf{S}),\mathbf{Z}\right)$ is the production function from the production set T'''' defined by $\left(y_k,\mathbf{x}_k,B(\mathbf{S}),\mathbf{Z}\right)\in\mathrm{T}_k''''$ and where T_k'''' has the same properties as T. Given the properties of the biodiversity indicator, relation (2.12) implies that:

$$\begin{cases} S_l \in \left]0;\bar{S}/K\right] \Rightarrow \dfrac{\partial y_k\left(\mathbf{x}_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial S_l} \geq 0 \\ S_l \in \left]\bar{S}/K;\bar{S}\right[\Rightarrow \dfrac{\partial y_k\left(\mathbf{x}_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial S_l} < 0 \end{cases} \tag{2.13}$$

Relation (2.13) specifies that as long as increasing S_l increases $B(\mathbf{S})$, y_k increases with S_l. However, when the augmentation of S_l reduces $B(\mathbf{S})$, y_c is decreasing with S_l. Contrary to the acreage literature, relation (2.13) specifies that yields of k are influenced by the evolutions of the area for all the other crops. Alternatively, specialization towards l leads to a decrease in biodiversity productive capacity, implying a loss of yields for $[1;K]$. This can be interpreted as an increase in pest pressure (Altieri and Nicholls, 2004) or a loss of pollinators around the farm (Kennedy et al., 2013). In particular, we have, as in the acreage literature, that $\partial y_k/S_k \leq 0$ when k becomes an over-represented crop (once S_k exceeds $\bar{S}/K$). When k is an under-represented crop, this relationship does not hold. Assuming no marginal decreasing return to acreage, relation (2.13) means that the production $Y_k\left(\mathbf{X}_k,S_k;B(\mathbf{S}),\mathbf{Z}\right)=S_k\times y_k\left(\mathbf{x}_k;B(\mathbf{S}),\mathbf{Z}\right)$ at the farm scale could be distinguished in three stages (Zilberman, 2004). Indeed, we have:

$$\begin{cases} S_k \in \left[0;\bar{S}/K\right] \Rightarrow \begin{cases} \dfrac{\partial Y_k\left(\mathbf{X}_k,S_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial S_k} > 0 \\ \dfrac{\partial Y_k{}^2\left(\mathbf{X}_k,S_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial S_k{}^2} > 0 \end{cases} \\ S_k \in \left]\bar{S}/K;\bar{S}\right] \Rightarrow \begin{cases} \dfrac{\partial Y_k\left(\mathbf{X}_k,S_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial S_k} > 0 \\ \dfrac{\partial Y_k{}^2\left(\mathbf{X}_k,S_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial S_k{}^2} < 0 \end{cases} \end{cases} \tag{2.14}$$

Relation (2.14) implies that Y_k reaches a maximum between $\left]\bar{S}/K;\bar{S}\right]$ for a crop-specific value $\widehat{S}_k$, determined such as the FOC of $Y_k\left(\mathbf{X}_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)$ relative to S_k is null, i.e.

$$\widehat{S}_k = y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)\left[\frac{\partial y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial B(\mathbf{S})}\frac{\partial B(\mathbf{S})}{\partial S_k}\right]^{-1} \tag{2.15}$$

Relation (2.14) implies that, during the first stage $\left[0;\bar{S}/K\right]$, the marginal productivity of S_k is positive and higher than its average productivity. The second stage $\left]\bar{S}/K;\widehat{S}_k\right]$, the marginal productivity of S_k is positive but lower than its average productivity. The third stage $\left]\widehat{S}_k;\bar{S}\right]$ is characterized by negative marginal productivity. As underlined by Zilberman (2004), the second stage is the economic region, characterized by positive but decreasing marginal productivity. This second stage is thus coherent with the production economics theory and, in particular, with the acreage literature. The possible "ecological" interpretation of the first stage is that the first units of land devoted to k increases the suitable habitat for bees or specific insects involved in biological control. The second stage may be interpreted as the loss of yield due to pest pressure, but this loss is compensated by an increase in S_k, meaning that the extensive margin effect (equal to $y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)$) dominates the intensive margin effect (equal to $S_k \times \left(\partial y_k\left(\mathbf{x}_k; B(\mathbf{S}), \mathbf{Z}\right)/\partial S_k\right)$). In the third stage, the increase of S_k no longer compensates for the loss of yields in the other areas, i.e., the loss of yields. The intensive margin effect dominates the extensive margin effect. This last stage represents the impact of monoculture on yields. This third stage may not appear at the farm scale. Indeed, some examples, such as the maize monoculture in southwestern France or the cacao monoculture in the Ivory Coast, indicate that some crops can present $\widehat{S}_k > \bar{S}$, i.e., that the corresponding crop yields are not too sensitive to $B(\mathbf{S})$.

2.3.2.2 Variable input demand and biodiversity productive capacity

In addition to its effects on crop yields, biodiversity productive capacity can also influence variable input applications. If the "biodiversity productivity" literature does not provide solid evidence of a possible substitution between biodiversity and variable input,[20] specialized literatures such as agronomy (Altieri and Nicholls, 2004), ecology (Geiger et al., 2010) or applied economic literature on soil conservation (Kim et al., 2000) or crop rotation (Hennessy, 2006; Thomas, 2003) do. Notably, it appears that biodiversity productive capacities can be substituted with fertilizers and pesticides. The technical interpretation of such substitution with pesticide is that crop diversity breaks the evolution of pest populations and increases the number of suitable insects involved in biological control. One technical interpretation of the substitution between biodiversity productive capacity and fertilizer is that it provides complementary relationships between species, which enhances nitrogen fixation. Overall, we observe input savings from biodiversity productive capacity, and we have $\partial \mathbf{x}_k / \partial B(\mathbf{S}) \leq 0$. First, like production, the properties of the biodiversity indicator imply that the introduction of an additional crop while maintaining an equally distributed acreage decreases variable input use on all farms' outputs. Second, we can specify the following:

$$\begin{cases} S_l \in \left[0; \bar{S}/K\right] \Rightarrow \dfrac{\partial \mathbf{x}_k\left(y_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_l} \leq 0 \\ S_l \in \left]\bar{S}/K; \bar{S}\right] \Rightarrow \dfrac{\partial \mathbf{x}_k\left(y_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_l} > 0 \end{cases} \quad \forall (k,l) \in [1;K] \times [1;K] \tag{2.16}$$

where $\mathbf{x}_k\left(y_k; B(\mathbf{S}), \mathbf{Z}\right)$ is the vector of required inputs to produce any y_k derived from T_c'''. $\mathbf{x}_k\left(y_k; B(\mathbf{S}), \mathbf{Z}\right)$ can include y_k null but cannot include null vector if y_k is non-null. Relation (2.16) specifies that $\mathbf{x}_k$ decreases with S_l until S_l reaches $\bar{S}/K$, where, above this point, augmentation of S_l increases $\mathbf{x}_k$. This second stage represents the effect of crop specialization. Like (2.14), we can also study the relation between $\mathbf{X}_k\left(Y_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)$ and S_k:

[20] Di Falco and Chavas (2006) do find a negative interaction between biodiversity productive capacity and pesticide applications on the variance of yields. Unfortunately, they did not examine such interactions on mean yields.

$$\begin{cases} S_k \in \left[0;\bar{S}/K\right] \Rightarrow \begin{cases} \dfrac{\partial \mathbf{X}_k\left(Y_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_k} \lesseqgtr 0 \\ \dfrac{\partial \mathbf{X}_k\left(Y_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_k{}^2} \lesseqgtr 0 \end{cases} \\ S_k \in \left]\bar{S}/K;\bar{S}\right] \Rightarrow \begin{cases} \dfrac{\partial \mathbf{X}_k\left(Y_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_k} > 0 \\ \dfrac{\partial \mathbf{X}_k\left(Y_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_k{}^2} > 0 \end{cases} \end{cases} \tag{2.17}$$

If we know that $\mathbf{X}_k$ increases convexly once S_k exceeds $\bar{S}/K$ and until it reaches a monoculture (where the farmer should apply its highest amount of variable input to maintain Y_k), we do not know the form of the function $\mathbf{X}_k\left(Y_k, S_k; B(\mathbf{S}), \mathbf{Z}\right)$ before $\bar{S}/K$. The function is either increasing and concave or decreasing and convex. The form of the function depends on the amplitude of the intensive margin effect, i.e., on the amplitude of $\partial \mathbf{x}_k / \partial S_k$. If $\partial \mathbf{x}_k / \partial S_k > 1$, then $\mathbf{X}_k$ is decreasing and convex, and the minimal application of variable input is reached for $\breve{S}_c > 0$. Otherwise, when $\partial \mathbf{x}_k / \partial S_k \leq 1$, $\mathbf{X}_k$ is increasing and quasi-concave, and the minimal application of variable input is reached for $\breve{S}_k^{'} = 0$.

2.3.2.3 Properties of gross margins with biodiversity productive capacity

As $B(\mathbf{S})$ increases yields and reduces variable input uses, we can determine that biodiversity productive capacity influences the gross margin $\pi_c^r\left(p_c, \mathbf{w}; B(\mathbf{S}), \mathbf{Z}\right)$. First, for positive $(p_c, \mathbf{w})$, the properties of the biodiversity indicator imply that the introduction of an additional crop while keeping for an equally distributed acreage increases the gross margins of all farms' outputs. Second, for positive $(p_c, \mathbf{w})$, relations (2.13) and (2.16) lead to:

$$\begin{cases} S_l \in \left[0;\bar{S}/K\right] \Rightarrow p_c \dfrac{\partial y_c\left(\mathbf{x}_c; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_c} - \mathbf{w}' \dfrac{\partial \mathbf{x}_c\left(y_c, \mathbf{w}; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_c} = \dfrac{\partial \pi_k^r\left(p_k, \mathbf{w}; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_l} \geq 0 \\ S_l \in \left]\bar{S}/K;\bar{S}\right] \Rightarrow p_c \dfrac{\partial y_c\left(\mathbf{x}_c; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_c} - \mathbf{w}' \dfrac{\partial \mathbf{x}_c\left(y_c, \mathbf{w}; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_c} = \dfrac{\partial \pi_k^r\left(p_k, \mathbf{w}; B(\mathbf{S}), \mathbf{Z}\right)}{\partial S_l} < 0 \end{cases} \tag{2.18}$$

For given $(p_c, \mathbf{w})$, the maximum π_k^r is reached when the acreage is equally distributed among crops. However, the relationship for the crop-specific profit function at the farm scale Π_k^r

displays complex relationships. Indeed, the maximum of Y_c is reached at $\widehat{S}_c$, and the minimum of $\mathbf{X}_c$ is reached at $\breve{S}_c$. As $\widehat{S}_c > \bar{S}/K$ and $\breve{S}_c < \bar{S}/K$, there is no simple maxima for Π_k^r.[21] The maxima depend on the marginal productivity and marginal input saving but also on $(\mathbf{p},\mathbf{w},\mathbf{a})$.

2.3.2.4 Biodiversity productive capacity and total area constraint

Farms face a total land constraint and must allocate their lands towards different outputs. Here, I first examine the marginal effect of S_k change on $B(\mathbf{S})$ when the total land constraint is included. I label this indicator $B(\mathbf{S};\bar{S})$. $B(\mathbf{S};\bar{S})$ evolves not only according to dS_k but also according to the whole indirect acreage reorganization induced by dS_k. The evolution of $B(\mathbf{S};\bar{S})$ due to dS_k is:

$$\frac{\partial B(\mathbf{S};\bar{S})}{\partial S_k} = \frac{\partial B(\mathbf{S})}{\partial S_k} + \sum_{l=1}^{K} \frac{\partial B(\mathbf{S})}{\partial S_l(\bar{S})} \frac{\partial S_l(\bar{S})}{\partial S_k} \tag{2.20}$$

with $\partial S_l / \partial S_k \le 0 \quad \forall l \in [1;K] - \{k\}$, i.e., the increase of S_k reduces the available areas for the other crops. An increase in S_k implies that the farmer chooses to reduce the area devoted to at least one other crop. If we know that the total land constraint implies that $dS_k = \sum_{\substack{l \in [1;K] \\ l \neq k}} dS_l$, the evolution of S_l according to change in S_k is impossible to derive because there is an infinity of solutions. In this case, the usual strategy is to arbitrarily determine a reference crop to "close" the total land constraint. Defining the reference crop as K, I can write (2.20) as:

$$\frac{\partial B(\mathbf{S};\bar{S})}{\partial S_k} = \frac{\partial B(\mathbf{S})}{\partial S_k} - \frac{\partial B(\mathbf{S})}{\partial S_K} \quad \forall k \in [1; K-1] \tag{2.21}$$

Relation (2.21) means that the evolution of the biodiversity indicator under constraint is equal to the marginal evolution of $B(\mathbf{S})$ due to the evolution of S_k minus its marginal evolution due

[21] In practice, the maxima is reached when $\frac{\partial Y_k(\mathbf{X}_k, \dot{S}_k; B(\mathbf{S}), \mathbf{Z})}{\partial \mathbf{X}_k(Y_k, \dot{S}_k; B(\mathbf{S}), \mathbf{Z})} = \frac{\mathbf{w}}{p_k}\left[1 - a_k \Big/ \left(\mathbf{w}\frac{\partial \mathbf{X}_k(Y_k, \dot{S}_k; B(\mathbf{S}), \mathbf{Z})}{\partial S_k}\right)\right]$

to the compensated effect of S_K. I discuss the implications of total land constraints on yield, input application and crop profit.

Using (2.21) and previous properties, the consequence of the total land constraint for yields is:

$$\frac{\partial y_k\left(\mathbf{x}_k;B\left(\mathbf{S};\bar{S}\right),\mathbf{Z}\right)}{\partial S_l}=\frac{\partial y_k\left(\mathbf{x}_k;B\left(\mathbf{S}\right),\mathbf{Z}\right)}{\partial B\left(\mathbf{S}\right)}\left(\frac{\partial B\left(\mathbf{S}\right)}{\partial S_l}-\frac{\partial B\left(\mathbf{S}\right)}{\partial S_K}\right)\quad \forall\left(k,l\right)\in\left[1;K\right]\times\left[1;K-1\right] \tag{2.22}$$

The yield of crop k depends on the direct effect of S_l on $B(\mathbf{S})$ and the "compensated effect" on $B(\mathbf{S})$ due to acreage reorganization. This compensated effect complicates the analysis of the effect of S_l on y_k. Indeed, if S_l has a positive direct impact on $y_k(\mathbf{x}_k;B(\mathbf{S}),\mathbf{Z})$ until S_k reaches $\bar{S}/K$, the sign of the impact of the compensated effect depends on the initial value of S_K. If S_K was strictly higher than $\bar{S}/K$, the impact of the compensated effect on y_k is positive but negative otherwise. Similarly, we have:

$$\frac{\partial \mathbf{x}_k\left(y_k;B\left(\mathbf{S};\bar{S}\right),\mathbf{Z}\right)}{\partial S_l}=\frac{\partial \mathbf{x}_k\left(y_k;B\left(\mathbf{S}\right),\mathbf{Z}\right)}{\partial B\left(\mathbf{S}\right)}\left(\frac{\partial B\left(\mathbf{S}\right)}{\partial S_l}-\frac{\partial B\left(\mathbf{S}\right)}{\partial S_K}\right)\quad \forall\left(k,l\right)\in\left[1;K\right]\times\left[1;K-1\right] \tag{2.23}$$

Like y_k, the evolution of $\mathbf{x}_k(y_k;B(\mathbf{S}),\mathbf{Z})$ due to S_l depends on the initial levels of both S_l and S_K. Finally, the derivative of $\pi_k^r\left(p_k,\mathbf{w};S_k,B\left(\mathbf{S};\bar{S}\right),\mathbf{Z}\right)$ yields $\forall\left(k,l\right)\in\left[1;K\right]\times\left[1;K-1\right]$:

$$\frac{\partial \pi_k^r\left(p_k,\mathbf{w},\mathrm{a}_k;B(\mathbf{S};\bar{S}),\mathbf{Z}\right)}{\partial S_l}=\left(p_k\frac{\partial y_k\left(\mathbf{x}_k;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B\left(\mathbf{S}\right)}-\mathbf{w}'\frac{\partial \mathbf{x}_k\left(y_k;B\left(\mathbf{S}\right),\mathbf{Z}\right)}{\partial B\left(\mathbf{S}\right)}\right)\left(\frac{\partial B\left(\mathbf{S}\right)}{\partial S_l}-\frac{\partial B\left(\mathbf{S}\right)}{\partial S_K}\right) \tag{2.24}$$

The relation between π_k^r and S_l depends on the initial levels of both S_l and S_K.

2.3.2.5 Profit and optimal acreage with biodiversity productive capacity

Here, I present a single profit function with similar properties to (2.1). This function is inspired by Carpentier and Letort (2012, 2014), who use a specific function (2.8) where crop-specific profits do not depend directly on $\mathbf{S}$. Here, I consider that crop-specific profits depend on biodiversity productive capacity, i.e., depend indirectly on $\mathbf{S}$. Using Carpentier and Letort (2012, 2014), we define the following restricted profit function:

$$\Pi^r\left(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{S},B(\mathbf{S}),\mathbf{Z}\right)=\sum_{k=1}^{K}S_k\pi_k^r\left(p_k,\mathbf{w},a_k;B(\mathbf{S}),\mathbf{Z}\right)-C(\mathbf{S};\mathbf{Z})$$
$$\text{s.t.}\sum_{k=1}^{K}S_k=\bar{S} \quad (2.25)$$

where $C(\mathbf{S};\mathbf{Z})$ is the "implicit cost" function of acreage, which increases with S_k. Usually, this function captures all the constraints and incentives that farmers face when trying to diversify their acreage. Here, I interpret $C(\mathbf{S};\mathbf{Z})$ as the "implicit cost" function to manage biodiversity productive capacity. Indeed, as the model captures the benefits of crop diversity through the effects of $B(\mathbf{S})$ on the π_k^r, I should have removed the benefits of crop diversity from $C(\mathbf{S};\mathbf{Z})$. Consequently, I define the sum of the effects of $B(\mathbf{S})$ on π_k^r as the benefits of biodiversity. The maximization of (2.25) on S_k leads to:

$$\pi_k^{r*}\left(p_k,\mathbf{w};B(\mathbf{S};\bar{S}),\mathbf{Z}\right)+a_k+\sum_{l=1}^{K}S_l\left(p_l\frac{\partial y_l\left(\mathbf{x}_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}-\mathbf{w}'\frac{\partial \mathbf{x}_l\left(y_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}\right)\left(\frac{\partial B(\mathbf{S})}{\partial S_k}-\frac{\partial B(\mathbf{S})}{\partial S_K}\right)$$
$$=\lambda''\left(\mathbf{p},\mathbf{w},\mathbf{a};\mathbf{Z}\right)+\frac{\partial C(\mathbf{S};\mathbf{Z})}{\partial S_k} \quad (2.26)$$

where λ'' is the shadow price value due to land constraint and π_k^{r*} is the optimized gross margin of the problem $\text{argmax}_{y_k,\mathbf{x}_k}\left\{p_k y_k-\mathbf{w}'\mathbf{x}_k\right\}$. The effects at the extensive margins are similar to those in (2.10), but those at the intensive margins are different. Indeed, whereas the acreage choice literature has considered that the intensive margins to acreage was $S_k p_k\,\partial y_k\left(\mathbf{x}_k;S_k,\mathbf{Z}\right)/\partial S_k$ at best, I consider that S_k also impacts $\mathbf{x}_l$ ($\forall l\in[1;K]$), i.e., that the management of acreage choice leads to potential gains in yields and input savings for all outputs and variable inputs. Relation (2.26) shows that the evolution of crop price indirectly influences the profitability of other crops. Indeed, the augmentation of a specific crop price p_k leads to the augmentation of both y_k (due to more variable input application) and S_k, which indirectly modifies biodiversity productive capacity and the predictability of all other crops. More particularly, the augmentation of subsidies devoted to one specific type of land modifies the marginal profitability of other crops. These relations highlight precautions required of policymakers designing new instruments. Because λ'' is the same for each output, the optimal land allocation is obtained when relations (2.26) are equal for each output. Relation (2.27) presents the optimal acreage conditions:

$$\begin{cases}
\pi_1^{r*}\left(p_1,\mathbf{w};B(\mathbf{S};\bar{S}),\mathbf{Z}\right)+a_1+\sum_{l=1}^{K}S_l\left(p_l\frac{\partial y_l\left(\mathbf{x}_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}-\mathbf{w}'\frac{\partial \mathbf{x}_l\left(y_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}\right)\left(\frac{\partial B(\mathbf{S})}{\partial S_1}-\frac{\partial B(\mathbf{S})}{\partial S_K}\right)-\frac{\partial C(\mathbf{S};\mathbf{Z})}{\partial S_1} \\
= \\
\ldots \\
= \\
\pi_k^{r*}\left(p_k,\mathbf{w};B(\mathbf{S};\bar{S}),\mathbf{Z}\right)+a_k+\sum_{l=1}^{K}S_l\left(p_l\frac{\partial y_l\left(\mathbf{x}_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}-\mathbf{w}'\frac{\partial \mathbf{x}_l\left(y_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}\right)\left(\frac{\partial B(\mathbf{S})}{\partial S_k}-\frac{\partial B(\mathbf{S})}{\partial S_K}\right)-\frac{\partial C(\mathbf{S};\mathbf{Z})}{\partial S_k} \\
= \\
\ldots \\
= \\
\pi_K^{r*}\left(p_K,\mathbf{w};B(\mathbf{S};\bar{S}),\mathbf{Z}\right)+a_K-\frac{\partial C(\mathbf{S};\mathbf{Z})}{\partial S_K}
\end{cases} \quad (2.27)$$

The choice of optimal acreage $\mathbf{S}^*$ depends on the gross margins and the subsidies of each output and on the costs and benefits of biodiversity productive capacity. The benefits of biodiversity productive capacity depend on additional yields ($\partial y_k / \partial B(\mathbf{S})$) and on variable input savings ($\sum_i \partial x_{ik} / \partial B(\mathbf{S})$) for each crop, i.e., on the sensitivity of each crop to biodiversity productive capacity. This result is a special case of the well-known results that producers' choices depend on the form of the technology. The costs linked to the management of biodiversity productive capacity depend on the form of $C(\mathbf{S};\mathbf{Z})$ according to acreage choices. These costs can have concave or convex form according to the considered input. For example, a labour output-specific labour peak encourages diversification, whereas output-specific capital encourages specialization. For example, maize requires costly irrigation systems, which are not especially useful for other crops. The costs of management of biodiversity productive capacity thus depend on farmers' long-term choices $\mathbf{Z}$. The range of $\mathbf{Z}$ values illustrates a part of the diversity of agricultural systems and may explain the differences between low-capital-intensive farms that have incentives to manage biodiversity productive capacity (e.g., permaculture, organic farming) and high-capital-intensive farms whose biodiversity management costs discourage them from managing biodiversity.

Under the assumption that input ES depend on land use and that farmers maximize their profit, relation (2.27) provides a framework to analyse farmers' management of input ES. Relation (2.27) expressly links $B(\mathbf{S})$ with economic context. In particular, it illustrates that a marginal change to a crop-specific subsidy or price influences the profitability of all other crops through two channels: the modification of the vector of input ES that influences π_k (through additional yields and input savings) and the modification of the allocation of fixed inputs that incurred costs at the farm level. In this particular case, all crops are equally profitable and subsidized (i.e., $\left(\pi_k^{r*} + a_k\right)$ are equal $\forall k \in [1;K]$), relation (2.27) states that the farmers make the marginal benefit of biodiversity productive capacity equal to the marginal cost of biodiversity management, i.e.,:

$$\sum_{k=1}^{K}\left[\sum_{l=1}^{K} S_l\left(p_l\frac{\partial y_l\left(\mathbf{x}_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})} - \mathbf{w}'\frac{\partial \mathbf{x}_l\left(y_l;B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})}\right)\left(\frac{\partial B(\mathbf{S})}{\partial S_k} - \frac{\partial B(\mathbf{S})}{\partial S_K}\right)\right] = \sum_{k=1}^{K}\frac{\partial C(\mathbf{S};\mathbf{Z})}{\partial S_k} \quad (2.28)$$

In fact, one can consider that relation (2.27) shares the basic settings of Orazem and Miranowski (1994) but in a static framework. Orazem and Miranowski (1994) consider both the marginal benefits and the marginal costs of crop rotations, with FOC values similar to those of (2.26) in a two-period framework. In their framework, the benefits of crop rotations are due to technical relationships (depending on $\mathbf{S}_{t-1}$), and the marginal costs are due to adjustment costs linked to the management of fixed inputs (depending on $\mathbf{S}_t$). However, most acreage works are conducted in a static framework and are thus consistent with my static framework on the management of input ES.[22]

More generally, relation (2.27) decomposes the sources of jointness faced by multioutput farms between the presence of biodiversity productive capacity, a public input at the origin of technical complementarities between crops, and the presence of allocable fixed input (see appendix 2.C. for a discussion of joint production in the considered framework). This decomposition illustrates the sources of scope economies faced by multioutput farms, which is a dynamic area of research in agricultural economics (Blancard et al., 2011, 2016; Chavas and Kim, 2010; Lansink and Stefanou, 2001). I believe that my structural approach can help interpret the differences arising from the measure of scope economies on reduced-form

[22] In addition, the effects of biodiversity productive capacity seem to be more important in the current year than in the following years (Di Falco and Chavas, 2008).

equations, which sometimes lead to opposite results, e.g., Chavas and Aliber (1993) underline scope economies, Blancard et al. (2011) underline scale economies and Lansink and Stefanou (2001) measure both scope and scale economies.

2.4 Discussion

This chapter presents a general framework to analyse the management of input ES by farmers at the farm scale and in a certain and static framework. This framework enables analysis of the acreage choices when these choices define the quality of biodiversity habitat at the farm scale and when biodiversity levels generate vectors of potential input ES. This framework can be useful to both the acreage choice and biodiversity productive capacity literature and in evaluating agroenvironmental policies. However, our framework is subject to several drawbacks linked to (i) the assumptions, (ii) the empirical issues and (iii) spatio-temporal specificities of ecosystem services.

2.4.1 Contributions

2.4.1.1 Contributions to the literature on the productivity of biodiversity

My model contributes to the literature on the productivity of biodiversity for three reasons. First, the FOC (2.27) can be estimated using a structural approach. The appeal of the structural approach is that it enables the estimation of all the mechanisms at stake in different interlinked equations. In contrast, the literature on the productivity of biodiversity has primarily relied on the estimation of reduced-form equations with the risk that the estimated parameters may capture different processes at the same time. For example, crop diversity may influence both the output-specific margins (through modification of input ES) and the costs linked to the management of fixed inputs. Therefore, an estimation of the marginal effect of crop diversity indicators into a single profit function may capture the two effects, preventing any conclusions on the real profitability of input ES. The estimation of the underlying structural model can shed light on the strengths of the different processes at stake.

Second, my framework characterizes some sources of potential endogeneity linked to biodiversity indicators that have not been identified by the literature on the productivity of biodiversity. In particular, acreage choices depend on the economic context and, notably, on prices. Authors have usually used a variety of variables to instrument biodiversity indicators, including the lagged value of the biodiversity index (Di Falco and Chavas, 2008, 2009), the distance with the closest input supplier (Chavas and Di Falco, 2012; Di Falco and Chavas, 2009), the distance from the nearest road and city (Di Falco and Chavas, 2009), the distance between plots and farms (Di Falco et al., 2010) or the shares of land cover categories with high biodiversity potential (Donfouet al., 2017). None of these papers have, to my knowledge, tried to correct for the effects of prices; thus, their productivity measures are suspected to still suffer from some endogenous biases.

Finally, if the literature provides estimations of the contribution of input ES to agricultural productivity, we do not know if farmers manage biodiversity and linked input ES or if they benefit from unanticipated productive externalities. Here, the proposed framework informs the optimal management of input ES by a rational and informed farmer. The estimation of model (2.27) can inform this management. I will return to this point later in the discussion.

2.4.1.2 Contributions to the literature on acreage choices

My framework can serve the literature on farmers' acreage choices for four reasons. First, the FOC (2.27) highlights that farmers make decisions that integrate the feedback effects on output profitability. In other words, relation (2.27) represents the farmers' management of input ES, or, from a more practical point of view, represents the program of an "agroecological" farmer (in the French spirit of "agroecology"). The single motive for the integration of such feedback effects in the acreage choice literature relates to the issue of decreasing return to scale, i.e., negative feedback. In a sense, my framework is a generalization of such effects on all outputs: the acreage choice for one crop modifies the probability of all the other crops, and the feedback effects can be positive or negative. This generalization is due to the improvement of the representation of agricultural technology due to the introduction of explicit links between acreage choices and input ES.

Second, relation (2.27) highlights that several causes may explain diversification. Here, I have decomposed the pure technical complementarities that are due to a public input (the biodiversity productive capacity $B(\mathbf{S})$) and the allocable fixed input problem. The proposed model may contribute to the analysis of the relative importance of the two causes, a question that remains largely unexplored (see appendix 2.C.). The distinction of the two sources is also crucial when investigating different policy instruments, notably for the question of variable input savings.

Third, my model decomposes between the impacts on yields and the impacts on variable input applications. The analysis of alternative practices to save variable input is a central point for the future of agriculture. Indeed, variable input applications are the source of several local pollutants. The fertilizer issue has received major attention from policymakers, with several successive reforms in Europe aiming to improve water quality. The pesticide issue is at the heart of many ongoing debates, with crucial questions on the impact of such chemicals on the environment and health. The required conditions to achieve variable input savings constitute an essential question for agricultural economists. My framework may contribute to identifying such conditions because it provides more details on the role of input ES in agricultural technologies. The improvement of the representation of the technology provides fine-grained details on farmers' behaviour in multi-output farms.

Fourth, relation (2.27) highlights that the farmers' fields are not independent from each other. This independence exists at the farm scale and possibly also at the landscape scale. Indeed, if acreage choices modify the level of input ES and those input ES modify the profitability of outputs, the farmers' acreage choices generate productive externalities for neighbouring farms. This feature has led to simulation works on coordinated management of productive ES at the landscape scale (e.g., Cong et al., 2014) but remains largely unexplored, notably by agricultural economists.

2.4.1.3 Contributions to policy evaluations

The proposed framework is well suited to analysing the impacts of new policy instruments. Indeed, the model informs variable input applications, acreage choices and on-farm biodiversity levels. It has interesting implications for usual agricultural policy instruments. For example, the evolution of a crop-specific area subsidy indirectly influences the profitability of other crops due to technical jointness: the augmentation of a specific crop price a_k leads to the augmentation of S_k, which modifies biodiversity productive capacity and thus modifies y_l and $\mathbf{x}_l$ ($\forall l \in [1;K]$). Thus, a crop-specific area subsidy may lead to an indirect negative impact on pesticide or fertilizer applications, depending on the initial reference acreage. Another example relates to taxes on variable input applications, such as pesticide taxation schemes, which is a deeply investigated question in agricultural economics (Femenia and Letort, 2016; Finger et al., 2017). The proposed model indicates that a tax on variable input applications would have an impact on acreage because rational profit-maximizing farmers will reorganize acreage to improve biodiversity levels and benefit from input savings. In other words, my model specifies an alternative for farmers, which may improve the effectiveness of a variable input tax and reduce farmers' economic losses. Finally, relation (2.27) shows that acreage choices that consider biodiversity productive capacity still depend on these fixed factors $\mathbf{Z}$ (the short-term choices still depend on long-term choices). Policy instruments aiming to modify the price of quasi-fixed inputs can thus influence the biodiversity of farms. These relations highlight that policymakers designing new instruments must exercise caution.

Finally, assuming that biodiversity indicators are also positively correlated with non-input ES, which are valorised by non-farmer agents, the proposed model could be used to analyse the effectiveness of policy instruments to reach the optimal level of on-farm biodiversity. Such policy instrument evaluations with endogenous environmental state variable levels are common in environmental economics (e.g., Laukkanen and Nauges, 2014; Mouysset et al., 2014). The development of this framework aims to identify the most efficient instruments to orient farmers' choices towards an optimal level of biodiversity. Here, the appeal of the proposed model is that it also integrates the productive feedback effects of biodiversity evolution. Indeed, if previous studies have integrated the costs of providing biodiversity or reducing environmental damage, they do not integrate the opportunity costs that may depend on biodiversity/environmental state variables. The proposed model illustrates such effects: an increase (decrease) in on-farm biodiversity levels improves (decreases) the productivity effects, which decreases (increases) the farmers' opportunity costs of improving on-farm biodiversity levels. Thus, on-farm biodiversity provisions may face (dis)economies of scale. To my knowledge, such a closed loop has not been considered in the literature, implying that the previous conclusions may be flawed.

2.4.2 The crucial assumptions

2.4.2.1 Land-based ecosystem services

The crucial assumption with regard to input ES is that their expression depends only on acreage choices. These ES are called "land-based" ES in the literature (Müller et al., 2016) and represent the majority of the ES literature. This assumption enables approximating input ES using biodiversity indicators, which are computed using landscape composition. The consequence is that input ES do not depend on farmer practices at the field scale. However, several works underline that the levels of ES provision depend negatively on the intensity of agricultural inputs per unit of land (e.g., fertilizers and pesticides by hectare, or cow per ha) but positively on biodiversity-friendly practices (e.g., reduced tillage practices). These practices modify the expression of ES (Le Coeur et al., 2002). These practices are the results of farmers choosing to enhance the expression input ES to the detriment of conventional inputs. In this thesis, I do not integrate the impact of conventional inputs on the level of biodiversity productive capacity, but I do consider that biodiversity productive capacity impacts the productivity of conventional inputs (see Chapter 3) and the application of conventional inputs (see Chapter 4). Because I have assumed that input ES depend only on land use, it may be possible that our framework is too specific to be reliable. To my knowledge, Omer et al. (2007) have been the only ones to measure biodiversity productivity considering that biodiversity levels depend on conventional input applications. Other authors, such as Brunetti et al. (2018), have proposed a theoretical framework in which biodiversity is an input that depends negatively on farmers' conventional input applications.

In the PhD manuscript, I consider that biodiversity productive capacity is a vector of potential input ES that are complementary between each other. The complementarity provision of such ES is a source of debate in the ecological literature (Müller et al., 2016; Power, 2010), but most of the empirical works on biodiversity indicators highlight complementary ES provision (Gardiner et al., 2009; Garibaldi et al., 2016; Kennedy et al., 2013; Letourneau et al., 2011; Mäder et al., 2002; Morandin et al., 2014). Nevertheless, the consequence is that I do not investigate the management of one specific type of input ES. This may be a limit of my framework because the diverse ES do not process at the same scale and they surely involve different behaviours.

2.4.2.2 Ecosystem services: the scale issue

Zhang et al. (2007) distinguish ES according to their scale of provision. According to them, input ES can be provided at the field, farm and landscape scale. The proposed framework is well suited to analysing farm-scale management of input ES thanks to on-farm biodiversity. However, it does not fit the two other scales. In particular, I do not examine the behaviour of farmers at the field scale. The management of the expression of input ES is studied not only by agronomists (see research on no-tillage) but also by economists (Wu and Babcock, 1998). Historically, economists have investigated the impact of erosion on agricultural productivity as well as strategies to limit erosion, notably for the implementation of conservation agriculture practices with the triptych of no-tillage, crop rotation and crop covers (Barbier, 1990; Hediger, 2003; Kim et al., 2000). In this framework, erosion can be seen as a limited mechanism in the expression of input ES, and the implementation of conservation agriculture corresponds to an investment in soil ES. A larger body of literature has focused on the effects of crop rotation at the field scale (Eckstein, 1984; Hendricks et al., 2014; Hennessy, 2006; Livingston et al., 2015).

Economists have paid explicit attention to soil ES in recent works, with similar references to productive and insurance values (Pascual et al., 2015). However, farmers' behaviours at the field scale are different, with deeper attention being paid to the trade-off between the decisions in the short and long terms, the theoretical results as they depend on time preferences, and technical relationships with conventional inputs (Issanchou et al., 2018). Contrary to the ES provided field scale, my framework can be adapted to ES provided at the landscape scale (see chapter 5).

2.4.2.3 Assumptions from the acreage choices framework

Like most of the literature on acreage choices, our framework relies on restrictive assumptions. First, I have assumed that farmers are fully aware of the effect of acreage choices on input ES. This assumption is surely incorrect. Indeed, studies on the relationship between input ES and landscape structure are still ongoing and appear to be rather complex. For example, Martel et al. (2017) find that hedgerow density positively influences the abundance of some carabid beetles but negatively influences some other carabid beetle species, making the management of carabid beetles for biological control rather complex. In addition, it appears that farmers have good knowledge about input ES but consider that most ES are moderately manageable (Smith and Sullivan, 2014). Note that Brunetti et al. (2018) proposed a theoretical model in which farmers are myopic agents who ignore the beneficial effects of biodiversity productive capacity, leading to interesting implications for policy instruments (e.g., a tax on inputs could increase profits).

Second, I have considered a static framework. Due to the annual nature of acreage choices, there are no existing frameworks analysing the long-term choices of acreage choice, or frameworks may do so only with a two-period framework (Hennessy, 2006; Lansink and Stefanou, 2001; Orazem and Miranowski, 1994; Thomas, 2003). However, the modification of quasi-fixed input levels in the long term modifies acreage choices, as illustrated by the implicit cost function in (2.27). In addition, the input ES are characterized by dynamic processes, which may influence farmers' behaviour in the long term, e.g., crop rotations and soil conservation practices. I propose an extension including dynamic in chapter 4.

Third, I have considered a certain framework. This approach may be an issue if farmers manage input ES to reduce production risk. In such cases, my structural approach may lead to biased parameters. The literature on acreage choices has proposed extensions to accommodate uncertainty and incertitude, for example, based on the theory of expected utility of profit

(Chavas and Holt, 1990; Sckokai and Moro, 2006), but these works also rely on restrictive assumptions, notably the form of price anticipations.

Finally, the proposed framework is only correct at the farm scale and if farms are spatially continuous and sufficiently large to manage ES. In reality, farms face farm fragmentation (Latruffe and Piet, 2014) and thus suffer from spillovers due to the acreage choices of their neighbours. In Chapter 5, I present a framework to analyse the management of ecosystem services among several farms. I succinctly present the motivations of such extensions in the third part of this section.

2.4.3 Empirical applications and extensions

I estimate the underlying model of (2.27) in chapter 4 using the empirical framework provided by Carpentier and Letort (2012, 2014). The appeal of their approach is that it easily integrates technical terms in supply and demand functions. Given the properties of the different indicators used (see Appendix 2.B), I use the Simpson index as the biodiversity indicator. As in most of the "biodiversity productivity" literature, I have considered a farm-scale, certain and static framework. This framework can be adapted to accommodate spatial, temporal and uncertain features, such issues arising in ecosystem functioning. I integrate these effects in the following chapters.

In chapter 3, I develop the presented general model by focusing on several types of biodiversity components, namely, those attached to crop on-farm biodiversity and those attached to permanent grasslands; permanent grasslands and attached landscape elements are considered as rich specific ecosystems (Baudry et al., 2000). The purpose of this chapter is to pay deeper attention to the properties of agricultural technologies when several types of input ES are considered. Indeed, papers on biodiversity productive capacity consider a single component of biodiversity and do not examine how several components can interact. They also do not focus on the productive interactions of input ES with conventional inputs, which are, however, a crucial point when considering variable input savings. The empirical counterpart is that I consider a very short-term optimization process, considering that land-use choices are already made.

In chapter 4, I estimate the presented model in a dynamic framework, considering biodiversity productive capacity as a special case of capital. Indeed, Di Falco and Chavas (2008) have estimated that the effects of current biodiversity indicators on the agricultural productivity of future periods represent 41% of current productive effects. Therefore, a rational and fully informed farmer integrates these future effects. The basic modification in (2.27) is that farmers integrate the benefits of current biodiversity productivity capacity levels in future periods, i.e., on future yields and future variable input applications. I also estimate the presented model in a static framework and Femenia and Letort (2016)'s model in a sample of French farms from "La Meuse" CER accounting agency.

In chapter 5, I investigate the benefits of collective management of input ES using simulations from an agent-based model. Starting with Cong et al. (2014), the collective management of input ES represents a growing literature that assumes that farmers manage input ES and perform simulation exercises to analyse the gains emerging from landscape-scale management of input ES, i.e., in a game theory framework, the gains emerging from the grand coalition. This literature relies, however, on restrictive assumptions, usually simulating homogenous agents and simple mosaic landscapes. The originality of our approach is that we consider a more complex modelling framework with heterogeneous agents and realistic landscapes from Martel et al. (2017). I use the results of chapters 3 and 4 to calibrate the productive impacts of carabid beetles, which are considered the only source of input ES.

2.5 Conclusion

In this chapter, I provide a theoretical framework to analyse the management of biodiversity productive capacity when on-farm biodiversity is considered the source of a potentially provided vector of input ES. This framework is inspired by the literature on biodiversity productivity and acreage choices. It provides interesting optimal conditions to explain the management of input ES by farmers, considering both its benefits (additional yields and input savings) and its costs (with regard to the management of fixed inputs such as capital and labour). To our knowledge, this is the first theoretical framework explaining land-use choice that considers the benefits and costs of biodiversity productive capacity in a static framework. I have presented the relevance of this framework to the two literatures and to the evaluation of different policy instruments aiming to increase the provision of environmental services. Finally, I discuss some assumptions of the theoretical framework and present possible extensions to analyse temporal and spatial specificities of input ES. The analysis of the management of biodiversity in the farmers' production process is essential because it enables a better understanding of the role of biodiversity for agricultural producers. A better allocation of public funds for biodiversity conservation depends heavily on the better comprehension of the link between biodiversity and the production of agricultural goods.

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2.7 Appendices

Appendix 2.A. Bibliometric analysis of ecosystem services in the economic literature

The aim of the bibliometric analysis is to examine the evolution of the ES concept in economics, especially in agricultural economics. The purpose of this work is not to produce a literature review on ES in economics but rather to illustrate the emerging links between literature and (sub-)disciplines. There are already several literature reviews, including recent reviews, on ES that relate ES to current developments in economics (Barnaud and Antona, 2014; Gómez-Baggethun et al., 2010). There are also similar reviews on the development of ES in agricultural sciences (Tancoigne et al., 2014) or from an interdisciplinary perspective stretching across academic disciplines (Chaudhary et al., 2015). The motivation to rely on ES in economics applied to agriculture is related to the tradition among agricultural economists to work on microeconomic choices involving biological mechanisms. For example, initial works on the management of pest pressure were developed in journals of agricultural economics (e.g., Feder and Regev, 1975; Lichtenberg and Zilberman, 1986; Regev et al., 1976). Similar patterns are observed in the study of irrigation choices (Caswell and Zilberman, 1985; Lichtenberg, 1989; Nieswiadomy, 1988) or on the dynamic management of nutrient stocks (Eckstein, 1984; Tegene et al., 1988). These works were developed at the same time as initial works on ES (Westman, 1977) and before the mainstreaming of ecological economics (Costanza et al., 1997). However, contrary to the valuation motive of ecological economics, agricultural economics aims to explain producers' choices in relation to environmental changes. Here, the aim of the analysis is to illustrate the potential links between different economic studies in order to examine farmers' management of ES.

The identification of the links is based on a bibliometric analysis. A bibliometric analysis is a statistical analysis applied to publications (e.g., scientific articles, books). Bibliometric analysis is frequently used to evaluate the publications of an author or an institution through the production of indicators (e.g., the h-index); it has been developed to describe scientific literature on a subject.

To perform the bibliometric analysis, we first proceeded to a selection of papers related to ES on Web of Science. To identify the maximum number of works related to this concept, we use a procedure with related keywords, [23], inspired by Tancoigne et al. (2014). After the selection of economic filters,[24] we identify 9496 references, which is our dataset reference. We know the title of the study, the name of the journal, the keywords, the name of the authors, the date of publication and the abstract of the study for each of these references. To provide sensitivity analysis tests, we also refine our dataset using two additional restrictions:

- The paper should use the term "agricult*" in the title, the keywords or the abstract;
- The paper should be published in a HCERES rank A review in economics.

These additional restrictions enable us to create three additional datasets:

- One containing 3121 papers due to restriction 1 (named the "agricultural database" hereafter);
- One containing 2055 papers due to restriction 2 (named the "HCERES database" hereafter);
- One containing 698 papers due to restrictions 1 and 2.

Due to the limited number of observations for the third sub-dataset, we only retain the first two sets ones to provide bibliometric analysis.

We used the CorTexT platform from IFRIS to run the bibliometric analysis. In particular, the statistical treatments were run using CorTexT Manager, an online tool developed by R software (R Core Team, 2012). Following instructions from CorTexT document, [25] we first run a lexical extraction, followed by sorting, and finally, we create a corpus indexation based on the information in each reference. The lexical extraction enables the same keywords to be changed from different spellings to a single and normalized keyword. The sorting step enables suppressing some observations that are not of interest for the study. In our case, we have deleted 723 references based on service quality, customer satisfaction, technology innovation or corporate social responsibility. Our cleaned reference database is thus composed of 8673

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[23] We implement this procedure in Web of Science: TS=((Agri* NEAR/5 (Function OR fonction OR system OR service OR good OR amenity OR externality)) OR (Agro* NEAR/5 (Function OR fonction OR system OR service OR good OR amenity OR externality)) OR (environment* NEAR/5 (Function OR fonction OR system OR service OR good OR amenity OR externality)) OR (ecologic* NEAR/5 (Function OR fonction OR system OR service OR good OR amenity OR externality)) OR (land* NEAR/5 (Function OR fonction OR system OR service OR good OR amenity OR externality)) OR (eco?system NEAR/5 (Function OR fonction OR system OR service OR good OR amenity OR externality)))

[24] We have selected three discipline categories in the Web of Science: "agricultural economics policy", "business" and "economics".

[25] All information on the usual operation treatment is available at https://docs.cortext.net/.

references. The corpus indexation step attributes the corrected keywords to each element of the final database. These indexes are then used to perform the statistical treatments.

Several statistical treatments are available in CorTexT. We chose to restrain our analysis to mappings. Mapping treatment provides links between groups of references sharing common indexes. Each link is made between two nods, each nod regrouping a cluster of references. The procedure is developed in three steps. The first step is the "raw data" step to compute the number of occurrences of each index. The second step is the "measuring proximities" step to normalize the occurrence measures (to prevent statistical biases) and identify the similarities and dissimilarities between each nod. This step provides a score of chi-2 for each group of indexes where the highest scores are used to create the map. Here, we test the creation of a map based on 50 or 100 nods. The higher the number of nods, the higher the level of detail in the final map, to the possible detriment of a clear representation. The third step is the "community detection" step to identify the networks between nods and identify underlying links between non-neighbouring nods. This step segregates groups of nods from each other and clusters nods in the map. This three-step procedure is run in the three datasets.

Results

Figure 2.1 displays the results from the bibliometric procedure on the whole reference database. In Figure 2.1, we see a link between "agricultural production" and "agricultural systems" (in the blue cluster), a link between "agricultural systems" and "land-use" (between the blue and yellow clusters) and then between "land-use" and "environmental impacts", which is also linked to the ecological economics literature (the green cluster). In addition, we see a clear link between "land use" and "ecosystem services" (in the yellow cluster). From our point of view, this indicates that the analysis of land-use choices is the key perspective from which to study the management of ecosystem services. Indeed, studies on land-use are at an interdisciplinary crossroads, not only between agricultural economics (blue cluster) and ecological economics (green cluster) but also with natural sciences such as landscape ecology (even if these links do not appear in Figure 2.1). In our reference database, the ecological economics cluster is the dominant one. It regroups works on the monetary valuation of ES and solutions to overcome externalities linked to the degradation of ES levels. The agricultural economics cluster aims to explain farmers' production choices, notably land-use choices. In summary, the ecological economics cluster aims to value ES, and agricultural economics aims to explain land-use choices without any direct link with the ES concept. The "land-use" nod indirectly links (i) ES,

(ii) governance mechanisms to overcome externalities due to ES modification and (iii) farmers' choices.

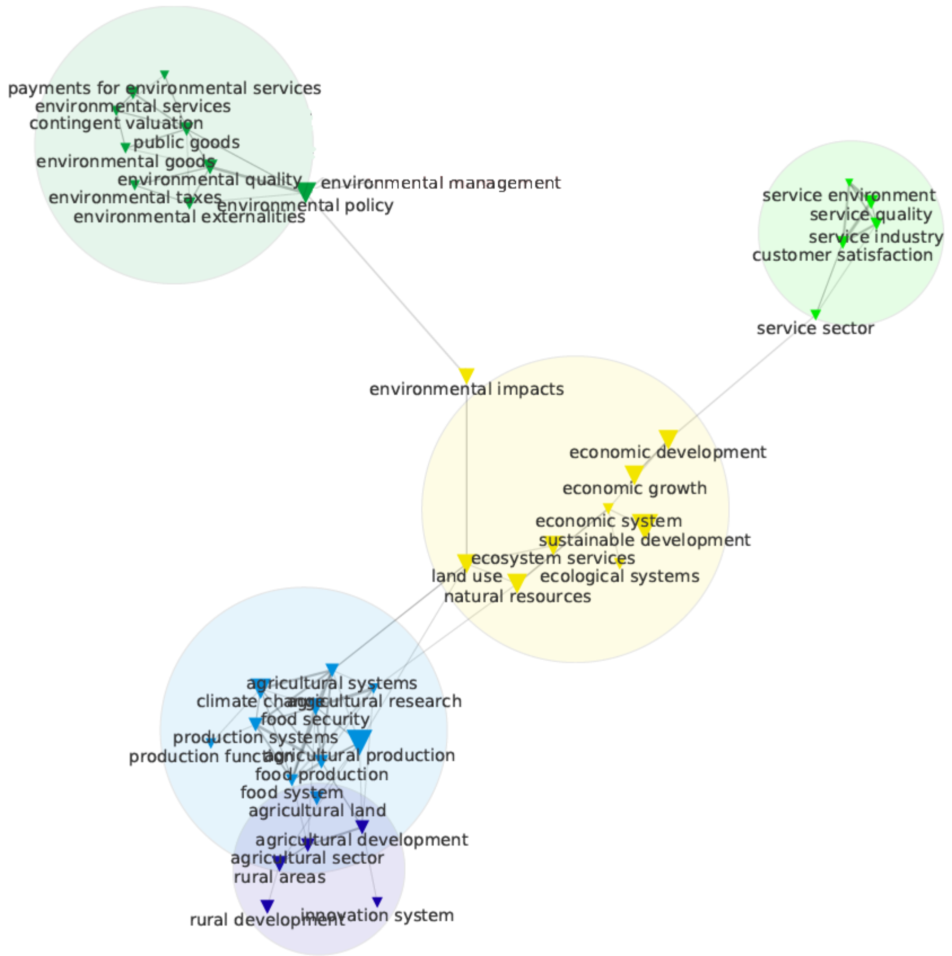

Figure 2.1. Mapping of the reference database (*Source: CorTexT Manager*)

Figure 2.2 displays the results of the bibliometric analysis on the "agricultural" database. Due to the restriction on the initial database, the results display no clear links between agricultural production and ES. However, once again, we do see that land use (green cluster) is directly linked to ES (red cluster).

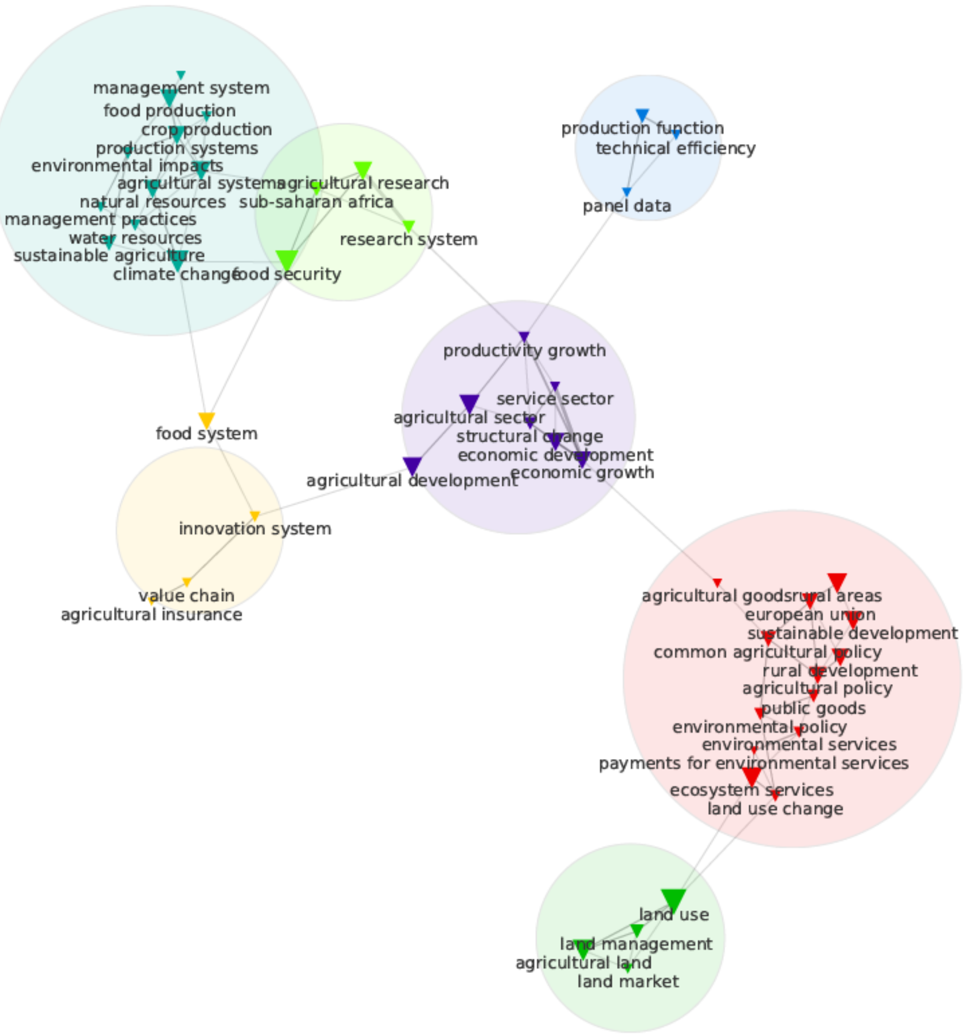

Figure 2.2. Mapping of the "agricultural" database (*Source: CorTexT Manager*)

Finally, Figure 2.3 displays the results of the bibliometric analysis of the "HCERES" database (i.e., the reference database restricted to top reviews in economics). Here, we see that "agricultural production" is clearly linked with "land-use" on one side and "ecosystem services" (through the "production systems" and "economic systems" nods) on the other side. This third database illustrates that agricultural economists are the researchers who have focused most on land use.

Figure 2.3. Mapping of the "HCERES" database (*Source: CorTexT Manager*)

Conclusion

The bibliometric analysis of the three databases has highlighted that land use is the main way to explore the management of ecosystem services. It also appears that the study of land use is clearly linked to agricultural economics, which has investigated these questions for several decades (e.g., Chavas et al., 1983). This short analysis motivates the approach developed in this PhD. In this PhD, I use existing models developed by agricultural economics to propose a framework to analyse the management of ES by farmers.

Appendix 2.B. Biodiversity indicators and the literature on the productivity of biodiversity

Because the most common information on landscape structure in agricultural microeconomic datasets pertains to acreage composition, economists have usually used this information to build indicators measuring crop inter-diversity (number of crops) or intra-diversity (number of varieties). Among the diversity of used indicators, we find the Shannon index (e.g., Di Falco and Chavas, 2008; Donfouet et al., 2017), the Margalef index (e.g., Di Falco and Chavas, 2009), the count index (Bangwayo-Skeete et al., 2012; Di Falco and Zoupanidou, 2017; Di Falco et al., 2007) or the dummy multicrop index (if mutlicrop farms 1, otherwise 0 - Karunarathna and Wilson, 2017). The Shannon index is an entropy measure based on the relative abundance of the different plant species in a given area. The count index is defined as the number of different plant species grown in a given area. The Margalef index is quite similar, but its weight is the number of different plant species in a given area by the inverse of the logarithm of the number of different plant species. To my knowledge, no papers from this literature except Smale et al. (1998) have used the Simpson index (which is similar to the Herfindahl index), despite its usefulness in landscape ecology.

The selection of the indicator is always a difficult issue, but there is still no proof of the possible superiority of one particular index over the others (Chavas and Di Falco, 2012a). In practice, most of the authors of the "biodiversity productive" literature have relied on the Shannon index. The Shannon index seems to have several advantages and is well suited to measuring habitat diversity (Mainwaring, 2001). First, like the Simpson and the Berger-Parker indexes, the Shannon index is sensitive to both evenness and abundance, limiting the bias that the indicator captures a sampling effect (Di Falco and Chavas, 2008). However, the Simpson and the Berger-Parker indexes are more used for situations where the dominant cover type is of interest (Nagendra, 2002). Another advantage of the Shannon index is that it is not sensitive to sample size (Keylock, 2005), unlike the count and the Margalef indexes (Gamito, 2010), even if its accuracy seems to decrease in practice when the scale of the landscape increases (Bailey et al., 2007). In fact, the literature on biodiversity indicators is still emerging, and the effectiveness of biodiversity indicators according to the scale and the accuracy of the information is debated (Bailey et al., 2007; Billeter et al., 2008). However, most of the papers using the Shannon index use aggregate data at the county or regional levels. As stressed by Donfouet et al. (2017), even if all the "biodiversity productivity" literature has concluded that there are productive effects of crop diversity regardless of the indicators used, farm-level studies rely more on the Margalef or count indexes. This could indicate that the Shannon index cannot be computed at the farm scale, but no reference was found in these farm-level papers to justify this choice. Moreover, in her farm-level study, Bezabih (2008) found no significant differences between the productivity of the count index and that of the Shannon index. In addition, if landscape ecologists insist on indicator computation on a continuous landscape, the choice to measure diversity using the Margalef or count indexes at the farm level does not overcome this issue.

Appendix 2.C. Comments on joint production in the considered framework

The aim of this appendix is to interpret relation (2.27) in light of the existing literature on joint production, notably in the case of multi-output farms.

Joint production: some definitions and sources

Joint production refers to situations where firms produce several outputs that are interlinked such that an evolution of the supply of one output affects the levels of the others. Even if several dual definitions have been proposed (Baumol et al., 1982; Hall, 1973), the first definition of joint technology was provided by Carlson (1939) in a primal framework. Carlson (1939) considers that a two-output technology is non-joint if:

$$\frac{\partial^2 \mathbf{X}}{\partial Y_1 \partial Y_2} = 0 \tag{2.29}$$

Relation (2.29) expresses that the application of inputs depends on a single output. In a joint technology, the application of an input on a specific output depends on at least one other output. According to Lau (1972), definition (2.29) is a case of input jointness, which is different from output jointness. Input jointness has received most of the attention of economists (Kohli, 1985).

Three types of jointness are usually distinguished (OECD, 2001): (i) technical interdependencies in the production process, (ii) non-allocable inputs or (iii) allocable fixed inputs. In the case of agriculture, the technical interdependencies are due to agronomical or ecological drivers, e.g., the beekeeper benefits from apple trees, and the apple producer benefits from the bees. Non-allocable inputs lead to joint technology because several outputs are produced by the same inputs. In the case of agriculture, this occurs when two outputs are cropped on the same land, e.g., agroforestry practices. Allocable fixed inputs can cause jointness because an evolution in the production of one output changes the availability of the input for the supply of the others due to firms' constraints. In the case of agriculture, farmers must allocate their capital and labour availability to the different outputs (Chambers and Just, 1989).

The source of jointness differentially impacts the behaviour of the agents. Indeed, using a dual definition of jointness, Dupraz (1996) notes that the multioutput cost function (Hall, 1973) displays the following properties in the first two cases of jointness:

$$\frac{\partial^2 C(\mathbf{Y},\mathbf{w})}{\partial Y_k \partial Y_j} \leq 0 \tag{2.30}$$

In these cases, the two outputs present weak cost complementarity, meaning that the marginal production cost of one output decrease when the production of another output increases. Dupraz (1996) highlights that this property is due to the existence of a public input that influences output production, i.e., inputs with public good characteristics of non-rivalry and non-excludability (Sandmo, 1972).

Using a similar function, Moschini (1989) finds the following in the case of allocable fixed inputs:

$$\frac{\partial^2 C(\mathbf{Y},\mathbf{w})}{\partial Y_k \partial Y_j} \geq 0 \tag{2.31}$$

In this case, the two outputs present weak cost substitutability, i.e., the marginal production cost of one output increases when the production of another output increases. In the case of allocable fixed inputs and normal technology (Sakai, 1974), the technology is characterized by increasing marginal costs when the quantities of other outputs increase and by decreasing marginal profits when the prices of other outputs increase. Inspired by Baumol et al. (1982), Dupraz (1996) qualifies the inputs at the source of the jointness as "quasi-public" inputs, in the sense that even if the inputs can be useful for two outputs, their utilization in the process of one output diminishes their use in another. Otherwise, if jointness is due to pure public inputs (leading to non-allocable inputs or technical interdependencies), the marginal cost of an output increases when the quantities of other outputs decrease, but the gross substitutability among outputs is not verified. Thus, different causes of jointness differentially influence the behaviour of the agents.

The acreage literature has studied production jointness, notably based on the Hall-derived definition from Shumway et al. (1984).[26] However, works have more investigated the consequences of jointness, the investigations being primarily motivated by empirical purposes, notably for evaluating the indirect effects of policy instruments (Guyomard et al., 1996; Koutchadé et al., 2018; Lansink and Peerlings, 1996). The crucial consequence of jointness in

[26] Shumway et al. (1984) define nonjointness in input as: $\partial y_k^* / \partial p_j = 0 \quad (k \neq j)$.

these cases is that it implies that a change in the economic context of one output modifies the supply and acreage of other outputs. Most of these works do not really investigate the sources of jointness. For example, Carpentier and Letort (2012, 2014) introduce jointness motive in the "implicit cost" function, which simultaneously captures (i) the unobserved variable costs associated with a given acreage such as energy costs, (ii) the constraints due to management of quasi-fixed inputs such as capital and labour and (iii) crop rotation constraints. These motivations refer to different sources of jointness, if any. The authors' specification of the cost function prevents them from doing any interpretations of their results regarding the source of the jointness. Some works have followed the seminal work of Chambers and Just (1989) and studied jointness through the angle of the allocable fixed input. Papers on acreage choices have rarely investigated other sources of jointness. In particular, they do not focus on the technical interdependencies between outputs, which is, however, thoroughly studied by the literature on the productive capacity of biodiversity (Chavas, 2009; Chavas and Di Falco, 2012a). One exception is the set of acreage works on crop rotation, which pay major attention to the dynamic production properties of multioutput technologies at the field level (Hendricks et al., 2014; Hennessy, 2006; Livingston et al., 2015). However, like papers on the productive capacity of biodiversity, papers on crop rotations ignore other underlying causes of jointness (notably because most studies are conducted at the field level). In summary, all these works focus on one type of jointness source but tend to ignore additional sources. Two exceptions should be noted. First, Carpentier and Gohin (2015) theoretically consider both the production properties of crop rotation (i.e., the dynamic technical interdependencies between outputs) and the allocable fixed input problem. Second, Orazem and Miranowski (1994) estimate farmers' choices by considering both the production properties of crop rotation and the allocable fixed input problem. In the two cases, the allocable fixed input problem concerns land. A noticeable difference between the two works is that Orazem and Miranowski (1994) construct an index at the farm level based on the previous acreage, while Carpentier and Gohin (2015) explicitly consider the previous crop at the field level.[27]

I consider two cases of jointness in my framework: technical interdependencies in the production process and allocable fixed inputs. To my knowledge, this is the first work that explicitly considers these two motives in a static framework. Expressions (2.26) and (2.27) enable the separate analysis of the two motives. On the left side, the joint process is due to

[27] The estimation of the model of Carpentier and Gohin (2015) would require both farm- and field-level data. These observations are difficult to obtain simultaneously, which explains why their work is purely theoretical.

technical interdependencies among outputs, while on the right side, the joint process is due to allocable fixed inputs.

Jointness due to technical interdependencies

The technical interdependencies are due to the input ES provided by the different output areas, whose quality is approximated by $B(\mathbf{S})$, i.e., the biodiversity productive capacity. Technical independencies due to diverse acreage lead to overyielding effects. In addition, $B(\mathbf{S})$ also impacts the application of variable inputs on each output, the two inputs being either cooperating or substitute inputs. In practice, several empirical measures indicate that input ES and chemical inputs are substitute inputs (Hennessy, 2006; Kim et al., 2000; Skevas et al., 2012; Thomas, 2003), leading to potential input savings. In particular, the multi-output multi-input cost function $C(\mathbf{Y},\mathbf{w}) = \sum_{k\in[1;K]} \mathbf{w}'\mathbf{X}_k\left(Y_k; B(\mathbf{S}),\mathbf{Z}\right)$ respects relation (2.30) on a share of the production set, implying that biodiversity productive capacity does present characteristics of a public input. Indeed, considering that $\partial^2 \mathbf{x}_k\left(y_k; B(\mathbf{S}),\mathbf{Z}\right) / \partial y_i \partial y_j = 0$, $C(\mathbf{Y},\mathbf{w})$ is equivalent to:

$$\frac{\partial^2 C(\mathbf{Y},\mathbf{w})}{\partial Y_i \partial Y_j} = \frac{\partial^2\left(\sum_{k\in[1;K]} S_k \mathbf{w}'\mathbf{x}_k\left(y_k; B(\mathbf{S}),\mathbf{Z}\right)\right)}{\partial Y_i\left(S_i;y_i\right)\partial Y_j\left(S_j;y_j\right)} = \frac{\partial^2\left(\sum_{k\in[1;K]} S_k \mathbf{w}'\mathbf{x}_k\left(y_k; B(\mathbf{S}),\mathbf{Z}\right)\right)}{\frac{\partial S_i}{\partial Y_i}\frac{\partial S_j}{\partial Y_j}} \tag{2.32}$$

where $S_i(Y_i; y_i, B(\mathbf{S}),\mathbf{Z})$ is the inverse function of $Y_i(S_i; y_i, B(\mathbf{S}),\mathbf{Z})$ increasing in Y_i. Because $\partial^2 S_k / \partial S_i \partial S_j = 0$ ($\forall (i;j;k) \in [1; K-1]$), (2.32) is equivalent to:

$$\frac{\partial^2 C(\mathbf{Y},\mathbf{w})}{\partial Y_i \partial Y_j} = \left(\sum_{k\in\mathbf{C}} S_k \mathbf{w}' \frac{\partial^2 \mathbf{x}_k\left(y_k; B(\mathbf{S}),\mathbf{Z}\right)}{\partial B(\mathbf{S})^2}\right) \frac{\partial^2 B(\mathbf{S})}{\frac{\partial S_i}{\partial Y_i}\frac{\partial S_j}{\partial Y_j}} \tag{2.33}$$

As we have $\partial \mathbf{x}_k\left(y_k; B(\mathbf{S}),\mathbf{Z}\right) / \partial B(\mathbf{S}) < 0$, and $\partial^2 B(\mathbf{S}) / \partial S_i \partial S_j \geq 0$ is possible for any $\left(S_i; S_j\right) \in \left[0; \bar{S}\right] \times \left[0; \bar{S}\right]$ and depends only on initial values of i, j and K, we have at least $\partial^2 \mathbf{x}_k (Y_k; B(\mathbf{S}),\mathbf{Z}) / \partial B(\mathbf{S})^2 \leq 0$ for a segment of the technical set, implying that (2.30) is verified on a segment of the technical set. According to Dupraz (1996), biodiversity productive capacity presents characteristics of a public input. Indeed, its productive services are available for all outputs of the firm, and its utilization in one process does not decrease its utilization for other outputs. Note that alternative causes of technical jointness are not considered in my framework, whereas there exist several examples of such effects, e.g., the beneficial interactions on mixed farms that grow crops and engage in animal production, where growing crops feed animals and manure can be used for organic crop fertilization (Dupraz, 1996).

Jointness due to allocable fixed inputs

The right-hand side of relation (2.27) illustrates that the management of fixed input for acreage is costly. The acreage literature does not provide strong evidence on the second-order relationship between $C(\mathbf{S};\mathbf{Z})$ and S_k. Indeed, the usual empirical assumption in the acreage literature is that the implicit cost function is convex in S_k, i.e., that $\partial^2 C(\mathbf{S};\mathbf{Z}) / \partial S_i \partial S_j \leq 0$. This convenient assumption represents all the incentives and disincentives for diversification. However, because we have removed some incentives for diversification from $C(\mathbf{S};\mathbf{Z})$ with the consideration of biodiversity productive capacity, we need a deeper examination of the evidence on the management of fixed input that would lead to relation (2.31).

One reason may be the management of farm capital, which is usually considered to present increasing returns to scale (Griliches and Jorgenson, 1966). In particular, capital-specific investments concerning one type of production are considered to present higher increasing returns to scale. According to Dupraz (1996), specific investments are one type of public input that only contributes to scale economies and may explain relation (2.31). The specific investments can lead to $\partial^2 C(\mathbf{S};\mathbf{Z}) / \partial S_i \partial S_j > 0$, explaining the monoculture. For example, maize requires costly irrigation systems, which are not especially useful for other crops. The costs incurred by $\partial^2 C(\mathbf{S};\mathbf{Z}) / \partial S_i \partial S_j > 0$ are alternatively interpreted as dynamic or static adjustment costs (Chambers and Just, 1989; Lansink and Stefanou, 2001; Orazem and Miranowski, 1994) or as the underemployment of fixed input (Dupraz, 1996).

Finally, relation (2.27) highlights that the costs link to management of biodiversity productive capacity depend on $\mathbf{Z}$, i.e., the long-term choices made by the farmers. The range of $\mathbf{Z}$ values illustrates part of the diversity of agricultural systems. The verification of property (2.31) can explain the differences between low-capital-intensive farms, which have incentives to manage biodiversity productive capacity (e.g., permaculture, organic farming) and high-capital-intensive farms, where biodiversity management costs discourage farmers from managing biodiversity.

To sum up

Relation (2.27) decomposes the joint technology between the presence of the public input (i.e., the biodiversity productive capacity) and the presence of allocable fixed input (i.e., the land). This decomposition illustrates the sources of scope economies in agricultural economies, which is a dynamic area of research in agricultural economics (Blancard et al., 2011, 2016; Chavas and Kim, 2010; Lansink and Stefanou, 2001). Indeed, considering a two-output firm with the cost function of one production equal to the sum of the variable costs and the implicit costs, scope economics are equal to (Baumol et al., 1982):

$$SC = \frac{[\mathbf{wX}_i(Y_i;\mathbf{Z}) + C(S_i;\mathbf{Z})] + [\mathbf{wX}_j(Y_j;\mathbf{Z}) + C(S_j;\mathbf{Z})] - [\mathbf{wX}(\mathbf{Y};B(\mathbf{S}),\mathbf{Z}) + C(\mathbf{S};\mathbf{Z})]}{\mathbf{wX}(\mathbf{Y};B(\mathbf{S}),\mathbf{Z}) + C(\mathbf{S};\mathbf{Z})} \tag{2.34}$$

where $\mathbf{wX}_i(Y_i;\mathbf{Z})$ and $C(S_i;\mathbf{Z})$ are the variable costs and the fixed costs to produce output *i* alone, and $\mathbf{wX}(\mathbf{Y};B(\mathbf{S}),\mathbf{Z})$ are the variable costs and the fixed costs to produce the 2 outputs at the farm scale, respectively. Relation (2.34) is similar to:

$$SC = \frac{\mathbf{wX}_i(Y_i;\mathbf{Z}) + \mathbf{wX}_j(Y_j;\mathbf{Z}) - \mathbf{wX}(\mathbf{Y};B(\mathbf{S}),\mathbf{Z})}{\mathbf{wX}(\mathbf{Y};B(\mathbf{S}),\mathbf{Z}) + C(\mathbf{S};\mathbf{Z})} + \frac{C(S_i;\mathbf{Z}) + C(S_j;\mathbf{Z}) - C(\mathbf{S};\mathbf{Z})}{\mathbf{wX}(\mathbf{Y};B(\mathbf{S}),\mathbf{Z}) + C(\mathbf{S};\mathbf{Z})} \tag{2.35}$$

The measure of scope economies can be decomposed as the addition of scope economies due to technical dependences in the multioutput firm and of scope economies due to the management of allocable fixed input. Because I have assumed $\partial \mathbf{x}_k\left(y_k;B(\mathbf{S}),\mathbf{Z}\right)/\partial B(\mathbf{S}) < 0$, the first term is positive. In the case of crop-specific capital and machinery, the second term is negative (Moschini, 1989). However, as noted by Dupraz (1996), agricultural capital can lead to economies of scope in the case where crops require similar types of machinery. In addition, labour is also considered to lead to economies of scope, allowing a better allocation of labour over the year. Thus, fixed input can be at the source of scope and scale economies (Dupraz, 1996), and we have $C(S_i;\mathbf{Z}) + C(S_j;\mathbf{Z}) \leq \geq C(\mathbf{S};\mathbf{Z})$. This decomposition illustrates the different sources of scope economies of multioutput farms.

CHAPTER 3. BIODIVERSITY PRODUCTIVE CAPACITY IN MIXED FARMS OF NORTHWEST FRANCE: A MULTI-OUTPUT PRIMAL SYSTEM [28]

This chapter focuses on the productive properties of productive ecosystem services (ES) in the defined agricultural technologies of the general framework (Chapter 2). In particular, it informs the productive relationships of biodiversity when two types of biodiversity components are distinguished, namely, those attached to crop on-farm biodiversity and those attached to permanent grasslands. These two components grasp most of the biodiversity in northwest Europe, where landscapes are complex combinations of these two components. We assume that crop diversity and permanent grassland are the main sources of ES farmers use as inputs. Similarly to the previous papers on the productivity of biodiversity, we estimate yield equations with biodiversity indicators being an input of agricultural technology. Contrary to previous papers, we explain that such yield equations can be derived from the very short term optimization programme, when land-use choices are already made (i.e., biodiversity indicators are exogenous) but when variable input applications still need to be optimized (i.e., variable inputs are endogenous). We estimate the first-order productivities of the two biodiversity components for cereals and milk on a sample of farms located in the northwest of France. We also estimate the second-order productivities of the two biodiversity components, informing on the complementary or substitutionary relationships between the two biodiversity components and the conventional variable inputs. Such interactions are crucial for understanding the effect of economic incentives, such as polluting input taxes or biodiversity component subsidies.

[28] This chapter was coauthored with Pierre Dupraz (INRA, SMART-LERECO).

3.1 Introduction

Modern human activities and, notably, agriculture have degraded biodiversity. Conversions of natural areas to arable lands have reduced the number of suitable habitat for biodiversity. The reduction of the number of crops have amplified this issue (Kleijn et al., 2009). This trend has led to interrogations on the possibility to combine intensive agriculture and biodiversity. Protection of biodiversity is crucial because biodiversity contributes to ecosystem functioning thanks to the interactions of species with each other. Ecosystem functioning influences the provision of many ecosystem services that are valorized by our societies (MEA, 2005). Certain authors consider that among the diversity of beneficiaries, the highest value of biodiversity accrues to farmers through its beneficial effects on production (Perrings, 2010).

Supporting and regulating ecosystem services (e.g. nutrient cycles and pest control) have been increasingly recognized as inputs for agriculture (Zhang et al., 2007). Several economic studies have analyzed the productive effects of these services for crop farms. For this purpose, they estimated production functions with biodiversity indicators considered as an input (Di Falco et al., 2010).[29] The biodiversity indicators rely on agricultural land-use shares and indicate the degree of diversity of biodiversity habitats within the considered agroecosystems. Even if the indicators correspond to a small component of the whole notion of biodiversity, they are correlated to species diversity and richness (Burel and Baudry, 2003). In particular, these indicators can be considered as proxies of productive ecosystem services. For example, higher on-farm crop diversity is correlated with higher soil structure (Mäder et al., 2002), pollination (Kennedy et al., 2013) and biological control (Letourneau et al., 2011). However, economists do not observe effectively the levels of these ecosystem services and can only assume that they are effectively provided to the farmers. Thus, biodiversity indicators correspond to an observable but inherently imperfect description of an ecosystem, which supports a vector of several productive ecosystem services that are potentially provided to the farmers. We refer to the capacity of an ecosystem to provide productive ecosystem services based on its observable characteristics as the “biodiversity productive capacity”. Several components of biodiversity could have this productive capacity.

[29] This method is often used in ecosystem services valuation studies (Perrings, 2010). Just and Pope (2002) have stressed the interest of production function estimations to get new insights on technology and farmers’ choices.

Previous studies on biodiversity productive capacity have emphasized that crop diversity increases mean agricultural yields, while decreasing their variance. This information is useful for policymakers because it highlights that high yields are compatible with diversified landscapes. However, most of the previous studies have only assessed the proper effect (or first order effect) of crop diversity on wheat, cereal or crop yields. In our view, they have four main limits that narrow the available knowledge on biodiversity productive capacity. Indeed, they do not estimate the biodiversity productive capacity considering (i) several products, (ii) several kinds of biodiversity components, (iii) the interactions between variable inputs and biodiversity productive capacity (i.e. the second-order productivities) and (iv) the potential endogenous bias linked to the simultaneity of farmers' optimal choices between variable inputs and objective yields. Our objective is to extend the current knowledge on the productive capacity of different components of biodiversity by assessing the productivity of crop diversity and permanent grasslands for cereals and milk, at the first and second orders, while controlling for the optimizing behavior of the farmers. These information are essential for policymakers as they may hinder the implementation of some policy measures.

Assuming that farmers maximize their very short term profit, we estimate a primal model with two yield functions (cereals and milk) and two biodiversity habitats (crop interspecific diversity and permanent grasslands)[30] on an unbalanced sample of mixed farms from the FADN (Farm Accountancy Data Network, a database used by the European Union to analyze the effects of CAP – Common Agricultural Policy – reforms on European farmers) between 2002 and 2013. Like crop diversity, permanent grasslands and related semi-natural elements (such as hedgerows) are considered as a vector of potential ecosystem services, notably for biological control (Aviron et al., 2005; Martel et al., 2017). The utilization of the very short term profit-maximizing framework allows assuming that acreage as well as the biodiversity indicators are exogenous and that the farmers optimize only on variable inputs. It allows correcting for the effects of prices on variable input applications, which is a source of endogeneity between variable inputs and yields that may affect the estimation of the productivities of our biodiversity indicators. It also allows allocating variable inputs between products when output-specific allocation of inputs is unobservable from the econometrician. In particular, we estimate three different specifications according to different possible properties of the variable inputs. The first model assumes that variable inputs are private inputs (which is an usual assumption in

[30] Interspecific diversity refers to crop species diversity (i.e. diversity among crops). This is different from intraspecific diversity, which refers to crop genetic diversity (i.e. diversity among the varieties of the same crop).

agricultural economics) and uses the optimal conditions on variable input application to derive parameter restrictions that allow allocating variable inputs between products. The second and third models assume that variable inputs are public inputs, allowing to remove the restrictions. These two last models can be considered as robustness checks of the economic specification. However, this specification also allows investigating the interactions between variable inputs and the two different biodiversity productive capacities in the third model. The interactions inform if variable inputs are complement or substitute with the supported productive ecosystem services. We estimate our model thanks to the general method of moment (GMM) on a sample of farms from northwest France, a region with diversified landscapes and high shares of semi-natural elements. We find that (i) crop diversity is an input for cereals and milk, (ii) permanent grasslands are an input for cereals, (iii) crop diversity and permanent grasslands are substitute for each other and (iv) the two components of biodiversity are substitute for mineral fertilizers and pesticides. We also find that ignoring the optimizing role of the farmers regarding variable input applications lead to an overestimation of the productivities of biodiversity.

The next section details the limits of the existing literature. The third section presents the theoretical analysis. We then present the empirical segment. The fifth section presents the results. We discuss them in the last section.

3.2 Literature review

Since the seminal works of Heisey et al. (1997) and Smale et al. (1998), the analysis of biodiversity productive capacity has benefited from a growing empirical literature in economics (e.g., Chavas and Di Falco, 2012; Di Falco et al., 2010; Donfouet et al., 2017; Finger and Buchmann, 2015; Matsushita et al., 2016). These studies estimate the productivity and/or the profitability of biodiversity indicators for agriculture. Most of them use primal approaches to estimate marginal effects of biodiversity indicators on mean and/or variance of yields, biodiversity indicators being measured through diverse functions of agricultural land-use. All these studies have found that biodiversity indicators are inputs for agricultural outputs. Studies based on profit analysis have concluded to a profitable effect of biodiversity indicators. Considering different components in the productive effects of crop biodiversity, Chavas and Di Falco (2012) confirmed that complementarity effects is the main source of additional productivity. In addition, it appears that (i) biodiversity has decreasing marginal returns on both yield and profit (Di Falco and Chavas, 2006; van Rensburg and Mulugeta, 2016), (ii) crop diversity is a suitable strategy for risk management (e.g. Di Falco and Perrings, 2005) but mainly (iii) when pesticide applications are low (Di Falco and Chavas, 2006) and (iv) for the driest years (Di Falco and Chavas, 2008). These evidences support the idea that biodiversity has both productive and insurance values (Baumgärtner, 2007; Chavas, 2009).

Despite the usefulness of these results, there are several shortcomings in this literature. First, studies have usually measured the productivity of biodiversity on a single product. Most of the studies have measured it on crop yields. To our knowledge, only van Rensburg and Mulugeta (2016) and Finger and Buchmann (2015) have analyzed the effects of biodiversity on animal and forage systems. Research needs to investigate the effects of biodiversity on other products. This is what we do focusing on two agricultural products: milk and cereals.

Second, these studies focus on a single kind of biodiversity component. They usually focus on intraspecific or interspecific crop diversity, considering crops as the main habitats within many agro-ecosystems but showing how narrowly biodiversity has been defined in these studies. Indeed, crop-orientated agroecosystems present a lower heterogeneity than many others, which usually present diverse landscape elements from crops to semi-natural elements. These areas may have productive cross-effects between them. For example, Klemick (2011) found that upstream forest fallows provide productive spillovers for crops but, still, did not focus on alternative biodiversity components. Similar to Donfouet et al. (2017), we consider that more studies need to be conducted on spillovers from semi-natural areas to better understand farmers' behavior regarding them. This is what we do by focusing on two components of agricultural biodiversity: crop diversity and permanent grasslands.

Third, there are still several uncertainties on the productive interactions between biodiversity productive capacity and conventional inputs, conventional inputs being at the source of several non-point pollutions. To our knowledge, only Di Falco and Chavas (2006) have examined these relationships. They have found that pesticides and crop diversity are substitutes for risk management. The lack of knowledge on the relationship to other variable inputs prevents the optimal implementation of instruments to promote biodiversity and/or reduce the application of polluting inputs. We control for these effects by focusing on the interactions between the productive capacities of the two biodiversity components with fertilizers and pesticides. Indeed, as both biodiversity productive capacities support biological control and other productive ecosystem services, they may interact with variable input productivities.

Fourth, most of the cited studies have estimated production functions. We argue that they do not capture farmers' behavior, notably regarding their response to prices. If most of the cited studies have instrumented biodiversity indicators, none of them has ever tried to correct for the endogenous bias between objective yields and variable input applications. Therefore, the conclusions of their studies may be biased. Here, we focus on the very short term profit maximization, considering that farmers only optimize on variable input application. It implies that, contrary to previous studies, we consider that farmers manage variable input applications but not biodiversity productive capacities.

The objective of our study is to provide an example that addresses these four issues by assessing the productivity of crop diversity and permanent grasslands for cereals and milk, at the first and second orders, while controlling for the optimizing behavior of the farmers. The next section presents the employed theoretical background to control for the optimization on variable input applications in a context where the productive capacity of biodiversity is explicitly considered.

3.3 Theoretical Analysis: a simple procedure to deal with jointness in multi-output farms

The current section develops a theoretical analysis to present and justify the specifications that we use in the empirical part, the different specifications may influencing the estimated parameters. We rely on the farmer's very short-term profit maximization (Asunka and Shumway, 1996) to represent the simultaneous choices of yield objectives and variable input applications between the different outputs of the multi-output farm. We also take into account the possibility of joint production in multi-output farms. Otherwise, the estimated productivity of biodiversity could be confounded with other technical complementarities between products. Finally, we present the first-order conditions (FOC) on variable input given farm's predefined acreage, which leads to the specification of parameter restrictions. We use these restrictions in a first model (Model 1). They provide a structure to estimate our two yield functions when only the total amount of bought variable inputs are registered at the farm level with unobserved input allocation between outputs. We compare Model 1 with two others, where we do not specify any hypothesis regarding variable input allocation.

We consider a risk-neutral farmer who maximizes her annual profit Π by adjusting the variable inputs gathered in vector $\boldsymbol{X}$, according to her quasi-fixed input dotation comprised in vector $\boldsymbol{Z}$, her biodiversity productive capacity levels included in vector $\boldsymbol{B}$ and her farm total area A. Given input price $\boldsymbol{w}$**,** she produces the agricultural goods gathered in vector $\boldsymbol{Y}$ sold at price $\boldsymbol{p}$. $\boldsymbol{Z}$ contains information on farm labor and capital. We assume that $\boldsymbol{Z}$, $\boldsymbol{B}$ and *A* are not adjusted in the short-term (Asunka and Shumway, 1996). This hypothesis differs from pre-existing literature in case of $\boldsymbol{B}$. Authors usually instrument the biodiversity indicator, i.e. implicitly assume that farmers optimize $\boldsymbol{B}$. This makes sense from one year to another since the usual indicator is crop diversity. However, the simultaneous choice between yields and variable inputs has to be taken into account as well. We deepen this issue by distinguishing a very-short-term time horizon (within year) where only variable inputs are optimized given $\boldsymbol{B}$, and not the opposite (Asunka and Shumway, 1996). More realistic for temperate and developed countries, our time sequence hypothesis enables the description of biodiversity and chemical input interactions in agricultural products. We discuss this hypothesis in section 6.3.1.

We write the general farmer's program as follows:

$$\Pi\ (E(\boldsymbol{p}), \mathrm{E}(\boldsymbol{w}), S, \boldsymbol{Z}, \boldsymbol{B}, A) = \max_{\boldsymbol{X}}\{E(\boldsymbol{p})'\boldsymbol{Y} - E(\boldsymbol{w})'\boldsymbol{X} + S; (\boldsymbol{Y}, \boldsymbol{X}, \boldsymbol{Z}, \boldsymbol{B}, A) \in \mathrm{T}\} \tag{3.1}$$

where $(E(\boldsymbol{p}), E(\boldsymbol{w}))$ are the farmer's expectation prices and S sums the area subsidies received by the farm.[31] T is the production feasible plan of the multi-output farm. Relation (3.1) defines the multi-output multi-input profit function that represents T if T is bounded compact and quasi-convex in ($\boldsymbol{X}$,$\boldsymbol{Y}$) for each $\boldsymbol{Z}$**,** $\boldsymbol{B}$ and *A* (McFadden, 1978). Program (3.1) represents the farmer's annual production choices.

In the empirical model, we estimate a multi-output technology by specifying a system of output-specific yield functions. However, the specification of output-specific yield function implies additional technology hypothesis on the allocation inputs between each output of the multi-output farm. Indeed, considering that the farmer produces *K* outputs (each sold at price p_k), the farmer allocates $\boldsymbol{X}$**,** $\boldsymbol{Z}$**,** $\boldsymbol{B}$ and *A* between the *K* outputs, each input unit may benefiting to one or several products with more or less rivalry between products. First, variable inputs can be applied to each unit of land. The application of variable inputs is rival between products because one unit of such input applied on a particular product cannot be applied to another. However, a part of it will benefit to other products in case of production jointness. For example, the use of fertilizer on crops increases crop production, which benefits to milk production in case of crop

[31] For the following developments of the model, note that, as area subsidies of the common agricultural policy of the European Union are decoupled from yields since the early nineties, they are not considered in the empirical estimation.

interconsumption. Hereafter, we propose two ways to represent the allocation of variable inputs between cereals and milk. On both cases, we allocate animal variable inputs (feed and health products in the empirical analysis) to milk production and we accordingly specify the production jointness due to organic fertilization. In the first case (Model 1), we use the application rivalry of variable inputs to derive the optimal conditions of variable input allocation. On the second case (Models 2 and 3), we simply model them as public inputs (Baumol et al., 1988), implying that variable inputs could be the source of unspecified output complementarities and are available to all outputs at the farm level. This second model is general enough to deal with either private inputs that are actually rival between products or public inputs, as well as allocated inputs with so many spillovers from one production to the others that they operate like public input at the farm level (Asunka and Shumway, 1996). Choosing between these two specifications is an empirical issue. Second, we make no assumption about $\boldsymbol{Z}$ and $\boldsymbol{B}$ allocation, modelling them as public inputs. Agricultural economists often use this specification for $\boldsymbol{Z}$ (e.g. Carpentier and Letort, 2012). The possible non-rivalry of $\boldsymbol{B}$ between outputs seems coherent as ecological processes could present many spillovers. Finally, we allocate farm land A between products to specify output-specific yield functions.

Following the preceding hypotheses, we split annual program (3.1) into a two-stage optimization process to isolate the estimated yield functions. The first stage occurs at the beginning of the agricultural campaign when the farmer sows her lands based on decoupled area subsidies s_k (with $S = \sum_k s_k a_k$) and expected margins per hectare $E(\pi_k)$, the farmer's acreage allocation being composed of K components a_k. $E(\pi_k)$ depends on farmer's price expectations during this stage (usually in October in France). Contrary to prices, s_k is known and only depends on land-use (arable or grasslands).[32] The second stage (i.e. the very-short-term optimization) occurs during the agricultural campaign when the farmer optimizes gross margins on each area based on variable input application given her acreage, which is assumed to be fixed (Asunka and Shumway, 1996). We assume that price expectations may differ between the two stages due to new information leading to differences between expected and realized margins (the second stage usually occurs during spring in France). Following Carpentier and Letort (2012), we assume that farmers correctly know the input prices in the second stage.

[32] Area subsidies being decoupled from yields, they influence farmland allocation between products but not yields.

In Model 1, we decompose (3.1) in the first-stage optimization (3.2) and the second-stage optimization (3.3):

$$\Pi\ (E(\boldsymbol{p}), E(\boldsymbol{w}), \boldsymbol{s}, \boldsymbol{Z}, A) = \max_{a_1;\dots;a_k} \left\{\textstyle\sum_{k=1}^{K} a_k \left[E\left(\pi_k(E(p_k), E(\boldsymbol{w}), \boldsymbol{Z}, A)\right) + s_k\right]; \sum_{k=1}^{K} a_k = A\right\} \quad (3.2)$$

$$\pi_k(E(p_k), \boldsymbol{w}, \boldsymbol{Z}, A) = \max_{\boldsymbol{x_k}}\{E(p_k)y_k - \boldsymbol{w}'\boldsymbol{x}_k;\ y_k \leq f_k(\boldsymbol{x}_k; \boldsymbol{Z}, \boldsymbol{B}, \boldsymbol{Y_{-k}}, A)\} \quad (3.3)$$

where π_k is the margin of output k. The vector $\boldsymbol{x_k}$ contains variable input applied per hectare of product k such that $\sum_k a_k \boldsymbol{x_k}$ are the components of $\boldsymbol{X}$. y_k is the yield of k, $f_k(\boldsymbol{x}_k; \boldsymbol{Z}, \boldsymbol{B}, \boldsymbol{Y_{-k}}, A)$ is the corresponding yield function and $\boldsymbol{Y_{-k}}$ represents the vector of products other than k (Asunka and Shumway, 1996). $\boldsymbol{Y_{-k}}$ is an input of $f_k(\cdot)$, meaning that multi-output farms may present joint production processes (e.g. cereal yields depend on organic fertilization). We assume that T is fully defined by the K output-specific frontiers $f_k(\cdot)$ such that $Y_k \leq a_k f_k(\cdot)$ where Y_k is the output production level at the farm level. $f_k(\cdot)$ is nonnegative, nondecreasing, linearly homogenous and concave in $\boldsymbol{x}_k$. Note that $f_k(\cdot)$ does not depend on $\bar{a}_k$ explicitly, i.e. that we assume that marginal short-run returns to area are constant in output area.[33] Without loss of generality, we consider two outputs ($k=1$ is cereals and $k=2$ is milk) and solve (3.3) on x_{l2}. With $Y_2 = \bar{a}_2 y_2$ and $\bar{a}_2 > 0$, we have the following FOC:

$$\frac{\partial f_2(\boldsymbol{x}_2; \boldsymbol{Z}, \boldsymbol{B}, Y_1, A)}{\partial x_{l2}} = \frac{w_l}{E(p_2) + \frac{\bar{a}_1}{\bar{a}_2} E(p_1) \frac{\partial y_1}{\partial y_2}}$$

where $\partial y_1/\partial y_2$ represents the additional cereal yields due to the increase of one unit of milk yield. Farmers apply x_{l2} on $\bar{a}_2$ until the sum of the anticipated marginal productivity of x_{l2} on y_2 and its indirect marginal productivities on y_1 equals w_l. Like the common short-term maximization conditions, the last relation highlights that an increase of the expected price of one output leads to increase input-use (because $f_k(\cdot)$ is concave in $\boldsymbol{x}_k$). Because it is valid for each input ($\forall l \in [1;4]$) and output, we have:

$$\frac{\partial f_1}{\partial x_{11}} \Big/ \frac{\partial f_2}{\partial x_{12}} = \cdots = \frac{\partial f_1}{\partial x_{41}} \Big/ \frac{\partial f_2}{\partial x_{42}} = \frac{E(p_2) + E(p_1)\frac{\bar{a}_1 \partial y_1}{\bar{a}_2 \partial y_2}}{E(p_1) + E(p_2)\frac{\bar{a}_2 \partial y_2}{\bar{a}_1 \partial y_1}} \quad (3.4)$$

[33] This assumption is also made by Carpentier and Letort (2012) for example. We have estimated the production functions assuming non-constant return to area but the estimated parameters were non-significant.

The ratios of marginal input productivities of cereals on milk are equal if variable inputs are really private inputs. We use relation (3.4) as parameter restrictions in Model 1.

In Models 2 and 3, we model variable input as public inputs. We decompose program (3.1) in (3.5) and (3.6). The farmer chooses its acreage in (3.5) and its variable input application in (3.6). Contrary to Model 1, the farmer cannot optimize each margin separately in the second-stage. We have:

$$\Pi\ (E(\boldsymbol{p}), E(\boldsymbol{w}), \boldsymbol{s}, \boldsymbol{Z}, A) = \max_{a_1;\ldots;a_k} \{\textstyle\sum_{k=1}^{K} a_k(E(p_k)E(y_k) + s_k) - E(\boldsymbol{w})' A E(\boldsymbol{x});\ \sum_{k=1}^{K} a_k = A\} \quad (3.5)$$

$$\Pi\ (E(\boldsymbol{p}), \boldsymbol{w}, \boldsymbol{Z}, A) = \max_{\boldsymbol{x}} \{\textstyle\sum_{k=1}^{K} \bar{a}_k(E(p_k)y_k + s_k) - \boldsymbol{w}' A \boldsymbol{x};\ y_k \leq g_k(\boldsymbol{x}; \boldsymbol{Z}, \boldsymbol{B}, \boldsymbol{Y_{-k}}, A)\} \quad (3.6)$$

where $\boldsymbol{x}$ is the vector of variable input applied per hectare at the farm level such that $\boldsymbol{X} = A\boldsymbol{x}$. $E(y_k)$ and $E(\boldsymbol{x})$ defined in (3.5) are the solutions of (3.6) with $\boldsymbol{w}$ being imperfectly known. The vector of yields $\boldsymbol{y}$ is composed of K yields y_k. $g_k(\boldsymbol{x}; \boldsymbol{Z}, \boldsymbol{B}, \boldsymbol{Y_{-k}}, A)$ is the yield function of y_k. We assume that T is fully defined by the K output-specific frontiers $g_k(\cdot)$ such that $Y_k \leq a_k g_k(\cdot)$. $g_k(\cdot)$ is nonnegative, nondecreasing, linearly homogenous and concave in $\boldsymbol{x}$. In Model 2, we assume that the second-order productivities of $\boldsymbol{B}$ and $\boldsymbol{x}$ are null, i.e. $\partial^2 g(\cdot)_1 / \partial \boldsymbol{B} \partial \boldsymbol{x} = 0$. Similar assumptions were implicitly made in Model 1 to reach (3.4). In Model 3, we explore these productive interactions between variable inputs and biodiversity productive capacities in the cereal yield function making no assumption on the second-order productivities, crop diversity and permanent grasslands could influencing the productivities of variable inputs through ecological processes. Due to the public input specification, the variable input optimization in (3.6) is performed on all the products at the same time. The very short-term optimization leads to the following FOC:

$$\bar{a}_1 E(p_1)\left(\frac{\partial f_1(\boldsymbol{x}_1; \boldsymbol{Z}, \boldsymbol{B}, Y_2, A)}{\partial x_l} + \frac{\partial y_1}{\partial y_2}\frac{\partial f_2(\boldsymbol{x}_2; \boldsymbol{Z}, \boldsymbol{B}, Y_1, A)}{\partial x_l}\right) + \bar{a}_2 E(p_2)\frac{\partial f_2(\boldsymbol{x}_2; \boldsymbol{Z}, \boldsymbol{B}, Y_1, A)}{\partial x_{l2}} = w_l$$

The sum of the direct and indirect marginal productivities of $\boldsymbol{x}$ is equal to $\boldsymbol{w}$, which prevents deriving parameter restrictions between outputs and inputs like in Model (1).

The interest of (3.3) and (3.6) is threefold for our empirical application. First, it introduces the estimated yield functions $f_k(\boldsymbol{x_k}; \boldsymbol{Z}, \boldsymbol{B}, \boldsymbol{Y_{-k}}, A)$ and $g_k(\boldsymbol{x}; \boldsymbol{Z}, \boldsymbol{B}, \boldsymbol{Y_{-k}}, A)$. Second, relation (3.3) (respectively (3.6)) highlights that y_k and $\boldsymbol{x_k}$ (respectively $\boldsymbol{x}$) are jointly chosen by the farmers. This is the issue of simultaneity leading to potential endogeneity bias in the estimation. The optimal demand of variable input $\boldsymbol{x_k^*}$ and $\boldsymbol{x}^*$ are obtained using the FOC of (3.3) and (3.6). It depends notably on input-output price ratios. We use these FOC to instrument variable inputs in the empirical models. Third, it underlines that each output-specific yield function depends on other products due to joint production. This technical interaction can bias the estimation of the yield functions if not taken into account. In particular, as manure production is correlated with permanent grassland area, its absence in the estimation would overestimate productivities of permanent grasslands. Thus, we add explicit representation of production jointness with the introduction of manure production inside the estimated equations of the three models. It disentangles the productivity of $\boldsymbol{B}$ and variable inputs from production complementarity due to organic fertilization.

The purpose of the theoretical analysis was to provide insights on the possible specifications of the system of production functions, the specifications could influencing the estimations of our parameters of interest (the first and second order productivities of crop diversity and permanent grasslands). The three models provide complementary information on biodiversity productive capacity but must also been understood as robustness tests. There are obviously remaining empirical issues (e.g. unobserved aspects of production such as land quality or manure production) that we treat in the following section.

3.4 Empirical model, biodiversity indicators and summary statistics

3.4.1 Biodiversity indicators

We select two kinds of biodiversity components: crop diversity (noted B_{1t} for year t) and permanent grasslands (noted B_{2t} for year t). We measure the two biodiversity components using two different biodiversity indicators based on land-use. We measure B_1 with the Shannon index (Baumgärtner, 2006), which is an indicator that is usually used to measure crop diversity (Donfouet et al., 2017). This index has the advantage to (i) correct for both species richness and evenness of their proportional abundance, (ii) be not sensitive to sample size and (iii) be well suited to measure habitat diversity (Mainwaring, 2001). Other usually used indices such as the count index do not usually correct for evenness. The Shannon index is an entropy measure based on land shares but, as we measure crop biodiversity, we correct for permanent grassland shares a_{Kt}. We compute B_{1t} as follows:

$$B_{1t} = -\sum_{k=1}^{K-1} \frac{a_{kt}}{1-a_{Kt}} \ln\left(\frac{a_{kt}}{1-a_{Kt}}\right)$$

B_{1t} takes the value 0 when the farm has a monoculture and increases when diversity increases. Here, we compute crop diversity using the whole diversity of FADN crops with 42 crops (41 annual crops including forages – maize, temporary grasslands – plus orchards, but without permanent grasslands). In most cases, Landscape ecologists have highlighted that B_1 increases when biodiversity increases (Burel and Baudry, 2003). Productivity of B_1 captures an augmentation of ecosystem services such as soil structure (Mäder et al., 2002), pollination (Kennedy et al., 2013) and biological control (Letourneau et al., 2011). The impact on soil structure explains that crop diversity may interact with fertilizer productivity while its impact on biological control explains that crop diversity may interact with pesticide application. We test these features in Model 3.

We choose B_{2t} as the permanent grassland share in the utilized agricultural area (UAA), i.e. $B_{2t} = a_{Kt}$. Permanent grasslands share is notably a proxy of the number of permanent semi-natural landscape elements (e.g. hedgerows - Thenail, 2002) that are susceptible to have productive effects on milk and crop products. These effects are (i) wind-break, (ii) habitat furniture for insects involved in biological control, (iii) influence on hydrological flux, (iv) reduction of erosion and (v) contribution to a microclimate (Baudry et al., 2000). High share of permanent grasslands increases also landscape complexity and provides suitable habitat for pollinators (Ricketts et al., 2008) or for insects involved in biological control (Martel et al., 2017). Both effects will be captured on the productive capacity of B_{2t}. Its impacts on hydrological flux, erosion and biological control also indicate that permanent grasslands may interact with fertilizers and pesticide productivities (Model 3).

Our biodiversity indicators may suffer from several biases. First, the choice of the indicators relies highly on data availability. The mobilization of the FADN database compels us to rely on indicators computed at the farm scale. Instead, landscape ecologists compute these indicators at the landscape scale (Burel and Baudry, 2003). However, Donfouet et al. (2017) have emphasized that there are no significant differences of crop diversity productivity according to the scale of the indicator computation in previous studies. Second, farmers' CAP declaration of permanent grasslands may be underreport due to constraining legislative specificities. Third, biodiversity indicators based on landscape structure do not consider farmer practices. If landscape elements can enhance agricultural production, the expressions of the related functionalities depend on agricultural practices, notably on chemical practices (Omer et al., 2007). We partly address this issue by considering productive interactions between the biodiversity productive capacities and the variable inputs in the Model 3. Fourth, we consider only a within-year optimization of variable inputs given B_{1t} and B_{2t}, implying that we ignore crop rotation. However, Di Falco and Chavas (2008) found that the productivity of B_{1t-1} for cereals were 59% less important than the productivity of B_{1t}, suggesting than the effects of crop rotation (see Hennessy, 2006) are only a minor component of the overall productive effects attached to crop diversity. Additional issues may originate from potential biases linked to economic confounders; for example, indicators can inform on fixed input organization. These issues are common to all economic studies on the measure of the productivity of biodiversity. If we have attempted to capture these effects, some results may be biased due to remaining confounders.

3.4.2 Empirical models

We consider two products: cereals (*k*=1) and milk (*k*=2). They are produced on separated areas $\bar{a}_{1t}$ and $\bar{a}_{2t}$, $\bar{a}_{2t}$ being the total size allocated to maize silage, temporary grasslands and permanent grasslands.[34] We measure cereal and milk yields in quantity by area. For cereals, we estimate a log-linear production function:[35]

$$log(y_{1it}) = \beta_{01} + \sum_{l=1}^{4} \beta_{l1} \frac{X_{lit}}{A_{it}} + \sum_{j=1}^{2} \beta_{j1} B_{jit} + \beta_{121} B_{1it} B_{2it} + \sum_{m=1}^{2} \beta_{m1} \frac{Z_{mit}}{A_{it}} + \beta_{A1} A_{it} + \sum_{n=1}^{12} \beta_{n1} c_{nit} + \varepsilon_{1it} \quad (3.7)$$

where t is the index for year t and i is the index for individual i. We consider four variable inputs: mineral fertilizer (*l*=1), pesticides (*l*=2), seeds (*l*=3) and fuel (*l*=4). The two fixed inputs *m* are available labor and farm capital. We add an interaction term β_{121} between B_{1it} and B_{2it} to capture their non-linearity effects on yields, informing on second-order effects of biodiversity productive capacities for the two products. The twelve variables c_{nit} are the control variables. They include ten climatic variables (see section 4.3. for details) and two variables for organic fertilization: the cattle manure production per hectare and the manure production per hectare

[34] Note that $\bar{a}_2$ and B_2 are different: B_2 informs only permanent grasslands. The areas for maize silage and temporary grasslands are ecosystem components captured into B_1.

[35] We have also estimated quadratic production functions. The principal issue is that we cannot estimate Model 1 with linear econometrics method. The results of Model 2 with quadratic production functions are coherent with the presented ones but the variable input productivities were less significant. Note that we have not estimated log-log production functions. The combination of the distribution of our explanatory variables would lead to the suppression of 85% of our sample (see section 4.3.).

from other livestock. We compute these quantities using the Agricultural French Ministry's formula based on the number of animal units at the farm scale (CORPEN, 2006).[36] ε_{1it} is the error term which captures the unobserved heterogeneity. The introduction of individual fixed effects allows controlling for fixed characteristics of farms that may bias the estimation of the productivities of the biodiversity indicators. In particular, it allows controlling for the unobservable soil quality (assume to be fixed here). Using the usual Durbin-Wu-Hausman test to compare random and fixed individual effects, we select the specification $\varepsilon_{1it} = u_{1i} + v_{1t}$ where u_{1i} is the farmer's fixed effect. Eventually, we estimate the within transformation of (3.7) (e.g. Baltagi, 2008) to suppress u_{1i} and thus to suppress the heterogeneity bias linked to the correlation of the unobservable fixed effects, such as soil quality, with explanatory variables.

We also estimate a log-linear production function for milk:

$$log(y_{2it}) = \beta_{02} + \sum_{l=1}^{6} \beta_{l2} \frac{X_{lit}}{A_{it}} + \sum_{j=1}^{2} \beta_{j2} B_{jit} + \beta_{122} B_{1it} B_{2it} + \sum_{m=1}^{2} \beta_{l2} \frac{Z_{mit}}{A_{it}} + \beta_{A2} A_{it} + \sum_{n=1}^{12} \beta_{n2} c_{nit} + \varepsilon_{2it} \quad (3.8)$$

We assume that the number of cows per hectare is fixed. Some non-linearity on milk yield per cow can be captured through the introduction of manure production per hectare, which is a function of number of animal. Because FADN does not provide information on forage yields, we must interpret the productivities of $\boldsymbol{B}$ and the four variable inputs on milk as a function of their productivities on forage. In addition to the four previous variable inputs (which benefit to milk production through forage production), we add purchased feed ($l=5$) and health and reproduction expenses ($l=6$). ε_{2it} is the error term of (3.8) with $\varepsilon_{2it} = u_{2i} + v_{2t}$. Similar to (3.7), we estimate the within transformation of (3.8). Because we estimate the within transformation of (3.7) and (3.8), the constant terms β_{0k} in (3.7) and (3.8) capture the average technical progress.

As we ignore x_{lkit} (the application of input l on output k for i in t) and only know X_{lit}/A_{it}, the β_{lk} in (3.7) and (3.8) do not only represent the marginal productivity of input l on output k. These parameters measure the product of the marginal productivity of l on k by an always positive multiplying factor ($A/\bar{a}_1 y_1$ in model 1 and A/y_1 in models 2 and 3 – see appendix

[36] Note that we do not introduce Y_{2it} in Y_{1it} explicitly because dairy cows are not the only sources of manure (other animals matter) and manure fertilizes forage crops used in milk production as well.

3.A1). The β_{lk} measure two effects that are impossible to estimate separately. However, as our parameters of interest are the β_{jk}, we only have to verify that β_{lk} are positive for each *l* and *k*.

Based on FOC (3.3) and (3.6), we instrument X_{lit}/A_{it} in (3.7) and (3.8) by the input-output price ratios, assuming naïve anticipation for outputs and rational anticipation for inputs.[37] We use decoupled subsidies and milk quota as additional instruments to capture heterogeneity of the farms' economic environment. As farmers are price-takers and milk quotas have never been tradable in France but administratively allocated, our prices and policy instruments are purely exogenous from the farmer's point of view and should be correlated with X_{lit}/A_{it} (as our theoretical analysis suggests). We verify these correlations in our empirical results where price ratios have significant effect and expected signs. We also instrument total labor using farm partners' labor, which is fixed in the short-term and can thus be considered as exogenous.

In Model 1, we use equation (3.4) to correct for the optimal allocation of variable input between cereals and milk. Equation (3.4) implies:

$$\beta_{11}/\beta_{12} = \beta_{21}/\beta_{22} \tag{3.9}$$

$$\beta_{21}/\beta_{22} = \beta_{31}/\beta_{32} \tag{3.10}$$

$$\beta_{31}/\beta_{32} = \beta_{41}/\beta_{42} \tag{3.11}$$

These restrictions are valid in the case of log-linear production functions and with X_{lit}/A_{it} instead of x_{lkit} (see Appendix 3.A1.). Model 1 is composed of within transformations of (3.7), (3.8) and restrictions (3.9), (3.10) and (3.11) with the instrumentation of the six variable inputs using GMM. GMM corrects for potential heteroscedasticity. In addition, we run a seemingly unrelated regression (SUR) and three-stage least square (3SLS) estimations to illustrate the interest of instrumentation for variable inputs.

Model 2 is composed of the within transformations of (3.7) and (3.8) without any parameter constraints. We instrument X_{lit}/A_{it} using GMM. The comparison of Models 1 and 2 allows determining the impact of restrictions on the estimated parameters, notably biodiversity productivity ones.

Model 3 is composed of the within transformations of (3.8) and (3.12) without any parameter constraints. Model 3 is similar to Model 2 but we replace the interaction term between B_{1it} and

[37] These are classic assumptions in agricultural economics (e.g. Carpentier and Letort, 2012).

B_{2it} with interaction terms between B_{jit} and X_{lit}/A_{it} (for $j \in [1;2]$ and $l \in [1;2]$) in the equation of cereal yields. [38] We focus only on the productive interactions of biodiversity with fertilizer and pesticides, the productivities seeds and fuel being insensitive to ecological processes in the current year. Equations (3.7) and (3.12) give different information on the second-order effects of biodiversity productive capacities: (3.7) focus on the interactions between B_{1t} and B_{2t} whereas (3.12) focus on the interactions between B_{jt} and variable inputs. We have:

$$log(y_{1it}) = \beta_{01} + \sum_{l=1}^{4} \beta_{l1} \frac{X_{lit}}{A_{it}} + \sum_{j=1}^{2} \beta_{j1} B_{jit} + \sum_{l=1}^{2} \sum_{j=1}^{2} \beta_{lj1} \frac{X_{lit}}{A_{it}} B_{jit} + \sum_{m=1}^{2} \beta_{m1} \frac{Z_{mit}}{A_{it}} + \beta_{A1} A_{it} + \sum_{n=1}^{12} \beta_{n1} c_{nit} + \varepsilon_{1it} \quad (3.12)$$

with $\varepsilon_{1it} = u_{1i} + v_{1t}$. We instrument and $B_j X_{lit} / A_{it}$, additional instruments being computed multiplying the previously identified instruments of X_{lit}/A_{it} by B_{jt}.

3.4.3 Description of the data and variables

We use the FADN on three regions of northwest of France from 2002 to 2013: Brittany ("*Bretagne*" in French), Lower-Normandy ("*Basse-Normandie*") and Western-Loire ("*Pays-de-la-Loire*"). These regions are orientated towards breeding (e.g., they produce approximately 60% of French milk) and present diversified acreages with high shares of permanent grasslands. We can consider that the set of financial supports were relatively homogenous during our sample period, data from 2002 being only used for price expectation. Indeed, farms from our sample only confront the 2008 Common Agricultural Policy (CAP) reform. The most notable changes are the suppression of fallow obligations, the gradual increase of milk quotas and the extension of decoupled subventions. We only select mixed farms that produced milk and that have allocated area to cereals, maize silage and temporary grasslands, which represent 75.8% of the FADN mixed and dairy farms in these regions. Our rotating panel sample is constituted of 999 farms that have been around an average of 3.96 years, constituting in total 3,960 observations. This selection is required to estimate the system of production functions but one could argue that we focus on farms with relatively high diversity already. However, as presented in Table 3.1, the crop diversity index present a dispersed distribution, the maximum value being 11 times higher than the minimum value (equal to 0.206, which indicates a real tendency to monoculture).

[38] It is difficult to have robust results with significant interactions when we consider both interactions between B_{jit} and variable inputs, and the interactions between B_{1it} and B_{2it}.

Table 3.1 presents the descriptive statistics. As input prices are not available in the FADN, we compute the quantity index for each input using the farm's individual purchases and the average regional prices for the three regions (base 100 in 2010). We have deflated prices and subsidies by the national consumption price index. Here, cereals include the production of soft wheat, durum wheat, rye, spring barley, winter barley, escourgeon, oat, summer crop mix, grain corn, seed corn, rice, triticale, non-forage sorghum and other crops. The yields of cereals are computed in constant euros using a Paasche index based on the mean price of each cereal in 2010. We use individual farmers received prices for milk. We have added annual climatic variables (i.e. variables on rainfall quantity, raining days, snow quantity, snowing days, wind speed, humidity rate and minimum, maximum and average temperature), but we do not report them in Table 3.1.

Table 3.1. Descriptive statistics (N=3,960)

	Mean	**Median**	**Q1**	**Q3**	**Min**	**Max**
Cereal yield (constant €/Ha)	1064.14	1074.04	918.15	1217.05	58.65	2455.44
Milk yield (kg/Ha)	6111.58	6171.39	4553.45	7852.81	276.81	20909.08
log(cereal yield)	6.942	6.979	6.822	7.105	4.071	7.806
log(milk yield)	8.718	8.727	8.423	8.968	5.623	9.947
Crop diversity (B_1)	1.246	1.207	1.021	1.496	0.206	2.287
Permanent grasslands (B_2)	0.10	0.015	0	0.14	0	0.89
UAA (Ha)	90.01	77.62	55.18	110.39	15.59	382.88
Main forage area (Ha)	60.95	53.64	37.27	76.39	8.16	290.9
Fertilizer (quantity index)	9899.41	8028.13	4778.82	12821.82	0	87025.84
Pesticides (quantity index)	6402.45	4843.92	2754.69	7837.9	0	71907
Seeds (quantity index)	6866.18	5575.39	3567.07	8462.67	0	73701.09
Fuel (quantity index)	57.19	47.58	30.56	72.89	0	311.41
Cow feed (quantity index)	282.52	225.19	131.31	368.81	1.702	2803.41
Health and reproduction (quantity index)	54.2	42.77	25.9	74.32	0	407.17
Cattle fertilizer (kg)	8871.66	7456.86	5093.1	10886.78	735.81	45234.26
Other livestock fertilizer (kg)	2076.85	0	0	0	0	95850
Capital (1000€)	299.88	258.30	158.94	383.41	0	3822.41
Labor (annual worker unit/100)	218.19	200	150	272	100	1200

Milk and cereals are the most profitable products of our sample. On average, 56.75% of the revenues originate from milk production, and 9.82% originate from cereal production. The byproducts of milk production are less profitable than cereals. Some farms have other activities, notably pig production (for 11% of farms).

3.5 Results

Table 3.2 reports the GMM estimation of Model 1. We find that crop diversity increases both cereal and milk yields. Permanent grasslands increase cereal yields but do not affect milk yields.

Interestingly, both biodiversity indicators interact negatively with each other for cereal yields, suggesting that they are substitute inputs. B_{2t} increases cereal yields only when its marginal productivity (equals to 0.261-0.217 B_{1t}) is positive, i.e., when B_{1t} is lower than 1.20. Based on the distribution of B_{1t} , B_{2t} increases cereal yields in 46% of our observations. Similarly, B_{1t} increases cereal yields in 89% of our observations (when B_{2t} < 0.35). At the average level of B_{2t} , increasing B_{1t} from an equally distributed acreage between three crops (B_{1t} = 1.099) to an equally distributed acreage between four crops (B_{1t} =1.386) increases cereal yields by 2.3% and milk yields by 2.6%. We find that B_{2t} does not influence cereal and milk yields at the average level of B_{1t} . However, we find that B_{2t} increases cereal yields for low level of B_{1t} . In the case where B_{1t} =1, an increase in B_{2t} from 0.1 to 0.2 leads to an increase of cereal yields by 0.4%, which is relatively small compared to the productivity of crop diversity. These effects can express that landscapes with high hedgerow and permanent grassland densities need a lower complexity of crop mosaic to achieve the same level of biological control in cereal fields than landscapes with low hedgerow density (Martel et al., 2017). It could also represent the benefits from pollination, some crops being sensitive to pollinators while forage do not (Free, 1970). The result that permanent grassland is not an input for milk may seem counterintuitive as permanent grasslands could be used for grazing, but permanent grassland is usually associated with extensive farming, notably with lower level of imported cow feed (Ryschawy et al., 2012).

The variable input productivities are all significantly positive, except for pesticides (non-significant). If the results are relatively similar regarding the first and second-order productivities of the two biodiversity components, the comparison between models 1 and 2 highlights different estimated values of variable input productivities (see Table 3.A1 in Appendices). Model 2 displays notably a negative pesticide productivity on milk if we do not correct for time trend (see models 2a and 2b in Table 3.A1).[39] Parameter restrictions correct for the negative productivity of pesticide on milk. However, estimation of Model 1 shows that the parameter restrictions are significant at 5%, i.e. they are binding constraints. Our sample does not support the validity of these restrictions. It means that there are unspecified complementarities or spillovers between milk and cereals associated with these inputs. Above

[39] Farmers have applied different pesticide types over our sample period. In addition, French legislation has provided signals to reduce pesticide utilization. As milk yields have increased over the whole sample period, this may be a temporal conjuncture confounder.

all, the two specifications of the multi-output technology have no impact on the parameters of interest, confirming that (i) B_{1t} increases crop and milk yields, (ii) B_{2t} can increase crop yields and (iii) B_{1t} and B_{2t} are substitute inputs for cereals.

Table 3.2. GMM estimations of Model 1 (N=3,960)

	log(y_cereals)	log(y_milk)
Biodiversity productive capacity		
B_1	0.077 **	0.096 **
	(0.026)	(0.028)
B_2	0.261 *	0.042
	(0.123)	(0.13)
B_1*B_2	-0.217 *	-0.069
	(0.093)	(0.11)
Variable inputs		
Fertilizer	0.001 ***	0.01 **
	(0.0003)	(0.0003)
Pesticides	0.0001	0.0001
	(0.0003)	(0.0002)
Seeds	0.001 °	0.001 *
	(0.0005)	(0.0004)
Fuel	0.34 **	0.276 **
	(0.108)	(0.09)
Cow feed		0.099 ***
		(0.010)
Health and reproduction		0.193 *
		(0.091)
Organic fertilizer proxies		
Cattle fertilizer/UAA	-0.094 *	-0.115 °
	(0.041)	(0.07)
Other livestock fertilizer/UAA	-0.016	-0.022
	(0.013)	(0.013)
Fixed inputs		
UAA	-2.50E-4	-9.15E-4 *
	(2.65E-4)	(4.16E-4)
Capital/UAA	-0.0001	-0.0006
	(0.0004)	(0.0005)
Labor/UAA	-3.57	2.45
	(2.42)	(2.63)
Technical progress	-0.002	0.002
	(0.015)	(0.002)
Restrictions		
Restriction 1	-2.109 *	
	(1.045)	
Restriction 2	-2.170 *	
	(1.044)	
Restriction 3	-2.310 *	
	(0.959)	

°, *, **, *** significance level at 10%, 5%, 1% and 0.1%. Standard errors in brackets.

SUR and 3SLS estimations (see Table 3.A2 in Appendices) display the same significant signs for biodiversity indicators than GMM ones. However, the levels of estimated productivity are overestimate in SUR and 3SLS. For example, the SUR estimation leads to crop diversity productivities twice larger than the estimated ones with GMM. It highlights the importance of instrumentation of variable input, which otherwise lead to overestimation of the productivity of the two biodiversity components. Our instrument equations display R^2 equal to 0.16 to 0.34. [40]

The addition of control variables is crucial in our estimation. All climatic variables affect significantly cereal yields. Only snow quantity and minimum, maximum and average temperature impact milk yields. The omission of meteorological information leads to negative productivities of certain variable inputs, highlighting that application of variable inputs are influenced by meteorological conditions. The estimation of our model without fixed effects also displays negative productivities. The introduction of weather variables and individual fixed effects reduce the unobserved heterogeneity, removing some endogenous biases. All fixed inputs have null productivity except UAA, which decreases milk yields. UAA captures the lower yields per area of extensive farms. The null productivity of other fixed inputs highlights the difficulty of measuring them effectively. Cattle manure decreases crop yields, but organic fertilization proxies are non-significant otherwise (at statistical level of 5%). It suggests an inefficient management for this public input, which may be due to the existence of legislative constraints on the application of organic fertilizers. The specification of alternative organic fertilization proxies does not influence the significance and the sign of the productivity of B_{2t} or the variable input productivities.

GMM estimation of Model 3 is available in Table 3.3. Like Model 2, we correct the negative pesticide productivity on milk by the addition of an interaction term with a trend. The parameters are overall less significant than in the two previous models, but the interaction terms between the biodiversity indicators and the variable inputs are all significantly negative (i.e. the second-order productivities are negative). High levels of biodiversity indicators decrease the productivity of pesticides and fertilizers. It suggests that the productive capacities of the two biodiversity components are substitute inputs for fertilizers and pesticides. On average, a 10% increase of crop diversity decreases fertilizer and pesticide productivities on cereals by 3,6% and 3,3% respectively. A 10% increase of permanent grassland shares decreases fertilizer and pesticide productivities by 0.6% and 0.9% respectively. The first-order productivities of the

[40] Available on request to the authors.

biodiversity indicators remain significant. At average points, productivities of B_{1t} and B_2 in Model 3 are consistent with those of Model 1 and Model 2, confirming that the different specifications of variable input allocation do not impact our results.

Table 3.3. GMM estimations of Model 3 (N=3,960)

	log(y_cereals)	log(y_milk)
Biodiversity		
B1	0.929 ***	0.095 ***
	(0.248)	(0.027)
B2	2.804 ***	0.038
	(0.589)	(0.055)
Variable inputs		
Fertilizer	0.007 **	0.0004
	(0.002)	(0.0004)
Fertilizer*B1	-0.004 *	
	(0.002)	
Fertilizer*B2	-0.011 ***	
	(0.003)	
Pesticides	0.013 **	0.004 °
	(0.004)	(0.002)
Pesticides*B1	-0.006 *	
	(0.003)	
Pesticides*B2	-0.030 ***	
	(0.008)	
Pesticides*trend		-0.0008 *
		(0.0003)
Seeds	0.001	0.002
	(0.001)	(0.0007)
Fuel	0.190	0.420
	(0.157)	(0.176)
Cow feed		0.066 ***
		(0.012)
Health and reproduction		0.246 **
		(0.090)
Organic Fertilizer proxies		
Cattle fertilizer/UAA	0.037	-0.066
	(0.058)	(0.0005)
Other livestock fertilizer/UAA	0.019	-0.025
	(0.019)	(0.017)
Fixed inputs		
UAA	-3.38E-4	-5.58E-4
	(5.28E-4)	(4.59E-4)
Capital/UAA	-0.0003	-0.0004
	(0.0005)	(0.0005)
Labor/UAA	-8.440	1.863
	(6.079)	(4.892)
Technical progress	0.001	0.005 *
	(0.002)	(0.002)

°, *, **, *** significance level at 10%, 5%, 1% and 0.1%. Standard errors in brackets.

3.6 Discussion and conclusions

Our paper extends the current knowledge on biodiversity productive capacity to (i) several kinds of biodiversity, (ii) several products and (iii) the interactions with conventional variable inputs.

3.6.1 First-order productivities of the two biodiversity components

First, we confirm that crop diversity is an input for cereals. In line with Donfouet et al. (2017), we confirm that crop diversity is also useful for wet regions. This may explain the augmentation of crop diversity in our studied regions between 2007 and 2010 (Desjeux et al., 2015). Second, we find that crop diversity is also an input for milk. We interpret it as the increasing of forage yields, meaning that forages are sensitive to biological control. It might also suggest that cows benefit from more diversified feed. To our knowledge, this is the first time that it is highlighted that crop diversity benefits to other products than crops in the economic literature. We also find that ignoring the optimizing role of the farmers regarding variable input applications lead to an overestimation of the productivities of biodiversity.

We also find that permanent grasslands increase cereal yields, confirming agronomical and ecological studies on the potential benefits of permanent grasslands and related landscape elements on crop production. The productivity of permanent grasslands on cereals emphasizes a productive spillover between semi-natural areas towards arable lands. Klemick (2011) highlighted a similar result on fallow forests in Brazil. Our result may explain the augmentation of grassland shares on crop-orientated French regions (Desjeux et al., 2015), although they are significantly lower than in dairy regions. Desjeux et al. (2015) have shown that permanent grasslands have declined in our case study regions. Our results suggest that it may be due to the lower productivity of permanent grasslands compared to crop diversity productivity. It also might be due to legislative constraints, which increase the cost of permanent grassland management (Nilsson, 2009).

Under the assumption that farmers maximize their profit, we find that biodiversity productive capacities increase yields, suggesting that farmers do manage biodiversity. The cost of their management is equal to the sum of their marginal productivities. We do not find any conflict between high yields and biodiversity but we highlight that the productivity of permanent grasslands is lower than the productivity of crop diversity.

3.6.2 Second-order productivities of the two biodiversity components

One of our most interesting results is the negative interaction term between crop diversity and permanent grasslands in cereal production, which suggest that both biodiversity productive capacities are substitutes for cereals. The elasticity of cereal yields to crop diversity is 0.10% considering only the first-order effect whereas it equals 0.07% when we consider the second-order effects. This result could confirm the recent results in landscape ecology; for example Martel et al. (2017) have found that landscapes with low hedgerow density need a high complexity of crop mosaic to achieve the same level of biological control of landscapes with higher hedgerow density. We conclude that farmers have no incentives to increase both biodiversity productive capacities at the same time. This explanation is consistent with Desjeux et al. (2015) who observed a trade-off between crop diversity and permanent grasslands in most French regions.

In Model 3, we emphasize that both biodiversity productive capacities interact with variable inputs. We find that crop diversity is substitute for pesticides, with an elasticity of pesticide productivity relatively to crop diversity of 0.33%. This extends Di Falco and Chavas (2006) who have found that crop diversity and pesticides are substitute inputs for risk management. We find that crop diversity is substitute for fertilizer, with an elasticity of fertilizer productivity relatively to crop diversity of 0.36%. Kim et al. (2000) have highlighted that soil quality and fertilizer are substitutes in the short-term in USA. Because crop diversity increases soil quality, our results confirm their previous analysis. However, Kim et al. (2000) have also found that soil quality and fertilizer are complements in the long-term. We cannot confirm this result because farmers are only present for four consecutive years in our sample. We should only consider our results valid in the short-term. Moreover, we stress that estimated biodiversity productive capacities are consistent locally and within intensive agricultural regions. The relationship between variable inputs and biodiversity productive capacity may differ in developing regions where variable inputs are limiting inputs.

We find that permanent grasslands are substitute for pesticides and fertilizers in the short-term (with elasticities of 0.09% and 0.06% respectively). This finding could confirm the beneficial role of permanent grasslands and the attached elements on biological control (Baudry et al., 2000). It appears that crop diversity interacts more with variable inputs than permanent grasslands, confirming its more important role in agricultural production. However, in contrast to crop diversity, permanent grasslands play a higher role in crop protection than in crop fertilization, which is consistent with ecological studies (Baudry et al., 2000).

In summary, we have found that (i) crop diversity is an input for both cereals and milk, (ii) permanent grasslands are an input for cereals, (iii) crop diversity and permanent grasslands are substitutes, and (iv) both biodiversity productive capacities are substitutes for mineral fertilizers and pesticides. These results are robust to econometric methods and production function specifications. Our results also contribute, to a larger extent, to the discussions on the benefits of mixed farming (Ryschawy et al., 2012).

3.6.3 Methodological limitations

The decomposition of farmers' annual choices in a two-stage optimization allows considering interactions between biodiversity productive capacities and variable inputs, notably in Model 3. Our results are only valid considering the sequence decision as correct, i.e. when farmers optimize variable input application given the acreage. Our models provide theoretically consistent results for variable input productivities, highlighting that the omission of the variable input instrumentation leads to biased parameters. We have explained that the second-stage optimization can be represented using different variable input allocation specifications in case of multi-output technology. However, the different specifications do not influence the estimated biodiversity productivities. However, we do find evidences that ignoring the optimizing role of the farmers regarding variable input applications lead to an overestimation of the productivities of biodiversity, which supports our theoretical analysis.

Our work still suffers from additional issues. One limit is due to the estimation of the within transformations of (3.7), (3.8) and (3.12) which only allows explaining a small portion of the total variability. Second, we have assumed that biodiversity productive capacities are fixed in the very-short-term. However, similar to variable inputs, acreage shares can be simultaneous to objective yields and may suffer from endogenous bias. Multicrop microeconometric models have stressed the sensitivity of farmers' acreage choices to prices. However, if acreage price elasticities are high between cereals, they are fixed between cereals and other outputs, at least in the short-term (Carpentier and Letort, 2012). This fixity is notably due to diversification costs that prevent farmers from significantly modifying their acreage each year. We can thus consider our biodiversity indicators as "predetermined" and exogenous. The instrumentation of the Shannon index by its lagged values in Di Falco and Chavas (2008) for example illustrates the quasi-fixity of acreage. The hypothesis of "predetermined" biodiversity is however less correct in the long-term. In this case, we should consider biodiversity productive capacities as quasi-fixed inputs and instrument them or construct a structural model that explicitly consider biodiversity dynamics, notably to capture crop rotation effects (Hendricks et al., 2014).

3.6.4 Implications for environmental policies

Policymakers aim to increase the levels of environmental quality and biodiversity due to their beneficial effects on social welfare. Our results can benefit to policymakers because they emphasize the incentives encountered by profit-maximizing farmers managing biodiversity. The first-order effects highlight that both biodiversity productive capacities increase cereal and milk yields, suggesting that there are no conflicts between high yields and biodiversity. The second-order effects stress the difficulty of designing optimal sets of policy instruments targeting crop diversity and permanent grasslands at the same time. Policy instruments providing incentives to the enhancement of crop diversity also favor a decrease of permanent grasslands and *vice-versa*. This substitution is amplified because crops and permanent grasslands are competitors for land and farmers have limited UAA. Thus, cross-compliance requirements introduced in the CAP 2014 reform may lead to counterintuitive acreage evolutions. Indeed, crop-orientated regions (with high dotation of crop diversity) receive incentives to enhance ecological focus areas and permanent grasslands; this, in turn, leads to a decrease of marginal productivity of crop diversity and finally, assuming profit-maximizing farmers, to reduction of crop diversity.

Finally, we want to emphasize the optimistic implications of the Model 3 results. We find that variable inputs and biodiversity productive capacity are substitutes, at least in the short-term and in intensive agricultural regions. Thus, the taxation of polluting inputs would provide incentives to farmers to increase biodiversity levels. Because we do find that biodiversity and variable input are non-complementary substitute inputs, biodiversity augmentation should not suffer from any mitigation effects. Similarly, biodiversity subventions should favor farmers to reduce the application of fertilizers and pesticides. Environmental policies could reach several objectives together.

If our results provide new insights on biodiversity management, they only concern yields (i.e. the biodiversity effects at the intensive margin). To really improve policy measures, future researches should focus on the effects of biodiversity on acreage choices (i.e. the biodiversity effects at the extensive margin), notably in a dynamic framework. This would better characterize the existing conflicts between agriculture and biodiversity.

3.7 References

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3.8 Appendices

3.A. Verification of parameter restrictions in case of log-linear production function and unobserved variable input application

We consider the system composed of (3.7) and (3.8). We verify the parameter restriction (3.4) in this system. We compute the marginal productivities of x_l ($l \in [1;4]$) on cereals (using (3.7)) and milk (using (3.8)). Noting that $X_{lit} = \bar{a}_{1it}x_{l1it} + \bar{a}_{2it}x_{l2it}$, we have respectively:

$$\begin{cases} \dfrac{\partial \log(y_{1it})}{\partial x_{l1it}} = \beta_{l1}\dfrac{\bar{a}_{1it}}{A_{it}} \\ \dfrac{\partial \log(y_{2it})}{\partial x_{l2it}} = \beta_{l2}\dfrac{\bar{a}_{2it}}{A_{it}} \end{cases}$$

Which is equivalent to:

$$\begin{cases} \dfrac{\partial y_{1it}}{\partial x_{l1it}} = \beta_{l1}\dfrac{\bar{a}_{1it}}{A_{it}}y_{1it} \\ \dfrac{\partial y_{2it}}{\partial x_{l2it}} = \beta_{l2}\dfrac{\bar{a}_{2it}}{A_{it}}y_{2it} \end{cases}$$

Thus, we have $\forall l \in [1;4]$:

$$\frac{\dfrac{\partial y_{1it}}{\partial x_{l1it}}}{\dfrac{\partial y_{2it}}{\partial x_{l2it}}} = \frac{\beta_{l1}\bar{a}_{1it}y_{1it}}{\beta_{l2}\bar{a}_{2it}y_{2it}}$$

Because $\bar{a}_{1it}y_{1it}$ and $\bar{a}_{2it}y_{2it}$ do not depend on x_l, we do have (3.9), (3.10) and (3.11).

These restrictions would hold as well if we introduce Y_{2it} in cereal yield function explicitly and vice-versa (see relations (3.4)).

Table 3.A1. GMM estimations of Model 2 (N=3,960)

	Model 2a		Model 2b	
	log(y_cereals)	log(y_milk)	log(y_cereals)	log(y_milk)
Biodiversity				
B1	0.081 **	0.117 ***	0.075 **	0.090 **
	(0.026)	(0.030)	(0.027)	(0.034)
B2	0.234 °	-0.049	0.225 °	-0.101
	(0.126)	(0.134)	(0.126)	(0.139)
B1*B2	-0.207 *	0.002	-0.195 *	0.012
	(0.094)	(0.116)	(0.094)	(0.121)
Variable inputs				
Fertilizer	0.002 ***	0.0001	0.002 ***	0.0005
	(0.001)	(0.0005)	(0.001)	(0.0005)
Pesticides	0.0003	-0.002 **	0.0003	0.005 °
	(0.0004)	(0.001)	(0.0004)	(0.003)
Pesticides*trend				-0.001 *
				(0.0004)
Seeds	0.001 °	0.001	0.001 °	0.001
	(0.001)	(0.0008)	(0.001)	(0.0008)
Fuel	0.118	0.539	0.136	0.518
	(0.131)	(0.139)	(0.131)	(0.143)
Cow feed		0.101 ***		0.101 ***
		(0.014)		(0.014)
Health and reproduction		0.189 °		0.171
		(0.113)		(0.121)
Organic fertilizer proxies				
Cattle fertilizer/UAA	-0.045	-0.167 *	-0.050	-0.192 *
	(0.048)	(0.079)	(0.048)	(0.080)
Other livestock fertilizer/UAA	-0.006	-0.032	-0.006	-0.040 °
	(0.013)	(0.019)	(0.014)	(0.021)
Fixed inputs				
UAA	3.70E-04	-0.0005	3.80E-4	-0.0007
	(3.21E-4)	(0.0005)	(3.21E-4)	(0.0005)
Capital/UAA	0.001	-0.0009	0.001	-0.001 °
	(0.001)	(0.0005)	(0.001)	(0.0006)
Labor/UAA	-4.186	4.556	-4.304	7.503
	(3.950)	(4.739)	(3.952)	(4.953)
Technical progress	-0.016	0.002	-0.018	0.004
	(0.026)	(0.002)	(0.026)	(0.003)

°, *, **, *** significance level at 10%, 5%, 1% and 0.1%. Standard errors in brackets.

Table 3.A2. SUR and 3SLS estimations of Model 1 (N=3,960)

	SUR		3SLS	
	log(y_cereals)	**log(y_milk)**	**log(y_cereals)**	**log(y_milk)**
Biodiversity productive capacity				
B1	0.132 ***	0.193 ***	0.132 ***	0.110 ***
	(0.021)	(0.017)	(0.021)	(0.033)
B2	0.281 *	-0.048	0.225 °	-0.154
	(0.115)	(0.093)	(0.119)	(0.139)
B1*B2	-0.210 *	-0.067	-0.197 *	-0.023
	(0.093)	(0.075)	(0.095)	(0.111)
Variable inputs				
Fertilizer	3.3E-5	0.0001	-0.0001	0.0005
	(3.1E-4)	(0.0001)	(0.0001)	(0.0005)
Pesticides	0.0001 *	0.0004 *	0.0002	-0.003 **
	(0.0006)	(0.0001)	(0.0003)	(0.0001)
Seeds	0.0001	0.0004 **	0.0001	-0.001
	(0.0005)	(0.0004)	(0.0001)	(0.0009)
Fuel	0.007	0.020	-0.043	0.502 **
	(0.006)	(0.016)	(0.055)	(0.16)
Cow feed		0.049 ***		0.134 ***
		(0.002)		(0.013)
Health and reproduction		0.081 ***		0.209 °
		(0.008)		(0.123)
Organic fertilizer proxies				
Cattle fertilizer/UAA	0.044	0.165 ***	0.006	-0.310 ***
	(0.030)	(0.025)	(0.037)	(0.07)
Other livestock fertilizer/UAA	-0.014	-0.017 °	-0.033 *	-0.063 ***
	(0.012)	(0.009)	(0.014)	(0.018)
Fixed inputs				
UAA	-2.39E-7	-8.7E-6 ***	5.9E-6 °	-2.6E-7
	(2.40E-4)	(1.95E-6)	(3.5E-6)	(4.12E-6)
Capital/UAA	-0.0001	0.001 ***	-0.0002	-0.001
	(0.0004)	(0.0003)	(0.0004)	(0.0005)
Labor/UAA	-0.529	2.057 ***	7.798 *	14.34 **
	(0.717)	(0.579)	(3.126)	(4.84)
Technical progress	-0.011 *	-0.003	-0.019	-0.001
	(0.005)	(0.002)	(0.019)	(0.003)
Restrictions				
Restriction 1	-2.376 *		-5.036 ***	
	(0.943)		(1.213)	
Restriction 2	0.30		3.582	
	(2.005)		(3.047)	
Restriction 3	0.754		8.765 **	
	(0.919)		(3.236)	

°, *, **, *** significance level at 10%, 5%, 1% and 0.1%. Standard errors in brackets.

CHAPTER 4. HOW DO FARMERS MANAGE CROP BIODIVERSITY OVER TIME? A DYNAMIC ACREAGE MODEL WITH PRODUCTIVE FEEDBACK [41]

Chapter 3 examines the productivity of biodiversity in the very short term (during agricultural campaign) when biodiversity can be considered exogenous. However, Chapter 2 theoretically links annual acreage choices and biodiversity productive capacity such that biodiversity should be considered endogenous in the short term (between two agricultural campaigns). This chapter develops and estimates the general model of Chapter 2 in the short term considering the supply functions for three crops and the variable input crop-specific demand for fertilizers and pesticides. We provide evidence that farmers manage biodiversity by using a dynamic framework, considering biodiversity productive capacity as a special case of capital (i.e., a quasi-fixed input). The short- and long-term elasticities of yields, variable input crop-specific demands and crop-specific gross margins highlight the incentives for farmers to conserve biodiversity. We estimate the impact of a 100% *ad valorem* tax on pesticides using three models: the developed dynamic model, the static model of Chapter 2 and Femenia and Letort's (2016) model where biodiversity productive capacity is not considered. The comparison of the tax impacts on pesticide applications and gross margins highlights the interest of our model: farmers respond more to pesticide tax when biodiversity productive capacity is explicitly considered as a quasi-fixed input.

[41] This chapter was coauthored with Elodie Letort (INRA, SMART-LERECO) and has been published in the European Review of Agricultural Economics under the reference: François Bareille, Elodie Letort; How do farmers manage crop biodiversity? A dynamic acreage model with productive feedback, European Review of Agricultural Economics, Volume 45, Issue 4, 1 September 2018, Pages 617–639, https://doi.org/10.1093/erae/jby011

4.1 Introduction

It is widely recognized that human activities, especially modern agriculture, have negative impacts on biodiversity (MEA, 2005). The simplification of habitats from natural areas to arable lands (and monoculture) has decreased biodiversity levels. Because biodiversity greatly contributes to the functioning of the ecosystem, this loss threatens the provision of valuable ecological functionalities. Biodiversity is a crucial issue not only for our society but also for the sustainability of agriculture. Indeed, these functionalities support the provision of ecosystem services that provide suitable agricultural production conditions (MEA, 2005). Few authors have emphasized the productive value of biodiversity for crop farms (see Di Falco, 2012 for a review). These authors have usually estimated the effects of crop biodiversity using primal production functions or reduced form profit functions. Because measures of species density on point maps are often unavailable in databases, biodiversity is generally approximated by indicators based on land use, such as the Shannon index to measure crop diversity (e.g., Donfouet et al., 2017). In this paper, like most agricultural economists, we focus on crop diversity to approximate the level of biodiversity at the farm level. [42] This approach is highly influenced by landscape ecology, which postulates that landscape structure, defined by both its composition and configuration, determines species dynamics and, hence, species density (Burel and Baudry, 2003). In particular, crop diversity increases the likelihood of species diversity (Di Falco, 2012). It also improves several ecosystem services such as the nutrient stock, the soil structure (Mäder et al., 2002), pollination (Kennedy et al., 2013) and biological control (Letourneau et al., 2011). Of course, crop diversity is only an indirect indicator and does not reflect the complexity of the notion of biodiversity.

From our point of view, the economic literature on crop biodiversity emphasizes two main empirical results. First, crop diversity increases the mean yield and reduces the variance yield. This finding has led authors to consider both a productive value of biodiversity (Chavas, 2009) and an insurance value of biodiversity (Baumgärtner, 2007). Second, crop diversity of the previous year increases current production (Di Falco and Chavas, 2008). This result suggests that the productive effects of biodiversity persist over time.

Because biodiversity levels depend on land use, the current productive capacity of biodiversity depends on current and past acreage decisions. In this case, a dynamic model is necessary to represent production and acreage decisions. Here, we propose a dynamic acreage model

[42] We use the terms "crop biodiversity" and "crop diversity" interchangeably.

considering that farmers manage their biodiversity as capital. Similar to how firms make certain investment decisions to benefit from the productive capacity of capital, we assume that farmers make cropland decisions to benefit from the productive capacity of crop biodiversity. Thus, our objective is to confirm that farmers make cropland decisions with the aim of maintaining their current and future productive capacities. Therefore, we compile literatures on the productivity of crop diversity and acreage choices (e.g., Chambers and Just, 1989). Compared to other studies on biodiversity productivity, we extend this analysis to land allocation and variable input applications. These choices partly explain farmers' behaviours regarding the productive capacity of biodiversity. This concept may be relevant, especially for impact analyses of the economic incentives associated with biodiversity management and for evaluations of agro-environmental measures designed to maintain and promote biodiversity. We consider only the mean effect of biodiversity on the yield and input use, but the literature indicates that biodiversity reduces also the probability of a low yield (Di Falco and Chavas, 2009). It is theoretically possible to consider risk aversion and the impact of biodiversity on production variability in our model. Nevertheless, in practice, this inclusion would complicate the model notably with regard to the number of parameters to be estimated. We focus on the estimation of dynamic effects, which is already a more complicated approach than the standard multicrop model of crop allocation.

To our knowledge, few papers have considered the dynamics of acreage allocation within a dynamic theoretical farm-level model. One exception is the work of Orazem and Miranowski (1994), who built a dynamic model of acreage allocation. They assumed that farmers' acreage allocation decisions are conditional on their current stock of soil capital, which depends on past acreage allocations. Orazem and Miranowski considered that some crops increase future soil quality and thus have positive productivity effects. The main idea of their paper is similar to that of ours. Nevertheless, there are several key differences. First, their soil indicator is defined by crops, while our biodiversity indicator is implemented at the farm level. Their assumption technically suggests that the soil indicator of a crop depends on the past acreage of all crops and on only the current acreage of the considered crop. Orazem and Miranowski used this assumption to represent crop rotation effects. Our biodiversity indicator depends on the current and past acreages of all crops. Our specification expresses that crop yields depend not only on past crop diversity but also on current crop diversity, which agrees with Di Falco and Chavas (2008). This dependence complicates the derivation of acreage equations but better represents farmers' behaviour. Second, Orazem and Miranowski (1994) did not consider the potential effects of soil quality on input use, such as fertilizer application. This issue requires the imposition of identifying restrictions and leads to a less efficient estimation of parameters associated with the productive effects of soil quality. Here, we propose to estimate together acreage, input application and output supply equations.

Another interesting paper is that of Thomas (2003), who presented a dynamic model of nitrogen management at the farm level considering root crops and fertilizer as the two sources of nitrogen. He measured farmers' fertilizer application decisions considering that farmers account for nitrogen accumulation, i.e., the nitrogen stock available for the next period as a result of current production decisions. Similar to Orazem and Miranowski (1994), Thomas (2003) provided a framework to explain crop rotation decisions with a temporal lag in acreage decisions. Although his dynamic optimization programme is quite similar to ours, his theoretical model differs in three main respects. First, he focused on the effect of the nitrogen stock on fertilizer decisions and did not consider the other productive effects of crop rotations, such as biological control. Second, his state variable, the carry-over nitrogen, is a function of past nitrogen levels in plots and does not depend on current acreage decisions. Third, he assumed that farmers can instantaneously adjust their land allocation, while Oude Lansink and Stefanou (2001) found that area adjustments are quite slow.

Indeed, Oude Lansink and Stefanou (2001) proposed a dynamic model of acreage allocation to derive dynamic measures of scope and scale economies. Contrary to Orazem and Miranowski (1994) and Thomas (2003), they estimated reduced-form equations rather than a structural model. The originality of their acreage model is associated with the use of adjustment costs. They consider that output-specific areas evolve over time and that these area adjustments are costly. These costs are associated with the underutilization of fixed inputs or the reorganization of the farm operation. Adjustment costs have already been used in investment and employment literature. The adjustment cost function captures the fact that the productivity effects of quasi-fixed inputs are not instantaneous because producers incur additional costs in adjusting their stocks of capital and labour. Carpentier and Letort (2012, 2014) and Kaminski et al. (2013) used a similar cost function within a static multioutput acreage allocation model. In these cases, these costs were interpreted as the implicit costs linked to the management of both crop rotation constraints and quasi-fixed input constraints.

Our work is also based on the concept of adjustment costs for land allocation, but our modelling is different in one important way. In Oude Lansink and Stefanou (2001), the long-term productive effects of crop diversity are captured by a cost function. Their dual approach does not allow them to differentiate these productive effects from the adjustment costs associated with adjusting areas. Similarly, the utilization of an implicit cost function in the static acreage literature does not allow for the examination of the beneficial effects of crop diversification because it captures both the costs of fixed input management for a multioutput firm and the "negative costs" (i.e., the benefits) of crop diversity linked to the productive capacity of crop biodiversity. Our framework allows for the disassociation of the benefits and costs of crop diversification. Another interesting feature of our model is that we use an explicit representation of production technology. The explicit representation of the technology is useful for testing various adjustment cost functions within a dynamic investment model (e.g., Gardebroek 2004) and for studying environmental problems within a static land allocation model (e.g., Femenia and Letort, 2016). In our model, the specification of production technology allows us to explicitly analyse the impacts of the productive capacity of crop biodiversity on output yields and variable input savings.

The next section presents the theoretical model and a discussion of the economic interpretation. In the third section, we propose an empirical counterpart to this theoretical framework. Output supply and input demand equations, as well as first-order conditions regarding acreage choices, are estimated for a sample of French farms between 2007 and 2012. The fourth section presents the results, and the final section concludes the paper.

4.2 The dynamic model of acreage decisions

In this paper, we consider the productive capacity of crop biodiversity as a quasi-fixed input. Inspired by the investment literature, we develop a model that combines a multioutput farm model with a specific representation of the production technology and the specific dynamics of quasi-fixed inputs. This multi-output farm model is presented in the first part of this section. The dynamic framework is described in the second part.

4.2.1 The multioutput model of acreage decisions

Our modelling framework relies on models that are derived from a profit maximization problem with land as an allocable fixed input. These models are well-known in the agricultural economics literature (see, e.g., Chambers and Just 1989, Moore and Negri 1992, Wu and Segerson 1995, Oude Lansink and Peerlings 1996, Fezzi and Bateman 2011, Carpentier and Letort 2012). In our approach, price-taker farmers produce multiple outputs for which they choose the optimal quantity of variable inputs and the optimal allocation of land given the amount of fixed inputs applied based on price and production expectations.

The total restricted profit function Π_t of year *t* is defined as the sum of the gross margins per hectare π_{kt} of each output *k* (*k* ϵ[1, *K*]) multiplied by the acreage S_{kt} minus the acreage management costs defined by the function $H(\mathbf{S}_t)$:

$$\Pi_t\left(\mathbf{x}_t, B_t, \mathbf{S}_t; \mathbf{z}_t\right) = \sum_{k=1}^{K} S_{kt}\pi_{kt}\left(\mathbf{x}_{kt}, B_t; \mathbf{z}_t\right) - H\left(\mathbf{S}_t\right) \tag{4.1}$$

The function $H(\mathbf{S}_t)$ is assumed to be convex in $\mathbf{S}_t$. The gross margin per hectare π_{kt} of output *k* depends on the vector of variable input quantities $\mathbf{x}_{kt}$, the biodiversity indicator B_t and the vector of fixed input quantities $\mathbf{z}_t$. We consider that the gross margins for each output *k* do not depend explicitly on $\mathbf{S}_t$ (i.e. present constant return to acreage), but do depend indirectly on $\mathbf{S}_t$ thanks to B_t (see the discussion on the model assumptions below and section 4.3.1. on the construction of the biodiversity indicator). In a static framework, farmers choose their acreage according to the following optimization problem:

$$\max_{\mathbf{S}_t} \Pi_t\left(\mathbf{x}_t, B_t, \mathbf{S}_t; \mathbf{z}_t\right) \text{ s.t. } \sum_{k=1}^{K} S_{kt} = L_t \tag{4.2}$$

where L_t is the total land quantity for crops $k = 1, ..., K$. The gross margin π_{kt} is derived from the following optimization problem:

$$\pi_{\mathrm{kt}} = \max_{\mathbf{x}_{kt}} \left\{ \begin{array}{c} p_{kt}y_{kt} - \sum_{i=1}^{I} w_{it}x_{ikt} \\ \text{s.t. } y_{kt} = F_{kt}\left(\mathbf{x}_{kt}, B_t; \mathbf{z}_t\right) \end{array} \right\} \tag{4.3}$$

where y_{kt} is the yield of the output *k* per hectare at time *t* and x_{ikt} (*i* ϵ[1, *I*]) is the quantity of variable input *i* applied to output *k* per unit of land at time *t*. $F_{kt}\left(\mathbf{x}_{kt}, B_t; \mathbf{z}_t\right)$ is the production function, which is non-decreasing in $\mathbf{x}_{kt}$ and strictly concave in $\mathbf{x}_{kt}$.

Our modelling framework differs from that of other models that treat land as an allocable fixed input based on three main points. These specific features are partly shared with the model proposed by Carpentier and Letort (2012, 2014). First, it relies on an explicit representation of crop production technology. Standard dual models are almost exclusively used to model farmers' behaviours regarding the explicit allocation of fixed factors. However, they are based on reduced-form functions and implicit production technology, which are not always well suited for analyses of environmental problems, such as the impact of input reduction policies (Femenia and Letort, 2016). In our model, the specification of production technology allows us to analyse the productive effects of crop biodiversity.

The second interesting feature is the utilization of the function $H(\mathbf{S}_t)$ in the total restricted profit function. This type of function has already been used in the investment and employment literature. The authors interpret this function as the adjustment costs linked to quasi-fixed input management and capture the non-instantaneous nature of the profitable effects of quasi-fixed inputs. Adjustment costs due to land allocation have previously been considered. For example, Oude Lansink and Stefanou (2001) found that although Dutch farmers have incentives for specialization, high adjustment costs prevent them from specializing. Carpentier and Letort (2012, 2014) and Kaminski et al. (2013) used a function similar to $H(\mathbf{S}_t)$ within a static multioutput acreage allocation model. They interpreted the function as the implicit costs linked to crop rotation management and quasi-fixed input constraints. Here, we use the same interpretation of the function. However, because we capture some crop rotation effects in the production functions, our cost function should mainly represent the farmers' fixed input constraints. An interesting consequence is that we capture the benefits of crop diversification on each of the gross margin π_{kt} and the costs of crop diversification (i.e., the management costs of quasi-fixed inputs) in the implicit cost function $H(\mathbf{S}_t)$. In addition, the adjustment cost model offers a methodological advantage: it provides a simple dynamic theoretical framework (which is presented in the next part).

Third, the modelling framework generally used by agricultural economists to represent farmers' acreage decisions considers one or two motives of crop diversification. The main motives of crop diversification are decreasing returns to scale (or more generally scale economies), risk spreading, crop rotation effects, and constraints associated with allocated quasi-fixed factors (other than land). Most multicrop econometric models that consider land as fixed but allocable focus on decreasing marginal returns to crop acreage (e.g., Chambers and Just 1989, Moore and Negri 1992) or on market risk spreading (e.g., Chavas and Holt 1990) as the motives for crop diversification. Crop rotation effects are more rarely considered in multicrop econometric models, likely due to the complexity of dynamic choice modelling (e.g., Orazem and Miranowski 1994, Thomas 2003). The constraints associated with allocated quasi-fixed factors are used as motives for crop diversification in some multicrop econometric models (e.g., Oude Lansink and Stefanou 2001, Carpentier and Letort 2012, 2014, Kaminski et al. 2013) and in some positive mathematical programming models (e.g., Howitt 1995). In our model, the motives of crop diversification are represented by the implicit cost function $H(\mathbf{S}_t)$, which approximates the constraints associated with the limiting quantities of quasi-fixed inputs, and the productivity effects of crop diversity captured in each of the gross margin π_{kt}. Consequently, our model relies on two main assumptions. The first one is farmers' risk neutrality. Although it appears restrictive, it is imposed in all multicrop model not considering risk issues.[43] The second is the assumption of constant returns to acreage as stated in the definition of the gross margins (4.3).[44] This assumption is used as a simplifying assumption in multicrop econometric models considering risk spreading or constraints associated with allocated quasi-fixed factors as motives for crop diversification.[45]

4.2.2 The dynamic framework

Although the productivity of crop biodiversity can be assessed within a static model, crop biodiversity levels will be misjudged because land-use dynamics are not considered. Indeed, acreage decisions affect biodiversity dynamics and, in turn, affects productive capacity of crop biodiversity in the future (Di Falco and Chavas, 2008). Therefore, we must consider that farmers maximize their acreage decisions taking into account that their acreage decisions influence current and future levels of the productive capacity of biodiversity. Accordingly, we assume that farmers maximize the expected value of future discounted profits over the entire period $[1;T]$:

$$\max_{\mathbf{S}_t} E_t \left\{ \sum_{t=1}^{T} \left(\frac{1}{1+r} \right)^{t-1} \Pi_t(\mathbf{x}_t, B_t, \mathbf{S}_t; \mathbf{z}_t) \right\} \tag{4.4}$$

---

[43] The examination of farmers' risk-reducing strategies in the context of crop biodiversity management should be a promising area of research. Indeed, crop biodiversity reduces the probability of low yield realization as well as the magnitude of the yield shortfall under stress (e.g. Di Falco and Chavas, 2008). Here, we only consider the mean effects of crop biodiversity on yields and ignore the potential implications of crop biodiversity properties for risk-averse farmers.

[44] Note, however, that gross margins depend indirectly on acreage thanks to the biodiversity indicator.

[45] Nevertheless, our model can be easily adapted for non-constant returns to crop acreage and allow scale effects in a simple way; therefore, the parameters of the production functions can be defined as linear functions of crop acreage (Carpentier and Letort, 2010).

where r is the interest rate. The productive capacity of biodiversity evolves according to:

$$B_t = (1-\delta_t)B_{t-1} + g(\mathbf{S}_t) \tag{4.5}$$

and

$$\sum_{k=1}^{K} S_{kt} = L_t \, . \tag{4.6}$$

We propose a dynamic form for the biodiversity equation. The productive capacity of biodiversity in t depends on the current acreages in t and past acreages (years before t). The $g(\mathbf{S}_t)$ function is the biodiversity indicator that depends on $\mathbf{S}_t$. Farmers can manage this function each year. Based on the investment literature, $g(\mathbf{S}_t)$ can be considered as an investment in the productive capacity of biodiversity. The term $(1-\delta_t)B_{t-1}$ represents the inherited portion of the productive capacity of biodiversity from years before t. Farmers cannot manage this factor in t because it depends on past acreage decisions. This representation agrees with those in the literature. Indeed, previous studies have noted that the beneficial effects of crop biodiversity on production can last more than two years, even if these effects decrease over time (Hennessy 2006, Di Falco and Chavas 2008). Theoretically, this parameter depends on the natural conditions, notably climatic variations (e.g. Di Falco and Chavas, 2008) or soil and moisture conditions, as these factors may influence species dynamics. Nevertheless, for empirical purposes, we consider a single parameter δ in the following, meaning that we implicitly assume that δ_t is fixed over time.

Below, we examine the implications of the different values of the parameter δ, which is a key parameter in the estimation. When $\delta = 1$, the productive capacity of crop biodiversity depends only on current acreage decisions; past acreage decisions have no effect on current production. When $\delta = 0$, the productive capacity of crop biodiversity equally depends on past and current acreage decisions. When $\delta < 0$, the past productive capacity of biodiversity has a greater effect than current acreage decisions, meaning that the benefits of biodiversity are irreversible and can accumulate over time. Finally, when $\delta > 1$, the past productive capacity of biodiversity has a negative impact on the current capacity. These last two cases are difficult to justify from an ecological point of view. Thus, this parameter should range between 0 and 1. In this case, the productive capacity of biodiversity increases every year, but this increase becomes increasingly less important. After an acreage change damages biodiversity (monoculture is an example), the productive capacity of biodiversity decreases, but not instantaneously. Overall, a δ value between 0 and 1 suggests that the past productive effects of crop biodiversity still have positive impacts on production, but these effects decrease over time (Hennessy 2006, Di Falco and Chavas 2008). These potential cases are illustrated in Appendix 4.A.

From a technical perspective, we propose another way of interpreting this equation. As biological protection and net primary production depend on the current acreage composition and configuration (Burel and Baudry, 2003), the productive effects of the current acreage can be interpreted as a spatial choice. In contrast, because crop rotation effects depend on the preceding crops (Hennessy, 2006), the productive effects of past acreage may be perceived as a temporal choice. Here, because equation (4.5) assumes that farmers manage their acreages to benefit from current and future productive effects at the same time, we consider acreage choices to be spatiotemporal choices. In this case, the δ parameter reflects the importance of the farmers' temporal acreage management versus the farmers' spatial acreage management.

Let $V_t(B_t)$ be the maximum value of the function in (4.4) at period *t*, where B_t is the state variable of the model. According to the maximum principle, the dynamic optimization problem can be resolved using the Bellman equation:

$$V_t(B_t) = \max_{\mathbf{S}_t} E\left\{\Pi_t + \frac{1}{1+r}\left[V_{t+1}(B_{t+1})\right]\right\} \tag{4.7}$$

Equation (4.7) illustrates the inter-temporal problem faced by farmers. Assuming an interior solution, the first-order conditions associated with the maximization of $V_t(B_t)$ according to x_{ikt} for $i \in [1; I]$ and $k \in [1; K]$ are defined by the following formula:

$$p_{kt}\frac{\partial F_{kt}}{\partial x_{ikt}} - w_{it} = 0 \tag{4.8}$$

Given optimal levels of B_t, farmers apply variable inputs such that the marginal cost of the last applied input unity equals its marginal benefit. The calculation of first-order conditions for acreage decisions are more complex. Farmers must optimize $\mathbf{S}_t$ according to $\mathbf{S}_{t-1}$ while anticipating the marginal effect of those choices on $V_{t+1}(B_{t+1})$. For a sake of simplification, we do not integrate the binding land constraint in this section but we present the derivation of the empirical model with the binding land constraint in Appendix 4.B. Without the binding land constraint, the first-order conditions for acreage are defined by:

$$\frac{\partial V_t}{\partial S_{kt}} = \frac{\partial \Pi_t}{\partial S_{kt}} + \frac{1}{1+r} E\left[\frac{\partial V_{t+1}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{kt}}\right] = 0 \tag{4.9}$$

with $$E\left[\frac{\partial V_{t+1}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{kt}}\right] = \frac{\partial \Pi_{t+1}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{kt}} + \frac{1}{(1+r)} E_t\left[\frac{\partial V_{t+2}}{\partial B_{t+2}} \frac{\partial B_{t+2}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{kt}}\right] \tag{4.10}$$

Noting that $\frac{\partial B_t}{\partial S_{kt-1}} = (1-\delta)\frac{\partial g(\mathbf{S}_{t-1})}{\partial S_{kt-1}} = (1-\delta)\frac{\partial B_t}{\partial S_{kt}}$, and following recursive reasoning, we have:

$$E\left[\frac{\partial V_{t+1}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{kt}}\right] = \sum_{i=1}^{\infty} \frac{(1-\delta)^i}{(1+r)^{i-1}} E\left[\frac{\partial \Pi_{t+i}}{\partial B_{t+i}} \frac{\partial B_{t+i}}{\partial S_{kt+i}}\right] \tag{4.11}$$

The first-order condition for acreage choice S_{kt} is then defined by:

$$\pi_{kt} + \sum_{j=1}^{K} S_{jt} \frac{\partial \pi_{jt}}{\partial B_t} \frac{\partial B_t}{\partial S_{kt}} - \frac{\partial H}{\partial S_{kt}} + \sum_{i=1}^{\infty} \frac{(1-\delta)^i}{(1+r)^i} E\left[\sum_{j=1}^{K} S_{jt+i} \frac{\partial \pi_{jt+i}}{\partial B_{t+i}} \frac{\partial B_{t+i}}{\partial S_{kt+i}}\right] = 0 \tag{4.12}$$

To interpret equation (4.12), let us compare the first-order conditions of acreage in different models. In a static framework, as reported by Letort and Carpentier (2012, 2014), the conditions become:

$$\pi_{kt} = \frac{\partial H}{\partial S_{kt}} \tag{4.13}$$

In this case, the optimal acreage for crop *k* is obtained when its gross margin, depending only on variable inputs, is equal to its marginal cost of adjustment.

In a static framework considering the productive effect of crop biodiversity, as defined by Di Falco and Perrings (2005) or Di Falco and Chavas (2006 and 2009), we have the following condition:

$$\pi_{kt} + \sum_{j=1}^{K} S_{jt} \frac{\partial \pi_{jt}}{\partial B_t} \frac{\partial B_t}{\partial S_{kt}} = \frac{\partial H}{\partial S_{kt}} \tag{4.14}$$

In this case, the marginal benefit of one additional unit of area devoted to crop *k* is defined as the gross margin of *k* plus the marginal profitability of the productive capacity of biodiversity on the other outputs linked to the reorganization of the total acreage. These effects include the productivity of crop biodiversity (i.e., $p_{kt}\,\partial F_{kt}/\partial B_t$) and the variable input savings due to the productive capacity of biodiversity (i.e., $w_{it}\,\partial x_{ikt}/\partial B_t$). These marginal benefits should be equal to the marginal cost of adjustment. Comparing our approach with the acreage literature (e.g., Carpentier and Letort, 2012), equation (4.14) illustrates the separation of the beneficial effects of crop diversity from the implicit cost function. Comparing our approach with the literature on the productive value of biodiversity, equation (4.14) also illustrates the importance of the effects of adjustment costs in explaining biodiversity levels at the farm scale. This model is the one we have discussed in Chapter 2.

In our dynamic framework, the conditions are defined by the following equation (considering an optimization problem with two periods):

$$\pi_{kt} + \sum_{j=1}^{K} S_{jt}\frac{\partial \pi_{jt}}{\partial B_t}\frac{\partial B_t}{\partial S_{kt}} + \frac{(1-\delta)}{(1+r)}E\left[\sum_{j=1}^{K} S_{jt+1}\frac{\partial \pi_{jt+1}}{\partial B_{t+1}}\frac{\partial B_{t+1}}{\partial S_{kt+1}}\right] = \frac{\partial H}{\partial S_{kt}} \tag{4.15}$$

These conditions state that the marginal revenue per hectare of crop *k* at time *t* should be equal to the marginal adjustment cost due to marginal change in area *k*. The marginal revenue is defined by the gross margin of crop *k* plus the marginal profitability of the productive capacity of biodiversity for all crops plus the discounted expected marginal value of the crop biodiversity gain at time *t+1*. In other words, farmers consider the future productive effects of crop biodiversity when making their current acreage decisions. Considering the discounted expected marginal value of the crop biodiversity gain at time *t+1* as the future benefits of the current productive capacity of biodiversity, equation (4.15) can be interpreted as the equality between the adjustment costs due to the current acreage and the sum of the current and future benefits due to the current acreage. Equation (4.15) illustrates that price expectations affect the current acreage choices. The influence of price expectations is more important when the future impacts of the productive capacity of biodiversity are high, i.e., when δ is low. Our empirical model aims to estimate the magnitude of the effects of the productive capacity of biodiversity and to estimate the value of δ.

4.3 The empirical model

In this section, we propose an empirical counterpart to the theoretical framework. The data and the sample used for the application are described in the first subsection. The set of estimated equations comprises output supplies, input demands and first-order conditions for acreage choices, all of which are presented in the second subsection.

4.3.1 Data and variables

We use a dataset from a sample of farms located in the French territorial division of *La Meuse* observed between 2007 and 2012. The dataset comes from a local accounting agency and provides information on acreage, yields, and output prices. Contrary to most alternative French economic databases, it provides the variable input quantities applied per crop. Femenia and Letort (2016) used this database to estimate a static acreage model and simulate pesticide taxation policies. Because we consider the dynamics of the acreage choices, we select farms that have been identified for at least two consecutive years. We explain farmers' choices regarding the three main crops of the region, i.e., wheat (26% of the total acreage), winter barley (14% of the total acreage) and rapeseed (17% of the total acreage).[46] To avoid corner solutions in the model, we select farms with these three outputs, which yields a sample of 771 observations and represents more than 80% of the initial farm sample.

Similar to several cited studies, we measure crop diversity $g(\mathbf{S}_t)$ using the Shannon index,[47] i.e., an entropy measure based on land shares. This indicator corrects for species abundance and sample size and is well suited for measuring habitat diversity (Mainwaring, 2001). We compute $g(\mathbf{S}_t)$ as follows:

$$g(\mathbf{S}_t) = -\sum_{n=1}^{N} s_{nt} \ln(s_{nt}) \tag{4.16}$$

where s_{nt} is the share of the land areas devoted to crops n ($n \epsilon [1, N]$). The n indexes refers to the endogenous crops (wheat, winter barley and rapeseed) plus all other land uses considered exogenous in the model (spring barley, peas, sunflower, forage maize, sugar beets, potatoes, permanent grasslands and other crops used as biofuels). The share s_{nt} is defined as S_{nt}/TL_t,

[46] We assume that the other land uses are exogenous. The evolution of permanent grasslands, which represent 28% of the total acreage on average, relies on medium- to long-term strategies. The acreage of fodder crops relies on livestock production decisions and is thus based on different decision-making criteria. Some crops such as sugar beets and potatoes can easily be considered exogenous because they are produced under quotas or contracts.

[47] We also calculate the Simpson Index, as defined by $g(\mathbf{S}_t) = \sum_{n=1}^{N} \left(s_{nt}^2 - (1/N)\right) / \left(1 - (1/N)\right)$. This index increases when crop diversity decreases. The estimation results are consistent with those obtained with the Shannon index, but the estimated parameters are overall less statistically significant. The results are available from the authors upon request.

with S_{nt} being the land devoted to output *n* and TL_t being the total agricultural area of the farm at time *t*. TL_t is the sum of L_t plus all the areas devoted to other exogenous land uses. We consider TL_t as fixed and exogenous. $g(\mathbf{S}_t)$ increases when habitat diversity increases, which reflects the augmentation of crop biodiversity (Burel and Baudry, 2003).

Table 4.1 presents the descriptive statistics of the variables used in the empirical analysis. We have deflated prices based on the national consumption price index. In addition, we use regional input price indexes from the French Department of Agriculture and monthly climatic variables at the municipality level obtained from the *Météo France* database.[48] To account for soil heterogeneity, we use a soil condition index at the municipal level obtained from the *Chambre d'Agriculture de Lorraine* (Hance, 2007).

Table 4.1. Descriptive statistics (N=771)

	Mean	Median	Q1	Q3	Min	Max
Wheat yield (100 kg/Ha)	72.22	72.50	67.02	78.39	38.95	106.96
Winter barley yield (100 kg/Ha)	65.33	66.10	58.42	72.79	33.27	89.24
Rapeseed yield (100 kg/Ha)	33.95	34.19	29.91	38.38	7.96	49.30
Wheat price (€/100 kg)	16.15	15.95	13.03	18.51	3.82	28.32
Winter barley price (€/100 kg)	14.20	14.14	11.10	16.69	7.58	30.82
Rapeseed price (€/100 kg)	33.62	32.74	29.00	37.94	19.96	57.78
Fertilizer on wheat (constant €/Ha)	126.72	119.97	108.76	136.55	3.80	210.15
Fertilizer on barley (constant €/Ha)	110.20	103.38	95.03	118.19	3.15	211.05
Fertilizer on rapeseed (constant €/Ha)	125.72	119.46	107.62	136.47	3.54	247.84
Pesticides on wheat (constant €/Ha)	162.20	160.07	132.94	186.06	44.43	326.58
Pesticides on barley (constant €/Ha)	154.86	153.11	124.65	181.54	41.28	357.65
Pesticides on rapeseed (constant €/Ha)	217.65	214.93	183.62	249.87	63.24	423.47
Fertilizer price index	1.13	1.03	1.00	1.34	0.91	1.51
Pesticides price index	0.98	0.97	0.94	1.00	0.94	1.01
Wheat area (Ha)	53.04	46.47	32.24	68.49	9.19	169.42
Winter barley area (Ha)	28.47	24.50	16.35	37.56	4.46	94.11
Rapeseed area (Ha)	35.33	31.47	19.66	45.73	0.77	123.59
Total area (Ha)	206.87	191.76	143.34	252.40	67.43	552.41
Biodiversity index	1.53	1.53	1.41	1.65	0.95	1.93

4.3.2 Empirical model and econometric strategies

We explain supply, input application and acreage choices for three outputs: soft wheat, winter barley and rapeseed. We consider two variable inputs: fertilizers and pesticides. The specification of our model requires assumptions about functional forms for the production functions and the adjustment cost function. We use the same forms as those employed by

[48] We only use climatic variables that are likely to impact crop production, i.e., average rainfall, temperature, solar radiation and number of frost days. We use these data to consider biological cycles of vegetation and pests, i.e., from February to July for crop yields and from April to June for variable input application.

Carpentier and Letort (2012) and Femenia and Letort (2016). For each output k, we use a quadratic production function:

$$F_{kt}(\mathbf{x}_t, B_t; \mathbf{z}_t) = \alpha_{kt}(B_t; \mathbf{z}_t) - \sum_{i=1}^{I}\sum_{j=1}^{I}\gamma_{ijk}\left(\mu_{ikt}(B_t; \mathbf{z}_t) - x_{ikt}\right)\left(\mu_{jkt}(B_t; \mathbf{z}_t) - x_{jkt}\right) \tag{4.17}$$

The advantage of this functional form is the simple interpretation of its parameters. Parameter α_k represents the maximum yield of output k, and the vector of parameters $\boldsymbol{\mu}_{kt} = \left(\mu_{1kt}, \mu_{2kt}\right)$ corresponds to the required level of fertilizers and pesticides to reach the maximum yield of crop k. These parameters are defined as functions of the productive capacity of biodiversity B_t and some pedo-climatic characteristics $\mathbf{z}_t$ such that:[49]

$$\alpha_{kt}(B_t; \mathbf{z}_t) = \alpha_{0k} + \alpha_{1k}B_t + \boldsymbol{\alpha}_{2k}\mathbf{z}_t \tag{4.18}$$

$$\boldsymbol{\mu}_{kt}(B_t; \mathbf{z}_t) = \boldsymbol{\mu}_{0k} + \boldsymbol{\mu}_{1k}B_t + \boldsymbol{\mu}_{2k}\mathbf{z}_t \tag{4.19}$$

where the parameter α_{1k} is the productivity of crop biodiversity on output k, and the parameter $\boldsymbol{\mu}_{1k}$ is the vector of the input savings on output k due to crop biodiversity. All these parameters are estimated. In particular, crop biodiversity affects production in several ways, namely, sampling, complementarity and facilitation effects (Hooper et al., 2005). The sampling effect implies that the likelihood of the presence of species with a large impact on ecosystem performance increases with crop biodiversity. The complementarity effect refers to the more efficient allocation of resources over time between species that need resources in different periods. The facilitation effect refers to the positive interactions among species that benefit from them. The complementarity and facilitation effects lead to the so-called overyielding effect, i.e., the additional yield of a species when grown with other species compared to its yield in a monoculture. These effects can also lead to marketed input savings if the associated ecological processes are substitute with chemical inputs (Hennessy, 2006). The matrix $\boldsymbol{\Gamma}_k \equiv [\gamma_{ijk}]$ determines the curvature of the function. A positive definite matrix guarantees the concavity of the production function.

The adjustment cost function is approximated using the following quadratic form:

[49] $\mathbf{z}_t$ could also depend on other variables, such as capital and labor (which are not included here).

$$H(\mathbf{S}_t) = \varphi + \sum_{k=1}^{K} \varphi_{0k} S_{kt} + 0.5 \sum_{k=1}^{K} \sum_{m=1}^{K} \varphi_{km} S_{kt} S_{mt} \tag{4.20}$$

where φ, φ_{0k} and φ_{km} are parameters to be estimated. The parameter φ_{0k} depends on the farm characteristics, such as capital, machinery and labour endowment, and the matrix $\mathbf{J}_k \equiv [\varphi_{km}]$ is symmetric. The adjustment cost function corresponds to the cost associated with the reorganization of the farms' fixed inputs.

Following Lucas' critique and similar to Gardebroek (2004), we assume rational price expectations for input and output prices in $t+1$,[50] i.e., that farmers know the underlying formation price mechanisms. The assumption of rational expectations allows for the replacement of the unobserved expected prices in t+1 with their realized counterparts and the addition of an expectation error term $\boldsymbol{\varepsilon}_{t+1}$. We thus write $E(\mathbf{p}_{t+1}) = \mathbf{p}_{t+1} + \boldsymbol{\varepsilon}_{t+1}$ and $E(\mathbf{w}_{t+1}) = \mathbf{w}_{t+1} + \boldsymbol{\upsilon}_{t+1}$ and assume that $E(\boldsymbol{\varepsilon}_{t+1}) = 0$ and $E(\boldsymbol{\upsilon}_{t+1}) = 0$. We also assume that $\boldsymbol{\varepsilon}_{t+1}$ and $\boldsymbol{\upsilon}_{t+1}$ are uncorrelated with any information in *t*. The properties of the error terms suggest that farmers anticipate the realized prices in each period on average.

Solving the farmer's optimization problem leads to $(K \times I)$ input demand and K output supply equations in matrix notation as follows:

$$\mathbf{x}_{kt} = \boldsymbol{\mu}_{0k} + \boldsymbol{\mu}_{1k} B_t + \boldsymbol{\mu}_{2k} \mathbf{z}_t - p_{kt}^{-1} \boldsymbol{\Gamma}_k^{-1} \mathbf{w}_t + v_{kt}^{x} \tag{4.21}$$

$$y_k = \alpha_{0k} + \alpha_{1k} B_t + \boldsymbol{\alpha}_{2k} \mathbf{z}_t - p_{kt}^{-2} \mathbf{w}_t' \boldsymbol{\Gamma}_k^{-1} \mathbf{w}_t + v_{kt}^{y} \tag{4.22}$$

Additionally, $(K-1)$ first-order conditions for acreage choices can be established assuming an interior solution. These first-order conditions include the binding land constraint (with *K* the reference crop) as follows (see appendix 4.B. for the details of the derivation):

[50] Alternative forms of price expectation do not change the signs of the parameter but modify the amplitude of the effects.

$$
\begin{aligned}
&\left(\pi_{kt}-\pi_{Kt}\right)-\left(\varphi_{0k}-\varphi_{0K}\right)-\left(\varphi_{kK}L_t-\varphi_{KK}L_t\right)-\sum_{m=1}^{K-1}S_{mt}\left(\varphi_{km}-\varphi_{kK}-\varphi_{Km}+\varphi_{KK}\right)\\
&-\left(\ln s_{kt}-\ln s_{Kt}\right)\left[\sum_{j=1}^{K-1}s_{jt}\left(p_{jt}\alpha_{1j}-p_{Kt}\alpha_{1K}-\mathbf{w}_t\boldsymbol{\mu}_{1j}+\mathbf{w}_t\boldsymbol{\mu}_{1K}\right)+l_t\left(p_{Kt}\alpha_{1K}-\mathbf{w}_t\boldsymbol{\mu}_{1K}\right)\right]\\
&-\frac{(1-\delta)}{(1+r)}\left(\ln s_{kt+1}-\ln s_{Kt+1}\right)\left[\begin{array}{c}\sum_{j=1}^{K-1}s_{jt+1}\left(p_{jt+1}\alpha_{1j}-p_{Kt+1}\alpha_{1K}-\mathbf{w}_{t+1}\boldsymbol{\mu}_{1j}+\mathbf{w}_{t+1}\boldsymbol{\mu}_{1K}\right)\\ +l_{t+1}\left(p_{Kt+1}\alpha_{1K}-\mathbf{w}_{t+1}\boldsymbol{\mu}_{1K}\right)\end{array}\right]+\nu_{kt}^{s}=0
\end{aligned}
\tag{4.23}
$$

where ν_{kt}^{x}, ν_{kt}^{y} and ν_{kt}^{s} are random terms accounting for unobservable heterogeneity among farmers and stochastic events that can impact production. Based on Oude Lansink and Stefanou (2001), we fix *r* at 0.04. The economic model composed of equations (4.21), (4.22) and (4.23) fully explains farmers' short-term production decisions. For output *k*, the marginal costs (the derivation of the adjustment cost function) of area *k* equal its marginal benefits (the gross margin plus the current and future marginal benefits due to the modification of productive capacity of biodiversity). Production decision equations and Euler equations are typically estimated using the Generalized Method of Moments (GMM, see Hansen and Sargent, 1980). We thus estimate equations (4.21), (4.22) and (4.23) with the GMM using SAS software. Note that some parameters are common to several equations and that equation (4.23) integrates the binding land constraint (4.6). Rapeseed is chosen as the reference crop, and thus, first-order conditions for acreage are estimated for wheat and barley with respect to rapeseed. A consequence is that the parameters φ_{0k}, φ_{0K}, φ_{km}, φ_{KK}, φ_{kK} and φ_{Km} can not be identified separately.

Our model has the advantage of being structural, meaning that we explicitly explain all the production decisions. This feature allows us to address the standard endogeneity problem between production decisions and acreage choices, defining explicitly the structure of the underlying endogeneity.[51] The single issue regarding endogeneity concerns the crop diversity index calculated from acreage areas and present in the output and input equations. To address this problem, we proceed in two steps. In the first step, we regress the acreages of wheat, barley and rapeseed based on all exogenous explanatory variables. We recalculate the diversity index using the predicted acreages of these three crops, still considering the other crops to be exogenous. In the second step, we estimate the complete model with the GMM technique using

[51] Input uses and output productions are generally considered endogenous in acreage equations because of the unobserved heterogeneity of farms, which may affect both production decisions and acreage choices. In our model, all production decisions are explicitly explained, meaning that acreage allocations depend only on the deterministic part of the production process.

the predicted diversity index as an instrument in the equations of the output supply and variable input demand. The other instrumental variables, as defined by the equation, correspond to the exogenous explanatory variables.

Our empirical model has two main potential limitations. First, we do not consider the possibility of corner solutions. All farms produce the three outputs considered in the application. In addition to the standard potential problem of selection bias, this assumption limits the results concerning crop biodiversity. Indeed, the diversity index varies according to the number of crops produced and the uniform repartition of crops over the total area. Given that the number of crops is fixed and cannot change over time, the variation in the biodiversity index is only due to a change in the allocation of land between crops. Second, the crop diversity index is not simultaneously estimated with the production and acreage decisions. We are not able to express the crop diversity index as a function of acreage predicted by the complete model because the model is composed of the first-order conditions for acreage and not the analytical solution of acreage choices.

4.4 Results and discussion

4.4.1 GMM estimation of the structural model

The estimation results are presented in Table 4.2. The R^2 criteria are rather low for the yield and input demand equations. This issue has been highlighted by Carpentier and Letort (2012) and reflects heterogeneity among farmers' production conditions. The term α_k corresponds to the potential yield value for crop *k*. The terms μ_{1k} and μ_{2k} represent the quantities of fertilizer (*i=1*) and pesticide (*i=2*) required to achieve the potential yield of crop *k*. A linear combination of control variables is introduced in these terms, and the parameters α_{0k}, μ_{01k} and μ_{02k} correspond to their average values. Due to space limitations, the estimated parameters of these control variables are reported in appendix 4.C. The parameters α_{1k}, μ_{11k} and μ_{12k} are the parameters associated with the crop diversity indicator for additional yields, for fertilizer savings and pesticide savings of output *k*, respectively.

Almost all estimated parameters are significantly different from zero at the 0.05 level. The parameter estimates satisfy the restrictions imposed by the concavity of crop production functions (γ_{1k} and γ_{2k} are positive, and $\gamma_{1k}\gamma_{2k} - \gamma_{12k}^2 > 0$ for the three crops). Similar to Femenia and Letort (2016), we find that fertilizers and pesticides are substitute inputs (the γ_{12k}

is negative for the three crops). The average potential yield value α_k, expressed in quintals per hectare, corresponds to the average value observed in the region. This value is 72.8 quintal per hectare for wheat,[52] 65.8 for winter barley and 34.5 for rapeseed (see Table 4.1). The estimated values of $\boldsymbol{\mu}_k$ reflect the fact that cropping rapeseed requires larger quantities of fertilizers and pesticides compared with barley and wheat. These results are consistent with agronomic considerations and other results obtained from French data (Carpentier and Letort 2012, Femenia and Letort 2016).

Table 4.2. Results of GMM estimation (N=771)

	Wheat	Winter barley	Rapeseed
Yield supply			
Average potential yield			
Average value α_{0k}	70.54 ***	60.86 ***	34.14 ***
	(4.46)	(5.29)	(2.90)
Crop biodiversity index α_{1k}	1.49 °	3.20 **	0.21
	(0.88)	(1.15)	(0.34)
Curvature parameters			
γ_{1k}	833.58 ***	525.48 ***	1947.04 ***
	(72.97)	(63.93)	(221.90)
γ_{2k}	1065.45 ***	672.03 ***	2583.69 ***
	(144.00)	(150.20)	(507.70)
γ_{12k}	-884.15 ***	-576.50 ***	-1862.01 ***
	(89.02)	(65.10)	(244.50)
R^2	*0.207*	*0.261*	*0.199*
Fertilizer demand			
Average required use			
Average value μ_{01k}	140.09 ***	116.26 ***	142.01 ***
	(23.17)	(18.98)	(23.54)
Crop biodiversity index μ_{11k}	-6.32 °	-3.87	-3.96
	(3.46)	(3.34)	(4.82)
R^2	*0.673*	*0.602*	*0.574*
Pesticides demand			
Average required use			
Average value μ_{02k}	210.21 ***	176.49 ***	316.76 ***
	(16.93)	(13.32)	(16.80)
Crop biodiversity index μ_{12k}	-29.71 **	-13.86	-56.89 ***
	(9.86)	(10.46)	(11.04)
R^2	*0.062*	*0.052*	*0.090*
Acreage			
$(\varphi_{0k} - \varphi_{0K})$	76.29 °	-392.90 ***	(*Ref*)
	(46.46)	(78.16)	

[52] It corresponds to $\left(\alpha_{01} + \alpha_{11}\bar{B}\right) = \left(70.54 + 1.49 \times 1.53\right) = 72.8$.

$\left(\varphi_{kK}-\varphi_{KK}\right)L_t$	-36.97 *		-40.71 *	(*Ref*)
	(16.62)		(19.52)	
$\left(\varphi_{kk}-\varphi_{kK}-\varphi_{Kk}+\varphi_{KK}\right)$	60.98 *		84.11 *	(*Ref*)
	(29.57)		(40.71)	
$\left(\varphi_{km}-\varphi_{kK}-\varphi_{Km}+\varphi_{KK}\right)$		46.21 *		(*Ref*)
		(20.40)		
Biodiversity dynamics				
δ	0.70 ***			
	(0.13)			

°, *, **, *** significance level at 10%, 5%, 1% and 0.1%. Standard errors in brackets.

With respect to the effects of the productive capacity of biodiversity on the average potential yield and average required use of pesticides, our model provides useful insights. First, we find that crop diversity increases yields of wheat and winter barley ($\alpha_{1k} > 0$). We do not find any significant effect of the productive capacity of biodiversity on the rapeseed yield. To our knowledge, this is the first time that crop diversity has been found to increase winter barley yields. This finding confirms that crop diversity increases cereal yields. However, this finding also stresses the need to carefully interpret the results of empirical applications that determine aggregate crop yields based on crop diversity, as some crops are sensitive to crop diversity, whereas others not.

Second, we find that the productive capacity of biodiversity leads to pesticide savings ($\mu_{12k} < 0$). Di Falco and Chavas (2006) found a beneficial effect of the productive capacity of biodiversity on pesticide application based on the estimation of the variance of cereal yields and concluded that the productive capacity of biodiversity reduces production risk. Here, we extend their results by confirming that the productive capacity of biodiversity is a substitute for pesticides. The impact of the productive capacity of biodiversity on fertilizer application is only significant for wheat (at the 10% statistical level). The estimation of our structural model suggests that farmers manage the productive capacity of biodiversity to increase average yields and reduce variable input applications. The productive capacity of biodiversity increases the gross margins of the three outputs, illustrating that farmers have incentives to diversify their acreage.

All the estimated parameters of the acreage equations are significantly different from 0 at the 5% statistical level. The parameter $\left(\varphi_{0k} - \varphi_{0K}\right)$, which measures the difference in fixed costs between wheat and rapeseed, is positive. This means that wheat incurs more costs for fixed inputs than does rapeseed. We find a negative value for winter barley, meaning that winter barley incurs more costs for fixed inputs than does rapeseed. As the determinant of $\mathbf{J}_k \equiv [\varphi_{km}]$ is positive, the concavity of the profit function is verified. Concerning the parameter sets $\left(\varphi_{km} - \varphi_{kK} - \varphi_{Km} + \varphi_{KK}\right)$, we estimate one per acreage equation (for $k = m$) plus one parameter set that is common between the two acreage equations (for $k \neq m$, see equation 23). If we do not include the impacts of crop biodiversity in the model,[53] the sign of the common estimated

[53] We have estimated the model developed by Femenia and Letort (2016), which relies on implicit cost function but does not include the effects of biodiversity on margins (see equation (4.13)). The results obtained with this model are available from the authors upon request.

parameter set is opposite to the one presented in Table 4.2. In this case, the implicit cost function captures all effects associated with acreage management, i.e., the beneficial effect of crop diversity and the management costs of quasi-fixed inputs. In explicitly considering the productive effect of crop diversity, we have separated the benefits and the costs of diversification. Our results agree with those of Oude Lansink and Stefanou (2001) and Chavas and Di Falco (2012), who observed opposite strengths between diversification and specialization, albeit based on different motives. However, the interpretation of the estimated parameters from our adjustment cost function is subject to limitations because the estimated parameters capture the difference between the true parameters of wheat and barley and those of rapeseed.

Finally, these results provide information regarding the management of the productive effects of crop biodiversity. The parameter δ associated with the dynamic effect of the productive capacity of biodiversity is equal to 0.70 (significantly different from 0 at the 0.1% level). This result reflects two important points.

First, similar to Di Falco and Chavas (2008), the estimation of our model indicates that farmers manage their acreage to benefit from the productive effects of past acreage but that the effects of the productive capacity on crop diversity in past years are lower than those in the current year. We confirm that the inherited portion of the productive capacity of crop biodiversity is low, i.e., that the productive capacity of crop biodiversity is primarily managed through current acreage decisions. This result may surprise agricultural economists. Indeed, the effects of the productive capacity of biodiversity are mainly considered dynamic due to crop rotation. A high value of δ does not mean that farmers do not use crop rotations. Indeed, we do not observe acreage spatial choices. Thus, we have to assume that farmers optimize their crop rotation between two periods. Because δ is less than one, the increase in acreage diversity in one period increases yields and variable input savings in future periods, which can be interpreted as more suitable possibilities for crop rotation.

Second, this result shows that the current levels of the productive capacity of biodiversity do not considerably influence farmers' choices over more than two periods. This result agrees with the research of Di Falco and Chavas (2008) and results of Hennessy (2006). Indeed, 30% of the effect of productive capacity of biodiversity on yields and input applications is from acreage choices in $t-1$, and only 9% is from acreage choices in $t-2$ (see Figures 4.A1 and 4.A2 in Appendix 4.B. for a graphic representation of the dynamic effect of biodiversity with δ being equal to 0.70).[54] Our results are robust to different levels of discount rates,[55] and different forms of price expectations.[56] Some precautions are required for interpretation, as the estimated parameter may capture some preference parameters due to price expectations that are not present in our risk-neutral agent model.

Four empirical limits may affect the estimation of δ. First, the crop diversity indicator does not substantially vary between the two periods and may bias and overvalue the estimation of δ. Second, only the acreage choices of three outputs are estimated. However, the sample is composed of heterogeneous farmers, and some of them present a high degree of specialization for wheat, while others demonstrate a high level of diversification. Accordingly, the existence of a corner solution limits the accuracy of our estimations and impacts the estimation of δ. Third, we estimate a single δ for the three crops, while Hennessy (2006) provided evidences that the dynamics of crop rotations are different between crops. Fourth, we have estimated a single δ for the entire period. Di Falco and Chavas (2008) emphasized that the current productive capacity of biodiversity and the rainfall over past year interact negatively interact in crop production, i.e., the dynamic effect of the productive capacity of biodiversity depends on climatic conditions. Future estimations of our model could integrate these information when estimating δ_t.

Some lessons regarding public policies can be drawn from the model and the results presented here. For example, this paper demonstrates that public policies aiming to reduce a pollutant input through pesticide taxation have a double positive impact on the environment: (i) a direct impact that is associated with input reduction (Femenia and Letort, 2016) and (ii) an indirect impact associated with increased marginal productivity of crop biodiversity. In fact, according to the theoretical model and the results, we obtain $\frac{\partial y_{kt}}{\partial B_t \partial x_{ikt}} = \sum_{j=1}^{I} \gamma_{ijk} \frac{\partial \mu_{jk}}{\partial B_t} < 0$ for each input *i* and each crop *k*. An input reduction leads to an increase in the marginal productivity of crop biodiversity. After implementing the policy, farmers are then encouraged to diversify their crops since the effects of the productive capacity of crop biodiversity on crop margins are higher.

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[54] $B_t = 0.30\, B_{t-1} + g(\mathbf{S}_t)$, and $B_t = (1-0.70)^2 B_{t-2} + (1-0.70)\, g(\mathbf{S}_{t-1}) + g(\mathbf{S}_t)$

[55] δ remains between 0.69 and 0.71.

[56] δ remains between 0.70 and 0.83.

Conversely, public policies that encourage crop diversity as proposed in agro-environmental contracts may allow for a reduction in the utilization of variable inputs. Farmers who adopt some agro-environmental measures (AEMs) by integrating a wide diversity of crops into their rotational cropping receive some payments in compensation for revenue loss. If the total impact of biodiversity on production decisions is not considered, these payments are likely misevaluated, especially in the long term.

4.4.2 Additional results

The previous results were published into the European Review of Agricultural Economics. We present here additional results to illustrate the interest of our model.

4.4.2.1 Elasticities at the intensive margins

The estimated parameters in Table 4.2 may be difficult to interpret as we estimate several parameters of interactions between conventional inputs and the biodiversity indicator. The effect of biodiversity on variable input applications, yields and gross margins may be assessed by computing the corresponding elasticities according to B_t (at the intensive margin) considering for all modelled interactions. However, the estimation of our acreage functions using dynamic first-order conditions prevents estimating acreage elasticities, which is usually one aim of the models with land as an allocable fixed input (Carpentier and Letort, 2014). Our dynamic approach allows estimating the intensive margin elasticities at both the short-term and the long-term. The estimated short-term elasticities for gross margin of output *k* (for example) regarding crop biodiversity are computed as:

$$\varepsilon_{B_t,ST}^{\pi_{kt}} = \frac{\partial \pi_{kt}}{\partial B_t}\frac{B_t}{\pi_{kt}} = \left[\begin{array}{c} \left(\alpha_{1k} + \frac{\gamma_{11k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2}\mu_{11k} + \frac{\gamma_{22k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2}\mu_{12k} - \frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2}\left(\mu_{11k} + \mu_{12k}\right)\right) p_{kt} \\ -\left(\mu_{11k} - \frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2}\mu_{12k}\right) w_{1t} - \left(\mu_{12k} - \frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2}\mu_{11k}\right) w_{2t} \end{array}\right] \frac{B_t}{\pi_{kt}}$$

The elasticity of gross margin of *k* is a function of yields and input demand elasticities.

Following Pesaran and Smith (1995) and Arnberg and Hansen (2012), the estimated long-term elasticities for gross margins regarding crop biodiversity are computed as:

$$
\begin{aligned}
\varepsilon^{\pi_{kt}}_{B_t,LT} &= \frac{1+r}{1-\delta}\varepsilon^{\pi_{kt}}_{B_t,ST} \\
&= \frac{1+r}{1-\delta}\left[\begin{array}{c}\left(\alpha_{1k}+\frac{\gamma_{11k}}{\gamma_{11k}\gamma_{22k}-\gamma_{12k}^{2}}\mu_{11k}+\frac{\gamma_{22k}}{\gamma_{11k}\gamma_{22k}-\gamma_{12k}^{2}}\mu_{12k}-\frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k}-\gamma_{12k}^{2}}\left(\mu_{11k}+\mu_{12k}\right)\right)p_{kt} \\ -\left(\mu_{11k}-\frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k}-\gamma_{12k}^{2}}\mu_{12k}\right)w_{1t}-\left(\mu_{12k}-\frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k}-\gamma_{12k}^{2}}\mu_{11k}\right)w_{2t}\end{array}\right]\frac{B_t}{\pi_{kt}}
\end{aligned}
$$

The long-term elasticities converge because the estimated δ is comprised between 0 and 1. The estimated elasticities are available in Table 4.3.

Table 4.3. Elasticities of yields, variable input demands and gross margins relatively to biodiversity

	Short-term				**Long-term**			
	Mean	**SD.**	**Min**	**Max**	**Mean**	**SD.**	**Min**	**Max**
yield_wheat_biodiversity	0.03	0.01	0.02	0.04	0.05	0.01	0.03	0.06
yield_barley_biodiversity	0.07	0.02	0.04	0.11	0.11	0.02	0.06	0.16
yield_rapeseed_biodiversity	0.01	0.001	0.005	0.02	0.01	0.002	0.007	0.02
pesticides_wheat_iodiversity	-0.28	0.05	-0.49	-0.17	-0.40	0.07	-0.70	-0.24
pesticides_barley_biodiversity	-0.14	0.02	-0.21	-0.08	-0.20	0.03	-0.30	-0.12
pesticides_rapeseed_biodiversity	-0.41	0.07	-0.65	-0.21	-0.58	0.10	-0.93	-0.30
fertilizer_wheat_biodiversity	-0.08	0.02	-0.14	-0.04	-0.11	0.02	-0.20	-0.05
fertilizer_barley_biodiversity	-0.06	0.01	-0.10	-0.03	-0.08	0.02	-0.14	-0.04
fertilizer_rapeseed_biodiversity	-0.05	0.01	-0.09	-0.02	-0.07	0.01	-0.13	-0.03
gross_margins_wheat_biodiversity	0.10	0.08	0.04	2.16	0.14	0.11	0.06	3.09
gross_margins_barley_ biodiversity	0.12	0.03	0.07	0.29	0.18	0.04	0.10	0.41
gross_margins_rapeseed_ biodiversity	0.13	0.03	0.06	0.29	0.18	0.04	0.08	0.41

Except the short and long-term elasticities for wheat gross margins, all elasticities are estimated with a good statistical precision (p-values are lower than 0.05). Elasticities of gross margins are comprised between 0.10 and 0.18, with higher values in the long-term. The higher effects are on pesticide savings, which is conform with previous results. Most part of gross margin elasticities for rapeseed is attributed to this pesticide saving. Even if the parameters of fertilizer savings $\boldsymbol{\mu}_{12}$ are not always significantly different from zero, the estimated elasticities for fertilizer application are significantly different from zero at a statistical level of 1%. This is due to the precise estimations of pesticide savings $\boldsymbol{\mu}_{11}$ and the parameters of substitution between pesticide and fertilizer $\boldsymbol{\gamma}_{12}$. Barley yields are more sensitive to crop biodiversity than the other outputs, explaining that, even if the pesticide savings for barley are lower than the two other outputs, the gross margin elasticities for barley are almost similar to the ones for rapeseed.

Additional yields lead to higher relative benefits than input savings. Overall, our elasticities are comparable with Di Falco et al. (2007) on farmers' revenues.

4.4.2.2 Comparison with other models

In order to highlight the interest of our model (noted hereafter Model 1), we compare our results with two other models (Models 2 and 3). Model 2 is the empirical counterpart of the static acreage model proposed in chapter 2 and whom FOC are interpreted in relation (4.14) in this chapter. Model 2 corresponds to a static management of crop biodiversity. Model 3 is similar to the model estimated in Femenia and Letort (2016), expect that we estimate acreage FOC instead of the acreage transformation proposed in their work. Model 3 corresponds to an absence of crop biodiversity as potential input. Mathematically, Model 2 is composed of three supply functions of type (A2), six input demands of type (B2) and two acreage function of type (C2):

$$\begin{cases} \text{(A2): } y_k = \alpha_k\left(B_t, \mathbf{z_t}\right) - p_{kt}^{-2}\,\mathbf{w}_t\,'\boldsymbol{\Gamma}_k^{-1}\mathbf{w}_t + v_{kt}^{y} \\ \text{(B2): } \mathbf{x}_{kt} = \boldsymbol{\mu}_k\left(B_t, \mathbf{z_t}\right) - p_{kt}^{-1}\,\boldsymbol{\Gamma}_k^{-1}\mathbf{w}_t + v_{kt}^{x} \\ \text{(C2): } \left(\pi_{kt} - \pi_{Kt}\right) - \sum_{j=1}^{K-1} s_{jt}\left(\ln s_{kt} - \ln s_{Kt}\right)\left(p_{jt+1}\alpha_{1j} - p_{Kt+1}\alpha_{1K} - \mathbf{w}_{t+1}\boldsymbol{\mu}_{1j} + \mathbf{w}_{t+1}\boldsymbol{\mu}_{1K} + l_t\left(p_{Kt}\alpha_{1K} - \mathbf{w}_t\boldsymbol{\mu}_{1K}\right)\right) \\ \qquad\qquad - \left(\left(\varphi_{0k} - \varphi_{0K}\right) + \left(\varphi_{kK}L_t - \varphi_{KK}L_t\right) + \sum_{m=1}^{K-1} S_{mt}\left(\varphi_{km} - \varphi_{kK} - \varphi_{Km} + \varphi_{KK}\right)\right) + v_{kt}^{s} = 0 \end{cases}$$

The functions (A2) and (B2) are similar to the supply and demand functions estimated in Model 1.

Mathematically, Model 3 is composed of three supply functions of type (A3), six input demands of type (B3) and two acreage function of type (C3):

$$\begin{cases} \text{(A3): } y_k = \alpha_k\left(\mathbf{z_t}\right) - p_{kt}^{-2}\,\mathbf{w}_t\,'\boldsymbol{\Gamma}_k^{-1}\mathbf{w}_t + v_{kt}^{y} \\ \text{(B3): } \mathbf{x}_{kt} = \boldsymbol{\mu}_k\left(\mathbf{z_t}\right) - p_{kt}^{-1}\,\boldsymbol{\Gamma}_k^{-1}\mathbf{w}_t + v_{kt}^{x} \\ \text{(C3): } \left(\pi_{kt} - \pi_{Kt}\right) - \left(\left(\varphi_{0k} - \varphi_{0K}\right) + \left(\varphi_{kK}L_t - \varphi_{KK}L_t\right) + \sum_{m=1}^{K-1} S_{mt}\left(\varphi_{km} - \varphi_{kK} - \varphi_{Km} + \varphi_{KK}\right)\right) + v_{kt}^{s} = 0 \end{cases}$$

The functions (A3) and (B3) are similar to the supply and demand functions estimated in Femenia and Letort (2016). They differ from those of Models 1 and 2 by the absence of crop biodiversity. Functions (C3) are different from the ones estimated in Femenia and Letort (2016), even if we share the same parameters (but not the same functions of parameters).

To illustrate the interest of our method, we simulate the impact of a 100% tax on pesticides. Pesticide taxation has often been advocated in the economic literature as one of the most cost effective policy instruments to reduce the use of pesticide (Femenia and Letort, 2016). However, the taxation could lead to a greater decrease in pesticide use when the farmers have access to a substitute to pesticide. Here, we consider two types of substitutes: fertilizers and crop biodiversity. We compare the impact of pesticide taxation in the three models, considering different degree of biodiversity management by the farmer: from a long-term management in Model 1 (Bareille and Letort, 2018) to the absence of management in Model 3 (Femenia and Letort, 2016). We simulate the impacts of an ad valorem tax on pesticide expenditure on the crop profitability at the intensive margin and on farmers' use of pesticide given the degree of biodiversity management.[57] These information are determinate using the elasticities of pesticide application and gross margins regarding pesticide price. The pesticide application elastic formulas are the same in the three models and are equal to:

$$\varepsilon_{w_{1t}}^{x_{1kt}} = -\frac{\gamma_{11k}}{p_{kt}} \frac{w_{1t}}{x_{1kt}}$$

The different elasticities in the three models depend thus only on the different estimation of γ_{11k}. The reduction of pesticide use after a 00% ad valorem tax on pesticides are equal to 100 times these elasticities. Table 4.4 presents the average results for a 100% tax on pesticide expenditure.

Table 4.4 Simulated impacts of a 100% tax on pesticides — average impacts on pesticide use in the three models (% age change compared with the initial situation)

	Model 1	Model 2	Model 3
pesticides_wheat	-41,99	-43,27	-43,02
pesticides _barley	-32,08	-32,47	-30,07
pesticides _rapeseed	-35,89	-35,71	-32,66

In contrast, the gross margin elasticity formulas for gross margins regarding the gross margins are different in the three models. In Model 1, we have:

[57] Femenia and Letort (2016) discuss the pertinence of simulating an ad valorem tax compared to differentiated taxes based on the toxicity of the products (e.g. debates on the taxation/prohibition of neonicotinoids in France). The information on the differentiated pesticide expenditures and correspondent toxicity indices is not available in our database.

$$\varepsilon_{w_{1t},LT}^{\pi_{kt}} = \frac{w_{1t}}{(1-\delta)\, p_{kt}} \left[-\frac{(\gamma_{11k} w_{1k} + \gamma_{12k} w_{2k})}{y_{kt}} \left(1 + \frac{\gamma_{11k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2} \mu_{11k} - \frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2} \mu_{12k} \right) + \frac{w_{1t}\gamma_{11k}}{x_{1kt}} + \frac{w_{2t}\gamma_{12k}}{x_{2kt}} \right]$$

In Model 2, we have:

$$\varepsilon_{w_{1t},CT}^{\pi_{kt}} = \frac{w_{1t}}{p_{kt}} \left[-\frac{(\gamma_{11k} w_{1k} + \gamma_{12k} w_{2k})}{y_{kt}} \left(1 + \frac{\gamma_{11k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2} \mu_{11k} - \frac{\gamma_{12k}}{\gamma_{11k}\gamma_{22k} - \gamma_{12k}^2} \mu_{12k} \right) + \frac{w_{1t}\gamma_{11k}}{x_{1kt}} + \frac{w_{2t}\gamma_{12k}}{x_{2kt}} \right]$$

In Model 3, we have:

$$\varepsilon_{w_{1t},CT}^{\pi_{kt}} = \frac{w_{1t}}{p_{kt}} \left[-\frac{(\gamma_{11k} w_{1k} + \gamma_{12k} w_{2k})}{y_{kt}} + \frac{w_{1t}\gamma_{11k}}{x_{1kt}} + \frac{w_{2t}\gamma_{12k}}{x_{2kt}} \right]$$

Table 4.5 presents the impacts of pesticide taxation on gross margins using these elasticities.

Table 4.5 Simulated impacts of a 100% tax on pesticides — average impacts at the intensive margin (gross margins) in the three models (% age change compared with the initial situation)

	Model 1	Model 2	Model 3
gross_margin_wheat	-11,99	-12.38	-14.24
gross_margin_barley	-12.69	-13,55	-10.50
gross_margin_rapeseed	-23.73	-24.91	-29.70

Table 4.4 shows that the integration of the management of crop biodiversity increases the effectiveness of the tax on barley and rapeseed, but decreases its effectiveness on wheat (but the estimated parameters deviates by 3% from each other maximum). The difference is more accentuated for rapeseed, with a 10% difference on the reduction of pesticide use between Model 1 and Model 3. Table 4.5 highlights that the impacts of pesticide taxation on profitability are usually lower with a deeper integration of biodiversity management (except for barley). In particular, a 100% tax incurs a loss of rapeseed profitability by only 23.7% in Model 1 (compared to 29.7% in Model 3), i.e. the estimated loss of profitability for rapeseed is overestimated by 25% when ignoring the effect of crop biodiversity as a substitute for pesticide (e.g. Femenia and Letort, 2016; Carpentier and Letort, 2011; Koutchadé et al., 2018). Our results suggest that, when alternative techniques (here biodiversity productive capacity) are taken into account, farmers have more freedom to adapt from an exogenous choc (here the pesticide tax). To sum up, our model provides estimators suggesting that pesticide use on rapeseed would decrease more than usually estimated in case of pesticide taxation scheme, notably because its effect on intensive-margin profitability are lower than the ones usually

estimated. The lower intensive-margin decrease for rapeseed would also impact extensive-margin choices, rapeseed areas should being less reduced than usually estimated.[58]

4.5 Conclusion

Our structural microeconomic model allows for the simultaneous estimation of supply, variable input demands and acreage functions. Inspired by multicrop microeconometric and investment literature, our approach considers (i) the productive effects of crop biodiversity, (ii) the dynamics of the productive capacity of crop biodiversity and (iii) the adjustment costs associated with fixed input management. We find that high levels of crop diversity lead to the augmentation of yields and to input savings. Compared to the research of Femenia and Letort (2016), the introduction of crop biodiversity effects inside gross margins allows the capture of only the acreage management costs inside the implicit cost function. The separation of the benefits and costs of diversification is supported by the results. To our knowledge, this is the first time that the costs and incomes associated with the productive capacity of biodiversity have been simultaneously considered. Previous studies have typically focused on a single dimension of the productive capacity of biodiversity or on a dual restricted profit function, neither of which allows for a full understanding of the economic and ecosystem mechanisms. Hence, the addition of the dynamic framework provides new insights into the intertemporal management of crop biodiversity. Our model allows for a generalization of the management models of the productive capacity of biodiversity that are proposed in the economic literature.

A potential limit of our framework is that it ignores the effects of crop biodiversity on variance yields. Indeed, the literature on crop diversity has stated that crop diversity reduces the probability of low yield realization and, thus, decreases production risk. Crop diversity also decreases market risk, as crop diversity can be considered as a portfolio strategy (Di Falco and Perrings, 2005). In addition to provide more flexibility for the analysis of crop biodiversity productivity, the consideration of the effects of crop biodiversity on variance yields has an impact on risk-averse farmers. Consequently, the presented results definitely underestimate the potential beneficial effects of crop biodiversity on farmers' profit. Additional gains can notably emerge from substitution between financial insurance and crop diversity. To our knowledge, if Baumgärtner (2007) has already theoretically dealt with this issue, no study has ever measured

[58] Note that we do not estimate acreage shares here. However, as acreage choices are usually modelled as gross margins comparisons modulo fixed acreage costs (e.g. Carpentier and Letort, 2012), the rapeseed reduction due to pesticide tax should be lower than previously estimated ones.

such substitution in an empirical study. Regarding the amount of subsided crop insurance in the world (not in France though), such measurement would be a great contribution to the literature and a valuable information for policymakers.

Because we rely on investment literature, our model offers substantial possibilities for extensions; e.g., we can introduce heterogeneous adjustment costs or threshold effects into the biodiversity dynamics. Future studies could also consider several dynamic parameters as well as the impacts of climatic conditions or the heterogeneity of dynamic effects on output yields and input savings. Our model can also provide new insights on the effectiveness of AEMs because it expresses the evolution of acreage diversity management based on market fluctuations. Furthermore, our results may benefit the design of suitable AEMs and could lead to a win-win situation in which both biodiversity and agricultural profitability increase. This need has already been stressed by Omer et al. (2007) in a study based on a stochastic production function with the introduction of a biodiversity indicator. However, an analysis based on a production function is not sufficient for evaluating the relevant incentives (Omer et al., 2007). We contend that our model can provide this type of information because it expresses farmers' responses to economic incentives and the associated effects in crop biodiversity management. We do not address this issue because the analysis of current AEM effectiveness requires the mobilization of special econometric methods to overcome the sample selection bias. However, the approach developed in this paper serves as a good basis for future work in this area.

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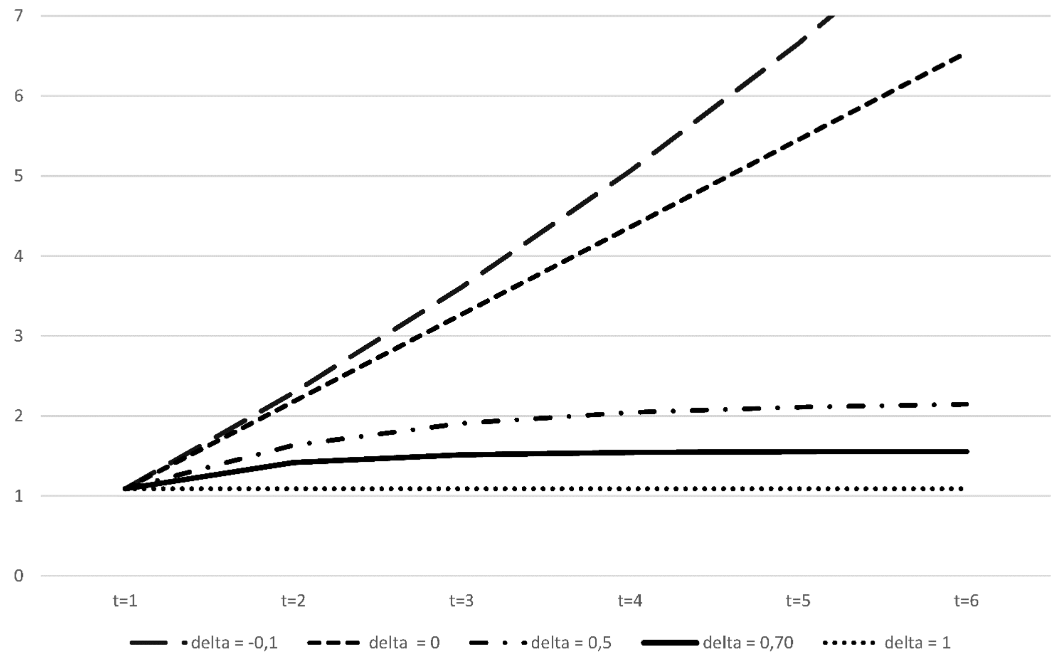
4.7 Appendices

Appendix 4.A. Temporal evolution of the productive capacity of biodiversity according to the potential values of δ .

As explained in the empirical section, we estimate the single parameter δ instead of several δ_t values. Thus, the dynamics equation of crop biodiversity is defined by $B_t = (1-\delta)B_{t-1} + g(\mathbf{S}_t)$.

We assume that $B_0 = 0$ and that farmers cultivate 3 crops (wheat, bailey and rapeseed). We compare two situations. First, the farmer equally allocates his land among these 3 crops (Figure 4.A1). The $g(\mathbf{S}_t)$ term is maximal, illustrating the positive effects of crop diversity on the yield and variable input savings. Second, he equally allocates his land between these 3 crops from $t = 1$ to 3 and decides to cultivate only one crop from $t = 4$ to 6 (Figure 4.A2). The $g(\mathbf{S}_t)$ term changes from its maximal value to its minimal value. In each case, we compare the evolution of the productive capacity of biodiversity B_t according to different values of the δ term. As presented in Table 4.2, the estimated value of the δ term is 0.83. The estimated evolution of B_t is represented by the solid line. The dotted lines correspond to the different potential values of δ (described on page 8).

Figure 4.A1. Evolution of the productive capacity of crop biodiversity



In Figure 4.A1, we observe three different evolutions.[59] First, when $\delta = 1$, B_t remains constant because it only depends on $g(\mathbf{S}_t)$, which remains constant. Second, when $\delta \in]0,1[$, B_t increases, but this increase is less significant over time. Third, when $\delta \le 0$, B_t increases with a constant or increasing slope.

Figure 4.A2. Evolution of the productive capacity of crop biodiversity with a change in acreage in $t = 4$

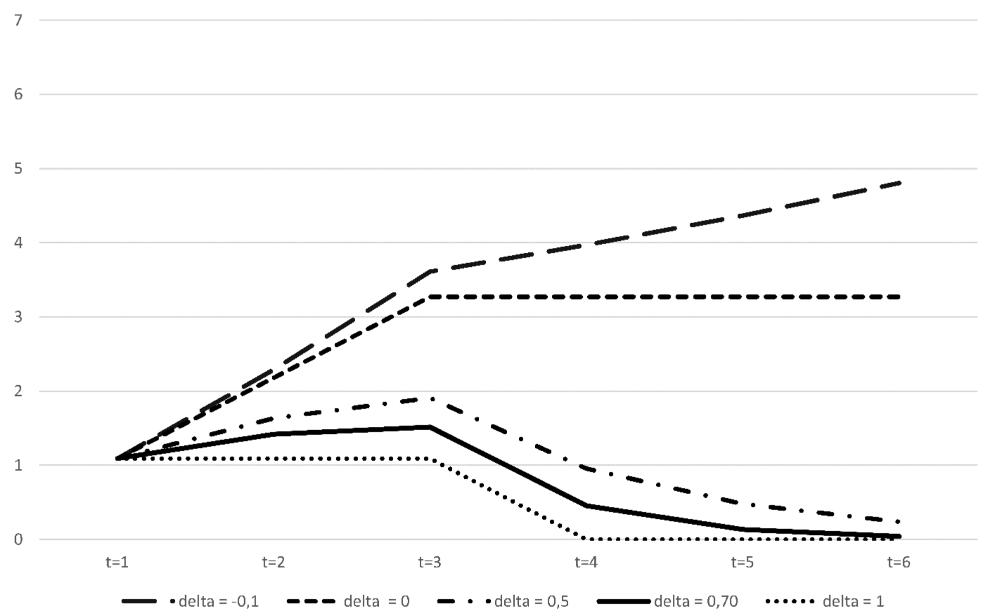


Figure 4.A2 presents the case in which a farmer simplifies his crop rotation by cultivating only one crop in year $t = 4$. From an ecological point of view, this decision has an adverse effect on biodiversity because of the reduction in habitat diversity. However, benefits of past practices may still influence the productive capacity of crop biodiversity B_t. Compared to these ecological considerations, some potential values of δ lead to the inadequate evolution of B_t. When $\delta \le 0$, the benefits of past acreages never decrease and can further increase in spite of

[59] We do not consider the case in which $\delta > 1$ because it leads to an uninterpretable evolution. For example, if $\delta = 2$, the productive capacity of biodiversity ranges between 0 and 1 from year to year.

the monoculture. When $\delta = 1$, the benefits of past acreages are null, and B_t is thus null. The more realistic situations correspond to the cases in which $\delta \in]0,1[$. B_t decreases at a variable rate, depending on the value of δ. The acreage decisions of the past year have a longer lasting effect as δ approaches 0.

Appendix 4.B. First-order conditions for acreage choices with integration of land constraint

The Lagrangian function associated to our maximization problem is defined by:

$$L\left(\mathbf{S}_t, \lambda\right) = \sum_{t=1}^{T} \left(\frac{1}{1+r}\right)^{t-1} \Pi_t(\mathbf{x}_t, B_t, \mathbf{S}_t; \mathbf{z}_t) - \lambda\left(\sum_{k=1}^{K} S_{kt} - L_t\right) \tag{4A.1}$$

with λ being the Lagrangian multiplier associated with the land constraint. Considering an optimization problem with two periods, it leads to the following first-order conditions for crop $k\ \left(k \neq K\right)$ and for the reference crop K:

$$\frac{\partial L}{\partial S_{kt}} = \pi_{kt} + \sum_{j=1}^{K} S_{jt} \frac{\partial \pi_{jt}}{\partial B_t} \frac{\partial B_t}{\partial S_{kt}} - \frac{\partial H}{\partial S_{kt}} + \frac{\left(1-\delta\right)}{\left(1+r\right)} E\left[\sum_{j=1}^{K} S_{jt+1} \frac{\partial \pi_{jt+1}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{kt+1}}\right] = \lambda$$

(4A.2)

$$\frac{\partial L}{\partial S_{Kt}} = \pi_{Kt} + \sum_{j=1}^{K} S_{jt} \frac{\partial \pi_{jt}}{\partial B_t} \frac{\partial B_t}{\partial S_{Kt}} - \frac{\partial H}{\partial S_{Kt}} + \frac{\left(1-\delta\right)}{\left(1+r\right)} E\left[\sum_{j=1}^{K} S_{jt+1} \frac{\partial \pi_{jt+1}}{\partial B_{t+1}} \frac{\partial B_{t+1}}{\partial S_{Kt+1}}\right] = \lambda \tag{4A.3}$$

$$S_{Kt} = L_t - \sum_{g=1}^{K-1} S_{gt} \tag{4A.4}$$

Given that $\frac{\partial \pi_{kt}}{\partial B_t} = p_{kt}\alpha_{1k} - \mathbf{w}_t \boldsymbol{\mu}_{1k}$, $\frac{\partial B_t}{\partial S_{kt}} = -TL_t^{-1}\left(\ln s_{kt} + 1\right)$ and $\frac{\partial H}{\partial S_{kt}} = \varphi_{0k} + \sum_{m=1}^{K} \varphi_{km} S_{mt}$, we obtain the following system of first-order conditions:

$$\begin{aligned}\frac{\partial L}{\partial S_{kt}} = \pi_{kt} - \sum_{j=1}^{K} s_{jt}\left(\ln s_{kt} + 1\right)\left(p_{jt}\alpha_{1j} - \mathbf{w}_t \boldsymbol{\mu}_{1j}\right) - \left(\varphi_{0k} + \sum_{m=1}^{K} \varphi_{km} S_{mt}\right) \\ - \frac{\left(1-\delta\right)}{\left(1+r\right)}\left[\sum_{j=1}^{K} s_{jt+1}\left(\ln s_{kt+1} + 1\right)\left(p_{jt+1}\alpha_{1j} - \mathbf{w}_{t+1} \boldsymbol{\mu}_{1j}\right)\right] = \lambda\end{aligned} \tag{4A.5}$$

$$\begin{aligned}\frac{\partial L}{\partial S_{Kt}} = \pi_{Kt} - \sum_{j=1}^{K} s_{jt}\left(\ln s_{Kt} + 1\right)\left(p_{jt}\alpha_{1j} - \mathbf{w}_t \boldsymbol{\mu}_{1j}\right) - \left(\varphi_{0K} + \sum_{m=1}^{K} \varphi_{Km} S_{mt}\right) \\ - \frac{\left(1-\delta\right)}{\left(1+r\right)}\left[\sum_{j=1}^{K} s_{jt+1}\left(\ln s_{Kt+1} + 1\right)\left(p_{jt+1}\alpha_{1j} - \mathbf{w}_{t+1} \boldsymbol{\mu}_{1j}\right)\right] = \lambda\end{aligned} \tag{4A.6}$$

$$S_{Kt} = L_t - \sum_{g=1}^{K-1} S_{gt} \tag{4A.7}$$

Equation (4A.5) minus equation (4A.6) leads to:

$$\left(\pi_{kt}-\pi_{Kt}\right)-\sum_{j=1}^{K}s_{jt}\left(\ln s_{kt}-\ln s_{Kt}\right)\left(p_{jt}\alpha_{1j}-\mathbf{w}_t\boldsymbol{\mu}_{1j}\right)-\left(\varphi_{0k}-\varphi_{0K}+\sum_{m=1}^{K}\left(\varphi_{km}-\varphi_{Km}\right)S_{mt}\right)$$
$$-\frac{\left(1-\delta\right)}{\left(1+r\right)}\left[\sum_{j=1}^{K}s_{jt+1}\left(\ln s_{kt+1}-\ln s_{Kt+1}\right)\left(p_{jt+1}\alpha_{1j}-\mathbf{w}_{t+1}\boldsymbol{\mu}_{1j}\right)\right]=0$$

(4A.8)

The inclusion of (4A.7) in (4A.8) leads to the following first-order condition for acreage choice of crop $k\ \left(k\neq K\right)$:

$$\begin{aligned}&\left(\pi_{kt}-\pi_{Kt}\right)-\left(\varphi_{0k}-\varphi_{0K}\right)-\sum_{m=1}^{K-1}S_{mt}\left(\varphi_{km}-\varphi_{kK}-\varphi_{Km}+\varphi_{KK}\right)-\left(\varphi_{kK}L_t-\varphi_{KK}L_t\right)\\&-\left(\ln s_{kt}-\ln s_{Kt}\right)\left[\sum_{j=1}^{K-1}s_{jt}\left(p_{jt}\alpha_{1j}-p_{Kt}\alpha_{1K}-\mathbf{w}_t\boldsymbol{\mu}_{1j}+\mathbf{w}_t\boldsymbol{\mu}_{1K}\right)+l_t\left(p_{Kt}\alpha_{1K}-\mathbf{w}_t\boldsymbol{\mu}_{1K}\right)\right]\\&-\frac{\left(1-\delta\right)}{\left(1+r\right)}\left(\ln s_{kt+1}-\ln s_{Kt+1}\right)\left[\begin{array}{c}\sum_{j=1}^{K-1}s_{jt+1}\left(p_{jt+1}\alpha_{1j}-p_{Kt+1}\alpha_{1K}-\mathbf{w}_{t+1}\boldsymbol{\mu}_{1j}+\mathbf{w}_{t+1}\boldsymbol{\mu}_{1K}\right)\\+l_{t+1}\left(p_{Kt+1}\alpha_{1K}-\mathbf{w}_{t+1}\boldsymbol{\mu}_{1K}\right)\end{array}\right]=0\end{aligned}\qquad(4A.9)$$

With $l_t=L_t/TL_t$ being the total acreage share of all endogenous crops on total agricultural area.

Appendix 4.C. results of GMM estimation for all estimated parameters (N=771)

	Wheat	Winter barley	Rapeseed
Yield supply			
Average potential yield			
Constant	56.38 ***	43.94 **	30.16 **
	(15.79)	(16.60)	(10.69)
Rain in March	0.04	0.05	0.03
	(0.04)	(0.04)	(0.03)
Rain in April	0.04	0.01	-0.03
	(0.05)	(0.06)	(0.03)
Rain in May	-0.02	-0.07 *	-0.09 ***
	(0.03)	(0.03)	(0.02)
Rain in June	-0.10 **	-0.12 ***	-0.07 **
	(0.03)	(0.04)	(0.02)
Frost in May	-3.07 °	-0.56	-0.25
	(1.83)	(1.93)	(1.44)
EVT in May	0.19	-0.14	-0.03
	(0.17)	(0.20)	(0.12)
EVT in June	0.35 ***	0.45 ***	0.20 ***
	(0.09)	(0.11)	(0.06)
EVT in July	-0.09	-0.04	-0.09
	(0.08)	(0.09)	(0.06)
Temperature in February	3.79 **	3.96 **	0.85
	(1.19)	(1.42)	(0.88)
Temperature in Mars	1.33 *	-0.13	0.95 *
	(0.56)	(0.60)	(0.40)
Temperature in April	2.92 ***	1.70 *	1.96 ***
	(0.68)	(0.67)	(0.41)
Temperature in May	0.22	0.86	0.67
	(0.96)	(1.07)	(0.71)
Temperature in June	-6.67 ***	-3.73 *	-2.45 *
	(1.44)	(1.49)	(1.04)
Soil index	22.41 ***	10.61 °	10.52 *
	(6.10)	(6.01)	(4.14)
Crop biodiversity index	1.49 °	3.20 **	0.21
	(0.88)	(1.15)	(0.34)
Curvature parameters			
γ_{1k}	833.58 ***	525.48 ***	1947.04 ***
	(72.97)	(63.93)	(221.90)
γ_{2k}	1065.45 ***	672.03 ***	2583.69 ***
	(144.00)	(150.20)	(507.70)
γ_{12k}	-884.15 ***	-576.50 ***	-1862.01 ***
	(89.02)	(65.10)	(244.50)
R^2	*0.207*	*0.261*	*0.199*
Fertilizer demand			
Average required use			
Constant	115.85 ***	52.47 °	125.47 ***
	(28.46)	(30.13)	(38.15)
Rain in April	0.99 ***	0.84 ***	0.97 ***
	(0.05)	(0.05)	(0.06)
Rain in May	0.13 ***	0.08 ***	0.10 **
	(0.02)	(0.02)	(0.03)

EVT in June	0.66 ***	0.74 ***	0.65 ***
	(0.15)	(0.13)	(0.18)
Temperature in April	10.05 ***	9.15 ***	10.13 ***
	(0.59)	(0.51)	(0.75)
Temperature in June	-11.77 ***	-8.32 ***	-11.83 ***
	(1.92)	(2.14)	(2.29)
Soil index	-16.97	-24.98 °	-19.09
	(13.38)	(13.11)	(15.63)
Crop biodiversity index	-6.32 °	-3.87	-3.96
	(3.46)	(3.34)	(4.81)
R^2	*0.673*	*0.602*	*0.574*
Pesticides demand			
Average required use			
Constant	-69.51	-151.06 °	67.91
	(77.69)	(84.06)	(96.22)
Rain in April	1.12 ***	0.94 ***	0.85 ***
	(0.16)	(0.19)	(0.21)
Rain in May	0.25 **	0.14	-0.05
	(0.09)	(0.09)	(0.10)
Rain in June	-0.22	0.19	-0.24
	(0.14)	(0.15)	(0.18)
EVT in April	0.12	-0.12	-0.17
	(0.51)	(0.53)	(0.56)
EVT in May	-0.11	0.33	0.43
	(0.34)	(0.35)	(0.42)
EVT in June	1.44 ***	1.48 ***	0.60
	(0.30)	(0.29)	(0.40)
Temperature in April	13.05 ***	10.11 ***	12.87 ***
	(2.23)	(2.68)	(2.70)
Temperature in May	-2.29	-6.46	-12.94 **
	(3.70)	(4.22)	(4.67)
Temperature in June	2.75	6.25	15.29 *
	(4.70)	(5.66)	(6.55)
Soil index	-98.56 **	-61.42 *	-69.20
	(33.11)	(30.50)	(43.16)
Crop biodiversity index	-29.71 **	-13.86	-81.76 °
	(9.85)	(10.47)	(44.97)
R^2	*0.062*	*0.052*	*0.090*
Acreage			
$\left(\varphi_{0k}-\varphi_{0K}\right)$	76.29 °	-392.90 ***	(*Ref*)
	(46.46)	(78.16)	
$\left(\varphi_{kK}-\varphi_{KK}\right)L_t$	-36.97 *	-40.71 *	(*Ref*)
	(16.62)	(19.52)	
$\left(\varphi_{km}-\varphi_{kK}-\varphi_{Km}+\varphi_{KK}\right)$	60.98 *	84.11 *	(*Ref*)
	(29.57)	(40.71)	
$\left(\varphi_{km}-\varphi_{kK}-\varphi_{Km}+\varphi_{KK}\right)$	46.21 *		(*Ref*)
	(20.40)		
Biodiversity dynamics			
δ	0.70 ***		
	(0.13)		

°, *, **, *** significance level 10%, 5%, 1% and 0.1%. Standard errors in brackets.

CHAPTER 5. PRODUCTIVE ECOSYSTEM SERVICES AND COLLECTIVE MANAGEMENT: LESSONS FROM A REALISTIC LANDSCAPE MODEL [60]

Up to this chapter, we have examined the properties and management of biodiversity productive capacity at the farm scale. However, farm territories are fragmented over space, and the ecological functionalities provided by biodiversity depend on the composition and configuration of land use at the landscape scale; i.e., the provision of input ES also depends on neighbouring farmers' land-use choices. This implies that unless neighbouring farmers cooperate, they generate productive externalities to each other. The aim of this chapter is to analyse the benefits of collective management of input ES using simulations from an extension of the agro-ecological agent-based model developed by Martel et al. (2017), where we explicitly introduce farmers' microeconomic and strategic behaviour. We examine the collective and individual gains arising from no management to coordinated management of carabid beetles, carabid beetles being considered natural pest predators. We contribute to the literature by considering heterogeneous agents and different initial conditions, as such elements influence the success of coordination (Costello et al., 2017). Such a degree of realism and detail is also required to explore the opportunity for farmers to shape the landscape and manage biodiversity productive capacity. This interest in realism follows the results from the two previous empirical chapters, in which we have considered several biodiversity components and more complex interactions than usually measured in the literature.

[60] This chapter was coauthored with Hugues Boussard (INRA, BAGAP) and Claudine Thenail (INRA, BAGAP).

5.1 Introduction

Occupying 37.5% of world lands, farming is the most land-intensive economic activity, making farmers responsible for a large part of earth's ecosystem. On the one hand, there has been increasing evidence about the way farmers affect the provision of diverse ecosystem services (ES). At the field scale, ecological functions involved in ES, such as natural pest control, depend on the variety of cropping practices used and farmers' land-use choices (*e.g.*, Seguni et al., 2011). Hypotheses and evidence have been presented regarding the relative influence of landscape structure (composition and configuration), field structure (*e.g.*, with or without hedgerows) and field management on key ecological functions (Tscharntke et al., 2012). For instance, there is a consensus about the negative effect of intensive farming practices in simplified landscapes on the biodiversity of pest predators (Chaplin-Kramer et al., 2011). The response of carabid species (generalist pest-predatory insects) richness to landscape heterogeneity has been notably explained by the quantity of interfaces in landscape mosaics, which controls resource availability for the carabids (Duflot et al., 2016).

On the other hand, these ES may influence the utility of diverse agents, including that of farmers, for instance, through agricultural productivity and profitability (Zhang et al., 2007). We refer to these services as productive ES, considering that they may be inputs into the production of other goods that are themselves marketed, in particular, agricultural goods (Barbier, 2007). Several works aimed at valuing productive ES at the field scale, *e.g.*, by considering yield gain or reductions in pesticide costs due to biological pest control (*e.g.*, Brainard et al., 2016). There have also been attempts to value productive ES at the farm scale, *e.g.*, by considering the share of favorable land use on farms and by calculating an average yield loss/gain from a representative sampling of farms (Klemick, 2011; Letourneau and Goldstein, 2001). Other studies extend beyond the valuation of productive ES to examine the management of productive ES by farmers at the farm scale. Relying on crop allocation choice models at the farm scale, these works demonstrated that farmers manage productive ES to benefit from them, either in terms of additional yields or input savings (Bareille and Letort, 2018; Orazem and Miranowski, 1994). These results suggest that productive ES are impure public goods and that they are not pure externalities; *i.e.*, farmers do internalize them, at least at the field and farm levels.

The knowledge obtained about the impact of the landscape structure on the provision of productive ES has highlighted new issues in terms of the collective agricultural management of productive ES (Zhang et al, 2007). One issue is estimating how and how much farmers' individual land-use choices generate externalities for other farmers sharing the same landscape due to the respective influence of these farmers' choices on mobile ES providers such as beneficial insects. Another related issue is estimating the potential benefits of the coordinated management of productive ES at the landscape scale. The analysis of the benefits of the coordinated management of productive ES has recently received the attention of economists, who have responded with either purely theoretical works (Costello et al., 2017; Zavalloni et al., 2018) or empirical works (Atallah et al., 2017, Epanchin-Niell and Wilen, 2014).

Answering such questions would help in the assessment of the impacts of different existing or novel policy instruments on the evolution of collective ES provision and management at the landscape scale. Nevertheless, we argue that to produce such operational outcomes, the considerable heterogeneity of fields, farms and farmers in agricultural landscapes must be considered. Farmers are heterogeneous, as they differ in terms of the systems (*e.g.*, organic or conventional farms), fixed input dotation (*e.g.*, farm size) and preferences (*e.g.*, risk preferences). The spatial heterogeneity of farm territories is also very high, at least in many European landscapes. The heterogeneity of field quality relates, for example, not only to the size and shape of the plots but also to the soil quality of these plots. Moreover, the fragmentation of farm territories, *i.e.*, both the parceling and the scattering in space of the whole set of fields of each farm, is highly heterogeneous. Agricultural landscape mosaics are largely made of these interwoven, and more or less fragmented, farm territories, which induces complex spatial interdependencies between ecological processes and agricultural management that should be taken into account when examining farmers' behavior in realistic situations (Martel et al., 2017; Sutherland et al., 2012).

To our knowledge, the first study that investigated the issue of economic and ecological interdependencies between crop production and biodiversity (mobile ES providers) at the landscape scale was Cong et al. (2014, 2016). The main result of Cong et al. (2014) was that the coordinated management of pollination at the landscape scale (called "landscape-scale management" and noted hereafter as LSM) increases the profit of each farmer more than the uncoordinated management of pollination (called "farm-scale management" and noted hereafter as FSM). Epanchin-Niell and Wilen (2014) and Atallah et al. (2017) stressed that LSM improves the profits of all farmers in most cases. Cong et al. (2016) showed that the achievement of the LSM solution is characterized by a landscape mosaic with a dispersed configuration of habitats, depending on the arrangement of the farms in the landscape.

However, these works rely on simple raster-stylized representations of landscapes, farms and plots, with homogeneous fields, homogeneous farmers and continuous farms. Therefore, several aspects of agricultural heterogeneity have not been fully addressed. In line with the sensitivity analysis conducted by Atallah et al. (2017), who highlighted that the benefits of collective management depend on the heterogeneity of the product quality of the two modeled farmers, more realistic modeling is suspected to change the conclusions from previous works.

Based on the first attempts of Cong et al. (2014, 2016), the aim of our study was to examine the benefits of collective ES management in realistic landscapes with heterogeneous farms/farmers. For this purpose, we simulated different biological control management strategies through an agronomic-ecologic-economic landscape model on a realistic landscape site that is representative of north-western France. The model we developed was derived from the first model, which allowed us to evaluate the impact of land-use allocation in several types of farms on landscape patterns and on populations of carabid beetles, which are considered to be a potential biological control (Martel et al, 2017). The ecological function that we propose in the present model is more complex than the usual ecological function that is used in the literature on the coordinated management of productive ES. For example, Epanchin-Niel and Wilen (2014) considered only the species dispersal from one field to that of the neighbors, and Cong et al. (2014) and Atallah et al. (2017) considered the decreasing probability of species dispersal using the distance between one field and the others. Here, we enhance the realism of the ecological modeling by integrating recent results in landscape ecology that highlighted the role of interfaces within agricultural landscape mosaics at 500 m buffer scales on the life cycle of carabids (Martel et al 2017).

In the present study, we considered several degrees of biological control management: (i) no management at all, (ii) a naïve-FSM strategy where farmers do not communicate with each other, (iii) a rational FSM strategy where farmers communicate with each other regarding their crop allocation intentions and (iv) the commonly simulated LSM strategy.

Therefore, three main hypotheses were tested. Our first hypothesis is that the landscape-scale total profits will gradually increase from scenario 1 (no management) to scenario 4 (LSM) due to the gradually increasing management of the carabid beetles. Second, in line with Cong et al. (2014), we hypothesized that total profits at the landscape scale would be the highest for LSM, but we depart from previous authors' results by considering that not all farmers will benefit from LSM due to their heterogeneity. Finally, Martel et al. (2017) found that both the share of the area and the relative crop patterns of the farms in the landscape influenced their contribution to carabid abundance in the landscape. Applying the same perspective, we hypothesized that the relative structural characteristics of the farm territories vis-à-vis the landscape site would influence the gains in all scenarios.

The paper is structured as follows. We first describe our model and empirical strategy (Section 5.2). We then present the results of one hundred simulation replicates (Section 5.3). We analyze the distribution of the overall and individual profits in the considered scenario and also analyze the farm and landscape characteristics as potential drivers. We finally discuss our results and methodological choices (Section 5.4).

5.2 Material and Method

5.2.1 Genesis of the landscape model

Our empirical approach consists of modeling different levels of collective crop allocation management to optimize profits based on a productive ES within a continuous landscape site. For this purpose, we adopted the models and data of Martel et al. (2017), who used the landscape modeling framework APILand (Boussard et al., 2010).

APILand is a JAVA® library that includes the following concepts and features: (i) a meta-model of landscape representation in terms of space, time, and theme that facilitates the combination of farm territories within a non-agricultural matrix (of, *e.g.*, roads, buildings, woodlots, and hedgerows); (ii) a set of simulation tools for managing the virtual experience plans; (iii) a spatio-temporal dynamic crop allocation module (CAPFarm) that explicitly takes into account farm system constraints and territories; and (iv) a landscape metrics analyzer (Chloe) using sliding windows to ecologically characterize agricultural landscapes.

The aim of Martel et al. (2017) was to understand the impact of farm spatial organization on carabid beetle populations to implement territorial management solutions. Their model called Agriconnect determines the abundance of carabid beetles depending on landscape connectivity (due to the size and dispersion of the plots) and composition (of the crops and other fixed elements, such as semi-natural elements). Agriconnect was implemented on two realistic landscape sites, one with few woody elements and one with many woody elements (woodlots and hedgerows). Both sites contained eight farms with heterogeneous farm territories and were selected from the entire Brittany region to minimize the number of farmers in a 500 m radius circle. Martel et al. (2017) also considered two realistic farm systems, "swine" and "cattle", with specific crop allocation rules. Those rules were translated into CAPFarm agronomical constraints based on farmers' interviews. The farmers validated ex-post the simulation results. Agriconnect also contained two statistically validated ecological models for two distinct carabid beetle species groups involved in biological control: the species associated with woody habitats (the 'woody' model) and the species associated with maize crops (the 'maize' model).

To focus on our issue, we adapted and further developed the Agriconnect model. We used the sole landscape site with few woody elements and extended the area from a 500 m radius circle to a 1 km radius circle (see part b.), leading to the modeling of ten farms (instead of eight in Agriconnect). We used only the "swine" farm system (see part c.). We also generalized the "maize" carabid model to the whole cropped area, including permanent grassland (see part d.). We added an economic module that defined crop-specific profitability, which depends positively on the abundance of carabid beetles, which are considered as a source of biological control (see part e.). These adaptations led to a new virtual experiment plan (see part f.) that considered four scenarios with distinct objective functions regarding the collective management of ES (see part g.).

We present the details of those different steps in the following parts.

5.2.2 The landscape site and the farm territories

The landscape site is a spatially continuous 1 km radius circle where (i) crops are allocated, (ii) carabid beetles abundance is computed and (iii) profit-based objective functions are maximized. The landscape site is a subzone of a larger area containing all the farm territories, which is necessary for running consistent allocations at the farm level (see Figure 5.1).

This whole landscape is composed of a non-agricultural part and an agricultural part. The non-agricultural part consists of fixed landscape elements (hereafter referred to as fixed elements), including artificial elements, such as roads and buildings; natural elements, such as water bodies; and semi-natural elements, such as woodlots, herbaceous field margins and hedgerows. These elements are not included in the simulation process: they cannot be modified by the farmers. The farmsteads are also fixed and are part of $\mathbf{L}$. In the agricultural part, all the covers are allocated, *i.e.*, are processed in the simulation, but permanent grasslands stay fixed because of allocation constraints.

Ten farm territories contribute to the agricultural part of the landscape site. The territory of each farm j is composed of one farmstead and a vector $\mathbf{I}_j$ of fields; all farms together contribute to a landscape mosaic of $\mathbf{I}$ fields. Each farm territory is managed by a farmer; farmers can neither exchange their fields nor modify the shape and size of their fields; *i.e.*, field boundaries are fixed. We use $\mathbf{I}_j^s$ to represent the subset of fields that are at least partly contained in the landscape site and that belong to farm *j*. The cover of those fields is the single variable of our model, and it is dynamically allocated by the CAPFarm solver (see part c.).

The landscape site is modeled as a raster data set with 31,214 pixels. Each pixel *p* is 100 m² and belongs to a single field or to a single fixed landscape element. In other words, each field in $\mathbf{I}_j^s$ consists of a specific vector of pixels $\mathbf{P}_i$. Thus, the farmers are heterogeneous in the sense that they have heterogeneous farm territories containing heterogeneous fields in terms of size, shape and localization.

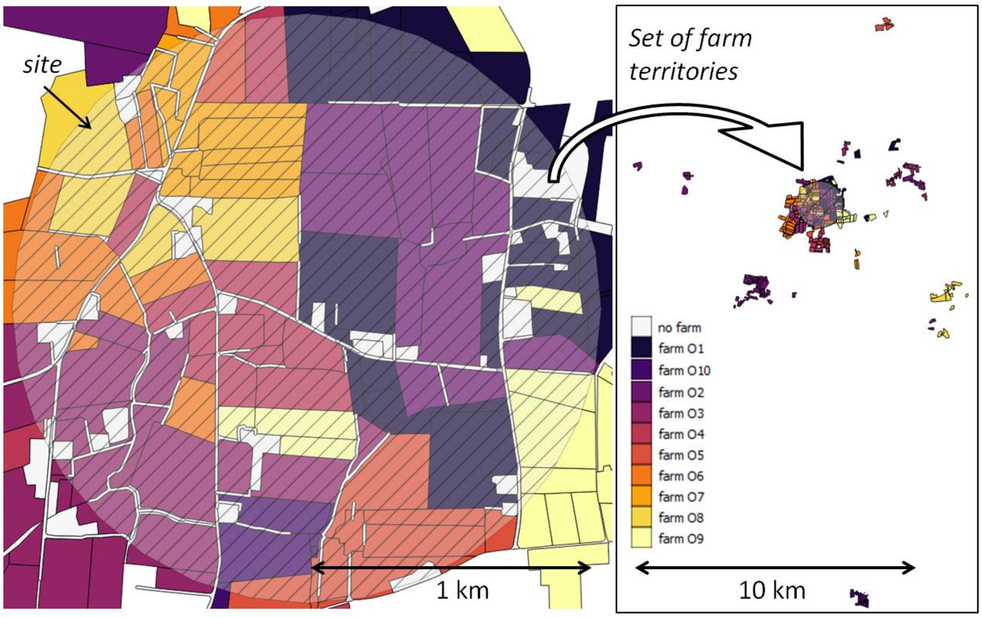


Figure 5.1. The landscape site included in the whole set of farm territories

The selected landscape site represents an area of 314.16 ha, with 272.33 ha of crop fields and the rest consisting of fixed elements, including 9.67 ha of woody elements. There are 120 fields either totally or partially included in the landscape site.

Table 5.1 displays the descriptive statistics of the fixed characteristics of the farm territories vis-à-vis the landscape site. In particular, the table indicates (a) the share of each farm territory included in the landscape site and (b) the share of the landscape site fields that belong to each farm. The table also indicates the length of the interfaces between (c) the fields that belong to the same farm (called fixed "intra"-interfaces hereafter) and (d) the fields that belong to distinct farms (called fixed "inter"-interfaces hereafter). In total, there are 21.22 km of fixed interfaces in the landscape site, 59% being fixed intra-interfaces. Note that the length of the fixed interfaces in the site by farm is considerable (110 m of fixed interfaces per ha, on average), illustrating how finely the farm territories are fragmented and interwoven in the landscape site.

Table 5.1. Descriptive statistics of the farms

Farm	UAA (ha)	UAA in landscape site (ha)	(a) Farm UAA in the landscape site (%)	(b) Farm UAA of the landscape site (%)	Fixed interface length of the site (km)	Fixed interfaces length of the site (%)	(c) Fixed "intra" interfaces length (km)	(d) Fixed "inter" interfaces length (km)
O1	104.22	56.13	53.86	20.61	7.28	34.31	3.99	3.29
O2	191.88	46.54	24.26	17.09	3.51	16.55	1.68	1.83
O3	130.59	38.99	29.86	14.32	2.37	11.15	0.88	1.48
O4	82.90	26.61	32.10	9.77	3.31	15.60	0.89	2.42
O5	54.93	23.65	43.06	8.69	2.82	13.29	1.36	1.46
O6	90.22	19.95	22.11	7.33	2.39	11.25	0.47	1.92
O7	28.23	17.43	61.73	6.40	3.02	14.25	1.66	1.36
O8	75.46	18.90	25.05	6.94	1.41	6.67	0	1.41
O9	63.95	15.32	23.95	5.63	2.57	12.10	1.21	1.36
O10	38.54	8.80	22.83	3.23	1.18	5.58	0.43	0.75

Legend. UAA: utilized agricultural area; "intra": within farms; "inter": between farms.

5.2.3 The crop allocation submodel

The crop cover of the fields is the single variable in our model. Next, we specify how the dynamic crop allocation model is built and computed through the CAPFarm solver.

We consider that each farmer *j* allocates the *K* crops he produces among his $\mathbf{I}_j$ fields according to a set of farm-level agronomic constraints adapted from Agriconnect's realistic "swine" farm system (Table 5.2). The CAPFarm solver randomly generates a crop allocation that verifies this set of spatial and temporal constraints; one field is covered by a single crop *k* for a given year. The cover of pixel *p* by *k* is denoted as $\gamma^s_{p,k}$, producing a landscape mosaic covered by the matrix $\boldsymbol{\gamma}^s$ of crop pixels (embedded into the matrix $\mathbf{L}^s$ of fixed elements). The cover function is also applied to the fields throughout the farm territories; each field *i* is covered, respectively, by $\boldsymbol{\gamma}^s_{i,k}$ and $\boldsymbol{\gamma}_{i,k}$ inside and outside the landscape site, and the whole farm territory is covered by $\boldsymbol{\gamma}_j$.

The "swine" farm system considers five crops with six types of constraints (see Table 5.2). The only change from Agriconnect is that we impose a non-null area for each crop while Agriconnect imposed a minimum area for cash-crop and on-farm pig-food productions. Here, our purpose is to precisely select the optimal landscape based on profit maximization. Note that in this farm system, we constrain farmers to maintaining permanent grasslands, meaning that some fields are fixed across simulations.

Table 5.2. The adapted "swine" farm system adapted from Martel et al. 2017

Crop	Code	Allowedpreviouscrop	Field size	Total area	Min. return time	Max. duration	Never on
Maize-Corn	MC	WH-BA	> 0.5 ha	> 0 ha	2	1	-
Wheat	WH	MC-RA	> 0.5 ha	> 0 ha	2	1	-
Barley	BA	WH	> 0.5 ha	> 0 ha	3	1	-
Rapeseed	RA	WH-BA	> 0.5 ha	> 0 ha	3	1	Drainedfield
Permanent Grassland	PG	PG	< 0.5 ha	-	-	-	-

Note that winter crops (WIC), as mentioned in the following text, include wheat, rapeseed and barley.

We generate a historical background of crop allocation to ensure that the dynamics of crop allocation respect temporal constraints (Table 5.2).

5.2.4 The ecological submodel

We need an ecological model that is applied to the whole landscape site to express a productive ES. For this purpose, we adapt the "maize" carabid beetle abundance model, as defined in Agriconnect. Martel et al. (2017) statistically estimated carabid beetles abundance in maize fields based on surrounding landscape metrics. While their model considered that only maize fields attracted carabids, here, we consider that similar to the maize fields, other crops and grasslands also attract carabids, according to the surrounding landscape pattern. Therefore, we were able to estimate the profits derived from ES for the whole agricultural area in the landscape site. Except this modification, the ecological function is the same as that in Martel et al. (2017). The function is computed on each pixel of the landscape site such as:

$$c_p\left(\boldsymbol{\gamma},\mathbf{L}\right)=e^{\left(4.98+\left(6.78^{E-04}E_{WIC-MA500,p}\right)-\left(7.05^{E-06}C_{W500},p\right)\right)} \tag{5.1}$$

where c_p is the abundance of carabid beetles on pixel *p*. The first defined landscape metric is the length of the interfaces between maize (MA) and winter crops (WIC)[61] in a 500 m radius circle around *p* (denoted as $E_{WIC-MA500,p}$). The second metric is the Hanski connectivity of woody elements (W) in a 500 m radius circle around *p* (denoted as $C_{W500,p}$). Function (5.1)

[61] The interfaces between the winter crops and maize crops will be hereafter referred to as "interfaces WIC_MA".

implies that carabid beetle abundance increases with the length of the interfaces between MA and WIC but decreases with the connectivity of woods. The abundance of carabid beetles depends on farmers' crop allocations γ through $E_{WIC-MA500,p}$. The impact of $C_{W500,p}$ on carabid beetles is exogenous to farmers' choices and is fixed. However, $C_{W500,p}$ is different in each pixel *p*.

We compute the used landscape metrics according to circle sliding windows with the Chloe software (Baudry and Boussard, 2012).

Our 500 m buffer analysis centered on each pixel *p* is influenced by "site edge effects". Indeed, there are missing values outside the landscape site, but the extent of farm territories around the landscape site provide additional information, leading to a different degree of spatial uncertainty regarding the abundance of carabid beetles. Figure 5.2 shows the uncertainty of the ecological model due to the site edge effect, which is 0.09, on average, and 0.44, at the most.

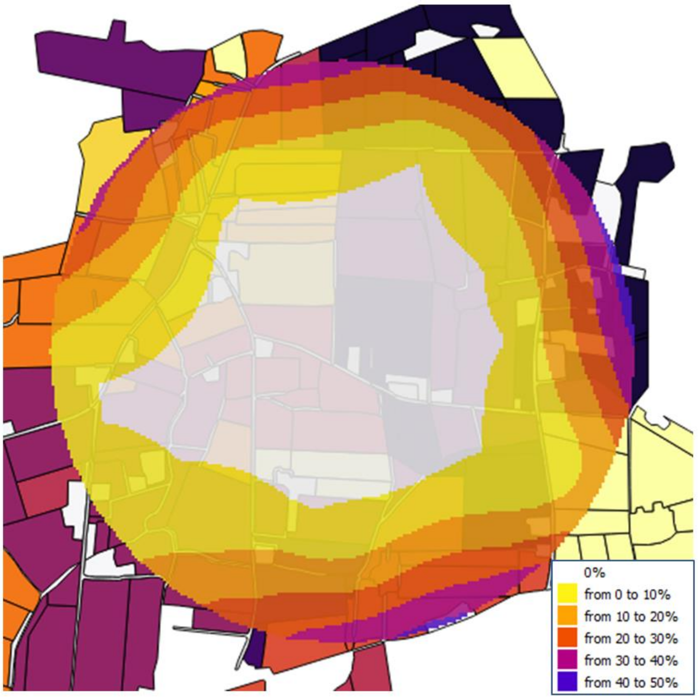


Figure 5.2. Distribution of the spatial uncertainty in the ecological model due to the landscape site edge effect

5.2.5 The economic submodel

In our model, we assume that the profits of the farmers depend on productive ES such that gross margins $\pi_{k,p}$ specifically differ from pixel to pixel depending on the abundance of carabid beetles. The profit Π_j of farmer *j* is the sum of the gross margins across all the pixels managed by *j* in the landscape site. We denote the profit of farmer j as:

$$\begin{aligned} &\Pi_j = \sum_{i \in \mathbf{I}_j^s} \sum_{p \in \mathbf{P}_i} \pi_{k,p} \left(c_p \left(\boldsymbol{\gamma}, \mathbf{L} \right) \right) \\ &s.t. \quad \mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B} \end{aligned} \tag{5.2}$$

where $\mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B}$ is the set of constraints used to generate the different possible crop allocations, which apply at the farm level across the whole farm territory (Table 5.2). Here, we consider that the gross margins $\pi_{k,p}$ depend only on carabid beetles abundance $c_p(\boldsymbol{\gamma}, \mathbf{L})$; all other elements that are suspected to influence the gross margins, such as capital or pesticide and fertilizer applications, are exogenous and assumed to be equal across farms. As noted in part d., carabid beetles abundance in *p* depends on the structure of the surrounding landscape, *i.e.*, on all farmers' crop allocation decisions $\boldsymbol{\gamma}_j$. We assume that the gross margins $\pi_{k,p}$ depend positively on $c_p(\boldsymbol{\gamma}, \mathbf{L})$ such as:

$$\pi_{k,p} = \left(\alpha_k + \beta_k \frac{c_p}{\bar{c}} \right) \bar{\pi}_k \tag{5.3}$$

Where $\bar{c}$ is the average abundance of carabid beetles computed by the Chloe software for 500 randomly generated landscapes by CAPFarm; $\bar{c}$ is equal to 56 carabid beetles per m². The gross margin for field *i* and crop product (output) *k* is a function of (i) α_k, the share of the gross margin independent from carabid beetles; (ii) β_k, the share of the gross margin depending on the ES provided by the carabid beetles such that we have $\alpha_k + \beta_k = 1$, (iii) $\bar{\pi}_k$, a parameter representing the normalized profitability for output k; and (iv) c_p. This notation is the translation of the production function used by Cong et al. (2014, 2016) to crop gross margins. In their case, α_k represented the crop yield, which is independent from the pollination, and β_k represented the crop yield that depends on pollination. Here, we adopt a similar interpretation even if the parameters are applied to gross margins; *i.e.*, β_k represents both the gains from additional yields and a reduction in the costs linked to the reduction in pesticide utilization. In contrast to Cong et al. (2014, 2016), who tested a different set of parameters (α_k, β_k), we specifically calibrate the parameters $\boldsymbol{\beta}$ by following Bareille and Dupraz (2017) and Bareille and Letort (2018), who, together, have identified these parameters for our five considered outputs from farm samples in northern France. These studies have estimated the productivities and input savings due to biodiversity indicators at the farm scale, allowing for the determination of the elasticities of the gross margin to these indicators (which correspond to β). Here, we assume that all the benefits from these indicators are due to carabid beetles.

Table 5.3. The parameters $(\bar{\pi}_k, \alpha_k, \beta_k)$ for the five considered crops (source: authors' own computation based on Bareille and Dupraz 2017 and Bareille and Letort 2018)

Crops	Normalized gross margins $\bar{\pi}_k$ (€/ha)	Gross margin share independent from ES α_k	Gross margin share dependent from ES β_k
Maize-Corn	830	0.82	0.18
Wheat	840	0.864	0.136
Barley	650	0.841	0.159
Rapeseed	740	0.862	0.138
Permanent Grassland	400	0.82	0.18

5.2.6 Resolution of simulations

The principle of the resolution of our model is the following. First, for each farm, we generate a random series of crop allocations for three years, which constitute the historical background of farmers' crop allocations and define the initial conditions. Second, for a given historical background of a specific random crop allocation, we simulate thirty crop allocations respecting the constraints for a single year per farm, leading to 30^10 possible crop allocations for the whole set of farm territories. We restrain the number of possible crop allocations for three reasons: i) the computation of profit for each pixel for one possible landscape (one loop) takes approximately five seconds (see Figure 5.3); (ii) the number of possible landscapes increases exponentially with the number of possible crop allocations per farm, increasing the required number of loops; and (iii) it would not be possible to explore the whole range of solutions. More crop allocations would have, of course, led to increased profits, as farmers have more flexibility, but these additional profits consume more time for computation.[62] In addition, we consider that selecting 30 crop allocations is sufficient to explore the range of alternative solutions a farmer may formulate in real conditions. Third, we perform the simulation loop described in Figure 5.3, which (i) generates the possible landscapes given the different farmers' crop allocations, (ii) computes the abundance of carabid beetles on each pixel using formula (1) for each possible landscape, (iii) computes the farmers' individual profits on each pixel for the considered

[62] Thirty crop allocations per farm corresponds to the number of crop allocations that provides the highest marginal information per unit of time when considering five to forty crop allocations.

landscape using formula (5.2), and (iv) selects the optimal farm crop allocations maximizing individual or collective profit functions according to the four scenarios (see part g.). Note that this process optimizes the profit in an *a posteriori* way since we cannot *a priori* solve the optimization problem when we introduce ES into the gross margin functions. Given the slow *a posteriori* procedure, we repeat this resolution procedure only 100 times (called replicates hereafter), which, according to the law of large numbers, leads to a maximum error risk of 10% for our results.

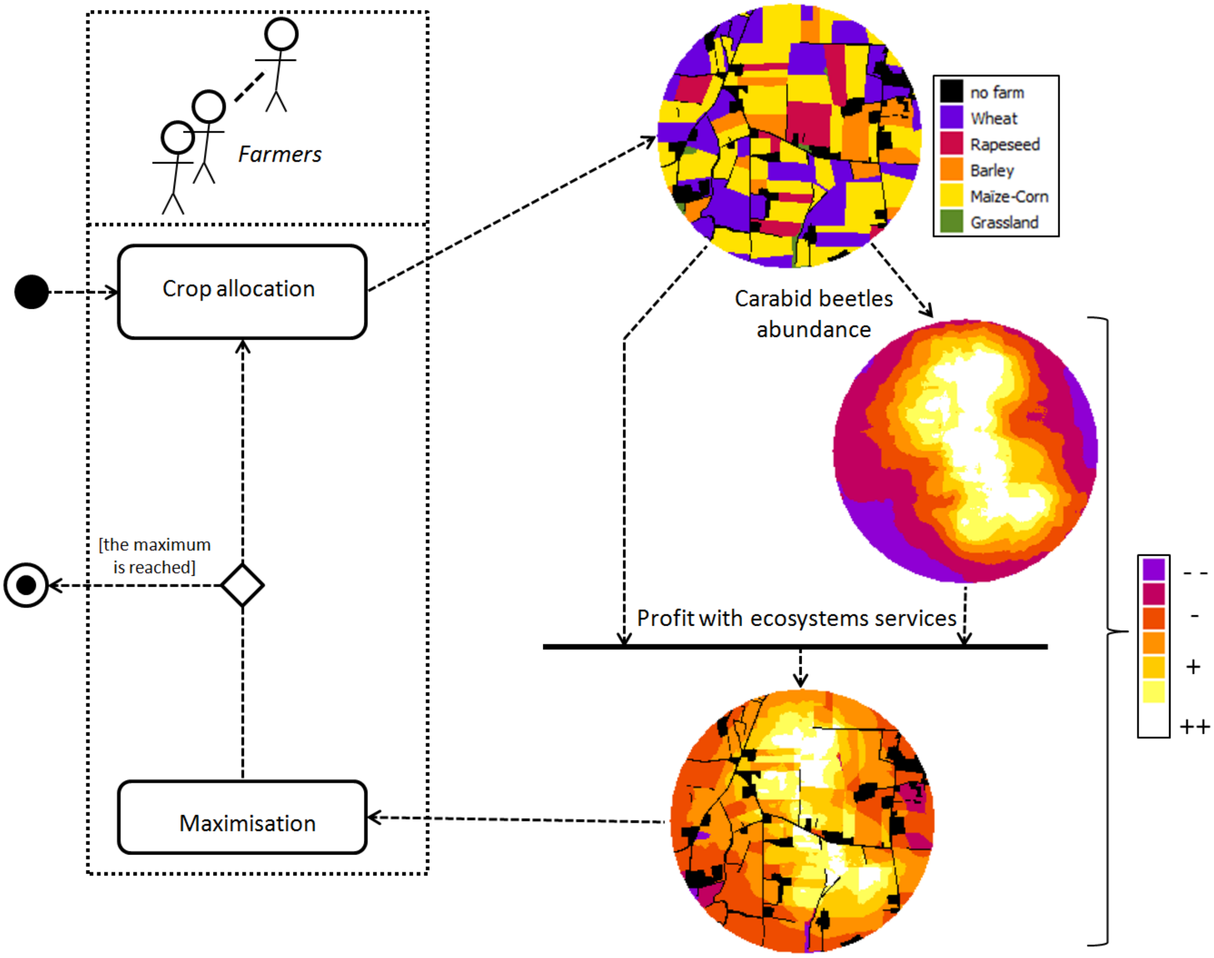


Figure 5.3. Resolution process of a simulation loop represented in a UML activity diagram (UML: unified modeling language)

5.2.7 Four management scenarios

The purpose of the four scenarios is to consider successive ES management possibilities, namely, no management (scenario 1), naive farm-scale management (scenario 2, referred to as naive-FSM hereafter), Nash farm-scale management (scenario 3, referred to as Nash-FSM hereafter) and landscape-scale management (scenario 4, referred to as LSM hereafter).

Scenario 1 (no management) represents constrained profit maximization with carabid beetles modeled as externalities. In other words, we consider that farmers ignore the fact that carabid beetles influence the profitability of the different crops. We consider that farmers maximize the following expected profit function:

$$\max_{\boldsymbol{\gamma}_j} E\left(\Pi_j\right) = \sum_{i \in \mathbf{I}_j^s} \sum_{p \in \mathbf{P}_i} \bar{\pi}_{k,p} \qquad (5.4)$$
$$s.t. \quad \mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B}$$

The expected profit of farmer j depends only on the direct benefits $\bar{\pi}_k$. Without any constraints $\mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B}$, the results of (5.4) would lead to the monoculture of the most profitable crop. The resolution of (5.4) for the ten farmers leads to the optimal landscape $\boldsymbol{\gamma}^{1*}$. The real profits generated from function (5.4) are computed using relation (5.2); the difference between the real and expected profits represent the externalities generated by the carabid beetles. Regarding Figure 5.3, the crop allocation selection for each farm is directly determined due to the optimization of profit without ES.

Scenario 2 (naïve-FSM) also represents constrained profit maximization, but, this time, farmers recognize that carabid beetles influence crop profitability. In this scenario, we consider that farmers do not communicate with each other; therefore, the farmers formulate false expectations regarding the other farmers' choices. In particular, our model assumes that one farmer considers as given the resulting crop allocation from scenario 1 and that the abundance of carabid beetles depends only on his own choices. In this context, the farmers maximize the following expected profit function:

$$\max_{\boldsymbol{\gamma}_j} E\left(\Pi_j\right) = \sum_{i \in \mathbf{I}_j^s} \sum_{p \in \mathbf{P}_i} \pi_{k,p}\left(c_p\left(\gamma_{i,k} \mid \boldsymbol{\gamma}_{-i,j}^{2*}, \boldsymbol{\gamma}_{-j}^{1*}, \mathbf{L}\right)\right) \qquad (5.5)$$
$$s.t. \quad \mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B}$$

where $\boldsymbol{\gamma}_{-i,j}^{2*}$ is farmer j's crop allocation choices for his fields other than i and $\boldsymbol{\gamma}_{-j}^{1*}$ represents the optimal crop allocation of the other farms in scenario 1. The profit of farmer j thus depends on the direct benefits due to the crop allocation choices and the indirect benefits generated by crop allocation choices through the evolution of the abundance of carabid beetles on his fields. The crop allocation decisions $\gamma_{i,k}^{*}$ depend on the anticipated effect of s_i on carabid beetles density; farmer j considers that the cover of the other farmers' plots are fixed at $\boldsymbol{\gamma}_{-j}^{1*}$ and knows that his other plots are $\boldsymbol{\gamma}_{-i,j}^{2*}$. However, as each farmer makes this same assumption, the real level of $\boldsymbol{\gamma}_{-j}$ is not $\boldsymbol{\gamma}_{-j}^{1*}$ but rather $\boldsymbol{\gamma}_{-j}^{2*}$. In other words, the farmers do not consider that the other farmers also seek to optimize the abundance of carabid beetles and thus face externalities. The resolution of relation (5.5) for the ten farmers leads to the optimal landscape $\boldsymbol{\gamma}^{2*}$, where, obviously, the real profits (5.2) differ from the expected profits resulting from relation (5.5). Regarding Figure 5.3, the crop allocation selection for each farm is realized by computing carabid beetles abundance based on the anticipated crop allocation of the nine other farms.

Scenario 3 (Nash-FSM) is similar to the second scenario but consists of changing the form of the farmers' expectations regarding the behavior of the other farmers. Here, we consider that the farmers communicate their ideal crop allocation plan with each other; *i.e.*, the farmers have rational expectations regarding the behaviors of the other farmers. This scenario is similar to the FSM strategy of Cong et al. (2014) and leads to another optimal landscape that corresponds to the Nash equilibrium. In this context, the farmers maximize the following expected profit function:

$$\begin{aligned} &\max_{\boldsymbol{\gamma}_j} E\left(\Pi_j\right) = \sum_{i \in \mathbf{I}_j^s} \sum_{p \in \mathbf{P}_i} \pi_{k,p} \left(c_p \left(\gamma_{i,k} \mid \boldsymbol{\gamma}_{-i,j}^{3*}, \boldsymbol{\gamma}_{-j}^{3*}, \mathbf{L} \right) \right) \\ &s.t. \quad \mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B} \end{aligned} \tag{5.6}$$

where $\boldsymbol{\gamma}_{-i,j}^{3*}$ is farmer j's crop allocation choices for his fields other than i and $\boldsymbol{\gamma}_{-j}^{3*}$ represents the optimal crop allocation of the other farmers. The farmers internalize the effects of the other farmers' decisions regarding carabid beetles abundance but maximize their profits individually. As shown in Figure 5.3, we solve this equilibrium by successively running the crop allocation decision models until the cover of each field remains fixed between two periods. This technical optimization is similar to the one used by Cong et al. (2014), who used this procedure to imitate rational anticipations.

Scenario 4 (LSM) is similar to the LSM strategy in Cong et al. (2014) and consists of simulating the grand coalition described in cooperative game theory. Here, all the farmers manage their crop allocations collectively to maximize the sum of the individual profits. In other words, while the first three scenarios maximize the private optimums, this fourth scenario maximizes the social optimum. By definition, one farmer in the grand coalition does not need to anticipate the other farmers' choices because the farmers in the grand coalition make their choices collectively. This scenario leads to a fourth optimal landscape. In scenario 4, the farmers maximize the total profit Π under the choices of γ such as:

$$\begin{aligned} &\max_{\gamma} \Pi = \sum_{j=1}^{10} \Pi_j = \sum_{j=1}^{10} \sum_{i \in I_j} \sum_{p \in \mathbf{P}_i} \pi_{k,p} \left(c_p \left(\gamma_{i,k} \mid \boldsymbol{\gamma}_{-i}^{4*}, \mathbf{L} \right) \right) \\ &s.t. \quad \mathbf{A}\boldsymbol{\gamma}_j \leq \mathbf{B} \quad \forall j \in [1,10] \end{aligned} \tag{5.7}$$

The total profit Π corresponds to the sum of the profits of the ten farmers; each farmer is still subject to farm-scale constraints. In this case, the farmers anticipate perfectly the crop allocations for all other fields $\boldsymbol{\gamma}_{-i}^{4*}$. As shown in Figure 5.3, the optimal landscape is obtained by directly considering the ten farms as one single farm, without any need for presenting anticipated landscapes.

5.3 Results

5.3.1 Analysis of total profits: is LSM the best strategy at the landscape scale?

Figure 5.4. presents the distribution of total landscape-scale profits (the sum of the farm-scale profits) among the four scenarios for the 100 replicates. Table 5.4. presents the relative total profits of the different scenarios.

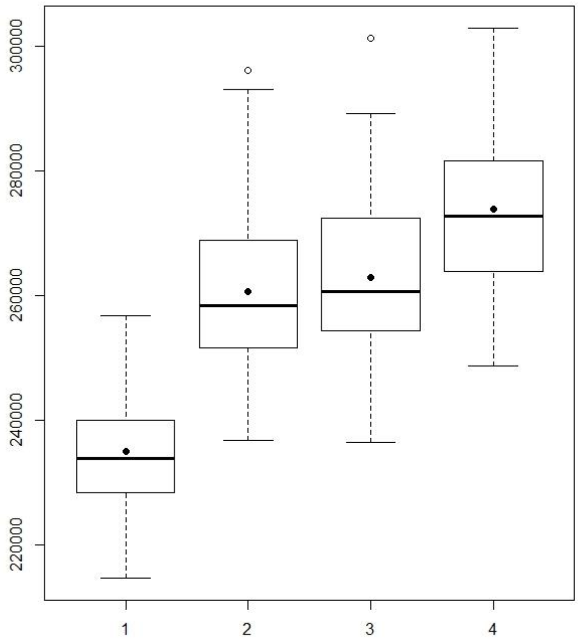


Figure 5.4. Box-plot representation (medians - quartiles) of the distribution of total profits at the landscape scale for the 4 scenarios (N=100). *The black points represent the means. Legend: 1: "no management", 2: "naïve-FSM", 3: "Nash-FSM" and 4 "LSM".*

We observe an increase in total profit among the four scenarios (Figure 5.4). On average, we find that LSM increases total profit by 16.7% compared to the absence of management (Table 5.4). We find no statistically significant differences between naive-FSM and Nash-FSM; the p-value of the Student test is 0.14, which indicates that the communication in the Nash-FSM does not significantly improve overall profits compared to the naive-FSM. However, we find a statistically significant difference between LSM and the other scenarios. In particular, we find results that are similar to those of Cong et al. (2014): on average, LSM increases the total profit by 4.2% compared to the Nash-FSM; the impact ranges from +0% +17% across the 100 replicates. Thus, the introduction of heterogeneous farmers does not change the previous results: the farmers benefit from a better allocation of the habitat across the landscape when they act collectively.

Table 5.4. Relative total profits for the no management, naive-FSM, Nash-FSM and LSM scenarios (N=100)

	mean	**median**	**min**	**max**
Profit Naive-FSM / Profit no management	1.109	1.099	1.026	1.227
Profit Nash-FSM / Profit no management	1.120	1.115	1.025	1.232
Profit LSM / Profit no management	1.167	1.170	1.063	1.348
Profit Nash-FSM / Profit Naive-FSM	1.009	1.000	0.945	1.099
Profit LSM / Profit Naive-FSM	1.052	1.051	1.000	1.125
Profit LSM / Profit Nash-FSM	1.042	1.038	1.000	1.170

Figure 5.5 presents the average spatial distribution of carabid beetles abundance and the gross margins for the four scenarios across the 100 replicates. We observe a progressive increase in the abundance of carabid beetles across the four scenarios. The carabid beetles are less abundant close to the boundaries due to both site edge effects and the presence of fixed elements, such as built areas (Figure 5.1). The gross margins also increase for most parts of the landscape, explaining why LSM is the best management strategy at the landscape scale.

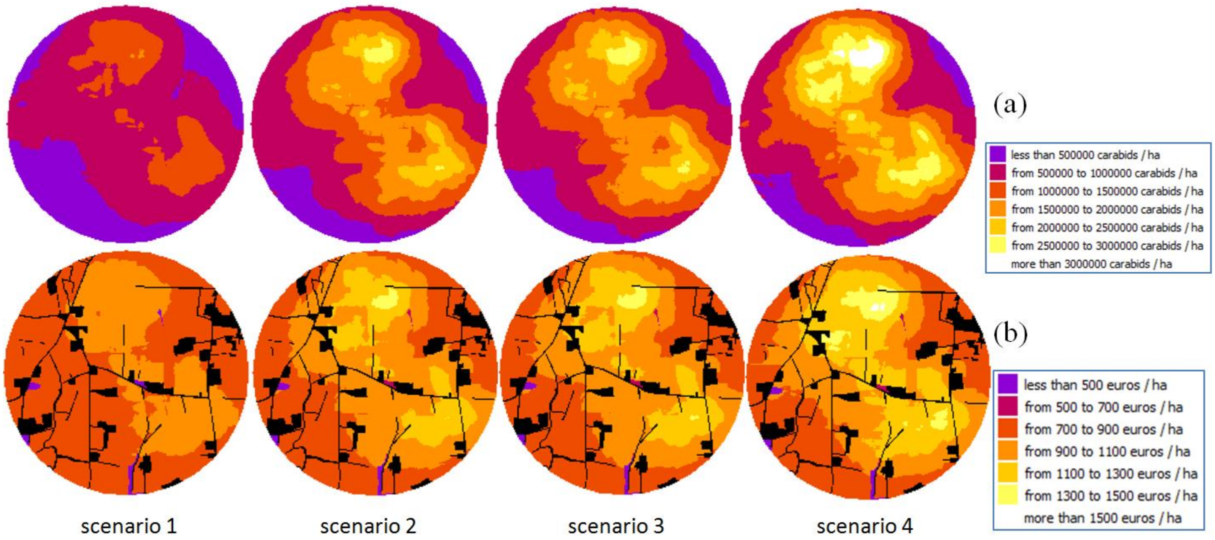


Figure 5.5. Average distribution of (a) carabid beetles abundance and (b) the gross margins across the modeled landscape

5.3.2 Analysis of the individual profits: is LSM the best strategy at the farm scale?

We now analyze the distribution of the LSM gains at the farm scale. Figure 5.6 presents the relative profits for the LSM case compared to the no management case and the Nash-FSM case. We find that, on average, farmers have higher individual profits for LSM than for Nash-FSM and no management, confirming the results in Cong et al. (2014) and Epanchin-Niell and Wilen (2014) and indicating that all farms would win if farmers act collectively. However, we find considerable heterogeneity in the individual profits across the ten farmers. For instance, farm O10 presents an average gain of +0.9% with LSM compared to the Nash-FSM, while farm O7 presents average gains of +10.2%, illustrating that the introduction of heterogeneous players leads to a greater difference in the results than that found by Cong et al. (2014) and Epanchin-Niell and Wilen (2014).[63]

[63] The relative gain from Nash-FSM to LSM between the farmer winning the most and the farmer winning the least was limited to four in Cong et al. (2014), whereas gains are 11 times higher for farm O7 relatively to farm O10 here.

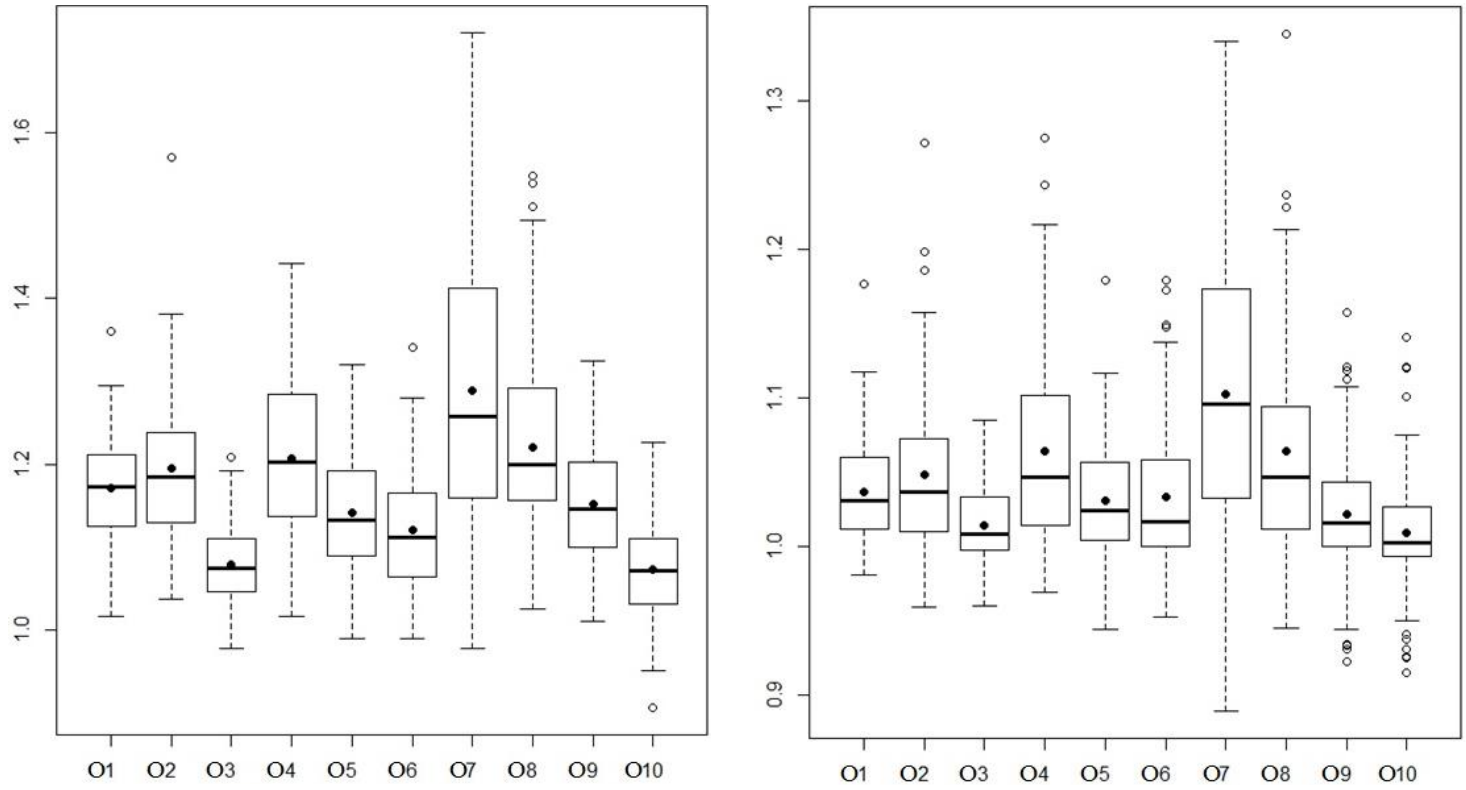


Figure 5.6. Box-plot representation (medians - quartiles) of the distribution of farmers' profits with LSM compared to (a) no management and (b) Nash-FSM. *The abscissa axis indicates the 10 farms. The ordinate axis indicates the profit relative gains. The black points represent the means.*

More importantly, we find that compared to Nash-FSM, LSM leads to gains for all farmers in only 15% of the replicates. Thus, in most cases, the highest total profit at the landscape scale is reached when at least one farmer agrees to lose. This result implies that, *ceteris paribus*, LSM can appear in only 15% of the cases. Indeed, according to the framework of the cooperative game theory, our modeling indicates that the internal stability criteria of the grand coalition is unverified in 85% of the replicates. This is a major finding, as previous results for the collective management of ES suggested that farmers' individual profits always increase with coordination (Cong et al., 2014; Epanchin-Niell and Wilen, 2014). Specifically, we find that the total landscape-scale profits for the cases where all farmers win due to collective management are significantly higher than those for the cases where at least one farm loses by 3.2%. We also find heterogeneity in the cases when at least one farmer faces losses. For instance, farmer O2 gains by acting cooperatively in 93% of the replicates, while farmer O10 improves his profit in only 56% of the replicates. This heterogeneity is due to the heterogeneity of the farm territories and the farmers' initial conditions, which determine the possible farm crop allocations.

We find similar results in terms of the heterogeneity of farmers' situations when comparing the individual profits of the no management and LSM scenarios (the ranking of the farmers is globally conserved; see Figure 5.6): although farmers increase their profits by 16.5% with LSM compared to no management, we find that at least one farmer should agree to reduce his profits in 12% of the replicates. Once again, this result is explained by the initial conditions and the heterogeneity of the farmers.

5.3.3 Analysis of the drivers of collective gains

We now examine the relationships between the profits of the four scenarios and the structural characteristics of the agricultural part of the landscape site to provide insight into the spatial aspects of the complex interdependencies between the ecological and economic processes.

Table 5.5 presents the regressions of total landscape-scale profit for the four scenarios, the difference in total profit between two successive scenarios and carabid beetle abundance, as a function of the characteristics of the dynamic crop mosaics. The advantage of analyzing the difference in total profits between two successive scenarios is removing the historical background effects and thus specifying the gains arising from the different strategies independent of the initial conditions. We selected the descriptors of crop diversity (computed as the Shannon index for the five crops, and farms are considered indistinctly) and the total length of the intra- or inter- interfaces WIC_MA; all farms are considered indistinctly.

The regressions on the four scenarios show that the two types of interfaces WIC_MA play a relatively similar role in total profits, even if the inter-interfaces seem to marginally increase the total profits more in the three scenarios with effective management than they do in the scenario with no management. This difference between the two types of interfaces is consistent with their effects on biological control; inter-interfaces WIC_MA explain 7% more carabid beetle abundance than intra-interfaces WIC_MA. At the average point, we find that an increase of 1% in inter-interfaces WIC_MA increases average carabid beetle abundance by 0.85%. The results for the differences in the scenarios confirm that the inter-interfaces WIC_MA play a greater role than the intra-interfaces WIC_MA. In particular, we find that the inter-interfaces WIC_MA explain 71.2% more of the gains achieved by Nash-FSM compared to naive-FSM than the intra-interfaces WIC_MA. Similarly, inter-interfaces WIC_MA explain 51.4% more of the gains achieved by LSM compared to Nash-FSM than intra-interfaces WIC_MA, highlighting the key impacts of the coordinated choices regarding interfaces WIC_MA in LSM. By contrast, even if the inter-interfaces WIC_MA still result in greater interest in naive-FSM than in the no management scenario, this advantage is limited to 31%. Therefore, at the landscape scale, the advantage of inter-interfaces WIC_MA over intra-interfaces WIC_MA increases across the scenarios.

Table 5.6 presents the regressions on individual profit in the four scenarios and the difference in individual profits in consecutive scenarios as a function of farm-scale descriptors. In addition to including descriptors of the dynamic structure of the farm territories (*i.e.*, Shannon diversity of crops and the length of interfaces WIC_MA), we consider the descriptors of the fixed structure of the farm territories (*i.e.*, descriptors of farm size and the length of the fixed interfaces).

Without considering the effects of the fixed structure of the farm territories (Table 5.6, right side), we find that the intra-interfaces WIC_MA play a greater role in all scenarios than the inter-interfaces WIC_MA, particularly in LSM, which differs from the previous landscape scale analysis. However, when we consider the variations among scenarios and control for fixed farm and historical background effects, we find that intra-interfaces WIC_MA play a smaller role than inter-interfaces WIC_MA in the naive-FSM to Nash-FSM gain (+47%) and in the Nash-FSM to LSM gain (+77%), while both types of interfaces play a similar role in the no management to naive-FSM gain. This result suggests that the communication in Nash-FSM and the coordinated management in LSM lead to additional gains mainly due to the reorganization of the inter-interfaces WIC_MA across the landscape. Overall, the results with and without farm fixed effects suggest that the farmers' choices of intra-interfaces WIC_MA are already relatively optimal in the naive-FSM case. These results are consistent with those for total profit at the landscape-scale.

Regarding the effects of the fixed structure of the farm territories (Table 5.6, left side), we find that the larger the share of the landscape a farmer manages in his farm, the more he benefits from collective management. We find that the share of the farm in the site has no impact, except in the LSM scenario: the greater extent to which the farm is included in the site, the more the farmer benefits from maximum-coordination management. Farmers with farms that are included in the landscape site to a lesser extent have fewer incentives to cooperate than the other farmers. Similarly, we find that the greater extent to which the farmer owns the fixed interfaces of the site on his farms, the more the farmer benefits from collective management. Finally, we find that the length of the fixed intra-interfaces increases individual profits as the degree of collective management increases (*i.e.*, from the no management scenario to the LSM scenario), whereas the length of fixed inter-interfaces decreases individual profits. In particular, the impact of the fixed inter-interfaces decreases individual profits more in LSM than in the three previous scenarios, suggesting that farms with more fixed “inter”-interfaces deviate from the private optimum to increase the social optimum in the LSM scenario.

Table 5.5. Landscape-scale drivers of the total profits

	No-management			**Naive-FSM**			**Nash-FSM**			**LSM**			**Naive-FSM – No-management**			**Nash-FSM – Naive-FSM**			**LSM – Nash-FSM**		
	estim.	std		estim.	std		estim.	std		estim.	std		estim.	std		estim.	std		estim.	std	
Constant	171,067	15,368	***	156,584	17,078	***	170,391	17,826	***	121,559	24,512	***	-	-	-	-	-	-	-	-	-
Crop diversity	-14,219	22,934		-83,567	23,501	***	-97,197	24,936	***	-64,393	34,266	°	-35,302	25,384		-83,318	22,550	***	-25,772	23,360	
Intra interfaces WIC_MA	7,533	416	***	12,221	644	***	11,704	638	***	13,710	845	***	8,005	688	***	6,708	837	***	7,813	965	***
Inter interfaces WIC_MA	6,881	483	***	12,619	701	***	12,643	610	***	14,141	1,052	***	10,493	860	***	11,520	542	***	11,843	517	***
Fixed effect (replicates)	No			No			No			No			Yes			Yes			Yes		
N	100			100			100			100			100			100			100		
R²	0.836			0.878			0.886			0.803			0.955			0,834			0,169		

°, *** mean significance level at 10% and 0.1%. Legend: WIC_MA interfaces: dynamic interfaces between winter crops and maize crops; "intra": within farms; "inter": between farms

Table 5.6. Farm-scale drivers of the individual profits

	No-management			**Naive-FSM**			**Nash-FSM**			**LSM**			**Naïve-FSM – no-management**			**Nash-FSM – Naïve-FSM**			**LSM – Nash-FSM**		
	estim.	std		estim.	std		estim.	std		estim.	std		estim.	std		estim.	std		estim.	std	
Constant	-3,433	278	***	-3,334	434	***	-3,460	436	***	-3,629	441	***	-	-	-	-	-	-	-	-	-
UAA in the site	-1	47		26	64		55	63		322	64	***	-	-	-	-	-	-	-	-	-
UAA of the site	1,702	129	***	2,775	188	***	2,747	186	***	2,727	191	***	-	-	-	-	-	-	-	-	-
Fixed interfaces of the site	216	25	***	345	35	***	362	35	***	379	35	***	-	-	-	-	-	-	-	-	-
Fixed intra interfaces	416	33	***	506	53	***	506	53	***	512	54	***	-	-	-	-	-	-	-	-	-
Fixed inter interfaces	-505	46	***	-674	64	***	-716	64	***	-1,029	66	***	-	-	-	-	-	-	-	-	-
Crops diversity	127,486	5,707	***	87,904	8077	***	90,042	8,066	***	92,536	8,036	***	-55,295	9,295	***	-72609	7,902	***	-75,224	8,740	***
Intra interfaces WIC_MA	2,761	191	***	4,070	317	***	4,190	321	***	4,820	315	***	2,182	195	***	1,909	248	***	1,309	296	***
Inter interfaces WIC_MA	1,522	126	***	1,840	164	***	1,911	155	***	1,983	160	***	2,020	156	***	2,808	143	***	2,316	151	***
Fixed effect (farms)	No			No			No			No			Yes			Yes			Yes		
N	1,000			1,000			1,000			1,000			1,000			1,000			1,000		
R²	0.984			0.976			0.977			0.977			0.216			0.079			0.169		

*** significance level at 0.1%. Legend: UAA: Utilized Agricultural Area; WIC_MA interfaces: dynamic interfaces between winter crops and maize crops; "intra": within farms; "inter": between farms; fixed interfaces: interfaces between fields irrespectively of crops.

Finally, the literature on the measure of productivity of productive ES has usually used crop diversity as an indicator of productive ES and found that crop diversity increases crop production and individual profits at a diverse array of scales, ranging from the farm scale (Bareille and Letort, 2018; Di Falco et al., 2010) to the regional scale (Di Falco and Chavas, 2008; Donfouet et al., 2017). If we find that crop diversity increases carabid beetle abundance at the landscape scale (with an elasticity of 0.21 at the average point - see Table 5.5), we find mixed results regarding its effects on profits. Indeed, if we find that on-farm crop diversity increases farmers' individual profits in the four scenarios (Table 5.6),[64] we find that crop diversity at the landscape scale, at best, has no effect and at worst, has negative effects on total profits according to the scenario (Table 5.5).[65] These opposite results could represent an aggregation effect with regard to greater heterogeneity of the Shannon index at the farm scale and to the non-linearity of the functions constituting the Shannon index. These effects are typical but have not been examined by the literature on the productivity of crop diversity.

5.4 Discussion and Conclusions

Our paper extends the current knowledge on collective ES management when considering heterogeneous farmers (in terms of heterogeneous farm territories, all other things being similar), realistic landscapes and ecological function.

5.4.1 Heterogeneous farmers and the emergence of coordination

The fact that our work considers heterogeneous farmers is a major contribution because previous studies considered homogeneous farmers (Cong et al., 2014, Epanchin-Niell and Wilen, 2014, Atallah et al. 2017). Indeed, if we find average gains in LSM that are similar to those found by Cong et al. (2014), we find that LSM improves all the farmers' profits in only 15% of the cases. By comparison, previous works considering homogenous farmers have concluded that coordination has a beneficial role in productive ES in all cases (Cong et al., 2014) or in most cases (Epanchin-Niell and Wilen, 2014, Atallah et al. 2017). The heterogeneity of the agent implies that, *ceteris paribus*, the probability that LSM will occur is 15%, casting doubts on the occurrence of LSM in real landscapes. This result confirms that the heterogeneity

[64] However, we find that as the degree of carabid beetle management increases (from naive-FSM to LSM), an increase in diversity decreases individual profits. However, the impact of this degree of crop diversity management on additional profits has not been investigated by the literature on crop diversity.

[65] These results could represent a confounding effect of the consideration of interfaces on the ecological model, but the results are robust to the omission of these variables. Crop diversity has a correlation of 0.6 with the two types of interfaces.

of farmers and the initial conditions of the landscape and farm territories are key elements when analyzing coordination processes (Atallah et al., 2017, Costello et al., 2017). We are, however, the first to empirically verify this result while focusing on the management of productive ES through land-use choices.

The use of LSM when considering heterogeneous farmers and realistic landscapes may however be influenced by other factors. First, the use of LSM may arise only if no alternative coalition structure improves the profit of at least one player (this is the principle of stability). Cong et al. (2014) noted that such a condition may not be respected in the case of the collective management of pollinators; the farmers still face incentives to avoid LSM. The consideration of heterogeneous farmers should increase these incentives (Costello et al., 2017). Second, farmers may design collective contracts such that the “winners” compensate the “losers” (Cong et al., 2014). Indeed, the probability of the occurrence of LSM can be improved by incorporating side-payments among the farmers of the coalition (Wätzold and Drechsler, 2014). The payments,, at a minimum, can be based on compensation for losses that occur as the farmers move from Nash-FSM to LSM, but alternative strategies can use payments based on either the marginal contribution of the farmers to the grand coalition or on the Shapley value (McGinty et al., 2012; Zavalloni et al., 2016). Third, the consideration of heterogeneous farmers makes the issue of inequity in cooperation even more important, and this inequity may lead to a non-efficient solution (Browning and Johnson, 1984). Indeed, a theoretical study explained that the aversion for inequity/differences may lead to a negative relation between heterogeneous coordination gains and coordination success (Fehr and Schmidt, 1999). This is a major issue, as unequal gains increase with the heterogeneity of the farmers (*e.g.*, the relative difference between the farmer who earns the most from coordination and the one who earns the least is 2.75 times higher in our study than that in Cong et al., 2014). However, laboratory experiments seem to indicate that such worries may be unfounded in the case of the coordinated management of a public input, as the choices of agents seem to be more driven by motives of social welfare maximization (Gueye et al., 2018).

5.4.2 Structure of farm territories at the origin of heterogeneous gains

Our results show that the fragmentation of farm territories generates complex spatial ecological-economic interdependencies that influence the gains in all scenarios. Two main issues are discussed: the heterogeneous spatial involvement of farms in the landscape site and the heterogeneity of intra- and inter-farm interfaces.

First, as the majority of farm territories stretch far beyond the landscape site (only three farms out of ten have more than 40% of their farm territory located within the landscape site), the interest for the coordinated management concerns a relative small part of these farms and hence, this type of management represents a small part of the entire farm profits. Therefore, it may be argued that because there is a lack of overlapping between farm territories and the landscape site, the landscape site is not an appropriate scale to for the gainful management by farmers of carabids-related ES. Nevertheless, our results showed that as more farm territories are spatially involved in the landscape site, more farmers are interested in managing ES associated with carabid beetles and in doing so in a coordinated manner. Finally, these findings do not lead to the rejection of the principle of a landscape site but show ways to enhance the relevance of a landscape site for the gainful management of carabids-related ES by 1) testing and revealing the most appropriate size for a landscape site and 2) differentiating the farmers' incentives according to their degree of spatial involvement in the landscape site.

The second aspect relates to the role of intra-farm and inter-farm interfaces in the landscape to foster the gainful management of carabid-related ES. Even in their simulation study with homogeneous one block farms, Cong et al. (2016) demonstrated that to achieve LSM, farmers' land-use allocation differs regardless of whether their farms have a few or numerous neighbors (whether the one-block farm is close to the site center or to the site edge); *i.e.*, the interfaces between farms matter in the LSM of ES. Our results provide the specifications for such spatial issues in the context of heterogeneous, fragmented farm territories. We showed that the coordinated management in LSM leads 1) to additional gains at the individual farm scale and at the landscape scale due to the length of inter-farm maize/winter crop interfaces and 2) to additional gains at the individual farm scale due to the length of the fixed intra-farm interfaces. These results suggest that 1) farm territory fragmentation in such a landscape cannot be only envisaged as a constraint decreasing profits when taking advantage of ES and 2) in coordinated management processes used to enhance the benefits of ES, the farmers' land-use allocations should not be considered without also considering land consolidation options.

To conclude, heterogeneous farm territories generate, through farmers' crop allocations, both heterogeneous landscape mosaics favorable for mobile ES providers (Martel *et al*, 2017) and complex spatial interdependencies leading to heterogeneous profits among farmers, which we were able to measure in this study. Such a diagnosis could lead to the development of novel perspectives for combining farm-scale land consolidation and management, with the LSM of

ES providers, which has rarely been accomplished thus far (Cong et al., 2016; Demetriou et al., 2012).

5.4.3 Methodological issues

Our work required us to make several trade-offs to consider realistic (and so complex) landscapes and ecological functions while ensuring the means to test our hypotheses (Cong et al, 2016; Sun et al, 2016). Our ecological function is more complex than the functions usually used for the coordinated management of productive ES, where species density depends on the distance to a specific area. This higher complexity is increased not only by the representation of realistic landscape and farm territories (heterogeneous fields, non-agricultural areas and interconnected farmers – see Figure 5.1) but also by the consideration of a larger number of decision variables (5 crops on several fields) than are usually considered. Our choice to focus on realism over method manageability prevents us from exploring the whole diversity of landscape solutions. We thus generate a subset of possible landscapes made possible by the generation of thirty possible farm territories subject to some farm-level agronomic constraints from Martel et al. (2017) for the ten farms, given a random landscape historical background. In addition, we analyze the results emerging from one hundred replicates of this procedure. This series of choices is well suited for an action research framework, as farmers from a real territory can communicate on the interest of coordinated management. This study also illustrates the interest in considering realism and heterogeneous farmers for theoretical and empirical studies on collective landscape management.

However, our method also has several drawbacks. First, the analysis on subpossibilities implies that we do not examine all the possible landscapes. In particular, we probably find the local optimum in the LSM scenario, even if we identify landscapes resulting from the optimization of the first three scenarios. Thus, our result showing that coordinated management improves individual profits in 15% of the cases is probably misestimated, even if the large difference between our result and that of previous studies stresses the need to consider heterogeneous farmers. Second, our model relies on several uncertain sources. Indeed, there were already uncertainties about carabid beetles abundance, as noted in Martel et al. (2017). We added some uncertainty by applying the abundance function to all crops, whereas the function was validated for maize crops only and considering that the gains from ecosystem functioning determined in Bareille et al. (2018) were linked only to carabid beetles. Since we apply the carabid abundance model to a 1 km radius circle, uncertainties stem from the site edge effects of the landscape site (see Figure 5.2). Third, we consider the optimization for a single year even though farming is characterized by dynamic and temporal choices (as illustrated by our constraints). Given the importance of the initial conditions on the emergence of the coordinated management solutions, long-term coordinated management is unlikely to arise over two or more agricultural campaigns (Embrey et al., 2017).

Our methodological choices are consistent with our objectives of considering a higher degree of realism and heterogeneity than is usually considered. We chose to study the interest in coordinated management for a particular type of farming system (swine production) using a particular landscape type in the Brittany region and a single ES. Therefore, our results should be interpreted as illustrative examples of this particular setting. Nevertheless, our method based on the APILand modeling framework (Boussard, 2010) may be adapted to different landscapes and different agricultural and ecological conditions.

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CHAPTER 6. DISCUSSION ON THE AGRICULTURAL MANAGEMENT OF PRODUCTIVE ECOSYSTEM SERVICES

6.1 Main contributions

The purpose of this first part of the PhD thesis was to investigate how farmers manage biodiversity productive capacity to benefit from its productive effects. This part includes four chapters that provide new insights regarding this research issue. Chapter 2 proposes a general theoretical framework. Relying on this general framework, Chapters 3 to 5 provide original empirical applications. In Chapter 2, the general theoretical framework links the literature on the productivity of biodiversity and the literature on acreage choices, disentangling several mechanisms at play in multi-output farms. In particular, considering that biodiversity productive capacity is an input of all crop-specific technologies and that it depends on the farmland area, a marginal change in the acreage of any crop impacts the yields and variable input application of all crops. This type of modelling, where yields depend on acreage, has already been considered in the literature but only with crop-specific decreasing return to acreage for each crop (Just et al., 1983). The proposed model can thus be considered as a generalization of existing acreage models and specifies that crop yields depend on the farm's overall ecosystem quality, here captured by the level of a biodiversity indicator.

My objective in this first part was primarily to introduce the farmers' behaviour into the literature dealing with the productivity of biodiversity. In this sense, the proposed structural model is valuable mainly for the literature on the productivity of biodiversity. Indeed, the decomposition of the farm-scale maximization problem in terms of "yields", "variable input applications" and "acreage choices" emphasizes that biodiversity productive capacity results from a series of choices, which are notably influenced by prices and crop-specific productive properties. Because they relied on reduced form equations, previous studies have provided a single measure of the productivity of biodiversity (most of the time on a single aggregated output), which provided a truncated picture of the whole of the agricultural technology (Mundlak, 2001). Here, we considered a mixed between dual and primal approaches to crop-specific supply and input demand functions and further integrated the fixed farm-scale inputs. This enabled us to disentangle the different steps of farmers' sequential choices and, if our assumptions on the farmers' behaviour are correct, to better measure the productivity of biodiversity and related productive ES with less potential confounders. Our measures of the productivity of productive ES, freed from these confounders, should thus be more precise than those in previous papers. In particular, the proposed model emphasizes that the lack of consideration of prices and related farmers' choices may lead to biased estimations of biodiversity productivity. The model of Chapter 2 implies that because they can be considered exogenous from farmers' point of view but drive part of their choices, prices should be considered as good variable instruments to treat the endogeneity issues arising in the estimations of the biodiversity productive capacity. If economic behaviour matters for the estimation of the productivity of biodiversity, the estimates depend on the validity of our hypotheses. In the case of misspecified behaviour, the estimations of the productivity of biodiversity may still be biased.

The model of Chapter 2 was estimated in Chapter 4, in both static and dynamic frameworks. In the latter case, farmers optimize the sum of the discounted profits anticipating the future prices, with biodiversity being a state variable similar to capital in the investment literature (Bond and Meghir, 1994). The estimation results confirm that crop yields increase when biodiversity rises. We also find that farmers reduce the application of pesticides and fertilizers when biodiversity increases. However, even if input application is a choice variable, this result does not imply that farmers manage biodiversity in purpose: it could still indicate that farmers benefit from unexpected beneficial effects from biodiversity and that they just adapt their production decisions.

The estimation in a dynamic framework suggests, however, that farmers do manage, at least partly, biodiversity productive capacity and that the beneficial effects are not pure externalities. Indeed, we find that, even if the dynamic parameter is rather small (the dynamic productive effects represent 30% of the total current effects), its estimation precision indicates that farmers, on average, do anticipate the future productive effects of biodiversity and manage their acreage in consequence, similarly to capital. In particular, one could consider that farmers diversify their acreage in the current period if they anticipate a higher relative price for a given crop in future periods. There is thus a positive cycle of biodiversity productive capacity: farmers can increase biodiversity in a given period to benefit from it in future periods. Such a positive cycle is the opposite of what Skevas et al. (2012, 2013) found for pesticide, where current pesticide application decreases future benefits by enhancing future pest resistance. To my knowledge, this is the first time that such evidence of effective management of biodiversity productive capacity at the farm scale has been found. However, we cannot know whether this management operates due to crop rotation, which is the usual driver considered in the agricultural economics literature (Hennessy, 2006), as we can observe only acreage composition. In fact, the analysis of the management of crop rotation requires additional information on acreage configuration. Recent results on crop rotation management have found similar mechanisms, where farmers can adapt rotation to crop prices (Hendricks et al., 2014). Our results are, however, complementary to those on the links between biodiversity and farmers' insurance choices, where some evidence suggests that farmers manage biodiversity for its insurance value (Di Falco et al., 2014).

The work provides additional insights. First, our three empirical papers confirm that crop diversity increases crop production at the farm scale, but the results from Chapters 3 and 4 suggest that all crop yields are not affected in the same way. For example, we find that barley yields are more sensitive to crop biodiversity than wheat yields. This is a major finding, as recent studies still tend to aggregate all crops as a single production (Donfouet et al., 2017, Fontes and Groom, 2018). In Chapter 3, we also find that crop diversity positively affects milk production, suggesting a positive effect on forage yields, and in particular, that forage yields are more sensitive to crop biodiversity than cereal yields.

Second, the results of Chapter 3 highlight that permanent grasslands may increase cereal yields when crop biodiversity is small, confirming the potential productive spillovers from semi-natural areas (Klemick, 2011; Matsushita et al., 2017), which are still largely ignored in the economic literature.

Third, in addition to confirming that crop diversity increases crop yields, the work highlights the impact of crop diversity on variable input savings. As for yields, variable input savings due to biodiversity productive capacity depend on the considered crop. In particular, rapeseed is identified as the most sensitive crop: the average farm in term of crop biodiversity applies 27.5% less pesticide on rapeseed than a (theoretical) farm in rapeseed monoculture. As rapeseed is also one of the most pesticide-intensive crops, the management of biodiversity productive capacity is an interesting strategy for increasing profit. Chapter 3 also highlights that both crop diversity and permanent grasslands were substitute inputs for pesticides and fertilizers, suggesting that semi-natural areas could also lead to variable input savings. The elasticities of yields, input application and gross margins with respect to biodiversity from Chapter 4 suggest that most of the additional gains were due to additional yields (see barley).

Fourth, we find in Chapter 5 that, while crop diversity at the farm scale does increase farm profits, crop diversity at the landscape scale (1 km^2) decreases total profits at the landscape scale. Such results could reflect some aggregation effects, the spatial dimension of the site affecting ES measures (Mitchell et al., 2015). Our results underline the need to conduct further research on these spatial dimension effects, as the literature that measures the productivity of biodiversity from the farm to the regional scale (Bellora et al., 2017; Di Falco and Chavas, 2008, 2009; Donfouet et al., 2017) does not mention these effects.

Fifth, Chapter 5 highlights that the coordinated management of carabid beetles at the landscape scale leads to a 4% increase in total profit, in line with Cong et al. (2014). The gains arising from the individual management compared to the absence of management were 12%, suggesting that farmers managing acreage on their own already capture most of the gains from biological control. The presented results, however, highlight that farmers generate productive externalities for each other when managing biodiversity at the farm scale, illustrating that the acreage choice literature relies on restrictive assumptions of independence between farms when analysing farmers' choices at the farm scale. The obtained results suggest that farms are not independent but are connected by the ecosystem functioning, which provides productive ES that spills over across neighbouring farms. This could imply that previous estimations, including ours in Chapters 3 and 4, are biased and that ex ante policy evaluations were subject to some slipups. Overall, we find that the consideration of heterogeneous agents and realistic landscapes casts doubts on the emergence of coordination on real landscapes. Indeed, we find that landscape-scale management increases all individual profits in only 15% of the cases, whereas previous studies with homogeneous agents identified that coordination would be beneficial in most cases.

6.2 Research limitations

While our results provide new insights, they are subject to some limitations. First, our results are derived from a model where farmers are fully rational and have a full knowledge of the ecosystem functionalities at stake. This is obviously false, ecologists themselves still having many uncertainties in describing and understanding the ecosystem functioning (Hooper et al., 2005). In addition, the management of the ecosystem requires labour- and knowledge-intensive work, inducing some practices that are largely ignored here (Ellis, 2018).

Second, our empirical works use biodiversity indicators based on acreage composition to inform the level of biodiversity, which leads to obvious simplifications and approximations, as these indicators ignore the diversity of contexts and the associated ES variability. For example, the same measure of biodiversity could indicate different quality of an ecosystem under different meteorological conditions (Di Falco and Chavas, 2008). The utilization of these indicators also requires the assumption that the same level of indicator informs similar bundles of ES, whereas some ES could be substituted to each other depending on the context (Müller et al., 2016). While we have relied on the usual Shannon and share indicators from landscape ecology, we have computed them at the farm scale, whereas the landscape ecology literature considers that they have meaning at the landscape scale. This limitation is common to most economic papers on the productivity of biodiversity, but the results from Chapter 5 highlight that this issue can matter. One must also remember that we examine the productive ES emerging at either the farm or the landscape scales but that we ignore the productive ES managed at the field scale, such as productive ES attached to soil (Barbier, 1990; Hediger, 2003; Issanchou et al., 2018). Crop rotation is one example of a practice to manage productive ES attached to soil, which has been ignored in this thesis. Indeed, even if crop rotation is correlated with crop diversity at the farm scale (Thomas et al., 2003), the two practices are different.

Chapter 5 also highlights that the configuration of the farm territory influences the degree of the possible management of biodiversity productive capacity. Such information is, however, not available on the usual economic dataset, such that we needed to develop a simulation modelling framework in Chapter 5 to investigate the configuration issue. Lamy et al. (2016) explained, however, that landscape composition is a more important driver of ES provision than landscape configuration.

Chapter 5 uses a detailed ecological function, which ensures that the measured biodiversity is not correlated with other non-observed factors, such as economic confounders. While we have tried to capture all these unobserved effects using available information on fixed inputs and climatic and topographic conditions, other potential joint production processes (see Chapter 3 and the manure issue) and confounders could bias our estimations. Unobservable conditions should, however, generate an interplay between farmer heterogeneity and the ecological factors that influence the incentives to manage biodiversity; e.g., pest control strategies are adopted first in regions with high pest pressure. This unobservable heterogeneity could bias our estimated parameters. This is a classical issue in agricultural economics because industry statistics for agriculture do not report all necessary information, notably with regard to the analytical accounting of inputs, except for land and some product-specific inputs, such as livestock. That is why we have used the analytical accounting dataset from “La Meuse”, but the issue still remains for fixed inputs, which prevents us from performing more analysis on the role of fixed inputs for managing biodiversity. We could have used alternative econometric methods to account for this unobservable heterogeneity, such as techniques based on random parameters, which have been proven to improve the estimation of the parameters (Koutchadé et al., 2018), but such routines still do not exist and would therefore require intensive coding. Alternative econometric techniques, such as nonparametric techniques, could also provide additional information on the effects of biodiversity on crop yields and input application, as nonparametric methods provide useful information on non-linear effects (e.g., Fontes and Groom, 2018).

While we find strong evidence of the benefits from crop biodiversity, our results provide limited insights into the cost of biodiversity management by farmers. Chapter 2 suggested that these costs are linked to the management of fixed costs, namely, labour and capital. However, the identification strategy in Chapter 4 requires that we introduce the land constraint in the estimated equations, with the consequence that we are unable in practice to identify the different crop-specific costs attached to fixed input management. We identified, however, that the function of parameters $\left(\varphi_{km} - \varphi_{kK} - \varphi_{Km} + \varphi_{KK}\right)$ is opposite when we introduce the biodiversity productive capacity effects, suggesting that farmers' management of fixed inputs does not present incentives for diversification but rather for specialization, which conforms with the results on economies of scale due to capital (Lansink and Stefanou, 2001). Labour is usually considered a polyvalent public factor, whereas equipment is more often considered a specialized public factor inducing higher return to scale than labour (Dupraz, 1996). Our results may thus suggest that capital leads to higher marginal biodiversity management costs. The management of biodiversity also incurred other costs that we have ignored, such as knowledge costs. Landscape-scale coordination also includes transaction costs (Banerjee et al., 2017), which have been ignored here and would decrease the coordination gains.

Finally, we must recall that our results should be considered valid locally and *ceteris paribus*. The real parameters may be different in the case of a real agroecological transition, where farmers could operate in different technological zones. The analysis of the robustness of our results on specialized ES-oriented farms, such as organic farms, would have indicated the validity of our estimations.

6.3 Policy assessment

The first part of the thesis highlights that the marginal cost of providing biodiversity depends in a complex way on biodiversity and ecosystem functioning and that it influences the application of other polluting inputs with public good characteristics. Our results emphasize the incentives encountered by profit-maximizing farmers managing biodiversity and suggest that there are no conflicts between high yields and biodiversity.

We find that variable inputs and biodiversity productive capacity are substitutes. As pesticide prices are relatively low, farmers have no particular incentives to substitute pesticides with biodiversity. Such substitution could arise in the context of policy interventions. In particular, we find in Chapter 4 that the consideration of biodiversity productive capacity as an additional input leads to higher efficiency of an ad valorem pesticide tax than previously estimated (e.g., Femenia and Letort, 2016). This is particularly the case for rapeseed, where pesticide application decreases by 10% more than previously estimated. This highlights that the previously ex ante estimations provided biased evaluations of such policy, farmers having indeed potential to adapt. Such a policy instrument aiming to reduce pesticide application is a recurrent objective of French governments, notably the present one, which went further than the European Commission on the Glyphosate issue with its planned ban in 2020. The results from Chapters 3 and 4 on the productive interactions between biodiversity and chemical inputs indicate that such policies would also provide incentives to farmers to increase biodiversity levels. This illustrates that environmental policies could reach several objectives simultaneously. Similarly, our results indicate that subsidies targeting biodiversity and biodiversity habitats should also encourage farmers to reduce the application of fertilizers and pesticides. Some of the existing agro-environmental measures defined in the Common Agricultural Policy (CAP) may thus also affect chemical applications.

The CAP also conditions one-third of the decoupled production payments from the first pillar, the so-called green payments, with respect to some environmental constraints, notably with regard to the maintenance of minimum levels of crop diversity, habitat friendly landscape features and permanent grasslands. However, the results from Chapter 3 on the productive interactions between crop diversity and permanent grasslands emphasize the difficulty of designing optimal sets of policy instruments targeting crop diversity and permanent grasslands at the same time, as farmers have incentives to enhance crop diversity when permanent grasslands decrease and *vice versa*. Thus, green payments may lead to counterintuitive acreage evolution. For example, the introduction of green payments encourages crop-oriented regions to enhance ecological focus areas and permanent grasslands; this, in turn, leads to a decrease in the marginal productivity of crop diversity and finally to a reduction of crop diversity. We thus find that environmental policies cannot always reach several objectives simultaneously.[66] This general result is in line with the Tinbergen principle of "one objective, one instrument" (Tinbergen, 1952). Such policies may, however, stimulate price-induced innovation (Caputo and Paris, 2013) and increase the diffusion of agroecological techniques (Rollins et al., 2017).

Chapters 2 and 4 suggest that biodiversity productive capacity generates costs to the farmers due to the management of quasi-fixed inputs, such as labour and physical equipment, including buildings and machinery. As already explained, our results from Chapter 4 may suggest that capital leads to higher marginal biodiversity management costs. However, public agricultural policy has largely supported physical investments with several mechanisms allowing for tax exemption (at least in France): e.g., benefit threshold for benefit taxes, investment deduction (French DPI: "Deduction Pour Investissement"), and European subsidies for investments. This leads to specific public support for capital in agriculture, to the detriment of labour (Manuelli and Seshadri, 2014). Public support for capital in agriculture is notably supposed to maintain French farms' competitiveness. Our results, however, underline that the reduction of policy support for capital could provide incentives to increase biodiversity productive capacity and thus increase the provision of joint environmental services. Alternatively, specific instruments to reduce the costs of labour in agriculture can increase the incentives to enhance biodiversity productive capacity. In particular, policymakers should subsidize biodiversity-specific labour and capital when possible. The identification of such environmentally friendly input and/or practices may be one of the ambitions of the next CAP, which would require the identification of suitable indicators (see COM(2018) 392).

The results from Chapter 5 indicate mixed evidence regarding policy recommendations for the configuration of farm territories and landscape. Indeed, French farm territories are rather fragmented, notably in municipalities without previous land consolidation programmes (Latruffe and Piet, 2014). One could argue that new land consolidation programmes could favour the concentration of farm territories, which would improve the manageability of productive ES. However, as already stated, the management of productive ES at the farm scale on existing farm territories captures most of the potential benefits of such management at the landscape scale. In addition, existing experiments on land consolidation suggest that farmers enlarge the fields to benefit from economies of scale due to equipment, removing existing boundaries (notably hedgerows) and decreasing on-farm crop biodiversity (Di Falco et al.,

[66] We also find, like Femenia and Letort (2016), that pesticides and fertilizers are substitute inputs. This indicates that a pesticide taxation scheme would lead to additional fertilizer application, decreasing the efficiency of the tax.

2010). Such a land consolidation programme in dairy areas could, however, benefit the conservation of permanent grasslands because the concentration of fields around the farmstead would decrease transport costs and thus increase the profitability of grazing compared to forage crops (Dumont et al., 2016).

6.4 Future research

Our framework could be improved to address related questions in future research. First, I have considered only the productive value of biodiversity, without any references to the insurance value of biodiversity. Most of the literature on crop biodiversity focuses on the effects of biodiversity on the variance of yields, production or profits (Baumgärtner and Quaas, 2010; Di Falco and Chavas, 2009; Finger and Buchmann, 2015). There are some evidences that pesticide applications and financial insurance are substitute with crop biodiversity for risk management (Di Falco and Chavas, 2006; Di Falco et al., 2014). The obtained results underestimate the beneficial effects of crop biodiversity on farms. The proposed structural model could be used to investigate how farmers manage such second-order effects because risk motive and portfolio strategy are some of the usual motives in acreage literature (Chavas and Holt, 1990). However, the management of risk-reducing input, such as biodiversity or pesticide, is usually considered to be more related to the productive effects on average yields; e.g., at most, only 15% of pesticide applications are explained by self-insurance behaviour (Carpentier, 1995).

Second, there is much literature on scope and scale economies in multi-output farms based on the estimations of reduced form equations (Ang and Kerstens, 2017; Blancard et al., 2011, 2016; Chavas and Aliber, 1993; Chavas and Kim, 2010; Kim et al., 2012), with remaining debates on the gains between specialization and diversification. These debates arise notably because the measures capture the different benefits or costs of diversification in terms of single parameters, which prevents any discussion on the underlying economic, technical and ecological processes. Our framework could contribute to this debate: it disentangles several processes, notably between the management of productive ES and fixed inputs, which could both generate scope economies. This improvement requires the utilization of more detailed information on farm-fixed inputs.

Third, while our results provide new insights into several productive effects of several types of biodiversity and ecosystem components, there are still many possible improvements. One improvement would be to measure and not only simulate the productive spillovers from one farm to another, notably those arising from semi-natural elements (such as permanent grasslands), as the economic literature remains focused on crops. This could be possible by matching economic datasets, such as the farm accounting database network (FADN) and the land parcel identification system (LPIS) or Corine land cover. Such matching could appear at different scales and allow us to investigate the spatial perimeter of such productive effects according to the considered biodiversity components. Other information could be obtained by examining temporal and dynamic specificities of the biodiversity productive capacities, which may differ according to the considered biodiversity components as well as external drivers, such as climatic and topological conditions (Di Falco and Chavas, 2008; Di Falco and Zoupanidou, 2017; Donfouet et al., 2017). The introduction of an effective measure of biodiversity, such as species abundance measures, instead of our biodiversity indicator in agricultural technologies could also provide detailed information on the different productive ES and on the potential competition between species behind these ES (Mouysset et al., 2014).

Finally, one could consider that the provided results contribute to the emerging knowledge on agroecological transitions, which should be deepened. Indeed, agroecological practices are knowledge intensive (Rollins et al., 2017). Such knowledge should be produced by the research, which exhibits high returns to scale, amounting to 30% in the agronomic research for example, but takes between 15 to 30 years to be applied (Chavas et al., 1997). There is currently urgent need to develop this research.

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PART TWO:

ECOSYSTEM SERVICES, AN ENVIRONMENTAL ECONOMICS APPROACH

The first part of the PhD analysed the demand for productive ES by the farmers themselves, providing new insights into the specificities of the supply of environmental services provided by farmers. As stated previously, our results could contribute to improving the design of agro-environmental policy. However, efficient agro-environmental policy also depends on the demand of other agents for the jointly produced public goods (PGs) generated by the environmental services. As a reminder, like Engel et al. (2008), we consider an "environmental service" as corresponding to an agent's action that modifies the ES flows. We also consider the value of this environmental service as dependent on the variation in the stocks of public goods induced by these modifications of the ES flows.

The aim of this second part is to examine the impact of the geographical scale of the PG demand on the design of agro-environmental policy, considering that farmers jointly generate local and global PGs. In Chapter 7, I provide a global assessment of the marketed and environmental effects of French pesticide regulations. In particular, I quantify the impacts of these regulations in terms of (i) fertilizer and pesticide applications, which lead to numerous forms of local pollution and (ii) carbon emissions, which are a threat to climate stability. In Chapter 8, using a spatial hedonic approach, I value the environmental services provided by the farmers at different geographical scales. In Chapter 9, I theoretically analyse the effectiveness of the decentralization of the design of agro-environmental subsidies according to the different levels of information obtained by the hierarchical governments on the value of local and global public goods. I parameterize the theoretical model to the case of wetland management in Brittany (France) based on the results of European project H2020 PROVIDE, reported in the fourth and fifth work packages (WP4 and WP5). Chapter 10 is a general discussion of the results of Chapters 7 to 10.

CHAPTER 7. SIMULATING THE MARKET AND ENVIRONMENTAL IMPACTS OF FRENCH PESTICIDE POLICIES: A MACROECONOMIC ASSESSMENT [67]

In this chapter, we perform a macroeconomic assessment to examine the impact of a 50% *ad valorem* pesticide tax in France on French pesticide and fertilizer applications as well as carbon emissions worldwide. Pesticides and fertilizers are sources of local pollution of water, air and soil, which are local public goods (PGs). Carbon emissions are a major driver of climate change, and climate stability is a global PG. Incentive instruments that can reduce chemical input applications or maintain land-use practices suitable for the preservation of these PGs are debated in both the political and societal spheres. Do such pollution reductions constitute environmental services? This theoretical issue is not discussed here. Rather, we investigate how policy instruments lead to unexpected induced effects due to trade. We examine this question by estimating the effects of a pesticide tax on French pesticide and fertilizer applications and assessing the induced carbon emissions in tier countries due to land-use changes. To that end, we estimate a structural model constituted of output supply, variable input demand and acreage functions derived from Carpentier and Letort (2014) on all French farm-regions and all farm activities. This model can be seen as a simpler model than those estimated in the first part of the PhD. The originality of this first step is that such estimations are usually performed on crop-oriented farms and at the farm level, a practice that has prevented the assessment of pesticide and fertilizer applications at the national scale. We then introduce the estimated parameters and specifications into the computable general equilibrium GTAP-Agr framework, which pays close attention to the representation of land uses. An exogenous pesticide tax is simulated, as is a more technical scenario where pesticide productivity is improved. Such scenarios influence French farm production, which influences the prices of agricultural goods, thus generating new incentives for foreign farmers who could use non-agricultural land for agricultural production. We do find that French policy influences both local and global PG. In particular, a pesticide tax in France would decrease French pesticide applications but would also increase global carbon emissions due to LUC (notably deforestation). Our results reveal a trade-off between local and global PG provision. For the French government, the optimal policy would thus depend on the French demand for these different PGs.

[67] This chapter is coauthored with Alexandre Gohin (INRA, SMART-LERECO)

7.1 Introduction

Over the last century, global food production has increased faster than the wealthier population, improving global food security. Enhanced crop protection has led to a massive increase in realized crop yields, limiting the expansion of arable lands and deforestation. Before, protecting crops against pests and weeds mostly involved the management of their natural enemies, a technique known as biological control, and some labour-intensive techniques such as weeding and tilling. The application of chemical products started in the 19th century with the utilization of copper on vineyards and potatoes to protect the crops from fungi damage. The utilization of synthetic products appeared at the beginning of the 20th century, starting with the commercialization of dichlorodiphenyltrichloroethane (better known as "DDT"). In the last century, increasingly complex synthetic pesticides were introduced. At the same time, new agronomic techniques and farm machines enhanced the application of these new products and saved professional farmers, as well as leisure gardeners, from painful labour.

However, societal concerns regarding the health and environmental impacts of pesticides have increased in recent decades, particularly for synthetic pesticides. Scientific evidence has accumulated indicating that significant exposure to these pesticides directly influences farmers' health and can cause cancer or chronic diseases such as Parkinson's disease (Alavanja et al., 2003; Multigner et al., 2010; Betarbet et al., 2000). Currently, intense scientific debates examine the indirect effects of pesticides on the health of food and water consumers. These debates focus specifically on the allowable concentration levels of individual molecules and on their interactions. In regard to the environment, pesticide residues unambiguously pollute water and soil resources. The impact of pesticides on biodiversity is more debated because pesticide use is correlated with landscape simplification, which reduces the habitats of biodiversity (Butchart et al., 2010). However, pesticides are suspected to be major contributors to losses of biodiversity, notably for common birds and aquatic invertebrates (Beketov et al., 2013).

These societal concerns call for public action. These concerns are addressed with different intensities and policy instruments across the world, ranging from command-and-control instruments (such as the ban on DDT adopted in the EU in the 70s) to market-based instruments (such as ad valorem taxes in Denmark). In this paper, we focus on the French case, which is characterized by significant pesticide use, a diversity of farm production and crop damage, a currently complex policy and many recent policy decisions. The current French pesticide policy is obviously consistent with the European policy that mainly defines authorized and banned pesticides. The French pesticide policy includes national bans in addition to European ones. In November 2017, the European Parliament and Council voted to reauthorize the use of synthetic pesticides with glyphosate for a period of 5 years. Similar to a few European countries, policy makers in France are considering the possibility of banning glyphosate in 2021 for farmers and have already voted to ban these pesticides for public use and private gardeners by 2019. The French policy goes further and includes some specific taxes for farmers who use synthetic pesticides, depending on their toxicity. This policy also includes research efforts to develop alternatives; these efforts were significantly increased with the Ecophyto 1 plan, which was implemented in 2008. Finally, the most recent pesticide reform that should be applied in 2021 includes new taxes for pesticide retailers unless pesticide retailers justify a decreasing of their sales.

Despite all the recent policy reforms, French pesticide policies regularly divide stakeholders, with environmental groups asking for more severe regulations and food and pesticide industries asking for the opposite. For French policy makers, defining the optimal pesticide policy is not straightforward due to scientific uncertainties regarding the health and environmental impacts and due to the multiple known, but imperfectly measured, trade-offs.

First, the optimal pesticide policy must obviously balance environmental and economic objectives. In the recent glyphosate debate, French farmers and pesticide lobbies stress that the banning of this herbicide will decrease their crop yields and increase their production costs, mostly due to the additional mechanical control of weeds that would become necessary. The income of the French farm sector would significantly decrease (estimates by Concorde 2017 and Ipsos 2017 vary between 1 and 2 billion euros; the average income of this sector in the last 5 years was approximately 13 billion euros). These results rely on the crucial assumption that farmers are technically and economically efficient, applying pesticides due to their marginal productivity and prices relative to crop prices. These results are based on the short-term view of fixed technologies and crop allocations. By contrast, other French scientific studies find that the total farm use of pesticides (including glyphosate and all other pesticides) can significantly decrease without reducing farmers' incomes (by 30% according to Jacquet et al., 2011, Boussemart et al., 2011, and Lechenet et al., 2014). These contradictory results rely on the crucial opposite assumption that some farmers are technically or economically inefficient. These studies also consider a larger set of alternatives to pesticides rather than solely considering mechanical control, including integrated cropping techniques and new crop allocations. These last studies are thus more relevant in the long run because it is well known that economic agents have more flexibility in addressing new constraints, as illustrated by Femenia and Letort (2016). French policy makers are thus currently informed by contradictory studies on the inevitable tension between farm competitiveness and pesticide use.

Second, French policy makers also have to manage the conflicts between different environmental objectives. The recent glyphosate debate again nicely illustrates some of these trade-offs. The same farm and pesticides lobbies stress that banning this synthetic pesticide will have a negative climate change impact by inducing farmers to manage weeds mechanically, which would contribute to more energy use and hence increase direct carbon emissions. Moreover, less carbon would be stored in the soil. By contrast, environmental groups suggest that banning glyphosate would not increase net carbon emissions if production systems are modified, for example, by developing associated crops to control weeds (Generation futures, 2017). The conflict between environmental objectives is however much more complex than these first ones. Some studies (such as Bareille and Letort, 2018) find that there are some substitutions between pesticides and mineral fertilizers for some crops and farmers, implying that, *ceteris paribus*, a constraint on pesticide use will increase fertilizer use, which may subsequently increase nitrogen pollution in waterways. French policy makers are well aware of this potential tension between the pollutions induced by the use of pesticides and fertilizers but lack of numerous scientific evidences. Moreover, stricter French regulation on pesticides may reduce overall French farm production, which may be partially compensated by increased imports depending on trade regulations. These imports may come from countries using relatively more pesticides than French producers and may also induce land use changes and related changes in carbon emissions in these countries. These “leakage” effects are well known in the climate change literature, as well as in the more recent biofuel issue (Searchinger et al., 2008). The quantification of land use changes induced by the use of European biofuels has recently been an intense empirical issue. These land use changes are not directly measured; instead, they are counterfactually simulated with market equilibrium models. These models are based on uncertain parameters, such as the reactions of agents to economic incentives (price and income elasticities), contributing to empirical contradictions. The existence of such leakage effects is now recognized in all French agri-environmental policy debates as the notion of imported deforestation. Again, empirical studies measuring these trade-offs are currently missing.

In this complex context characterized by many trade-offs and uncertainties, French policy makers and more generally, the French society at large, have highlighted the need for transparent scientific results to guide their decisions and positions. Numerous synthetic reports have been published by French/European/world health and environmental agencies in recent years. However, the different economic and environmental trade-offs just mentioned are not simultaneously addressed and quantified (Reboux et al., 2017).

Our main objective in this paper is to partially fill this gap by offering a macroeconomic quantification of some of the economic and environmental impacts of two contrasted French pesticide policies. The first simple but radical policy scenario is the implementation of significant pesticide taxes similar to those implemented in a few other countries and those often suggested in the academic economic literature (Carpentier et al., 2010). Hereafter, we refer to this first scenario as the tax scenario. The second contemplated policy scenario is more in the spirit of the recent reforms, and hereafter, we refer to it as the technological scenario. The latter favours the adoption of potentially new pesticide-saving technologies by boosting public/private researches and disseminations of their results to farmers. In other words, we clearly define two very contrasted and stylized policy scenarios because we assume that a policy-induced technical change occurs in the second scenario, while there is no price-induced technical change in the first scenario. Note that we are not looking for the optimal French pesticide policy but more modestly measure some economic and environmental trade-offs that such a policy must address.

For this purpose, we develop an original methodology with three distinctive features. First, we perform econometric estimations to identify the economic behaviour of French farmers regarding their use of pesticides and fertilizers and how they choose their acreage. In this way, we can avoid any assumptions regarding whether they are technically or economically efficient or not. Second, we introduce all farm activities, including the often-neglected fodder crops consumed by livestock sectors. These first two distinctive features rely on the often-overlooked regional agricultural economic accounts. These yearly accounts include data from 1990 to the present, are publicly available and cover all farm activities. We develop generalized maximum entropy procedures to address the limited number of observations. This database does not separate the different types of farm technologies and pesticides but aggregates the synthetic and chemical pesticides used by both conventional and organic farmers. Our macroeconomic assessment is thus complement to microeconometric analyses performed with databases covering particular farm, technologies and/or pesticides. We find a large number of statistically significant price coefficients; hence, farmers' use of pesticides depends on prices. We find that the French price elasticity of pesticide use amounts to -0.8, which is higher but consistent than other available microeconometric estimates. We also find significant variations in elasticities among activities, such as lower responses for cereals than vineyards, livestock and vegetable elasticities, and among French regions.

Our third distinctive feature is the simulation of some of the economic and environmental impacts of our two scenarios at the world level. We develop an original computable general equilibrium (CGE) framework, which is based on the standard global trade analysis project (GTAP)-Agr model (Keeney and Hertel, 2005). This model, which is based on the GTAP database, does not isolate pesticides from other chemical products such as mineral fertilizers. We thus improve the representation of the French economy by specifying the particular role of pesticides and mineral fertilizers used by French farmers and by introducing the previously estimated elasticities. This CGE framework allows us to simultaneously measure the impacts of our two scenarios on global economic indicators and the pesticide use of French farmers. We also measure the global net carbon emissions by taking into account the indirect effects occurring through market reorganization, induced by the livestock sectors for example. Ultimately, we provide some rudimentary estimates regarding the evolution of the nitrogen surplus in France. We find, as expected, that the tax scenario has a negative economic impact on the French farm and food processing sectors and leads to a reduction in their pesticide use. Reduced French production is partly compensated by increased imports, benefiting, in particular, Latin American producers. We obtain a meaningful reduction in French livestock production, which does not compensate for changes in global carbon emissions induced by global land use changes. We also obtain a higher French nitrogen surplus as cereal yields and exports contract much more than French livestock production. On the other hand, our technological scenario leads to very small crop market effects, reduces the application of pesticides and increases French economic indicators. Interestingly, we also find that all of our environmental trade-offs are solved, which is partly explained by increased French production of protein crops and reduced imports of these products. Finally, this scenario quantifies some of the economic benefits of R&D efforts.

Below, section two details our econometric efforts. Our simulated policy scenarios are analysed in section three. The last section concludes with some policy and research recommendations.

7.2 Econometric identification of French farmers' behaviour

The effectiveness of any pesticide policy partly depends on the behaviour of farmers. Some studies (such as Concorde 2017 and Jacquet et al. 2011) postulate the behaviour of farmers and then perform policy simulations with calibrated models. By contrast, many other studies analyse the behaviour of farmers with statistical techniques. The main results of current econometric studies are summarized in Skevas et al. (2013) and Bocker et Finger (2017). These scholars find some consistent results across studies such as higher price responses in the long run (compared to the short run) or at the aggregate level (compared to the individual level). However, some conflicting results remain, such as the overuse vs underuse of pesticides by farmers or the exact levels of the price responses for different pesticides and crops. These conflicting results can be partly explained by the datasets, statistical procedures and economic specifications used in these studies.

The economic specifications can be separated into three groups. The first group uses a production function approach where technological relationships are statistically estimated (recent French applications include Boussemart et al., 2013; Desbois et al. 2016, and Urruty et al., 2015). One critical challenge of this approach is controlling for the potential endogeneity of the explanatory variables (Griliches, 1957; Griliches and Mairesse, 1995), the results being often sensitive to the choice of instrumental variables. The second group relies on duality theory to directly estimate price elasticities (one recent French application is found in Fadhuile et al., 2016). These studies usually do not identify the underlying technological relationships and consider a limited set of decision variables (for example, focusing only on pesticide application without considering the use of fertilizers, cropping practices, and acreage decisions). The third group can be presented as a mix of the two previous groups with the explicit representation of some technological relationships and the explicit specification of exogenous price incentives on many interrelated decision variables (such as variable input applications and acreage choices). Carpentier and Letort (2012, 2014) explain the virtues of their structural approach and Femenia and Letort (2016) provide a French application that focuses on pesticides. Their dataset is limited to individual cereal producers located in the French department La Meuse and covers a limited number of years (2007-2012). We elaborate on this approach and apply it to a larger (but less detailed) dataset. We implement this specification in both this statistical section and for the policy CGE simulation in the next section.

7.2.1 Economic specifications

We consider a multi-output farm r maximizing its restricted profit $\Pi_{r,t}$ each year. The modelled decision variables are the annual application of the variable inputs on each output and the acreage choices of some annual crops. The maximization programme is subject to the expected output and input prices, the level of fixed factors, technological possibilities and regulatory constraints. The yields are assumed to be crop-specific quadratic functions depending on the variable input applications with constant returns to acreage. Compared to the often-used damage control function, this quadratic function does not impose rigid separability of the variable inputs (Carpentier and Weaver, 1997). The restricted profit function is defined as the sum of the gross margins per hectare $\pi_{k,r,t}$ for each output k multiplied by the respective acreage minus a cost function $C\left(\mathbf{S}_{r,t};\bar{\mathbf{S}}_{r,t},\mathbf{Z}_{r,t}\right)$ depending on the acreage allocation of endogenous areas $\mathbf{S}_{r,t}$. This cost function captures all the constraints and motives for crop diversification. These constraints can be due to the management of fixed inputs (capital and labour, $\mathbf{Z}_{r,t}$) at the farm scale, decreasing returns to scale, crop rotations or risk diversification motives. This function ensures the convexity of the profit function, allowing the determination of the optimal acreage.

Formally, the maximization programme can be solved in two steps. In the first step, we solve for the optimal application of the variable inputs for each crop. In the second step, we solve for the optimal acreage choices. The first programme is given by:

$$\pi_{k,r,t} = \max_{\mathbf{x}_{k,r,t}} \left\{ \begin{array}{l} p_{k,r,t-1} y_{k,r,t} - \sum_{i=1}^{I} w_{i,r,t} x_{i,k,r,t} \\ \text{s.t.}\, y_{k,r,t} = f_{k,t}\left(\mathbf{x}_{k,r,t}\right) \end{array} \right\} \tag{7.1}$$

where $x_{i,k,r,t}$ is the quantity of the variable input i applied to one hectare of area k, $w_{i,r,t}$ is its price, $p_{k,r,t}$ is the price of output k and $y_{k,r,t}$ is the yield of output k. The yield is equal to a function of the variable input application $f_{k,t}(\cdot)$. Note that we assume that farmers have naïve anticipation for output prices but perfect anticipation for input prices. This assumption is common in most agricultural economics works with short-term profit-maximization problems. Indeed, French farmers sow their land a few weeks after the harvest of campaign $t-1$ without knowing the output prices of campaign t, but pesticides and fertilizers are used during the spring of campaign t. The dynamic process of plant growth is such that most authors consider these form of anticipations. Formally, the production function is given by:

$$f_{k,t}\left(\mathbf{x}_{k,r,t}\right)=\alpha_{k,r}+\alpha_{t,k,r}t-\frac{1}{2}\left(\beta_{1,k,r}^{-1}\left(b_{1,k,r}-x_{1,k,r,t}\right)+\beta_{2,k,r}^{-1}\left(b_{2,k,r}-x_{2,k,r,t}\right)\right) \tag{7.2}$$

This quadratic production function includes easily interpretable parameters (Just and Pope, 2003). The parameters $\alpha_{k,r}$ and $\alpha_{t,k,r}$ represent the maximal yields of output k that depend on time (represented by a trend t), and $\alpha_{t,k,r}$ represents technical progress. The parameters $b_{1,k,r}$ and $b_{2,k,r}$ represent the maximum required variable inputs to reach the maximal yields. The parameters $\beta_{1,k,r}$ and $\beta_{2,k,r}$ represent the responses of the yields to variable inputs and are directly related to the price responses (see below). We consider only two variable inputs: pesticides ($i=1$) and fertilizers ($i=2$). We omit second-order interactions between the two variable inputs due to multicollinearity issues in our dataset.

The resolution of (7.1) with (7.2) leads to the following functions:

$$x_{i,k,r,t}=b_{i,k,r}-w_{i,k,r,t}\,p_{k,r,t-1}^{-1}\beta_{i,k,r}^{-1} \tag{7.3}$$

and:

$$y_{k,r,t}=\alpha_{k,r}+\alpha_{k,r,t}-\sum\nolimits_{i=1}^{2}w_{i,k,r,t}^{2}\,p_{k,r,t-1}^{-2}\beta_{i,k,r}^{-1} \tag{7.4}$$

where (7.3) is the demand function of the variable inputs, and (7.4) is the crop yield function. The estimations of the parameters in (7.3) and (7.4) allow the determination of the optimal gross margins $\pi_{k,r,t}^{*}$ that are needed to determine the optimal acreage choices. Formally, the second programme is given by:

$$\begin{aligned}&\max_{\mathbf{S}_t}\Pi_{r,t}=\sum_{k=1}^{K}S_{k,r,t}\pi_{k,r,t}^{*}\left(\mathbf{x}_{k,r,t}\right)+\sum_{k=K+1}^{\bar{K}}\bar{S}_{k,r,t}\pi_{k,r,t}^{*}\left(\mathbf{x}_{k,r,t}\right)-C\left(\mathbf{S}_{r,t};\bar{\mathbf{S}}_{r,t},\mathbf{Z}_{r,t}\right)\\&\text{s.t.}\sum_{k=1}^{K}S_{k,r,t}+\sum_{k=K+1}^{\bar{K}}\bar{S}_{k,r,t}=UAA_{r,t}\end{aligned} \tag{7.5}$$

In the following, we consider that $\sum_{k=1}^{K} S_{k,r,t} = S_{tot,r,t}$, where $S_{tot,r,t}$ is the total area devoted to crops with an endogenous area in the considered region in t. For the cost function, we use a parsimonious entropic function:

$$C\left(\mathbf{S}_{r,t};\bar{\mathbf{S}}_{r,t},\mathbf{Z}_{r,t}\right) = A + \sum_{k=1}^{K} c_{k,r} S_{k,r,t} + a_r \sum_{k=1}^{K} S_{k,r,t} \ln\left(S_{k,r,t}\right) \tag{7.6}$$

The term A represents the fixed costs of the farm that do not depend on acreage choices. The vector of parameter $\mathbf{c}_r$ represents crop-specific costs that do not depend on variable inputs. The parameter a_r plays a key role in determining the optimal area. Indeed, by resolving (7.5), we obtain:

$$S_{k,r,t}^{*}\left(\pi_{k,r,t}^{*}\right) = S_{tot,r,t} \frac{\exp\left(a_r\left(\pi_{k,r,t}^{*} - c_{k,r,t}\right)\right)}{\sum_{l=1}^{K} \exp\left(a_r\left(\pi_{k,r,t}^{*} - c_{l,r,t}\right)\right)} \tag{7.7}$$

The optimal acreage of crop k depends positively on the total area of the endogenous crops and the gross margin of k but negatively on the gross margins of the other crops. In particular, an exogenous shock on input prices impacts acreage decisions. The expression of (7.7) in the logarithm leads to:

$$\ln\left(\frac{S_{k,r,t}^{*}}{S_{l,r,t}^{*}}\right) = a_r\left(\pi_{k,r,t}^{*} - \pi_{l,r,t}^{*}\right) - a_r\left(c_{k,r,t} - c_{l,r,t}\right) \tag{7.8}$$

Equation (7.8) shows that the evolution of the ratio of the optimal areas directly depends on the margin differences and the parameter a_r. If a_r is high, then the farmer can easily modify his/her optimal acreage. If parameter a_r is null, then the areas are independent of the margins and thus independent of market prices.

The aim of using this statistical approach is estimating the deep parameters $\left(\boldsymbol{\alpha}_r, \boldsymbol{\beta}_r, \mathbf{b}_r, a_r, \mathbf{c}_r\right)$. In particular, the estimations of $\boldsymbol{\beta}_r$ allow the elasticities of yields and input demands regarding input and output prices to be determined, and a_r allows the elasticities of area regarding input and output prices to be determined.

7.2.2 Econometric procedures

Several issues prevent the direct estimations of the behavioural parameters. First, we do not observe crop-specific input demand but only the regional consumption of pesticides and fertilizers $\mathbf{X}_{r,t}$. This is a classical issue when estimating crop-specific input demand functions. We thus estimate:

$$X_{i,r,t} = \sum_{k=1}^{K} S_{k,r,t} \left(b_{i,k,r} - w_{i,k,r,t} p_{k,r,t-1}^{-1} \beta_{i,k,r}^{-1} \right) + \sum_{k=K+1}^{\bar{K}} \bar{S}_{k,r,t} \left(b_{i,k,r} - w_{i,k,r,t} p_{k,r,t-1}^{-1} \beta_{i,k,r}^{-1} \right) + \varepsilon_{i,r}^{X} \tag{7.9}$$

where $\varepsilon_{i,r}^{X}$ is the random term accounting for unobservable heterogeneity among farmers and stochastic events that can impact production.

Second, due to the total land constraint, the parameters $(a_r, \mathbf{c}_r)$ can only be determined if a reference crop is defined. Thus, we estimate only $K-1$ acreage equation functions such that:

$$\ln\left(\frac{S_{k,r,t}^{*}}{S_{K,r,t}^{*}} \right) = a_r \left(\hat{\pi}_{k,r,t}^{*} - \hat{\pi}_{K,r,t}^{*} \right) - a_r \left(c_{k,r,t} - c_{3,r,t} \right) + \varepsilon_{k,r,t}^{S} \tag{7.10}$$

where $\varepsilon_{k,r,t}^{S}$ is the random term accounting for unobservable heterogeneity.

In total, we estimate a system composed of $\bar{K}$ yield equations, 2 demand equations (for $i \in [1;2]$), and $K-1$ acreage equations. The crop yield equations are:

$$y_{k,r,t} = \alpha_{k,r} + \alpha_{k,r,t} t - \sum\nolimits_{i=1}^{2} w_{i,k,r,t}^{2} p_{k,r,t-1}^{-2} \beta_{i,k,r}^{-1} + \varepsilon_{k,r,t}^{y} \tag{7.11}$$

where $\varepsilon_{k,r,t}^{y}$ represents the error term. We estimate this system for each French region, assuming that the set of parameters is specific for each one. This decomposition also allows the error terms to be disentangled from the regional fixed effects.

We estimate our system of equations using the generalized maximum entropy (GME) method (Golan et al., 1996). Indeed, van Akkeren et al. (2002) show that the GME method has better finite-sample properties and is more robust regarding the distribution of errors than the usual method of moments.[68] In this more recent method, the estimated parameters are defined as the product of (endogenous) probabilities and (exogenous) support values. Assuming that the value of parameter θ_n ranges between $[z_{n1}, z_{nR}]$, the econometrician defines the set of support values $\mathbf{z}_n = [z_{n1}, z_{n2}, ..., z_{nR}]$ with the associated probability weights $\mathbf{P}_n = [P_{n1}, P_{n2}, ..., P_{nR}]$, where $P_{nr} \geq 0 \ \forall n \in [1; N]$ and $\forall r \in [1; R]$. Each parameter is defined as:

$$\theta_n = \sum_{r=1}^{R} z_{nr} P_{nr} \tag{7.12}$$

This optimal probability distribution maximizes the entropic criteria defined by:

$$H(\mathbf{P}) = -\sum_{m=1}^{m=M} P_m \ln(P_m) \tag{7.13}$$

In the GME method, the entropic criteria includes the probability distributions associated with both the deep parameters and error terms. Accordingly, this method avoids making assumptions regarding the specific distributions of these error terms. Tests can be performed using entropic ratio tests that are similar to the likelihood ratio test used in the maximum likelihood approach. Below, we use standard asymptotic results for statistical inference. It appears that we need to correct for the autocorrelation of the error terms.

The GME method has gained popularity in recent years, but similar to Bayesian econometrics, it remains sensitive to the determination of the support values. In alignment with most studies using GME, we consider only three support values for each parameter. Due to the agronomic interpretation of our parameters, we use some technical information to help us define the support values of some of the deep parameters. Specifically, we assume that the $\alpha_{k,r}$ parameter (maximum yield) represents between 50% and 150% of the observed maximal yield. We assume that the annual trend parameter $\alpha_{t,k,r}$ represents between -50% and 50% of the observed mean yield. The parameters $b_{1,k,r}$ and $b_{2,k,r}$ measure the variable inputs required to reach the maximum yields and are assumed to be between zero and 25% of the maximum observed crop receipts. For the crucial price response parameter $\boldsymbol{\beta}_r$, we rely on the values from prior studies to guide our support values. As seen from equation (7.4), these parameters are directly related

[68] Note that some studies on the estimation of pesticide demand have already used the GME method (e.g. Oude Lansink and Carpentier, 2001).

to the elasticities of the yields with respect to variable input prices. When defining the support values of these crucial parameters, we assume that the elasticities of these yields are negative and higher than -0.5. The robustness of our econometric results for these support values is reported in the Appendix. In regard to the other crucial parameter a_r governing acreage decisions, we again rely on the literature and assume that the own price elasticity of land use is positive and lower than 0.5. Finally, we assume large negative and positive support values for the crop-specific cost parameters $\mathbf{c}_r$.

7.2.3 Data and descriptive statistics

We use the agricultural economic accounts (AEA) of the 21 former metropolitan and continental French regions (all metropolitan regions except Corsica) between 1991 and 2015. Produced by the INSEE (the French Institute of Statistics), this database provides information on the different elements of agricultural incomes (production, sales, intermediate inputs, subsidies, wages, profits, etc.).[69] In addition to providing information that covers a relatively long period of time, this database provides information on the values of different fodders, which is usually unavailable in other farm datasets.

We distinguish five outputs (i.e., $\bar{K}=5$): cereals, industrial crops (mostly oilseeds and sugar beets), corn silage, other fodder (mostly from grasslands) and other crops. This last category is an aggregate of likely pesticide-intensive crops such vegetables, fruits and vineyards. We consider that the acreage of the first three outputs is determined each year by the farmers, while the last two types of land are more permanent crops. The acreage of these two last types of outputs is treated as exogenous in the estimation procedure. Table 7.1 provides the summary statistics for the 21 (number of regions)*25 (number of years) observations.

The statistics for these areas highlight that the most cultivated lands are those used for other fodders, even if there are large disparities among the regions (notably between the regions of the Paris Basin and the ones in the mountains where permanent grasslands represent the main agricultural area). Cereals are the second most cultivated lands. The statistics on variable input consumption confirm that the two most consumed variable inputs used for crop activities are pesticides and fertilizers (seed expenditures are much lower). The AEA database only reports the aggregated consumption of pesticides; therefore, we are not able to distinguish between the

[69] See Annequin et al. (2009) for details on this database.

different types of pesticides (insecticides, fungicides and herbicides) or between the different practices and their outputs (organic versus conventional farming). According to this database, pesticide applications have increased between 1991 and 2008 but have decreased since; 2015 levels are the same as 1991 levels. For this period, pesticide expenditures represent less than 8% of farmers' incomes. Pesticide prices are rather stable over the first 15 years, and they modestly increase in the last 10 years (possibly due to the banning of more synthetic pesticides).

Table 7.1. Descriptive statistics (N=525)

	Mean	**S.D.**	**Min**	**Max**
price index of pesticides (base 1990)	123.49	95.15	92.67	665.19
price index of fertilizers (base 1990)	142.94	62.37	86.35	531.10
value of pesticides (€)	123.81	73.52	10.60	347.63
value of fertilizers (€)	142.45	77.28	29.98	565.34
price of cereals (€)	137.61	36.29	78.63	288.61
price index of industrial crops (base 1990)	72.18	21.02	39.63	154.07
price index of maize fodder (base 1990)	115.76	42.23	49.69	349.44
price index of other fodder (base 1990)	118.80	41.03	56.88	331.03
price index of other crops (base 1990)	101.88	24.46	57.29	229.19
cereals area (1000 Ha)	433.48	275.19	73.67	1339.48
industrial crop area (1000 Ha)	140.85	125.22	4.38	529.39
maize forage area(1000 Ha)	71.25	85.42	1.35	384.42
other fodder areas (1000 Ha)	613.27	356.01	23.33	1365.90
other crops area (1000 Ha)	128.86	154.15	6.32	775.72
yields of cereals (tons/Ha)	6.77	1.41	2.86	10.73
yields of industrial crops (quantity index/Ha)	16.01	6.73	7.06	48.13
yields of maize forage (quantity index/Ha)	4.78	1.85	0.07	13.02
yields of other fodders (quantity index/Ha)	2.78	1.32	0.67	6.48
yields of other crops (quantity index/Ha)	85.62	54.74	18.67	295.10

7.2.4 Econometric results

For each region, we estimate 33 deep parameters. Table 7.2 reports the estimated deep parameters governing the biological/price responses to pesticides for all regions and outputs.

The estimated parameters for cereals, industrial crops and other crops are statistically significant in most regions, particularly in regions with mixed farms (e.g., Pays de le Loire). We also find that industrial crops are more price sensitive than cereals, which is consistent with Carpentier and Letort (2012). Corn silage and other fodder crops are less sensitive to pesticide prices, possibly because more complex crop rotations are implemented in the livestock farms. The absence of response by maize in some regions may also be explained by the development of hoeing techniques, which decrease the required pesticide levels. Crop farms have less freedom to implement such alternative techniques and rely more on pesticide application to manage plant health. Finally, fodder prices vary less than the prices of cereals and industrial crops, which makes it more difficult to statistically identify price responses.

Table 7.2. Estimated response parameters to pesticide prices by region and crop

	Cereals		Industrial crops		Corn silage		Other fodder		Other crops	
Ile de France	0.19		1.00	*	1.03		0.49		15.24	
Champagne Ardennes	0.55		1.41	*	1.11		0.30		11.79	
Picardie	0.39		2.06	**	0.80		1.09		7.89	**
Haute Normandie	0.60	*	1.04	**	0.62		0.42		9.82	**
Centre	0.22		0.53	*	0.55		0.34		5.38	
Basse Normandie	0.61		1.23	*	0.62		0.12		9.02	*
Bourgogne	0.38		0.32		1.19		0.26		7.52	
Nord pas de Calais	0.50	*	3.06	**	0.70		0.76		1.29	
Lorraine	0.03	**	0.06	**	0.08	*	0.00		1.29	*
Alsace	0.57	**	1.86		1.16		0.54		10.24	
Franche comté	0.31		0.42		1.31	*	0.30	**	18.69	
Pays de la Loire	0.48	**	0.73	**	0.55	*	0.17		5.92	**
Bretagne	0.17		0.90	**	0.60		0.27	*	4.08	*
Poitou Charentes	0.42	*	0.21		0.69		0.58	**	6.00	**
Aquitaine	0.93	**	0.37		0.51		0.35	**	2.21	**
Midi Pyrénées	0.31	*	0.15		0.76		0.36	**	2.87	*
Limousin	0.33	*	0.98		0.56	*	0.02		2.13	
Rhône Alpes	0.90	**	0.78		1.08		0.22	**	0.74	
Auvergne	0.47	**	0.47		1.01	*	0.00		7.20	*
Languedoc Roussillon	1.67	*	0.67	*	0.53		0.32	*	0.88	**
PACA	1.70	**	1.48		0.22		0.19		6.96	**

** and ** represent the 10% and 5% significance levels, respectively.*

We find that these estimated parameters are robust to the choices of the support values (see Tables 7.A1. and 7.A2. in the Appendix). The parameters for fertilizers are estimated with less precision, which is probably due to the substitution of chemical fertilizers with organic fertilizers. In regard to the other estimated parameters, we include a trend in the crop yield equations that proxies the effects of technical changes and climate effects. These trends are statistically positive for cereals and industrial crops, representing 0.5% and 0.8% of the annual growth, respectively. These parameters illustrate the gains obtained using the same levels of inputs and considering technical progress or meteorological conditions. These parameters are not significant for other crops and fodders, which is possibly due to decreased R&D efforts for these activities.

Table 7.3. Aggregated estimated elasticities for France

		Cereals	Industrial crops	Maize forage	Other fodders	Other crops	Aggregated
Yield elasticities	Output price	0.07	0.19	0.26	0.17	0.10	
	Pesticide price	-0.04	-0.10	-0.14	-0.09	-0.06	
	Fertilizer price	-0.04	-0.07	-0.11	-0.08	-0.03	
Input own-price elasticities and crop-specific consumption	Pesticide price	-0.34	-1.30	-2.71	-1.01	-0.99	-0.82
	Fertilizer price	-0.23	-0.44	-1.15	-0.54	-0.43	-0.39
	Pesticide repartition	0.13	0.13	0.04	0.04	0.66	
	Fertilizer repartition	0.14	0.15	0.05	0.04	0.63	
Acreage elasticities	Cereal price	0.07	-0.14	-0.14			
	Industrial crop price	-0.05	0.18	-0.04			
	Maize forage price	-0.01	-0.01	0.10			
	Pesticide price	-0.007	0.02	0.01			
	Fertilizer price	-0.01	0.03	0.02			

Table 7.3 reports the estimated elasticities at the national scale. The aggregated own-price elasticity of pesticide application is estimated to be -0.82 (and remains at -0.78 and -0.80 in the robustness checks when the support values are divided by two or multiplied by two for all the crops and regions). This value lies in the upper range of those found in the microeconometric literature and aligns with the utilization of aggregated data and the consideration of the diversity of agricultural outputs. We note that the latest microeconomic attempts in France find comparable elasticities (Fadhuile et al., 2016). Moreover, we find that the pesticide demand for cereals is more inelastic than for other crops (Table 7.3). Our estimated elasticity for cereals is indeed close to the median of previous estimations (Böcker and Finger, 2017) that usually focus on these outputs. We find higher own price elasticities for other categories, particularly for corn silage. Such high levels of elasticities have been estimated in the past for cereals and aggregated agricultural outputs (Carpentier and Weaver, 1997, Chambers and Lichtenberg, 1994, Chen et al., 1994), but they lie in the upper range of those found in the literature (Böcker and Finger, 2017). The literature rarely estimates pesticide elasticities for corn silage and other crops, which complicates the verification of our results. However, we recognize that the discussion on crop-specific elasticities may be complicated as the crop-specific parameter $b_{1,k,r}$ is not precisely estimated. Thus, our crop-specific input demands are not precisely estimated. The joint estimation of such parameters using the farm-scale equation (7.10) is always a tricky task (Carpentier and Letort, 2012) and even more so when there is a limited number of observations. We find that the other crops category represents the largest share of pesticide expenditures and the fodder crops the smallest share, which is consistent with the agronomic literature (Urruty et al., 2015, IONOSYS, 2016a, PEREL, 2015). Less consistent is our finding that the shares of

pesticide expenditures for cereals and industrial crops are similar, implying a per-hectare application for cereals that is too low. However, this does not raise any doubts regarding the sign and level of the aggregated elasticity, which is significantly different from 0 at the 5% level, or the fact that crop-specific pesticide demand is sensitive to pesticide prices (see Table 7.2).

We also compute the price elasticities of crop yields. Our estimated crop yield elasticities are consistent with the economic literature, with lower levels for cereals and higher levels for industrial crops. We find that the highest yield elasticities are for corn silage, which may indicate that a higher price in the previous period (i.e., the anticipated price is higher) corresponds to a lack of fodder for livestock feeding. Finally, we find that the acreage elasticities are lower than the yield elasticities, which is consistent with Carpentier and Letort (2012). This result illustrates that it is more difficult for farmers to modify their acreage than to modify their practices at the intensive margins.

Overall, our econometric results show that crop and input prices influence farmers' decisions, which aligns with the assumption that regional farm optimize at the aggregate scale. Our results imply that a pesticide tax will effectively modify pesticide use, which is the aim of our tax scenario in the simulation exercise. We also find a significant positive yield trend for cereals and industrial crops, possibly capturing technical progress. In our second technological scenario, we explore the impacts of increasing R&D efforts to reduce pesticide use.

7.3 CGE policy simulations

All public policies have some direct and indirect effects on economic and environmental indicators. The indirect effects are generally more difficult to measure but may eventually counterbalance the direct ones, leading to complex policy debates. Global economic models are the inescapable tools for measuring these effects when considering "significant" public policies. Below, we elaborate on the GTAP-Agr framework, which has been utilized to assess the indirect effects of several agri-environmental policies, including those that affect the use of biofuel (Hertel et al., 2010), Genetically Modified Organisms (Mahaffey et al., 2016) and organic farming (Bellora et Bureau, 2016) and a ban on glyphosate (Brookes et al., 2017).

7.3.1 The starting GTAP-Agr framework

The GTAP-Agr framework is a comparative static CGE model accounting for a large diversity of goods produced by many sectors (Keeney and Hertel, 2005). This framework covers the world and considers the heterogeneity of climatic and topographic conditions, distinguishing between several agro-ecological zones within each country. The GTAP-Agr model distinguishes firms, which maximize their profits, and households, which maximize their utility. By default, this model assumes that economic agents are price takers. The GTAP-Agr model departs from a textbook CGE model mostly due to its rich specification of agricultural and food sectors and markets. Pervasive farm policies are also finely modelled; the specificities of farm production and food consumption are captured by nested structures of globally regular production/utility functions.

The GTAP-Agr model relies on the GTAP database, which compiles social accounting matrices for many countries. The quality of this database continuously improves and is beneficial for several types of global economic analysis (Corong et al., 2017). The last available database covers the economic flows of 2011. This GTAP database includes 20 agricultural and food products and explicitly considers land as a primary factor of production. This database is well suited for measuring carbon emissions linked to land use changes. The GTAP database also distinguishes energy and livestock products, which are responsible for some greenhouse gas (GHG) emissions.

7.3.2 The specifications of the French economy

The GTAP-Agr framework cannot be directly used to perform simulations of French pesticide policies, in particular because pesticides cannot be isolated in different products. One strategy consists of tailoring policy shocks to the model structure (for example, Brookes et al. introduce taxes on chemicals, labour, capital and land productivity shocks to assess the impacts of a glyphosate ban). This strategy is easy to implement in the CGE framework, but it does not explicitly reflect the response of economic agents to the policy. The second strategy consists of modifying the model structure, with product/factor disaggregation and economic specifications that differ by country (for example, Adams et al., 1997). We pursue this strategy by developing new specifications for the French economy inside the GTAP-Agr framework. We built a new social accounting matrix for the French economy using 2011 economic data. We start with the macroeconomic tables produced by the INSEE. Fortunately, French trade data are similar to the

GTAP-Agr trade data. Then, we analyse the farm and food sectors using additional statistical information provided by the French Ministry of Agriculture, including the agricultural economic accounts. Information on farmers' use of pesticides is obtained from these economic accounts. We assume that these pesticides are offered by a perfectly competitive, multi-product chemical industry. This industry also offers mineral fertilizers. However, we do not isolate pesticides used by non-farmers due to missing economic values.

In regard to economic specifications applied to the farm sectors, we depart from the standard nested CES/CET specifications implemented in the GTAP-Agr framework. Rather, we implement the supply/demand equations described in the previous section. Specifically, we build the quadratic production functions for each crop and an entropic cost function that governs land allocation (note that this approach is locally similar to the standard CET specification). The price parameters of these production/cost functions are calibrated using the econometric elasticities calculated in the previous section. Pesticide use by crops is not estimated with great precision. We rely on the technical literature (IONOSYS 2016a, 2016b, PEREL, 2015) to provide initial value shares. For the three animal activities that we explicitly isolate (livestock, pigs and poultry) we proceed similarly. We construct a quadratic production function for each type of animal activity. The level of production depends on the level of use of different feeds (cereals, oil meals, maize fodder, other fodder, and compound feeds). We also construct an entropic cost function that specifies the number of animals. Here, we obtain the price responses from a literature review, adopting a substitution elasticity of 0.5 for feed commodities (Suh and Moss, 2016).

7.3.3 Results of the tax scenario

We first simulate the economic and environmental impacts of an ad valorem tax of 50% on pesticides, assuming that the deep parameters are policy invariant. This tax level approximates the current level in Denmark. Moreover, according to our estimated price elasticity of pesticides, this tax should reduce French pesticide use by approximately 40*% ceteris paribus*, which is close to the objective of the initial Ecophyto plan defined in 2008.

We indeed find that this tax would decrease farmers' use of pesticides by 37%. The difference is explained by crop price effects (see below). Table 7.4 below reports the evolution by crops and the main impacts on the French market. The obtained reductions are consistent with our elasticities. The application of pesticides to cereal areas declines the least (by 17%), which translates into lower wheat yield and production and subsequently creates a shortage in the world wheat market and increases French wheat prices. This output price effect slightly dampens the direct impact of the pesticide tax on yield and production. Overall, French wheat production declines by 4%. The impact of the tax on the oilseed sector is greater due to both higher initial applications of pesticides and higher price sensitivity: French oilseed production declines by 9%, and it appears that the application of pesticides on corn silage nearly disappears (reduction by 86%). This result is consistent with our previous estimated elasticity and again, the assumption of policy-invariant deep parameters. The market price of non-traded corn silage increases significantly (by 14%), thus limiting the reduction of corn silage production through an acreage effect. Corn silage areas slightly increase (by 3%) to the detriment of cereal areas. Because the application of pesticides is initially low on other fodder areas, the introduction of the pesticide tax has a limited effect on their production. We still obtain a significant increase in the price of other fodder (by 6%), which is pushed up by the corn silage price. Both products are substitutes for livestock feeding. The areas devoted to wine, fruits and vegetables are also nearly unchanged, and their production declines, which is similar to the yield effects (by 1%).

Table 7.4. French market impacts of the tax scenario (in % with respect to the observed 2011 levels)

	Area	Yield	Production	Price	Pesticide use
Wheat	-0.8	-2.7	-3.5	0.9	-17.1
Oilseed	0	-9.4	-9.4	1.7	-61.7
Sugar beets	1.5	-6.9	-5.4	4.2	-56.5
Forage maize	2.7	-11.1	-8.4	13.7	-85.9
Grasslands	0	-2	-2	5.6	-42
Beverages	0	-0.8	-0.8	0.3	-49.6
Vegetables and fruits	0	-1.4	-1.4	0.4	-49.5
Milk			-1.6	1.7	
Cattle meat			-1.9	1.2	
Pork meat			-1.4	1.3	

Interestingly, we find that our tax scenario has a non-marginal impact on the animal sectors. The French production of milk, cattle, pigs and poultry declines between 1% and 2%, due to less fodder availability and the higher prices of other feeds (including oil meals, by 1%). Animal market prices increase due to the higher production costs.

The French final consumption of food products is price and income inelastic. We thus observe a very limited decrease in French food consumption (0.2% for dairy and meat products). The reduction in French food production is thus equilibrated by trade flows (table 7.5). We find significant decreases in French exports (up to 9% for sugar and rapeseeds) and significant increases of French imports (up to 14% for sugar and 7% for soybeans). These trade impacts enhance farm and food production in other countries. We obtain the largest production impacts in other EU member states (production of oilseeds and sugar increases by nearly 1%). The impacts on third countries are more limited, due to import protections and preferences (captured using the standard Armington model for trade flows). The positive impact on animal production is only discernible for other EU member states.

Table 7.5. World market impacts of the tax scenario (in % with respect to the observed levels for 2011)

	French exports	French imports	USA production	Brazil production	Rest EU production
Wheat	-5.5	6	0.4	0.3	0.5
Oilseed	-8.8	7.1	0.1	0.2	0.7
Meat	-1.7	4.2	0	0	0.1
Dairy products	-2.8	5.7	0	0	0.3
Sugar	-8.7	14.5	0	0.1	0.9
Vegetables and fruits	-0.8	2.3	0	0.1	0

The production impacts on other countries may seem modest in terms of percentages, but they are consistent with the French share in the world food markets (French production represents less than 5% of world production for most products). We obtain similarly small percentage impacts on land use changes. Overall, the amount of world acreage devoted to arable crops increases by 32 thousand hectares. Malaysian and Indonesian areas devoted to palm oil increase by 2 thousand hectares (to compensate for reduced French rapeseed oil production). Expansions are found in (Brazilian) sugar cane areas (1 thousand hectares), pasture areas (19 thousand hectares) and deforestation (14 thousand hectares). These land use changes lead to a “one shot” 5.7 million tons of carbon emissions (CO2 equivalent). We also obtain an increase in direct carbon emissions due to the increased use of chemicals in other countries (by 0.9 million tons) and reduced carbon stored in biomass (by 2.1 million tons). Overall carbon emissions increase by 8.8 million tons. The reduction in worldwide animal consumption is not sufficient to counterbalance the carbon emissions related to land use changes and crop intensification in other countries.

Therefore, it appears that the French pesticide tax does not solve the trade-off problem between French pesticide use and climate change.[70] At the French level, we also obtain an increase in nitrogen surplus by 2 kg/ha. Three complementary reasons explain this result. First, the total French use of mineral fertilizers slightly increases by 1%, which is mostly explained by an increase in the output price. Second, French imports of oilseed products also increase (by 2% for soya meals). Third, French animal production decreases (see previous discussion). Finally, the pesticide tax negatively affects French economic welfare. As expected, farmers are the most penalized: farm value added decreases by 638 million euros, mainly due to a 19% reduction in land prices. The food industry also suffers from the tax (by 261 million euros), as it processes fewer French farm products. On the other hand, the tax receipts of the government increase (by 859 million euros), but French consumers suffer from an increase in food prices.

In total, French economic welfare, as measured by the equivalent variation, decreases by 108 million euros. It should be clear that this welfare criteria only includes the market effects captured by our CGE framework, which is not sufficient for defining the optimal pesticide policies, which should also take into account long-term human health and environmental effects (such as reduced water pollution from pesticides). More modestly, our results provide a macroeconomic assessment of some economic and environmental trade-offs that a simple pesticide tax alone cannot resolve unless a credible announcement of a significant pesticide tax could induce important technological change. This is the purpose of our technological scenario.

7.3.4 Results of the technological scenario

Although the current French pesticide policy is complex, its main philosophy is to avoid a punitive version and foster a positive version by supporting research and development on pesticide-saving technologies and farming practices. There are many possibilities, such as organic farming or using genetically modified (GM) crops, that have pros and cons as well as supporters and opponents. Our CGE framework with aggregated data does not permit us to individually analyse these alternatives. Golub et al. (2009) show how to combine detailed engineering and agronomic studies in a CGE framework to analyse GHG saving technologies. We follow their example and rely on our previous statistical results indicating that the French

[70] We are not able to accurately measure the increasing use of pesticides in other countries as the GTAP database does not distinguish pesticides from other chemical products. However, a good approximation is given by the total use of chemical products for farming activities in other regions because price effects are limited in those countries. This total use increases by 0.08%. Given that our tax scenario leads to a 37% reduction in French pesticide use, the world use of pesticides for farming very likely decreases, benefiting the health of the average food consumer.

farm sector was able to produce more annually with the same level of variable inputs (yield increasing technology). In the technological scenario, we assume that French research and development efforts are tailored to technologies and practices reducing pesticides while maintaining crop yields.

To implement this scenario in our CGE framework, ideally, we should identify the required level of R&D expenditures and the time necessary to develop these technological improvements. However, this is clearly beyond the scope of this paper and not easy to perform with the available databases. There are indeed many economic studies on policy-induced innovations. Alston (2018) summarizes this literature and finds that there are high social payoffs for agricultural R&D investment, which implies a very significant failure of the government in terms of the provision of agricultural R&D. This author also recognizes that it is very difficult to clearly document the payoffs for different technologies. Accordingly, we simulate a very simple technological scenario where we assume that the technical change reduces pesticide use per hectare by 30% for all crops. This level is obtained from academic papers produced during the Ecophyto 1 negotiations (see the introduction). In practise, we reduce the value of the parameters $\mathbf{b}_{1,r}$ (i.e., the vector of maximum required amount of pesticides to reach the maximal yield for each output k).

Table 7.6 below shows the evolution by crops and the main impacts on the French market. We find that pesticides are reduced by 30% for each crop. In fact, the price effects of this scenario are very limited. The most discernible impact is a reduction in the price of sugar beets (by less than 1%). The production, acreage and yield impacts are also muted. The most notable result is a small reduction in fodder outputs and the corresponding small increase in their prices, which stems from the fact that the initial application of pesticides on these areas is smaller than applications on arable crops. Therefore, these arable crop activities become more profitable following technological improvement, leading to a small acreage reallocation. For example, the sugar beet area increased by nearly 1%. In contrast, the corn silage area decreased. The reduced availability of fodder crops has a very marginal impact on livestock production (bovine production reduced by 0.01%) due to the substitution between the different types of feeds.

Table 7.6. French market impact of the technological scenario (in % with respect to 2011 values)

	Area	Yield	Production	Price	Pesticide use
Wheat	0.1	0	0.1	0	-30
Oilseed	0.2	0	0.2	0	-30
Sugar beets	0.7	-0.1	0.6	-0.5	-30
Forage maize	-0.5	0.3	-0.2	0.3	-30
Grasslands	0	0	0	0.1	-30
Beverages	0	0	0	0	-30
Vegetables and fruits	0	0	0	0	-30
Milk			0	0	
Cattle meat			0	0	
Pork meat			0	0	

Because the impacts on the French output market are marginal, the world impacts are logically also very limited. For example, the world area devoted to arable crops and palm oil decreased by 0.5 and 0.3 thousand hectares, respectively (because French sugar beet and oilseed output both increase). We do not obtain information on reforestation, and there was only a small increase in global pasture areas (by 0.9 thousand hectares). These limited land use changes lead to marginal carbon saving in soils. In fact, the main carbon impact is savings from chemical production activities. In total, this carbon emission is reduced by 0.2 million tons in this scenario. At the French level, we find no impact on nitrogen surplus. The very limited decrease in nitrogen exports caused by animal production is compensated by the reduction in French imports of protein crops. This scenario improves the economic welfare of French farmers (by 829 million euros) and marginally, that of the food industry (by 12 million euros). As we assume that the technological improvement is a free lunch, the expenditures of the French government remain stable. French consumers benefit from slightly lower prices (primarily for sugar and vegetable oils). In total, French economic welfare increases by 1611 million euros. This level is higher than the initial reduction in pesticide expenditures (by 825 million euros) due to the general equilibrium effects on the markets that benefit the French economy (terms of trade and allocation effects). Again, this level does not take into account all the health and environmental impacts induced by the reduced level of French pesticide applications and only provides an indication of the value of R&D expenditures that could be devoted to reduce the application of pesticides by 30%.

7.4 Concluding remarks

Pesticide use by farmers has generated a growing debate in France regarding its economic, environmental and health impacts. This paper contributes to these debates by offering an original macroeconomic quantification of some of the economic and environmental impacts. First, we statistically identify the influence of prices on pesticide use for all farm activities over the last 25 years. We find that the prices of crops and pesticides influence their use in many French regions and for many crops. The overall estimated own-price elasticity of pesticide demand amounts to -0.8, pesticide application on cereals being less price sensitive than other crops. Second, we simulate the market and welfare effects of two very different and thus illustrative reforms of French pesticide policy. Our CGE simulations show that a 50% tax on pesticides will reduce French farmers' pesticide consumption by 37%. This reduction would, however, have some side effects. The French farm and food industry would lose nearly 1 billion euros annually, and the nitrogen surplus would increase by 2 kg/ha. Moreover, world net carbon emissions would increase by approximately 9 million tons (CO_2 equivalent), mostly due to land use changes in other countries. Some deforestation would occur in some Latin American countries. These induced emissions is equal to 10% of the actual carbon emissions from French agricultural sector (Pellerin et al., 2017). We also find that the French animal sector would be significantly affected, mainly through less fodder availability.

We find that our second technological policy scenario solves these economic and environmental trade-offs, but such a scenario could only emerge in the long run due to inevitable innovation delays. Indeed, this second illustrative scenario relies on the crucial assumption of free-lunch new technologies. Some alternative technologies might not be implemented because they require some costly and specific investments in machines or knowledge. Our analysis is indeed limited by the quality of our databases: information on farm labour and capital devoted to crop protection are not easily accessible. It would be interesting for future research to gather these information. A more detailed representation of the production processes, such as the distinction of several pesticides (herbicides, fungicides and insecticides) or the consideration of biological processes (organic farming and crop rotation), would also improve our macroeconomic assessment (Chavas et al. 2010).

In the meantime, our analysis shows that French regulators are faced with economic and environmental trade-offs. We contribute by quantifying these trade-offs to help regulators sort out the lobbies' arguments. We highlight that a significant tax on pesticides would have side effects on several dimensions. However, these negative side-effects do not mean that regulators should maintain the existing legislative context. In contrast, it means that a pesticide taxation scheme could effectively reduce pesticide use, but other instruments should be jointly implemented to limit these side-effects.

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7.6 Appendices

Table 7.A1. Response parameters to pesticide prices with support values divided by two

	cereals		industrial crops		maize forage	other fodders		other crops	
Ile de France	0.31		0.88		0.71	0.34		15.78	
Champagne Ardennes	0.65		1.12		0.76	0.30		13.70	
Picardie	0.56	**	1.63	**	0.66	0.65		7.40	**
Haute Normandie	0.69	**	0.96	*	0.54	0.34		8.45	*
Centre	0.32		0.44		0.43	0.27		5.38	
Basse Normandie	0.68	*	1.00		0.53	0.14		7.72	
Bourgogne	0.48	*	0.34		0.75	0.21		8.30	
Nord pas de Calais	0.59	*	2.41	**	0.54	0.54		2.13	
Lorraine	0.03	**	0.05	**	0.05	0.00		0.70	
Alsace	0.64	**	1.67		0.90	0.41		10.79	
Franche comté	0.39	*	0.38		0.98	0.29	**	13.91	
Pays de la Loire	0.48	**	0.56	**	0.41	0.18		5.72	**
Bretagne	0.23		0.72	*	0.54	0.28	*	4.36	**
Poitou Charentes	0.54	**	0.26		0.51	0.41	*	4.50	**
Aquitaine	0.94	**	0.37		0.45	0.24		2.67	**
Midi Pyrénées	0.45	**	0.20		0.50	0.24	**	2.79	**
Limousin	0.38	*	0.67		0.45	0.03		2.34	
Rhône Alpes	0.95	**	0.68		0.76	0.15	**	1.26	*
Auvergne	0.53	**	0.45		0.67	0.01		6.33	*
Languedoc Roussillon	1.21	*	0.45		0.35	0.23	*	1.12	**
PACA	1.45	**	1.25		0.16	0.11		7.02	**

** and ** represent the 10% and 5% significance levels, respectively.*

Table 7.A2. Response parameters to pesticide prices with support values multiplied by two

	cereals		industrial crops		maize forage		other fodders		other crops	
Ile de France	0.12		0.97	*	1.07		0.50		12.82	
Champagne Ardennes	0.46		1.44	*	1.17		0.21		9.33	
Picardie	0.28		2.16	**	0.81		1.03		6.89	*
Haute Normandie	0.47	*	0.96	*	0.62		0.69		9.66	**
Centre	0.15		0.54	*	0.55		0.30		4.28	
Basse Normandie	0.52		1.21	*	0.61		0.09		9.02	*
Bourgogne	0.33		0.23		1.30		0.22		7.04	
Nord pas de Calais	0.40		3.11	**	0.69		0.77		0.70	
Lorraine	0.03	*	0.05	**	0.10	**	0.00		1.50	*
Alsace	0.51	**	1.84		1.18		0.54		8.85	
Franche comté	0.24		0.33		1.32	*	0.34	**	18.95	
Pays de la Loire	0.42	**	0.75	**	0.56	**	0.13		5.75	**
Bretagne	0.12		0.92	**	0.59		0.22		3.73	*
Poitou Charentes	0.28		0.11		0.70		0.63	**	6.53	**
Aquitaine	0.82	**	0.24		0.50		0.37	**	1.83	*
Midi Pyrénées	0.24		0.07		0.81		0.36	**	2.79	*
Limousin	0.28		1.10		0.57	**	0.01		1.73	
Rhône Alpes	0.90	**	0.76		1.14		0.13	*	0.72	
Auvergne	0.42	**	0.35		1.06	**	0.00		7.15	*
Languedoc Roussillon	1.67	**	0.70	*	0.56		0.34	**	0.59	
PACA	1.74	**	1.48		0.23		0.20		6.83	**

** and ** represent the 10% and 5% significance levels, respectively.*

CHAPTER 8. DECOUPLING VALUES OF AGRICULTURAL EXTERNALITIES ACCORDING TO SCALE: A SPATIAL HEDONIC APPROACH IN BRITTANY [71]

If local and global public goods (PGs) can be distinguished by the scale of demand for them, the literature dealing with the distance-decay of the willingness to pay (WTP) also stresses that local PGs can influence agents' utility across different geographical scales. Thus, the environmental service provided by a farmer can influence agents' utility across different scales. Assuming that the environmental service is a non-marketable service whose value is capitalized on private assets (particularly on houses), this chapter examines the shape of the value of externalities provided by different agricultural activities across space. The externality value of a considered activity in the municipality where the production occurs (called a direct effect) is distinguished from its value in other municipalities (called a spillover effect). We contribute to the literature on the hedonic valuation of agricultural externalities by disentangling the values at different scales. Previous studies have valued the externalities generated by a given agricultural activity using a single parameter only. Based on a simple theoretical model, we illustrate that this parameter actually captures the sum of the values of the different PGs generated by the activity. Using insights from distance-decay literature, we explain that this parameter depends on the distance to the source of the externality. Hence, each externality affects residents' utility at a different spatial scale. We run spatial hedonic pricing models on Breton rural house prices with explicit spatial interactions of agricultural activities. The model computes both the direct and the spillover effects for every explanatory variable. We illustrate that some of the agricultural activities located in a given municipality influence the residents' utility in neighbouring municipalities. We find that the signs of the externality values can be opposite at the two considered scales, illustrating that the different local PGs supported by the same activity are not sensitive in the same ways at the different scales.

[71] This chapter is coauthored with Abdel Osseni (INRA, SMART-LERECO) and Pierre Dupraz (INRA, SMART-LERECO). The results of this chapter contributes to the PROVIDE H2020 project.

8.1 Introduction

Agriculture is a multifunctional activity that ensures the joint production of marketable and non-marketable goods. These externalities impact the population's utility, either positively (e.g. conservation of biodiversity) or negatively (e.g. odor pollution). They have public good features: non-rivalry between consumers and/or non-excludability, especially for nuisances. As highlighted by the literature on distance-decay (e.g. Ay et al., 2016), their values for the consumers decline with the distance to the source of the externality. The modernization of agriculture has led to a gradual increase in negative externalities. The authorities have thus implemented several policies to internalize these effects. For example, the Common Agricultural Policy (CAP) offers payments to maintain specific areas (e.g. permanent grasslands) or to help farmers to modernize their farms and buildings to reduce pollution. The role of the authorities is to establish the most efficient instruments and to allocate an appropriate agro-environmental budget, which notably depends on the benefits captured by the population.

These benefits should be estimated using monetary valuation methods. The hedonic pricing method is a cornerstone of this literature (Rosen, 1974). Based on Lancaster's theory (1966), the hedonic pricing method is based on the principle that prices of marketable goods are defined by the combination of their attributes, which allows the value of each attribute to be determined. This method has been frequently used to estimate the population's willingness to pay (WTP) to improve environmental conditions, such as water quality (Leggett and Bockstael, 2000), or to reduce negative externalities, such as noise pollution (Fernández-Avilés et al., 2012). The hedonic pricing method is often applied to real estate observations, the theory being that, *ceteris paribus*, houses with superior amenities (negative externalities) have a higher (lower) price corresponding to the capitalization of the externality in the houses' value.

Several studies have valued agricultural externalities using this method. Le Goffe (2000) found that to double nitrogen concentration at the municipality scale decreases Breton bed and breakfast renting prices by 3%. Ready and Abdalla (2005) found that a new livestock farm located 500 meters from a house decreases its value by 6.4%. Herriges et al. (2005) stated that animal facilities reduce property values by 15% when they are located 0.25 miles upwind from houses. Bontemps et al. (2008) found that nitrogen surplus at the municipality scale decreases Breton house prices up to 7% but has no additional effect after 80 kg/Ha. They also found that the municipal share of temporary grassland decreases house prices up to 3%. Cavailhès et al. (2009) found that farmed activities have higher impacts when they are visible from the house.

Even if these papers provide remarkable insights on the impacts of agriculture on residents' utility, they have estimated the hedonic function at a given spatial scale, either the municipal scale (Bontemps et al., 2008; Le Goffe, 2000) or a lower one (Cavailhès et al., 2009; Ready and Abdalla, 2005). They do not provide information on the impacts of agriculture at higher scale, which is however important when designing agro-environmental policies. Indeed, using declared preference methods, the distance-decay literature highlights that residents are willing to pay (WTP) to conserve distant sources of amenities (even located from more than one hour to their house), even if the WTP decreases with the distance to the amenity source (Pate and Loomis, 1997). As rural households use to move over larger distance than urban ones to reach a place (for their job or leisure activities), agricultural activities can influence the housing market at larger scales than the previously examined ones and at least in neighboring municipalities. In addition, farms are dispersed over space and operate rarely on a single municipality. For example, Breton swine farmers are willing to apply manure at 70 kilometers from their headquarters (Gaigné et al., 2011), which imply that the externalities should not be contained in the municipality where the swine production occurs.

In addition, previous paper have ignored that agriculture supports the joint provision of several public goods. For example, agricultural wetlands provide habitat for remarkable biodiversity, which can be valorized by hikers, hunters and anglers, but agricultural wetlands are also located in areas with higher flooding risk. One can thus consider that an agricultural activity is a proxy of several public goods, whom quantities are unobserved in the usual datasets. As the distance-decay literature highlights that each public good affects agents under its own spatial range of impacts (e.g. Ay et al., 2016; Rolfe and Windle, 2012), one can even consider than an agricultural activity at a given localization is the proxy of several externalities, each of them impacting the residents' utility according to its own spatial range. The consequence is that one agricultural activity can have a positive (negative) impact at a narrow scale and a negative (positive) impact at a larger scale.

The objective of our paper is to distinguish the value of the agricultural externalities arising from the same agricultural activity at two different scales: the infra-municipal scale (where the residents and the agricultural activities are localized in the same municipality) and the extra-municipal scale (where the residents and the agricultural activities are localized in different municipalities), the distance to the considered activity being smaller in the infra-municipal scale. Our results could inform policymakers on the strengths and forms of the agricultural externalities over space, which should impact the design of agro-environmental policies.

For our purpose, we estimate a spatial hedonic model on the rural housing market of Brittany between 2010 and 2012. Spatial hedonic studies has been developed since the seminal work of Leggett and Bockstael (2000) (see Anselin and Lozano-Gracia, 2009 for a review) but have mainly relied on the spatial autoregressive (SAR) model or the spatial error model (SEM), which capture the whole spatial effect in a single parameter (McMillen, 2012). Here, we use econometric models that specify spatial effects for each of the explanatory variable, which are more flexible in modeling spatial spillover effects, i.e. the impact of a change in the variable level at one localization on the dependent variables of other places (Halleck Vega and Elhorst, 2015). The distinction between direct (i.e. the impact of a change in the variable level at one localization on the dependent variables of this localization) and spillover effects allow disentangling the value of agricultural effects at the different identified scales. We find that swine and poultry breeding activities impact house prices even in neighboring municipalities, suggesting a larger spatial impact than what had been previously estimated. We find that cattle activities (animal density, areas of temporary and permanent grasslands) have a direct negative impact on house prices but a positive spillover on neighboring house prices.

The next section presents a brief theoretical analysis on the measure of agricultural externalities at different scales and explain in more details the interest of the used spatial econometric models. The third section presents the empirical model and the descriptive statistics of the data. The fourth section presents the results of our estimations and the sensitivity analysis. We discuss the results in the last section.

8.2 Advances in spatial hedonic pricing

This section first explains the signification of the estimated parameters in hedonic method when considering a given agricultural activity as the support of different externalities with specific spatial range of impacts. We then present the developments of spatial econometrics to capture the spillover effects at the extra-municipal scale arising from the explanatory variables.

8.2.1 Hedonic pricing method in a spatial framework

The hedonic pricing method considers that goods, and in particular houses, are functions of their attributes (Ball, 1973). Denoting $\mathbf{y}_i$ as a vector of characteristics $(y_{1i},...,y_{ni})$ of house i ($i \in [1;I]$), which can be considered as marketable attributes, $\mathbf{z}_j$ as a vector of characteristics $(z_{1j},...,z_{mj})$ of localization j ($j \in [1;J]$), including the agricultural activities at the source of the externalities, and P_{ij} as the price of house i at localization j, the hedonic price function is written as follows:

$$P_{ij} = P(\mathbf{y}_i, \mathbf{z}_j) \tag{8.1}$$

Assuming that the consumer utility U_{ij} localized in house i in municipality j is a function of the consumer's composite consumption (x), $\mathbf{y}_i$ and $\mathbf{z}_{ij}$, U_{ij} is defined as follows:

$$U_{ij} = U\left(x, \mathbf{y}_i, \mathbf{z}_j\right) \tag{8.2}$$

Under the assumption that consumers maximize their utility under their income constraint $R = p_x x + P$, with P being the hedonic price and p_x the price of the composite good x, we reach the following first-order condition:

$$\frac{\partial U_{ij} \big/ \partial z_{kj}}{\partial U_{ij} \big/ \partial x} = \frac{\partial P_{ij}}{\partial z_{kj}} \tag{8.3}$$

The term $\partial P \big/ \partial z_{kj}$ is the consumers' marginal WTP for the attribute z_{kj} (the k[th] element of $\mathbf{z}_j$). In particular, z_{kj} can be an agricultural attribute, whom values follow a continuous distribution (e.g. an area or an animal density). Previous studies have focused on the estimation of $\partial P_{ij} \big/ \partial z_{kj}$, information on the household valuation of z_{kj}. Assuming a negligible impact of agricultural contractible labor on residents' localization choices, it means that z_{kj} support the provision of goods and/or services with public good characteristics.

We note $\left\{z_{kj}^{(1)}, ..., z_{kj}^{(Q)}\right\}$ the set of Q public goods supported by z_{kj}, the elements could being be null for some agricultural activities and non-null for the others. We assume that the production of the public good $z_{kj}^{(q)}$ depends only on z_{kj} such that $z_{kj}^{(q)} = \phi_k^{(q)}\left(z_{kj}\right)$, $\phi_k^{(q)}$ being the production function of the public good q supported by the activity k. In particular, we assume that the local conditions influence poorly the provision of the public goods and that the other activities do not enter in $\phi_k^{(q)}$. Each of the Q public goods is valued by the households such that $U_{ij} = U\left(x, \mathbf{y}_i, \mathbf{z}_j^{(1)}, ..., \mathbf{z}_j^{(Q)}\right)$, with $U\left(\cdot\right)$ being linear. In this framework, $\partial P \big/ \partial z_{kj}$ is in fact the sum of the value of all Q externalities supported by the attribute z_{kj}. Indeed, relation (8.3) gives:

$$\frac{\partial U_{ij} \big/ \partial z_{kj}}{\partial U_{ij} \big/ \partial x} = \sum_{q=1}^{Q} \frac{\partial P_{ij}}{\partial z_{kj}^{(q)}} \dot{\phi}_k^{(q)} \tag{8.4}$$

where $\dot{\phi}_k^{(q)}$ is the marginal productivity of z_{kj} for the production of the public good q, which is independent to the distance, and $\partial P_{ij} \big/ \partial z_{kj}^{(q)}$ is the value of the externality q supported by activity z_{kj} on house i. This value can be positive or negative. Relation (8.4) highlights that the estimated WTP for a specific agricultural activity in previous studies is equal to the sum of the values attributed to the externalities jointly produced by the activity. In particular, a positive value in relation (8.3) is positive implies that z_{kj} provides more positive externalities than negative ones. However, as $z_{kj}^{(q)}$ is often unobserved by the econometrician, the $\partial P_{ij} \big/ \partial z_{kj}^{(q)}$ cannot be measured independently.

Yet, as illustrated by distance-decay literature, each public good q supported by activity z_{kj} can impact the residents' utility in other locations (i.e. $U_{ij} = U(x, \mathbf{y}_i, \mathbf{z}_j^{(\mathbf{q})}, \mathbf{z}_{\mathbf{l}}^{(\mathbf{q})})$ whit $\mathbf{z}_{\mathbf{l}}^{(\mathbf{q})}$ being the matrix of the $J-1$ vectors of the set of public goods $\mathbf{z}_l^{(\mathbf{q})}$ in the other locations than j). Assuming that the I houses are randomly distributed over space, with I_j being the number of houses in localization j (such that $I = \sum_{j=1}^{J} I_j$), the total effect of a marginal change of z_{kj} is:

$$\underbrace{\sum_{l=1}^{J} \sum_{i \in I_l} \frac{\partial P_{il}}{\partial z_{kj}}}_{\substack{\text{total impact} \\ \text{of attribute } k \\ \text{in location } j}} = \underbrace{\sum_{i \in I_j} \sum_{q=1}^{Q} \frac{\partial P_{ij}}{\partial z_{kj}^{(q)}} \dot{\phi}_k^{(q)}}_{\substack{\text{direct impact} \\ \text{of unobserved} \\ \text{externality } q \\ \text{supported by } k \\ \text{in location } j}} + \underbrace{\sum_{\substack{l=1 \\ l \neq j}}^{J} \sum_{i \in I_l} \sum_{q=1}^{Q} \frac{\partial P_{il}}{\partial z_{kj}^{(q)}} \dot{\phi}_k^{(q)}}_{\substack{\text{spillovers} \\ \text{of unobserved} \\ \text{externality } q \\ \text{supported by } k \\ \text{in location } j}} \tag{8.5}$$

where $\partial P_{il} \big/ \partial z_{kj}^{(q)}$ is the spillover of the externality generated by $z_{kj}^{(q)}$ in the localization l. Indeed, due to data limitation on the spatial distribution of the agricultural activities, we can only know the municipal implantation of the different activities but, contrary to houses, we ignore their precise localization in the considered municipality.[72] Hence, the modeling framework integrates the insights from distance-decay considering two discrete zones: the municipality j where the agricultural activity z_{kj} occurs and all the other municipalities, the first area being localized closer to the externality sources than the second area.[73] These two areas allow to measure the externalities at the two identified scales. According to the distance-decay literature, the direct impact of the marginal change of $z_{kj}^{(q)}$ should be stronger than any spillover effect, i.e. $\left|\partial P_{ij} / \partial z_{kj}^{(q)}\right| > \left|\partial P_{il} / \partial z_{kj}^{(q)}\right|$. For example, a permanent grassland would impact more the utility of hunters that live in the same location than the utility of the hunters that live in other locations. However, as the number of hunters outside from j is supposed greater the number of hunters in j, the sum of the spillover effects can be higher than the sum of the direct impact depending on the strength of the distance-decay effect, i.e. $\sum_{i \in I_j} \left|\partial P_{ij} / \partial z_{kj}^{(q)}\right| \leq \sum_{\substack{l=1 \\ l \neq j}}^{J} \sum_{i \in I_l} \left|\partial P_{il} / \partial z_{kj}^{(q)}\right|$. This means that the sum of the utility of hunters that live in the same location than the considered grassland can be lower than the sum of the utilities of the hunters that live in other locations.

The measure of these direct (at the infra-municipal scale) and spillover (at the extra-municipal scale) effects is of major importance when designing an agro-environmental policy. However, as $z_{kj}^{(q)}$ is unobserved, the econometrician can only assess $\partial P_{ij} / \partial z_{kj}$ and $\partial P_{il} / \partial z_{kj}$ ($\forall l \in [1; J] - j$). Summing over the Q public goods, relation (8.5) leads to:

$$\underbrace{\sum_{l=1}^{J} \sum_{i \in I_l} \frac{\partial P_{il}}{\partial z_{kj}}}_{\substack{\text{total impact} \\ \text{of attribute } k \\ \text{in location } j}} = \underbrace{\sum_{i \in I_j} \frac{\partial P_{ij}}{\partial z_{kj}}}_{\substack{\text{direct impact} \\ \text{of attribute } k \\ \text{in location } j}} + \underbrace{\sum_{\substack{l=1 \\ l \neq j}}^{J} \sum_{i \in I_l} \frac{\partial P_{il}}{\partial z_{kj}}}_{\substack{\text{spillovers} \\ \text{of attribute } k \\ \text{in location } j}} \tag{8.6}$$

Given the wide range of values of the supported externalities and the form of the distance-decay effects over space, we can observe cases where the direct impact and spillover impacts have the same sign and other where they have opposite ones. The design of agro-environmental policy is easier in the first case than in the second one. Indeed, in the second case, an agro-

[72] We here assume that the $\mathbf{z}_j$ are localized on the centroid of the municipality.

[73] Given our assumptions, one can thus consider that the houses of municipality *j* are closer to the source of the externalities $\mathbf{z}_j$ than the houses located in other municipalities. Of course, there are cases where houses located in neighboring municipalities can be closer to at least one house of *j* but this is true on average.

environmental policy that support a specific activity will impact differently residents according to their localizations. For example, assuming that hunting conditions impact residents on a larger scale than flood risk but that the direct impact of flooding is higher (in absolute value) than suitable hunting conditions, a subsidy to maintain permanent grasslands in j would reduce the utility of residents in j but could increase the utility of residents in other localizations, i.e. at higher scales. Such distinction between direct impacts at the infra-municipal scale and spillover impacts at the extra-municipal scale has never been done in hedonic valuation of agricultural externalities. This is the aim of this paper.

8.2.2 Advances in spatial econometrics: integrating spillovers

Elhorst (2014) considered three types of spatial interactions to address the spatial effects: (i) interactions among dependent variables, (ii) interactions among explanatory variables and (iii) interactions among the error terms. To our knowledge, three studies have used spatial econometrics to assess the value of agricultural externalities, Kim and Goldsmith (2009) using the SAR model, Eyckmans et al. (2013) using the spatial autoregressive model with autoregressive disturbances (SARAR) model and Yoo and Ready (2016) using the SEM. These models do not considered interactions among explanatory variables, SEM only controlling for spatial autocorrelation of the error terms and SAR and SARAR controlling for interactions among dependent variables to measure both the direct impact of z_{kj} and the induced market adaptations on the other $I-1$houses by imposing *a priori* restriction on the spillover effects.[74] The spillovers from SAR and SARAR are defined as the global spillovers, i.e. the impact of a change in the level of z_{kj} that is transmitted to all other locations based on the infinite series expansion of the defined diffusion processes over all localizations (LeSage and Pace, 2009).[75] Even if we can compute spillover effects for each attribute, the SAR and the SARAR models capture the whole spatial effect in a single parameter (McMillen, 2012), with the consequence

[74] The induced market adaptations is notably linked to the assumption that sellers and buyers obtain information about nearby properties and use it to determine the prices of other houses. This assumption implies that $P_{ij} = P(\mathbf{y}_i, \mathbf{z}_j, \mathbf{P})$ where $\mathbf{P}$ is the vector of house prices in the considered market. Thus, a marginal change in z_{kl} will indirectly impact P_{ij} through price reorganization. The indirect impacts captured by SAR and SARAR do not capture the defined effects in relation (8.6).

[75] Basically, a marginal change of z_{kl} impacts house prices in localization l, which in turn, impact house prices in other locations, whom marginal change impact house prices in other locations, etc.

that two distinct activities present the same relative spillover impacts relatively to the direct ones. The SAR, the SEM and the SARAR models are thus not adapted to measure the defined spillovers in (8.6), which are defined in the spatial econometric literature as local spillovers. Contrary to the global spillovers, local spillovers do not disperse recursively through prices and concern only the impact of a change of z_{kj} on neighboring observations.

By contrast the spatial lag of exogenous variable (SLX) model the spatial Durbin model (SDM), the spatial Durbin error model (SDEM) and the general nesting spatial (GNS) model allow to measure the defined spillovers because they consider the interactions among explanatory variables (LeSage and Pace, 2009).[76] The SLX model and SDEM do not impose *a priori* restrictions between the spatial effects, explicitly considering both the direct impact $\partial P_{ij} / \partial z_{kj}$ and the local spillover impact $\partial P_{il} / \partial z_{kj}$ for each independent variable. By adding the interactions among dependent variables, the SDM and the GNS models consider specific global spillover effects for each independent variables. These models are thus well suited to study the forms and strengths of externalities over space (Halleck Vega and Elhorst, 2015). For this reason, Halleck Vega and Elhorst (2015) suggested taking the SLX model as the point of departure when estimating a spatial model and to successively develop it, if necessary, using the SDEM, the SDM or the GNS model.

To the best of our knowledge, Brasington and Hite (2005) were the first to use the SDM in hedonic analysis for environmental attributes. Comparing the OLS model, the SAR model, the SEM and the SDM, Montero et al. (2011) showed that the SDM was the most suitable model for valuing noise pollution in Madrid. In particular, Fernández-Avilés et al. (2012) highlighted that variable-specific spillovers correct for the nonlinearities of air pollution over space. Some more recent spatial hedonic studies have also tested the SLX model and the SDEM. Mihaescu and Vom Hofe (2013) were the first to use these specifications in the hedonic valuation of environmental attributes. Maslianskaïa-Pautrel and Baumont (2016) used the SLX model, the SDM and the SDEM to estimate the spillovers of environmental attributes. Notably, they found that the high prices on the shoreline are more determined by the impact of neighboring house prices (i.e., from the global spillovers) than by the positive amenities from seaboard proximity (i.e., from the local spillovers). To the best of our knowledge, no hedonic study on

[76] The SLX model contains only the interactions among the explanatory variables. The SDEM contains the interactions among the explanatory variables and among the disturbance terms. The SDM contains the interactions among house prices and among the explanatory variables simultaneously. The GNS model contains the three different spatial interactions presented by Elhorst (2014).

environmental valuation has ever used the GNS model, despite its apparent generality at first glance.

8.3 Empirical models and data description

We measure the direct and spillover impacts of agricultural activities on the house prices of rural and noncoastal municipalities of three departments of Brittany: Finistere, Morbihan and Côte d'Armor. We present the agriculture of Brittany and its environmentally related issues in the first part of this section. We then introduce the estimated models and the econometric strategy. Finally, we present the descriptive statistics of our sample.

8.3.1 Presentation of the study area

Brittany is the western region of France (Figure 8.1). In 2014, the utilized agricultural area covered 1.6 million ha, i.e., approximately 60% of the total region area. Breeding is the main agricultural activity in Brittany, Brittany representing 56% of French swine production and 44% of national egg production. Breton farms are mainly oriented toward dairy production, with 22% of French milk being produced in Brittany. Dairy production favors the maintenance of permanent grasslands and a typical "Bocage" landscape composed of hedgerows and earth banks. Owing to its countryside, its regional culture and its long seacoasts, Britany is the third highest French region for tourism. However, the environmental qualities of the region are threatened by intensive breading activities. Indeed, swine, poultry and, to a lesser extent, dairy productions contribute to nitrogen and phosphate spills in Breton watercourses and groundwater. The average nitrogen surplus of Brittany is 117 kg/Ha/year, i.e., approximately four times more than the national average (Peyraud et al., 2014). These surpluses led to high nitrogen concentrations in regional waters, which lead to several environmental negative effects such as water acidification, eutrophication, dystrophication and greenhouse gas emissions. In addition, the high nitrogen concentration rates have led to the proliferation of green algae on Breton seacoasts, whom decomposition produces the malodorous and potentially toxic hydrogen sulfide. It is suspected that several wild and domestic animal deaths have been due to hydrogen sulfide poisoning in recent years.[77] Thus, green algae negatively impacts the utility of local residents and tourists (MEEM, 2017). Local authorities have implemented several plans

[77] In 2009, the death of a horse due to green algae decomposition led authorities to launch the first green algae plan. In 2011, 36 wild pigs were found dead in a green algae zone. In 2016, the death of a jogger around the green algae zone led authorities to demand tests to determine the cause of the death. Today, no proof makes it possible to conclude that his death was due to hydrogen sulfide inhalation, but court actions are under process for the jogger and other potential victims.

to reduce green algae pollution, notably in 2017 with the promulgation of a 55 million euro plan for the period 2017-2021, who follow the 134 million euro plan for the period 2010-2016.

8.3.2 Empirical models and econometric strategy

We assume that all buyers and sellers are informed of the attribute levels at every possible housing location that they can move to utility-maximizing positions and that the Breton rural housing market is at the equilibrium. We focus on the relatively homogenous rural Breton housing market, constituted of noncoastal and rural municipalities in the 3 NUTS3 regions. The selection of a homogenous submarket should prevent most issues of spatial heterogeneity (Luc Anselin and Lozano-Gracia, 2009). Temporal heterogeneity is addressed using the observations for three consecutive years (2010 to 2012). We do not have to report any significant exogenous shocks to agricultural activities (i.e., similar agricultural and environmental policies), but the average prices slightly decrease over the period from 124,122 2012€ to 121,853 2012€.

We estimate the eight spatial hedonic models presented above (Table 8.1). The hedonic models are estimated under the semi-log form, which, according to Cropper et al. (1988) and Wooldridge (2015), is the best specification to mitigate the issue of heteroskedasticity and to limit unobserved heterogeneity biases.[78] The linear hedonic model we estimate is:

$$\ln\left(P_{ijt}\right) = \beta_0 + \boldsymbol{\beta_1}\mathbf{Y_i} + \boldsymbol{\beta_2}\mathbf{X_j} + \boldsymbol{\beta_3}\mathbf{C_j} + \varepsilon_{ijt} \tag{8.7}$$

where P_{ijt} is the selling price of house *i* located in municipality *j* in year *t*, $\mathbf{Y_i}$ is the vector of the intrinsic variables of house *i*, $\mathbf{X_j}$ is the vector of agricultural variables in municipality *j*, and $\mathbf{C_j}$ is the vector of the control variables in municipality *j*. We decompose the error term ε_{ijt} of (8.7) such that $\varepsilon_{ijt} = \boldsymbol{\alpha}\mathbf{t} + \varepsilon_{jt}$, where α is the vector of the temporal fixed effects. $\left(\beta_0, \boldsymbol{\beta_1}, \boldsymbol{\beta}_2, \boldsymbol{\beta}_3, \boldsymbol{\alpha}\right)$ is the set of vectors to be estimated, with $\boldsymbol{\beta_2}$ being the vector of our parameters of interest. The $n \times n$ matrix $\mathbf{W}$ is the spatial weight matrix that is required to estimate the seven spatial hedonic models in Table 8.1, which is symmetric and constituted of exogenous off-diagonals elements and null diagonal elements. The set of parameters $\left(\lambda, \rho, \boldsymbol{\eta}\right)$ is the specific parameters of the spatial econometric models, with $\boldsymbol{\eta} \equiv \left(\boldsymbol{\eta_1}, \boldsymbol{\eta_2}, \boldsymbol{\eta_3}\right)$. The

[78] We have also estimated the model using linear and log-log specifications. The results remain sensibly the same; they are available from the authors upon request.

successive introduction of these parameters leads to the different spatial econometric models. We estimate the linear hedonic model using the OLS and use the maximum likelihood estimation for the spatial hedonic models (Ord, 1975).

Table 8.1. Summary of the estimated spatial models (*Source: adapted from Halleck Vega and Elhorst, 2015*)

Models	Description	Direct effects	Spillover effects
Linear	$\ln(\mathbf{P}) = \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\varepsilon}$	Elements of $\boldsymbol{\beta}$	0
SEM	$\ln(\mathbf{P}) = \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\varepsilon}$ with $\boldsymbol{\varepsilon} = \lambda\mathbf{Wu}$	Elements of $\boldsymbol{\beta}$	0
SAR	$\ln(\mathbf{P}) = \rho\mathbf{W}\ln(\mathbf{P}) + \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\varepsilon}$	Diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}\boldsymbol{\beta}$	Off-diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}\boldsymbol{\beta}$
SLX	$\ln(\mathbf{P}) = \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\eta}_1\mathbf{WI} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\eta}_2\mathbf{WX} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\eta}_3\mathbf{WC} + \boldsymbol{\varepsilon}$	Elements of $\boldsymbol{\beta}$	Elements of $\boldsymbol{\eta}$
SARAR	$\ln(\mathbf{P}) = \rho\mathbf{W}\ln(\mathbf{P}) + \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\varepsilon}$ with $\boldsymbol{\varepsilon} = \lambda\mathbf{Wu}$	Diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}\boldsymbol{\beta}$	Off-diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}\boldsymbol{\beta}$
SDM	$\ln(\mathbf{P}) = \rho\mathbf{W}\ln(\mathbf{P}) + \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\eta}_1\mathbf{WI} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\eta}_2\mathbf{WX} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\eta}_3\mathbf{WC} + \boldsymbol{\varepsilon}$	Diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}(\boldsymbol{\beta} + \mathbf{W}\boldsymbol{\eta})$	Off-diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}(\boldsymbol{\beta} + \mathbf{W}\boldsymbol{\eta})$
SDEM	$\ln(\mathbf{P}) = \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\eta}_1\mathbf{WI} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\eta}_2\mathbf{WX} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\eta}_3\mathbf{WC} + \boldsymbol{\varepsilon}$ with $\boldsymbol{\varepsilon} = \lambda\mathbf{Wu}$	Elements of $\boldsymbol{\beta}$	Elements of $\boldsymbol{\eta}$
GNS	$\ln(\mathbf{P}) = \rho\mathbf{W}\ln(\mathbf{P}) + \beta_0\boldsymbol{\iota} + \boldsymbol{\beta}_1\mathbf{I} + \boldsymbol{\eta}_1\mathbf{WI} + \boldsymbol{\beta}_2\mathbf{X} + \boldsymbol{\eta}_2\mathbf{WX} + \boldsymbol{\beta}_3\mathbf{C} + \boldsymbol{\eta}_3\mathbf{WC} + \boldsymbol{\varepsilon}$ with $\boldsymbol{\varepsilon} = \lambda\mathbf{Wu}$	Diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}(\boldsymbol{\beta} + \mathbf{W}\boldsymbol{\eta})$	Off-diagonal elements of $(\mathbf{I} - \rho\mathbf{W})^{-1}(\boldsymbol{\beta} + \mathbf{W}\boldsymbol{\eta})$

Table 8.1 presents the decomposition of the direct and spillover effects for all estimated models. By construction, the linear model and the SEM provide only information on the direct effects of the explanatory variables. The SAR model and the SARAR provide information on both direct and global spillover effects (Anselin, 2003). On a technical side, global spillovers are the induced effects from a change in the variable level at one localization to all other locations by

the development of the spatial multiplier matrix (with non-null ρ) on immediate neighbors (first-order), neighbors of neighbors (second-order), etc. The global spillovers also include the feedback effects, i.e., the effects that pass through the neighboring localization back to the place from whence the change originated (LeSage and Pace, 2009). By comparison, the SLX model and the SDEM provide information on direct and local spillover effects, these spillovers only affecting the connected observations in $\mathbf{W}$ (with non-null η). The SDM and the GNS model provide information on direct and global spillover effects. However, the global spillovers are specific for each variable as SDM and GNS model consider explicit η for the explanatory variables, i.e. do not impose any prior restrictions between the direct and spillovers effects.

We use the specific-to-general approach first presented by Florax et al. (2003) and extended by Halleck Vega and Elhorst (2015) to select the best hedonic model specification. This approach consists in testing the spatial autocorrelation in the models by starting from simple models (OLS or SLX models) to more general models. However, it prevents the comparison between the SLX model and the SAR model, the SEM and the SARAR (Halleck Vega and Elhorst, 2015). For this reason, we use two alternative criteria to select the most suitable model, namely, the goodness of fit (measured here by the log likelihood, the Akaike information criterion (AIC) and the Nagelkerke R^2 tests (1991)) and the quality prediction (measured here by the normalized root mean square error – NRMSE –).[79] The combination of these two criteria and the specific-to-general approach has been used by Chakir and Lungarska (2017).

We estimate our models using the 40-nearest neighbor matrix (noted W1). Indeed, if the inverse-distance matrix is often used in environmental valuation studies within urban housing market, it is considered to be ineffective in rural housing markets (Kim and Goldsmith, 2009). By contrast, the k-nearest neighbor matrix is more adapted to the larger daily journeys and the larger geographic area of rural housing markets (Kim and Goldsmith, 2009). We specify the K-nearest neighbor matrix for the first 40 neighbors as, in our data, the municipality with the highest number of sold houses is 35 (the average number of sales per municipality is 5). The matrix is specified such that the K number of neighbors accounts for at least one house located in a neighboring municipality. W1 assumes that the 40 closer neighbors have the same impact on each other. In addition to W1, we also run the eight models with six alternative matrices (see

[79] We compute the NRMSE as NRMSE= $\sqrt{\frac{\sum_{i=1}^{n}(\hat{y}_{ik}-y_{ik})^2}{n}} \Big/ \sigma_{y_k}$ where $\hat{y}_{ik}$ is the predicted value of the estimated model, y_{ik} is the observed value of the dependent variable of the model, and σ_{y_k} is the standard deviation of the observed dependent variable.

appendix 8.A1.): the inverse of the Euclidean distance (denoted W2, with d_{mn} being the distance between observations m and n), the inverse of the Euclidean distance with the threshold (denoted W3 and W4), the square of the inverse of the Euclidean distance with the threshold (denoted W5 and W6) and the "queen"contiguity matrix between municipalities (denoted W7). [80] We use the contiguity weighting matrix W7 for municipality-aggregated data, decreasing the number of observations but controlling for the fact that houses in the municipality share the similar environmental and control variables. This should limit a "double-counting" effect for the measure of the spillovers, even if the number of sales is less than 10 for 85% of the municipalities.

8.3.3 Descriptive statistics

Our dataset merges information from the notarial house prices in Brittany (i.e., the MIN database), the agricultural census of 2010, Corine Land Cover, the INSEE population census of 2010 and the PIEB.[81] The descriptive statistics and the origins of the used variables are presented in Table 8.2.

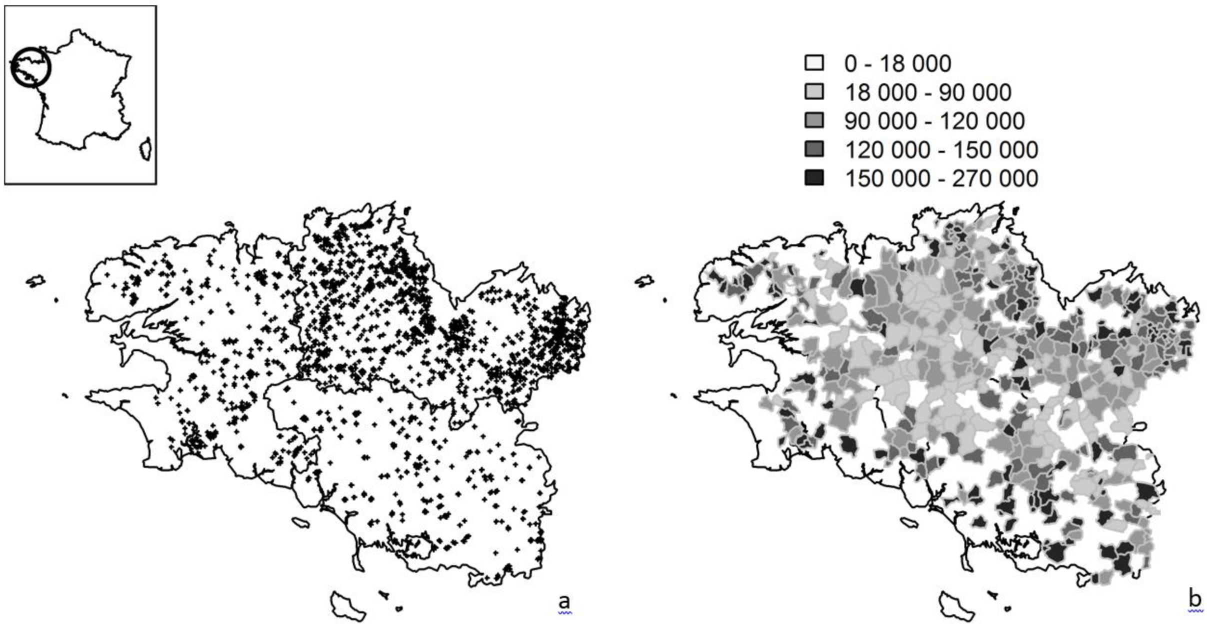

Figure 8.1. Maps of *(a)* the localization of the observations and *(b)* average house prices by municipality (*Source*: authors' own computation)

[80] Note that the maximum distance between the 40 closer neighbors is 25 kilometers, explaining the setting of the threshold in W4 and W6 to 25 kilometers. The average distance between the 40 closer neighbors is 10 kilometers, explaining the setting of the threshold in W3 and W5 to 10 kilometers.

[81] INSEE is the French acronym of "Institut National de la Statistique et des Etudes Economiques". PIEB is the acronym of "Portail de l'Information et l'Environnement en Bretagne".

Table 8.2. Descriptive statistics and variable definitions (N=2,476)

Variables	Mean	Std.dev	Min	Max	Description	Sources
House price	124214.50	57488.01	10000	448000	House prices in 2012€	MIN Dada
Intrinsic variables						
Nb_bathroom	1.31	0.46	1	2	Number of bathrooms	MIN Dada
Nb_room	4.96	1.38	3	9	Number of rooms	
Nb_floor	2.95	0.58	1	6	Number of floors	
Garden_area	2487.91	6334.04	42	178349	Garden area (square meter)	
Variables of interest						
Oilseeds_area	0.03	0.04	0	0.18	Oilseeds and proteins area (%UAA – Usable Agricultural Area -)	Agricultural cencus
Cereals_area	0.35	0.19	0	0.99	Cereals arean (%UAA)	
Othercrops_area	0.01	0.04	0	0.14	Othercrops area (including industrial crops) (%UAA)	
Perm_grassland_area	0.16	0.10	0	0.45	Permanent grassland area (%UAA)	
Temp_grassland area	0.13	0.18	0	0.72	Temporary grassland area (%UAA)	
Fallow_area	0.01	0.04	0	0.14	Fallow_area (%UAA)	
Shannon index	1.15	0.31	0.04	1.95	Shannon index	
Swine_poultry_N	49.21	72.72	0.00	534.12	Quantity of nitrogen from swine and poultry (KgN/TAM - Total Area of the Municipality)	
Cattle_N	34.39	23.76	0.00	100.22	Quantity of nitrogen from cattle (KgN/TAM)	
D_algae	19.06	11.48	3.22	50.48	The minimum distance from municipalities to sea affected by green algae (Km)	
Ratio_algae	0.87	0.16	0.31	1	The ratio of the minimum distance to sea on the minimum distance to green alga	
Control variables						
Waters_area	0	0.01	0	0.19	Water area (lake, rivers, etc.) (%TAM)	Corine Land Cover
Wetlands	0	0.01	0	0.29	Proportion of non-agricultural wetlands area (%TAM)	
Shrubs_area	0.01	0.03	0	0.29	Shrubs area (%TAM)	
Forest	0.10	0.09	0	0.77	Forest area (%TAM)	
Greenspace_area	0	0.01	0	0.07	Greenspace area (%TAM)	
Landfills_area	0	0.01	0	0.05	Landfill area (%TAM)	
Intdustries_area	0.01	0.02	0	0.18	Industrialized area (%TAM)	
Shops_area	0.08	0.14	0	0.92	Urbanized area (%TAM)	
D_sea	17.67	12.49	2.22	51.08	The minimum distance to sea (Km)	Authors' calculations
D_city	27.94	13.18	2.78	51.67	The distance to the closest city (Km)	
Pop_density	1.43	2.65	0.09	20.43	Population density (population/TAM)	INSEE
Revenues	20.04	3.21	12.39	38.82	Average income (income / populations in k€)	
Services	21.54	14.57	1.00	69	Number of services (e.g. school) in the municipality	
Dummies						
Year 2010	0.27	0.44	0	1	Sale in 2010	
Year 2011	0.47	0.50	0	1	Sale in 2011	
Year 2012	0.26	0.43	0	1	Sale in 2012	

The dataset provides exhaustive information on 2,476 house transactions between 2010 and 2012. We have the spatial coordinates of each observation (Figure 8.1a). The prices range from €10,000 to €448,000 in 2012€ and appear to be spatially correlated (Figure 8.1b). The intrinsic variables are available for the 2,476 observations. The agricultural variables are available only at the municipality scale, implying that the observations in the same municipality have the same explanatory variables (they share the same environment), and they provide information on the different types of crop cultivation and the nitrogen quantity released by each breeding activity. We also have information on green algae pollution, with the Euclidean distance between the houses to the closest municipality affected by green algae.[82] We compute the ratio of the minimal distance of municipalities to the sea to the minimal distance of municipalities to coastal municipalities affected by green algae. This ratio measures the relative proximity of municipalities to coastal municipalities polluted by green algae to the closest coastal municipality; its value ranges between zero and one. When the value is equal to one, the nearest coastal municipality of the house (and thus the closest beach) is polluted by green algae. When it is less than one, the nearest beach to municipalities is not affected by green algae. High values of this ratio express the loss of households' opportunity to enjoy nonpolluted beaches in their area. We also compute a Shannon index of farmland use in each municipality to represent land-use diversity, which may be considered as a proxy of landscape quality. The Shannon index is an entropy measure based on land shares; it increases with cultural diversity and decreases when it tends toward monoculture. The control variables contain additional environmental and accessibility variables that should influence the house price determination. Among the control variables, four variables are crucial for estimating the hedonic pricing model: population density, the municipalities' incomes, the distance to the closest CDB and the distance to the sea.[83] Because the first two variables are development and wealth indicators, their introduction in the model make it possible to correct for the heterogeneity of the considered market. The two last variables are major drivers of house prices.

8.4 Results

Moran's I for the residuals of the OLS model is significantly positive (p-value of 1.31E-10) with W1 (see Table 8.A2), highlighting the spatial autocorrelation in our data. In line with Kim

[82] The information on green algae pollution is provided by the 2013 report of the CEVA (the French organization for algae studies). The report is available at: http://www.ceva.fr/fre/MAREES-VERTES/Connaissances-Scientifiques/Marees-Vertes-en-Chiffres/Denombrement-des-sites-touches-par-des-echouages-d-ulves [consulted the 01/08/2017].

[83] The main cities considered are Rennes, Brest, Quimper, Saint-Brieuc, Guingamp, Vannes and Lorient.

and Goldsmith (2009), section 8.4.1 presents the selection of the most suitable spatial models with W1 and section 8.4.2 presents the estimated parameters of the selected models with W1. We present the robustness checks in section 8.4.3.

8.4.1 Selection of the model for the 40 nearest neighbors

Table 8.3 provides the results for the LM tests and its robust versions for the residuals of the OLS and SLX models using W1 to W7. The results for the OLS model reveal that the spatial parameters for both the lagged dependent variable and the disturbance term are significant at the 0.1% level. Thus, we reject the hypothesis of non-spatial autocorrelation for both house prices and the errors terms. We hold that SARAR specifications are relevant to correct for the spatial autocorrelation of our data. The LM tests for the SLX model reveal the non-significance of the spatial parameters for both the lagged dependent variable and the disturbance term, indicating that it is less appropriate to extend the SLX model to the SDM, the SDEM and the GNS model. This result indicates that the SLX model is the most suitable specification.

Table 8.3. Results for the spatial autocorrelation tests for the hedonic models with W1-W7

LM Test	W1	W2	W3	W4	W5	W6	W7
OLS versus SEM (Ho: λ=0)							
LM error	20.89***	27.35***	28.44***	44.32***	29.39***	39.14***	6.183*
RLM error	0.94	0.05	2.96°	4.17*	1.47	1.98	6.216*
OLS versus SAR (Ho: ρ=0)							
LM lag	51.49***	39.08***	45.69***	74.25***	43.57***	57.97***	0.050
RLM lag	31.55***	11.78***	20.23***	34.09***	15.64***	20.81***	0.083
OLS versus SARAR (Ho: ρ= λ=0)							
LM lag + error	52.43***	39.13***	48.64***	78.41***	45.03***	59.95***	6.266
SLX versus SDEM (Ho: λ=0)							
LM error	0.07	12.51***	24.20***	22.79***	27.75***	31.94***	2.679°
RLM error	1.17	0.26	2.34	0.91	0.45	3.02°	4.176*
SLX versus SDM (Ho: ρ=0)							
LM lag	2.16E-04	12.332***	25.48***	25.034***	28.40***	34.233***	0.009
RLM lag	1.10	0.07	3.62°	3.15°	1.1	5.31*	1.506
SLX versus GNS (Ho: ρ= λ=0)							
LM lag + error	1.17	12.59***	27.82***	25.94***	28.85***	37.25***	4.185
SAR versus SAC (Ho: λ=0)							
LM error	2.97°	0.12	3.76°	6.47*	1.92	3.36°	6.253*
SDM versus GNS (Ho: λ=0)							
LM error	1.18	1.78	3.87*	1.5	1.45	5.40*	4.256

***, **, *, ° stands for p-value of 0.1%, 1%, 5%, and 10% respectively.

The goodness-of-fit criteria, summarized in Table 8.4, reveal that all our spatial specifications improve the estimation quality compared to the OLS model. The results show that the GNS specification provides the highest R^2 and maximum likelihood estimation values. However, the results show that the SLX model provides the smallest value of the AIC, i.e., the SLX model minimizes the loss of information. The R^2 values of the SLX and GNS models are the highest and are almost equal. The smallest value of the NRMSE (Table 8.4) is provided by the GNS specification, indicating that the GNS model provides the best prediction quality. Although it is not the smallest value, the NRMSE of the SLX model ranks second with the SDM and SDEM. By combining the goodness-of-fit results with the quality prediction, we can indicate that the GNS model is the best specification for estimating our model, followed closely by the SLX model. Connecting these results with the LM results, we ultimately retain the SLX and GNS models as the best specifications.

Table 8.4. Goodness-of-fit and prediction quality of the different model specifications with W1-W6

	W1				W2				W3				W4				W5				W6			
	R^2	LL	AIC	NR MSE	R^2	LL	AIC	NR MSE	R^2	LL	AIC	NR MSE	R^2	LL	AIC	NR MSE	R^2	LL	AIC	NR MSE	R^2	LL	AIC	NR MSE
OLS	0.421	-1101.9	2267.7	76.1	0.421	-1101.9	2267.7	76.1	0.421	-1101.9	2267.7	76.1	0.421	-1101.9	2267.7	76.1	0.421	-1101.9	2267.7	76.1	0.421	-1101.9	2267.7	76.1
SEM	0.426	-1090.8	2247.6	75.5	0.428	-1088.0	2242.1	75.4	0.428	-1088.3	2242.6	75.5	0.431	-1081.5	2229.0	75.2	0.428	-1087.5	2241.0	75.4	0.430	-1083.1	2232.1	75.2
SAR	0.430	-1083.2	2232.3	75.4	0.429	-1084.5	2235.0	75.4	0.431	-1081.0	2228.0	75.2	0.436	-1071.0	2208.0	74.8	0.431	-1081.3	2228.6	75.2	0.434	-1075.2	2216.3	75.0
SLX	0.447	-1045.1	2214.2	74.3	0.440	-1060.2	2244.4	74.8	0.433	-1076.3	2276.5	75.3	0.443	-1053.8	2231.6	74.6	0.432	-1078.7	2281.3	75.4	0.436	-1069.3	2262.6	75.1
SARAR	0.431	-1081.0	2230.0	75.1	0.430	-1084.4	2236.9	75.3	0.432	-1078.7	2225.5	74.5	0.437	-1067.3	2202.6	74.1	0.431	-1080.2	2228.3	74.8	0.435	-1073.0	2214.1	74.3
SDM	0.447	-1045.1	2216.2	74.3	0.443	-1053.6	2233.2	74.5	0.439	-1064.3	2254.7	74.8	0.448	-1042.3	2210.7	74.1	0.439	-1065.0	2256.1	74.8	0.444	-1053.1	2232.3	74.4
SDEM	0.447	-1045.1	2216.2	74.3	0.444	-1052.9	2231.8	74.4	0.438	-1064.8	2255.7	74.8	0.448	-1043.2	2212.3	74.1	0.438	-1065.3	2256.5	74.8	0.443	-1053.9	2233.9	74.4
GNS	0.448	-1044.3	2216.5	74.0	0.444	-1052.8	2233.8	74.5	0.439	-1062.6	2253.1	72.9	0.449	-1041.5	2211.0	73.5	0.439	-1064.1	2256.2	73.3	0.445	-1050.5	2228.9	72.6

8.4.2 Spatial hedonic results using the 40 nearest neighbors matrix

We first present the results of the OLS model (Table 8.5). The coefficients are corrected for issues of heteroskedasticity using the White approach. Our results reveal that the crops that influence the Breton population are cereals and temporary grassland (significant effects at the 10% level). Each additional 1% of the cultivated cereal area increases house prices by 0.16% at the average point (from 35% of the UAA to 36%). By contrast, a relative increase of 1% of the temporary grassland area decreases house prices by 0.17%. We find no effect of permanent grasslands and the Shannon index on house prices. The diversity of the landscape is possibly already taken into account by the six agricultural land categories as explanatory variables. The effect of nitrogen on utility is controlled by swine, poultry and cattle impacts on house prices. The combined effect of swine and poultry is negative and significant at the 1% level. On average, house prices will decrease by 1.80% if we double the swine and poultry density. Similarly, the results show that cattle nitrogen negatively influences the Breton population by decreasing house prices by 2.88% if we double the cattle density. Finally, our results indicate that the moves from the first to the third quantiles for green algae pollution (from 0.8 to 1) decrease house prices by 2.7% (effect significant at the 10% level). This decrease is a relatively important effect, even if it is valued 5 times less than Wolf and Klaiber (2017). This difference may be explained by the two distinct submarkets, Wolf and Klaiber focusing on properties within 500 meters around the algae pollution whereas we have explicitly excluded these observations. We find that all intrinsic variables are significant at the 0.1% level. Regarding the control variables, the expected effects are found.

We now investigate the results from the SLX and the GNS models (Table 8.5). The structure of the GNS model implies that the estimated coefficients in Table 8.5 are not the marginal effects. Table 8.A3 in the appendices summarizes the marginal effects for the SLX and GNS models.

Table 8.5. coefficients for the linear and selected spatial hedonic models with W1

Variables	OLS model			SLX model						GNS model					
	Est. Coef	Std. Err		Coef.	Std. Err		Coef. (lag)	Std. Err		Coef.	Std. Err		Coef. (lag)	Std. Err	
Constant	10.63	0.14	***	9.70	0,47	***	-	-		6,20	1,91	**	-	-	
Nb_bathroom	0.26	0.02	***	0.26	0,02	***	0.11	0.14		0.25	0.02	***	0.02	0.13	
Nb_room	0.11	0.01	***	0.11	0,01	***	-0.07	0.05		0.11	0.01	***	-0.10	0.04	*
Nb_floor	-0.06	0.01	***	-0.06	0,01	***	-0.01	0.10		-0.06	0.01	***	0.01	0.07	
Garden_area	9.41E-06	1.64E-06	***	9.80E-06	1,23E-06	***	2.54E-07	1.00E-05		9.79E-06	1.21E-06	***	-3.30E-06	8.25E-06	
Oilseeds_area	-0.11	0.52		-0.43	0,58		-0.63	2.02		-0.48	0.58		-0.61	1.66	
Cereals_area	0.16	0.09	°	-0.34	0,14	*	1.33	0.27	***	-0.33	0.14	*	1.00	0.26	***
Othercrops_area	1.49	3.52		4.33	4,82		-41.26	21.61	°	4.51	4.80		-33.56	17.95	°
Perm_grassland_area	-0.17	0.21		-0.63	0,29	*	1.41	0.61	*	-0.58	0.29	*	1.04	0.53	*
Temp_grassland area	-0.17	0.09	°	-0.51	0,15	***	0.78	0.32	*	-0.52	0.15	***	0.71	0.27	**
Fallow_area	-1.14	3.59		-4.76	4,90		43.29	22.09	°	-4.85	4.87		35.17	18.39	°
Shannon index	-4.53E-03	0.06		0.03	0,08		-2.76E-03	0.17		0.04	0.08		-0.02	0.14	
Swine_poultry_N	-3.62E-04	1.25E-04	**	-6.78E-05	1,43E-04		-8.60E-04	3.58E-04	*	-5.86E-05	1.42E-04		-5.60E-04	3.44E-04	°
Cattle_N	-8.46E-04	4.06E-04	*	-1.13E-03	4,62E-04	*	2.29E-03	1.40E-03	°	-1.14E-03	4.61E-04	*	1.94E-03	1.14E-03	°
D_algae	-4.61E-04	1.50E-03		-4.59E-03	0,01		0.01	0.01		-4.47E-03	4.91E-03		0.01	0.01	
Ratio_algae	-0.13	0.07	°	-0.12	0,12		0.05	0.20		-0.12	0.12		0.10	0.17	
Waters_area	-0.16	0.81		-3.60E-04	1,12		-1.43	2.22		-0.03	1.11		-1.08	1.84	
Wetlands	-0.55	0.56		-0.69	0,79		0.74	2.71		-0.65	0.79		0.48	2.24	
Shrubs_area	0.37	0.27		0.32	0,35		-0.44	1.03		0.29	0.34		-0.22	0.86	
Forest	-0.11	0.10		-0.04	0,11		0.04	0.27		-0.04	0.11		0.01	0.22	
Greenspace_area	0.29	1.33		-0.32	1,41		-1.23	4.15		-0.06	1.41		-1.31	3.41	
Landfills_area	0.56	1.88		-0.49	1,77		5.94	5.78		-0.61	1.77		3.64	4.70	
Intdustries_area	0.25	0.33		-0.22	0,45		-0.93	1.14		-0.30	0.45		-0.65	0.94	
Shops_area	-0.34	0.22		-0.39	0,26		0.05	0.61		-0.42	0.27		0.29	0.53	
D_sea	-0.01	1.69E-03	***	2.29E-04	0,01		-0.01	0.01		-2.00E-04	0.01		-0.01	0.01	
D_city	-5.93E-04	7.29E-04		-3.55E-03	3,82E-03		3.27E-03	4.30E-03		-4.80E-03	3.63E-03		4.91E-03	4.03E-03	
Pop_density	0.02	0.01	°	0.02	0,01	°	0.02	0.03		0.02	0.01	°	0.01	0.03	
Revenues	0.03	3.08E-03	***	0.01	4,69E-03	**	0.04	0.01	***	0.01	4.65E-03	**	0.02	0.01	°
Services	1.77E-04	7.30E-04		1.80E-03	8,00E-04	*	-3.97E-03	1.83E-03	*	2.19E-03	8.06E-04	**	-4.17E-03	1.57E-03	**
Time FE	Yes			Yes						Yes					
R²	0.421			0.447						0.448					
LL	-1101.86			-1045.121						-1044.267					
AIC	2267.7			2214.241						2216.533					
ρ	-			-						0.358 *					
λ	-			-						-0.533 °					

***, **, *, ° stands for p-value of 0.1%, 1%, 5%, and 10% respectively

In the SLX model, both the direct and indirect effects of the cereals area, the permanent grassland area and the temporary grassland area are significant. Our results show that they are negatively correlated with the selling prices of the houses within their municipality boundaries. The indirect impacts of the cereals area, the permanent grassland area and the temporary grassland area are positive and higher in absolute term than the direct effects, meaning that they positively influence the utility of the inhabitants living in neighboring municipalities. As permanent grasslands are mainly agricultural wetlands in Brittany, this effect could reflect the local disutility of permanent grasslands due to the presence of flood risk but the positive effects of other externalities, such as biodiversity and landscape beauty, at a larger scale. In the linear model, the result for permanent grasslands was nonsignificant at the 10% level. This result could indicate that we have disentangled the scale effects of the different externalities by the management of permanent grasslands. Similarly, the results suggest a negative local effect of temporary grasslands and cereals at the infra-municipal scale, which may be due to agricultural practices, but positive local spillovers at the extra-municipal scale, which may be attributed to the attractiveness of this landscape. These results are nonsignificant at the 10% level under the GNS model.

We find an impact of both the indirect and the total impacts of the swine and poultry density in the SLX and GNS models that is negative and significant at the 5% level. This result suggests that the negative externalities of swine breeding are perceived far from the production zone. However, we found that the direct effect is negative and significant in the GNS model but nonsignificant in the SLX model. The results reveal that the direct impact of swine breeding tends to be negative but may not be robust. This result could represent that the recent investments of swine and poultry farms in renovating their buildings (notably with the PMPOA 1 and 2 programs). One consequence of these investments is that farmers must transport and spread manure out of their farms, which could explain why the local and global spillovers of the swine and poultry density are negative and significant. Overall, using the SLX and GNS results, we find that if we double the swine and poultry density, house prices are reduced by 5.38%, i.e., approximately three times what was estimated in the OLS model. The results of the SLX and GNS models are more in line with what we find in the literature, notably the results of Bontemps et al. (2008), who used a nonparametric hedonic function in Brittany.

We find that in the SLX model, both the direct and indirect impacts of the cattle density are significant at the 10% level. The direct impact is negatively correlated with house prices, but the local spillover effect is positive. This result could illustrate the negative impact of the odor nuisance within the municipalities and the contribution of pastures to landscape attractiveness. In addition, we find that the positive externalities of the cattle density have an impact that is two times greater than the impact of the negative externalities. In the GNS model, the direct impact of the cattle density is also significant and negative, showing that the direct effect is robust. The indirect impact in the GNS model is nonsignificant, suggesting that only the local spillover impacts residents' utility. Similar to the swine and poultry density, we find that the effects of the cattle density are underestimated in the OLS model compared to the SLX and GNS models.

We find that the D_ALGAE and RATIO_ALGAE variables are nonsignificant at the 10% level in both the SLX and GNS models. This result means that the negative effects of green algae pollution found in the linear model are not robust when we correct for spatial autocorrelation. Indeed, even in the SEM (see Appendix 8.A4.), we find that this effect disappears. This result suggests that green algae pollution is spatially correlated with an omitted variable that influences residents' utility.

Finally, our results for the control variables reveal that population income is positive and significant at the 1% level and positive for both the direct and the indirect impacts for both SLX and GNS models. This result reflects the homogeneity of the submarket within the rural housing markets of Brittany. The population density is also significant in the SLX model but only for the direct impact. As in the OLS model, the direct effects of the intrinsic variables are significant at the 1% level in both the SLX and GNS models. The indirect effects of the intrinsic variables are nonsignificant except for the garden area, which is significant at the 10% level in the GNS model (see Table 8.A3).

8.4.3 Robustness checks

8.4.3.1 Impact of the spatial matrix

We provide here the robustness analyses to examine the sensitivity of our results to the different spatial matrices. All criteria in tables 8.3 and 8.4 indicate that the GNS model is the most suitable for specifying spatial autocorrelation for the five matrices. We find that the direct impact of cattle breeding is robust (see Table 8.A5 in the appendices), while both the direct and indirect impacts of swine and poultry are not significant. Even if we find the same sign and amplitude for the indirect effect for cattle than in W1, this result is no longer significant. These results show that our previous insights depend on the specified matrices. Furthermore, our sensitivity analysis reveals that the impacts of D_ALGAE are significant for matrices W3 and W5 and increase house prices while the impacts of RATIO_ALGAE are significant for matrices W4 and W6 and decrease house prices. Utilizing alternative matrices than W1 confirms our linear results that green algae pollution decreases house prices, even if the significance of the impacts depends on the matrices used.

Regarding the selection of the spatial matrix, we find that W1 present the second highest R^2 and the second lowest log-likelihood (after W4). Overall, we agree with Kim and Glodsmith (2009) that the 40-nearest spatial matrix presumably provides the most interesting results in rural housing market.

8.4.3.2 Results at the municipal aggregated database

One of the limits of our approach is that, even if we know the specific location of each observation, the information on the agricultural variables is available at the municipal scale. Therefore, neighboring observations share similar agricultural and control variables. This feature is common to several hedonic studies (e.g. Bontemps et al., 2008). Table 8.A6 in the appendices presents the goodness-of-fit models and the prediction quality criteria for W7. The SDEM and the GNS model provide the highest R^2 and log likelihood values and the smallest values for NRMSE. The results of LM indicate that in this case, the SEM and/or SDEM are the most suitable (see Table 8.3). Using these criteria, we select the SDEM specification as the most appropriate for the aggregated model. Table 8.A7 in the appendices presents the results of the OLS model and the SDEM with W7. We notably confirm the results for swine, poultry and cattle breeding activities and, to a lesser extent, we confirm our results for grasslands.

8.5 Discussion and final remarks

Our hedonic application aims to value the externalities generated by agriculture in Brittany at different spatial scales, taking into account the spillover effects at the extra-municipal scale. Our results confirm that, on average, the residents of Brittany negatively value breeding activities, which is in line with the results of Le Goffe (2000) and Bontemps et al. (2008) in Brittany. However, in contrast to those studies, we distinguish between cattle and swine activities, allowing us to examine separately the effect of the two types of breeding. The results of the linear model highlight that swine and poultry activities impact residents’ utility more than do cattle activities.

Our spatial econometric results show that the externalities arising from these two types of breeding activities have opposite forms over space. First, the direct impact at the infra-municipal scale of the cattle density is negatively correlated with house prices, but the local spillover effect at the extra-municipal scale is positive, meaning that the effect of cattle breeding on residents' utility depends on the scale of the demand to the different externalities generated by cattle farms. Within the municipalities where production occurs, the negative impact could illustrate the impact of the odor nuisance. Outside the municipalities, the positive impact could represent the impact of grazing on landscape attractiveness, with landscape attractiveness impacting the resident inhabitants to a larger extent than the odor nuisance. We find similar results for temporary and permanent grasslands, where the direct impacts are negative but the local spillover impacts are positive. We interpreted the negative impacts by the increase in flood risks within the municipality where the production occurs and the positive impacts as the provision of some cultural and recreational services such as landscape attractiveness and biodiversity habitat that could benefit hunting activities (Mensah and Elofsson, 2017). As these areas are primarily managed by cattle farms, the tradeoff faced by residents in regard to cattle farms is reinforced: cattle farms reduce the utility of residents at a narrow scale but increase it at a larger scale. Second, in line with all the studies on effects of swine facilities on house prices, we find that swine and poultry activities have negative impacts on residents' utility. On average, the combined effect of swine and poultry leads to a 5.4% decrease in house prices if we double the animal density, which is quite similar to previous results (e.g., Bontemps et al., 2008). However, our spatial approach indicates that the negative impacts overlap with the municipality where the production occurs. The distance to swine activities has already been stressed to highlight the large impact of swine activities on house prices, but our results are larger than those previously estimated using linear econometrics with GIS data (e.g., Ready and Abdalla, 2005). In addition, we find that the direct impacts at the infra-municipal scale are lower than the local spillover impacts at the extra-municipal scale. We interpret this result as the reallocation of the odor nuisance due to the renovation of swine and poultry buildings and its replacement by manure spreading, sometimes far from buildings (Gohin et al., 2012; Peyraud et al., 2014). Overall, the spillover effects suggest that agricultural externalities overlap on neighboring municipalities, meaning that instruments design by municipal governance should be not optimal and that higher level of governance should be privileged.

Our results highlight the necessity of using spatial econometrics in the hedonic valuation of environmental goods. Correcting for the spatial autocorrelation of the observations modifies the significance, the sign and the amplitude of the parameters. For example, we find tha permanent grasslands have a negative impact on residents' house in the linear specification, which could question the involvement of the Common Agricultural Policy for their conservation. However, when controlling for the spillover effects, we find a positive impact of the grasslands, but this impact appears at larger scale than the municipal one. Regarding cattle farms, we find a negative impact in the linear specification but a potentially positive impact in the SLX specification, as the positive externalities are valued as two times greater than the negative externalities. Regarding swine and poultry activities, we find that the non-specification of the spatial correlation leads to an underestimation of their negative impacts on house prices by 2.5 in the case of the SLX model and even by three in case of the GNS model. These figures highlight the usefulness of spatial autocorrelation correction for the unbiased estimations of the parameter of interest when panel data are unavailable, the unbiased estimation of externalities being crucial for agro-environmental policy design. As repeat sales are rarely provided in real estate databases (at least for a short period of time such as ours), we advocate for a generalization of the utilization of spatial econometrics in hedonic valuation studies.

In particular, in line with Halleck Vega and Elhorst (2015), we advocate for a generalization of the utilization of spatial econometric models that specify the spatial relationships between the explanatory variables. Indeed, our results reveal that these models are the most appropriate for the seven tested matrices. The *a priori* restrictions between the direct and the spillovers effects in the SAR model, the SEM and the SARAR reduce the explanatory power of the explanatory variables, without mentioning that these restrictions reduce the information on the forms of externalities over space. In addition, we find that the SDM was not the most suitable model for specifying spatial dependence for the seven tested matrices. This result is particularly interesting, as, except for Maslianskaïa-Pautrel and Baumont (2016), all the environmental hedonic studies specifying the spatial relationships between the explanatory variables have used the SDM. Similar to Halleck Vega and Elhorst (2015), we call for a generalization to take the SLX model as the point of departure when estimating the spatial hedonic model and to then test for additional spatial effects using the specific-to-general approach (or any other procedure). Finally, in line with Chakir and Lungarska (2017), our results stress that the GNS model is often the best model for specifying the spatial autocorrelation of the observations. This result suggests that the three types of spatial interactions (autocorrelation, diffusion, heterogeneity) appears in our hedonic study. However, we find estimated parameters that are less significant than those in other models, which is a common feature of GNS models due to the complexity of the modeled spatial relationships (Halleck Vega and Elhorst, 2015).

Our work suffers from several potential limitations, with some of them having been investigated in the sensitivity analysis section. First, the results from the hedonic method are valid under several assumptions presented in section 3.2. The study of house prices in three NUTS3 departments may call into question the assumption of a homogenous market. To limit the heterogeneity of housing markets, we have focused on rural and noncoastal municipalities. We have also added population density and revenues as additional explanatory variables to capture some heterogeneity. We have limited temporal heterogeneity using time fixed effects. Finally, the mobilization of spatial econometrics models with a spatial effect on house prices (the SAR model, the SARAR, the SDM and the GNS model) “homogenizes” the Breton housing market. All these measures should prevent high unobserved heterogeneity in our data. Second, the choices of the spatial matrices can impact the results. We have proved that some of our results are robust but others only appears for some matrices. This suggests that the different matrices lead to different integration of space viscosity that could be more or less suitable for the capture of spatial processes. Third, observations from the same municipality share the same agricultural variables, which partly explains why we have tried several spatial weighted matrices. The fact that most of the interpretable results are derived from the 40-nearest neighbors matrix and the contiguity matrix highlights that this feature is important. Indeed, the other five matrices display a low number of significant estimated parameters. As these matrices are based on the inverse distance, more weight is placed on neighboring observations, which share explanatory variables (except intrinsic variables). It could be interesting to use GIS data for all observations to compute unique variables for each observation. However, the description of nonpoint source externalities such as nitrogen pollution is more adapted using the concentration (or share) rather than the closest distance to a potential source of a pollution (Bontemps et al., 2008). Overall, we agree with Kim and Glodsmith (2009) that the k-nearest spatial matrix presumably provides the most interesting results in rural housing market. Finally, our results relied only on parametric functional forms. Even if we had used several functional forms (see footnote 5), the utilization of a nonparametric method can lead to substantial gains in the precision of the estimation (Bontemps et al., 2008). There exist developments of nonparametric and semiparametric models within a spatial framework (McMillen, 2012). The applications of these models in our data may improve our results, but their utilization falls beyond the scope of our study. Similarly, other developments in the spatial econometrics literature, such as the mobilization of an endogenous spatial weighted matrix (Halleck Vega and Elhorst, 2015), should be considered in future hedonic valuations of agricultural and environmental externalities.

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8.7 Appendices

Table 8.A1. Definition of the used spatial weight matrices

Specifications	Matrices	Description
K-nearest weighting	W1	$\mathbf{W} = w_{mn} = \begin{cases} 1/K \text{ if } n \in [1;K] \\ 0 \text{ if } n \notin [1;K] \end{cases}$ with $K = 40$
Inverse distance weighting	W2	$\mathbf{W} = w_{mn} = d_{mn}^{-1}$
Inverse distance eighting with threshold	W3	$\mathbf{W} = w_{mn} = \begin{cases} d_{mn}^{-1} \text{ if } d_{mn} \leq 10\text{km} \\ 0 \text{ if } d_{mn} > 10\text{km} \end{cases}$
	W4	$\mathbf{W} = w_{mn} = \begin{cases} d_{mn}^{-1} \text{ if } d_{mn} \leq 25\text{km} \\ 0 \text{ if } d_{mn} > 25\text{km} \end{cases}$
Squared-inverse istance weighting with threshold	W5	$\mathbf{W} = w_{mn} = \begin{cases} d_{mn}^{-2} \text{ if } d_{mn} \leq 10\text{km} \\ 0 \text{ if } d_{mn} > 10\text{km} \end{cases}$
	W6	$\mathbf{W} = w_{mn} = \begin{cases} d_{mn}^{-2} \text{ if } d_{mn} \leq 25\text{km} \\ 0 \text{ if } d_{mn} > 25\text{km} \end{cases}$
Contiguity weighting	W7	$\mathbf{W} = w_{op} = \begin{cases} 1 \text{ if munipalities } o \text{ and } p \text{ are contiguous} \\ 0 \text{ if not} \end{cases}$

Table 8.A2. Spatial autocorrelation of the OLS residuals in the seven matrices

	W1	W2	W3	W4	W5	W6	W7
I of Moran	0.02	0.03	0.09	0.06	0.10	0.10	0.09
p-value	1.31E-10	1.17E-09	1.70E-09	1.38E-14	1.52E-09	3.15E-12	0.0011

Table 8.A3. Direct, indirect and total impacts of the SLX and GNS models with W1

Varibles	SLX model						GNS model					
	DE		IE		TE		DE		IE		TE	
Nb_bathroom	0.256	***	0.112		0.368	*	0.283	***	0.056		0.340	***
Nb_room	0.113	***	-0.065		0.048		0.122	***	0.000		0.122	***
Nb_floor	-0.062	***	-0.012		-0.074		-0.146	***	0.054		-0.092	
Garden_area	9.80E-06	***	2.54E-07		1.01E-05		7.55E-06	**	6.11E-06	°	1.37E-05	**
Oilseeds_area	-0.434		-0.632		-1.067		-0.816		0.454		-0.362	
Cereals_area	-0.343	*	1.328	***	0.985	***	0.009		0.093		0.102	
Othercrops_area	4.326		-41.261	°	-36.935	°	3.520		-33.228	*	-29.708	
Perm_grassland_area	-0.631	*	1.405	*	0.774		-0.552		-0.297		-0.849	
Temp_grassland area	-0.509	***	0.780	*	0.271		-0.446	*	0.128		-0.318	
Fallow_area	-4.763		43.294	*	38.531	°	-3.582		33.830	*	30.248	°
Shannon index	0.032		-0.003		0.029		0.093		-0.037		0.056	
Swine_poultry_N	-6.78E-05		-0.001	*	-0.001	**	-3.36E-04	°	-0.001	**	-0.001	***
Cattle_N	-0.001	*	0.002	°	0.001		-0.001	*	0.001		-1.58E-04	
D_algae	-0.005		0.007		0.002		0.001		-0.002		-0.001	
Ratio_algae	-0.124		0.046		-0.078		0.093		-0.214		-0.121	
Waters_area	-3.60E-04		-1.427		-1.427		-0.677		1.959		1.282	
Wetlands	-0.690		0.742		0.052		-0.663		0.875		0.212	
Shrubs_area	0.321		-0.439		-0.118		0.537		0.007		0.545	
Forest	-0.045		0.042		-0.002		-0.184		0.040		-0.144	
Greenspace_area	-0.323		-1.232		-1.555		0.691		-1.802		-1.111	
Landfills_area	-0.493		5.938		5.446		0.752		2.698		3.451	
Intdustries_area	-0.216		-0.928		-1.144		-0.718		1.356		0.637	
Shops_area	-0.395		0.047		-0.347		-0.632		-0.772		-1.403	
D_sea	2.29E-04		-0.009		-0.008	**	-0.012		0.004		-0.008	*
D_city	-0.004		0.003		-2.83E-04		0.003		-0.005		-0.002	
Pop_density	0.023	°	0.019		0.042		0.028		0.004		0.032	
Revenues	0.014	**	0.036	***	0.050	***	0.018	**	0.016		0.035	***
Services	0.002	*	-0.004	*	-0.002		0.002		-0.001		0.001	
Time FE	Yes						Yes					

***, **, *, ° stands for p-value of 0.1%, 1%, 5%, and 10% respectively.

Table 8.A4. SEM, SAR, SARAR, SDM and SDEM results with W1

Variables \ Coefficients	SEM	SAR	SAC	SDM		SDEM	
				X	Lag.X	X	Lag.X
Constant	10.894 ***	6.531 ***	5.100 ***	9.713 ***	-	9.703 ***	-
Nb_bathroom	0.256 ***	0.256 ***	0.254 ***	0.256 ***	0.113	0.256 ***	0.118
Nb_room	0.115 ***	0.115 ***	0.113 ***	0.113 ***	-0.065	0.113 ***	-0.067
Nb_floor	-0.062 ***	-0.060 ***	-0.057 ***	-0.062 ***	-0.012	-0.062 ***	-0.012
Garden_area	9.51E-06 ***	9.56E-06 ***	9.58E-06 ***	9.80E-06 ***	2.62E-07	0.000 ***	0.000
Oilseeds_area	-0.336	-0.269	-0.277	-0.435	-0.631	-0.441	-0.598
Cereals_area	-0.035	0.096	0.167 *	-0.343 *	1.329 ***	-0.342 *	1.320 ***
Othercrops_area	3.523	2.510	1.260	4.325	-41.277 °	4.329	-40.973 °
Perm_grassland_area	-0.412 °	-0.152	-0.015	-0.631 *	1.406 *	-0.627 *	1.392 *
Temp_grassland area	-0.263 *	-0.133	-0.073	-0.509 ***	0.781 *	-0.510 ***	0.782 *
Fallow_area	-3.426	-2.228	-0.918	-4.762	43.309 *	-4.758	42.942 *
Shannon index	0.012	0.008	0.013	0.032	-0.003	0.032	-0.003
Swine_poultry_N	-2.24E-04 °	-2.48E-04 *	-2.68E-04 *	-6.78E-05	-0.001 *	0.000	-0.001 *
Cattle_N	-0.001 *	-0.001 *	-0.001 *	-0.001 *	0.002 °	-0.001 *	0.002 °
D_algae	-0.001	1.62E-05	4.92E-04	-0.005	0.007	-0.005	0.007
Ratio_algae	-0.151 °	-0.082	-0.044	-0.124	0.045	-0.123	0.047
Waters_area	0.053	0.075	0.075	-0.001	-1.428	-0.009	-1.432
Wetlands	-0.635	-0.437	-0.307	-0.690	0.741	-0.689	0.714
Shrubs_area	0.380	0.325	0.270	0.321	-0.438	0.318	-0.409
Forest	-0.094	-0.081	-0.071	-0.045	0.042	-0.044	0.034
Greenspace_area	0.111	0.170	0.143	-0.323	-1.236	-0.306	-1.309
Landfills_area	-0.209	0.449	0.976	-0.492	5.944	-0.497	5.869
Intdustries_area	0.265	0.279	0.234	-0.216	-0.927	-0.225	-0.909
Shops_area	-0.302	-0.260	-0.234	-0.395	0.046	-0.398	0.049
D_sea	-0.007 **	-0.005 **	-0.004 ***	2.28E-04	-0.009	0.000	-0.009
D_city	-0.001	-4.59E-04	-2.89E-04	-0.004	0.003	-0.004	0.003
Pop_density	0.017	0.014	0.013	0.023 °	0.019	0.023 °	0.020
Revenues	0.023 ***	0.020 ***	0.017 ***	0.014 **	0.036 ***	0.014 **	0.036 ***
Services	0.001	0.001	0.001	0.002 *	-0.004 *	0.002 *	-0.004 *
Time FE			0.045 *	0.048 *	0.383 **	0.048 *	0.380 **
R²	0.426	0.430	0.431	0.447		0.447	
LL	-1090.8	-1083.167	-1080.987	-1045.121		-1045.08	
AIC	2247.576	2232.334	2229.974	2216.241		2216.16	
ρ	-	0.36 ***	0.48 ***	-0.001		-	
λ	0.43 ***	-	-0.37 *	-		-0.03	

***, **, *, ° stands for p-value of 0.1%, 1%, 5%, and 10% respectively.

Table 8.A5. GNS results for W2 – W6 (direct, indirect and total impact)

Variables	W2			W3			W4		
	DE	IE	TE	DE	IE	TE	DE	IE	TE
Nb_bathroom	0.253 ***	0.212	0.465 **	0.257 ***	0.097 **	0.354 ***	0.256 ***	0.157 °	0.413 ***
Nb_room	0.113 ***	-0.025	0.088	0.114 ***	0.016	0.130 ***	0.113 ***	0.021	0.134 ***
Nb_floor	-0.057 ***	0.023	-0.035	-0.060 ***	-0.054 *	-0.114 ***	-0.057 ***	-0.060	-0.117 *
Garden_area	0.000 ***	0.000	0.000	0.000 ***	0.000	0.000 *	0.000 ***	0.000	0.000 °
Oilseeds_area	-1.104	2.089	0.985	-0.682	0.486	-0.196	-0.094	-1.537	-1.630
Cereals_area	-0.157	1.027	0.870	-0.033	0.221	0.187	-0.390 *	0.978 ***	0.587 **
Othercrops_area	0.553	3.618	4.171	23.742 °	-23.582 °	0.160	22.455	-40.677	-18.222
Perm_grassland_area	-0.821 *	1.539	0.718	-0.323	0.278	-0.045	-0.686 °	0.935	0.249
Temp_grassland area	-0.335 °	0.512	0.177	-0.248	0.133	-0.115	-0.296	0.248	-0.048
Fallow_area	-1.342	1.428	0.085	-24.459 °	25.061 *	0.602	-23.988	44.879 °	20.890
Shannon index	0.135	-0.264	-0.129	0.038	-0.058	-0.020	0.075	-0.076	-0.001
Swine_poultry_N	0.000	-0.001	-0.001	-0.000	-0.000	-0.000 °	-0.000	-0.000	-0.000
Cattle_N	-0.002 **	0.004	0.002	-0.001 °	0.001	-0.001	-0.001 *	0.001	0.000
D_algae	0.001	-0.005	-0.004	-0.011	0.011	0.000	-0.017 *	0.019 *	0.003
Ratio_algae	0.021	-0.407	-0.386	-0.062	-0.073	-0.134	-0.093	0.046	-0.047
Waters_area	-1.582	5.094	3.512	-3.848 *	4.626 *	0.778	-0.664	1.102	0.438
Wetlands	-3.781 °	25.730	21.949	-0.754	0.048	-0.706	-3.125 °	5.565	2.440
Shrubs_area	0.310	-0.109	0.201	-0.068	0.498	0.430	0.300	-0.304	-0.004
Forest	0.023	-0.818	-0.795	0.023	-0.140	-0.117	-0.039	-0.041	-0.080
Greenspace_area	0.972	-1.422	-0.450	0.777	-0.502	0.275	-0.344	-0.112	-0.455
Landfills_area	0.691	-3.687	-2.996	0.169	1.544	1.714	-0.806	4.236	3.430
Intdustries_area	0.504	-1.505	-1.001	-0.592	1.087	0.495	0.110	-0.153	-0.044
Shops_area	-0.516	-0.177	-0.694	0.021	-0.348	-0.327	-0.221	-0.216	-0.436
D_sea	-0.010 **	0.013	0.003	0.007	-0.015	-0.008 ***	0.012	-0.022 *	-0.010 ***
D_city	-0.003 °	0.003	0.000	-0.012 °	0.011	-0.001	-0.010	0.010 °	0.000
Pop_density	0.025	-0.007	0.017	0.014	0.000	0.014	0.018	0.002	0.020
Revenues	0.019 **	0.048 °	0.067 **	0.011	0.017 °	0.028 ***	0.006	0.030 **	0.036 ***
Services	0.003 *	-0.006	-0.003	0.001	-0.001	0.000	0.001	-0.001	0.000
Time FE	Yes			Yes			Yes		
R^2	0.444			0.439			0.449		
LL	-1052.892			-1062.567			-1041.501		
AIC	2233.783			2253.133			2211.001		
ρ	0.046			0.313 ***			0.362 ***		
λ	0.273			-2.773 **			-0.207 °		

Table 8.A5. (continuation): GNS results for W2 – W6 (direct, indirect and total impact)

Variables	W5			W6		
	DE	IE	TE	DE	IE	TE
Nb_bathroom	0.258 ***	0.076 **	0.333 ***	0.261 ***	0.107 *	0.368 ***
Nb_room	0.114 ***	0.012	0.127 ***	0.113 ***	0.011	0.124 ***
Nb_floor	-0.061 ***	-0.046 *	-0.107 ***	-0.058 ***	-0.055 *	-0.113 ***
Garden_area	0.000 ***	0.000	0.000 **	0.000 ***	0.000	0.000 *
Oilseeds_area	-0.910	0.861	-0.048	-0.695	0.639	-0.056
Cereals_area	0.127	0.037	0.164	-0.154	0.402	0.248 °
Othercrops_area	22.823	-22.142 °	0.681	44.260 °	-46.577 °	-2.317
Perm_grassland_area	-0.095	0.014	-0.080	-0.401	0.410	0.010
Temp_grassland area	-0.010	-0.131	-0.141	0.046	-0.164	-0.119
Fallow_area	-23.110	23.010 °	-0.100	-45.559 °	48.644 *	3.085
Shannon index	0.009	-0.022	-0.013	0.153	-0.182	-0.029
Swine_poultry_N	-0.000	-0.000	-0.000 *	-0.000	-0.000	-0.000 °
Cattle_N	-0.002 *	0.001	-0.001	-0.001 °	0.001	-0.001
D_algae	-0.009	0.009	0.000	-0.017 *	0.018 *	0.001
Ratio_algae	-0.089	-0.058	-0.148 °	-0.013	-0.111	-0.124
Waters_area	-3.562 *	4.096 *	0.534	-1.866	2.431	0.565
Wetlands	3.394	-4.123	-0.730	-2.280	1.729	-0.551
Shrubs_area	-0.142	0.619	0.478	0.487	-0.129	0.358
Forest	-0.031	-0.073	-0.104	-0.033	-0.065	-0.097
Greenspace_area	0.528	-0.039	0.489	0.628	-0.676	-0.048
Landfills_area	-1.277	3.052	1.774	-0.996	3.657	2.661
Intdustries_area	-0.148	0.625	0.477	0.297	0.086	0.383
Shops_area	0.056	-0.353	-0.297	-0.173	-0.130	-0.304
D_sea	0.007	-0.014	-0.008 ***	0.012	-0.020	-0.008 ***
D_city	-0.010 °	0.010	-0.001	-0.010	0.009 *	-0.001
Pop_density	0.011	0.003	0.013	0.014	0.000	0.014
Revenues	0.013	0.015	0.028 ***	0.005	0.026 **	0.031 ***
Services	0.001	-0.001	0.000	0.001	-0.001	0.000
Time FE	Yes			Yes		
R^2	0.439			0.445		
LL	-1064.086			-1050.460		
AIC	2256.171			2228.920		
ρ	0.262 ***			0.345 ***		
λ	-0.171 *			-0.233 ***		

Table 8.A6. Goodness-of-fit and prediction quality for hedonic models with W7

	Goodness-of-fit			Prediction quality
	R^2	**ML**	**AIC**	**NRMSE**
OLS	0.494	-51.4	186.13	71.1
SEM	0.502	-47.6	161.13	70.2
SAR	0.494	-51.3	168.66	71.1
SLX	0.534	-31.1	186.13	68.2
SAC	0.502	-47.5	163.04	70.2
SDM	0.534	-31.1	188.12	68.2
SDEM	0.537	-29.3	184.67	67.8
GNS	0.537	-28.9	185.77	67.7

Table 8.A7. OLS and SDEM results with W7 in the aggregated model

Variables	OLS Model			SDEM Model					
	Coef.	Std. Err		DI		II		TI	
Constant	11.174	0.228	***	-		-		-	
Nb_bathroom	0.295	0.055	***	0.284	***	0.060		0.344	***
Nb_room	0.116	0.021	***	0.121	***	-0.001		0.120	***
Nb_floor	-0.153	0.039	***	-0.149	***	0.030		-0.119	
Garden_area	7.91E-06	2.26E-06	***	7.47E-06	**	6.11E-06		1.36E-05	**
Oilseeds_area	-0.508	0.727		-0.861		0.498		-0.363	
Cereals_area	0.052	0.122		0.025		0.008		0.033	
Othercrops_area	4.691	6.977		3.862		-35.155	*	-31.293	
Perm_grassland_area	-0.704	0.319	*	-0.543		-0.474		-1.017	
Temp_grassland area	-0.353	0.153	*	-0.437	*	0.052		-0.385	
Fallow_area	-4.529	7.017		-3.983		35.772	*	31.789	
Shannon index	0.064	0.082		0.113		-0.066		0.047	
Swine_poultry_N	-0.001	2.16E-04	*	-3.26E-04	°	-0.001	**	-0.001	***
Cattle_N	-0.001	0.001	*	-0.001	*	0.001		-2.85E-04	
D_algae	-4.97E-04	0.003		0.003		-0.005		-0.002	
Ratio_algae	-0.066	0.119		0.124		-0.285		-0.161	
Waters_area	-0.507	1.069		-0.758		2.268		1.510	
Wetlands	0.084	0.597		-0.812		0.991		0.179	
Shrubs_area	0.705	0.316	*	0.568		0.037		0.605	
Forest	-0.166	0.138		-0.192		0.024		-0.167	
Greenspace_area	0.842	1.943		0.851		-1.939		-1.088	
Landfills_area	0.483	1.842		0.757		2.427		3.184	
Intdustries_area	-0.514	0.727		-0.679		1.433		0.753	
Shops_area	-0.566	0.387		-0.623		-0.784		-1.407	
D_sea	-0.009	0.003	***	-0.014		0.006		-0.007	°
D_city	-0.001	0.001		0.005		-0.007		-0.002	
Pop_density	0.024	0.018		0.029		-7.49E-05		0.029	
Revenues	0.024	0.005	***	0.020	*	0.013		0.033	**
Services	0.002	0.001	°	0.002	°	-0.001		0.001	
Time FE	Yes			Yes					
R^2	0.494			0.53699					
LL	-51.35			-29.33534					
AIC	186.13			184.67					
λ	-			0.123 °					

***, **, *, ° stands for p-value of 0.1%, 1%, 5%, and 10% respectively.

CHAPTER 9. DECENTRALIZATION OF AGRO-ENVIRONMENTAL POLICY DESIGN: THE CASE OF ABANDONED WETLANDS IN BRITTANY [84]

The purpose of this chapter is to examine which geographical level of governance should be in charge of the design of agro-environmental payments when local and global PGs are jointly produced by agriculture and when different governments have different levels of information on the preferences for these two types of public goods (PGs). In particular, we assume that lower-level (local or regional) governments have an informational advantage with regard to the heterogeneity of local PG values over space (see Chapter 7), whereas higher-level governments can internalize the externalities due to global PG provision (see Chapter 6). Inspired by the literature on environmental federalism (Oates, 2001), this chapter contributes to the debate on the future CAP reform (see COM(2018) 392), which continues to require 4 to 5 billion euros each year for agro-environmental payments. First, we theoretically examine the gains emerging from partial or full decentralization of the agro-environmental policy design. We find that partial decentralization is optimal and that decentralization would lead to a decrease in total payments (i.e. a decrease of the agro-environmental budget) in most cases. Based on the estimated values of two local PGs and two global PGs supported by agricultural wetlands with risk of abandonment, we find that national decentralization of the design of agro-environmental payments such as those planned in the next CAP reform could improve the welfare of European residents by 67%.

[84] This chapter is coauthored with Matteo Zavalloni (UNIBO) and its results contribute to the PROVIDE H2020 project.

9.1 Introduction

Agriculture jointly produces private agricultural goods and environmental public goods (PGs), such as biodiversity, water quality or carbon sequestration, which affect the welfare of the population (OECD, 2015). The impacts of PGs on welfare depend on the geographical scale of their demand. In particular, the beneficiaries of global PGs are located all over the world, whereas local PGs benefit people in delimited areas around the provision locations.

The lack of market solutions for environmental PGs justifies the intervention of a public regulator. For example, in Europe, between 4 and 5 billion euro are allocated each year to farmers for the provision of environmental goods through the Agri-Environment-Climate Measures (AECMs) in the context of the Common Agricultural Policy (CAP). Given their structure, design, objectives and budget, the AECMs are largely decided on and bargained over at the EU level, with limited involvement of local authorities (Beckmann et al. 2009). Such centralized control has often been debated given the high heterogeneity of agricultural and environmental contexts across the EU and the informational advantage of lower levels of government (Beckmann et al. 2009; Droste et al. 2017). Indeed, the lack of integration of the heterogenous benefits and costs of PG provision leads to potential spatial mismatches between supply and demand for PGs produced by agriculture. The European Commission (EC) addresses this issue in the proposal for the new CAP, claiming that each member state will have the flexibility to implement specific instruments tailored to their local needs (COM(2018) 392).

The economic literature on environmental federalism addresses the question of which level of government should design and implement environmental policy by applying the "fiscal federalism" literature to environmental problems (Oates 2001). The basic assumptions of this literature are that (i) there are several levels of government (i.e., a federal system), (ii) local government can more effectively target public spending, but (iii) local government generates externalities to other jurisdictions and (iv) may face more deadweight losses than the central government. The literature examines the trade-off between the strengths of centralization and decentralization. The main conclusion is that instruments generating benefits contained within the boundaries of local jurisdictions present a high interest for decentralized management, whereas global environmental problems require central government intervention (Tiebout and Houston 1962). This conclusion is the essence of Oates' decentralization theorem (Oates, 1972): in the absence of interjurisdictional externalities and differentiated transaction costs between hierarchical governments, fiscal responsibilities should be decentralized.

The objective of this paper is to assess the welfare effects of decentralized decision-making concerning agro-environmental payments, taking into account the joint agricultural provision of local and global PGs. In addition to identifying the optimal level of decentralization, we compare the effectiveness of fully centralized vs. fully decentralized agro-environmental policy-making. Analysis of the effectiveness of decentralized decision-making is lacking in the case of agro-environmental European payments and more generally in the case of agriculture. This information is, however, central for the future CAP 2020 reform.

We examine this question in two steps. First, we develop a theoretical model in which we explicitly consider different hierarchical governments, ranging from regional to central ones. Under the assumption that governments are characterized by different information qualities over PG preferences, we evaluate whether decentralization potentially represents a suitable strategy to improve the effectiveness of agro-environmental policies. Many theoretical and empirical studies explore the diverse motives for decentralization (Besley and Coate 2003; Sigman 2005, 2014; Eichner and Runkel 2012; Harstad and Mideksa 2017; Droste et al. 2017). Some specific features of our model follow. First, we address the problem by assuming joint production global and local PGs on specific lands managed by agriculture. We consider that the farmers' choices to devote their lands to PG production depend only on agro-environmental payments. This implies, in contrast to most of the fiscal federalism, but in accordance with Bougherara and Gaigné (2008), that the suppliers of PGs are not part of the public sector but a private (agricultural) sector. This feature is shared with Harstad and Mideksa (2017), who focus on decentralization of the payment for environmental services in the case of deforestation, considering different motives for decentralization. One difference is that we consider not only a global PG (carbon sequestration in Harstad and Mideksa, 2017) but also a local PG, and the interactions between the two PGs drive most of our results. Second, we consider that the value of a local PG is heterogeneous over space (in each region). The heterogeneity of the value of a local PG in a relatively small area can be considered a characteristic feature of agriculture given the high heterogeneity of agricultural production conditions. Third, we consider that both suppliers and consumers of PGs are immobile, i.e., that there is no competition between local jurisdictions and that residents cannot "vote with their feet" (Tiebout, 1956). Fourth, we consider that hierarchical governments face different levels of information on the heterogeneity of local PG. The easier access to information is a classical argument of the fiscal federalism literature, which considers that local governments possess better knowledge of local conditions (Oates 1999). However, the better knowledge of the heterogeneity of local PG value is not a common feature of this literature, which rather considers different degrees of information on the heterogeneity of local preferences when regions are heterogeneous in tastes (Tiebout, 1956) or on the heterogeneity of conditions of PG provision (Harstad and Mideksa, 2017). Fifth, we consider that hierarchical governments face different agency costs when managing agro-environmental budgets. However, contrary to the usual assumption in fiscal federalism, we assume either economies or diseconomies of scale, in agreement with the debates on transaction costs in agro-environmental policies (Ahmad 2006; Mettepenningen et al. 2011; Weber 2015). Based on these specific features and welfare analysis, we determine the best level of government to design agro-environmental payments and compare the two extreme cases between regional and central governance. Our work shows that the total amount of financed lands decreases with decentralization to the benefit of the management of the most valuable lands; i.e., decentralized governance reduces global PG provision to the advantage of local PG provision. The effectiveness of decentralization compared to centralization depends on the value derived from local and global PGs produced on each unit of land, on the heterogeneity of local PG values, on the different agency costs of the governments and on the PG cost function.

Second, we apply our model to the empirical case of abandoned wetlands in Brittany (France), with farmers choosing to either manage wetlands or abandon them (given that wetland drying is forbidden in France). This specific case is representative of the more general PG provision loss due to land abandonment, which is a common risk across Europe (Terres et al. 2015). We use values from two local and two global PGs that have been valued using avoiding cost and transfer methods (Bareille et al., 2017). Contrary to the theoretical part, we consider the whole complexity of the costs faced by farmers when providing PGs, introducing heterogeneous costs and land constraints for each farmer. Based on this specific (but representative) case, we compare the welfare associated with regional, national and federal governments and determine which government should be responsible for the agro-environmental payments.

The article is organized as follows. The next section presents the theoretical model that analyzes the trade-offs between the centralized and decentralized governments. Section 3 is devoted to the empirical applications of the analytical results. We discuss the theoretical and empirical results in the fourth section. Section 5 concludes the paper.

9.2 Theoretical analysis

9.2.1 Description of the model

Assume an economy composed of R homogenous regions ($R \geq 2$). We assume no mobility of inhabitants between regions. Each region j contains a farming sector consisting of two farmers $i \in \{1;2\}$. The farmers own $\overline{X}_{ij}$ units of land that can be devoted to PG provision, X_{ij} being the total units of land devoted to PG provision on farm i. We consider that there is joint production of local and global PGs, and both PGs are provided in a fixed amount on one unit of X_{ij}. The management of X_{ij} incurs a net cost represented by the quadratic cost function $cX_{ij}^2/2$ ($c > 0$). We can interpret such a cost as the opportunity cost of environmentally friendly land management. In the case of land devoted to catch crops or permanent grasslands, the cost function corresponds to the opportunity cost of managing X_{ij} compared to the most profitable outputs (e.g., wheat). In the case of land with risk of abandonment, the cost function corresponds directly to the cost of maintaining agriculture on these lands, i.e., maintaining a nonprofitable activity on these lands. The farmers' profit function for PG provision is given by:

$$\Pi_{ij} = p_{ij} X_{ij} - \frac{1}{2} c X_{ij}^2 \tag{9.1}$$

Without loss of generality, we consider that the farmers face homogeneous costs (c is the same for the two farmers). Heterogeneous costs for the two farmers would only marginally change our results, which are here driven by the governmental information on the different scale of the demand for PGs.

The benefits obtained by each region depend on the provision of a local PG and a global PG. The local PG value is captured within the region where the production occurs, but the value is heterogeneous: farmer 1 supports the provision of a local PG of value v_{1j}, whereas farmer 2 supports the provision of a local PG of value v_{2j}. The global PG value is captured by the whole economy, and the value is homogenous. The benefits obtained by region j from PG provision are as follows:

$$B_j = \sum_i v_{ij} X_{ij} + \sum_i w X_{ij} + \sum_{\substack{k=1 \\ k \neq j}}^{R} \sum_i w X_{ik} \tag{9.2}$$

where v_{ij} is the marginal benefit derived from the consumption of the local PG on X_{ij} and w is the marginal benefit derived by the inhabitants of region *j* from the provision of global PG. We assume that we have $v_{1j} > v_{2j} \geq 0$ and $w > 0$. Note that benefits derived from global PG provision depend on the managed lands both inside and outside region j. Over the whole economy, the marginal value of a global PG is wR.

Public intervention is needed to provide the efficient level of PG. This intervention can be set or decided at different levels of governance, from the regional to the federal/central governments. To increase the PG provision on X_{ij}, the different governments can subsidize the farmers of the regions they are in charge of. We consider that a single level of government is in charge of the agro-environmental policy for S regions, with $S \in [1;R]$. For example, a fully centralized government is in charge of $S = R$ regions, whereas a regional government is in charge of $S = 1$ region. The level of government in charge of the agro-environmental policy has the responsibility to constitute a budget through the income taxation of the different regions under its responsibility and to design the most suitable policy instruments.

This setting matches the existing European case: the European Union is the only government in charge of agro-environmental payments, even if each European region contributes to the agro-environmental budget. Indeed, due to the possible introduction of concurrence distortions inside the EU common market, the first paragraph of article 107 of the Treaty on the Functioning of the European Union currently states that regional subsidies to private companies, including farms, are not allowed. Decentralized governments can still implement alternative agro-environmental instruments but without any public payments towards the farmers. We examine the effectiveness of agro-environmental payments at different levels of governance, with each government being in charge of S regions. Because the regions are homogenous, there are R/S governments in the economy.

The government in charge of the design of the agro-environmental policy aims to maximize the utility from PG provision but has different information quality on the values of the local PGs (while there is perfect knowledge on the supply side, which is surely the case in the CAP). In particular, we assume that the probability that the information on the local PG value is correct decreases with each step up in the governmental hierarchy. Such a probability is given by:

$$\theta(S) = 1 - \frac{(S-1)}{2(R-1)} \tag{9.3}$$

This parameter is equal to 1 in the case of complete decentralization. In this case, the government has perfect knowledge of the heterogeneous distribution of local PG values. The parameter is equal to 0.5 when there is complete centralization, implying full uncertainty of the value of local PGs. Indeed, the expected values of local PGs for a government of size *S* are $v_{1j}(\theta(S)) = \theta(S)v_{1j} + (1-\theta(S))v_{2j}$ and $v_{2j}(\theta(S)) = \theta(S)v_{2j} + (1-\theta(S))v_{1j}$. Thus, the central government considers that X_{1j} and X_{2j} provide the same expected value of local PGs, which is equal to the average value of the local PGs, i.e., $v_{1j}(\theta(R)) = v_{2j}(\theta(R)) = (v_{1j} + v_{2j})/2$. Without the loss of generality, we assume that $v_{1j} = v$ and $v_{2j} = 0$; v should be interpreted as the difference in the local PG values between the two areas rather than the real PG value of v_{1j}. This implies that $v_{1j}(\theta(S)) = \theta(S)v$ and $v_{2j}(\theta) = (1-\theta(S))v$ and furthermore that the different hierarchical levels have similar information on the average local PG values in the considered region but different information on the variance of local PG values. In addition, a government of size S considers that the global PG value is $w(S) = wS$. This implies that only the central government can internalize the entire value of global PGs, whereas lower hierarchical governments generate externalities to regions that are not under their governance.

In addition, we consider that governments may face different agency costs $\tau(S)$ for managing agro-environmental payments. These agency costs represent public administration costs, a special case of transaction costs (Mettepenningen et al. 2011). The literature on fiscal federalism considers that the highest levels of government face lower transaction costs, i.e., that public money management presents economies of scale (Ahmad 2006). These economies of scale are explained by the marginal agency costs that decrease with the size of the agency. The economies of scale would imply that the decentralization process presents $\tau(R) - R\tau(1) \leq 0$. However, the central government could also face a higher level of transaction costs than the regional government in the case when it assembles material on local conditions (Crémer et al., 1996). This material compilation appears in the CAP structure, where regional agencies provide estimates of the farmers' opportunity costs of PG provision in their region to the EC such that the EC sets agro-environmental payments equal to the median opportunity costs (Beckmann et al. 2009; Mettepenningen et al. 2011). The communication (or coordination) between agencies and central governments leads to specific transaction costs, implying that the decentralization process would present $\tau(R)-R\tau(1)\geq 0$. The parameter τ is thus the addition of two forces: economies of scale and communication/coordination costs. The result is that τ can be either increasing or decreasing with S : τ is positive (negative) when the savings of transaction costs due to economies of scale are higher (lower) than the coordination costs.

Given this structure, we examine the optimal level of decentralization of the agro-environmental policy. Two forces, working in opposite directions, are at stake. First, there is an informational advantage for lower levels of government. Second, there is a spillover-internalization advantage for higher levels of government. This implies a trade-off on the effectiveness of the decentralization of agro-environmental payments. We also examine the impact of agency costs on the effectiveness of decentralized governance, as agro-environmental policy is characterized by high transaction costs for public agencies (Mettepenningen et al. 2011).

Based on welfare analysis, we identify the optimal level of decentralization of the agro-environmental payments when such payments represent variable decisions. The stage of the game is the following. First, the social planner decides the level of governance based on welfare maximization of the overall economy. Second, the governments maximize the utility of the regions they are in charge of by determining the optimal agro-environmental payments based on the available information. Third, the farmers respond to payments by maximizing (9.1). We identify the optimal degree of decentralization that would maximize the welfare of the overall economy. In particular, we compare the welfare in two different cases: the *full centralization* case where the central government is in charge (i.e., the actual case) and the *full decentralization* case where the responsibility returns to the regional governments.

9.2.2 Solution of the model

9.2.2.1 Resolution of the program for a government of size S

Our theoretical analysis aims to determine whether the European Union should consider complete or partial decentralization. For this purpose, we solve the theoretical program using backward induction with three steps. We solve the problem by determining (1) the first-order conditions (FOC) of the farmers for given payments, (2) the payments for a given government and (3) endogenously, the optimal size of the governments.

Following the backward induction, in the third stage, given the payment levels p_1 and p_2, farmers determine the land allocated to PGs by maximizing equation (9.1). Under the assumption that the land constraint is not binding, the usual FOC yields:

$$X_{ij}^* = \frac{p_{ij}}{c}$$

In the second stage, the government of size S maximizes the utility function U_S, depending on the benefits (9.2) and the costs of agro-environmental payments. The government of size S maximizes:

$$\begin{aligned} &\max_{p_1\, p_2} U_S = SX_1\left[v_1(S)+w(S)\right]+SX_2\left[v_2(S)+w(S)\right]+(R-S)\left(\bar{X}_1+\bar{X}_2\right)w(S)-ktYS \\ &\text{s.t. } S\left(p_1X_1+p_2X_2+\tau(S)\right)=tYS \text{ and } X_{ij}^*=\frac{p_{ij}}{c} \end{aligned} \tag{9.4}$$

where t is the tax rate, Y is the total income of a region, k is the distortion of t and $\tau(S)$ is the cost of managing the schemes under S. The choice variables of the studied government are the payments p_1 and p_2, whose levels influence the levels X_1 and X_2 in the S governed regions. The lands $\bar{X}_1$ and $\bar{X}_2$ are the subsidized lands in the other $(R-S)$ regions, whose levels do not depend on the studied government. The regions under the size-S government benefit from the PG provision in these other regions. The government integrates the costs $ktYS$ incurred by the subvention of X_1 and X_2 into the regions to raise the agro-environmental budget tYS. Integrating the relations, equation (9.4) is equivalent to:

$$\begin{aligned} \max_{p_1\, p_2} U_S = &S\frac{p_1(S)}{c}\left[v_1(S)+w(S)\right]+S\frac{p_2(S)}{c}\left[\left(v_2(S)+w(S)\right)\right]+(R-S)\left(\bar{X}_1+\bar{X}_2\right)w(S) \\ &-kS\left[\frac{\left(p_1(S)\right)^2}{c}+\frac{\left(p_2(S)\right)^2}{c}+\tau(S)\right] \end{aligned} \tag{9.5}$$

The FOC of (9.5) on p_1 and p_2 leads to:

$$p_1(S)=\frac{v_1(S)+w(S)}{2k} \tag{9.6}$$

$$p_2(S) = \frac{v_2(S) + w(S)}{2k} \tag{9.7}$$

The payments are set such that the marginal costs of taxation (i.e., $2kp_1(S)$ for X_1) equals the expected marginal benefits (i.e., $v_1(S) + w(S)$ for X_1). Introducing these payments under the farmers' FOC leads to:

$$X_1(S) = \frac{v_1(S) + w(S)}{2kc} \tag{9.8}$$

$$X_2(S) = \frac{v_2(S) + w(S)}{2kc} \tag{9.9}$$

The levels of X_1 and X_2 increase when v and w increase but decrease when the cost parameters k and c increase.

Proposition 1: *The structure of the landscape depends on S. The level of X_2 increases with S due to the conjugate actions of information losses on local PG values and better integration of the global PG value. The level of X_1 increases with S if $0.5v < (R-1)w$ but decreases otherwise. Thus, the higher the relative increase in the local PG value with respect to the global PG value is, the more X_1 decreases with S.*

9.2.2.2 Optimal size of the government

The optimal government size is determined by maximizing the welfare of the entire economy, given the subsidy levels that each government level would apply. Under null transaction costs, the welfare of the economy under the governance of governments of size S is:

$$W(S) = R\Big[(v_1 + wR)X_1(S) + wRX_2(S) - k\big(X_1(S)p_1(S) + X_2(S)p_2(S)\big)\Big] \tag{9.10}$$

The welfare in relation (9.10) is the equivalent of (9.4) with perfect information on both local and global PGs at the scale of the whole economy. As the landscape structure depends on the payments $\{p_1(S); p_2(S)\}$, the welfare function depends only on the government levels. Noting that $\dot{v}_{1S} = -v/2(R-1)$ and $\dot{v}_{2S} = v/2(R-1)$, the FOC of relation (9.10) relative to S leads, after some mathematical simplifications (available in appendix 9.A.), to:

$$S^* = \frac{4R(R-1)^2 w^2 + v^2}{4(R-1)^2 w^2 + v^2} \tag{9.11}$$

Relation (9.11) highlights that the optimal level of governance is independent of the cost parameter c of the farmers and the deadweight loss rate k. Given (9.11), $S^* = 1$ in the only case when $R = 1$ and $S^* = R$ when $v = 0$: the two extreme cases are ruled out by definition. The derivatives of S^* relative to local and global PG values highlight that the decentralized strategies are more suitable when the heterogeneity of local PG (v) increases, whereas centralized governments are more suitable when the global PG value (w) increases. Indeed, because we have assumed that v and w are positive and R is higher than 2, we have:

$$\frac{\partial S^*}{\partial v} = -8(R-1)^3 vw^2 \Big/ \left(4(R-1)^2 w^2 + v^2\right)^2 < 0 \tag{9.12}$$

$$\frac{\partial S^*}{\partial w} = 8(R-1)^3 w^2 v^2 \Big/ \left(4(R-1)^2 w^2 + v^2\right)^2 > 0 \tag{9.13}$$

The two derivatives (9.12) and (9.13) indicate that the relative strengths of centralization (w increases) are greater than the strengths of decentralization (v increases) if w is greater than 1.

Proposition 2*: The two extreme cases, full centralization and full decentralization, are never optimal in our framework. The optimal level of governance increases with the value of global PG, while it decreases with the value of local PG. The amplitudes of the strengths towards centralization or decentralization depend on the value of* w*: the strengths are higher for centralization in cases where* $w > 1$*, while the strengths are higher for centralization otherwise.*

In Figure 9.1, we depict the optimal governance level (y-axis) for different values of v (x-axis) and w (z-axis) under an economy composed of 10, 50, 100 and 200 homogenous regions.

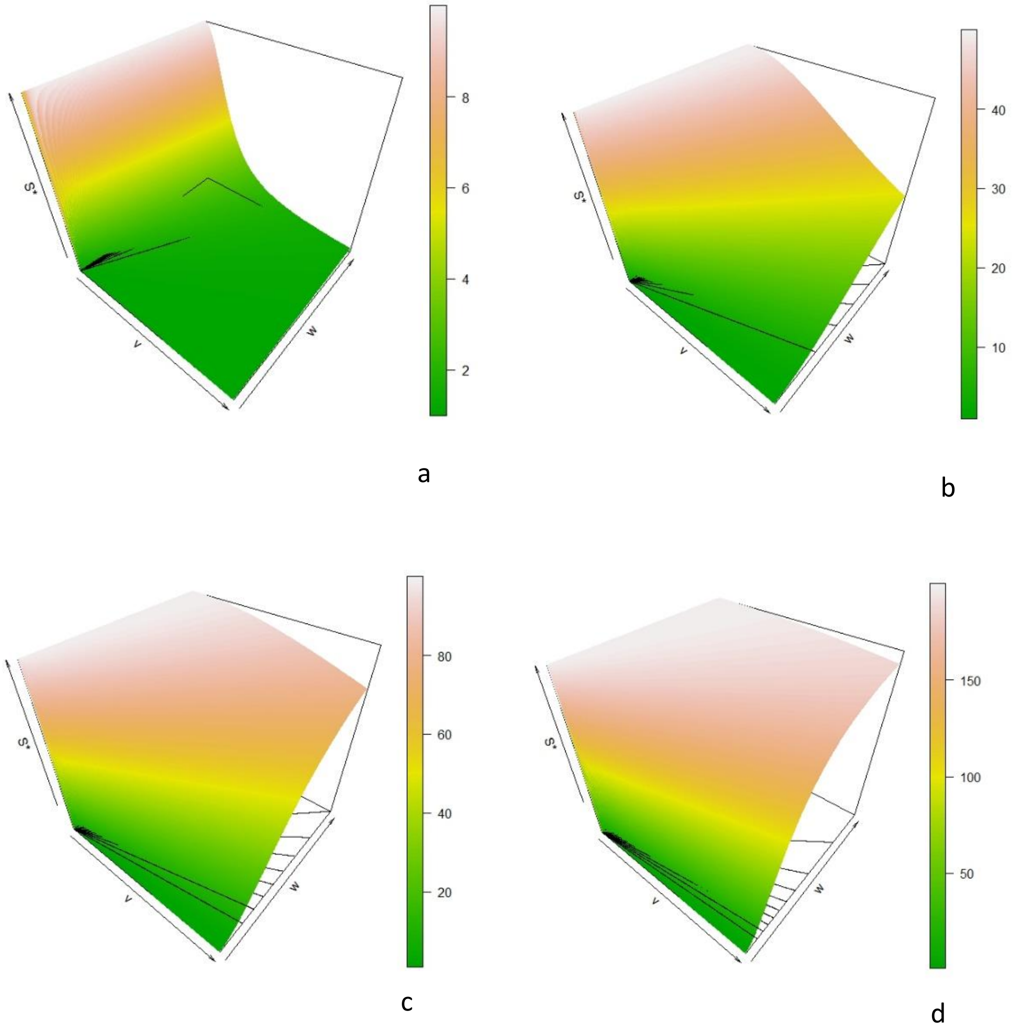

Figure 9.1. The optimal level of governance for an economy composed of (a) 10 regions, (b) 50 regions, (c) 100 regions and (d) 200 regions *(Source: authors' own computations)*. *In the figures,* $v \in [0; 2000]$ *and* $w \in [0, 20]$.

Figure 9.1 shows that the higher the number of regions R, the higher the need for centralization is, which is explained by the relative increase in the value of the global PG when R increases. In other words, an increase in R implies a lower relative value for the local PG and thus a relatively smaller importance of the heterogeneity of local PG values, leading to a tendency towards centralization. Figure 9.1 suggests that a relatively high heterogeneity in the local PG values is required for decentralization.

One key indicator for the design of agro-environmental instruments is the expenditures entailed by the policy. Here, we are interested in the evolution of the total transfers from society to farmers with the level of decentralization. Using relations (9.6) to (9.9), we find that the total transfers $T(S)$ is:

$$T(S) = \frac{1}{4k^2c}\left[\left(1 - \frac{S-1}{R-1}\right)v^2 + 2w^2S^2 + 2wvS\right] \tag{9.14}$$

The derivative of $T(S)$ relative to S leads to:

$$\frac{\partial T(S)}{\partial S} > 0 \Leftrightarrow S > \frac{v^2 - 2(R-1)wv}{2(R-1)w^2} \tag{9.15}$$

The transfers increase with the degree of centralization once S exceeds a threshold $\bar{S} = \left[v^2 - 2(R-1)wv\right] / \left[2(R-1)w^2\right]$. This threshold can be positive or negative. After examining the properties of the trinomial $v^2 - 2(R-1)wv - 2(R-1)w^2 = 0$, we conclude that $\bar{S}$ is higher than 1 (i.e., the lower level of governance) when $v \in \left[\left(R-1-\sqrt{R^2-3}\right)w; \left(R-1+\sqrt{R^2-3}\right)w\right]$, i.e., in the case where $v \in \left]0; \left(R-1+\sqrt{R^2-3}\right)w\right]$ given that $v > 0$ and $R \geq 2$. Otherwise, the threshold is negative, and the total transfers from society to the farmers always increase with the degree of centralization. Given the values of v in reality, this second case is likely to appear in most cases (see part 3 for a discussion on PG values for the specific case of agricultural wetland management).

Proposition 3: *The total transfers from society towards the farmers always increase with the degree of centralization when* $v \in \left]\left(R-1+\sqrt{R^2-3}\right)w; +\infty\right[$, i.e., *in most cases. Otherwise, in the case when* $v \in \left]0; \left(R-1+\sqrt{R^2-3}\right)w\right]$, *the total transfer decreases until* S *reaches the threshold* $\bar{S}$ *and increases after it. The minimal transfers are reached for* $S = \bar{S}$.

This result is consistent with some empirical results on the volume of public spending with decentralization (e.g., Arends 2017).

Finally, in the case of nonnull transaction costs, the optimal size of the government is:

$$S^* = \frac{4R(R-1)^2 w^2 + v^2 - 2ck^2 (R-1)^2 \dot{\tau}_S}{4(R-1)^2 w^2 + v^2} \quad (9.16)$$

The computations are available in appendix 9.B. The additional transaction costs faced by decentralized or centralized governments impact the optimal level of decentralization. If the transaction costs decrease with the size of the government, as is supposed by the fiscal federalism literature, the optimal size of the government is higher than that determined in (9.11). If the transaction costs increase with the size of the government, as is supposed by the fiscal federalism literature, the optimal size of the government is lower than that determined in (9.11). The introduction of transaction costs introduces the parameter costs of PG supply into the optimal size of the government. Following the fiscal federalism assumption that $\dot{\tau}_S < 0$, the higher the supply cost parameter is, the higher the interest in centralization is. The same appears for the deadweight loss rate k induced by taxation of the regional incomes.

Proposition 4: *The characteristics of the supply side and the deadweight losses impact the optimal degree of centralization only in the case of differentiated transaction costs among governments.*

9.2.2.3 Comparison of the two extreme cases

As a special case, we compare the welfare of the economy under full centralization ($S = R$, i.e., the actual CAP case) and full decentralization ($S = 1$) to identify the second-best government level. Under null transaction costs, the welfare of the economy under the governance of size-S governments is given by (9.10). In particular, we have:

$$W(1) = \frac{R}{2kc}\left(v(v+wR) + Rw^2\right) - \frac{R}{4kc}\left((v+w)^2 + w^2\right) \quad (9.17)$$

in the case of full decentralization and:

$$W(R) = \frac{R}{2kc}\left(\left(\frac{1}{2}v + wR\right)(v+2wR) - \left(\frac{1}{2}v + wR\right)^2\right) \quad (9.18)$$

in the case of full centralization. A comparison of (9.17) and (9.18) determines in which case regulators should prefer full centralization to full decentralization. The relation $W(R) > W(1)$ leads to:

$$W(R) > W(1) \Leftrightarrow (R-1)w > \frac{v}{2} \tag{9.19}$$

The details of the calculation are provided in appendix 9.C. The left-hand term $(R-1)w$ represents the global PG value captured outside the region where production occurs. In other words, it represents the marginal externalities generated by one unit of managed land in one region to the other regions. The right-hand side $v/2$ represents the average value of local PGs under the two types of land. In other words, it represents the marginal value attributed by the central government to the local PG due to information losses. Because the local PG value is v, $v/2$ also represents the difference in value between the real local PG value and the central government subjective value. Thus, the full centralization case leads to higher welfare when the value of the externalities generated by one unit of land is higher than the misjudged local PG value of one unit of land due to information losses.

Proposition 5*: Full decentralization of agro-environmental policies improves welfare if the externalities generated by the regional government to other regions are lower than the difference between the actual and expected values of local PG.*

The choice between centralized or decentralized provision involves a basic trade-off between the gains from the internalization of spillovers under centralization and the greater sensitivity of local outputs to heterogeneous preferences under decentralization (Oates 2005). The higher the global PG is, the greater the interest is in centralized governance. The higher the heterogeneity of the local PG value is, the greater the interest is in decentralized governance. This result is consistent with Oates' decentralization theorem (1972).

One can also compare the structure of the landscape under centralized and decentralized agro-environmental policy design. Under complete centralization, relations (9.8) and (9.9) indicate that $X_1(R) = X_2(R) = (wR + v/2)/2kc$, i.e., that the landscape would be homogenous. The homogeneous landscape indicates that the payments under full centralization are homogenous, which can easily be verified with (9.6) and (9.7). In contrast, relations (9.8) and (9.9) indicate that the landscape will be heterogeneous under complete decentralization, with $X_1(1) > X_2(1)$. A comparison of X_2 highlights that full decentralization decreases the level of X_2.[85]

[85] $X_2(R) - X_2(1) = \left(w(R-1) + v/2\right)/2kc > 0$

Comparing X_1 under full centralization and full decentralization highlights that decentralization decreases the level of X_1 when (9.19) is verified but increases it otherwise (when $0.5v > (R-1)w$). Finally, a comparison of the total amount of financed lands $X_1 + X_2$ under full centralization and full decentralization highlights the following:

$$\left(X_1(R)+X_2(R)\right)-\left(X_1(1)+X_2(1)\right)=\frac{w(R-1)}{kc}>0 \tag{9.20}$$

Proposition 6: *The total amount of financed lands decreases under full decentralization to the profit of more valuable lands, whose level increases if full decentralization increases the welfare.*

The decentralization of agro-environmental policy design would decrease the total amount of subsidized lands. Indeed, the regional governments would use their information to support the more valuable lands (i.e., X_1). However, the last unit of financed X_1 under decentralization incurs more costs for the farmers than the last unit of land under centralization. This means that regional governments would prefer to propose higher payments for the most valuable farmers, even if it reduces the total amount of financed lands.

9.3 Empirical application: abandonment of wetlands in Brittany

9.3.1 Provision of public goods from agricultural wetlands of the Odet watershed

In this section, we parameterized the theoretical model to the case study of wetland abandonment in the Odet watershed in Brittany (France). Wetland management is a good empirical counterpart to our theoretical model because wetlands face the risk of abandonment in Brittany and their agricultural management increases both local and global PG provision more than their abandonment, which leads to afforested wetlands in the long term (Bareille et al., 2017). Although the Odet watershed is not a NUTS2 region, we consider that watersheds are the empirical counterpart to our theoretical regions, which makes sense because the benefits of the local PG (e.g., water quality) are captured inside the watersheds where the PG provision occurs. Therefore, focusing on one specific watershed or on all watersheds of a considered region is the same if all benefits are captured inside the watershed boundaries and if the regional government has all the information on the heterogeneity of the local PG value. We assume that this is the case here because each of the 110 watersheds of Brittany is managed by local agencies to improve water quality, and we also assume that regional government representatives are part of these agencies and can inform the decentralized governments on the heterogeneity of the preferences and conditions inside each watershed. We consider that the global PG value is captured by all European Union inhabitants. Based on a realistic parametrization of the theoretical parameters, we examine the welfare emerging from the Odet watershed landscape under regional (Brittany), national (France) and central (European Union) governance.

The Odet watershed is a territory of 724 km², representing 2.64% of the size of the Brittany region (Figure 9.2). The territory consists of 27 municipalities and presents a density of 174 inhabitants per km². The main city of the watershed is Quimper, the third largest city of Brittany. Eight watercourses cross the watershed, and they are all grouped within the Odet coastal river. Agricultural wetlands represent 3,700 Ha, i.e., 5.1% of the watershed area. In 2014, 1,800 Ha of agricultural wetlands were abandoned in the Odet watershed (Figure 9.2).

The hydric and soil characteristics of agricultural wetlands provide an ecosystem distinct from other land types. Wetlands support the provision of several ecosystem functionalities contributing to water purification, flood control, biodiversity habitat and carbon sink. Based on benefit transfer functions and cost accounting, Bareille et al. (2017) find an estimated conservative value of 452 €/Ha for PGs provided by agricultural wetlands at the watershed scale. This value is computed as the difference of values of water filtration, fished salmon and trout, carbon sink and biodiversity habitat provided with and without agricultural management (Engel et al., 2008). Indeed, Bareille et al. (2017) consider that abandoned wetlands become afforested lands in the long term, which decreases PG provision compared to the agricultural management of wetlands (e.g., Pykälä 2003). This value is subdivided into 410 €/Ha for local PG (i.e., water quality and fishing) and 42€/Ha for global PG (i.e., carbon sink and biodiversity habitat).

The costs of agricultural production on wetlands incentivized farmers to turn wetlands into arable lands through drainage works. Since the drainage of wetlands has been forbidden in France since 1992, farmers are incited to sell or abandon their wetlands. In this context, farmers managing wetlands receive a payment of 120 €/ha thanks to an AECM (operation “Herbe_13” defined in Measure 10 of the 2014-2020 Rural Development Program for Brittany). Conditions of the AECM contract state that subsided areas should respect the maximum animal density of 1.4 per ha, a maximal nitrogen fertilization and the interdiction of pesticides and tillage. Despite this subsidy, abandonment of wetlands remains an issue in Brittany. Based on a wetland census of 2014 in Finistère (NUTS3 region), Bareille et al. (2017) determined that 46% of the agricultural wetlands were abandoned.

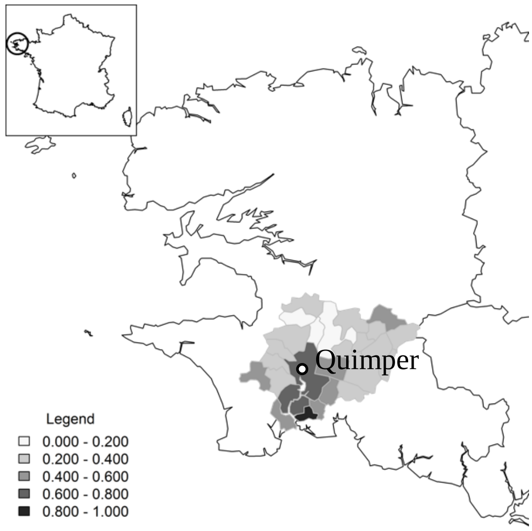

Figure 9.2. Wetland abandonment rate in the Odet watershed (source: Bareille et al., 2017)

9.3.2 Data and empirical model description

Contrary to the theoretical part, we account for the heterogeneity in the opportunity cost of wetland management across municipalities, assuming that each of the 27 municipalities represents a single farm ($i \in [1;27]$). We consider that all hierarchical governments are aware of this heterogeneity in the opportunity costs. In the absence of detailed costs, we calibrate the individual cost parameter of wetland management from the current observed levels of wetland abandonment X_i^o ($i \in [1;27]$, see Figure 9.1) and the Herbe_13 homogenous subsidy p^0 of 120€. Following the farmers' FOC, the cost parameters are given by $c_i = p^o / X_i^o$ (see appendix 9.D. for the calibrated values). This procedure leads to cost parameters whose levels decrease with decreasing distance to the seacoast, which is consistent with the usual theoretical and empirical results (Cavailhès and Wavresky 2003).

To consider the heterogeneous contribution of the wetlands to the local PG value, we use the results from the distance-decay literature. This literature states that the value of the local PG decreases with the distance between the consumer and the locality where the local PG is provided (see Pate and Loomis, 1997 for an application on wetlands). Therefore, we write the utility of the watershed (i.e., the region) as follows:

$$U_{regional} = \int_0^{\bar{d}} \left[\left(\mathbf{1} \frac{v_{water}}{d_i} + \frac{v_{fishing}}{d_i} + w \right) X_i \right] dd_i$$

where d_i is the distance in kilometers between the centroid of the municipality *i* to the centroid of Quimper, $\bar{d}$ is the distance between Quimper and the farthest municipality and $\mathbf{1}$ is an indicator function taking the value 1 (respectively 0) for municipalities located upstream (respectively downstream) Quimper.[86] Hence, all the wetlands of one municipality have the same value, which depends only on d_i. We parametrize v_{water} and $v_{fishing}$ using the unspatialized values of Bareille et al. (2017), such that:

$$v_{water} = 300 \cdot 17 \Big/ \sum_{i=1}^{17} \frac{1}{d_i}$$

and

$$v_{fishing} = 110 \cdot 27 \Big/ \sum_{i=1}^{27} \frac{1}{d_i}$$

yielding an average value for the local PG of 269 €/ha. The difference in the average values between our study and Bareille et al. (2017) is that they did not consider the different contributions between upstream and downstream wetlands. The sum of the two local PG values ranges from 89 to 1076 €/Ha over the 27 municipalities, implying that parameter v equals 3108 in the empirical and theoretical analysis.

In addition, we have for all wetlands $wR = 42\,€/\mathrm{Ha}$. Under the assumption that each European region derives the same utility for a global PG, we allocate the value between the region and the rest of the EU at the *pro rata* of inhabitant density, leading to a value of $w=0.009$. Given that there are R=281 regions in the EU, proposition 3 implies that the total payments always increase with marginal centralization in cases where $v \in \left]0;5\right]$ (see proposition 3), which is the case here.

We calibrate parameter *k* to 2.1871 such that the generated landscape represents the existing one under the AECMs (Figure 9.2). This implies that each 1€ spent for agro-environmental payment incurs 1.1871€ of deadweight loss in the examined watershed. Given the numerous

[86] As the water treatment factory is located in Quimper, the only valuable wetlands are located upstream of Quimper.

uncertainties in the transaction costs faced by the different hierarchical governments, we assume that the level of agency costs is identical for all governments.

The mathematical formulation of the empirical model is differentiated from the theoretical section with respect to the increase in the number of farmers (from 2 to 27), the specification of heterogeneous costs and the functional form for the heterogeneity of local PGs. We also consider that governments can propose either homogenous or heterogeneous payments.

9.3.3 Results

Table 9.1 presents the results of the empirical model in the cases of (i) European governance (full centralization), (ii) regional governance (full decentralization) and (iii) national governance (partial decentralization). We first assume no differences in the agency costs (i.e., the theoretical case of null transaction costs) and consider both homogenous and heterogeneous payments. Despite the numerous details on the supply side, the results clearly follow the theoretical analysis.

First, we find that the possibility of heterogeneous payments increases the efficiency of agro-environmental payments. Indeed, while still under the condition of fully centralized policy-making, we find that European governance with heterogeneous payments improves welfare by 28% compared to the current centrally determined homogenous payments. This result is not developed in the theoretical analysis because it is already well known in the case of agro-environmental payments (Latacz-Lohmann and Van der Hamsvoort 1997). However, we do find that such gains are amplified with decentralization. Indeed, we find that, compared to the actual case, regional government would increase welfare by 25.7% in the case of homogeneous payments and by 66% in the case of heterogeneous payments. Comparing regional and central governance with heterogeneous payments, we find that regional governments would still increase welfare by 30%, which is more important than the single gains from a move towards heterogeneous payments, highlighting the interest of decentralization in realistic environmental issues. As expected, we find that the payments under regional governance would decrease sharply, divided by 56% (64.5%) in the case of heterogeneous payments (homogenous payments), with the consequence of an increase in wetland abandonment. A regional government in charge of agro-environmental payments for wetland management would thus decrease the agro-environmental budget to increase the regional utility. However, the evolution of abandoned wetlands is heterogeneous across municipalities, and the abandonment rate decreases in the 8 upstream municipalities closest to Quimper (Figure 9.2). This suggests that regional governments would finance the most valuable lands to the detriments of the other lands (especially since the agro-environmental budget decreases). Indeed, by integrating all information on the heterogeneity of local PGs but ignoring global PG spillovers, regional governments consider that downstream municipalities produce lower levels of PGs than the central government expects. This leads to a sharp decrease in the payments to the downstream municipalities.

We find similar patterns for national governance. Overall, national governance increases welfare by 26.4% in the case of homogeneous payments and by 67% in the case of heterogeneous payments. This implies that national governance yields better performance than regional and central governance. We can explain this feature because national governance integrates more global PG spillovers while still integrating most of the heterogeneity of the local PG. This implies that national governments spend relatively more than regional governments, most of these additional funds being targeted to downstream municipalities (see appendix 9.E.), with the consequence that the welfare gains are captured outside of the watershed boundaries. This suggests that the decentralization of agro-environmental payments represents an option worth exploring, either partially or totally. This, however, must be further evaluated to observe whether such a result holds in the presence of transaction/coordination costs (see the sensitivity analysis in the next section). The relative value of the two types of PG is a major driver of the results. The small difference between the full decentralization and the partial decentralization is due to the relatively high value of the local PG with respect to the value of the global PG. We provide a sensitivity analysis on that point in the next section.

Table 9.1. Summary of results on the Odet watershed

	Regional government		**National government**		**European government**	
	Hom. Pay.	Het. Pay.	Hom. Pay.	Het. Pay.	Hom. Pay.	Het. Pay.
Welfare	670,879.30	888,201.35	674,408.94	891,434.70	533,731.40	684,391.65
Welfare evolution (% actual case)	0.26	0.66	0.26	0.67	-	0.28
Payment level (€, average)	72.41	68.36	74.45	70.03	120.00	102.42
Payment level (average/std)	-	0.69	-	0.69	-	0.68
Total expenditures (€)	242,627.21	341,992.60	256,491.54	357,777.61	666,492.45	777,251.41
Abandonment rate (%, average)	0.63	0.65	0.62	0.64	0.39	0.53
Abandonment rate (average/std)	0.11	0.33	0.12	0.34	0.19	0.69

9.3.4 Sensitivity analysis on global public good value

In this section, we present the results of a sensitivity analysis on the global PG values. We modify the value by multiplying w by a coefficient $0 \leq a \leq 5$ with *0.25* steps. The maximum value for the global PG that we account for in the sensitivity analysis is 210€/ha, which is still lower than the average local PG value.

Figure 9.3 shows the welfare effects emerging from such a sensitivity analysis in the case of heterogeneous payments. We find that, even when the global PG value decreases by 75%, the national government remains the best governance level. EU governments would become the best if the global PG increased by more than three times. In this case, the level of abandonment would be 37% for central governance and 63% for national governance.

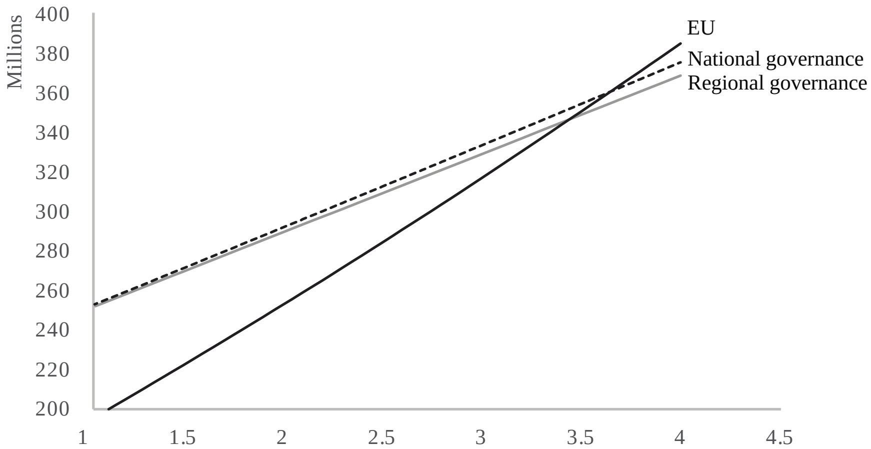

Figure 9.3. Welfare for different values of the global PG (as a % of the original welfare) under regional (gray line), national (dotted line) and EU governance (solid black line).

9.4 Discussion and conclusions

Our analysis, albeit relatively simple, provides some theoretical background for the potential decentralization of the design of agro-environmental payments in the future CAP. Indeed, by integrating the complexity of PG provision from agriculture, our results show that total or partial decentralization could improve the welfare of the whole economy. We find that the benefits of decentralization increase as the heterogeneity of local PG values increases and as the spillovers (the global PG values) decrease. This result is consistent with Oates' decentralization theorem (1972) but within a given jurisdiction. We find that, in most cases, the total transfers from society to farmers would decrease with decentralization, in line with the idea that local governments can do "more with less" (Benassy-Quere et al. 2007; Arends 2017). This is a major result given the sensitivity of this question for European and agricultural stakeholders. We also contribute to the literature on fiscal federalism with this result because the links between public spending and decentralization remain an open question (Arends 2017). We find that the optimal level of governance depends on the farmers' program and on the deadweight losses only when hierarchical governments face different agency costs. We also find that the total amount of financed lands decreases under (full) decentralization to the profit of more valuable lands, which increase if (full) decentralization increases the welfare. Finally, we find that the full decentralization of agro-environmental policies improves welfare if the externalities generated by the regional government to other regions are lower than the difference between the actual and expected values of the local PG.

Our empirical application provides a numerical illustration of the potential gains from the CAP reform. In a simple application to agricultural wetlands facing the risk of abandonment, the landscape resulting from decentralized governance always improves welfare compared to centralized governance. Comparing regional and national decentralization, it appears that national governance is the best level of decentralization. In total, a move from homogenous centrally determined payments to heterogeneous nationally determined payments would lead to welfare gains of 67%. In fact, 60.5% of these gains are explained by the heterogeneous payments, in line with the quantification of welfare gains from a move towards more heterogeneous regulations, from either the AECMs (Latacz-Lohmann and Van der Hamsvoort 1997; van der Horst 2007) or other instruments (Perino and Talavera 2013). The informational advantage of decentralized governments still contributes to 39.5% of these gains. Further decentralization would, however, decrease the welfare compared to national governance. Regional decentralization would decrease welfare reached by national governance by 5.2% in the case of homogeneous payments but by only 3.7% in the case of heterogeneous payments, highlighting the crucial links between decentralization and the possibility of heterogeneous payments. We also find that decentralized governments would decrease the agro-environmental budget by 200% to 300%. In any case, national decentralization in the spirit of the EC's proposal COM(2018) 392 is of interest for the retained PG values. Our sensitivity analysis confirmed that such decentralization would be beneficial even if PG values increase by a sensible percentage *ceteris paribus*.

Our empirical results are, however, subject to some limitations. First, the abandonment of wetlands is a specific example with the advantage that its agricultural management increases local and global PG provision at the same time. We can imagine cases where payments would improve the provision of one type of PG but decrease the provision of the other. Such a context could lead to competition between hierarchical governments, which is nonexistent in our case. Second, our results hold under the assumption that the single source of revenue from wetlands is the subsidy. However, wetlands can also generate market revenues for farmers. Regarding their role as pasture lands, agricultural wetlands can benefit farmers depending on milk and feed prices and fixed input dotation. More generally, extensive dairy farms can valorize these lands without any subsidies. As a result, our simulation leads to more contrasted landscapes than would emerge in reality. Third, the results depend on the valuation of the considered PGs, which are subject to their own limits (see Bareille et al. 2017 for a complete discussion). Our spatialization of the local PG values is also based on rough assumptions from the distance-decay literature, which can bias our welfare quantification.

The main interest of this research is its use of the fiscal federalism literature as a way to analyze the potential future reform of the CAP. We introduce two motives to model the advantages and disadvantages of different levels of government, namely, the different information on PG values held by the hierarchical governments and the different agency costs of managing public money. These two concepts are part of the second-generation theories on fiscal federalism (Oates 2005). The heterogeneous information partly explains why one government is more suitable to implement specific instruments (Boadway 1997). In our framework, the information of the heterogeneity of local PG values leads to an advantage for the local government, but the knowledge of global PG preferences of the central government allows the internalization of externalities. The differentiated agency costs have usually been examined in the fiscal literature considering that transaction costs face economies of scale, giving an advantage to the central government (Oates 1999). However, we have here considered that the transaction costs are due to not only economies of scale but also communication between different agencies (e.g., European and regional agencies). Indeed, as interestingly suggested by Crémer et al (1996), the central government can spend resources to obtain information on local conditions. This is precisely the case of the existing CAP, where the EC subsidizes farmers based on the average estimated opportunity costs reported by the regional agencies (Beckmann et al., 2009). Both economies of scale and communication costs have been observed in the literature on the effectiveness of agro-environmental payments. For example, economies of scale in transaction costs faced by the English administration have been highlighted by Falconer et al. (2001), and Weber (2015) found that more than 50% of the transaction costs are due to coordination between the EC, national governments and regional governments. To our knowledge, no study has estimated the resulting transaction costs when considering both economies of scale and communication costs. This gap in the literature is a large drawback to studying the effectiveness of the decentralization of agro-environmental payments, as already emphasized by Beckmann et al. (2009) and Mettepenningen et al. (2011). Our sensitivity analysis on agency costs illustrates this lack of information.

Our theoretical results are, however, subject to some limitations, notably due to our assumptions. First, we have considered that all hierarchical governments have the same level of information on the farmers' opportunity costs but different levels of information on the heterogeneity of the local PG values. This feature is relatively unusual in fiscal federalism, as the literature usually assumes that hierarchical governments face different levels of information on costs. However, as explained, this feature is justified in the case of the CAP due to the coordination between regional agencies and the central government on farmers' opportunity costs. This also explains why we do not explicitly introduce land constraints and heterogeneous costs in the theoretical analysis and focus on the heterogeneity of information on local PG values. Second, we consider that both firms and residents are immobile, whereas the literature on fiscal federalism considers that firms and/or residents are mobile. We justify our choice regarding farmers' immobility given the lower sensitivity of farmers' labor to short-term changes, with farmers' mobility occurring rather in the long term when farmers start their business, leave agriculture or retire (Gaigné and Bougherara, 2008). Similarly, given the shares of the agro-environmental budget in the total and regional budgets, it is unlikely that residents would change locations for agro-environmental tax raising. Third, we have also made restrictive simplifications with the reality of European agro-environmental payments. For example, we have considered that each regional/national government could be in charge of raising its own agro-environmental budget. However, for the time being, all states/regions contribute to the European budget proportionally to their wealth and development levels, and the European budget is then split between the European objectives (including the agro-environmental budget). As this decision is made at the European level, the regions have no impact on this budget, and one can even consider that the agro-environmental budget is exogenous. This form of organization where regional or local governments are constrained by central directives is in fact close to what Inman (2003) called “administrative federalism”. Thus, decentralization with an endogenous agro-environmental budget may not be the type explored by the EC for the following CAP reform. Decentralization of agro-environmental payments with an exogenous budget is more likely to emerge in the short term. A second restrictive simplification is that we have not considered that the regional governments and agencies co-financed 25% of the agro-environmental payments. Given that AECM decisions are made at the European level, this simplification does not impact our results if regions are homogenous. However, our framework is obviously restrictive on that point, as the EU is constituted by heterogeneous states and regions (for numerous reasons going beyond the agriculturally provided PGs). This restrictive assumption implies that we ignore that residents can “vote with their feet” (Tiebout 1956). This motive is, however, a central point of the fiscal federalism literature, as it could induce competition between regional governments (Tiebout 1956; Oates 1972; Besley and Coate 2003; Rhode and Strumpf 2003). We do not observe such competition here, but future studies could investigate this possibility.[87]

Fiscal federalism has also considered additional reasons for the effectiveness of decentralization. Most studies address political economy issues (Besley and Coate 2003). Explicit consideration of the objective functions of the governments and their representatives provides interesting justifications for centralization and can explain the existing expenses in the CAP framework. Indeed, it is probable that the two governments do not weight the farming population the same way (Bougherara and Gaigné 2008), potentially leading one government to favor farmers’ revenues rather than environmental outcomes. Here, we have decided not to introduce farmers’ revenues in the welfare maximization. Their introduction could notably lead to government competition if farmers’ profits are not weighted the same way in their objective function.[88] In particular, it is probable that central governments face a higher aversion for inequality among farmers (leading them to favor homogenous subsidies) as well as among regions (potentially leading to homogeneous budgets). Future works could study strategic interactions between regions and between government levels in a more decentralized context

[87] Note that the AECMs are specific to each region, in agreement with the regional needs. One can thus consider that the EU has information on the heterogeneity of preferences across regions but that it ignores the heterogeneity of local PG values inside a given region.

[88] It is also possible that the weight the local government gives to farmers would lead it to deviate from its objectives due to “reputational effects”, potentially leading to corruption (Oates 2005).

(e.g., moral hazard and adverse selection – Epple and Nechyba, 2004). Such questions should be of interest in the analysis of the decentralization of agro-environmental payments.

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9.6 Appendices

Appendix 9.A. Proof of proposition 2

The FOC of welfare (9.10) according to the size of the governments is:

$$\frac{\partial W(S)}{\partial S}=\frac{R}{2kc}\left[\begin{array}{l}\left(-\frac{1}{2}\frac{v}{R-1}+w\right)(v+wR)+\left(\frac{1}{2}\frac{v}{R-1}+w\right)wR \\ -\left[\left(-\frac{1}{2}\frac{v}{R-1}+w\right)\left(\left(1-\frac{1}{2}\frac{S-1}{R-1}\right)v+wS\right)+\left(\frac{1}{2}\frac{v}{R-1}+w\right)\left(\frac{1}{2}\frac{S-1}{R-1}v+wS\right)\right]\end{array}\right]=0$$

which is equivalent to:

$$\left(-\frac{1}{2}\frac{v}{R-1}+w\right)\left(v+wR-\left(1-\frac{1}{2}\frac{S-1}{R-1}\right)v-wS\right)+\left(\frac{1}{2}\frac{v}{R-1}+w\right)\left(wR-\frac{1}{2}\frac{S-1}{R-1}v-wS\right)=0$$

The successive factorization and developments of the relationship lead to the following:

$$\left(-\frac{1}{2}\frac{S-1}{(R-1)^2}v^2\right)+2w^2(R-S)=0$$

Multiplying this last expression by $(R-1)^2$ leads to:

$$4Rw^2(R-1)^2+v^2=\mathrm{S}\left(4w^2(R-1)^2+v^2\right)$$

which is equivalent to relation (9.11), leading to proposition 2.

Appendix 9.B. Proof of proposition 4

The FOC of welfare (9.10) according to the size of the governments is:

$$\frac{\partial W(S)}{\partial S}=\frac{R}{2kc}\left[\begin{array}{l}\left(-\frac{1}{2}\frac{v}{R-1}+w\right)(v+wR)+\left(\frac{1}{2}\frac{v}{R-1}+w\right)wR\\ -\left[\left(-\frac{1}{2}\frac{v}{R-1}+w\right)\left(\left(1-\frac{1}{2}\frac{S-1}{R-1}\right)v+wS\right)+\left(\frac{1}{2}\frac{v}{R-1}+w\right)\left(\frac{1}{2}\frac{S-1}{R-1}v+wS\right)+2k^2c\dot{\tau}_S\right]\end{array}\right]=0$$

which is equivalent to:

$$\left(-\frac{1}{2}\frac{v}{R-1}+w\right)\left(v+wR-\left(1-\frac{1}{2}\frac{S-1}{R-1}\right)v-wS\right)+\left(\frac{1}{2}\frac{v}{R-1}+w\right)\left(wR-\frac{1}{2}\frac{S-1}{R-1}v-wS\right)-k^2c\dot{\tau}_S=0 \quad \text{(A6)}$$

The successive factorization and developments of the relationship lead to the following:

$$\left(-\frac{1}{2}\frac{S-1}{(R-1)^2}v^2\right)+2w^2(R-S)-k^2c\dot{\tau}_S=0$$

Multiplying this last expression by $(R-1)^2$ leads to:

$$4Rw^2(R-1)^2+v^2-2k^2c\dot{\tau}_S=\mathrm{S}\left(4w^2(R-1)^2+v^2\right)$$

which is equivalent to relation (9.16), leading to proposition 4.

Appendix 9.C. Proof of proposition 5

The difference between (9.18) and (9.17) leads to:

$$W(R)-W(1)>0 \Leftrightarrow \frac{R\left(\frac{1}{2}v+wR\right)(v+2wR)-R(v+w)(v+wR)-(wR)^2}{2kc}$$
$$+\frac{R}{4kc}\left((v+w)^2+w^2-2\left(\frac{1}{2}v+wR\right)^2\right)>0$$

which is equivalent to ($R/2kc>0$):

$$\left(\frac{1}{2}v+wR\right)(v+2wR)-(v+w)(v+wR)-Rw^2+\frac{1}{2}(v+w)^2+\frac{1}{2}w^2-\left(\frac{1}{2}v+wR\right)^2>0$$

Trivial calculations lead to the following:

$$4(R-1)^2 w^2 > v^2$$

As the root square function is strictly concave, this last relation is equivalent to relation (9.19).

Appendix 9.D. Map of cost parameters

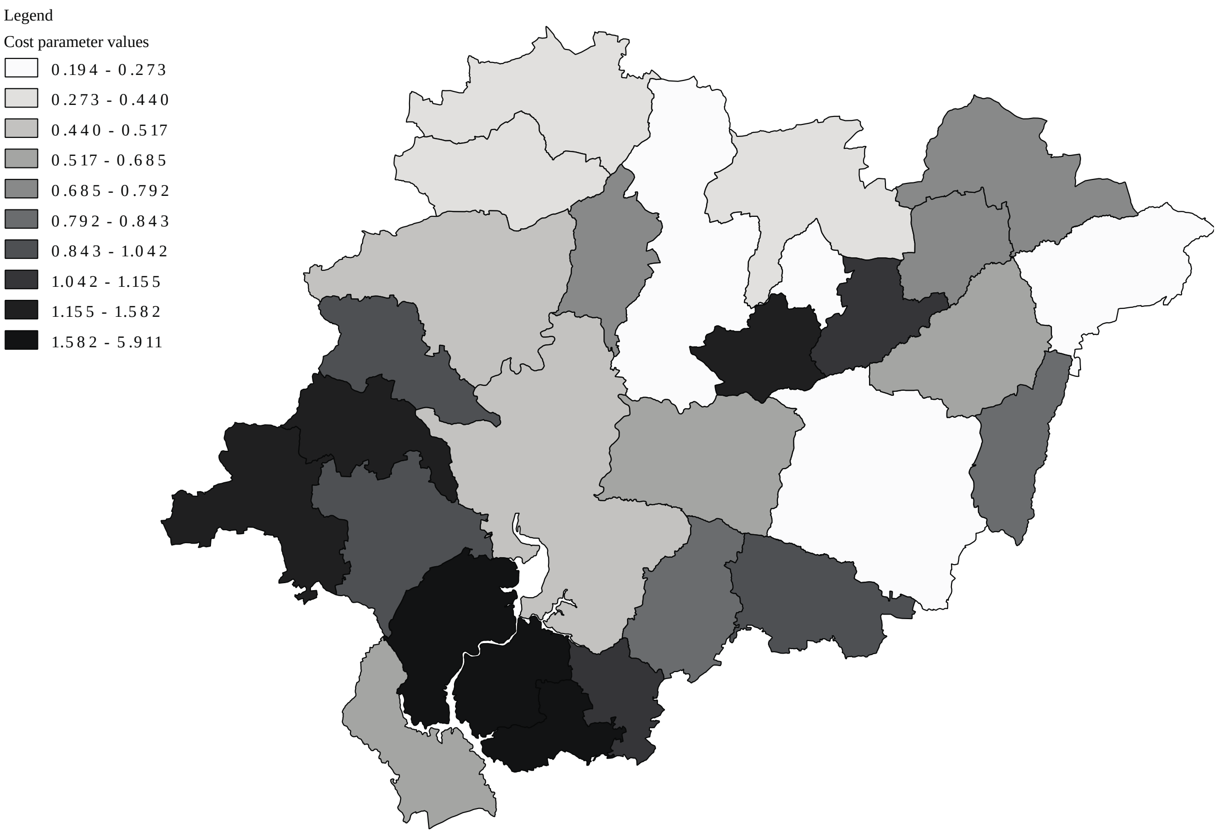

Figure 9.A1. Map of cost parameters in the Odet watershed

Appendix 9.E. Maps of wetland abandonment rate under different levels of governance.

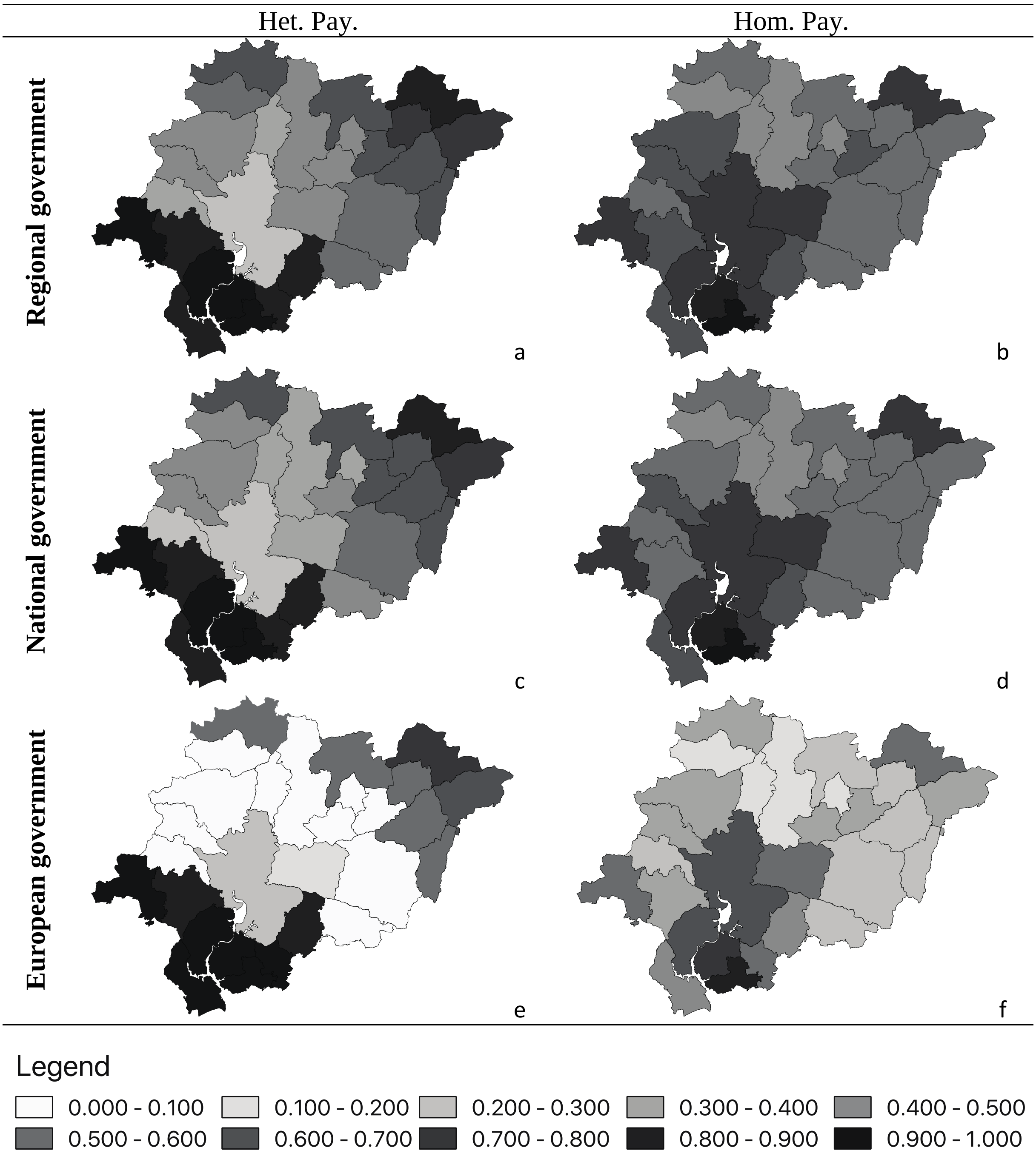

Figure 9.A2. Results of simulations on regional, national and EU governance (rows) with homogenous and heterogeneous payments (columns) with τ=0 (*Source: authors' own computation*).

CHAPTER 10. DISCUSSION OF THE GEOGRAPHICAL SCALE OF THE DEMAND FOR ENVIRONMENTAL SERVICES PROVIDED BY AGRICULTURAL MANAGEMENT

10.1 Principal contributions

The purpose of this second part of the PhD thesis was to investigate how geographical scale of demand for jointly provided public goods (PGs) from the farmers' environmental services may influence policy design. This question was raised during the writing of the PhD after my contribution to work package 4 of the European project H2020 PROVIDE. In that package, I valued, jointly with Jules Couzier and Pierre Dupraz, the environmental services provided by the farmers managing wetlands in Brittany. In this particular context, we use valuation methods based on benefit transfer and avoided costs. We found that the sum of the jointly provided local PGs were valued about ten times more than the sum of the jointly provided global PGs. We also showed that most of the identified spatial heterogeneity was due to the local PGs, notably in relation to the spatially heterogeneous demand for water quality (Bareille et al., 2017). It appears that such results are in line with the literature on the valuation of local and global PGs (Johnston and Ramachandran, 2014; Jørgensen et al., 2013; Logar and Brouwer, 2018; Schaafsma et al., 2013). Part 2 of this thesis investigated the impact of such properties on the design of environmental policies, here applied to agriculture. This part was divided into three chapters that aimed to provide new insights regarding this general research question.

First, Chapter 7 illustrates the externalities induced by trade where an exogenous shock induces shifts in supply and demand for different goods and services worldwide. In particular, we valued the induced land-use changes, as well as the changes in carbon emissions and the applications of fertilizer and pesticides. This macroeconomic assessment illustrates that a national policy targeting a local PG (here the pollution caused by pesticide applications) affects the provision of global PGs (or public bads). If these public goods are not accounted for, unexpected externalities are generated. Our major contribution in Chapter 7 is related to the literature on pesticide application, and more particularly on pesticide taxation schemes (see Finger et al., 2017 for a review). Indeed, the literature has mainly focused on the impacts of such schemes on pesticide reductions and on the related opportunity costs at the farm level. It ignores such induced impacts. Our results highlight that the global market effects induced by a national policy reduce the efficiency of such a policy by modifying crop and pesticide prices. However, our results indicate that such effects are limited. The induced impacts mostly concern other environmental PGs, in this case, global carbon emissions and national and global fertilizer applications. This illustrates that the benefits of pollution control for the citizens in a given region depend on regulatory and private production activities both within and outside the jurisdiction. Even if I did not examine this issue in the present thesis, this feature could lead to a “race to the bottom” where local governments would set lax environmental standards to decrease the costs of pollution controls for firms, a tactic that would result in inefficiently high levels of pollution (Harstad and Mideksa, 2017). For the purpose of our macroeconomic assessment, we also provided original estimations of pesticide applications at the French regional level, notably for agricultural products that are typically ignored, i.e., forages and vineyards; these products still account for a large portion of national applications of pesticides. The aggregation of farms at the regional scale displays higher elasticities than do usual estimates at the farm level. Our work is limited by the fact that if we have estimated economic welfare for producers and consumers, we have considered neither the welfare arising from environmental and health benefits nor the damages resulting from reduced national pesticide applications or increased carbon emissions. If the latter could rather easily be addressed (see Bareille et al., 2017 for example), the former constitutes a research area in itself (Wilson and Tisdell, 2001). Nevertheless, the addition of local and global PG values into the CGE model would definitely improve our contribution, notably by helping to determine the optimal policy design.

Chapter 8 contributes to the literature on the valuation of agricultural externalities. Chapter 8 stands among other works on the hedonic valuation of PGs by measuring the value of the agricultural externalities at two scales, namely the usual infra-municipal scale and the less-explored extra-municipal scale. This objective is motivated by the fact that other works tend to ignore the fact that the different agricultural activities jointly generate several local PGs and that these PGs affect residents differently over space, notably because of the distance-decay effect (Schaafsma et al., 2013). Specific spatial econometric models applied to residents’ house prices enable us to disentangle the values of the agricultural externalities between the two scales. We find that swine activities present negative effects at all scales whereas dairy cattle activities, including grassland management, present negative effects at the infra-municipal scale but positive ones at the extra-municipal scale. From a resource allocation perspective, this suggests that supports for cattle activities in nearby neighbourhoods are substitutes, whereas supports for swine and poultry activities are complements. This piece of research more generally contributes to the literature on heterogeneous PG value over space (Bateman et al., 2006; Lanz and Provins, 2013; Schaafsma et al., 2012). However, we are unable to explain whether these distinct values are due to spatially differentiated biogeochemical characteristics of the region, to spatial dimensions of ecosystem functioning (Lewis et al., 2015), to substitution effects (Schaafsma et al., 2013) or only to distance decay effects (Bateman et al., 2006; Jørgensen et al., 2013). Despite its importance for the optimal design of policy (see the Samuelson conditions), this Chapter is the only PG valuation exercise presented in the PhD manuscript. It highlights that even if methods are relatively well established, there is room for research focusing on the specificities of PG demand, as illustrated here by the spatial components of local PG demand.

Chapter 9 presents the optimal decentralization of agro-environmental policy decision-making when hierarchical governments have different degrees of information on PG demands and when the same environmental services jointly support different local and global PGs. In a sense, Chapter 9 examines some political economy properties of the Samuelson conditions. The chapter does not present the first-best conditions where all PG demands are considered but rather seeks to determine the second-best strategy. Such an endeavour is rather new in agricultural economics despite the public money allocated to agro-environmental policy. We contribute to the literature on environmental federalism by considering the specificities of the agricultural supply and ecosystem functioning, in particular with regard to the joint production of several PGs, to the heterogeneity of production conditions (and local PG demand) across space and, to a lesser extent, to farmers' total land constraints. Indeed, the complexity of ecological systems implies that policy decisions concerning a specific kind of pollution in a given jurisdiction generally indirectly affect more than one ecological component, although the effect is sometimes complex, time-lagged and difficult to predict. Previous works on environmental federalism have usually relied on a single type of PGs, which, according to Dalmazzone (2006), presents properties of non-renewable resources where depletion has little impact on the stock of other resources or on the rest of the ecosystem. This is not the case with the farmers' management of the agroecosystem. We find, notably, that the decentralization of European agro-environmental payments would improve welfare but decrease environmental quality. The local governments target the most valuable wetlands for local PG provision but may undervalue global PGs. Our settings indicate that decentralization would result in a decrease in total payments, contributing to the debates on public spending and decentralization

(Arends, 2017). We also find that the consideration of heterogonous payments would improve the efficiency of agro-environmental payments more than decentralization would. This finding highlights the importance of research on spatial targeting and auction mechanisms (Latacz-Lohmann and Van der Hamsvoort 1997; van der Horst 2007). As already stressed in Chapter 9, our work was, however, subject to several limits. We could add that in line with the limits of Chapter 7, our work examines the decentralization of a single instrument: agro-environmental subsidies based on farmers' effort (White and Hanley, 2016). Such an environmental policy design would, however, require the setting of multiple instruments to handle different environmental objectives (Bennear and Stavins, 2007). The introduction of these different instruments could lead to different conclusions, notably if several hierarchical governments implemented these different instruments.

10.2 Policy assessment and design

The second part of the PhD highlights that the marginal benefits of farmers' provision of environmental services depends, in a complex way, on the demand for the jointly produced public goods. Our three chapters highlight that the question "who are the beneficiaries of the environmental service?" is already a tricky one, preventing the easy application of the Samuelson conditions. Several insights for existing agro-environmental policy can be drawn from our conclusions.

First, Chapter 9 illustrates that the optimal policy design depends on the demand for the jointly provided PGs by the considered environmental service. This result is obvious (it is the purpose of the Samuelson proposition) but, in the cases of environmental services provided by agriculture, this trivial result is often overlooked. A representative example of this issue is the European agro-environmental policy. Indeed, the agro-environmental measures of the CAP second pillar are settled based on the average opportunity costs of the farmers, without any references for PG demand, as highlighted by the European Court of Auditors (European Court of Auditors, 2011). In the case of agricultural management of wetlands, the proposed agro-environmental payments is settled at 120 €/ha, while the conservative benefits from such management are valued at 440 €/ha (Bareille et al., 2017). This example, despite its limits, illustrates the ample room for improvement in agro-environmental design by paying deeper attention to PG demand.

Chapter 7 also illustrates how French policymakers underestimate the effect of trade on policy design. Indeed, both the French president, Emmanuel Macron, and the (recently) former French minister for the environment, Nicolas Hulot, have the ambition to reduce French pesticide applications by either introducing a specific ban (on glyphosate) or by increasing the pesticide tax and, at the same time, they want to reduce national and global carbon emissions to respect the Paris agreement to “make our planet great again”. Our results illustrate that these two environmental goals would not be attained with the same instrument, underlining the need to apply the Tinbergen principle of “one goal, one instrument”. In the illustrative case of Chapter 7, additional instruments to limit carbon emissions could be cap-and-trade instruments, for example (Coase, 1960), which would require trans-national cooperation. Indeed, the literature on environmental federalism, and particularly Chapter 9, illustrate that the optimal level of governance should be settled so that the environmental benefits would be contained within the jurisdictional boundaries. There are no such boundaries in the case of climate change. Such global environmental objectives thus require international cooperation, which generates multiple research questions (Stern, 2008) that are beyond the scope of this PhD.

Even if such international cooperation is doubtful, Chapter 9 illustrates that the increase in global PGs values privilege centralization. However, in the actual context, we find that a decentralization of agro-environmental policy design in Europe would be beneficial. Chapters 8 and 9 illustrate that this decentralization should not be settled at a too-narrow scale, as even local PGs could generate externalities from one municipality to another. This illustrates the complexity of the policy design: the literature suggests that municipalities have the highest levels of knowledge of both supply and demand for PGs among their residents, such that one could argue that they are well suited to design environmental policy (Deacon and Schläpfer, 2010; Droste et al., 2018; Lanz and Provins, 2013); however, municipalities would also generate externalities for neighbouring municipalities. These transboundary forms of pollution are the subject of many empirical works, notably on water pollution (Eichner and Runkel, 2012; Ogawa and Wildasin, 2009; Sigman, 2005, 2014), and these approaches could be extended to other PGs (Droste et al., 2018; Perrings and Halkos, 2012). The propositions for the next CAP reform to decentralize agro-environmental design at the national level (see COM(2018) 392) are supported by our results in Chapter 9. The results highlight that such decentralization could be even more efficient in cases where heterogeneous payments between farmers are allowed.

10.3 Future research

Our framework could be improved to treat related questions in future research. First, both Chapters 7 and 9 aimed to contribute to optimal policy design, but they ignore the environmental benefits in the first case and rely on functions that are perhaps overly simplified in the latter case. In particular, the integration of the demand functions for fertilizer and pesticide applications and carbon emissions could determine the optimal level of the pesticide tax rate. This addition would, however, require additional research on the demand for reductions in fertilizer, and particularly pesticide, applications.

Second, as already stated, we have examined some impacts of the demand specificities on the design of a single instrument (either a pesticide tax or an agro-environmental subsidy). The environmental policy design would, however, require the setting of multiple instruments to handle the different environmental objectives, which could generate strategic behaviours from hierarchical or horizontal governments. Future research should focus more on the setting of this policy mix when considering different demands for the different jointly provided PGs.

Third, although I have paid attention to some properties of the geographical scales of demand, there remain many uncertainties about PG demand. For example, we have assumed linear utility functions. However, basic economic theory suggests that consumers face decreasing marginal utility with the consumption of any good. This situation explains, for example, the spatial heterogeneity of the demand for PGs, as the distance between two neighbouring sites influences the degree of substitutability (Jørgensen et al., 2013; Schaafsma et al., 2013). One could argue that the scale effects in Chapter 8 could be related to these substitution effects, explaining why we often have opposite effects in the two distinct scales. We have also assumed separability between different public goods while, like private goods, different PGs present different degrees of substitutions. As in Chapter 6, we also highlight that we have considered a single utility function even though individuals have specific preferences. Such properties definitely influence the optimal design of PG provision, as illustrated by the Lindahl conditions (Foley, 1970; Lindahl, 1958). The examination of such properties is possible using revealed preference methods such as the hedonic pricing method.

Finally, if the presented chapters highlight the additional gains that could be obtained from the examination of the scale and spatial properties of the PG, a dimension that has been largely ignored in the past (Johnston et al., 2002; Smith, 1993), the temporal properties of the demand also affect the optimal policy design. In particular, ecological functions have different and variable temporal cycles; some of the impact of may manifest with a lag of seasons, years or decades. The integration of such features would definitely improve the design of agro-environmental policies. One question could be to examine the decentralization efficiency when hierarchical governments face heterogeneous discount rates for the same global PGs.

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CHAPTER 11. CONCLUSION

The concept of ecosystem services is an interdisciplinary one, referring to "the benefits people obtain from ecosystems" (MEA, 2005). My PhD thesis aimed to study ecosystem services from an economic perspective, particularly by analysing the specificities of supply and demand for public goods influenced by the flows of ecosystem services. The economic literature has mainly investigated the concept of ecosystem services by focusing on their monetary valuation, with limited reference to the behaviour of the agents. By contrast, the emerging literature on payment for environmental services has paid deeper attention to the behaviour of agents with regard to the management of ecosystem services. This literature considers the environmental service as corresponding to the ecosystem manager's action that leads to the modification of the flows of ecosystem services, and the ecosystem manager is considered to respond to economic incentives. The literature on payment for environmental services is, however, primarily concerned with the empirical measure of the additionality of the payments, i.e., with the specificities of the supply of environmental services; however, there are few references to the specificities of the demand for environmental services.

By contrast, I have paid deeper attention to the specificities of the demand for environmental services, particularly in the case of agriculture. If, like the literature on payment for environmental services, I still considered farmers to be the suppliers of environmental services, I have considered two types of consumers. The first type of consumer is the farmers themselves (i.e., the suppliers themselves). Indeed, there is a literature on measuring the productivity of productive ecosystem services that suggests that farmers demand some of these services. However, there is a lack of evidence that farmers consciously manage the provision of ecosystem services in order to benefit from them. The second type of consumer corresponds to non-farming agents, i.e., consumers who are not suppliers of environmental services. This second category consumes the public goods provided by the farmers who manage ecosystem services. This category is often considered when examining the effectiveness of a public (agro)environmental intervention. The originality of my research efforts regarding this second category of consumers is to consider that the consumers of a single environmental service may be localized all over the world because the supply of the environmental service influences the joint provision of local and global public goods. If most of the demand for environmental services is located around the localization of provision, the demand in more distant areas also affects the effectiveness of public intervention.

The first part of the PhD aimed to provide evidence that farmers manage the provision of ecosystem services to meet their own demand for productive ecosystem services. In other words, the first part of the PhD contributes to the analysis of the specificities of the supply of the environmental services provided by the farmers. The second part of the PhD aimed to investigate the role of the geographical scale of the demand for environmental services with regard to the optimal design of (agro)environmental instruments. In other words, the second part analysed the specificities of the demand for the environmental services provided by the farmers.

I bridged several literatures to investigate these questions. In the first part, I combine the literature on the productivity of productive ecosystem services with the literature on farmers' microeconomic behaviour to prove that farmers do manage the provision of productive ecosystem services. The literature on farmers' microeconomic behaviour uses the observed farmers' choices, notably variable input application and acreage choices, to determine the responses of farmers regarding economic incentives. Building on the advantages of the first literature, which approximates biodiversity with indicators based on land-use, Chapter 2 proposes a unified theoretical model that specifies farmers' behaviour with regard to productive ecosystem services. Chapters 3 to 5 are empirical works where I estimate different versions of this theoretical model. Using a dynamic framework, Chapter 4 provides evidence that farmers manage biodiversity and related productive ecosystem services to benefit from their productive effects. Based on farmers' observed behaviour, Chapters 3 and 4 also provide new insights into the productivity of different types of biodiversity components for a series of disaggregated outputs, including detailed interactions with chemical inputs. These results suggest that the productive ecosystem services supported by on-farm crop biodiversity and permanent grasslands (i) benefit differently to the different outputs, (ii) are substitute pesticides and fertilizers and (iii) have dynamic productive effects on future periods. Finally, in Chapter 5, I investigated the impacts of coordinated management of productive ecosystem services, which has been suggested in the literature to be a promising strategy. Based on a realistic landscape model with heterogeneous farmers, the results indicate that if coordinated management does lead to collective and individual gains on average, these gains are relatively limited and unequally distributed over the coordinated farmers, hampering the emergence of coordinated management strategies in real landscapes. I hope that the decomposition and the formulation of the proposed models will trigger further research in this area, notably about risk production management, an issue that I did not take into account despite the numerous discussions on the insurance value of biodiversity and related productive ecosystem services.

In the second part of the PhD, I introduce the principle that farmers managing agroecosystems jointly produce local and global public goods in three commonly used environmental economics literatures: the literature on the link between trade and environmental quality, the literature on hedonic pricing valuation and the literature on environmental federalism. In Chapter 7, I introduce this property of joint production in a standard general computable equilibrium model of international trade to investigate the induced impacts of the regulation of a local public good on global public good provision. Applied to the case of the reduction of pesticide applications, which generate numerous types of local pollution (affecting not only health but also environmental public goods), I highlight that if a French pesticide taxation scheme of 50% could reduce national pesticide applications by 40%, the market effects lead to the emission of 9 million tons CO2 equivalent in other localities of the world. These emissions, mostly due to land use changes in other countries and particularly due to deforestation in some Latin American countries, are equal to 10% of the actual carbon emissions from the French agricultural sector. In Chapter 8, I introduce the principle that farmers managing agroecosystems jointly produce local and global public goods in usual hedonic pricing models. Using the insights from the distance-decay of willingness-to-pay, I explain that farmers managing agroecosystems generate complex externalities over space. Using spatial econometric methods, the empirical results highlight that even if most of the value of the externalities are captured inside the municipality where production occurs, the distance-decay effects attached to the jointly provided local public goods also affect the welfare of neighbouring municipalities. For example, if swine activities present negative effects at all scales, dairy cattle activities, including grassland management, present negative effects in the municipality where production occurs but positive effects in the neighbouring municipalities. Finally, Chapter 9 is inspired by the literature on environmental federalism. I introduce the principle that farmers managing agroecosystems jointly produce local and global public goods into a theoretical model of a federal economy to study the effectiveness of the decentralization of the agroenvironmental policy design. In particular, I consider that hierarchical governments present different levels of information on the demand for the jointly provided global and local public goods, which affects the optimal degree of decentralization. These three chapters highlight that even if most of the value of an environmental service is captured locally, the demand for environmental services from larger and more distant areas influence social welfare and thus the design of the optimal policy.

The issue of the optimal policy has been an underlying but important question of this PhD thesis. I have explained in Chapter 1 that the usual Bowen Lindahl Samuelson conditions are subject to many uncertainties in the case of the environmental services provided by the farmers, preventing easy implementation in practice. This difficulty is illustrated by the existing French and European agroenvironmental policies. As highlighted by the European Court of Auditors (2011), most of the existing instruments are indeed focused on the supply side, with limited integration of the demand for environmental services. This situation is particularly clear in the development of "agroecology" in France. Recent French governments have promulgated a series of plans to support this "new" form of agriculture, ranging from a reorientation of agricultural research (notably from INRA, the French research institute for agriculture where I have conducted the presented research) to the introduction of specific subsidies. In the first part of the PhD, I have stressed that in the specific case where productive ecosystem services are private inputs, farmers already have incentives to manage agroecosystems depending on the prices and the properties of the productive ecosystem services. Even if different types of policy instruments could foster this transition, there is no need to encourage the farmers to mobilize productive ecosystem services in this case.

There are, however, needs for public intervention in the case of jointly produced public goods, which benefit agents that are not suppliers of environmental services. The first-best policy would consist of the implementation of the Tinbergen principle, with as many instruments as jointly provided public goods. For this reason, I argue that support for a specific form of agriculture is not optimal. For example, the simulated public support for the agroecological transitions in Chapter 7 highlights that the social benefits derived from the reduction of pesticide applications are reduced by the induced global emissions. This feature is common with organic farming: it improves the provision of some public goods but reduces the provision of others. Debates on which type of agriculture should be supported are nonsense to economists, who prefer debates on which type of public goods society wants. For example, instead of directly subsidizing agro-ecology with the aim of reducing pesticide applications, the regulator should encourage farmers to reduce pesticide applications by targeting pesticide applications directly (e.g., thanks to a pesticide taxation scheme): as highlighted by Chapters 2 to 5, a profit-maximizer farmer will shift on his own towards more agro-ecological practices. Subsidizing agronomical research on agroecology is useful because it provides technical solutions to farmers.

Chapters 7 to 9 illustrate, however, how the question of which public goods society wants is already a complex one. Indeed, each hierarchical government has different information on the demand for the public goods jointly provided by the environmental service. Local governments have better information on the demand for local public goods, which encourage decentralization but also generate externalities to other jurisdictions, either from trade (Chapter 7) or from joint production of local and global public goods (Chapters 7 to 9), encouraging centralization. In such a case where the first-best policy is unlikely to arise, the role of economists is also to investigate promising second-best situations. This was the aim of the second part of the PhD. For example, the results of Chapter 8 suggest that agroecological governance should not be settled at the scale of municipal government but rather at a larger scale. Similarly, I have concluded in Chapter 9 that even if it does not lead to a Pareto-optimal situation, national governments are the most suitable governments to design agro-environmental policies. Chapter 7 illustrates, however, that if trade and joint production are not taken into account, national intervention could lead to unexpected effects in the locality where the initial instrument is initially implemented. I hope that these works, together with the increasing literature on the role of distance on PG valuation, will encourage researchers to integrate the geographical scale of the demand when analysing the multiple dimensions of the effectiveness of agro-environmental instruments.

The recent promulgation of economic incentives to support more environmentally friendly agricultural practices has led to an increasing number of discussions within society. This is the case in France, where the development of agroecology and the provided incentives have led to numerous debates among diverse stakeholders: farmers and industrial lobbies, environmentalist lobbies, and policymakers, just to name few. From my perspective as a PhD student in economics, I wanted to contribute to these debates by investigating the concepts of ecosystem services and environmental services from the economic perspective, with a detailed focus on the supply of and the demand for the public goods influenced by the flows of ecosystem services. By deepening the Samuelson conditions in this specific case, I hope that these works will contribute to improving the effectiveness of agroenvironmental policies. I am already pleased that some insights from this PhD have been adopted in the policy brief of the PROVIDE H2020 project addressed to the European Commission. I am even more pleased that my works have contributed to the creation of a payment for environmental services scheme in Brittany to support the maintenance of traditional agricultural landscape features such as agricultural wetlands and hedgerows. These two examples illustrate the practical reasons for an economist to perform participatory research. If I were able to formulate policy recommendations, the complexity of the agroecosystem's functioning would require additional research on the numerous underlying specificities of the supply and the demand for environmental services before I could ultimately achieve the corresponding Samuelson condition.

Titre : Gestion agricole des services écosystémiques : éclairages à partir de l'économie de la production et de l'économie de l'environnement

Mots clés : analyse de l'offre ; analyse de la demande ; services écosystémiques ; services environnementaux ; biens publics ; politique agroenvironnementale

Résumé :

La thèse étudie théoriquement et empiriquement la gestion des services écosystémiques par les agriculteurs sous le prisme économique. La thèse se divise en deux parties. Dans la première partie, je m'intéresse à l'offre et à la demande de service écosystémique productifs en analysant le comportement des agriculteurs. J'introduis des indicateurs de biodiversité dépendants des assolements dans des modèles existants issus de l'économie de la production. Ma principale contribution à la littérature est de prouver, à partir de l'analyse des comportements observés des agriculteurs, que les agriculteurs gèrent consciemment les services écosystémiques productifs. J'apporte d'autres éléments à la littérature, comme e.g. des nouveaux éléments sur la technologie agricole ou en montrant que la gestion collective des services écosystémiques productifs ne peut que rarement émergée spontanément dans des paysages réels.

Dans la deuxième partie, j'étudie la demande de services écosystémiques non-productifs fournis par les agriculteurs. J'applique plusieurs cadres d'analyse développés en économie de l'environnement aux spécificités de l'agriculture, i.e. le service environnemental influe le plus souvent sur la fourniture de multiple biens publics, biens publics présentant des distributions spatiales de la demande différentes. Je contribue à la littérature en montrant que, bien que la plupart de la demande pour les services environnementaux fournis par les agriculteurs soit capturée localement (à l'échelle de la municipalité), une partie de la demande s'exprime à des échelles plus importantes. Cela a des implications pour les politiques agroenvironnementales, que j'explore à travers deux exemples : la réduction de l'application des pesticides et le maintien des zones humides agricoles.

Title : Agricultural management of ecosystem services : insights from production and environmental economics

Keywords : supply analysis; demand analysis; ecosystem services; environmental services; public goods; agro-environmental policy

Abstract :

The thesis studies both theoretically and empirically the management of ecosystem services by farmers in two parts. In the first part, I study the supply and demand for productive ecosystem services by analyzing farmers' behavior. I introduce biodiversity indicators that depend on acreage into existing models from production economics. My main contribution to the literature is to prove, from the analysis of farmers' observed behavior, that farmers consciously manage productive ecosystem services. I bring other elements to the literature, such as new elements on the agricultural technology or showing that the collective management of ecosystem services rarely arises spontaneously in real landscapes.

In the second part, I study the demand for non-productive ecosystem services. I apply several analytical frameworks developed in environmental economics to the specificities of agriculture, i.e. the environmental service influences the supply of multiple public goods with different spatial distribution of the demand. I contribute to the literature by showing that while most of the demand for environmental services provided by farmers is captured locally (at the municipal level), a part of the demand is expressed at larger scales. This has implications for agri-environmental policies, which I explore through two examples: the pesticide savings and the maintenance of agricultural wetlands.