Integrated modelization of land use in France: from the crop choice to the choice of economic sector

Anna Lungarska

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Anna LUNGARSKA

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Modélisation intégrée de l'allocation des terres en France : du choix cultural au choix sectoriel

Directeur de thèse : **Pierre-Alain JAYET**

Jury

Mme Catherine BAUMONT, Professeur, Laboratoire d'Économie et de Gestion, Université de Bourgogne	Rapporteur
M. Pierre DUPRAZ, Directeur de Recherche, UMR 1302 SMART, INRA/AgroCampus Ouest	Rapporteur
M. Lionel RAGOT, Professeur, UFR SEGMI, Université Paris Ouest Nanterre la Défense	Examinateur
M. Valentin BELLASSEN, Chargé de Recherche, UMR 1041 CESAER, INRA/AgroSup	Examinateur
M. Pierre-Alain JAYET, Directeur de Recherche, UMR Économie Publique, INRA/AgroParisTech	Dir. de thèse

INRA/AgroParisTech
UMR 210 Économie Publique
Avenue Lucien Brétignières, 78850 Thiverval-Grignon

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На семейството
и учителите ми

À ma famille
et à mes enseignants

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Résumé

Cette thèse de doctorat met au centre de la recherche le problème de l'allocation des terres selon deux perspectives : i) au sein de l'usage agricole, entre différentes activités (cultures, prairies), et ii) entre secteurs économiques (agriculture, forêt, urbain, etc.). Deux méthodes sont mobilisées : des modèles de programmation mathématique sectoriels pour la forêt et l'agriculture et des méthodes économétriques. Le modèle d'offre agricole européenne, AROPAj, permet d'étudier d'une manière très fine les décisions des agriculteurs en matière de choix des cultures et d'intrants (intrants azotés principalement). Puisque les agents économiques pris en compte dans le modèle cherchent à maximiser leur profit, les résultats obtenus sont directement utilisables dans des modèles économétriques d'allocation des terres intégrant la rentabilité des autres secteurs demandeurs en terres. Trois cas d'étude sont proposés. Dans le premier cas, on démontre l'importance de la prise en compte du choix des cultures lorsqu'une taxe sur l'apport azoté est introduite. Le deuxième cas d'étude est centré sur les prix des terres agricoles résultant de la concurrence entre les différents usages, à savoir les grandes cultures et les prairies, la viticulture, l'urbain et le tourisme. Comme les prix de la terre et les revenus agricoles sont souvent utilisés pour approximer la rente agricole dans les modèles économétriques d'allocation des terres, le troisième cas d'étude porte sur la comparaison des modèles économétriques dans lesquels ces données et les sorties du modèle économique servent de variables explicatives. L'emploi combiné de ces deux méthodes peut être utilisé pour l'étude des effets du changement climatique sur l'allocation des terres. Par ailleurs, le modèle économique permet aussi de tester des différents scénarios de politiques publiques.

Abstract

The research work presented in this doctoral thesis is devoted to the study of land use. The question is examined from two angles: i) land use within the farm (the choice of crops, pastures), and ii) land use between economic sectors (forests, urban, agriculture, etc.). Two methods were employed: mathematical programming models for the agriculture and forestry sectors and econometric methods. The supply-side agricultural model, AROPAj, allows us to model farmers' decision in terms of crops and nitrogen input quantities. Since its economic agents are profit maximizers, the results from the model are directly forwarded to an econometric land use model integrating the returns for the other land demanding sectors. Three case studies are proposed. In the first one, we prove that the choice of crops should be taken into account when evaluating the economic and environmental impacts of an input tax on fertilizers. The second case study is focused on agricultural land prices as the result from the competition among the different land uses, namely field farming and pastures, viticulture, urban and tourism. As land prices and agricultural revenues are often used as proxies for the agricultural rent in the econometric land use models, in the third case study we compare the results of econometric models when these values and the estimates from the agricultural model are employed as explanatory variables. The combined use of these two modeling methods can be valuable for the study of the climate induced land use change. Furthermore, the agricultural model allows us to simulate multiple public policy scenarios.

Table des matières

Contents

Table des figures

Liste des tableaux

Liste des acronymes

N Nitrogen

NO_3 Nitrate

AG Adour-Garonne River Basin District

AP Artois-Picardie River Basin District

AROPAj Modèle de l'offre agricole européenne (Jayet et al., 2015).

BRGM Bureau de recherches géologiques et minières

CAP Common Agricultural Policy of the European Union (ref. PAC).

CLC CORINE Land Cover

FADN Farm Accountancy Data Network

FFSM++ French Forest Sector Model, (Lobianco et al., 2015b).

FOC First Order Condition in mathematical optimization problems

GAM Generalized Additive Model

GHG Greenhouse gas

GI Geographical indicators of agri-food products

GIS Geographical Information Systems

GT Groupe type défini dans le modèle AROPAj

IDPR Network Persistence and Development Index

LB Loire-Bretagne River Basin District

LUC Land Use Change

LUCAS Land Use/Cover Area frame Survey

NPK Nitrogen (N), Phosphorus (P) and Potassium (K)

NPV Net Present Value

NRMSE Normalized root-mean-square error

NVZ Nitrate Vulnerable Zone

OLS Ordinary Least Squares technique

ORACLE Projet "Opportunités et Risques pour les Agro-ecosystèmes et les forêts en réponse aux changements CLimatiquE, socio-économiques et politiques en France (et en Europe)" financé par l'Agence Nationale pour la Recherche.

PAC Politique Agricole Commune de l'Union Européenne

RBD River Basin Districts

RICA Réseau d'Information Comptable Agricole

RM Rhin-Meuse River Basin District

RMC Rhône-Méditerranée et Corse River Basin District

SAFER Société d'aménagement foncier et d'établissement rural

SAR Small agricultural region (étude sur les prix de la terre)

SAR Spatial autoregressive model (étude comparative des proxies de la rente agricole)

SEM Spatial error model

SN Seine-Normandie River Basin District

SPS Single Payment Scheme of the Common Agricultural Policy

STICS Modèle agronomique de cultures (Brisson et al., 1998).

UAA Utilized agricultural area

UE Union européenne

WFD Water Framework Directive

Première partie

Introduction générale

L'activité humaine a toujours eu un impact sur l'espace environnant. La croissance économique et démographique font que cet impact est de plus en plus important et en nette accélération depuis la révolution industrielle. Cela a mené certains scientifiques à désigner l'époque actuelle comme l'*Anthropocène*. Le bouleversement du système climatique terrestre est indéniablement un indicateur majeur de l'ampleur de l'influence que l'humanité exerce sur la planète. Un des vecteurs principaux de la pression anthropique est le changement d'allocation des terres. Selon Pongratz et al. (2008), à l'échelle mondiale, 5 millions km^2 de végétation naturelle ont été convertis en terres arables et pâturages entre l'an 800 et 1700 ap. J.-C. Cependant, juste au cours de la deuxième moitié du XXème siècle, la surface agricole a augmenté par le même chiffre. Durant les 150 dernières années, 35% des émissions de CO_2 liées à l'activité humaine ont eu pour origine le changement d'usage des sols (Houghton, 2003). Par ailleurs, le système climatique n'est pas le seul à en être affecté. La biosphère et l'hydrosphère, et par conséquent l'humanité elle-même, en subissent également les effets. La modification de l'assolement n'est pas seulement une cause de bouleversement des processus naturels, elle est influencée à son tour par l'évolution de ces processus. Cela rend l'étude de ces phénomènes difficile et d'autant plus intéressante.

Au vu des interdépendances qui régissent nos sociétés et notre environnement, la recherche sur ces questions est de plus en plus poussée vers des approches interdisciplinaires. Les sciences modernes, extrêmement spécialisées, sont forcées à trouver un langage commun afin de faire avancer la connaissance sur ces sujets. Les outils employés dans le cadre des travaux présentés ici ont été largement développés dans cet esprit. Le prisme adopté est celui de la science économique puisque cette dernière est fondamentalement intéressée par la production des biens et services et leur distribution au sein des sociétés humaines. Ce sont justement ces processus qui motivent le changement d'usage des sols. Néanmoins, il est essentiel de prendre en compte les conditions physiques et biologiques ainsi que le cadre institutionnel qui déterminent l'activité économique. L'accent est mis sur l'agriculture en tant qu'activité la plus demandeuse en terre.

L'étude du secteur agricole est pertinente eu égard à la diversité des cultures, des pratiques et des impacts environnementaux qui en découlent. L'hétérogénéité du secteur est également source de nombreuses opportunités pour son adaptation aux changements globaux.

Pour intégrer les multiples dimensions de l'analyse de l'allocation des sols, il est nécessaire de recourir à la modélisation. Dans les présents travaux, plusieurs modèles sont mobilisés et mis en lien afin de prendre en compte les facteurs pertinents. Les secteurs forestier et agricole sont étudiés du point de vue biophysique et économique. Ainsi, un modèle générique de cultures est couplé au modèle d'offre agricole, alors que les estimations d'un modèle statistique biophysique de la forêt sont intégrés au modèle de secteur forestier. De cette manière, les modèles économiques prennent en compte les effets du changement climatique. Grâce à un modèle économétrique qui combine les résultats des deux modèles de secteur, nous pouvons faire des prédictions sur l'assolement dans le contexte du changement climatique et des politiques publiques.

Le but de la présente thèse de doctorat a d'abord été d'identifier, de comprendre et d'expliquer l'intérêt et les implications de l'utilisation d'une telle approche de modélisation en cascade. Dans un second temps, la validation de l'approche a été recherchée. L'objectif final a été l'application de cette approche dans l'étude des effets du changement climatique et des politiques publiques sur l'assolement. Différentes questions se sont imposées au cours des travaux de recherche :

— Quelles sont les caractéristiques des modèles proposés ?

— Pour chaque modèle, quels résultats intégrer dans les autres modèles ?

— En fonction des enjeux environnementaux étudiés, quelle est l'échelle géographique appropriée pour l'analyse ?

— Par rapport au changement climatique : quelles options d'adaptations sont prises en compte dans l'approche proposée et dans les études déjà menées ?

— Comment les variables centrales de notre analyse se rapportent-elles aux variables employées dans la littérature ?

— Quelles perspectives ouvre l'approche proposée et quels scénarios de politiques publiques tester ?

Certaines de ces questions peuvent paraître triviales. Cependant, dans le cadre d'une approche pluridisciplinaire comme la nôtre, elles ne sont pas sans intérêt.

Pour mener à bien les objectifs de la thèse, l'allocation des sols a été étudiée sous deux angles : i) au sein de l'usage agricole entre différentes activités (cultures, prairies) ; et ii) entre secteurs économiques (agriculture, forêt, urbain, etc.). Ainsi dans le Chapitre 3, la sensibilité de l'agriculture en terme d'allocation des sols est démontrée lorsque des politiques publiques sont simulées. Le Chapitre 4 établit un lien entre les prix observés des terres agricoles et la productivité marginale des terres, estimés par le modèle d'offre agricole, en tenant compte de l'influence que les autres usages peuvent avoir sur les prix observés. Dans le Chapitre 5, ces deux types de variables sont utilisés dans des modèles économétriques d'allocation des terres afin de comparer les résultats obtenus et de simuler différents scénarios de politiques publiques et de changement climatique.

Chapitre 1

Revue de littérature

L'usage des sols est un des sujets inhérents à l'analyse économique. La quête d'une bonne gestion des domaines agricoles donne naissance au terme Οικονομικὸζ ou *économie* employé par les philosophes grecs Xénophon et Aristote au IVème siècle av.J.-C. L'importance primordiale de la terre a été aussi reconnue en 1776 par Adam Smith dans son traité sur la richesse des nations (Smith, 1776) où elle est considérée comme l'un des facteurs fondamentaux de production, à côté du travail et du capital. Dans la société agraire de Smith, où la terre était rarement la propriété de ses exploitants, la question centrale était celle de la rente foncière. Plusieurs siècles plus tard, les débats sur le sujet ne sont toujours pas clos alors que les problématiques associées à la terre ne font que s'élargir.

Dans les dernières décennies, la branche de l'économie dédiée à la terre s'est beaucoup concentrée sur le changement d'allocation des sols (*land use change* ou LUC). Celui-ci peut être dû à des différents facteurs dont la démographie, le progrès technique, la croissance et le développement économique, la régulation publique, l'évolution des prix des biens et services ou encore le changement climatique. Cependant, les relations causales sont difficiles à déterminer (Duke and Wu, 2014). Les décisions sur l'allocation des terres ont un grand impact sur nos sociétés et notre environnement et les effets de ces décisions se font ressentir à toutes les échelles spatiales, du champ jusqu'au niveau global.

Les travaux présentés ici portent sur l'allocation des terres vue sous deux angles. Le premier relève de l'arbitrage entre différentes cultures fait par les agriculteurs en tant qu'agents économiques rationnels maximisant leur profit. Le deuxième angle couvre la conversion des terres d'un usage à l'autre (urbanisation, dé- ou afforestation, etc.). Dans les deux cas, le choix de l'allocation des terres est justifié par la comparaison des revenus relatifs à chaque option possible. Pour aborder ces deux problématiques, deux méthodes de modélisation ont été employées : la modélisation mathématique et l'économétrie. Les modèles de programmation mathématique, AROPAj pour l'agriculture et FFSM++ pour la forêt, ont comme fonction objectif la maximisation du profit des agents économiques. Ce même objectif est sous-jacent pour le modèle économétrique employé dans le Chapitre 5 où l'allocation des terres au sens large est étudiée.

1.1 L'allocation des terres au sein du secteur agricole

L'usage agricole est dominant à l'échelle mondiale avec 38% de la surface terrestre, la forêt en couvrant 31% (FAO, 2015). Depuis les années soixante, la surface agricole a augmenté de 10% et s'est stabilisée autour de 50 millions km^2 depuis 1995. L'agriculture se distingue des autres usages des sols par le caractère annuel du choix de culture [1]. Les différentes plantes sont liées à des pratiques et à des niveaux d'intrants spécifiques. Ainsi, l'impact environnemental de chaque culture est différent. La prise en compte de cette dimension lors de l'évaluation des politiques publiques s'avère très importante.

Le cas d'étude proposé dans le Chapitre 3 porte sur la pollution des eaux par les nitrates d'origine agricole due à l'épandage intensif d'engrais minéraux et/ou d'effluents d'élevage. Il s'agit d'un sujet largement traité en économie, de manière théorique et empirique. Les nombreuses difficultés dans la conception des politiques publiques justifient le grand intérêt que les économistes y portent. En premier lieu, les émissions de nitrates ne sont directement observables qu'à un

1. Ceci est surtout valable pour les cultures de champs. L'arboriculture et la viticulture, par exemple, sont liées à des choix pluriannuels.

coût de contrôle élevé, voire prohibitif (Shortle and Horan, 2002). Elles sont stochastiques à cause des aléas climatiques et impliquent un grand nombre de pollueurs, hétérogènes dans leurs activités et dans leur impact sur le système hydrique. Ainsi, les politiques dites de premier rang visant directement les émissions s'avèrent peu réalistes. Après Griffin and Bromley (1982) qui proposent une nouvelle approche prenant en compte le caractère diffus de la pollution (par opposition à la pollution ponctuelle), Segerson (1988) développe un système de récompenses-pénalités collectives par rapport à une norme de pollution ambiante. Shortle and Horan (2002) donnent une vue d'ensemble sur les instruments de la politique publique qui ciblent ce type de pollution. Afin de mieux cerner la problématique étudiée dans les travaux ici présentés, la revue de littérature se concentre sur la "taxe".

La taxe en tant qu'instrument économique de la politique environnementale a été proposée par Arthur Cecil Pigou en 1920. Ainsi, aujourd'hui, une taxe dont l'objectif est de corriger une externalité négative (p.ex. la pollution) en internalisant les coûts qui y sont associés est appelée taxe pigouvienne. Cependant, même Pigou reconnaît que la bonne définition de ce type de taxes exige que le régulateur soit parfaitement informé, ce qui est rarement vrai (Pigou, 1937). En cas de pollution diffuse, l'assiette de la taxe peut être choisie parmi différentes options : i) taxe sur les intrants ; ii) taxe sur l'estimation des émissions ; et iii) taxe sur la qualité de l'eau. Ces instruments sont souvent conçus dans l'objectif de minimiser le coût social tout en respectant une norme environnementale donnée (dans la continuité des travaux de Baumol and Oates, 1971), car les dommages liés à la pollution sont difficiles à estimer.

Tietenberg (1974) and Xepapadeas (1992) ont démontré de manière théorique que les taxes prenant en compte l'hétérogénéité spatiale sont plus coût-efficaces que les taxes uniformes. Par extension, en supposant connue l'hétérogénéité des pollueurs [2], les taxes optimales devraient être définies pour

2. Plusieurs facteurs influencent les fonctions de transformation de l'intrant (azoté) en émissions nocives (nitrates) et, par la suite, l'accumulation de ces émissions dans les eaux. Par exemple : la localisation géographique des champs avec ces conditions pédoclimatiques ; les pratiques em-

chaque agent et chaque intrant associés à la pollution. Les études empiriques sur l'efficacité-coût des politiques différenciées concernent souvent une seule culture (Helfand and House, 1995; Xabadia et al., 2008; Claassen and Horan, 2001; Martínez and Albiac, 2006; Lacroix et al., 2010) et/ou une seule masse d'eau (Xabadia et al., 2008; Fleming and Adams, 1997; Helfand and House, 1995; Claassen and Horan, 2001; Westra and Olson, 2001; Martínez and Albiac, 2006). Fleming and Adams (1997) étudient cinq cultures alors que Westra and Olson (2001) concentrent leur recherche sur deux. Les animaux en tant que source d'azote organique produit au sein de l'exploitation agricole sont, à ma connaissance, absents des études empiriques.

Le modèle d'offre agricole employé dans cette étude, AROPAj, couvre à la fois des activités animales et végétales dont huit cultures pour lesquelles les rendements sont représentés par des fonctions de dose-réponse par rapport à l'azote (voir Section 2.1), à l'échelle de l'Union Européenne. Le modèle a été employé dans plusieurs études sur la pollution des eaux par les nitrates (Bourgeois et al., 2014; Bourgeois, 2012; Jayet and Petsakos, 2013). Bourgeois et al. (2014) utilisent AROPAj pour comparer différentes politiques : taxe sur les pertes racinaires de nitrates avec et sans subvention sur une culture demandant moins d'azote (miscanthus géant). Jayet and Petsakos (2013) regardent les effets de la Politique Agricole Commune (PAC) sur l'efficacité d'une taxe sur les engrais azotés. Lors de cette dernière étude, les auteurs ont identifié un effet paradoxal de la taxe. Celui-ci est dû au changement d'usage des sols décidé par les agriculteurs dans le contexte du nouveau prix de l'intrant. Ainsi, les cultures choisies en réponse à la taxe peuvent s'avérer plus polluantes même si elles sont associées à des quantités inférieures d'azote appliquées aux champs. Cet effet paradoxal peut être évité si les taux de la taxe sont différenciés par rapport aux cultures (Goetz et al., 2005). Cependant, l'introduction d'un tel schéma de taxation est très peu réaliste [3] (Bourgeois, 2012).

ployées par l'agriculteur ; les cultures choisies.

3. Les agriculteurs sont incités à acheter de l'engrais à prix faible en déclarant qu'ils l'apporteraient à la culture qui pollue le moins et, par la suite, l'utilisant sur les cultures où le profit marginal

Le modèle économétrique structuré développé par Fezzi and Bateman (2011) permet de tenir compte des effets du LUC pour l'évaluation des politiques publiques ciblant la pollution diffuse d'origine agricole. Dans le Chapitre 5, un modèle économétrique d'allocation des terres est présenté. Il vise à estimer les effets du LUC d'une taxe sur les engrais à l'échelle de la France. Ainsi, les conséquences de la taxe sur la marge intensive (la quantité d'engrais apportée à l'hectare) et sur la marge extensive (la surface agricole utile en hectares) sont évaluées.

Extensions : choix de cultures et le changement climatique

Dans le cadre des analyses des effets du changement climatique sur l'agriculture, le choix de cultures est un des vecteurs d'adaptation. Les premières études sur le sujet utilisent des modèles agronomiques de croissance des plantes où les données climatiques sont modifiées selon les scenarios de changement climatique pour les États-Unis (D'Arge, 1975; Bach, 1979; Decker et al., 1986; Dudek, 1988; Adams, 1989; Adams et al., 1990; Rosenzweig and Parry, 1994). Mendelsohn et al. (1994) désignent ces premières évaluations comme l'approche par des fonctions de production et soulignent l'absence de possibilités d'adaptation pour les agriculteurs. Ils proposent une nouvelle technique basée sur la théorie ricardienne des prix de la terre (voir Section 1.2 du présent chapitre). La méthode est construite en deux étapes. Dans un premier temps, les prix de la terre agricole observés sont expliqués par des variables climatiques et édaphiques (températures, précipitations, qualité des sols). Ensuite, en faisant varier le climat, des prédictions sur les prix sont faites. Cette technique a la vertu d'être relativement plus facile à appliquer grâce à la disponibilité des données sur les marchés fonciers et sur le climat. Des nombreuses études ont ainsi été réalisées sur différents pays du monde : Seo et al. (2005); Seo and Mendelsohn (2008); Sanghi and Mendelsohn (2008).

Suite aux critiques faites par Mendelsohn et al. (1994) aux modèles basés sur

de l'intrant est le plus élevé.

des simulations agronomiques, des modules économiques ont été employés afin de prendre en compte la possibilité d'adopter des cultures plus rentables en climat futur (Easterling et al., 1993; Adams et al., 1995; Leclère et al., 2013). Leclère et al. (2013) testent, grâce aux modèles AROPAj et STICS, trois scenarios d'adaptation de l'agriculture de l'Union Européene à 15 : i) pas d'adaptation (surfaces par culture fixes) ; ii) adaptation juste par le changement de cultures ; et iii) changement dans les pratiques (dates de semis et de récolte, variétés). Leurs résultats montrent que sans aucune adaptation l'effet sur la marge brute agricole varie entre -11% et +5% pour le scénario B1 et entre -6% et +4% pour le scénario A2. Avec adaptation, les résultats sont entre +4% et +27% (B1) et entre +6% et +29% (A2). Les principales hypothèses des scénarios simulés sont données dans la Figure 1.1.

A1 – croissance économique rapide – croissance modérée de la population – progrès technique important – hausse des temp. de 1.4 à 6.4 °C	A2 – croissance économique modérée – croissance forte de la population consommation d'énergie élevée – hausse des temp. de 2.0 à 5.4 °C
B1 – croissance économique modérée – croissance faible de la population – durabilité environnementale – hausse des temp. de 1.1 à 2.9 °C	B2 – croissance économique faible – croissance moyenne de la population – durabilité environnementale – hausse des temp. de 1.4 à 3.8 °C

FIGURE 1.1 – Résumé des principaux scénarios SRES de IPCC (2000).

1.2 Rente et prix de la terre

Pour le propriétaire privé d'une parcelle de terre, le choix de l'usage qu'il en fera est résumé par le problème de la maximisation de son revenu (Bell et al., 2006). Le revenu du propriétaire foncier est communément appelé rente foncière. Historiquement, les deux noms qui se démarquent lorsqu'il est question de rente foncière sont ceux de Ricardo et de von Thünen (Ricardo, 1817; von Thünen, 1826). Cependant, d'autres auteurs comme Petty (1623 - 1687), Quesnay (1694 - 1774), Smith (1723 - 1790) et Malthus (1766 - 1836) se sont aussi intéressés au problème (Guigou, 1982a). Pour ces derniers, la rente a une origine naturelle. Tout en restant dans cette logique, Adam Smith se distingue et influence le raisonnement de Ricardo et Marx.

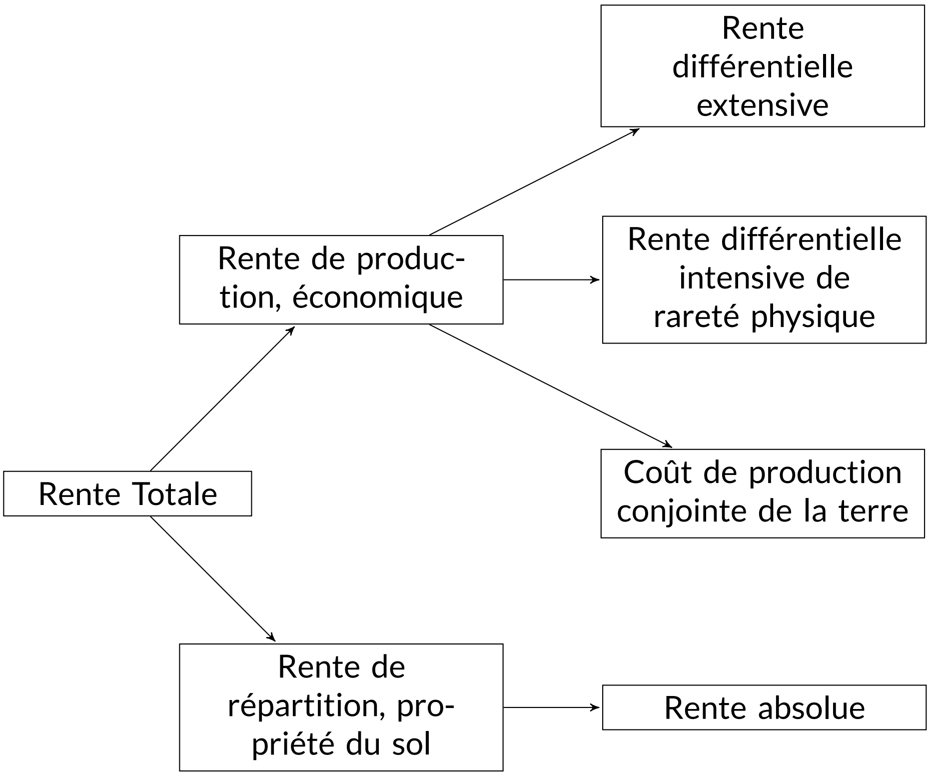


FIGURE 1.2 – La rente foncière d'après Guigou (1982b), définition inspirée par les travaux de David Ricardo, Karl Marx et Piero Sraffa.

La contribution de Ricardo est surtout associée à l'idée que la rente foncière est différenciée par rapport à la fertilité de la terre (Ricardo, 1817). Cette observation a été déjà faite par Smith (1776) mais Ricardo en fait une analyse plus approfondie. Ces deux auteurs ont aussi fait remarquer que cette différenciation dans les rentes peut être aussi due à la distance entre les champs et le marché (la ville) où la production est vendue. L'analyse de ce dernier sujet par von Thünen (1826) définit la méthode de l'économie spatiale. Smith (1776) regarde la rente comme un prix de monopole et ce concept est davantage développé par Marx (1894), dans ce que l'auteur appelle la rente absolue due à la propriété de la terre. Les auteurs cités ci-dessus partagent la même définition de la terre en tant que ressource naturelle rare. Sraffa (1926) propose une nouvelle vision où la différente productivité des terres est liée aux techniques de production qui y sont appliquées.

En se basant sur les travaux de Ricardo, de Marx et de Sraffa, Guigou (1982b) définie la rente comme un revenu bi-dimensionnel où une partie de la rente est due à des facteurs d'ordre économique (rentes différentielles et les coûts associés aux améliorations apportées à la terre [4]), alors que l'autre partie est liée à la distribution du surplus de l'économie (Figure 1.2). Ainsi, une augmentation de la demande de denrées alimentaires donne lieu à une hausse des rentes différentielles de deux manières. La première est liée à l'hétérogénéité où l'augmentation de la demande pousse à la mise en exploitation des terres moins fertiles ou plus éloignées (marge extensive). Ces terres marginales sont associées à des coûts de production supérieurs à ceux des terres de meilleure qualité, ce qui se traduit par des profits (rentes) plus élevés pour les terres initialement exploitées. Il s'agit ici de la rente différentielle extensive. La deuxième voie de formation de la rente différentielle est la marge intensive due à la rareté des terres. Pour expliquer ce processus, supposons que les terres sont rares et de qualité uniforme. Lors d'une hausse de la demande, les prix augmentent ce qui induit l'introduction de nouvelles techniques sur certaines parcelles. Ces techniques sont liées à des rendements par hectare plus élevés mais aussi à un coût supérieur. Ainsi, pendant que les deux techniques sont employées simultanément, une rente pour les terres exploitées avec la technique la moins chère se forme suite aux coûts inférieurs et aux prix plus élevés. À la différence de la rente différentielle, la rente absolue s'applique à toutes les terres, y compris les terres marginales. De ce point de vue, elle est liée à la répartition du surplus produit dans la société et distribuée selon le rapport des forces (Guigou, 1982b).

Sraffa (1926) est à l'origine du concept de production conjointe, où des mar-

4. Guigou (1982b) appelle cette rente "coût de production conjointe de la terre" puisqu'il considère que la terre peut être analysée en tant que produit et non pas comme ressource naturelle.

chandises et du capital fixe sont produits grâce à d'autres marchandises. Ainsi, les gains en productivité des terres liés aux investissements doivent être récompensés au même taux que tout autre placement du même risque. Cette vision va donner suite à une analyse qui ne considère plus la terre en tant que ressource naturelle mais simplement comme une forme du capital (Randall and Castle, 1985). L'analyse néo-classique de la rente foncière est basée sur l'idée que la terre est un bien comme les autres et que la rente n'est que le résultat de l'ajustement de l'offre et de la demande sur le marché (Guigou, 1982b). Tout surplus dû à l'augmentation des prix au-delà des coûts n'est que temporaire et le jeu de la concurrence l'élimine à terme. Pour Say (1803), la rémunération de la terre se partage en deux : i) rémunération du capital foncier (au niveau du taux d'intérêt), et ii) rémunération de la terre en tant que facteur de production (rente foncière). Ce dernier revenu est absent de l'analyse faite par Walras (1860) pour qui la terre est juste une forme de capital dont la rémunération est la rente foncière mais nullement un surplus. La différence des prix est due au fait que le bien est hétérogène. Allais (1943) parle de "la valeur de la production marginale de la terre" ou de "son produit marginal" mais ne trouve dans la théorie néo-classique ou ricardienne qu'une description de la rente et non pas une explication.

La question de la rente mène à celle de la localisation des activités, d'abord étudiée par von Thünen (1826) dans son analyse détaillée de la rente de localisation. Pour cet auteur, la distance est surtout associée au coût de transport. De ce fait, la rente qu'il étudie est expliquée par les économies de transport qui sont faites par les terrains proches de la ville et donc du marché (Guigou, 1982a). Cette analyse basée sur les distances est reprise par Alonso (1964) qui regarde la question de la localisation dans l'optique de la concurrence entre les différents usages, y compris l'agriculture. Ses travaux fondent la base de l'économie urbaine.

La prise en compte de la distance dans la modélisation des villes et régions péri-urbaines s'avère très pertinente pour expliquer les écarts entre le prix des terres agricoles et la somme actualisée des rentes. Généralement, depuis le temps des Classiques, le prix de la terre est regardé comme la somme actualisée des rentes futures [5]. Cependant, de nombreuses études ont démontré que la rente ne suffit pas pour expliquer les prix observés (Clark et al., 1993; Gutierrez et al., 2007; Dupraz and Temesgen, 2012; Karlsson and Nilsson, 2013). Une des raisons de cette dichotomie est le fait que les rentes futures peuvent provenir d'autres usages que l'agriculture. Guiling et al. (2009) développent la formule de la somme actualisée pour y englober la possibilité de changement d'usage.

Une telle transition a été déjà étudiée par Capozza and Helsley (1989), qui s'intéressent à la question de la date optimale de transformation des terres agricoles en terrains constructibles. Plantinga et al. (2002) donnent une mesure explicite de l'influence que la possibilité d'urbaniser a sur les prix des terres agricoles aux États-Unis. Les travaux de Cavailhès and Wavresky (2003) démontrent que les terres à proximité immédiate des villes ont des primes importantes dues au développement urbain mais ces primes baissent vite avec l'éloignement des villes. Dans une logique similaire, la proximité du littoral est aussi susceptible d'influencer les prix des terres agricoles (Dachary-Bernard et al., 2011). L'urbanisation est souvent régulée par des zonages et Geniaux et al. (2011) mettent en avant les effets de cette régulation sur les attentes des propriétaires fonciers et donc sur les prix. Ils étudient aussi l'influence de la présence des appellations d'origine des vins sur les prix de la terre ce qui s'avère positive et statistiquement significative.

Une autre voie d'exploration de la rente et du prix de la terre est l'étude de la capitalisation des aides publiques apportées aux agriculteurs. Floyd (1965) est le premier à mettre en lumière de manière théorique ce processus. Latruffe and Le Mouël (2009) présentent une revue de littérature approfondie et concluent que les prix des terres agricoles sont plus sensibles aux subventions publiques qu'aux profits du marché. En France, pour parer à l'appropriation de rentes liées aux politiques publiques par les propriétaires fonciers, une régulation des prix de location et de vente est introduite. Goodwin et al. (2003) utilisent des données à l'échelle

5. Le calcul usuel consistait à ajouter les rentes sur une période de 20-25 ans ce qui. La somme des rentes actualisées est proche à ce résultat, lorsque le taux d'actualisation est compris entre 4 et 5%. La valeur actuelle des rentes futures s'écrit : $L_t = \frac{R}{r}$ où L_t est le prix de la terre à l'instant t, R est la rente agricole supposée constante dans le temps et r est le taux d'intérêt pratiqué.

de la ferme pour démontrer que les modèles basés sur les revenus observés des agriculteurs peuvent être biaisés puisqu'il n'est pas rare que les revenus observés diffèrent des revenus espérés. Les auteurs soulignent aussi l'effet de la pression urbaine sur les prix des terres.

Dans la littérature économique, la notion de terre a été souvent utilisée pour désigner les ressources naturelles dont la disponibilité est limitée. Avec le progrès technique, les économistes (dont Solow, 1956) vont considérer que, grâce à la substituabilité entre les facteurs de production, la croissance économique n'est pas menacée à long terme par l'épuisement des ressources naturelles. Pourtant, c'est justement dans l'optique des ressources naturelles épuisables que la «terre» va retrouver de l'intérêt pour les économistes après le premier choc pétrolier et le Club de Rome. À ceci s'ajoutent les nouveaux défis liés à la gestion des sols dans le contexte de l'exploitation intensive qui en est faite depuis la Révolution verte dans l'agriculture.

1.3 Modèles d'allocation des terres, rente agricole et changement climatique

L'usage des sols fait l'objet d'un nombre important d'études qui se distinguent sur plusieurs points : i) la méthode employée (modèle économétrique ou modèle de programmation mathématique) ; ii) les différents usages pris en compte (cultures au sein de l'agriculture seule, agriculture et urbain ou agriculture et sylviculture, etc.) ; iii) les modèles sur des choix discrets ou continus (part de la surface) ; iv) le caractère statique ou dynamique des modèles ; v) la question qu'ils étudient (problématiques environnementales, de localisation optimale, de date de conversion optimale, d'urbanisme, etc.). Chakir (2013) présente une revue de littérature des modèles économétriques et développe la question de la prise en compte de la corrélation spatiale dans la modélisation. Quelques applications de modèles mathématiques d'allocation des terres entre différentes cultures ou usages sont données par Kaiser et al. (2011).

Dans les modèles de programmation mathématique, les revenus associés à chaque usage des sols sont souvent endogènes. Ainsi, les différentes rentes peuvent être d'abord évaluées et puis comparées afin de déterminer l'allocation optimale. Au contraire, les rentes dans les modèles économétriques sont généralement exogènes ou évaluées préalablement de manière explicite. Pour l'agriculture (mais aussi pour la forêt), les rentes ne sont pas toujours disponibles directement et les modélisateurs se voient obligés d'employer d'autres variables pour les approximer (des variables dites *proxy*). Ces variables sont souvent agrégées à des échelles administratives (Miller and Plantinga, 1999). Par exemple, Wu and Segerson (1995) utilisent les prix des intrants alors que Plantinga (1996) calcule le produit des prix fois les quantités. Dans leur étude sur l'influence de l'urbanisation sur les prix des terres agricoles, Plantinga et al. (2002) se basent sur le profit des fermes par hectare en y rajoutant les aides publiques perçues. Ahn et al. (2000), dans leur modèle de part des usages (*land shares model*), approximent les rentes agricoles par les profits des trois cultures majeures (soja, maïs et coton), pondérés par les surfaces dans les comtés d'Alabama. Dans Lubowski et al. (2006), la mesure de la rente agricole employée est plus élaborée : la distinction est faite entre les profits des cultures et ceux des pâturages et les aides publiques sont prises en compte. Pour calculer le profit des terres arables, les profits par hectare d'un nombre important de cultures (au moins 20) sont agrégés en pondérant par les surfaces de chacune. Des données de différentes sources sur les prix, les coûts et les rendements sont employées dans les calculs. L'indicateur pour les pâturages est basé sur leur rendement multiplié par le prix des fourrages.

La question du proxy employé pour la rente agricole se pose aussi dans le cadre des études sur les effets du changement climatique sur l'allocation des sols. En général, pour évaluer la profitabilité de l'agriculture en climat futur, les chercheurs ont le choix entre l'application de modèles agronomiques avec ou sans modules économiques ou la modélisation économétrique (discuté dans Section 1.1). En liant l'indicateur de la profitabilité de l'agriculture proposé par Lubowski et al. (2006) à la productivité primaire nette des écosystèmes, Haim et al. (2011) cherchent à faire évoluer l'usage des sols en climat futur. Deschênes and Greenstone (2007) se basent sur des données de panel sur le climat et les revenus des fermes pour évaluer les effets des déviations dans la météorologie à l'échelle des comtés américains. Avec cette méthode, les prix de la terre ne peuvent pas être employés comme indicateurs puisqu'ils reflètent les rentes à long terme en climat moyen et, en conséquence, ne captent pas les déviations annuelles du climat.

Ay et al. (2014a) s'inspirent de l'approche Ricardienne (Mendelsohn et al., 1994) et utilisent le prix de la terre comme indicateur de la rente agricole. Fezzi et al. (2015) se concentrent quant à eux sur la marge brute des agriculteurs. Dans ces études, la variation de la rentabilité de l'agriculture en climat futur est déduite grâce à des modèles économétriques qui prennent en compte la qualité des sols, le relief et le climat (températures et précipitations). Une des applications du modèle de LUC présenté ici (Chapitre 5) porte aussi sur les effets du changement climatique mais le proxy de la rente agricole est estimée par le modèle de programmation mathématique AROPAj. De la même manière, la rente forestière utilisée est issue du modèle d'équilibre partiel FFSM++. Cette méthode permet d'intégrer plusieurs options d'adaptation des agriculteurs et des agents forestiers. Les résultats obtenus montrent les tendances dans la répartition de l'espace suite à l'évolution du climat et peuvent servir d'appui à la décision publique.

1.4 Modélisation du secteur agricole

Les modèles dédiés au secteur agricole sont très divers du point de vue de leurs objectifs, hypothèses, méthodes, ainsi que des activités modélisées et de la couverture spatiale et temporelle. Selon Schoemaker (1982), les objectifs des modèles sont au nombre de quatre : i) description ; ii) prédiction ; iii) postdiction ; et iv) prescription. Souvent les modèles sont utilisés à plusieurs fins à la fois. Par exemple, les modèles descriptifs peuvent être employés pour prédire. Swinton and Black (2000) soulignent que le processus de modélisation est parfois l'occasion de révéler des lacunes dans la science ou dans les données disponibles. Ceci rend la modélisation intéressante non seulement au vu des résultats directement obtenus mais aussi de l'avancement de la connaissance en dehors de la modélisation en soi. La Table 1.1 rassemble les différents types de modèles selon plusieurs critères. Parmi les modèles agricoles, on distingue les modèles statiques et dynamiques. Souvent, les modélisateurs choisissent l'échelle géographique en fonction de la question à laquelle leur modèle répond. Dans le cas de pollution diffuse des masses d'eaux, par exemple, l'échelle appropriée est définie par la dynamique de la pollution (comme le bassin-versant, entre autres).

Critère	Type de modèle
Objectif	description
	prédiction
	postdiction
	prescription
Utilisateurs	chercheurs
	agents privés
	décideurs publics
Couverture temporelle	statique
	dynamique
Couverture spatiale	à l'échelle de la parcelle
	à une échelle biophysique
	à une échelle administrative
	à l'échelle de plusieurs pays
	à l'échelle mondiale
Techniques	simulation
	optimisation
	statistiques/économétrie
Prix exogènes	modèles d'offre
ou endogènes	équilibre partiel
	équilibre général

TABLE 1.1 – Classification des modèles agricoles inspirée (en partie) par Swinton and Black (2000).

D'une manière générale, l'économie agricole a largement contribué au développement des techniques d'optimisation et d'économétrie (Just, 2007). Le problème de l'allocation des terres entre cultures et élevage a été fondamental pour la discipline depuis le XIXème siècle. Grâce aux avancements dans le domaine informatique, l'analyse s'est élargie à d'autres activités et à d'autres contraintes. Selon Just (2007), ces études sont à la base de l'économie de la production. Les techniques de modélisation employées sont réparties en trois catégories : i) simulation ; ii) optimisation ; et iii) statistiques. Les modèles de simulation et d'optimisation sont basés sur des algorithmes mathématiques. Les modèles d'optimisation sont structurés par rapport à des fonctions objectif comme la maximisation du profit des agriculteurs. Les techniques statistiques sont souvent utilisées afin de tester des hypothèses ou d'estimer différents paramètres pour les modèles de simulation ou d'optimisation (Swinton and Black, 2000). Elles servent également à étudier le marché des denrées alimentaires. Différentes méthodes de programmation sont utilisées dans l'analyse économique du secteur agricole, notamment la programmation linéaire et non-linéaire, la programmation dynamique de contrôle optimal, la programmation stochastique, etc. Les modèles agricoles peuvent aussi être classés selon leurs hypothèses sur les prix des biens et services. Les prix sont exogènes pour les modèles d'offre et (partiellement) endogènes pour les modèles d'équilibre partiel et général.

Historiquement, dans les années 1950, la programmation linéaire a été dominante dans l'optimisation à l'échelle de la ferme. Dans les années 1970, l'application de ces techniques s'est élargie à la modélisation de fermes-types afin de simuler des politiques publiques. Les modèles de programmation linéaire ont généralement besoin de beaucoup de données statistiques. Ils peuvent donner des solutions très contrastées en réponse à des faibles variations des variables du modèle ("*jumpy behaviour*"). Cela est dû aux solutions en coin typiques de modèles linéaires qui peuvent aussi prédire une spécialisation excessive ("*over-specialization*") des systèmes agricoles. Les limitations de la programmation linéaire ont freiné son utilisation. Pendant les années 1980, des modèles avec prix endogènes sont apparus. Ceux-ci permettaient de simuler les équilibres de marché pour plusieurs produits agricoles et plusieurs pays ainsi que l'équilibre général sur l'ensemble des marchés (modèles d'équilibre général calculable). Ces modèles ont bénéficié du développement des modèles non-linéaires. Ces dernières années, la nécessité d'étudier les interactions entre l'agriculture et l'environnement a poussé les chercheurs vers des modèles bio-économiques. De nombreuses plate-formes informatiques ont été développées afin de permettre le couplage de modèles provenant de différentes branches de la science.

Quelques modèles agricoles sont présentés dans la Table 1.2. Chaque type de modèle a ses avantages et ses inconvénients. La bonne définition des questions de recherche est primordiale pour choisir le modèle approprié. Dans ces travaux de thèse, la question centrale est l'allocation des sols au sein de l'agriculture et entre secteurs économiques. Le modèle AROPAj a été utilisé pour capter les effets du changement climatiques et des politiques publiques sur le secteur agricole. La spécification du modèle est intéressante pour les études prospectives parce que le choix de cultures n'est pas contraint. De fait, ce qui est considéré comme un inconvénient majeur de ce type de modèles peut être particulièrement utile dans le cas des analyses à horizon lointain. Les modèles d'équilibre général calculable prennent en compte des interactions complexes et, pour cette raison, il est difficile de retracer l'enchaînement des processus sous-jacents ("boîte noire"). Leur intérêt principal est l'endogénéité des prix. Cependant, les estimations de ce type de modèle peuvent être employées dans des modèles aux prix exogènes. Dans le cas du modèle CAPRI par exemple, des modèles d'offre et d'équilibre général et partiel sont interconnectés. Aghajanzadeh-Darzi (2014) intégre les prix issus du modèle GLOBIOM dans le modèle AROPAj.

Modèle	Type	Portée
Global Trade Analysis Project (GTAP), Hertel (1997) `https://www.gtap.agecon.purdue.edu/`	équilibre général calculable	globale
ORANI-G `http://www.copsmodels.com/oranig.htm`	équilibre général calculable	nationale
IMPACT `http://www.ifpri.org/program/impact-model`	équilibre partiel	plusieurs régions et plusieurs marchés
Common Agricultural Policy Regionalised Impact Modelling System (CAPRI) `http://www.capri-model.org`	modules d'offre, du marché et d'équilibre général calculable	globale, Europe
GLOBIOM `http://www.globiom.org/`	équilibre partiel	globale
FARMIS `https://www.ti.bund.de/en/infrastructure/the-thuenen-modelling-network/models/farmis/`	modèle d'offre	certains pays européens
AROPAj, Jayet et al. (2015)	modèle d'offre	Union européenne

TABLE 1.2 – Présentation de quelques modèles agricoles.

1.5 Modélisation du secteur forestier

Les débuts de la modélisation de la forêt remontent aussi au XIXème siècle. Faustmann (1849) déduit une règle pour estimer la valeur des forêts et déterminer leur gestion optimale. Son approche est microéconomique et ne concerne que le choix individuel des forestiers. Les modèles agrégés portant sur la filière dans son ensemble n'apparaissent qu'un siècle plus tard. Leur objectif est, entre autres, d'étudier les conséquences des chocs pétroliers des années 1970.

Une revue détaillée des modèles est proposée par Caurla (2012) où ils sont classés selon : i) la dynamique temporelle ; ii) l'endogénéité des prix ; iii) les anticipations des agents ; et iv) la couverture spatiale. En ce qui concerne la dynamique temporelle, la gestion du temps est faite soit en dynamique récursive soit en optimisation intertemporelle (Latta et al., 2013). Comme dans le cas de la modélisation du secteur agricole, il existe des modèles d'offre et d'équilibre général et partiel. Les anticipations des agents sont soit myopes, soit adaptatives, soit parfaites.

Plusieurs des modèles présentés en Section 1.4 ont des modules dédiés à la forêt. Du fait de sa très large portée, le modèle GTAP n'étudie pas en profondeur le secteur forestier, mais son analyse est pertinente du point de vue macroéconomique. Le modèle GLOBIOM s'appuie sur le *Global Forest Model* (G4M) pour évaluer la profitabilité de l'usage forestier du sol. Les deux modèles principaux sont actuellement le *Global Forest Products Model* (GFPM, Buongiorno et al., 2003) et le *Global forest sector model* EFI-GTM (Kallio et al., 2004). Ces modèles ont une portée mondiale et permettent de captent les tendances dans le secteur forestier.

D'autres modèles ont été développés avec pour objectif l'étude approfondie des secteurs forestiers nationaux ou régionaux en fonction de leurs spécificités. C'est par exemple le cas des modèles *Norwegian Trade Model*, (NTM, Bolkesjø, 2004) et *U.S. Forest Products Module* (USFPM, Ince et al., 2011). Le modèle FFSM++ employé dans le Chapitre 5 s'inscrit dans cette logique de modélisation et représente uniquement le secteur forestier français. Il s'agit d'un modèle de dynamique récursive où les prix nationaux sont endogènes et les prix internationaux exogènes. Les agents sont considérés myopes par rapport à leurs anticipations des prix et le choix des essences. Les dernières modifications apportées au modèle (Lobianco et al., 2015a,b) ont permis de simuler les effets du changement climatique. Le revenu forestier issu de ce modèle n'est pas directement comparable au revenu agricole d'AROPAj. Cependant, grâce au modèle économétrique d'usages des sols, l'exploitation conjointe des résultats des deux modèles est possible.

Chapitre 2

Descriptif des modèles employés

Deux modèles mathématiques ont été mobilisés à différentes fins dans les travaux ici présentés : le modèle d'offre agricole AROPAj et le modèle d'équilibre partiel du secteur forestier FFSM++.

2.1 AROPAj

Le modèle AROPAj est un modèle mathématique basé sur la programmation linéaire qui représente l'offre de produits agricoles de l'Union européenne (UE). Son développement a commencé dans les années quatre-vingt-dix avec le début des réformes du régime de la Politique Agricole Commune (PAC) de l'UE. Au fil des années, le modèle a été élargi de plusieurs manières. Initialement développé pour la Grande-Bretagne et la France, il a été étendu à l'échelle des pays membres de l'UE à 15, à 24 et puis à 27 [1]. Cette évolution suit celle de l'Union elle-même, ainsi que l'évolution des données du RICA disponible (voir plus loin). Par ailleurs, différents modules complémentaires ont été ajoutés pour améliorer le lien entre le monde physique et l'activité des agents économiques du modèle. Une description détaillée du modèle est donnée dans Jayet et al. (2015).

1. Avec l'adhésion de la Croatie, l'UE compte, désormais, 28 États membres. La version d'AROPAj en développement, basée sur les données de 2009, porte sur 27 pays membres.

Données

Une liste non-exhaustive des données employées dans les différents modules du modèle est données en Table 2.1. La source d'information au cœur du modèle est la base de données du Réseau d'Information Comptable Agricole (RICA, FADN pour l'acronyme anglais). Les données sont annuelles et couvrent l'activité agricole au sein de l'UE. Les exploitations enquêtées sont représentatives à l'échelle des régions telle que définies par le RICA (proche au niveau NUTS2 [2] de l'UE). Pour la France, ces régions correspondent aux régions administratives établies dans les années 1950. La localisaiton exacte des fermes est inconnue et aucune information résultant d'une estimation à partir du FADN et mobilisant moins de 15 individus de l'échantillon ne peut pas être communiqué. Ainsi, pour modéliser le comportement des fermiers, un regroupement des individus s'impose.

Données	Module	Couverture	Échelle
RICA	AROPAj	UE	Région RICA
Sols, JRC European Soil Portal	Fonc. dose-rép.	UE	1 :1 000 000
	Spatialisation		
Climat, JRC MARS	Fonc. dose-rép.	UE	50 × 50 km grille
	Spatialisation		
Climat, ECHAM5 A2 et B1	Fonc. dose-rép.	UE	50 × 50 km grille
CORINE Land Cover	Spatialisation	UE	1 ha
LUCAS, Land use and land cover survey	Spatialisation	UE	2 x 2 km grille
Relief, GTOPO30 DEM	Spatialisation	Globe	~ 1 km

TABLE 2.1 – Données employées dans AROPAj.

Diverses sources de données sont mobilisées pour représenter le monde physique et économique qui contraint et caractérise l'activité agricole. Les informations nécessaires portent sur les conditions météorologiques, les caractéristiques des sols, les prix des facteurs de production, etc. Ces données sont utilisées en fonction des thèmes étudiés et des modules disponible selon les versions du modèle AROPAj. Ainsi, les données météorologiques servent à la fois pour définir

2. Pour plus de détails, voir `http://ec.europa.eu/eurostat/web/nuts/overview`.

les capacités productives des agents et pour la désagrégation (spatialisation) des sorties AROPAj à l'échelle fine. La diversité des sources d'entrées du modèle peut susciter des préoccupations en ce qui concerne la cohérence des données. Souvent les données ne sont pas toutes accessibles à la même échelle, ce qui oblige à l'introduction d'hypothèses supplémentaires et l'utilisation de valeurs moyennes ou agrégées. Les choix qui ont été faits visent à limiter les incohérences et à assurer la meilleure représentation possible. La dimension géographique qui est une des caractéristiques importantes du modèle implique le recours privilégié à des données accessibles à l'échelle européenne.

Agents économiques

Les agents économiques modélisés dans AROPAj représentent des regroupements d'au moins 15 exploitations observées. La représentativité est supposée acquise à l'échelle de la région RICA, un "poids" étant associé à chaque individu de l'échantillon. La pondération est fournie par le RICA. Le nombre d'agents économiques représentés dans le modèle dépend, en partie, du nombre d'individus présents dans l'échantillon. Le regroupement est fait par classification automatique à partir de l'orientation technico-économique (OTEX), l'altitude, la "dimension économique" et, pour la dernière version du modèle, aux surfaces irriguées. Par exemple, l'agriculture française est représentée par 157 groupes types (GT, version $V2$, RICA 2002) localisés dans les 22 régions FADN de la métropole du pays. La production des cultures majeures est modélisée par des fonctions dites dose-réponse où les effets de différents niveaux d'apport d'azote sont simulés avec le modèle de cultures générique STICS (Brisson et al., 1998). Chaque groupe type k maximise sa marge brute π_k (la différence entre les revenues et les coûts de production variables) en respectant l'ensemble des contraintes qui définissent l'ensemble possible de production. Le programme d'optimisation individuelle des agriculteurs est présenté de manière formelle dans Équation 2.1. Les différentes activités productives sont donnés par le vecteur x_k de dimension n, g_k étant leur contribution respective dans la marge brute. La matrice A_k, de dimensions $m \times n$, définit les coefficients d'entrées-sorties et z_k les m quantités contraignantes. Le vecteur des multiplicateurs de Lagrange associés aux m contraintes est donné par λ_k.

$$\begin{aligned} & \max_{x_k} \quad \pi_k(x_k) = \max \ g_k x_k & \\ \text{s.c.} \quad & A_k x_k \leq z_k & (\lambda_k) \\ & x_k \geq 0 & (\mu_k) \end{aligned} \tag{2.1}$$

Une des contraintes du modèle porte sur la surface agricole disponible. En maximisant leur profit, les agriculteurs décident des cultures et des surfaces qui leur sont dédiées ainsi que du niveau d'engrais azotés appliqué (grâce aux fonctions de dose-réponse). La contrainte est associée à une valeur duale de la terre qui, à l'optimum, traduit la valeur d'une unité supplémentaire de surface agricole. Selon le théorème de la dualité, la valeur duale de la terre est égale à sa productivité marginale. Celle-ci, à l'équilibre, correspond à la rente agricole dans l'approche néo-classique.

Fonctions dose-réponse

Les cultures représentées par des fonctions de réponse du rendement aux apports d'azote sont : le blé dur et le blé tendre, la betterave sucrière, le colza, le maïs, l'orge d'hiver, les pommes de terre et le tournesol. Pour définir une telle fonction, il faut d'abord déterminer un ensemble de simulations : pour chaque GT, plusieurs types de sols possibles (les cinq sols les plus répandus dans la région), deux cultures précédentes, trois dates de semis ou trois variétés culturales, en situation irriguée ou non irriguée (Leclère et al., 2013; Godard, 2005). Pour chaque combinaison de ces paramètres, plusieurs niveaux d'apport d'azote sont simulés avec STICS. Les résultats ainsi obtenus sont ensuite utilisés pour ajuster des courbes de type exponentiel dont la forme est donnée par l'Équation 2.2.

Ainsi, le rendement, y, dépend de la quantité d'azote apportée, N. Le rendement maximal, non-limité en azote, est donné par le paramètre B, alors que la quantité A est obtenue lorsque $N = 0$. La sensibilité du rendement par rapport à l'azote est captée par le paramètre τ.

$$y(N) = B - (B - A)e^{-\tau N} \tag{2.2}$$

Parmi toutes les courbes possibles, une seule est retenue. Les courbes qui ne permettent pas d'atteindre le niveau de production par hectare observé dans le FADN sont éliminées. Ensuite, la courbe dont la pente de la tangente est la plus proche du rapport des prix (prix de l'azote divisé par prix de vente de la culture) est sélectionnée.

Grâce à cette méthode, deux voies d'exploration s'ouvrent. La première est liée au fait que les fonctions dose-réponse sont estimées à partir des données climatiques. Ainsi les effets biologiques du changement climatique peuvent être captés et intégrés dans AROPAj (Galko, 2007; Leclère et al., 2013; Aghajanzadeh-Darzi, 2014). La deuxième voie porte sur la quantité d'engrais appliquée et les polluants qui en résultent (Bourgeois, 2012). Ceci est possible grâce au modèle STICS qui fournit aussi une estimation des émissions de nitrates, de protoxyde d'azote et ammoniac (NO_3, N_2O et NH_3). À partir de ces résultats, des fonctions d'émissions sont ajustées et ensuite employées dans AROPAj (Bourgeois et al., 2014).

Désagrégation spatiale des sorties

La politique de protection des données personnelles du FADN ne permet pas la communication de résultats utilisant la localisation géographique des agriculteurs. Pour obtenir une représentation plus fine de l'activité agricole au sein des Régions, une technique est donc proposée par Chakir (2009) et développée pour le cas de la version V2 d'AROPAj dans Cantelaube et al. (2012). Dans un premier temps, plusieurs informations sont combinées pour estimer les probabilités de présence de chaque GT sur le territoire de sa région. Les étapes sont les suivantes :

1. Des données ponctuelles sur l'usage des sols (LUCAS, cultures) sont croisées avec des informations sur le climat, le relief, le type de sol et sa couverture (CORINE). Grâce à un modèle économétrique de type Multinomial Logit, les probabilités de présence des différentes cultures sont estimées.
2. Sur la base des estimations de l'étape précédente, la distribution des cultures sur le territoire de la région est calculée. Cette distribution initiale est affinée par la technique de l'entropie croisée généralisée (*Generalized cross entropy*) en employant les informations du FADN sur les surfaces dédiées aux différentes cultures.
3. À partir des résultats sur la distribution des cultures, les probabilités de présence des GT sont calculées. Pour la version V5 du modèle AROPAj, où l'irrigation joue un rôle important, des zones susceptibles d'être irriguées sont identifiées grâce aux données LUCAS. Les probabilités de présence des GT sont aussi évaluées en tenant compte des surfaces irriguées.

La Figure 2.1 présente un résultat de la spatialisation des groupes types. Grâce à ces estimations, le résultat des simulations du modèle peuvent être désagrégés à une échelle très fine (1 hectare).

Afin d'appliquer cette méthode à la version V5 d'AROPAj de façon automatique, nous utilisons un certain nombre de logiciels et de procédures informatiques. Les différents types de données sont croisées à l'aide d'algorithmes informatiques. Les cultures représentées dans le RICA sont regroupées en classes selon leur représentativité dans la région. Lorsqu'une culture couvre plus de 5% de la surface agricole dans l'échantillon RICA, elle est modélisée indépendamment (Table 2.2). La même classification est appliquée aux points LUCAS.

Les observations dans la base de données LUCAS sont utilisées comme variable dépendante dans le modèle Multinomial Logit. Un grand nombre de variables explicatives sont testées pour trouver l'équation qui maximise le nombre

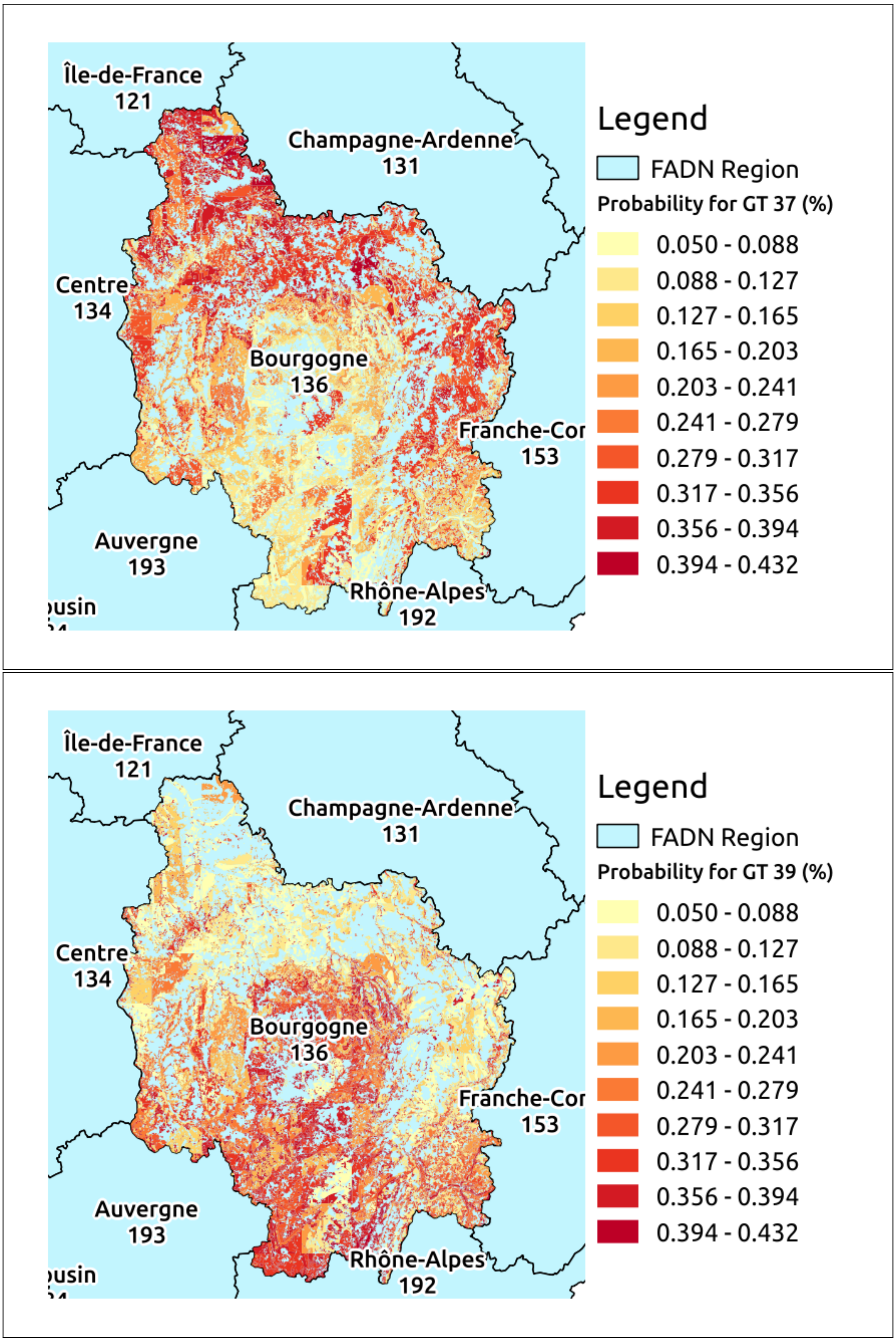


FIGURE 2.1 – Probabilités de présence des groupes types 37 et 39 de la région RICA 136, Bourgogne.

Culture	Culture >5% (Cat.1)	Culture <5% (Cat.2)	Cat.2 < 5% (Cat.3)
Céréales			
Blé (tendre)	Blé (tendre)	Céréales	Céréales
Blé (dur)	Blé (dur)	Céréales 2	Céréales
Orge	Orge	Céréales 2	Céréales
Avoine	Céréales 2	Céréales 2	Céréales
Seigle	Céréales 1	Céréales 1	Céréales
Other Céréales	Céréales 1	Céréales 1	Céréales
Maize	Maize	Céréales 2	Céréales
Riz	Riz	Céréales	Céréales
Cultures industrielles			
Légumineuses	LEG 1	IND	IND
Pommes de terre	PDT	IND	IND
Betterave sucrière	BTS	IND	IND
Tobac	TAB	IND	IND
Cotton	COT	IND	IND
Lin	LIN	IND	IND
Autres cultures industrielles	AIN	IND	IND
Colza	CLZ	PR1	PRO
Tournesol	TOR	PR1	PRO
Soja	SOJ	PRO	PRO
Cultures fourragères	FOU	PPR	PPR
Friches	FRIC	PPR	PPR
Prairies temporaires	FRIC	PPR	PPR
Autres cultures arables	AUC	AUC	AUC
Prairies permanentes	PPR	PPR	PPR

TABLE 2.2 – Regroupement des cultures selon leur représentativité dans la région RICA.

de points correctement prédits par le modèle. Un algorithme a été développé sous `R` afin d'automatiser la sélection des variables . Trois boucles sont exécutées sur l'ensemble des variables explicatives potentielles. Dans chaque boucle, chaque variable est ajoutée à l'équation du modèle Multinomial Logit et la performance du modèle est comparée à celle des modèles précédents. Si la dernière variable permet d'atteindre au moins la meilleure performance déjà enregistrée, cette variable est retenue dans la formule. À la fin des trois boucles, la formule qui donne le score maximal avec le moins de variables explicatives est sélectionnée. Ensuite, les estimations du modèle Multinomial Logit sont affinées grâce à la technique d'entropie croisée généralisée. Cela implique l'emploi d'une procédure `R` qui génère des programmes `GAMS` et lance leur exécution.

Pour prendre en compte l'irrigation dans la spatialisation des GT, des informations supplémentaires ont été utilisées. Elles sont obtenues à partir de la base des données LUCAS qui recense l'irrigation à des fins agricoles. Dans un premier temps, une grille régulière de 10 x 10 km est définie et la présence de points irrigués est testée pour chaque maille de la grille. Les mailles où de tels points sont observés sont retenues et considérées comme étant plus susceptibles d'abriter des GT qui irriguent. Ensuite, la probabilité de présence de chacun des GT est évaluée en tenant compte de l'irrigation et de l'altitude (Jayet et al., 2015, Section 8.2.2).

2.2 FFSM++

Dans le Chapitre 5, des sorties du modèle FFSM++ (*French Forest Sector Model*) en climat présent et futur sont utilisées. Le modèle FFSM++ est un modèle économique d'équilibre partiel développé au Laboratoire d'économie forestière (LEF) de l'INRA. Sa structure récursive repose sur deux modules : le premier est dédié à la dynamique de la ressource forestière et le deuxième à la dynamique du secteur (Caurla et al., 2010). La ressource forestière de la France est modélisée par 2574 cellules selon la région administrative, la gestion, les espèces et la classe de diamètre (Wernsdörfer et al., 2012). Les produits sont divisés en 10 groupes dont 4 concernent les produits bruts. Enfin, trois types d'agents sont représentés dans chaque région : les fournisseurs de bois (ou les gestionnaires), l'industrie de transformation et les consommateurs. Les prix sont endogènes sur le marché national et exogènes par rapport au marché international (Figure 2.2).

Chaque année, le module de ressource donne de l'information sur la quantité de ressource disponible au module du marché. Ce module calcule l'équilibre du marché en fonction de la demande nationale, les prix du marché international et estime la récolte de bois. Cette récolte est déduite de la ressource disponible dans le module de la forêt et, après l'application de l'accroissement annuel de la ressource, les quantités disponibles pour l'année suivante sont évaluées.

Suite au programme ORACLE, les ressource forestières du modèle ont été spatialisée pour mieux tenir compte de la diversité au sein des régions (Lobianco et al., 2015b). De plus, un module supplémentaire est intégré afin de prendre en compte les choix des gestionnaires forestiers (Lobianco et al., 2015a) et leur comportement vis-à-vis du risque.

Cette nouvelle version FFSM++ permet d'explorer des scénarios climatiques grâce aux variables décrivant la croissance et la mortalité des arbres. Ces deux variables sont estimées par un modèle statistique de type GAM (Wood, 2006) développé par Pierre Mérian [3] et Jean-Daniel Bontemps [4]. Les gestionnaires ont des attentes différentes pour le climat futur et pour les prix de marché. Ils peuvent modifier leur choix d'essences et de replantation en fonction de ces attentes. Ils maximisent l'équivalent d'un revenu annuel (*equivalent annual income*) où seul le revenu obtenu à la récolte finale est pris en compte. Ce revenu ne peut pas directement être comparé à la marge brute agricole (Lobianco et al., 2015b). Néanmoins, c'est un indicateur élaboré de la profitabilité du secteur forestier, sensible au changement climatique. C'est un indicateur particulièrement intéressant dans le cadre d'études sur l'usage des sols en climat futur.

3. IGN, Laboratoire de l'inventaire forestier.
4. AgroParisTech UFR FAM.

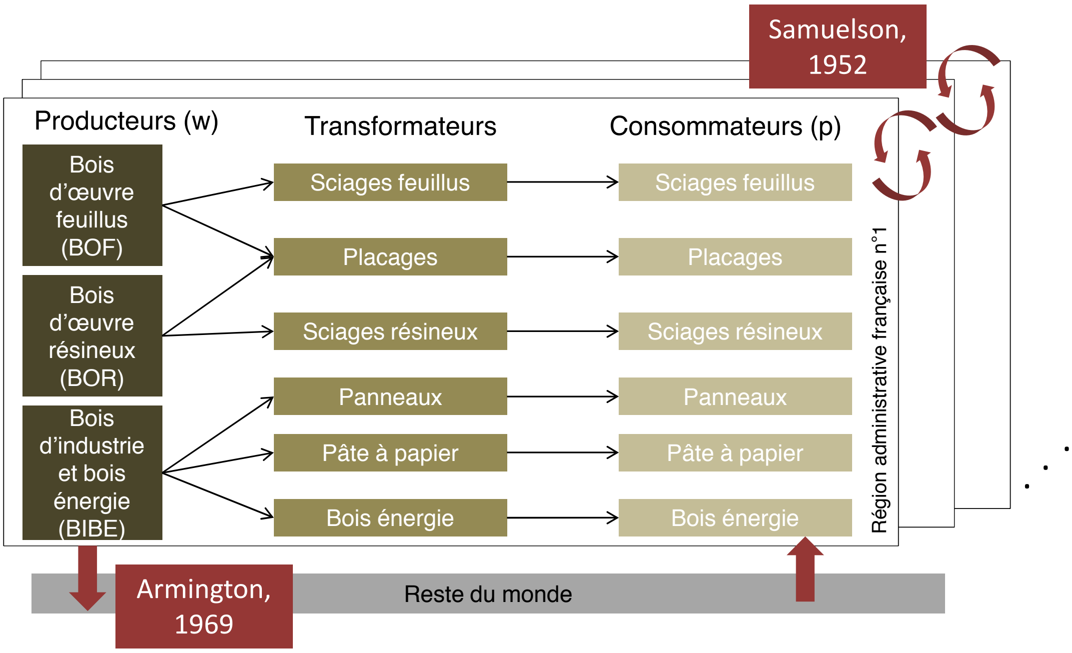


FIGURE 2.2 – Résumé du module économique du modèle FFSM++, figure extraite de Caurla (2012).

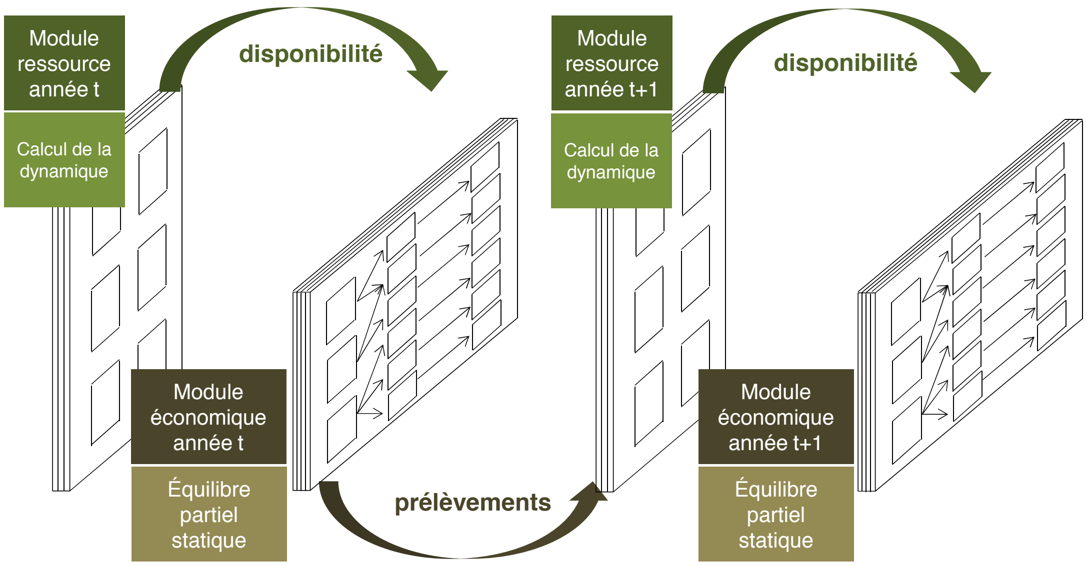


FIGURE 2.3 – Flux d'information entre les modules du modèle FFSM++, figure extraite de Caurla et al. (2013).

Deuxième partie

L’allocation des sols au sein du secteur agricole et sa prise en compte dans l’évaluation des politiques publiques

Le choix de cultures est primordial pour la rentabilité des agriculteurs. Plusieurs facteurs influencent ce choix, notamment les prix des denrées alimentaires et des intrants, les contraintes biophysiques, techniques ou institutionnelles, ainsi que l'aversion au risque des fermiers. L'allocation des sols entre cultures permet alors aux fermiers de répondre aux chocs exterieurs, comme le changement climatique ou l'introduction de nouvelles politiques publiques.

L'analyse présentée dans cette partie porte sur les effets d'une politique environnementale qui vise la pollution par les nitrates d'origine agricole. Un de ces effets est, précisément, le changement dans l'allocation des sols entre différentes cultures. Pour réaliser cette étude, nous utilisons le modèle AROPAj. La technique de désagrégation des résultats donne la possibilité d'évaluer les résultats de nos simulations à une échelle très fine. Nous nous y appuyons pour tester les gains potentiels en termes d'efficacité-coût liés à la différenciation spatiale de la politique publique étudiée.

Les conséquences environnementales de l'activité agricole se font ressentir à la fois à l'échelle planétaire à travers ses émissions de gaz à effet de serre, et à l'échelle locale avec la pollution par les nitrates. Le système hydrique est également directement affecté également par l'irrigation. De ce fait, l'analyse des usages des sols au sein de l'agriculture à une échelle fine peut être très intéressant pour l'étude des effets du changement climatique sur la ressource en eau, tant du point de vue de sa disponibilité que de sa qualité.

Notre travail au sein du projet ORACLE nous a mené à interagir avec des hydrologues. Cette collaboration a ouvert de nombreuses perspectives de recherche sur les effets de l'activité agricole sur l'eau dans le contexte du changement climatique.

Chapitre 3

La pollution par les nitrates et la différentiation spatiale des politiques publiques dans le cas de la France

Dans ce chapitre est traitée la question de la différentiation spatiale d'une taxe sur l'intrant azoté, étudiée en collaboration avec Pierre-Alain Jayet.

Dans le cas d'une pollution diffuse, unse taxe spatialement différenciée a théoriquement un meilleur rapport coût-efficacité qu'une taxe uniforme. Cependant, les résultats des études empiriques ne sont pas concordants. Le cas d'étude présenté ici est centré sur la France et compare plusieurs scénarios de politique publique à différentes échelles. Le modèle AROPAj nous permet de tenir explicitement compte des effluents d'élevage utilisés comme engrais ainsi que du changement de cultures opéré par les agriculteurs lorsqu'ils s'adaptent aux conjonctures simulées. Cette dernière dimension s'avère importante puisque, dans de nombreux cas, l'introduction de la taxe sur les engrais n'engendre pas les effets escomptés. En effet, le changement de cultures peut générer une augmentation des émissions polluantes par rapport à la situation sans intervention publique. Cet effet paradoxal de la taxe sur l'intrant est plus facile à percevoir lorsque l'échelle de l'étude est fine.

Nitrate pollution and spatial differentiation of taxation schemes applied to France

Abstract. Agricultural emissions of nitrates and their environmental consequences are a difficult problem for policy-makers to address. The nonpoint source character of the emissions and the legal constraints limit the regulatory options. The goal of the present study is to estimate the economic effects on farmers' profits of the spatial differentiation of pollution fees on mineral fertilizer and livestock units. We compare three policy scenarios, namely : 1) a benchmark case where farmers are constrained individually ; 2) uniform fees per river basin district ; and 3) differentiated fees. The estimates are from an agricultural supply model (AROPAj) coupled with a crop model (STICS) explicitly accounting for animals and animals manure. Results show that differentiated policies are less costly. We find numerous cases in which the second-best fees have paradoxical effects regardless of the taxation scenario. This calls for a greater flexibility when policies are established given the specific conditions and agricultural practices.

3.1 Introduction

Each year French households spend 1-1.5 billion Euros on additional water treatment because of agricultural pollution, or 7-12% of their water bill (Commissariat Général au Développement Durable, 2011). In the most polluted areas such costs can reach 494 Euros per household annually. These numbers do not include the negative financial impact on tourism or the additional health expenses. Although agricultural pollution has been addressed by numerous studies and public policies, the problem persists. A possible instrument for reducing it at its very source is an environmental tax. Such taxes can rarely be conceived in a socially optimal way (as Pigou himself recognized, Pigou 1937) but, nevertheless, offer a cost-effective solution. In this paper we investigate the potential environmental and economical effects of the spatial differentiation of a pollution fee in France in the context of the European Union Water Framework Directive (EU WFD, European Community 2000), the European Union Nitrates Directive (The Council of the European Communities, 1991) and the national legislation of the country.

Nitrogen (N) pollution from agriculture consists of three different polluting emissions, namely nitrates (NO_3), nitrous oxide (N_2O) and ammonia (NH_3) resulting from fertilizer or animal manure application (Bourgeois et al., 2014). As these pollutants are associated with different negative externalities their combined regulation is quite challenging and involves mostly second best regulatory solutions. For instance, in the case of water pollution by nitrates, the exact nitrate leaching from agriculture could only be observed at great costs. The sources of nitrogen can be marketed (generally mineral fertilizers) or produced on-farm (animal manure and other organic residuals). In order to predict nitrate emissions, the relation between crop - nitrogen source - pedo-climatic conditions should be taken into account in its integrity. However, public policies based on such a perfect knowledge, even when conceivable, may not be easily implementable because of national legal landscape or social disapprobation. In the present study we propose an assessment of a second best spatially differentiated taxation schemes with two pillars, namely mineral fertilizer used on field and livestock. In the case of nitrate pollution of water bodies the damage and the abatement costs are unknown. Hence, the regulation can be designed following Baumol and Oates (1971), where a uniform standard is defined and the most cost-effective way of achieving it is applied.

Generally the literature on nitrate pollution is focused on the policy design allowing the respect of an environmental quality standard. The difficulties stemming from the fact that nitrates pollution is mostly non-point sourced have defined the major topics on this question. Although the problem can be formulated with greater attention to the issue of pollution location (proximity to drinkable water sources, for instance), we center our study on the policy design with respect to the differentiation of the contribution of non-point sources (farmers) of nitrate pollution. The effect on ambient pollution levels from the application of nitrogen on fields is difficult to establish not only because of the complexity of the geological systems but also because of the stochastic nature of climatic variables. One of the options for the regulation of the pollution would be the establishment of field-specific nitrogen quotas designed in a way guaranteeing the respect of the environmental standard. This possibility is becoming more and more realistic as the costs for remote sensing and observation are getting lower. Unfortunately, at the present time, these costs remain prohibitive.

As theory in environmental economics have shown, the objective of an environmental standard could also be achieved through the introduction of the so called Pigouvian fee (Figure 3.2). Nitrate pollution is defined at the scale of a water body. These water bodies and the dynamics that define their respective nitrate pollution levels may be very different. If the policy-maker has to introduce a unique Pigouvian fee for more than one water body and she is obliged to respect the same environmental standard in both cases, then the higher tax should be imposed so that standard is enforced everywhere. This tax would restrain the polluting activity more than the necessary for the water body where the standard is respected at a lower cost, which is leading to an inefficiency of the policy.

Furthermore, the various sources of nitrate leaching have differing effects on different water bodies with respect to their location and water run-off. Thus, the individual contribution of farmers to the nitrate concentrations in a given water body varies following the physical reality of their respective fields. In such a setting, if the environmental issue is to be tackled via Pigouvian fees, Tietenberg (1974) and Xepapadeas (1992) have shown theoretically that taxation schemes are more efficient when the spatial heterogeneities are taken into account. Numerous studies have evaluated the cost-effectiveness of such policies in the cases of nitrate and/or phosphate concentrations in water. Nevertheless, evidences from the empirical literature are contradictory. Westra and Olson (2001) used a positive mathematical programming model of a stylized watershed for two different crops, phosphorous effluents and tax differentiation based on “agroecoregions”, which are distinguished by their pedo-climatic specificities. The study by Lacroix et al. (2010) is based on the crop model STICS combined with an economical model to test the effects of soil/practice differentiation of policies for a given water body. These two studies prove that there can be substantial gains from tax differentiation, whereas Fleming and Adams (1997) show benefits that are considered insufficient to cover the higher transaction costs. The latter used an integrated three-step modeling procedure, where farmers first decide their profit maximizing quantity of fertilizers which is then sent as an input to a soil model and linked forward to a groundwater model. The tax distinction is based on different soil types. Their findings are in agreement with those of Helfand and House (1995). Helfand and House (1995) also use a crop model (EPIC) to simulate the nitrate effluents from lettuce for two soil types and link these results to profit maximizing linear program representing farmers' decisions. Claassen and Horan (2001) use endogenous prices which is amplifying the differences between uniform and non-uniform policies. Xabadia et al. (2008) introduce time in their analysis of stock pollution in order to study the dynamic effects on technology and land use.

Recent developments in the Geographical Information Systems (GIS) and the resulting spatialization of mathematical models, render research at greater geographical extents possible. In our study, we use an agricultural supply model, covering metropolitan France [1] at the regional level. The model is applying to crop and animal breeding activities where animal manure is taken into account for fertilization. This represents a major difference in comparison to models used in literature. Estimates from the crop model STICS are used as dose-response functions linking crops' yields and nitrogen inputs. Results are down-scaled and aggregated at the level of hydrological sectors [2]. The model covers different levels of production heterogeneity, such as soils, crops, climate and other productive and economic specificities of farmers. This methodology allows us to investigate the effects of input-based water body-specific pollution fees. The recent studies of Jayet and

1. Except Corsica. The model uses FADN data. For Corsica the privacy policy of FADN does not allow us to apply the model's methodology.
2. A subdivision of the river basin districts (RBD) established in the EU WFD.

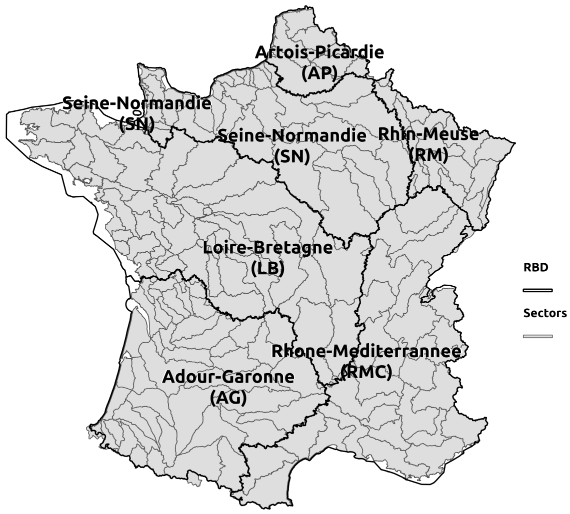
Petsakos (2013) and Bourgeois et al. (2014) are based on the same bio-economic model as our investigation. The former is focused on the comparison of the nitrate emissions abatement costs under different scenarios of the Common Agricultural Policy (CAP) of the European Union. Bourgeois et al. (2014) compare the cost-effectiveness of a uniform nitrogen tax to a mixed policy (a tax on nitrogen and a subsidy on a low-input plant, miscanthus) in terms of the abatement of the three pollutants resulting from fertilization.

Except Fleming and Adams (1997) and Westra and Olson (2001), most of the empirical studies on tax differentiation consider a single crop and thus the effects on land-use change are neglected (Goetz et al., 2005). The switch between crops can result in paradoxical taxation effects as reported in Jayet and Petsakos (2013). We study multiple crops on a broader scale and obtain results for a larger number of water bodies. In comparison to previously existing literature and to the best of our knowledge, the present study is the first to cover livestock as a source of nitrogen in the context of a spatially differentiated pollution fee.

In the course of this article we will first describe broadly the case of nitrate pollution in France and its environmental legislation on the matter. The legal landscape in the country forbids the implementation of the first best taxation scheme derived in Section 3.3. In Section 3.4, we compare the results for three policy scenarios for France at the scale of river basin districts (RBD). These scenarios are : 1) a benchmark scenario where policy is individualized ; 2) a uniform RBD fee only in polluted areas ; and 3) a differentiated fee in polluted areas. In order to approximate abatement costs, we use the farmers' loss of gross margin and the tax revenues. Our results show that there can be substantial gains from the individualization of pollution fees. Nonetheless, when taxes are defined at finer levels they induce lower losses. We find numerous cases in which the input-based tax can have paradoxical effects regardless of the taxation scenario. This calls for a greater flexibility when policies are established given the specific conditions and agricultural practices.

3.2 Background

Nitrate pollution in France is very unevenly distributed across the country. The highest nitrate loads are observed in the North and North-West. Concentration levels in certain regions exceed the EU norm of $50\ \mathrm{mg/l}$ and as such the country is running the risk of sanctions. In 2013, 55% of the utilized agricultural land in France is classified as Nitrate Vulnerable Zone (NVZ) following the European Union Nitrates Directive (The Council of the European Communities, 1991). These zones are defined as "areas of land which drain into polluted waters or waters at risk of pollution and which contribute to nitrate pollution". Because of the nitrogen cycle, nitrates occur naturally in water but in small quantities. A groundwater concentration of less than $25\ \mathrm{mg/l}$ is considered natural (Viennot et al., 2009) or subject to low anthropogenic pressure.



RBDs :

- Adour-Garonne (AG)
- Artois-Picardie (AP)
- Loire-Bretagne (LB)
- Seine-Normandie (SN)
- Rhin-Meuse (RM)
- Rhône-Méditerranée et Corse (RMC)

FIGURE 3.1 – River basin districts (RBDs) and their associated sectors in France.

France has special institutions that govern the country's water resources. These are the Water agencies. They correspond to the RBD division recommended by the EU Water Framework Directive (WFD). Figure 3.1 shows the six Water agencies in France, subdivided into hydrological sectors. Water agencies are empowered to collect pollution fees and to distribute subsidies and loans on their territory. The decisions are made by agencies' parliaments which represent the different water users.

Recent modifications in French Environmental Code address nonpoint source pollution and introduce pollution fees on livestock units [3]. Article L213-10-2 of the Environmental Code states (Legifrance, 2014) that pollution fees are fixed by water agencies at a coherent geographical unit level and with respect to the : 1) state of water bodies, 2) the risk of infiltration of pollutants, 3) prescriptions by water or other police, and 4) the objectives decided at the RBD level. Following this legislative framework, we focus our study on the economic effects of geographically differentiated tax on nitrogen fertilizers.

Such taxes have some limitations originating from the asymmetric information on input use. There are incentives for agents facing lower taxation rates to resale fertilizers to others (Helfand and House, 1995) making it a moral hazard problem. In France this could be less problematic as farmers in Nitrate Vulnerable Zones (NVZ) are obliged to keep records of the nitrogen applied. False declaration can result in exclusion from the Single Payment Scheme (SPS). This way, the tax can be levied on the base of farmers' declarations. Supposing a competitive market for fertilizers, the input's supply curve should remain unaffected in the short run by the policy.

Every regulation scheme induces transaction costs related to its implementation, monitoring and control. These costs can be even more important in the case of tax differentiation. Their amount can finally surpass the gains from rate differentiation (Helfand and House, 1995). Nevertheless, in most countries there is an administrative infrastructure already in place at different levels. In France, the RBD agencies exist since the 1960s. They have been levying fees for industrial and domestic pollution from firms and households since their establishment and from stockbreeding farms since 2006. Thus, a tax can be introduced at a relatively low additional cost.

3. The pollution fee is applied when there are more than 1.4 livestock units per hectare of utilized agricultural land in the farm. The charge is due for the units exceeding a certain threshold (40 units).

Another important shortcoming is related to the volatility of the prices of agricultural commodities. This is also a limiting factor for the real-world efficiency of input-based taxes. Hence, the more flexible the tax is the better it will achieve its goals.

Ultimately, there is the question of feasibility which makes one taxation scheme preferable over another when policies should be implemented. Environmental regulations should be designed with respect to countries' institutions and legal landscape which can impose significant constraints to policy-makers. In Section 3.3 we define the properties of an input-based tax where farmers' profit is maximized under the environmental constraint imposed by the Nitrate Directive. Then, in Section 3.4 we propose a feasible water body-specific pollution charge with two bases, mineral fertilizer and livestock,and estimate the result from its application in France .

3.3 Theoretical model

The starting point of our static theoretical model is provided by Shortle and Horan (2002). They consider different cases regarding the environmental constraints (damage or concentration levels) and the design of policies (input or ambient-based instruments). We focus our research on a concentration constraint and an input-based taxation scheme. Our study also refers to Segerson (1988) because we administer the environmental policy according to ambient pollution levels.

The equivalence between a pollution standard and a tax has been well established in environmental economics literature (Helfand et al., 2003) in the case of a fixed number of price-taking firms. For nitrate concentrations in water bodies the problem can be tackled in a similar way. On the abscissa of Figure 3.2, the level of nitrate concentration is presented and on the ordinate is given the associated marginal abatement costs (MAC) of the polluting firms. We consider the marginal damage (MD) infinite after a certain limiting value (L) in order to take

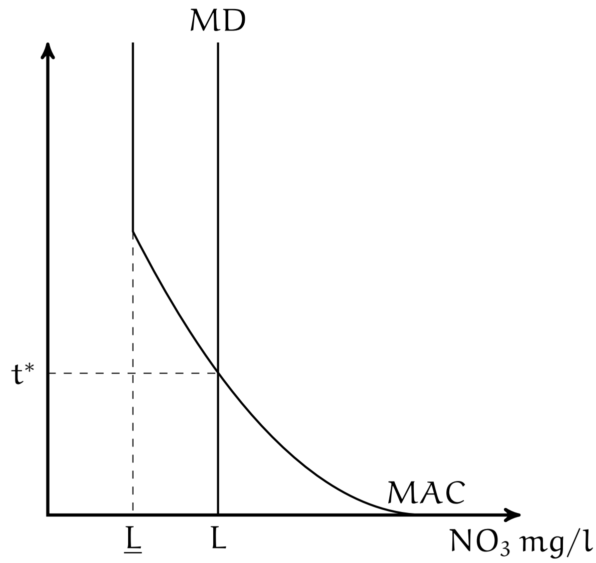


FIGURE 3.2 – Equivalence between the environmental norm and a Pigouvian fee (t^*). The marginal damage (MD), the marginal abatement cost (MAC), the natural level of nitrate concentration ($\underline{L}$) and the limiting value (L) are given on the graph. The Pigouvian fee, t^*, leads to the same level of pollution as a norm fixed at L would.

into account the European Union Nitrates Directive threshold value of $50\ mg/l$. The value $\underline{L}$ represents the naturally present level of nitrates, which cannot be abated by the polluters at any possible cost as they are not the source of this pollution. Nitrate loads are the cumulative effect of the individual nitrate emissions. Thus, in order to implement any of the two environmental policies (a tax or a standard), the policy maker should know the individual contribution of each polluter. The need for individualized taxes or standards arises when the marginal contribution to the pollution differs among polluters. Later on in the article, we use the environmental standard for the estimation of the parameters for different taxation schemes and as a benchmark case. We should stress out that the objective of the environmental tax here is not the allocative efficiency among producers. The sole objective is the respect of the imposed norm at the lowest possible cost for society.

Pollution in water results from point and non-point source nitrate leaching. For the sake of simplicity, in our theoretical model we address only non-point source nitrate leaching originating from the use of mineral fertilizers and animal manure by farmers. The framework includes a set of farms (indexed by $i = 1, ..., i, ..., n$) characterized by specific profit functions and geographically referenced positions associated with a water body ($w = 1, ..., l$). This is the first difference from Shortle and Horan (2002) in which only one water body is considered. One farm can have fields connected to different water bodies. Farmers maximize their profit (π_{iw}) by choosing the levels of their inputs (x_{iw}). We apply standard assumption for the profit function, namely concavity ($\partial^2 \pi_{iw}/\partial x_{iw}^2 < 0$) and the existence of internal solutions for $\partial \pi_{iw}/\partial x_{iw} = 0$ where $x_{iw} > 0$. The nitrate emissions, r_{iw}, of the farm i affecting the water body w are determined by her input use, x_{iw}, site characteristics and stochastic environmental variables(z_{iw} [4]) following the relation : $r_{iw}(x_{iw}, z_{iw})$.

Emissions define pollution concentrations following a water body-specific pollution fate and transport function, i.e. , $a_w(r_{1w}, ..., r_{iw}, ...)$ [5] where $\partial a_w/\partial r_{iw} \geq 0 \ \forall i$. The policy-maker introduces a uniform [6] environmental standard (L) limiting the pollution. Thus, we take into account only the pollution concentration and not the resulting damage to the society as in Shortle and Horan, 2002. The environmental standard is supposed to be the threshold value above which pollution is unacceptable by society. The policy-maker programme consists of a maximization of the sum of farmers' profits and an environmental constraint comparing the expected pollution concentration ($\mathbb{E}(a_w)$) with the environmental standard (L).

$$\left| \begin{array}{ll} & \max\limits_{x_{iw}} \sum\limits_{i} \sum\limits_{w} \pi_{iw}(x_{iw}) \\ \text{s.t.} & \mathbb{E}(a_w) \leq L \qquad (\lambda_w) \end{array} \right. \tag{1}$$

We first focus on the case where the optimal solution is obtained with a non-binding environmental constraint ($\mathbb{E}(a_w) < L$) for the w-th water body. This implies a zero value for the related dual variable (i.e. $\lambda_w^* = 0$). The first order opti-

4. See Shortle and Horan (2002) for the full specification of the emission function
5. Idem.
6. As the one defined by the European Union Nitrates Directive of 50 NO_3 mg/l.

mality condition (FOC) related to the input is presented in Equation (2).

$$\frac{\partial \pi_{iw}(x^*_{iw})}{\partial x_{iw}} = 0 \quad \forall i, w \tag{2}$$

The asterisked primal and dual variables denote the solution of this programme. Hence, in the case where the environmental constraint is non-binding, the policy-maker's FOC coincides completely with farmers' private FOC.

The above statement does not hold when the environmental constraint is binding ($\lambda^*_w > 0$) as Equation (3) shows.

$$\left| \begin{aligned} & \mathbb{E}(a_w) = L \\ & \frac{\partial \pi_{iw}(x^*_{iw})}{\partial x_{iw}} = \lambda^*_w \, \mathbb{E}\left(\frac{\partial a_w}{\partial r_{iw}} \frac{\partial r_{iw}}{\partial x_{iw}}\right) \qquad \forall i, w \end{aligned} \right. \tag{3}$$

Equation (3) balances the marginal profit and the expected contribution to pollution concentration of an additional unit of x_{iw}. These two figures are not uniform accros geographical space. Consequently, the policy-maker is addressing nitrate pollution via a tax it would be *a priori* iw-dependent. The introduction of a fee on polluting activities is modifies farmers' objective function. Farmers maximize after-tax profit $\pi_{iw}(x_{iw}) - t_{iw}x_{iw}$. The FOC (Equation (4)) leads to equality between marginal profit of the activity and the tax.

$$\frac{\partial \pi_{iw}(x^*_{iw})}{\partial x_{iw}} = t_{iw} \tag{4}$$

From Equation (3) and (4), we define the level of the tax which has non-zero value when the environmental constraint is binding. Equation 5 represents the first best water body and firm-specific tax.

$$t_{iw} = \lambda_w \, \mathbb{E}\left(\frac{\partial a_w}{\partial r_{iw}} \frac{\partial r_{iw}}{\partial x_{iw}}\right) \tag{5}$$

As appealing as Equation (5) might be, it masks a certain degree of complexity. Bourgeois et al. (2014) estimate functions for nitrate emissions for the six French river basin districts (RBD) and different crops. The functions are assumed to be linear and the coefficients (intercept and slope) differ widely between crops and between RBDs (soil and climate conditions). Furthermore, nitrate emissions are also dependent on the type of fertilizer and animal manure applied on field. Thus, a first best input tax should also be differentiated by the crop and the type of fertilizer. These complications could be bypassed by basing the tax directly on nitrate emissions which, unfortunately, remain difficult to assess in the real world. Our second best policies aim at regulating the use of marketed and on-farm produced nitrogen by introducing a tax on mineral fertilizers and livestock units.

Equation (5) also shows that the level of the tax is indexed t_{iw} because it is firm- and water body-specific. We should note that firm-specific tax differentiation would be illegal in France because of the fiscal equity principle. For this reason, later in Section 3.4, we investigate pollution fees differentiated solely between water bodies and compare them to an optimal scenario.

3.4 A feasible water body-specific pollution fee on nitrogen fertilizers and livestock in France

Hypothesis

Water bodies from the model in Section 3.3 are supposed to coincide with hydrological sectors, subdivisions of RBD. There are more than 200 such sectors in metropolitan France (Figure 3.1). As discussed in Section 3.3, the pollution fee should be firm- and water body-specific [7] but French law would not permit such specification. Nevertheless, the geographical differentiation of water pollution fees is established in the Environmental Code and is already implemented. For instance, in the Seine-Normandie Water agency different coefficients are applied to the industrial pollution fees depending on firms' location and on the state of the water [8].

7. In Section3.3 we even argue that the tax should be crop- and nitrogen source-specific.
8. Seine-Normandie Water agency `http://www.eau-seine-normandie.fr/index.php?id=5145`

Identifying the zones with potentially binding cases

Equation (5) relates the pollution charge level to the dual value of the environmental constraint (λ_w) and the marginal effect of the nitrogen input on pollution concentrations. The pollution fee is applied only when the environmental constraint is binding ($\lambda_w > 0$). In order to determine whether the constraint is binding or not, we intersect sectors with NVZs. If a NVZ is present on the territory of a sector, we consider that potentially $\lambda_w > 0$. Figure 3.3a shows the geographical distribution of the NVZ and the maximal concentrations between surface and groundwater in the sectors (Service de l'observation et des statistiques, 2011).

The sector-specific tax

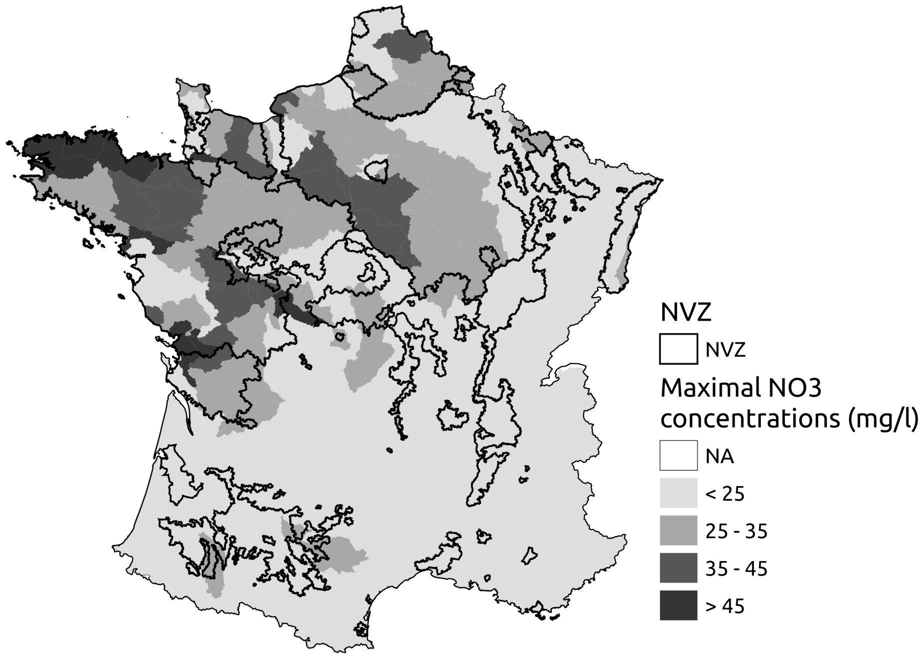
Given the regulatory limits, we focus on a water body-specific pollution fee. Equation (5) imposes a tax that is equal to the expected marginal pollution concentration resulting from the application of one supplementary unit of input by the i-th firm in the w-th water body multiplied by λ_w. Now, a water body-specific fee would equal the marginal profit of farmers across the water body.

In order to evaluate the marginal effect of nitrogen on pollution concentrations, we use the Simplified Vulnerability Index (Network Persistence and Development Index, IDPR) developed by BRGM [9]. The index range spans between 0 and 2000 where lower values indicate high infiltration of the surface water towards groundwater (Figure 3.3b). Values above 1000 mark the predominance of water run-off into adjacent surface water bodies. Using geographical information systems, we aggregate the IDPR at the sectors level. This index is used in France to estimate broadly the vulnerability of groundwater [10].

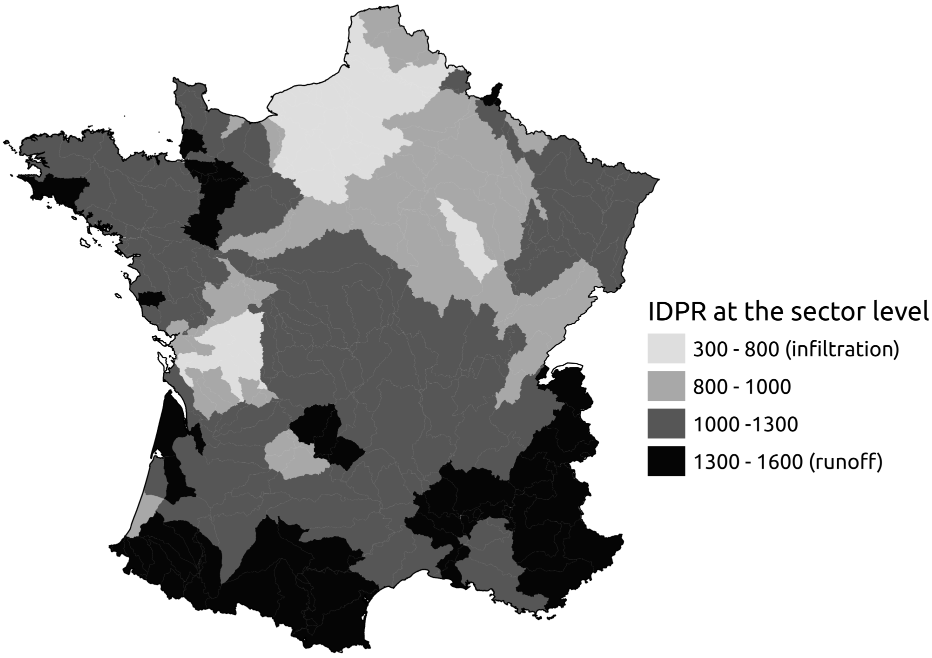
In our study, we apply this index in a statistical model that explains the observed nitrate concentrations in surface water with the average quantity of fertilizers and the average number of livestock units at the scale of the hydrological

9. Bureau de recherches géologiques et minières, `http://www.brgm.fr/`.

10. Numerous reports of the BRGM (`http://brgm.fr`) use a combination of IDPR and the depth of the unsaturated zone to evaluate the vulnerability (e.g., the simplified vulnerability assessment of the SN RBD available at `http://sigessn.brgm.fr/spip.php?article71`) . In this study different weights are assigned to the IDPR and the depth of the unsaturated zone. Our study uses only the IDPR.



(a) NVZ and nitrate concentration (maximal value between groundwater concentration and surface water, Service de l'observation et des statistiques, 2011).



(b) Network Persistence and Development Index (IDPR) provided by BRGM (`brgm.fr`).

FIGURE 3.3 – NVZ delimitation, nitrate concentrations and IDPR.

sector. This model allows us to approximate the marginal contribution to nitrate concentration of one extra unit of fertilizer or of livestock on the average.

A nitrate concentrations statistical model at the water body scale

We propose a simple statistical model given in Equation (6) which we estimate through ordinary least squares (OLS) technique. The model is relating the observed nitrate concentrations (surface water) to the two sources of nitrogen from agriculture : mineral fertilizers and livestock manure.

$$\begin{aligned} Surface\, water\, \mathrm{NO_3}\; concentration_w = \alpha_0 + \alpha_1 \mathrm{IDPR}_w + \alpha_2 \mathrm{Fertilizers}_w * \mathrm{IDPR}_w \\ + \alpha_3 \mathrm{Livestock}_w * \mathrm{IDPR}_w + \epsilon_w \end{aligned} \tag{6}$$

The data on the explanatory variables is supplied by the AROPAj agricultural supply-side model (see Subsection 3.4). The intercept is close to the value of 25 mg/l which is considered to be the $\mathrm{NO_3}$ concentration that occurs naturally. The model performs well in terms of variation explained (R^2 of 0.68) and all coefficients are reported as significant at the 1% confidence level. The actual levels of concentration (indexed in ascending order) are plotted on Figure 3.4b. The dashed line on the figure connects the predicted values. As the figure shows, the model tends to underestimate the concentrations for values above 23 mg/l. As for observed concentrations, the values for surface pollution do not exceed 50 mg/l and for groundwater only six sectors surpass the norm. We focus on surface water as the statistical model is performing better for this level of nitrate pollution (R^2 of 0.68 versus 0.39 for groundwater). On average, groundwater pollution is 25% higher than in surface water and the two are positively correlated (Pearson r of 0.68). Because of the high level of correlation between the two concentration values, we consider the concentrations on the surface as a good proxy for the groundwater pollution. These limitations of the statistical model oblige us to lower the value for the environmental constraint. The norm we are enforcing is of 25 $\mathrm{mg/l}$ which, according to our statistical model, is infringed by 26 of the 169 sectors in consideration.

By the model specification, we implicitly suggest that the relation between concentrations and inputs is linear [11]. This assumption is necessary for the simulation of the tax schemes by the AROPAj model (see Subsection 3.4). The true relation between the nitrogen sources and the concentrations is beyond the scope of our present work. Nevertheless, the good coefficient of determination of our statistical model makes us confident in our approximations. Undoubtedly, a panel data on the concentration levels at the scale of the hydrological sectors would have provided us with a better insight. Unfortunately, to the best of our knowledge, such information is currently unavailable.

This statistical model has two purposes. First, by estimating Equation (6) we can introduce the optimal taxation scheme as an environmental standard (thanks to the equivalence between the environmental standard and the optimal Pigouvian fee) in the farmers' profit maximization problem and thus deduce the values of λ_w. Second, the estimated coefficients for Equation (6) are used as approximations for the marginal contribution of the additional unit of fertilizer or livestock. By combining the dual value and the estimated coefficients we obtain the values for the sector-specific tax. We should note, that by using the interaction terms between fertilizers and the IDPR ; and the livestock units and the IDPR (Equation (6)), we obtain sector-specific marginal effects.

Taxation schemes scenarios

In order to evaluate the potential benefits of the tax differentiation, we estimate and compare the loss of gross margin for the following taxation schemes :

1. A benchmark case where policy is individualized among farmers ;
2. A uniform tax rates per RBD in the hydrological sectors where there are NVZs ;
3. Two definitions for sector-specific tax rates in the sectors where there are

11. A *log-linear* specification gives an adjusted R^2 of 0.74 and an *exponential* model 0.78. Both models underestimate the high levels of concentration.

(Intercept)	30.61***
	(2.35)
IDPR	-4.82***
	(0.38)
Fertilizers† * IDPR	5.98***
	(0.75)
Livestock† * IDPR	1.56***
	(0.14)
Number of observations	169
R^2	0.68
adj. R^2	0.67
Resid. standard deviation	5.96
Residuals distribution :	

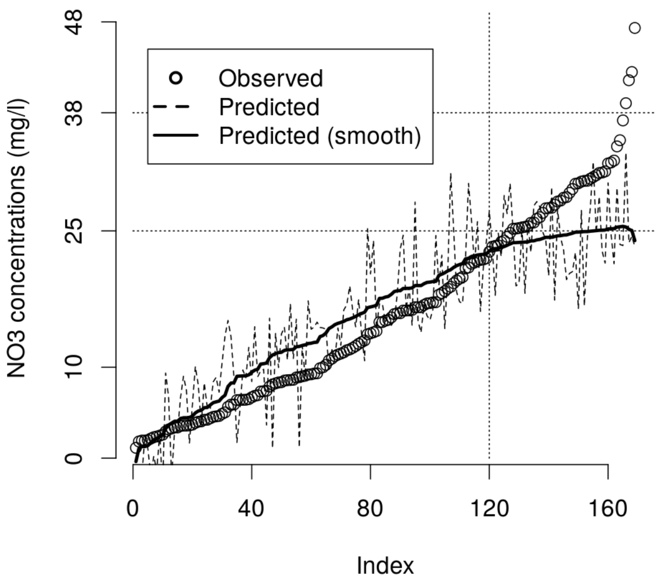
Min	1Q	Median	3Q	Max
-12.52	-4.12	-0.8	3.24	24.05

Standard errors in parentheses

* significant at $p < .05$; **$p < .01$; ***$p < .001$

† Fertilizers and livestock are expressed in units per hectare of the geographical surface of the hydrological sector.

(a) Statistical model (OLS) of nitrate pollution concentrations (NO_3 mg/l) observed in surface water in France.



(b) Plot of the observed nitrate concentrations (NO_3 mg/l) in surface water and the predicted values.

FIGURE 3.4 – Statistical model of nitrate pollution concentrations (NO_3 mg/l) observed in surface water in France.

NVZs.

Methodology

For our study we use the AROPAj agricultural supply model in concert with the crop model STICS. AROPAj is a static optimization model based on classical microeconomic assumptions and which uses the Farm Accountancy Data Network (FADN) database. A thorough description of the model is provided in Jayet et al. (2015). The economic agents are farmers grouped in representative "group-types" (GT) depending on their activity, economic size and altitude. Such models are commonly used quantitative methods applied to agricultural, environmental and resource economics (Kaiser et al., 2011). Originally, AROPAj is structured as an assembly of mixed integer linear programming models. Non-linearity occurs through integer variables and the use of dose-response functions linking nitrogen inputs and crop yields.

$$\left| \begin{array}{ll} & \max_{x_i} \pi_i(x_i) \\ \text{s.t.} & A_i x_i \leq s_i \\ & x_i \geq 0 \end{array} \right. \tag{7}$$

Equation (7) gives a summarized description of the optimization program of the group-types in AROPAj. Each economic agent of the model maximizes her gross margin (π_i) or the difference between the revenues and the variable costs. The optimization program is bounded by technical and structural constraints some of which related to the Common Agricultural Policy of the EU. GTs are indexed by i as are the farmers in our theoretical model in Section 3.3. The vector x_i has a broader sense, it integrates the productive activities of the economic agent i while g_i stands for the valuation of the activities in the gross margin. In our case, the activities of interest are the mineral fertilization and animal breeding. Livestock manure available to the farm is accounted for through explicit variables of the model and is applied on fields when there is any crop activity. The matrix A_i is represents the input-output constraints of the profit maximization problem and the right-hand side is given by the s_i vector of capacities. We are using the V2 version of the model (Jayet et al., 2015) with the 157 group-types comprising French agriculture.

In order to render the economic agents more adaptive to policy and/or price shocks, dose-response functions were introduced for each crop-group-type combination, (Godard et al., 2008). These functions are estimated via the crop model STICS (Brisson et al., 1998). An additional block is dedicated to the nitrogen balance and related to the N-to-yield functions. N-inputs are mineral fertilizers and five N-organic compounds sourced from manure. Jayet and Petsakos (2013) use the same method in their study of nitrogen input-based taxation. A similar approach is common in the literature (Helfand and House, 1995; Larson et al., 1996; Martínez and Albiac, 2006).

STICS estimates crops yields for different quantities of N fertilizer at the parcel level. The crop model entries include climate, soil, irrigation and some agronomical variables (e.g., varieties and preceding crops). GTs face specific biophysical conditions, the dose-response functions are estimated individually for each GT. The functional form used is exponential, increasing and concave (Equation (8)). It is, thus, compatible with the standard economic assumption of decreasing marginal productivity of inputs [12]. The function calls for the sum of the mineral and organic N. Nitrogen sources are indexed by o and the "rate of increase" of the function is captured by the parameter τ_{io}. This parameter is estimated initially for a mineral fertilizer of reference and then weighted fallowing and agronomical rule of equivalence between N sources. The quantities Y_i^{min} and Y_i^{max} represent the minimal and the maximal yield and do not depend on the type of N. The maximal yield is estimated asymptotically as the yield obtained under no N limitations for the crops. The yield when $N_o = 0, \forall o$ gives the parameter Y_i^{min}.

12. There are other specifications for the dose-response in literature (Gallego-Ayala and Gómez-Limón, 2009; Martínez and Albiac, 2006) which are also increasing and concave.

$$Y_i = Y_i^{max} - (Y_i^{max} - Y_i^{min})e^{-\sum_{io}\tau_{io}N_o} \tag{8}$$

where

Y_i is yield, $Y_i \in \left[Y_i^{min}; Y_i^{max}\right]$, and

N_o is the quantity of each type of nitrogen applied$(N_o \geq 0)$.

This procedure is applied for nine crops : common wheat, durum wheat, barley, maize, rapeseed, sunflower, soybean, potato and sugar beet (Leclère et al., 2013). Thus, the model solves for the optimal mineral fertilizer quantities, crop allocation, livestock units and their feeding and gross margin among others. The introduction of the dose-response functions makes the input use decision endogenous. This methodology allows great flexibility to economic agents while taking into account the technical constraints characterizing agricultural activities. However, as it is a static model, no feedback on prices of inputs and outputs is considered.

Nitrate emissions are also evaluated by the STICS model. They represent the losses of NO_3^- ions at the root level. Bourgeois et al. (2014) estimate linear emission functions (Equation (9)) based on the simulated results from STICS for each group-type i and each of the nine crops, c, mentioned above. As each GT is associated with only one FADN region, the GT specific coefficients β_{0ic} and β_{1ic} are estimated for the specific climatic and edaphic conditions of the region to which the GT belongs.

$$r_i(N) = \beta_{0ic} + \beta_{1ic}N \tag{9}$$

Another important aspect of AROPAj is the inclusion of livestock in the model. As animal manure is a source of nitrogen, we allow farmers to adjust their livestock capital in the range of $\pm$ 15% of the initial values. This limitation of the possible variation of livestock is necessary because the model does not account for the fixed capital related to animal breeding. Hence, we are considering the variations beyond $\pm$ 15% out of the calibration interval of the model.

Livestock manure application is limited to 170 kg of nitrogen per hectare per year for the territories in NVZ (The Council of the European Communities, 1991). This restriction is taken into account in the model. As our economic agents are all associated with FADN regions where there are NVZs and as the exact location of the agents inside the region is *a priori* unknown we apply this restriction to all group-types. We find that this limitation is rarely binding except in regions with important manure excess which are under NVZ.

The different fertilizers (ammonium nitrate, urea, etc.) available on the market are represented by a N-compound fertilizer for each group-type and each crop. Mean N content for this N-compound fertilizer is about 18% for a price of around 1 Euro per kilogram of N. This corresponds roughly to the price of nitrogen units in ammonium nitrate (NPK of 33.5-0-0). Similar value (0.90 Euros per kilogram) is found in Martínez and Albiac (2006). The model provides estimates for crop areas, yields, the amount of nitrogen used, nitrate emissions and farmer's gross margin and accounts for five types of animal manure. The results are obtained for each group-type associated with a unique FADN region.

Spatialization of AROPAj results

The exact geographical location of GTs inside the region is unknown due to the privacy policy of the FADN data. AROPAj results are therefore spatialized (Cantelaube et al., 2012) following the technique presented in Chakir (2009). FADN regions are divided into a 100 m $\times$ 100 m grid cells. First, remote-sensing data for land cover [13] is combined with land use survey data [14], weather [15] and soils [16]. A multinomial logit model is then estimated relating the land use (crop) with the

13. Corine Land Cover, for information see : `http://www.eea.europa.eu/publications/COR0-landcover`.
14. LUCAS, for information see : `http://epp.eurostat.ec.europa.eu/portal/page/portal/lucas/introduction`.
15. JRC MARS AGRI4CAST Interpolated Meteorological Data, for information see : `http://mars.jrc.ec.europa.eu/mars/About-us/AGRI4CAST/Data-distribution/AGRI4CAST-Interpolated-Meteorological-Data`.
16. JRC EUSOILS, for information see : `http://eusoils.jrc.ec.europa.eu/`.

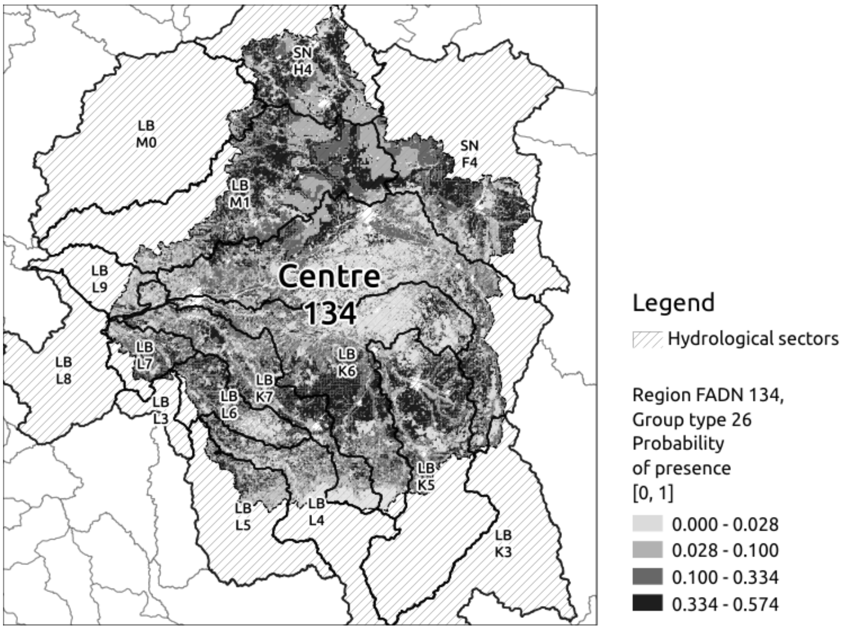


FIGURE 3.5 – The intersection between hydrological sectors, the FADN region 134 and the probabilities of presence for the farm group-type 26.

other physical data. Ultimately, probabilities of presence of GTs are defined for each cell (Cantelaube et al., 2012).

We focus our study at the level of the hydrological sectors which do not coincide with FADN regions. In order to represent the results at that scale, we use the grids obtained through the spatialization procedure described before and aggregate the data with respect to the sectors' borders and area. Similar approach is also employed by Martínez and Albiac (2006) who use remote-sensing and crop-field surveys for the spatialization of their model. The uncertainty of the localization of the group-types is a crucial point. As we are aiming at defining a sector-specific tax, all potentially present GTs in the hydrological sector are taken into account. Figure 3.5 represents the intersection between sectors and the French FADN region "Centre". The darker the filled polygons, the greater the probability of presence of a given GT in the polygon.

The modelling approach

Our modeling approach is constrained by the uncertainty of the localization of the economic agents. We are following the algorithm described below and summarized in Figure 3.6 :

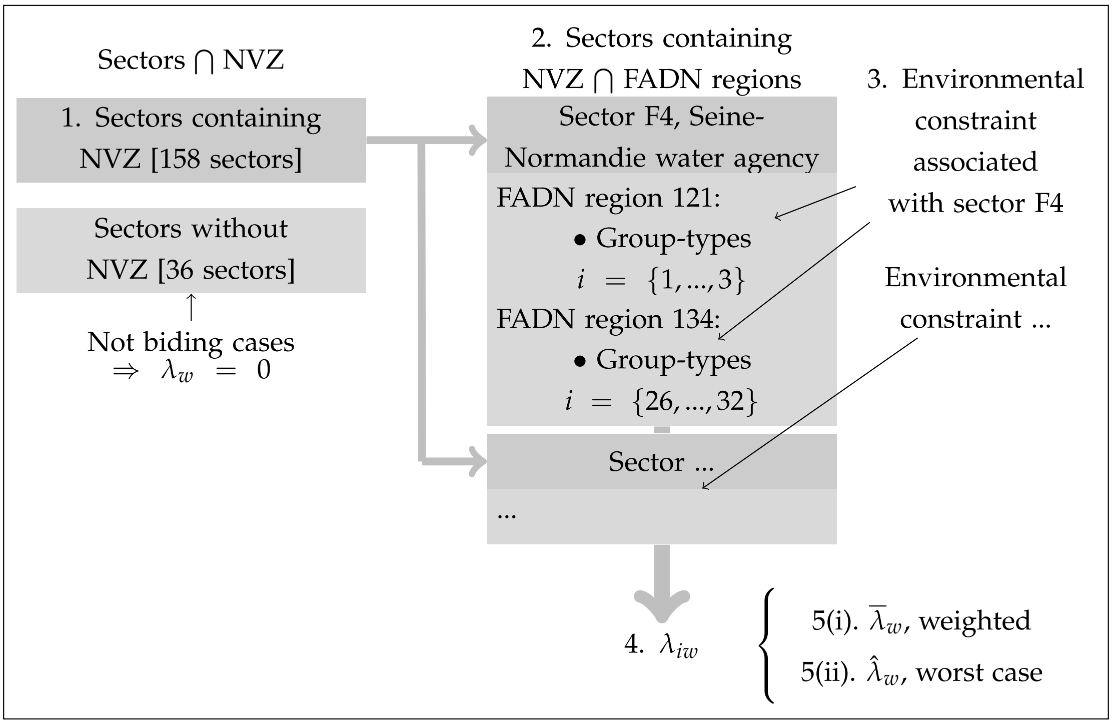


FIGURE 3.6 – The modelling approach employed for the estimation of the environmental constraint dual values. Example provided for sector F4 of the River basin district Seine-Normandie and FADN regions 121 ("Île-de-France"), 134 ("Centre") and the associated group-types in the AROPAj model.

1. Select the hydrological sectors containing NVZ (see paragraph "Identifying the binding case" above).

2. Select the FADN regions and the associated GTs intersecting with each of the previously selected sectors.

3. Introduce the environmental constraint as it is parametrized for the sector in the profit maximization problem of each GT localized within its borders. Thus, for each combination of a GT and a sector we perform a model simulation. As GTs do not interact in the model (each GT maximizes its gross margin individually) we cannot control for the total amount of fertilizers or livestock units. Hence, we impose the environmental constraint to each GT as if its fertilizer quantity and number of animals is the average in the sector. This assumption is introducing a bias into our results. Further discussion is provided in Subsection 3.4.

4. Solve the problem and define the dual value which, under this modeling pro-

cedure, is also indexed by the GT. For a given sector there are multiple values of λ_w depending on the number of GT involved.

5. We explore two definitions for the estimation of the dual λ_w : (i) an average of the obtained values, weighted by the probabilities of presence of the GTs ($\overline{\lambda}_w$) ; and (ii) a worst-case scenario where $\hat{\lambda}_w$ is defined as the greatest among the possible values. In order to evaluate the gains from the spatial differentiation by hydrological sector, we must also evaluate a scenario in which the greatest among the $\hat{\lambda}_w$ is assigned to the entire RBD. This scenario represents the case in which the differentiation can only be applied at the RBD scale.

We base the taxation schemes on the estimated dual values and compare the policies. The parameters of Equation (6) are estimated for the fertilizer use and the livestock units are both estimated for values per hectare of the geographical area of the hydrological sector. Without such an estimation, the model would not account for the size and the importance of the agricultural activity inside different sectors. For the sake of realism of the taxation schemes, we have decided to use average hectarage per sectors. Nevertheless, this assumption leads to a modification of the pollution fees' values. Farmers with greater per hectare load of animals and fertilizers are taxed less than they should be. The contrary holds as well for the farmers with more extensive type of production which are in consequence taxed more.

Results

In order to analyze the results of the different policy scenarios we present five indicators : farmers' gross margin, tax revenues, fertilizers use, nitrate emissions and the livestock units per hectare. We aggregate them at the level of the six RBDs in France. Figure 3.7 presents the evolution of the indicators under the different simulation scenarios. The parameters employed in each case are summarized in Table 3.1. The "No LU variation" is a scenario where the total number of livestock units is fixed while in the "No limit" case it can vary within a $\pm 15\%$ interval. The environmental constraint is only active in the "Optimal" scenario which provides us with the values of the λ_{iw} coefficients employed afterwards in the three "Lambda" scenarios. The geographical disparities observed for the nitrate concentrations clearly manifest themselves in the final results. The six RBDs are divided in two groups. Artois-Picardie, Seine-Normandie and Loire-Bretagne are the basins where greater nitrate loads are observed and where the simulated scenarios have more impact.

Name of the simulation	LU variation	Environmental constraint	λ_w
No LU variation	0%	No	-
No limit	15%	No	-
Optimal	15%	Yes	-
Lambda mean	15%	No	Average value ($\bar{\lambda}_w$)
Lambda worst	15%	No	Worst-case value ($\hat{\lambda}_w$)
Lambda RBD	15%	No	RBD worst-case of the $\hat{\lambda}_w$ of the associated sectors

TABLE 3.1 – Summary of the simulation scenarios' parameters.

For all RBDs the gross margin is reduced the most under the “Lambda RBD” scenario. The greatest reduction is observed for the Seine-Normandie (SN) where the gross margin falls by 24% in comparison to the “No limit” case. In Artois-Picardie (AP) the difference amounts up to 18%. The “Optimal” scenario is less expensive by 22% and 14% respectively. More important differences are also observed for the Loire-Bretagne (LB). Nevertheless, when the tax revenue is taken into account the differences are less pronounced. The other three RBDs, Adour-Garonne (AG), Rhin-Meuse (RM) and Rhône-Méditéranné-Corse (RMC), are subject to low anthropogenic pressure with a great part of their territory outside the NVZ and, thus, less concerned by the policies studied.

The “Lambda worst” and “Lambda RBD” lead to a reduction in the number of li-

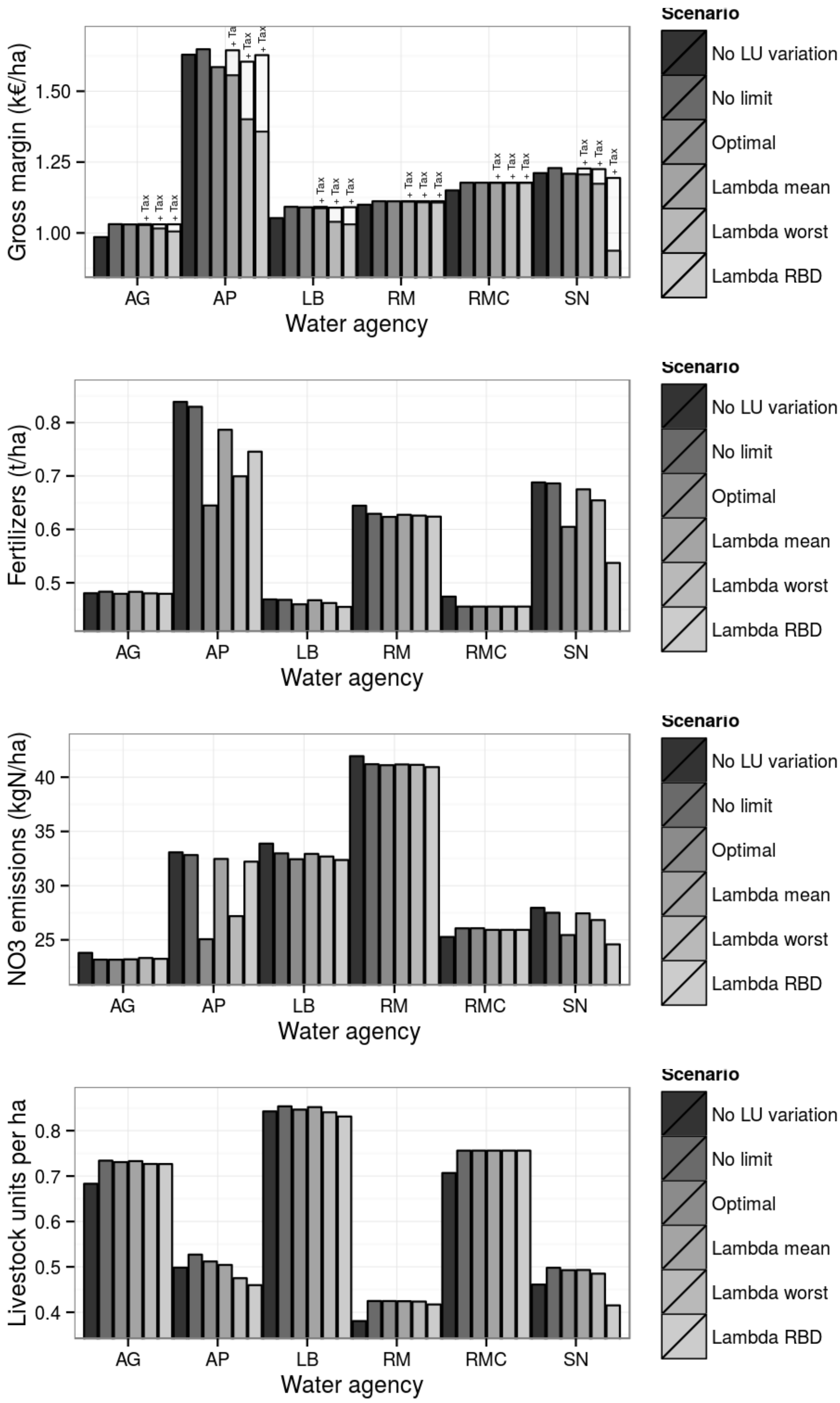


FIGURE 3.7 – Evolution of the indicators at the scale of the RBD under the different simulation scenarios. "No LU variation" stands for the case where the number of animals observed in the statistics is maintained unchanged. The "No limit" scenario is simulated for 15% livestock variation and no environmental constraint while for the "Optimal" case the nitrate concentration constraint is activated. The scenarios "Lambda mean" and "Lambda worst" result from the introduction of pollution fees on fertilizers and livestock. Results for "Gross margin" show also the tax revenue per hectare (white part of the barplots marked by "+ Tax").

vestock units, exceeding the one obtained in the "Optimal" scenario. On the other hand, the application of fertilizers remains greater than for "Optimal" except for Seine-Normandie in the "Lambda RBD" case. For instance, in Artois-Picardie the decline in animal population compensates for the less pronounced reduction in fertilizers use. Moreover, the overall nitrate losses at the root level are close to the levels seen in the benchmark scenario ("Optimal"). Nevertheless, the cost in terms of loss of gross margin is substantially higher. As Figure 3.8 shows, the decrease in fertilization in the "Optimal" case is associated with an increase in the fallow lands in the RBD which is not attained in the "Lambda worst" scenario. Similar results are obtained for the Seine-Normandie basin, but to a lesser extent.

The indicators for the three RBDs less concerned by the nitrate pollution (AG, RM and RMC, mentioned above) are evolving following the same pattern. For the scenarios where the livestock units are allowed to variate by 15%, their number increases. In RM and RMC the mineral fertilizers use is declining while in AG it remains stable. However, in RMC the nitrate emissions rise while in the other two RBDs fall. As the nitrate concentrations constraint is, for the most part, non-binding, the positive effect on the gross margin from the increased number of animals, is not canceled out by the simulated environmental policies. This is not the case for the northern and northwestern RBDs.

Figure 3.10a gives the resulting concentrations for the "No limit", "Optimal", "Lambda mean", "Lambda worst" and "Lambda RBD" scenarios at the scale of the hydrological sectors. As there is almost no difference between the simulations for the concentration values below 15 $\mathtt{mg/l}$ (except for the "Lambda RBD" scenario), the figure is focused on the higher nitrate loads. On the abscissa the sectors are given in ascending order according to the concentration levels for the "No limit" scenario. As Figure 3.7 shows, the two sectoral taxation schemes do not attain the same reduction levels in terms of fertilizer use and livestock units as in the "Optimal" case. This is also visible on Figure 3.10a when it comes to nitrate concentrations. On the contrary, the "Lambda RBD" leads to a greater nitrate reduction than the one for "Optimal". A more pronounced difference between the

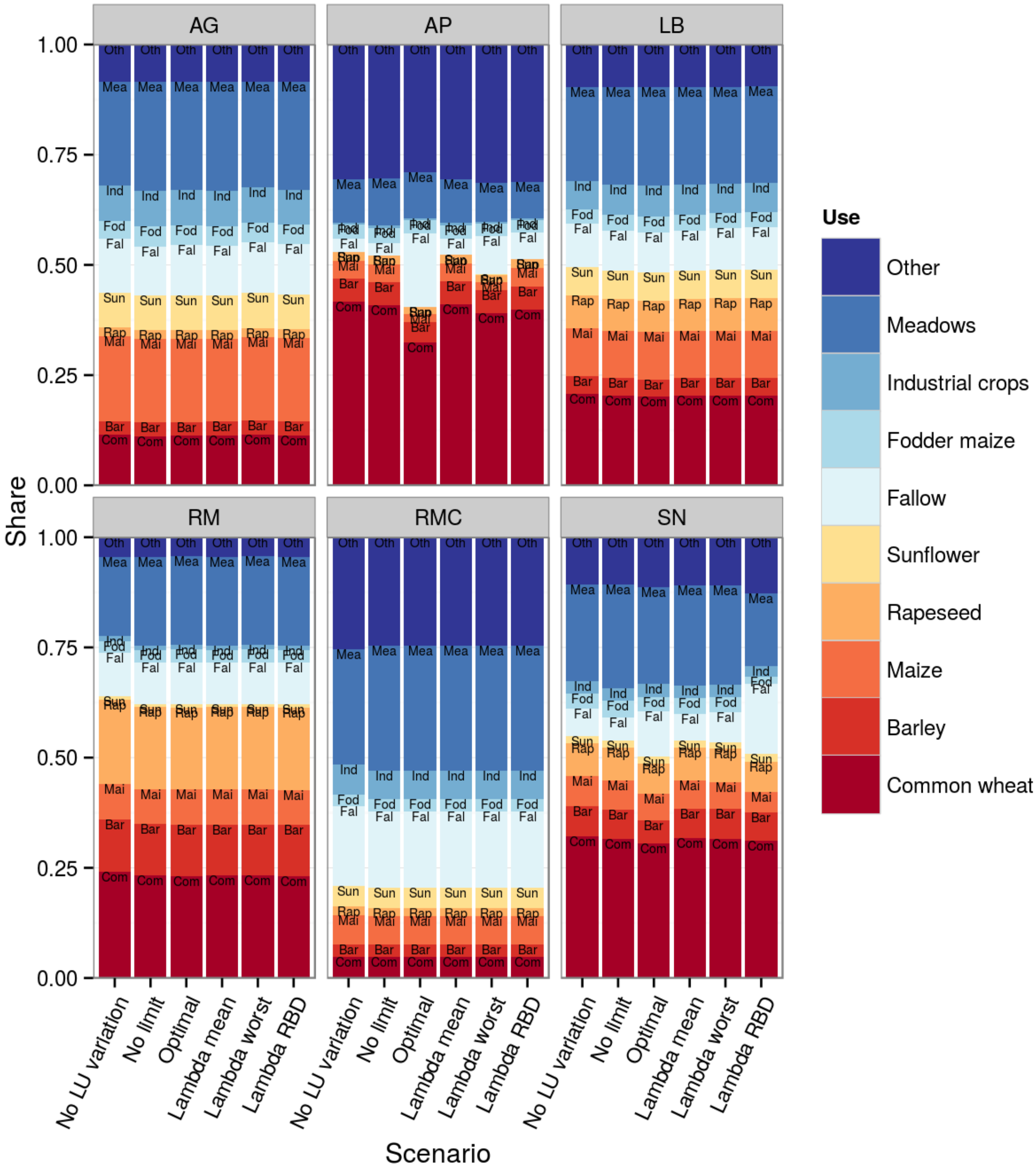


FIGURE 3.8 – Variation in the crop surfaces following the simulated scenarios. "No LU variation" stands for the case where the number of animals observed in the statistics is maintained unchanged. The "No limit" scenario is simulated for 15% livestock variation and no environmental constraint while for the "Optimal" case the nitrate concentration constraint is activated. The scenarios "Lambda mean" and "Lambda worst" result from the introduction of pollution fees on fertilizers and livestock.

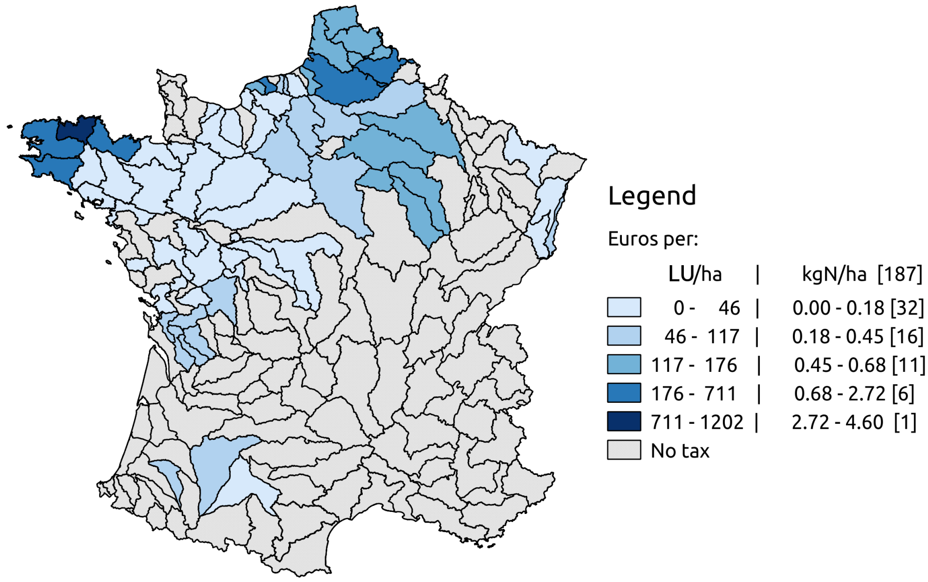


FIGURE 3.9 – Geographical distribution and rates for the "Lambda worst" scenario.

scenarios is visible for nitrate loads above 22 $\mathrm{mg/l}$, where for most of the group-types the environmental constraint is binding.

There are reductions in the concentration levels for values below 25 $\mathrm{mg/l}$ in each scenario studied. This is due to our modeling strategy in the “Optimal” scenario where every group-type is considered to be representative for the sector under simulation. Thus, farms with intensive crop or animal breeding activities are constrained by the environmental limitation. As a result, their production mix is modified, leading to a decrease in nitrate concentrations. In the “Lambda worst” scenario the dual values for these binding group-types are used in the tax definition and applied to all the other group-types associated with the hydrological sector. This effect is perceivable on Figure 3.10a as for some sectors the abatement in the “Lambda worst” scenario is greater than that of the “Optimal” scenario. Averaging the dual values yielded by the “Lambda mean” scenario attenuates the effects of the simulation bias.

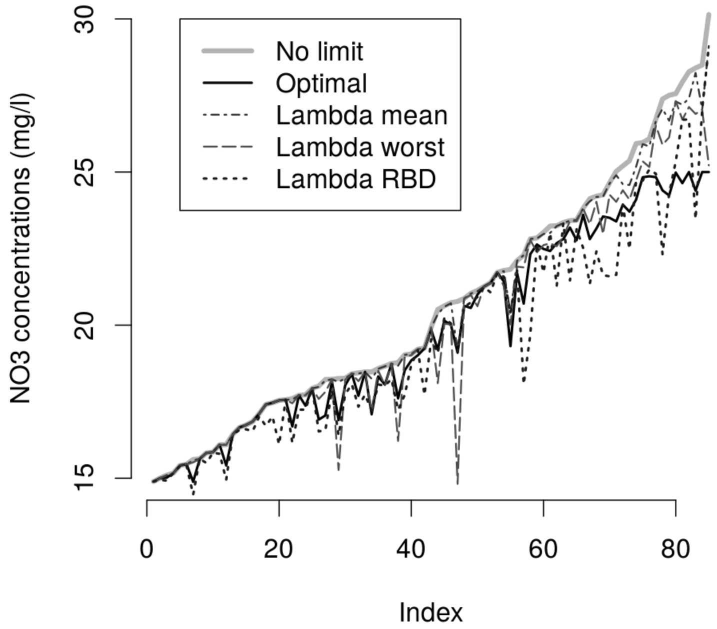
Furthermore, in the tax definitions, we use the average hectarage of the sector to estimate the fertilizer use and animal population per hectare. This, in turn, lowers the pressure on intensive farms and results in a reduction in the polluting activities which is smaller than that of the “Optimal” scenario. For this reason, post-taxation concentration levels under the differentiated schemes remain greater than those obtained in the optimal case, and in some cases, even higher than the assigned environmental limitation of 25 $\mathrm{mg/l}$.

The abatement curves presented in Figure 3.10b are obtained by smoothing. It should be noted that the different abatement levels are evaluated for different physical conditions (here given by the IDPR factor and the weight of agriculture in the sector in terms of geographical surface). The abatement cost for “Lambda mean” are close to those for “Lambda worst” for the low levels of concentration reduction. The maximum abatement engendered by the "Lambda mean" taxes is lower than 2 $\mathrm{mg/l}$. There are more points corresponding to “Lambda RBD” nitrate abatement for levels higher than 1 $\mathrm{mg/l}$ than there are for the “Lambda worst” case. The “Lambda RBD” tax is in general higher and applied to more sectors.

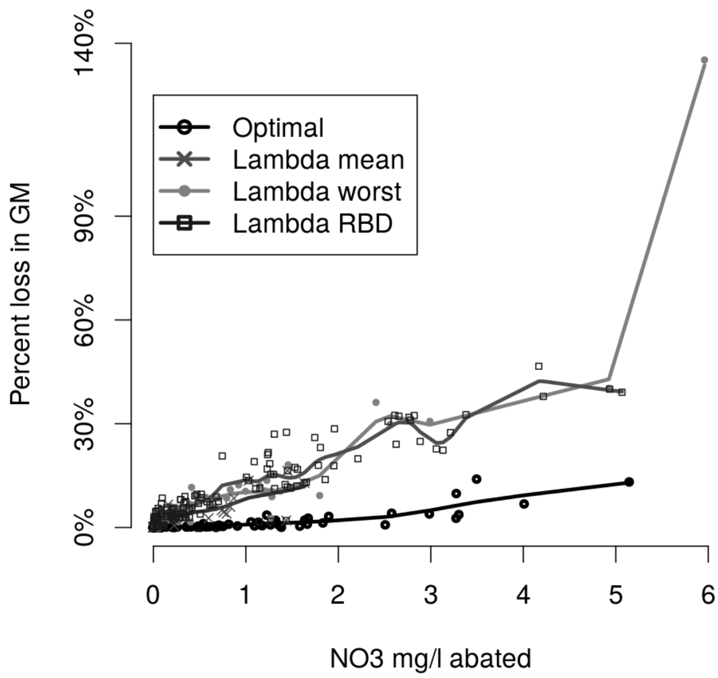
Input-based taxes have two effects on agricultural production. On one hand, they affect the quantity of mineral fertilizers used (intensive margin), and on the other hand, they can provoke a land use switch (extensive margin). Goetz et al. (2005) focus on the interaction of these two effects and propose a land use-specific tax that limits the introduction of crops that pollute more. However, the feasibility of such a policy is questionable.

This switching of crops can result in farmers switching to crops that are less efficient in nitrate fixation (more polluting). Such a paradoxical effect is reported in Jayet and Petsakos (2013). Our study is based on results at the hydrological sector level. Sectors cover smaller territory than FADN regions. Distinguishing the paradoxical effect at the sectors’ scale is easier because it is closer to the true scale of the switching process, namely the field level.

Table 3.2 summarizes the number and the extent of the paradoxical effects observed in the hydrological sectors for the different taxation policies. Approximately, one sixth of the sectors are affected. The percent increase in nitrate emis-



(a) Reference NO_3 concentrations (livestock variation and no environmental constraint) and the resulting concentrations from the application of the four scenarios, namely the optimal adjustment and the taxation with average, maximal and aggregated λ_w values. The x-axis is giving the index of the sectors in ascending order following the reference NO_3 concentrations.



(b) Nitrate mg/l abated and the corresponding abatement cost in the optimal case and the three pollution fees scenarios. The plot represents the smoothed curves for the estimated values (given by the points on plot).

FIGURE 3.10 – Concentrations post policy implementation and abatement costs.

sions is relatively small (below 3% of the values without taxation). Nevertheless, this bias of the second best taxation schemes is rather evident. AROPAj would allow us to simulate the effects of taxation schemes based on the estimated emissions. Under such framework, the paradoxical effect is eliminated.

		Lambda mean		Lambda worst		Lambda RBD	
RBD	All sectors	Number	Percentage	Number	Percentage	Number	Percentage
AG	48	9	0.47%	9	2.32%	19	0.9%
AP	7	5	1.9%	5	2.02%	3	2.58%
LB	48	6	0.22%	9	0.61%	9	0.72%
RM	18	2	0%	-	-	-	-
RMC	29	-	-	-	-	-	-
SN	34	9	0.6%	5	2.38%	4	2.53%
All	184	31	0.52%	28	1.39%	35	1%

TABLE 3.2 – Number of sectors and percent increase in nitrate emissions due to the paradoxical effect of the taxation policies. The paradoxical effect is defined as a case where the nitrate emission after taxation are supirior than for the "No limit" scenario while the nitrogen consumption and the livestock units are lower.

Results show that input-based tax policy should be accompanied by sector-specific measures relative to the agricultural production mix. The goal of these measures should be to limit the switch to more "polluting" crops or the promotion of environmentally friendly practices. Financing could be assured by the tax revenues or funded by the RBD agencies' budget. Such transfers are common to the water agencies as they are currently backing other environmental measures within their territory in accordance with the solidarity principle.

3.5 Conclusion

In the course of the present article we theoretically derived the level of an input-based, firm and water body-specific tax that allows for the attainment of a uniform environmental constraint. The expression is used in the case of nitrate pollution of water bodies originating from mineral fertilizer and animal manure use on field by French farmers. Since firm-specific taxation is in conflict with the country's current legislation on the fiscal equity principle, in our case study on tax differentiation, we use a water body-specific taxation rates for the two nitrogen sources. These specific rates depend on a simplified vulnerability index and the current anthropogenic pressure on the water quality. The environmental policy is introduced when the environmental constraint is binding, i.e. with the presence of NVZs defined in the EU Nitrate Directive (The Council of the European Communities, 1991). We use an agricultural supply model (AROPAj) coupled with a crop model (STICS) for our estimations. We compare three policy scenarios : a benchmark case where an environmental constraint is imposed on farmers, water body-specific taxes in hydrological sectors containing NVZ, and uniform rates per RBD in hydrological sectors containing NVZ.

Intensive animal breeding is an interesting case. Big pig farms (with more than 1000 animals) are found generally in the region of Brittany, part of the Loire-Bretagne RBD. The concentration of animals and the insufficient investment in manure processing facilities (anaerobic digesters) has led to the persistent nitrate problem in the region. Furthermore, the swine market is characterized by cyclic prices that variate in a large range. Since this kind of production is capital intensive leading to high fixed costs which represent a serious entry and exit barrier, farms oscillate between profits and losses from one year to another.

The introduction of water body-specific rates is associated with lower losses of gross margin for farmers and a better targeting of sectors concerned by high levels of nitrate concentrations. However, we observe large number of sectors where a paradoxical taxation effect occurs. These results can be alarming and are a testament to the need for a case-by-case approach. Such tailor-made policies could be relatively easily established in the current framework for RBD governance in France. French water agencies take management decisions within their RBD committees that have existed for more than 40 years now. In uniting all the stakeholders, including farmers, they are paragons of "water democracy" and participative decision-making. In such institutional context, a nitrogen fertilizer tax and a tax on livestock would be better accepted than if the same are imposed directly through national legislation.

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List of Acronyms :

EU European Union.

WFD Water Framework Directive.

NO_3 Nitrate.

N Nitrogen.

NPK Nitrogen (N), Phosphorus (P) and Potassium (K).

GIS Geographical Information Systems.

RBD River Basin Districts.

NVZ Nitrate Vulnerable Zone.

FADN Farm Accountancy Data Network.

AG Adour-Garonne.

AP Artois-Picardie.

LB Loire-Bretagne.

SN Seine-Normandie.

RM Rhin-Meuse.

RMC Rhône-Méditerranée et Corse.

SPS Single Payment Scheme.

FOC First Order Condition.

IDPR Network Persistence and Development Index.

BRGM Bureau de recherches géologiques et minières.

Troisième partie

Allocation des sols entre secteurs économiques

Le thème central de cette partie est l'allocation des sols entre secteurs économiques. Un premier cas d'étude est focalisé sur l'effet que les options de conversion d'usage ont sur les prix des terres agricoles. Un deuxième porte sur la comparaison des différentes mesures de la profitabilité de l'agriculture au sein des modèles économétriques d'usages des sols.

Dans l'étude des prix agricoles, la productivité marginale de la terre en usage agricole est approximée par la valeur duale de la terre estimée grâce au modèle AROPAj. Les résultats montrent que les deux sont positivement corrélés mais aussi que les prix de la terre tels qu'ils sont observés dans les statistiques du marché foncier reflètent aussi des primes liées à la conversion des terres vers d'autres usages. D'après ces conclusions, la valeur duale de la terre a été proposée comme approximation (proxy) de la rente agricole dans le deuxième cas d'étude. Les résultats des estimations étant en faveur de cette *proxy*, deux scénarios de politique publique visant les engrais azotés en climat futur (dans le cadre du changement climatique) et en climat présent sont présentés. Ces simulations mettent en lumière les perspectives que l'utilisation d'un modèle mathématique d'offre agricole ouvre lorsque les résultats de ce dernier sont intégrés dans un modèle économétrique d'usage des sols.

Chapitre 4

Prix des terres agricoles en France et le jeu de la concurrence entre les différents usages des sols

Ce chapitre est issu des travaux menés avec Pierre-Alain Jayet.

La théorie économique néoclassique suggère que la rente agricole est la productivité marginale de la terre. Le modèle AROPAj permet justement d'évaluer cette productivité marginale. En la confrontant aux prix de marché des terres agricoles, nous avons pu démontrer que les deux sont fortement corrélées mais que le prix de la terre n'est pas la simple somme des rentes agricoles futures actualisées. L'étude proposée cherche à distinguer l'influence de plusieurs options de conversion, notamment l'urbanisation (résidentielle ou récréationnelle) et la plantation de vignes. Il s'agit d'une décomposition hédonique des prix de la terre grâce à un modèle économétrique. Sa forme fonctionnelle est choisie suite aux estimations faites avec un modèle additif généralisé. Les résultats obtenus peuvent être valorisés dans le cadre d'autres analyses économiques basées sur les prix des terres agricoles.

Agricultural land prices in France: between pure agricultural productivity and conversion options

Abstract Theory and evidence show that agricultural land prices are influenced by factors other than the agricultural productivity and transportation costs as Ricardo and von Thünen suggested in the XIX century. Such factors are public policies and potential urbanization of land. In this study we extend the land development options by including vineyards and recreational activities (tourism). Results confirm the link between productivity and land prices but also the importance of the development options. Our findings could be used in public policy design and they have implications on economic analysis based on agricultural land prices.

4.1 Introduction

For more than three hundred years, agricultural land prices have been puzzling the scholars. Over the past few decades, research is generally focused on the effects of agricultural policy (following the work by Floyd, 1965) and the potential urban development (initiated by Capozza and Helsley, 1989). Nevertheless, land can be converted to other than the urban use. Thus, its price should be considered as the result of the capitalization of the gains from the present and the future land uses.

In this paper, we investigate the effects of several land uses on the agricultural land prices in France. Our model accounts for the current agricultural use, the potential urban and touristic development, and a switch to viticulture. The returns from agriculture are measured through land shadow prices obtained by a supply-side model accounting for public policy and heterogeneous farms. To the best of our knowledge, the use of land shadow prices in a hedonic land price model is an original contribution to the literature. In the course of the article we (i) present the theoretical background concerning land prices; (ii) propose a hedonic statistical model in support of the ideas presented; and (iii) make some conclusions on the possible implications of our findings in terms of public policy and future research.

4.2 Underlying theories

The remuneration of land as a factor of production (land rent) is one of the first subjects studied in economics theory before and after its emancipation as a separate discipline. There is a multiplicity of theories concerning land rent, the most influential amongst them being the work of Ricardo (1817) and von Thünen (1826). From the point of view of these two scholars, land rent originates from its scarcity and is differentiated with respect to its fertility (Ricardo) and the distance from the market (both Ricardo and von Thünen). Rents are established in comparison with marginal land where the average costs of production or provision are the highest.

Furthermore, land is considered as an asset on financial markets. Thus, property rights on land give access to certain revenues (Trivelli, 1997; Guigou, 1982a). It is generally assumed that land prices [1] reflect the Net Present Value (NPV) of future land rents. This straightforward link between observed prices and rents is, nevertheless, questionable (Clark et al., 1993; Gutierrez et al., 2007; Dupraz and Temesgen, 2012; Karlsson and Nilsson, 2013). Guiling et al. (2009) extend the NPV formula in order to include the possibility of land conversion to other uses.

Real options theory [2] and its applications to land pricing bring to light the relation between potential future land uses and current agricultural land prices (Capozza and Helsley, 1989; Plantinga et al., 2002). Land price is the NPV of future rents but these can arise not only from agricultural activity but also from other land uses, such as urban, to which land may be converted in the future. For two possible land uses, agricultural and urban, Capozza and Helsley (1989) propose a representation which defines agricultural land price (P^A) as a function of

1. We use agricultural land price (rent) and land price (rent) interchangeably.
2. Initially applied to land in the context of environmental protection and irreversibility by Arrow and Fisher (1974) and Henry (1974).

time (t) and space (z) in relation to a central business district (CBD). They summarize it in Equation (4.1).

$$P^A(t,z) = \int_t^{t^*} Ae^{-\delta(\tau-t)}d\tau + \int_{t^*}^{\infty} R(\tau,z)e^{-\delta(\tau-t)}d\tau - Ce^{-\delta(t^*-t)}, \qquad t \in [0,t^*]. \quad (4.1)$$

Agricultural rent is given by A, R is the rent from developing land for residential needs, C is the conversion cost, δ is the discount rate and t^* sets the moment when land is converted (irreversibly). Plantinga et al. (2002) base their research on a similar model and empirically show that option values of future land development are capitalized into agricultural land prices. Unlike previous studies where distance to cities has been associated only with a "von Thünen-type" rent, here the vicinity to cities is related to an option value for future development and its influence on agricultural land prices is explicit. Cavailhès and Wavresky (2003) show that farmland prices in immediate proximity to cities incorporate high premiums for urban conversion which fall sharply with the increase in distance.

Based on Equation (4.1), we consider land price as being composed of agricultural rent and option values for other land uses, where agricultural rent originates from a variety of factors. These include land productivity, capital invested (such as irrigation infrastructure, buildings), rent due to the proximity to markets (Cavailhès et al., 1996; Guigou, 1982a).

$$P^A = \underbrace{f(\text{productivity}) + k(\text{capital}) + m(\text{distance to markets})}_{\text{agricultural rent}} + \underbrace{\sum_{i=1}^{n} l_i(\text{land use}_i)}_{\text{option values}} \quad (4.2)$$

Functions f, k, l_i are supposed increasing and function m decreasing [3]. Each element in Equation (4.2) has its own temporal horizon. For this reason we assume that there are separate functions defining the way elements affect observed land prices. As there are many possible land uses other than the agricultural one, con-

3. For the sake of simplicity, we assume separability between these components.

version to which may occur at different moments in time, we suppose also that there are separate functions specific for each one of them. Agricultural rent depends on commodity prices established on international markets [4]. Furthermore, as Latruffe and Le Mouël (2009) summarize in their study, land prices are generally found to be "more responsive to government-based returns than to market-based returns". We address this specificity of the land market by using results from a supply-side model of French agriculture which integrates the European Union Common Agricultural Policy (CAP).

We introduce in the analysis the urban and touristic development options along with the one for planting vineyards. The division between urban and touristic development is imposed by the data available. The urban pressure is generally captured by the population density and thus the touristic inflow is not taken into account. The latter can be, however, important and so demanding in terms of space needed for accommodation and amenities provision. Dachary-Bernard et al. (2011) propose an original study demonstrating the effects of the proximity to the seacoast on agricultural land prices.

In our study we also include viticulture, distinguishing it from the other agricultural uses, namely cereals, oilseeds, protein and other crops, and meadows. French wine exports accounted for 7.83 billion euro in 2012 [5]. Thus, viticulture represents the third largest export sector of the country. The plantation of new vines is currently limited to the areas where geographical indicators (GI) are established. These GI are either the Protected designation of origin (PDO) or the Protected geographical indication (PGI) as they are defined by the European Union legislation. This regulation is, in fact, restraining the wine production from expanding into regions other than those where it already exists. As we shall see in the "Materials and methods" (Section 4.3), vineyards' prices are significantly higher than those for arable land and meadows. Thus, when a parcel of arable

4. This is true for France since 2006 when a reform in the Common Agricultural Policy took place.

5. FranceAgriMer, Le marché du vin en 2012, http://www.franceagrimer.fr/filiere-vin-et-cidriculture/Vin/La-filiere-en-bref/Le-marche-du-vin-en-2012

land is subject to GI attribution due to its location, its price would, supposedly, reflect it. Geniaux et al. (2011) estimate the anticipations for land conversion under land use regulation through zoning and confirm the anticipations' influence on land prices for the French Provence region. One of the indicators they use is the presence of geographical labeling for wines distinguishing four different appellations of origin. All four have a positive effect on parcels' prices, two of which being significant at the 1% confidence level.

4.3 Materials and methods

Agricultural land market and land prices. Land market in France is regulated by a special structure called SAFER (Société d'aménagement foncier et d'établissement rural). Notaries have to inform SAFER for all property transfers envisaged. The latter can then execute its pre-emption right and acquire the land. In 2012, for instance, SAFER bought almost 18% of the total agricultural land traded, i.e. some 90,000 hectares (7,000 ha through its pre-emption right [6]). In our study we use agricultural land prices (of arable land and meadows) communicated by the French Ministry of Agriculture [7]. They are averaged at the scale of (groups of) small agricultural region (SAR [8]). Land traded differs widely in its characteristics and Ministry statistics account for arable land and meadows alike (vineyards and orchards are not taken into account). In 2012, arable land price on average was 6,560 euro/ha (cereals), meadows are traded for an average of 4,220 euro/ha in regions with predominant animal husbandry, while a hectare of vineyards under PDO was traded for 131,700 euro (SAFER, 2013b). Special attention is paid to the selection of property right transfers taken into account by Ministry statistics. For instance, transactions of less than 70 ares or where there are buildings present are not considered in the statistics. Only parcels with a certain agricultural vocation are included. The number of transactions per year is

6. SAFER, `http://www.safer.fr/missions-safer.asp`.
7. Statistical service of the Ministry, Agreste, `http://agreste.agriculture.gouv.fr/`.
8. SAR's area is varying from some 1,000 ha to more than 400,000 ha.

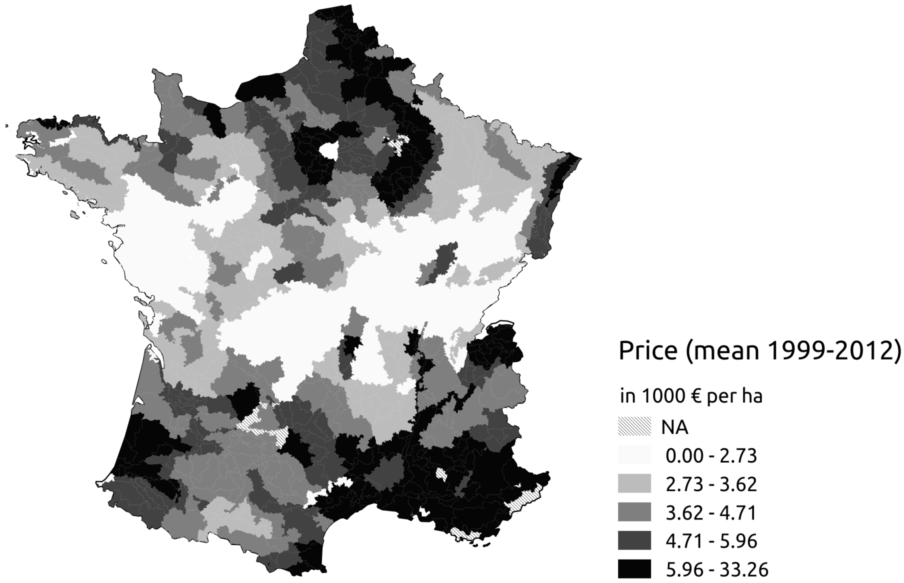
restraint and the characteristics of the land lots exchanged can vary from year to year. So, in order to have a better representation of land prices, we use the values for a 14 years period (from 1999 until 2012), expressed in 1999 euros [9], and calculate an average price. Figure 4.1a displays the territorial distribution of land prices.

Agricultural data. We use data from the general agricultural census conducted in 2010 [10] and provided by the statistical service of the French Ministry of agriculture. Data concerns the vineyards and the utilized agricultural area (UAA, all in hectares). The information is available at the French canton scale (there are approximately 4,000 cantons in France) and later aggregated at the (groups of) SAR level in order to ensure consistency with the data on land prices. A map of the share of GI vineyards in the total UAA is given in Figure 4.1b.

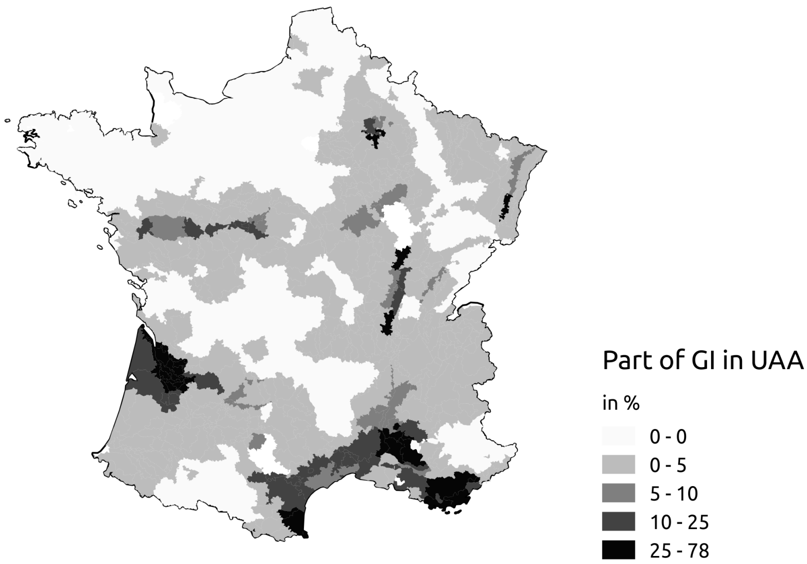
Demography and tourism. The data on population and revenues is provided by the French national statistical institute (INSEE) at the scale of the French commune. Density is then calculated and aggregated at the SAR level. The revenues across communes in the same (group of) SAR are averaged and expressed in euros per capita. INSEE provides us also with information concerning the domestic tourism consumption per administrative region (in millions of euro) and the number of vacation properties. This information is combined with the tourism density (number of beds per km^2) obtained from the French Ministry of Sustainable Development [11].All data transformations and aggregations are done using the R and the QGIS softwares.

Agricultural land rent. In order to identify the part of the agricultural returns capitalized in land prices, we use the land shadow prices calculated by the supply-side agricultural model AROPAj. Returns from agriculture in France depend not only on land quality and productivity. They depend as well on the attribution of different agricultural payments and production quotas defined in the Common

9. Inflation estimates for the period coming from the World Bank, `http://data.worldbank.org/indicator/NY.GDP.DEFL.KD.ZG`.
10. Agreste, `http://agreste.agriculture.gouv.fr/recensement-agricole-2010/`.
11. Data available at `http://geoidd.developpement-durable.gouv.fr/`



(a) Land prices per hectare at the level of (group of) SAR.



(b) Share of GI vineyards in total UAA at the level of (group of) SAR.

Figure 4.1 – (**a**) Land prices per hectare (mean value) and (**b**) share of GI vineyards.

Agricultural Policy (CAP) of the European Union. The model (for detailed description is provided by Jayet et al., 2015) focuses on the supply-side and is based on linear programming. It covers the European Union agriculture and takes into account the CAP. AROPAj agents are representative for about 85% of the UAA in France, 18 crops and 5 animal species. Our economic model is partly coupled with the crop model STICS (Godard et al., 2008) and is calibrated with the data from the Farm Accountancy Data Network (FADN). Economic agents in the model are groups of farms (group-types), each one maximizing its gross margin. There are 157 group-types in France and they are representative at the FADN region level. Their exact geographical location is unknown [12]. Results from AROPAj are disaggregated (spatialized) at a 100 x 100 m scale grid and the probabilities of presence of the groups of farms in each grid cell are calculated through the method developed by Chakir (2009) and Cantelaube et al. (2012). Then, results are summarized at the (group of) SAR level and the average values per hectare are obtained.

An essential element of AROPAj is the optimal allocation of land between different agricultural activities. This is translated by a total surface constraint associated with a shadow price (dual value), the groupe-type's UAA being considered as a quasi-fix factor of production. This shadow price reflects the opportunity cost of the most profitable land use. We use the land shadow prices estimated by AROPAj and calibrated to FADN data for 2002. Their spatial distribution is represented on Figure 4.2. For perfectly competitive markets, land rent and dual value should coincide. This is not true in the case of France where the rental prices are fixed in an interval defined by the administrative authorities [13]. Shadow prices can be estimated econometrically (Dupraz and Temesgen, 2012) or derived through the maximization of farmers' profit function as it is done in the AROPAj model. Dupraz and Temesgen (2012) report a mean shadow price

12. Privacy policy of the FADN.
13. French Rural Code, Article L411-11. In reality, in some parts of the country new tenants are often constrained to make additional payments under-the-counter to the former tenants or the landlords in order to obtain the lease or the property.

of 550 euros/ha/year which is a result close to the one obtained by AROPAj of about 570 euros/ha/year. The latter figure corresponds to a partially decoupled CAP policy following the Luxembourg reform of 2003. For both models data is provided by the FADN.

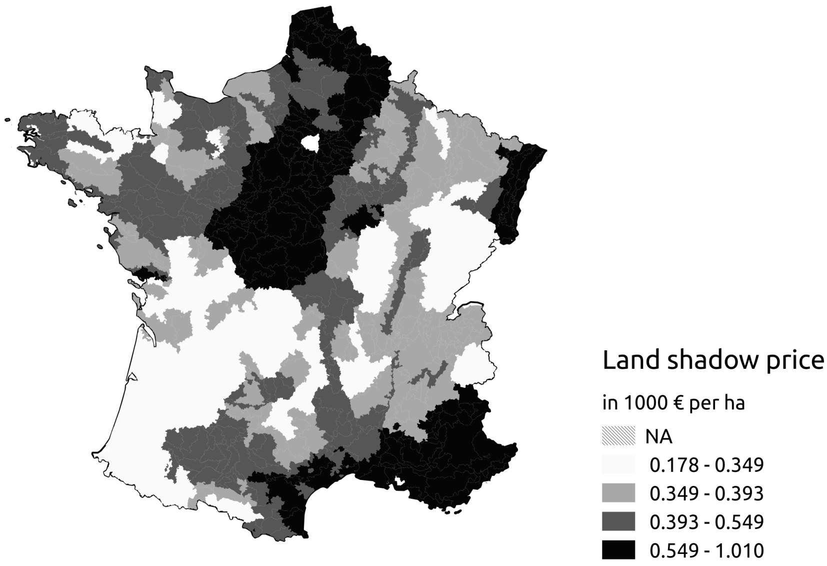


Figure 4.2 – Land shadow price estimated by the AROPAj supply-side model (mean values per ha at the scale of (group of) SAR.

An important limitation of FADN data is the lack of labour costs. Thus, the remuneration of labour is redistributed to the other bounded factors of production that are taken into account in the objective function (duality theorem), in our case these are land, irrigation, CAP quotas and other (e.g. livestock, supposed to be quasi-fixed on the short term). By using the dual values calculated by AROPAj, we assume that labour costs per heactare are uniform between different agricultural activities.

4.4 Results

Using the data described in Section 4.3, we test the influence of the three options for land conversion, namely urban, touristic and vines. To do so, we use the hedonic model of the land prices presented in Section 4.2.

Urban rent. We approximate the rent for urban development by multiplying the population density with the mean average revenue. In order to refer this rent to the agricultural land, we divide by the UAA in the (group of) SAR instead of the total surface. This procedure is applied to the other proxies (for tourism and for vines) as well.

Tourism. The three major touristic regions in France are the Parisian region (Île-de-France), the Riviera region (Provence-Alpes-Côte d'Azur) and the Rhône-Alpes region with its summit, the Mont Blanc. The inflow of tourists is not captured by population density. Nevertheless, it plays an important role in the local economy. To evaluate its influence we propose the composed indicator presented in Equation 4.3.

$$\text{Tourism rent} = \left(\begin{array}{c}\text{Vacation}\\ \text{proper-}\\ \text{ties}\end{array} + \begin{array}{c}\text{Number}\\ \text{of beds}\end{array} \right) * \frac{\begin{array}{c}\text{Domestic touristic consumption}\\ \text{in the adm. region}\end{array}}{\text{UAA}} \tag{4.3}$$

Vineyards. The variable we use to approximate the possibility to convert arable land or meadows into vines is the share of vineyards under GI in the total UAA in the (group of) SAR. It would have been preferable to distinguish the rent from GI and that for viticulture in general, but in reality the share of GI vineyards in the UAA and the share of viticulture in the UAA are highly correlated (the value of the Pearson r coefficient is of 0.91). Furthermore, as mentioned before, the potential vine plantation is associated with the establishment of a GI.

The first assessment of the impact of these variables that we present is based on a Generalized Additive Model specification (GAM, Hastie and Tibshirani, 1990; Wood, 2006). This is a type of generalized linear model. The dependent variable (here land prices) is explained by the sum of unknown smooth functions of the independent variables. The GAM model is aiming at estimating these smooth functions. Hence, it provides us with an insight on the functional forms under which the variables influence land prices. Formally the model is given in

Equation 4.4 where we use a link function (g, a logarithmic specification gives a better fit for the data) and assume a Gaussian distribution of land prices. This link function is relating the expected value of land prices ($\mathbb{E}(P^A)$) and the smooth functions (here f_i, where $i = 1, 2, 3, 4$) of the different explanatory variables along with a constant term (β_0). One of the inconveniences of the GAM is that the results are difficult to interpret, e.g. there is no single coefficient per variable. This is why such models are often represented by graphics that depict the estimated functions. The results obtained are given in Figure 4.3 while the approximate significance of the estimated functions is provided in Table 4.1. The model explains about 76.3% of the total deviance. All terms are reported significant.

$$g(\mathbb{E}(P^A)) = \beta_0 + f_1(\text{Shadow price}) + f_2(\text{Urban rent}) + f_3(\text{Tourism rent}) + f_4(\text{Vineyards}) \tag{4.4}$$

As theory suggests, the dual value of land explains well land prices. Prices increase as the dual values increase. There is a slight decrease in the interval between 600 and 800 euro per hectare and a steep rise afterwards. Values in that interval (600 – 800 euro) are mainly observed in the region Centre, where agriculture is dominated by large cereal farms (more than 55% of the UAA is concentrated in farms of more than 100 hectares [14]). Thus, the lower per hectare price can be explained by the fact that in this region land is traded by big surface lots diminishing the per hectare price. Eliminating observations from this region results in an almost linearly increasing function between land price and dual value.

Vineyards have a positive influence on land prices for GI vines' share lower than 10%. The effect stabilizes for greater values. For the urban rent, the estimated function has a parabolic form. We should mention that the second half of the parabola is evaluated on a low number of observations resulting in a large confidence interval. We address this issue in Subsection 4.4. The tourism func-

14. http://agreste.agriculture.gouv.fr/recensement-agricole-2010/

tion is estimated as an increasing linear one after an abrupt drop at lower values. Equation 4.5 is based on the functional forms estimated in the GAM model (the logarithmic transformation for the urban and vineyards' rents) and uses the logarithm of the land price as dependent variables. We evaluate it with the standard Ordinary least squares (OLS) technique. Results from the estimation are presented in Table 4.2.

$$\log(P^A) = \beta_0 + \beta_1 * (\text{Shadow price}) + \beta_2 * \log(\text{Urban rent}) + \\ + \beta_3 * (\text{Tourism rent}) + \beta_4 * \log(\text{Vineyards}) \tag{4.5}$$

All coefficients are significant at the 1% confidence level. The residuals satisfy the normal distribution hypothesis (the Jarque Bera test reports a p-value of 98%). Because of the cross-sectional character of our data, we run the Breusch-Pagan and the White tests for homoscedasticity. The p-values reported by the two tests are 0.64 and 0.88 respectively, which means that we cannot reject the homoscedasticity hypothesis. We can, thus, keep to our OLS estimates. Figure 4.4 represents the Pearson r coefficients of correlation between the model variables. Although positive correlations between variables are reported, the values are lower than 70%. We, thus, consider the estimated model coefficients reliable.

Sensitivity analysis

Estimated coefficients remain robust (significance, sign and to some extent values) when explanatory variables are dropped one by one. As we mentioned above and as Figure 4.3 shows, the estimated functions for the Urban and Tourism rent are based on too few observations for the upper values. After we eliminate the maximal values for these two rents, we reestablished the GAM functions (three observations are dropped out of the 350 initially available). Newly estimated functions resemble those on Figure 4.3, except that the Urban function is strictly increasing. The Tourism function keeps the stark switch in its functional form at lower values. The scale of the effect for Tourism is significantly

Parametric coefficients:					
	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	8.23040	0.02	414.8	<2e-16	***
Approximate significance of smooth terms:					
		edf	Ref.df	p-value	
Shadow price		6.947	7.682	<2e-16	***
Urban		2.992	3.284	4.31e-11	***
Tourism		6.591	7.128	1.18e-15	***
Vineyards		4.594	4.912	1.55e-4	***
Significance codes: 0 ’***’ 0.001 ’**’ 0.01 ’*’ 0.05 ’.’ 0.1 ’ ’ 1					
Adjusted R^2 = 0.761, Deviance explained = 77.5%, n = 350					

Table 4.1 – Intercept and approximate significance of smooth terms (Std. Error – standard error, edf – effective degrees of freedom, Ref.df – degrees of freedom for reference distributions).

Parametric coefficients:					
	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	6.588	0.18	37.183	<2e-16	***
Shadow price	0.9925	0.092	10.765	<2e-16	***
log(Urban)	0.1109	0.019	5.923	7.65e-09	***
Tourism	5.076e-07	1.363e-07	3.723	0.00023	***
log(Vineyards)	0.5921	0.2237	2.646	0.00851	**
Significance codes: 0 ’***’ 0.001 ’**’ 0.01 ’*’ 0.05 ’.’ 0.1 ’ ’ 1					
Multiple R^2 = 0.4535, Adjusted R^2 = 0.4471					
F-statistic: 71.56 on 4 and 345 DF, p-value: < 2.2e-16					

Table 4.2 – Parametric coefficients and significance (Std. Error – standard error, DF – degrees of freedom).

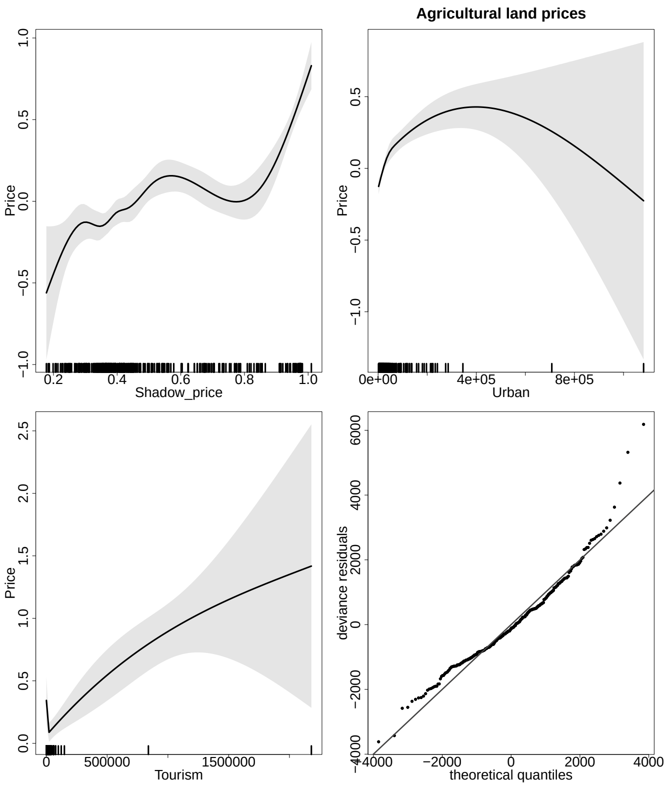


Figure 4.3 – GAM smooth functions and residuals quantile-quantile plot. The observations' distribution is presented on the horizontal axis. On the vertical axis are land prices (0 equals the mean value). Plots per variable are done by assuming all other variables at their mean values. The shaded area represents the confidence intervals.

	price	Shadow_price	Urban	Vineyards
Shadow_price	0.53 (0.45,0.60)			
Urban	0.54 (0.46,0.61)	0.22 (0.12,0.32)		
Vineyards	0.34 (0.25,0.43)	0.16 (0.05,0.26)	0.15 (0.05,0.25)	
Tourism	0.61 (0.54,0.67)	0.18 (0.07,0.28)	0.69 (0.64,0.74)	0.17 (0.07,0.27)

Figure 4.4 – Pearson r correlation coefficients for the regression variables. Between brackets is given the confidence interval at 95% level.

modified as the comparison between Figure 4.3 and 4.5 shows. This is due to the fact that the highest value for the tourism rent is 130 times bigger than the mean one. The SAR where the maximal value is observed is on the Riviera coast and has a great number of vacation properties (in the top 5% of the distribution) and an above average domestic touristic consumption. However, the importance of the touristic rent in this SAR is mostly due to the low denominator as the UAA is the lowest in the sample (some 2000 ha). The results of the OLS model remain robust for the Shadow Price and the Urban rents. Tourism's coefficient increases tenfold (from $5.076*10^{-7}$ to $3.314*10^{-6}$). The overall explicative power is lowered (the R^2 drops from 45% down to 40%) and the significance terms of the Vineyards and the Tourism coefficients deteriorate (from 0.00851 and 0.00023 to 0.04021 and 0.00696 respectively).

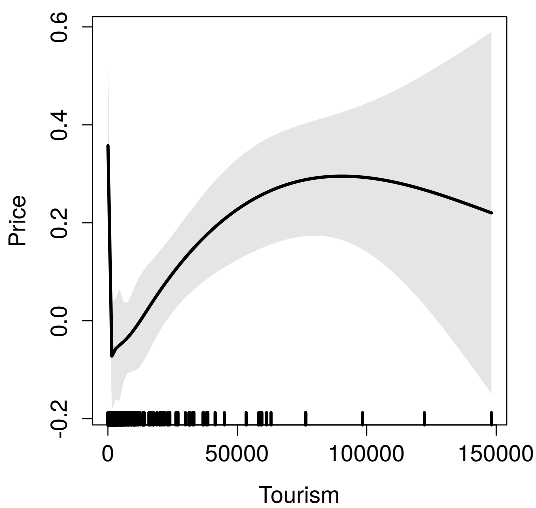


Figure 4.5 – GAM smooth function for Tourism rent with maximal values for Urban and Tourism eliminated (three observations).

4.5 Conclusion and perspective

Land is a production factor not only for agriculture. As such, its price reflects the competition between different activities. Land rent theory puts agricultural land prices in relation to other land uses along with the pure agricultural land productivity. In order to demonstrate this, we propose a hedonic model decomposing French land prices into the different economic rents that define it. We capture the agricultural rent by using the land shadow price evaluated by a supply-side model which accounts for the CAP of European Union. We confirmed the influence of potential land conversions on land prices. The options we identified are the urban and touristic development along with the plantation of vineyards.

What implications do these results have? In 2014, France and some other wine producing countries succeeded in gaining cause to the European Union commission and, after months of struggle, the liberalization of the vine plantation was abandoned. The regime change proposed in the new CAP never took place and an alternative scheme is now under consideration. One of the arguments for the maintain of the production control was that vineyards currently occupy marginal lands. The potential liberalization, thus, could have resulted in the loss of arable lands due to vines' plantation because of their higher productivity. Our study shows that land prices are influenced positively by the possibility to convert to vineyards (related to the presence of GI). So, the replacement of arable land by vines is a real prospect.

From methodological point of view, the present results confirm the capacity of economic models to evaluate the productivity of agricultural land. The advantage of such models is that the public policies could be easily translated into parameters or constrains of the maximization program. Thus, the effects of different regulations on agricultural activity could be assessed through their direct consequences on land returns. Furthermore, the land markets are not always well developed or land transactions could be rare. The shadow price can, in such cases, be used to obtain or refine estimations of land prices.

Another application of these results is in the sense of the Ricardian approach to climate change impacts assessment. This method, proposed initially by Mendelsohn et al. (1994), consists in explaining land prices by the present climate and evaluating the future prices following climate projections. One of its major hypothesis is the perfect mobility of the economic activities along with the well functioning land markets. As we showed in our analysis, land prices are not only influenced by the pure agricultural productivity but also by other factors some of which fixed in space (like the Riviera coast or the Alpes, for instance, as well as the GI wines). Thus, a rigorous assessment of climate change impacts based on this method should take into account these factors and capture their part in the land prices in order to prevent biases. For instance, Schlenker et al. (2005) in their analysis account for the effects of urban pressure on agricultural land prices following the foundings in Plantinga et al. (2002).

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Chapitre 5

La rente agricole dans les modèles économétriques d'allocation des terres : comparaison entre les proxies les plus fréquemment utilisées

Ce chapitre est issu d'un travail réalisé avec Raja Chakir.

Les modèles économétriques d'allocation des sols ont tous comme hypothèse de base la maximisation des rentes relatives à chaque usage étudié. Or les valeurs de ces rentes sont rarement accessibles dans les données statistiques. De fait, les chercheurs sont obligés d'avoir recours à des valeurs dites de *proxy* ou à des approximations des rentes. Dans le cas de l'usage agricole, le plus couramment ces *proxies* sont les revenus agricoles ou les prix de la terre. Dans ce chapitre, ces deux variables sont comparées au prix implicite de la terre estimé par le modèle AROPAj. Cela permet de valider la pertinence de ce dernier indicateur de la rente agricole dans le contexte des études sur l'usage des sols. Le modèle AROPAj permet aussi d'évaluer les effets des politiques publiques et du changement climatique sur l'agriculture.

Agricultural rent in land use models: Comparison of frequently used proxies

Abstract The objective of this paper is to compare land use models based on three different proxies for agricultural land rent: farmers' revenues, land prices, and shadow land prices derived from a mathematical programming model. We estimate a land use share model for France at the scale of a homogeneous (8 km x 8 km) grid. We consider five land use classes: (1) agriculture, (2) pasture, (3) forest, (4) urban, and (5) other. We investigate what determines the alternative land use shares using economic, physical, and demographic explanatory variables. The data on land use are from Corine Land Cover, the remote sensing database. We model spatial autocorrelation between grid cells, and compare the prediction accuracy as well as the estimated elasticities between different model specifications. Our results show that the three rent proxies are similar for prediction quality of different models. They show also that including spatial autocorrelation in land use models improves the quality of prediction (RMSE - root mean square error indicators). One of our econometric land use models is used to simulate the effects of a nitrogen tax, and show land use change (LUC) projections for France under two IPCC (Intergovernmental Panel on Climate Change) climate scenarios.

5.1 Introduction

Land use and LUC are the main human derived pressures on the environment (Foley and al., 2005). Some LUCs such as deforestation and overturning of permanent pasture, can have adverse effects on the environment such as reduced biodiversity (Sala and al., 2000), carbon release into the atmosphere (Rhemtulla et al., 2009), changes to water cycles (Stevenson and Sabater, 2010) and loss of ecosystem services (Schroter and al., 2005). Other LUCs such as the establishment of permanent grassland or afforestation, can store carbon in the soil, and thus contribute to the reduction of greenhouse gas (GHG) emissions and preservation of the environment.

The empirical economic literature on land use has increased significantly. Although individual studies have particular objectives and exploit particular data sets and estimation methods, all this work is based on a common economic theory of land use which assumes profit maximization by landowners. The optimal land use is determined by comparing the rents associated with each possible use. According to this theory, these rents will vary depending on characteristics such as the fertility (Ricardo, 1817) and location (von Thünen, 1826) of the land. However, other factors can affect the land use decision for a given plot. These include socio-economic factors such as production prices, and policy variables such as taxes or subsidies. Econometric studies of land use generally examine the relationship between land use choices and a set of explanatory variables, namely the rents derived from different land uses, or proxies such as input and output prices, subsidies, and soil and climatic variables (slope, altitude, soil quality, temperature, precipitation, etc.). Land rent is a rather complex notion and there are several concepts of economic rent advanced in the literature [1].

Since land use rents usually are not directly observed, most studies are approximating them by other variables. These proxies for land use rents vary from one study to another but the most frequent proxies for agriculture and forestry are producer's revenue, the agricultural land price, output or input prices, yields, land quality, and government payments (*e.g.* Wu and Segerson, 1995, Plantinga, 1996, Stavins and Jaffe, 1990, Plantinga and Ahn, 2002). The objective of the present paper is to compare land use models based on three different proxies for the land rent from agriculture - (i) the farmer's revenue, (ii) the land price, and (iii) the shadow land price derived from a mathematical programming model (AROPAj). The farmer's revenue is the most commonly used in the literature. Data on returns from agriculture is often directly observed, or can be derived from agricultural censuses or surveys (Stavins and Jaffe, 1990; Plantinga and Ahn, 2002; Lubowski et al., 2008; Chakir and Le Gallo, 2013). The second proxy,

1. See Randall and Castle (1985) for a detailed presentation on the concept of land rent.

land price, is generally assumed to be the net present value (NPV) of future land rents (Ricardo, 1817). However, the literature shows that the standard NPV formula ignores the possibility of agricultural land being converted to other uses (Clark et al., 1993; Gutierrez et al., 2007; Karlsson and Nilsson, 2013). Ay et al. (2014a) use agricultural land price [2] to approximate land rents in the econometric model and to study the impacts of climate change on land use and common birds in France. Our third proxy, land shadow price, to the best of our knowledge, has not been used in econometric land use shares models so far. The shadow price corresponds to the marginal productivity of the land estimated using a European Union mathematical programming model for agriculture. The economic supply-side model AROPAj (for detailed description see Jayet et al., 2015) is based on the Farm Accountancy Data Network (FADN).

In order to compare the impacts of different agricultural rent proxies, we estimate land use share models at the resolution of a homogeneous 8km x 8km cell grid covering the territory of metropolitan France. We consider five land use classes: agriculture, pasture, forestry, urban, and other. Data on land use are derived from the remote sensing database, Corine Land Cover (CLC [3]). We model the spatial correlations between land uses in neighboring grid cells. Most studies in the literature assume spatial independence of land use choices. Some recent exceptions include Ay et al. (2014b); Chakir and Le Gallo (2013); Li et al. (2013); Sidharthan and Bhat (2012); Ferdous and Bhat (2012); Chakir and Parent (2009). Incorporating spatial autocorrelation into land use models improves their prediction accuracy but could raise several issues related to econometric estimation, hypothesis testing, and prediction (Anselin, 2007; Brady and Irwin, 2011).

2. Also known as Ricardian approach, following (Mendelsohn et al., 1994).
3. For more information: `http://land.copernicus.eu/pan-european/corine-land-cover`.

5.2 The Model

Land use share model

In line with the literature on LUCs, we estimate a land use share model. Such models have been widely employed in the literature (Lichtenberg, 1989; Stavins and Jaffe, 1990; Wu and Segerson, 1995; Plantinga, 1996; Miller and Plantinga, 1999). The first step in the modeling procedure assumes that the landowner derives the optimal land allocation from his/her profit-maximization problem. In this paper we focus on the landowner's decision to allocate land among five possible uses: agriculture, pasture, forest, urban and other. As in Plantinga (1996) and Stavins and Jaffe (1990) landowners allocate land to the use providing the greatest net present value of the profits. In the second step, and following the literature, we aggregate the optimal allocations by individual landowners to derive the observed share of land in the grid cell i in use k, denoted y_{ki}.

In this paper we use grid-level data, where shares are defined as the percentage of total grid area devoted to given uses. The observed share of land use k ($k = 1, ..., K$) in grid cell i ($i = 1, ..., I$) is expressed as:

$$y_{ki} = p_{ki} + \varepsilon_{ki} \quad \forall i = 1, \ldots, I, \quad \forall, k = 1, \ldots, K, \tag{5.1}$$

where p_{ki} is the expected share of land allocated to use k in grid cell i. The observed land allocation y_{ki} may differ from the optimal allocation due to random factors such as bad weather or unanticipated price changes. These random events are assumed to have a zero mean.

As in Wu and Segerson (1995) and Plantinga et al. (1999), we assume a logistic [4] specification for the share functions as follows:

4. The logistic share models are mainly used for three reasons: (i) they ensure that the predicted share functions (strictly) lie in the interior of the zero-one interval, (ii) they are parsimonious in parameters and (iii) they are empirically tractable thanks to the so-called log-linear transformation.

$$p_{ki} = \frac{e^{\beta_k' X_i}}{\sum_{j=1}^{K} e^{\beta_j' X_i}} \tag{5.2}$$

where X_i are explanatory variables and β_k' measure the effect of the explanatory variables on the expected shares.

Following Zellner and Lee (1965), the natural logarithm of each observed share normalized on a common share (here y_{Ki}) is approximately equal to:

$$\widetilde{y}_{ki} = \ln(y_{ki}/y_{Ki}) = \beta_k' X_i + u_{ki} \text{ for } \forall i = 1, \ldots, I, \quad \forall, k = 1, \ldots, K, \tag{5.3}$$

where u_{ki} is the transformed error term. The model in 5.3 is identified if we constrain $\beta_K = 0$.

Spatial autocorrelation

In the context of aggregated land use share models, spatial autocorrelation could result from a structural spatial relationship among the values of the dependent variable, or a spatial autocorrelation among the error terms. The former is viewed as a fundamental characteristic of spatial processes which are characterized by potentially complex interactions, and dependent structures among neighboring values. Spatial autocorrelation due to a spatially correlated error structure is essentially a data measurement problem. For example, it can arise from data measurement errors involving the boundary of the spatial phenomena differing from the boundaries used for the measurement, or from omitted variables which are spatially correlated [5].

An econometric model that does not include spatial autocorrelation when the data generating process is spatial, could be adversely affected by this omission (bias in the regression coefficients, inconsistency, inefficiency, masking effects of spillovers, prediction bias). There are several procedures that can be used to test statistically for the presence of spatial dependence against the null hypothesis

5. See LeSage and Pace (2009) who provide motivations for regression models that include spatial autoregressive processes.

of spatial independence (Anselin, 1988). The most commonly used measure of spatial autocorrelation is Moran's (1948) I statistic which indicates the degree of spatial association reflected in the data. Considering spatial autocorrelation in an econometric model can be achieved in different ways by including spatially lagged variables, that is, weighted averages of the observations of "neighbors" of a given observation (Anselin, 1988). These spatially lagged variables can be the dependent variable (spatial auto-regressive - SAR - model), explanatory variables (spatial cross regressive model), or the error terms (spatial error model), or any combination of these options, which results in a wide range of spatial models (Elhorst, 2010).

Here, we consider two specifications for spatial autocorrelation: an additional regressor in the form of a spatially lagged dependent variable (*spatial autoregressive model*, *SAR*), and in the error structure (*spatial error model*, SEM). The SAR model is appropriate if the focus of interest is assessment of the existence and strength of spatial interaction. This is interpreted as substantive spatial dependence in the sense that it is directly related to a spatial model (e.g., a model that incorporates spatial interaction, yardstick competition, etc.)

The SAR model can be written as follows (Anselin, 1988):

$$\widetilde{y_i} = f(\widetilde{y_1}, ..., \widetilde{y_{i-1}}, \widetilde{y_{i+1}} ..., \widetilde{y_n}) \tag{5.4}$$

This provides the following equation:

$$\widetilde{y} = \rho W \widetilde{y} + X\beta + \varepsilon \tag{5.5}$$

W is an $n \times n$ spatial weight matrix and ρ is the spatial autoregressive parameter that expresses the magnitude of the interaction between grids.

The SEM takes account of the interactions between non-observed factors that affect the agricultural land use conversion decision. The interactions in the error terms can be expressed as follows:

$$\varepsilon_i = f(\varepsilon_1, ..., \varepsilon_{i-1}, \varepsilon_{i+1}..., \varepsilon_n) \tag{5.6}$$

where ε_i is the spatial residual in the grid i and $\varepsilon_1, ..., \varepsilon_{i-1}, \varepsilon_{i+1}..., \varepsilon_n$ are spatial residuals in the other grids.

$$\begin{aligned} \widetilde{y} &= X\beta + \varepsilon \\ \varepsilon &= \lambda W\varepsilon + u \end{aligned} \tag{5.7}$$

The parameter λ expresses the interaction between residuals and u is an *iid* [6] error term such that $u \sim \mathrm{iid}(0, \sigma^2 I)$.

We estimate the SEM and SAR model using the `R` package `spdep` (Bivand et al., 2013; Bivand and Piras, 2015). The spatial neighbourhood matrix, W, is obtained by triangulation of the centroids of the grid cells and its values are consequentially row-weighted.

5.3 Data presentation

Land use data

The land use data are from the CLC database for France at the scale of 100 m x 100 m (1 ha) grids and for the year 2000. The land cover classes are agriculture, pasture, forest, urban, and other. Table 11 in Appendix A summarizes the rules applied for the aggregation of the land use classes. The resulting map is given in Figure 5.1. Then, we calculate the share of each land use class for each (8 km x 8 km) grid cell; we know that each cell includes a maximum of 6,400 ha. Land use shares are expressed as the sum of the same land use classes in hectares divided by the surface of the grid cell. Although these cells are generated as homogeneous, they are changed by their intersection with the French borders. For

6. Independent and identically distributed random variable.

instance, grid cells on the coast are restricted to their parts on dry land.

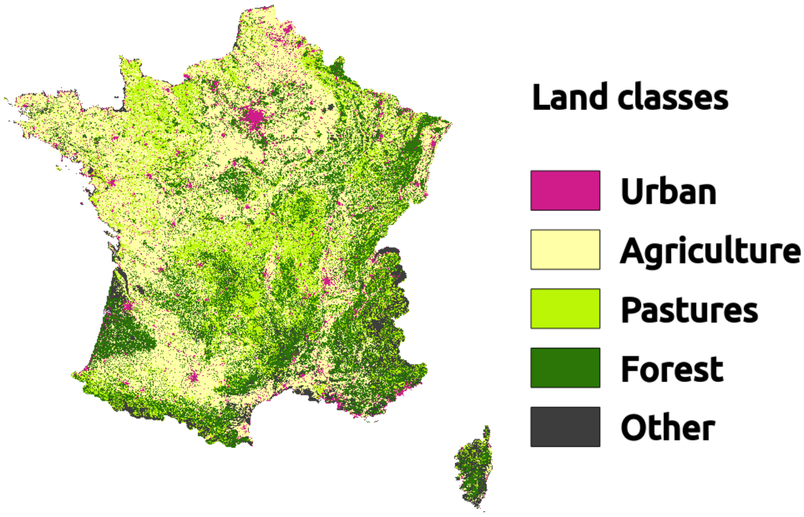

Figure 5.1 – Corine Land Cover (CLC) data aggregated in five land use classes for the year 2000.

Agricultural and forestry rent proxies

General information and descriptive statistics of the variables used in the study are summarized in Table 5.1.

Farmers' revenues Data on farmers' revenues are provided by the European Union's FADN at the European NUTS 2 level (Gross Farm Income, SE410). We focus on the revenues from crop production (cereals, oleaginous and other field crops) and animal breeding. Revenues from viticulture, horticulture, and other perennial crops are excluded because of the high profits per hectare and their limited areas (Table 12 in Appendix A). For instance, viticulture in France accounts for only 1.5% of the metropolitan territory but provides about 15% of agricultural sector value [7].

Land price As already mentioned, land prices are generally assumed to be the NPV of future land rents (Ricardo, 1817). The literature shows that this might be

7. FranceAgriMer, `www.franceagrimer.fr`.

a rather strong assumption since the standard NPV formula ignores the possibility of land conversion to other than agricultural use (Clark et al., 1993; Gutierrez et al., 2007; Karlsson and Nilsson, 2013). To address this, Guiling et al. (2009) extend the NPV formula. The most profitable conversion is land development (switch to urban use). In the case of France, Cavailhès and Wavresky (2003) find that immediate proximity to a city results in a high development premium which falls sharply as the distance from the city increases. In land use models, land prices are often used in the context of a hedonic approach to climate change impact assessments, otherwise known as the Ricardian method proposed initially by Mendelsohn et al. (1994) and focused solely on agricultural use. Because of the influence of urbanization on agricultural land prices, Schlenker et al. (2005) account for population density in their analysis of climate change effects on U.S. agriculture. Ay et al. (2014a) use the Ricardian approach in order to assess the effects of climate change on land use, and consequently common birds in France. Annual data on land prices, used in this paper, is provided by the statistical department of the French Ministry of Agriculture (Agreste) at the scale of the French small agricultural region.

Land shadow price In this study, we test land shadow price as a proxy; to our knowledge, this is the first such application in econometric land use shares models. Shadow prices capture the non-market value that farmers attribute to their production. They account for the complex interactions between on-farm activities. We use the values estimated by a European Union mathematical programming model for agriculture, applied to France. The economic supply-side model AROPAj (for a detailed description see Jayet et al., 2015) is based on FADN data and accounts for the Common Agricultural Policy. The economic agents in the model are representative farms grouped by farm types maximizing their gross margins (revenue minus variable costs). For each farmer the only publicly available information concerning location is the FADN region in which he/she operates. In order to maximize their profits, farmers in the model allocate their land to different crops while respecting a total area constraint. We use the Lagrange multiplier (LM) associated with this constraint in our comparative study of agricultural returns proxies. Microeconomic theory tells us that at the optimum, this Lagrange multiplier (shadow price or dual value) should be equal to the annual agricultural rent [8].

Shadow prices are used when real market values are not available or existing ones do not include some particularity of the good in question, as in the case of traditional maize varieties in Mexico (Arslan, 2011). In the case of France, land rental prices are administered by public authorities [9], and thus the shadow price and the observed rental prices do not coincide (Dupraz and Temesgen, 2012). Furthermore, agriculture is a complex system where some of the products are consumed on-farm [10], and thus, are not valued on the market.

Data on the agricultural rent proxies are available at different scales and for different years. Some aggregations were necessary in order to obtain data at the same scale. Thus, farmers' revenues and land prices are averaged over five and six year periods respectively [11]. In the absence of information on land prices for a given small agricultural region, we use the mean value for the corresponding French *département*. The data on land shadow prices from the AROPAj model are considered at their original scale, namely the FADN region (corresponding to the NUTS 2 level).

Forestry rents are approximated by the expected returns estimated by the partial-equilibrium model FFSM++ (Caurla and Delacote, 2012; Caurla et al.,

8. Agricultural rent is the remuneration of land as a factor of production. The equality between the Lagrange multiplier associated with the total land constraint and agricultural rent results from the application of the duality theorem to the profit maximization problem. Following this approach the profit maximization problem is equivalent to the cost minimization problem. For a general description see McFadden (1978).

9. French Rural Code, Article L411-11. In some regions this regulation is circumvented and new tenants are often obliged to pay under-the-counter former ones in order to obtain rights on land.

10. For instance, manure could be used as a fertilizer on crops while some of the biomass produced could be destined for animal feeding.

11. Inflation estimates for the period are provided by the World Bank, `http://data.worldbank.org/indicator/NY.GDP.DEFL.KD.ZG`.

2013; Lobianco et al., 2015a) developed by the Forestry Economics Laboratory at the French Agricultural Research Institute (INRA) in Nancy. The expected returns are calculated for 2006 at the scale of the French administrative region (NUTS2) and for coniferous and broadleaved forests. We use an average of these two values.

Both the AROPAj and the FFSM++ models include biological modules. AROPAj is partly coupled with the generic crop model STICS (Brisson et al., 2003, 2009), while FFSM++ uses parameters (mortality and growth of trees) derived from statistical data. Their biological modules allow both models to take into account the effects of climate change (the linkage between modules is detailed in Leclère et al., 2013 in the case of AROPAj). Also, the economic components of the models allow the simulation of different price and policy scenarios [12].

In this paper we assess the LUCs induced by the introduction of a tax on the mineral nitrogen fertilizers used by farmers (Section 5.4). We also evaluate the effects of the policy in the context of climate change based on the estimates provided by the biological and economical components of AROPAj and FFSM++ (Section 5.4).

Demography

Approximation of the urban rent is based on population density (in terms of number of households per ha) and household revenues. Both indicators are provided by the French statistical institute (INSEE), revenues are available at the scale of the *commune*, and the number of households is available for a regular 200 m x 200 m grid [13]. We use projections on demographic evolution from the INSEE (at the *département* level, 2040 horizon) and estimates from Center for International Earth Science Information Network (2002) for the simulation of climate induced land use change (Section 5.4).

12. For instance, an obligatory set-aside clause increases the demand for low quality land and consequently its rent.

13. INSEE, http://www.insee.fr/fr/themes/detail.asp?reg_id=0&ref_id=donnees-carroyees&page=donnees-detaillees/donnees-carroyees/donnees_carroyees_diffusion.htm.

Physical data

We also use data on topography of land.

Soils are represented by the data provided by the Joint Research Centre (JRC Panagos et al., 2012) at the scale of 1:1000000 and further aggregated at grid cell level. The indicator we use for soil quality is soil texture according to four levels. The lowest quality, level 1, is used as the reference. Land quality is an important variable in land use models (Chakir and Le Gallo, 2013; Ahn et al., 2000; Lubowski et al., 2008).

Relief (altitude and slope) is derived from the digital elevation model (DEM) GTOPO, available at the scale of 30 arc seconds (approximately 1 km). In the model only slope is introduced because of the high correlation between slope and altitude. Slope is also leading to better results in terms of fit of the models.

5.4 Estimation results

In order to compare estimations and evaluate the gains associated with different spatial autocorrelation specifications, we consider three estimators for each land use share model:

1. pooled ordinary least square (OLS) which ignores spatial autocorrelation;
2. the SEM estimator which takes account of the autoregressive spatial error autocorrelation
3. the SAR model which takes account of the autoregressive spatial autocorrelation

We estimate the OLS model, followed by the SEM and SAR models. Each specification (OLS, SEM, SAR) is estimated for the three proxies for agricultural land rents - shadow price, farmer's revenue, and arable land and pasture land prices. The results are presented in tables 13, 14 and 15. For each of the OLS models

Variable	Description	Mean	St. dev.	Min	Max
Land use					
s_{ag}	Share of agricultural use	0.438	0.276	0	1
s_{pa}	Share of pastures	0.181	0.181	0	0.94
s_{fo}	Share of forests	0.262	0.22	0	0.989
s_{ur}	Share of urban	0.053	0.097	0	0.99
s_{ot}	Share of other uses *Source*: CLC 2000 *Scale*: aggregated at 8 km x 8 km	0.065	0.133	0	1
Shadow price	Land shadow price (k€/ha) *Source*: AROPAj v.2 (2002) *Scale*: NUTS 2	0.576	0.197	0	1.029
Agri revenue	Farmers' revenues (k€/ha) *Source*: FADN, mean 1995-1999 *Scale*: NUTS 2 scale	0.651	0.153	0.19	0.975
Land price	Price for arable land (k€/ha) *Source*: Agreste, mean 1995-2000 *Scale*: French small agricultural region or *département*	3.035	1.485	0	20.256
For revenue	Forestry revenues (€/ha) *Source*: FFSM++, 2006 *Scale*: NUTS 2 scale	65.295	34.279	0	133.915
Pop revenues	Households' revenues (k€/ year/ household) *Source*: INSEE, 2000 *Scale*: French *commune*	12.424	3.213	0	44.642
Pop density	Households density (households/ ha) *Source*: INSEE, 2010 *Scale*: 200 m x 200 m grid	5.541	2.973	2.75	140.131
Slope	Slope (%) *Source*: GTOPO 30 *Scale*: 30 arc sec ~ 1 km	4.363	6.211	0	44.2
TEXT	Soils' texture classes	*1*	*2*	*3*	*4*
	Number of cells *Source*: JRC, Panagos et al. (2012) *Scale*: 1:1000000	1180	4258	2859	525

Table 5.1 – Summary statistics of land use shares and the explanatory variables.

the Moran's I score is significant meaning that the null hypothesis of no spatial autocorrelation is rejected. The estimated spatial autocorrelation, λ for the SEM models and ρ for the SAR models, are significant at 1%.

The value of the log-likelihood function of the OLS models increases for the SEM and SAR models. This holds for all land use equations and all the agricultural rents considered. To identify whether the SAR or the SEM model better describes the data, we use the classic LM-tests proposed by Anselin (1988) as well as the robust LM-tests proposed by Anselin et al. (1996). Table 16 presents the results of these tests. Using the classic tests, both hypotheses of no spatially lagged dependent variable and no spatially autocorrelated error term are rejected at 5% significance for all the models. The robust LM test results show that both SAR and SEM specifications are relevant for $\ln(pst/oth)$ and $\ln(agr/oth)$. Concerning the forest land share, on the basis of the robust LM tests the SAR model is more appropriate. For urban use, the SEM is more appropriate for the specifications with land prices and agri-revenue.

In the OLS specification, the agricultural vs. other use and the urban vs. other use models perform better in terms of explained variance, with R^2 close or superior to 40%. The other two models, forest vs. other use and pasture vs. other use, do not score as highly; the R^2 coefficients are lower than 20%. Population density and households revenue are significant and have the expected signs (positive) for the urban vs. other model regardless of the agricultural rent proxy employed and the model specification. Furthermore, the coefficients of these two explanatory variables remain stable. These two findings are valid for forestry revenues in the forest vs. other use models. In the he SEM and SAR specifications the coefficients of determination improve, and their results are close. Thus, in the agriculture vs. other use model, the R^2 increases by about 25% while in the forest vs. other use and pastures vs. other models the increase is over 40%, and in the Urban vs. other use model the increase is of the order of 20%. Also, these model results seem to be independent of the agricultural proxy employed.

In the agriculture vs. other use models, the parameters associated with the land shadow price is significant for all specifications. The parameter associated with agricultural revenue is positive but significant on in the OLS model. The impact of the arable land price on the agricultural share is significant at the 10% confidence level in the OLS and SEM models, and at the 1% confidence level in the SAR model. The three proxies are reported to have a negative impact on the share of pasture, which is significant in all the model specifications.

Elasticities

We calculate the elasticities for agricultural land shares with respect to agricultural rents based on Equation 5.8. Appendix B provides more details on the calculus of these elasticities.

$$\frac{\partial s_{ag}}{\partial Agr\,rent} * \frac{Agr\,rent}{s_{ag}} = \beta_{agr\_rent} * Agr\,rent \tag{5.8}$$

Agr rent	Model	Min.	1st Qu.	Median	Mean	3rd Qu.	Max	St.Dev
Shadow price	OLS	0	0.3166	0.3558	0.4183	0.4949	0.7478	0.143
Shadow price	SEM	0	0.3766	0.4232	0.4975	0.5886	0.8895	0.170
Shadow price	SAR	0	0.2164	0.2432	0.2859	0.3382	0.5111	0.098
Land price	OLS	0	0.0976	0.1255	0.1381	0.168	0.8198	0.065
Land price	SEM	0	0.1676	0.2156	0.2372	0.2884	1.408	0.111
Land price	SAR	0	0.2835	0.3647	0.4014	0.488	2.382	0.188
Agri revenue	OLS	0.07943	0.2338	0.2563	0.2715	0.3161	0.407	0.064
Agri revenue	SEM	0.09455	0.2783	0.3051	0.3232	0.3763	0.4846	0.076
Agri revenue	SAR	0.01576	0.04639	0.05085	0.05387	0.06272	0.08076	0.013

Table 5.2 – Elasticities of agricultural land with respect to different agricultural rent proxies.

The estimated elasticities are stable for the Shadow price proxy under the three model specifications. In the case of Land prices proxy, the elasticities are greater for the SEM and the SAR models. The values for the Agri revenue proxy are similar for the SEM and the OLS models, and lower for the SAR model.

Predictions

The fitted values used in the SEM and SAR models are obtained by exploiting the R package spdep package. The estimates use the available response variables (Equations 5.5 and 5.7). Since these variables are unknown when the predictions are formulated, we use Equations 5.10, 5.11 and 5.12. From a technical point of view, we build on Equation 5.3.

$$\widetilde{y}_k = \beta_k^{'} X_k + u_k \quad \forall k = 1, \ldots, K \tag{5.9}$$

For the models ignoring spatial autocorrelation, estimated by OLS, the predictor for the ith cell for equation k is simply:

$$\widehat{y}_{ik}^{OLS} = X_{ik} \widehat{\beta}_{k,OLS} \tag{5.10}$$

where X_{ik} is the matrix of data for observation i in equation k and $\widehat{\beta}_{k,OLS}$ is the pooled OLS estimator obtained for equation k.

In case of the SEM model allowing for spatial autocorrelation of error terms, the predictor is similar as follows:

$$\widehat{y}_{ik}^{SEM} = X_{ik} \widehat{\beta}_{k,SEM} \tag{5.11}$$

where $\widehat{\beta}_{k,SEM}$ is the SEM estimator obtained for equation k.

In case of the SAR model the predictor is as follows:

$$\widehat{y}_{ik}^{SAR} = (I - \widehat{\rho}_k W)^{-1} X_{ik} \widehat{\beta}_{k,SAR} \tag{5.12}$$

where $\widehat{\beta}_{k,SAR}$ is the SAR estimator and $\widehat{\rho}_k$ is the estimated autocorrelation coefficient for equation k.

Table 5.3 presents the values of the normalized root-mean-square errors (NRMSE) for the three proxies and the three model specifications. Accounting for spatial autocorrelation is reducing the NRMSE when predicting agriculture,

	Land use	Shadow price			Land price			Agri revenue		
		OLS	*SEM*	*SAR*	*OLS*	*SEM*	*SAR*	*OLS*	*SEM*	*SAR*
1	s_ag	0.2265	0.1221	0.1268	0.2223	0.1206	0.1252	0.2232	0.1215	0.1260
2	s_fo	0.1891	0.1125	0.1169	0.1872	0.1116	0.1155	0.1911	0.1125	0.1168
3	s_ot	0.0978	0.0647	0.0643	0.0995	0.0647	0.0646	0.0998	0.0648	0.0646
4	s_pa	0.1906	0.0885	0.0884	0.1888	0.0876	0.0882	0.1909	0.0882	0.0886
5	s_ur	0.0632	0.0470	0.0504	0.0624	0.0474	0.0504	0.0614	0.0470	0.0503

Table 5.3 – Normalized root-mean-square error for the different proxies and the three model specifications.

pastures and forests regardless of the proxy for the agricultural rent. The urban and other uses seem less influenced by spatial autocorrelation.

Environmental policy simulations: nitrogen pollution from agriculture

The intensification of agriculture since the Second World War has enabled a significant increase in the production of food in developed countries. Nevertheless, the high quantities of fertilizers and pesticides that this has necessitated have caused environmental problems including soil erosion, water and air pollution, loss of biodiversity, etc. For instance, the use of fertilizers has resulted in three types of nitrogen pollution: (i) nitrate pollution of soils and water, (ii) nitrous oxide emissions, and (iii) atmospheric emissions of ammonia (Bourgeois et al., 2014). Given the high nitrate concentrations observed in French water bodies, a tax on nitrogen fertilizers could be an environmental policy tool to reduce anthropogenic pressure on water resources. Use of fertilizers is associated also with GHG emissions (nitrous oxide), which need to be reduced to mitigate climate change.

Agriculture mathematical programming models offer the possibility to simulate different public policy scenarios through sets of parameters or constraints. For instance, AROPAj takes account of the major animal and crop activities observed in the European Union. These activities are parameterized in detail. For eight of the major crops dose-response functions are estimated (Leclère et al., 2013) using the crop model STICS (Brisson et al., 2003, 2009). These functions define the crops' responses to nitrogen and serve as production functions. STICS also evaluates nitrogen pollutant emissions allowing pollution functions to be fitted and introduced in AROPAj (Bourgeois et al., 2014). This methodology allows us to test farmers' reactions in terms of quantity of nitrogen fertilizer used and resulting pollutant emissions when there is a price shock. Were a tax per unit of nitrogen to be imposed this would represent a price shock.

Assuming a price of about 1 Euro per kg of nitrogen content in fertilizers, we test two nitrogen tax policies. In the first case we increase the price 50% and in the second we increase it 100% [14]. Such policies would reduce the profitability of agriculture (ceteris paribus, no price feedback is considered), and consequently the land shadow price (Figure 5.2). Using the econometric land use models presented above, we can evaluate the effects of the tax in terms of LUC. The results are summarized in Tables 5.5, 5.6, 5.7 and maps are provided in the Appendix E. Table 5.4 presents the emissions abatement per ha, and the change in agricultural area following the introduction of the taxation policies. The results show that the impact of policy is reinforced because pollutants emissions per ha are reduced (intensive margin) and also the total number of ha in agricultural use (extensive margin) is reduced.

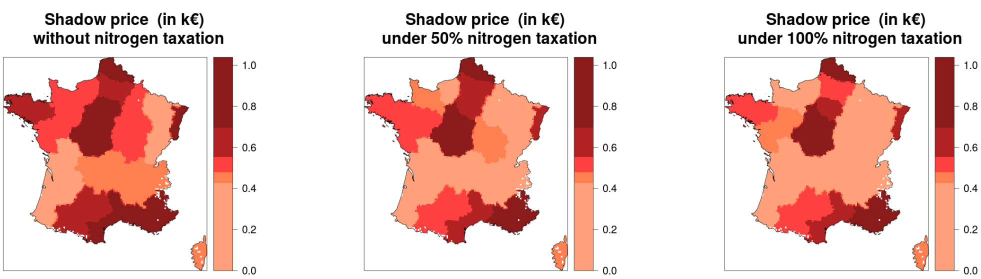

Figure 5.2 – Land shadow price in the BAU case and under nitrogen taxation policy schemes.

14. Jayet and Petsakos (2013) simulate numerous tax levels and Common Agricultural Policy scenarios using the AROPAj model

Policy scenario	Nitrates per ha (%)	Nitrous oxide per ha (%)	Agr. area (%) *OLS*	*SEM*	*SAR*
BAU	100.00%	100.00%	100.00%	100.00%	100.00%
Tax 50%	86.39%	75.06%	97.42%	98.07%	93.92%
Tax 100%	77.22%	59.23%	95.39%	96.53%	89.25%

Table 5.4 – Emission abatement and change in agricultural area.

Adaptation and mitigation of climate change

LUC is one of major sources of GHG with 12.2% of total emissions in 2005 (Herzog, 2009), making it one of the causes of climate change. However, LUC can also result from climate change in the form of economic adaptation. The methodology proposed for the study of climate induced LUC is summarized in Figure 5.3. The biological modules of the two mathematical models, AROPAj and FFSM++, allow simulation of climate change scenarios that take account of the switch to other crops and tree species. Here, we evaluate two IPCC emissions scenarios - A2 (pessimistic) and B1 (optimistic). For the land shadow price we use the results in Leclère et al. (2013) based on the ECHAM5 model [15]. Forest rents are based on the simulations of the ARPEGE Model by Meteo-France [16]. The population and revenue projections are from INSEE (2010) and the Center for International Earth Science Information Network (CIESIN, 2002). We also evaluate the effects of two climate change mitigation policies - the two nitrogen input tax levels presented in Section 5.4, in the context of the IPCC Special Report Emissions Scenarios.

The predicted impacts of climate change on land use using the OLS model are reported in Table 5.8. We present the impacts of climate change on LUCs in ha as well as in percentages. Results show that the impacts of climate change on agriculture, pasture, and forest are very similar for the A2 and B1 scenarios. In these two scenarios agricultural land use share increases by 20%, pasture decrease by 50%, and forests decrease by 22%-25%. The results for urban land show a contrast with an increase in urban area of 7.6% under A2 and a decrease of 4% under

15. `http://www.mpimet.mpg.de/en/wissenschaft/modelle/echam/echam5.html`
16. `http://www.cnrm-game.fr/spip.php?article124&lang=en`

	Min.	1st Qu.	Median	Mean	3rd Qu.	Max.
			Tax=+50%			
Land share changes						
ag	-0.04	-0.018	-0.014	-0.014	-0.01	0.126
pa	-0.046	0.004	0.007	0.008	0.011	0.031
fo	-0.073	0.004	0.006	0.006	0.008	0.029
ur	-0.003	0	0	0.001	0.001	0.015
ot	-0.02	0	0	0	0	0.007
Land use change in ha						
ag	-255.3	-115.1	-89.53	-87.41	-57.61	25.33
pa	-10.14	25.57	44.74	48.33	69.76	197.9
fo	-51.01	25.57	38.29	38.11	51.17	185.1
ur	-19.2	0	0	3.499	6.372	95.96
ot	-127.7	0	0	-2.796	0	19.15
			Tax=+100%			
Land share changes						
ag	-0.077	-0.033	-0.026	-0.025	-0.018	0.12
pa	-0.044	0.007	0.013	0.014	0.019	0.055
fo	-0.069	0.007	0.01	0.011	0.014	0.054
ur	-0.006	0	0	0.001	0.001	0.026
ot	-0.037	0	0	-0.001	0	0.006
Land use change in ha						
ag	-491.4	-204.9	-160	-156.4	-108.8	24.12
pa	-8.844	44.75	76.82	87.62	121.6	351.5
fo	-95.86	44.78	63.96	67.23	89.42	344.6
ur	-31.98	0	0	6.139	6.398	166.3
ot	-236.3	0	0	-4.975	0	38.3

Table 5.5 – Nitrogen tax simulations (OLS estimates).

B1. The results also show that a tax on nitrogen mitigates the impacts of CC on LUC. A tax that implies a 100% increase in the price of fertilizers leads to a small reduction in forest area (15%-18% instead of 22%-25%).

The SEM and SAR model estimates (Tables 5.9 and 5.10) both predict an increase in cropland of some 34% and a reduction in pasture and forests of 62% and 24%-28% respectively. The spatial dimension of our land use model allows us to present the geographical impacts of CC more precisely - see Figures 10, 13 and 16.

	Min.	1st Qu.	Median	Mean	3rd Qu.	Max.
			Tax=+50%			
Land share changes						
ag	-0.026	-0.014	-0.011	-0.011	-0.008	0.115
pa	-0.088	0.006	0.009	0.009	0.011	0.027
fo	-0.02	0.001	0.002	0.001	0.003	0.048
ur	-0.003	0	0	0.001	0.001	0.013
ot	-0.006	0	0	0	0	0.006
Land use change in ha						
ag	-165.9	-89.43	-70.33	-66.77	-44.83	23.12
pa	-18.98	38.38	51.22	54.49	70.36	172.6
fo	-127.7	5.607	12.77	6.013	19.15	57.5
ur	-12.8	0	0	3.774	6.386	83.11
ot	-6.398	0	0	1.283	0	31.92
			Tax=+100%			
Land share changes						
ag	-0.051	-0.025	-0.021	-0.019	-0.015	0.11
pa	-0.084	0.011	0.016	0.016	0.02	0.053
fo	-0.04	0.001	0.003	0.002	0.005	0.043
ur	-0.004	0	0	0.001	0.001	0.022
ot	-0.005	0	0	0	0.001	0.012
Land use change in ha						
ag	-325.5	-153.6	-127.9	-119.7	-89.6	22.11
pa	-16.9	70.24	96.22	99.28	128	338.7
fo	-255.4	6.396	19.19	9.626	25.62	102.2
ur	-25.6	0	0	7.029	6.401	140.7
ot	-12.78	0	0	3.043	6.388	57.45

Table 5.6 – Nitrogen tax simulations (SEM estimates).

	Min.	1st Qu.	Median	Mean	3rd Qu.	Max.
			Tax=+50%			
Land share changes						
ag	-0.113	-0.028	-0.013	-0.017	-0.001	0.116
pa	-0.001	0	0	0.003	0.001	0.111
fo	-0.049	0	0	0	0	0.021
ur	-0.115	0.002	0.012	0.015	0.024	0.079
ot	-0.065	0	0	-0.001	0	0.001
Land use change in ha						
ag	-721.3	-179	-76.79	-109	-6.398	31.8
pa	-0.26	0	0	16.43	6.392	710.2
fo	-312.4	0	0	1.264	0	134.2
ur	-274.2	6.402	70.37	95.9	153.5	504.3
ot	-415.5	0	0	-4.879	0	6.397
			Tax=+100%			
Land share changes						
ag	-0.201	-0.051	-0.022	-0.031	-0.002	0.094
pa	0	0	0	0.005	0.001	0.202
fo	-0.091	0	0	0	0	0.037
ur	-0.094	0.003	0.02	0.027	0.044	0.149
ot	-0.122	0	0	-0.001	0	0.002
Land use change in ha						
ag	-1283	-320.1	-134.4	-192.8	-12.78	18.89
pa	0	0	0	31.55	6.404	1289
fo	-580.2	0	0	2.283	0	236.5
ur	-510.2	12.8	121.6	167.5	275.2	951.2
ot	-779.8	0	0	-8.805	0	12.79

Table 5.7 – Nitrogen tax simulations (SAR estimates).

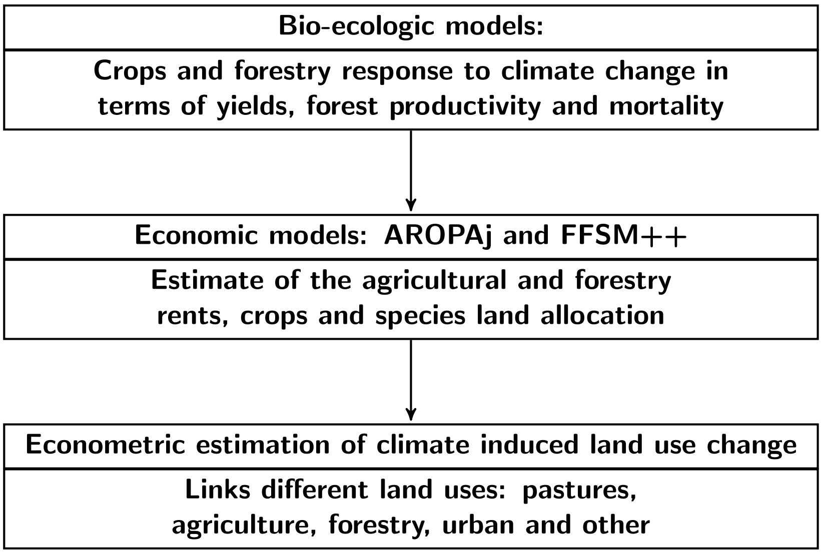

Figure 5.3 – Methodology for the assessment of the climate induced land use change.

Scenario	s_ag	s_pa	s_fo	s_ur	s_ot
Land use change in %					
CC=A2	20 %	-50.7 %	-25.2 %	7.6 %	46 %
CC=A2, t=50%	16.8 %	-45.4 %	-21.4 %	10.9 %	44.5 %
CC=A2, t=100%	14.3 %	-40.8 %	-18.3 %	13.4 %	42.8 %
CC=B1	19.8 %	-50.2 %	-21.8 %	-4.2 %	36.3 %
CC=B1, t=50%	16.8 %	-45 %	-18.2 %	-1.5 %	34.5 %
CC=B1, t=100%	14.4 %	-40.4 %	-15.2 %	0.6 %	32.8 %
CC=BAU, t=50%	-2.6%	6.2%	2.5%	1.3%	-1.4%
CC=BAU, t=100%	-4.6%	11.3%	4.4%	2.2%	-2.5%
Land use change in 1000 ha					
CC=A2	5979	-3477	-3431	187	742
CC=A2, t=50%	5038	-3115	-2908	269	716
CC=A2, t=100%	4268	-2797	-2492	331	690
CC=B1	5936	-3448	-2969	-104	585
CC=B1, t=50%	5040	-3088	-2471	-37	556
CC=B1, t=100%	4302	-2770	-2073	14	528
CC=BAU, t=50%	-771	426	336	31	-25
CC=BAU, t=100%	-1380	773	593	54	-44

Table 5.8 – Simulations of climate change and nitrogen tax (OLS estimates).

Scenario	s_ag	s_pa	s_fo	s_ur	s_ot
Land use change in %					
CC=A2	13.8 %	-46.3 %	-12.9 %	8.2 %	22.2 %
CC=A2, t=50%	11.4 %	-39 %	-11.7 %	11.5 %	26.4 %
CC=A2, t=100%	9.4 %	-32.5 %	-10.9 %	14 %	29.7 %
CC=B1	13.9 %	-45.3 %	-10.7 %	-4.3 %	16.1 %
CC=B1, t=50%	11.6 %	-38.4 %	-9.6 %	-1.6 %	19.6 %
CC=B1, t=100%	9.7 %	-32.2 %	-8.8 %	0.4 %	22.3 %
CC=BAU, t=50%	-1.9%	7.8%	0.4%	1.5%	1.6%
CC=BAU, t=100%	-3.5%	14.2%	0.6%	2.5%	2.8%
Land use change in 1000 ha					
CC=A2	4216	-2855	-1829	207	261
CC=A2, t=50%	3465	-2406	-1660	290	311
CC=A2, t=100%	2849	-2003	-1549	354	349
CC=B1	4228	-2794	-1514	-110	189
CC=B1, t=50%	3531	-2368	-1352	-41	231
CC=B1, t=100%	2956	-1986	-1244	11	263
CC=BAU, t=50%	-589	481	53	33	11
CC=BAU, t=100%	-1056	876	85	62	27

Table 5.9 – Simulations of climate change and nitrogen tax (SEM estimates).

Scenario	s_ag	s_pa	s_fo	s_ur	s_ot
Land use change in %					
CC=A2	10.5 %	-76.7 %	-76.3 %	-1.7 %	6.3 %
CC=A2, t=50%	1.4 %	-75.3 %	-75.8 %	2.7 %	4.5 %
CC=A2, t=100%	-5.5 %	-73.7 %	-75.4 %	6 %	2.9 %
CC=B1	34.8 %	-82.7 %	-67.7 %	-14 %	14.4 %
CC=B1, t=50%	26.3 %	-80.7 %	-66.7 %	-9.9 %	12.6 %
CC=B1, t=100%	19.6 %	-78.6 %	-66 %	-6.7 %	11 %
CC=BAU, t=50%	-6.1%	25.3%	1.1%	2.5%	-1.4%
CC=BAU, t=100%	-10.7%	48.2%	2%	4.4%	-2.5%
Land use change in 1000 ha					
CC=A2	1660	-445	-845	-567	197
CC=A2, t=50%	228	-436	-840	908	141
CC=A2, t=100%	-864	-427	-835	2034	92
CC=B1	5515	-480	-750	-4736	450
CC=B1, t=50%	4163	-468	-740	-3349	394
CC=B1, t=100%	3105	-455	-731	-2263	345
CC=BAU, t=50%	-962	145	11	846	-43
CC=BAU, t=100%	-1701	278	20	1478	-78

Table 5.10 – Simulations of climate change and nitrogen tax (SAR estimates).

5.5 Conclusion and perspectives

The objective of this paper was to compare land use models based on three different proxies for agricultural land rent: farmers' revenues - land prices, and shadow land prices - derived from a mathematical programming model. We estimated a land use shares model for France at the scale of a homogeneous (8 km x 8 km) grid and considered five land use classes: (1) agriculture, (2) pasture, (3) forest, (4) urban and (5) other uses. We investigated what determines the shares of land in alternative uses using economic, pedoclimatic, and demographic explanatory variables. We modeled spatial autocorrelation between grid cells, and compared prediction accuracy and estimated elasticities for the different model specifications.

Our results show that results for predicted quality are similar for the three rent proxies. We found also that including spatial autocorrelation in land use models improves the quality of the predictions (RMSE indicators). We used the estimated models to simulate the impact of an input-based tax on fertilizer in terms of a LUC. We simulated two tax levels: 50% and 100% increase in the nitrogen price. Our results show very heterogeneous regional disparities with a decrease in the national agricultural area of 0.77 million ha and 1.4 million ha, and an increase in pasture land area of 0.42 million ha and 0.77 million ha for the 50% tax and 100% respectively.

We also used our econometric land use models to project land use in France under two IPCC climate change scenarios (A2, B1). We found that climate change mostly affects cropland area which increased by 6 million ha in both scenarios. The area to pasture fell by 3.4 million ha in both scenarios while forest decreased by 2.9 million ha in scenario B1 and by 3.4 million ha in scenario A2. Our methodology allows us to take into account of farmers' and forest managers' autonomous adaptation capacity: to switch crops, tree species, and management practices.

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A Data

Land Cover class	CLC value	LU class
1 Artificial Surfaces	1, ..., 11	Urban
2 Agricultural Areas	12, ..., 17 and 19, ..., 22	Agriculture
2.3 Pastures	18	Pastures
3 Forest and Semi Natural Areas	23, 24 and 25	Forest
3.2.1 Natural grasslands	26	Pastures
3.2.2 Moors and heathland	27	Other
3.2.3 Sclerophyllous vegetation	28	Other
3.2.4 Transitional woodland-shrub	29	Other
3.3 Open spaces with little or no vegetation	30, ..., 34	Other
4 Wetlands	35, ..., 39	Other
5 Water bodies	40, ..., 44	Other

Table 11 – Extract from the CLC classification and the corresponding LU aggregation.

Agricultural activity	Profit before tax (1000 euros)	Average farm surface (ha)
Cereals and protein crops	24.1	68
Horticulture	30.7	7
Wine under geographical label	52,9	12*
Other wine	13.1	12*
Fruits and others	10.5	13
Bovine (milk)	28.8	58
Bovine (meat)	24.2	46
Bovine (mixed)	33.1	75
Sheep and other	17.6	25
Pig, poultry and other	36.6	34
Mixed farming	27.0	48

* Average for viticulture in general

Table 12 – Average farmers' profits for 2005 per agricultural activity. The data on farms' size in hectares is for 2000. Source: Agreste, French Ministry of agriculture.

B Elasticities

Calculus for the elasticities.

$$
\begin{aligned}
\frac{\partial \ln\left(\frac{s_{ag}}{s_{ot}}\right)}{\partial Agr\,rent} &= \beta_{agr\_rent} \\
\frac{\partial\left(\frac{s_{ag}}{s_{ot}}\right)}{\partial Agr\,rent} * \frac{s_{ot}}{s_{ag}} &= \beta_{agr\_rent} \\
\frac{\partial s_{ag}}{\partial Agr\,rent} * \frac{1}{s_{ot}} * \frac{s_{ot}}{s_{ag}} &= \beta_{agr\_rent} \\
\frac{\partial s_{ag}}{\partial Agr\,rent} &= s_{ag} * \beta_{agr\_rent} \\
\frac{\partial s_{ag}}{\partial Agr\,rent} * \frac{Agr\,rent}{s_{ag}} &= s_{ag} * \beta_{agr\_rent} * \frac{Agr\,rent}{s_{ag}} \\
\frac{\partial s_{ag}}{\partial Agr\,rent} * \frac{Agr\,rent}{s_{ag}} &= \beta_{agr\_rent} * Agr\,rent
\end{aligned}
\tag{13}
$$

C Models estimates

Tables 13, 14 and 15 present the estimated parameters for the three agricultural rent proxies under the different model specifications.

Table 13 – Regional dual

	Dependent variable:											
	ln(pst/oth)			ln(agr/oth)			ln(for/oth)			ln(urb/oth)		
	OLS	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
Constant	1.928***	2.207***	0.508***	1.504***	2.339***	0.477***	2.536***	1.896***	0.739***	−4.919***	−4.180***	−2.342***
	(0.254)	(0.493)	(0.178)	(0.215)	(0.408)	(0.166)	(0.204)	(0.377)	(0.161)	(0.215)	(0.375)	(0.185)
Shadow price	−1.138***	−1.029**	−1.147***	0.727***	0.864**	0.497***	−0.434**	0.307	−0.297**	0.086	0.299	−0.404***
	(0.214)	(0.514)	(0.150)	(0.181)	(0.416)	(0.144)	(0.172)	(0.382)	(0.133)	(0.181)	(0.369)	(0.147)
For revenue	0.027***	0.018***	0.023***	0.014***	0.014***	0.007***	0.016***	0.016***	0.014***	0.018***	0.017***	0.015***
	(0.001)	(0.003)	(0.001)	(0.001)	(0.003)	(0.001)	(0.001)	(0.002)	(0.001)	(0.001)	(0.002)	(0.001)
Pop density	−0.134***	−0.094***	−0.307***	−0.139***	−0.099***	−0.274***	−0.173***	−0.117***	−0.324***	0.081***	0.122***	0.150***
	(0.014)	(0.012)	(0.009)	(0.011)	(0.012)	(0.009)	(0.011)	(0.011)	(0.008)	(0.011)	(0.012)	(0.010)
Pop revenues	−0.122***	−0.021	−0.098***	0.091***	0.075***	0.161***	0.058***	0.051***	0.122***	0.315***	0.279***	0.403***
	(0.013)	(0.016)	(0.009)	(0.011)	(0.014)	(0.008)	(0.010)	(0.014)	(0.008)	(0.011)	(0.015)	(0.009)
Slope	−0.056***	−0.062***	−0.066***	−0.292***	−0.258***	−0.341***	−0.058***	−0.033***	−0.053***	−0.218***	−0.217***	−0.244***
	(0.007)	(0.013)	(0.005)	(0.006)	(0.011)	(0.005)	(0.006)	(0.010)	(0.004)	(0.006)	(0.010)	(0.005)
TEXT2	1.563***	0.428***	2.035***	1.696***	0.635***	2.232***	0.021	0.053	0.030	1.011***	0.462***	1.323***
	(0.118)	(0.111)	(0.081)	(0.100)	(0.104)	(0.077)	(0.095)	(0.099)	(0.028)	(0.100)	(0.110)	(0.084)
TEXT3	2.479***	0.817***	3.393***	2.647***	1.168***	3.470***	0.721***	0.290**	0.911***	1.683***	0.827***	2.036***
	(0.125)	(0.129)	(0.086)	(0.106)	(0.120)	(0.083)	(0.100)	(0.114)	(0.058)	(0.106)	(0.126)	(0.089)
TEXT4	2.934***	1.044***	4.317***	2.883***	1.403***	4.063***	0.669***	0.197	0.649***	1.516***	0.570***	1.583***
	(0.188)	(0.177)	(0.128)	(0.159)	(0.166)	(0.122)	(0.151)	(0.158)	(0.108)	(0.159)	(0.175)	(0.133)
Moran's I	0.567***			0.456***			0.458***					
λ		0.792***			0.731***			0.714***			0.644***	
ρ			0.779***			0.704***			0.705***			0.612***
N	8822											
R^2	0.182	0.629	0.63	0.403	0.657	0.657	0.133	0.491	0.493	0.38	0.578	0.57
Log Lik.	-23704.86	-20892.41	-20848.63	-22230.67	-20325.95	-20285.2	-21752.48	-19910.07	-19878.68	-22223.33	-20923.09	-20956.42

Note: $^{*}p<0.1$; $^{**}p<0.05$; $^{***}p<0.01$

The values for the SAR models represent the total effect of the explanatory variables. The reported significance is for the estimated coefficients $\hat{\beta}_{k,SAR}$.

Table 14 – Land prices

	Dependent variable:											
	ln(pst/oth)			ln(agr/oth)			ln(for/oth)			ln(urb/oth)		
	OLS	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
Constant	1.605***	2.082***	0.449***	1.909***	2.753***	0.524***	2.495***	2.392***	0.764***	−4.995***	−4.344***	−2.508***
	(0.217)	(0.356)	(0.144)	(0.185)	(0.305)	(0.143)	(0.174)	(0.282)	(0.117)	(0.184)	(0.290)	(0.159)
Land price	−0.390***	−0.270***	−0.428***	0.053*	0.090*	0.153***	−0.231***	−0.138***	−0.247***	0.129***	0.195***	0.156***
	(0.032)	(0.056)	(0.022)	(0.027)	(0.049)	(0.021)	(0.026)	(0.046)	(0.020)	(0.027)	(0.047)	(0.023)
For revenue	0.030***	0.020***	0.026***	0.012***	0.012***	0.006***	0.017***	0.015***	0.015***	0.018***	0.017***	0.016***
	(0.001)	(0.003)	(0.001)	(0.001)	(0.002)	(0.001)	(0.001)	(0.002)	(0.001)	(0.001)	(0.002)	(0.001)
Pop density	−0.128***	−0.092***	−0.297***	−0.138***	−0.099***	−0.276***	−0.169***	−0.115***	−0.317***	0.078***	0.119***	0.145***
	(0.013)	(0.012)	(0.009)	(0.011)	(0.012)	(0.009)	(0.011)	(0.011)	(0.008)	(0.011)	(0.012)	(0.010)
Pop revenues	−0.092***	−0.016	−0.065***	0.092***	0.074***	0.150***	0.078***	0.058***	0.144***	0.302***	0.270***	0.383***
	(0.013)	(0.016)	(0.009)	(0.011)	(0.015)	(0.008)	(0.010)	(0.014)	(0.008)	(0.011)	(0.015)	(0.010)
Slope	−0.041***	−0.061***	−0.049***	−0.297***	−0.260***	−0.347***	−0.050***	−0.034***	−0.045***	−0.222***	−0.220***	−0.247***
	(0.007)	(0.013)	(0.005)	(0.006)	(0.011)	(0.005)	(0.006)	(0.010)	(0.004)	(0.006)	(0.010)	(0.005)
TEXT2	1.603***	0.445***	2.070***	1.685***	0.625***	2.216***	0.042	0.056	0.051	1.001***	0.453***	1.314***
	(0.117)	(0.111)	(0.080)	(0.100)	(0.104)	(0.077)	(0.094)	(0.099)		(0.100)	(0.110)	(0.084)
TEXT3	2.511***	0.842***	3.415***	2.661***	1.158***	3.461***	0.748***	0.302***	0.942***	1.662***	0.813***	1.993***
	(0.124)	(0.129)	(0.086)	(0.106)	(0.120)	(0.083)	(0.100)	(0.114)	(0.046)	(0.106)	(0.126)	(0.089)
TEXT4	3.015***	1.066***	4.378***	2.858***	1.392***	4.033***	0.711***	0.205	0.690***	1.496***	0.558***	1.570***
	(0.187)	(0.177)	(0.128)	(0.159)	(0.166)	(0.122)	(0.150)	(0.158)	(0.104)	(0.159)	(0.175)	(0.133)
Moran's I	0.562***			0.457***			0.453***			0.38***		
λ		0.789***			0.732***			0.71***			0.643***	
ρ			0.775***			0.704***			0.702***			0.61***
N	8822											
R^2	0.193	0.629	0.63	0.403	0.657	0.657	0.14	0.491	0.493	0.381	0.578	0.57
Log Lik.	-23645.51	-20883.08	-20840.54	-22236.85	-20326.42	-20283.39	-21715.63	-19905.85	-19872.08	-22212.19	-20914.75	-20953.39

Note: $^{*}p<0.1$; $^{**}p<0.05$; $^{***}p<0.01$

The values for the SAR models represent the total effect of the explanatory variables. The reported significance is for the estimated coefficients $\hat{\beta}_{k,SAR}$.

Table 15 – Agri revenue

	Dependent variable:											
	ln(pst/oth)			ln(agr/oth)			ln(for/oth)			ln(urb/oth)		
	OLS	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*	*OLS*	*spatial error*	*spatial autoregressive*
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
Constant	1.782***	2.305***	0.501***	1.756***	2.668***	0.557***	3.397***	2.783***	1.035***	−5.476***	−4.603***	−2.581***
	(0.254)	(0.518)	(0.169)	(0.215)	(0.420)	(0.128)	(0.202)	(0.383)	(0.135)	(0.214)	(0.377)	(0.184)
Agri revenue	−1.159***	−1.450**	−1.400***	0.417*	0.497	0.083	−2.305***	−1.227**	−2.324***	1.243***	1.160**	0.697***
	(0.271)	(0.698)	(0.179)	(0.229)	(0.553)		(0.215)	(0.499)	(0.167)	(0.228)	(0.479)	(0.190)
For revenue	0.031***	0.022***	0.028***	0.012***	0.011***	0.005***	0.020***	0.017***	0.018***	0.016***	0.015***	0.015***
	(0.001)	(0.003)	(0.001)	(0.001)	(0.002)	(0.001)	(0.001)	(0.002)	(0.001)	(0.001)	(0.002)	(0.001)
Pop density	−0.137***	−0.094***	−0.311***	−0.137***	−0.098***	−0.272***	−0.174***	−0.116***	−0.322***	0.081***	0.122***	0.148***
	(0.014)	(0.012)	(0.009)	(0.011)	(0.012)	(0.009)	(0.011)	(0.011)	(0.008)	(0.011)	(0.012)	(0.010)
Pop revenues	−0.123***	−0.020	−0.097***	0.095***	0.076***	0.165***	0.072***	0.056***	0.136***	0.305***	0.276***	0.393***
	(0.013)	(0.016)	(0.009)	(0.011)	(0.015)	(0.008)	(0.010)	(0.014)	(0.008)	(0.011)	(0.015)	(0.009)
Slope	−0.054***	−0.062***	−0.064***	−0.295***	−0.259***	−0.343***	−0.061***	−0.036***	−0.056***	−0.216***	−0.216***	−0.241***
	(0.007)	(0.013)	(0.005)	(0.006)	(0.011)	(0.005)	(0.005)	(0.010)	(0.004)	(0.006)	(0.010)	(0.005)
TEXT2	1.543***	0.423***	2.010***	1.700***	0.633***	2.230***	−0.038	0.041	−0.030	1.045***	0.474***	1.344***
	(0.118)	(0.111)	(0.080)	(0.100)	(0.104)	(0.077)	(0.094)	(0.099)		(0.100)	(0.110)	(0.084)
TEXT3	2.436***	0.813***	3.349***	2.673***	1.167***	3.488***	0.696***	0.286**	0.885***	1.692***	0.838***	2.024***
	(0.125)	(0.129)	(0.085)	(0.106)	(0.120)	(0.083)	(0.100)	(0.114)	(0.045)	(0.106)	(0.126)	(0.089)
TEXT4	2.907***	1.036***	4.280***	2.885***	1.401***	4.058***	0.571***	0.184	0.547***	1.572***	0.588***	1.624***
	(0.188)	(0.177)	(0.127)	(0.159)	(0.166)	(0.122)	(0.150)	(0.158)	(0.100)	(0.159)	(0.175)	(0.133)
Moran's I	0.567***			0.457***			0.449***			0.378***		
λ		0.792***			0.732***			0.71***			0.642***	
ρ			0.779***			0.704***			0.701***			0.61***
N	8822											
R^2	0.181	0.629	0.63	0.403	0.657	0.657	0.143	0.491	0.493	0.382	0.578	0.57
Log Lik.	-23709.75	-20892.24	-20848.73	-22237.04	-20327.71	-20285.76	-21698.72	-19907.44	-19870.24	-22208.62	-20920.5	-20955.95

Note: $^{*}p<0.1$; $^{**}p<0.05$; $^{***}p<0.01$

The values for the SAR models represent the total effect of the explanatory variables. The reported significance is for the estimated coefficients $\hat{\beta}_{k,SAR}$.

Table 16 – Lagrange miltiplier tests

		Models											
		ln(pst/oth)			*ln(agr/oth)*			*ln(for/oth)*			*ln(urb/oth)*		
Proxy	LM Test	Statistic	df	p-value	Statistic	df	p-value	Statistic	df	p-value	Statistic	df	p-value
Shadow price	LMerr	8402.81	1	0	5502.83	1	0	5508.71	1	0	3824.11	1	0
	LMlag	8593.49	1	0	5513.07	1	0	5642.74	1	0	3674.2	1	0
	RLMerr	46.38	1	0	173.39	1	0	6.04	1	0.014017401	194.62	1	0
	RLMlag	237.05	1	0	183.63	1	0	140.07	1	0	44.71	1	0
	SARMA	8639.86	2	0	5686.46	2	0	5648.77	2	0	3868.82	2	0
Land prices	LMerr	8256.74	1	0	5513.64	1	0	5402.61	1	0	3817.44	1	0
	LMlag	8432.28	1	0	5529.6	1	0	5533.84	1	0	3652.69	1	0
	RLMerr	52.54	1	0	174.19	1	0	7.84	1	0.0051083418	204.61	1	0
	RLMlag	228.07	1	0	190.15	1	0	139.07	1	0	39.86	1	3e-10
	SARMA	8484.82	2	0	5703.79	2	0	5541.68	2	0	3857.3	2	0
Agri revenue	LMerr	8422.24	1	0	5510.55	1	0	5301.49	1	0	3785.84	1	0
	LMlag	8612.21	1	0	5524.85	1	0	5468.63	1	0	3629.37	1	0
	RLMerr	46.31	1	0	173.59	1	0	2.85	1	0.0912817067	196.27	1	0
	RLMlag	236.28	1	0	187.89	1	0	169.99	1	0	39.8	1	3e-10
	SARMA	8658.52	2	0	5698.44	2	0	5471.49	2	0	3825.65	2	0

D Predicted land use shares

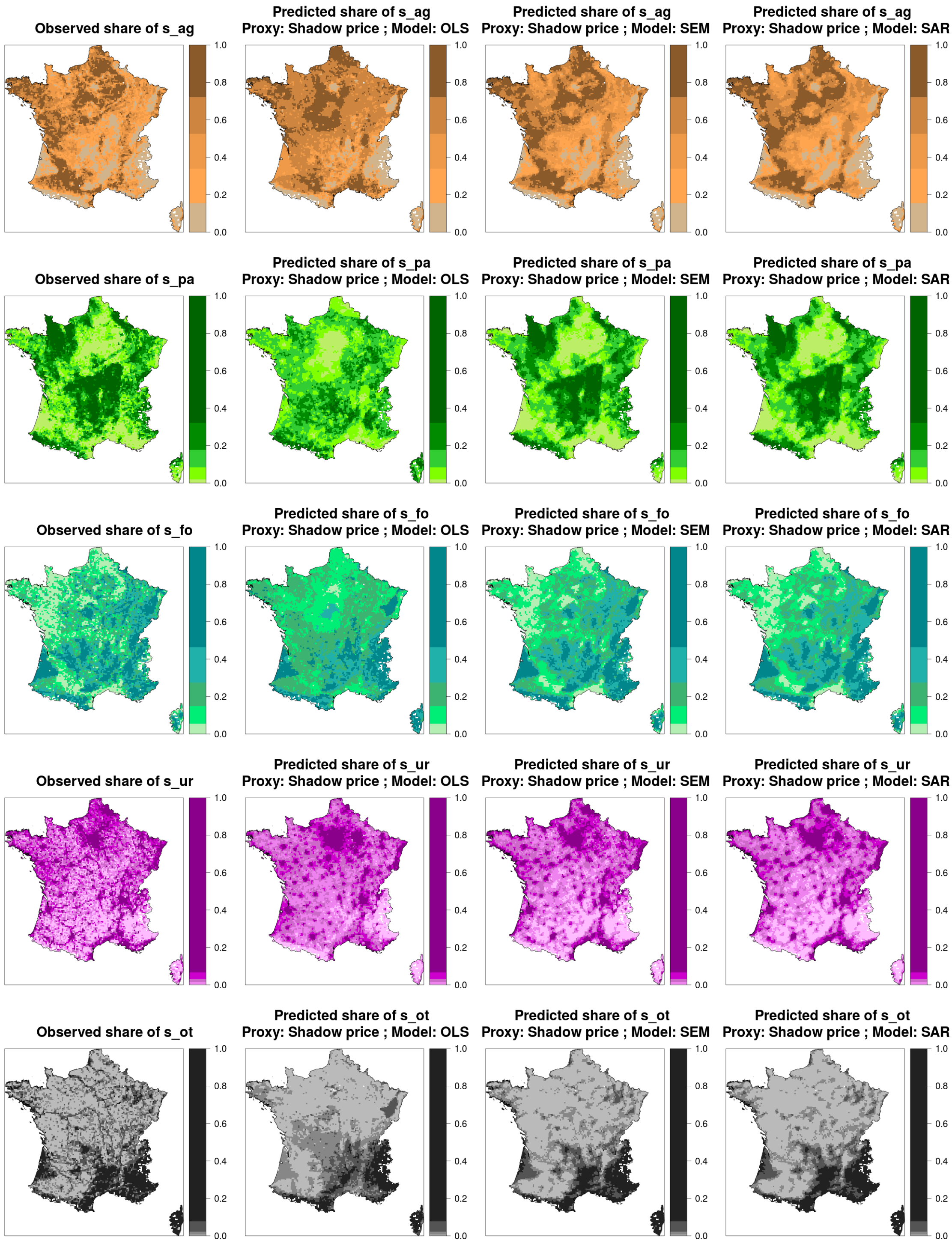

Figure 4 – Observed land use shares (left pane) and predicted (OLS, SEM and SAR models on the right). Proxy for the agricultural rent: shadow price.

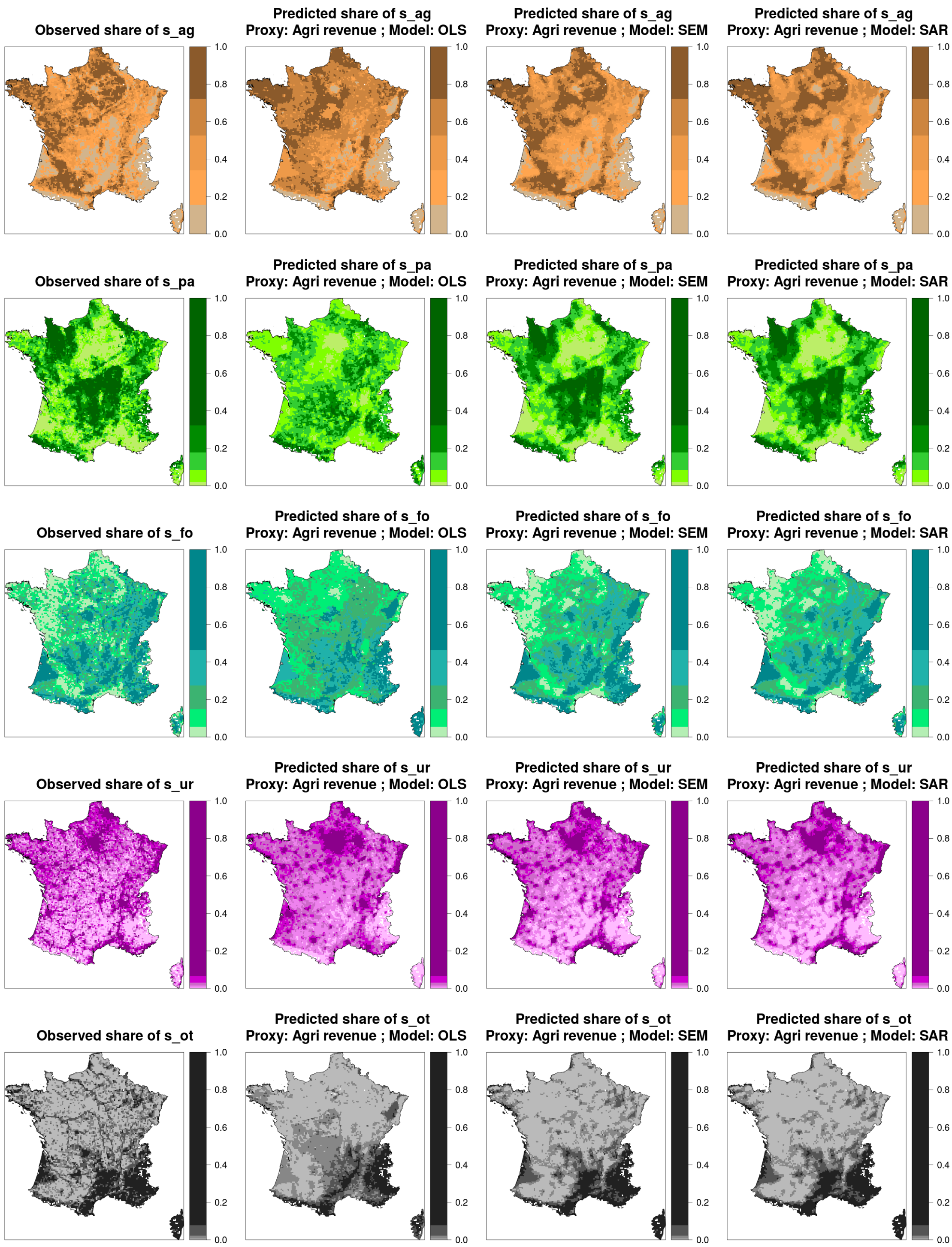

Figure 5 – Observed land use shares (left pane) and predicted (OLS, SEM and SAR models on the right). Proxy for the agricultural rent: farmers' revenues.

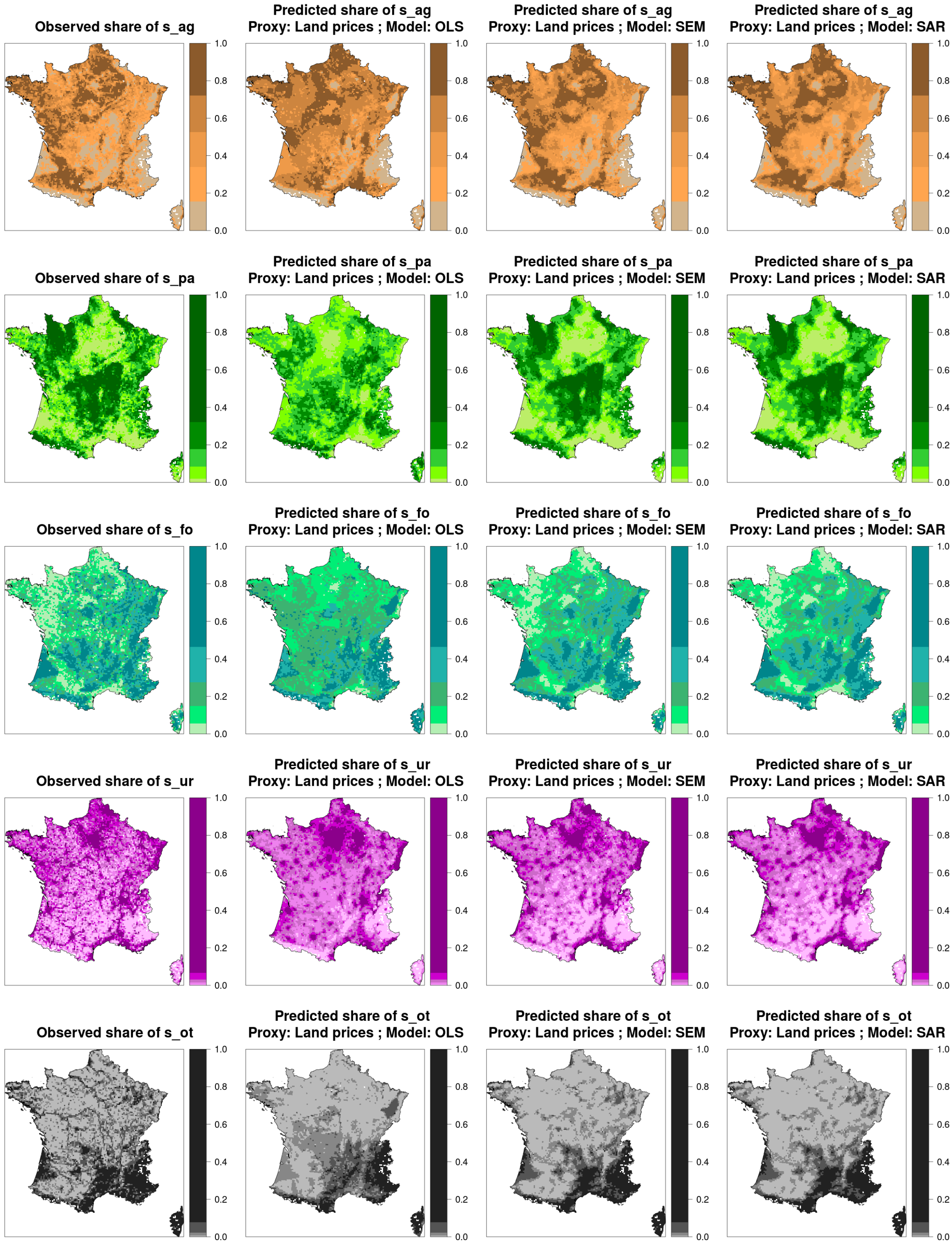

Figure 6 – Observed land use shares (left pane) and predicted (OLS, SEM and SAR models on the right). Proxy for the agricultural rent: agricultural land prices.

E Land use shares and nitrogen input tax

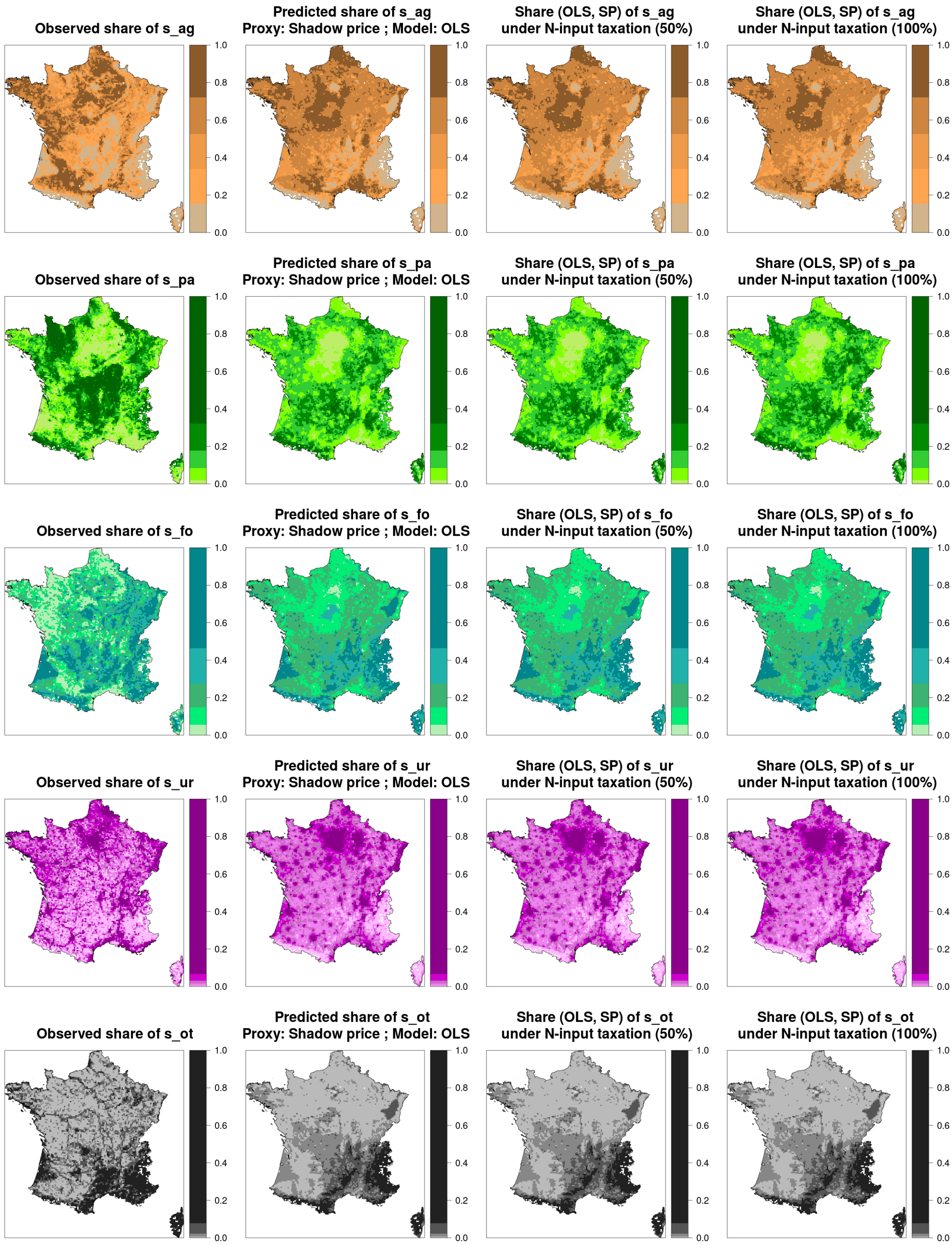

Figure 7 – Observed and predicted (via OLS) land use without tax (left), with 50% and with 100% tax (right) on the mineral nitrogen input. Proxy for the agricultural rent: shadow price.

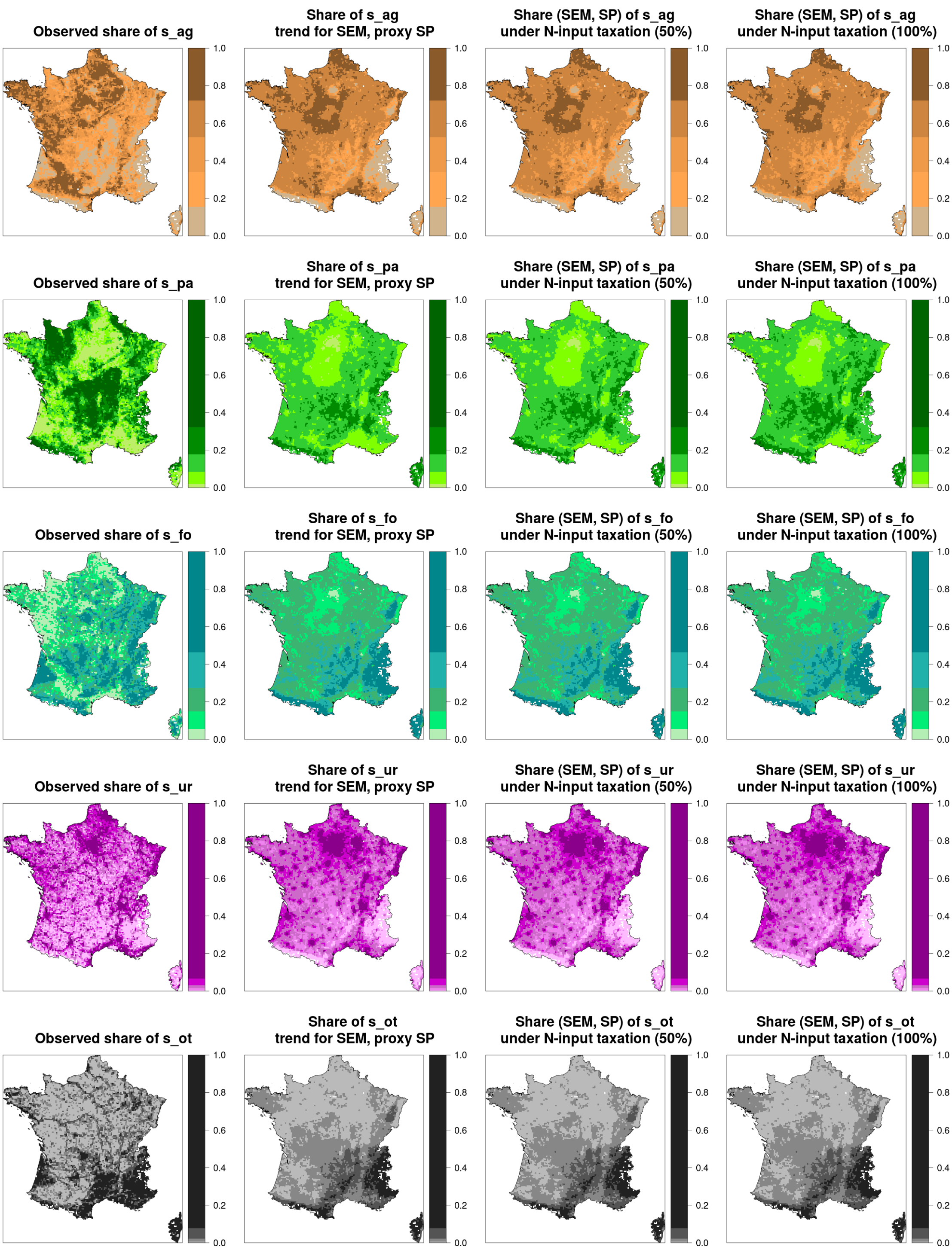

Figure 8 – Observed and predicted (via SEM) land use without tax (left), with 50% and with 100% tax (right) on the mineral nitrogen input. Proxy for the agricultural rent: shadow price.

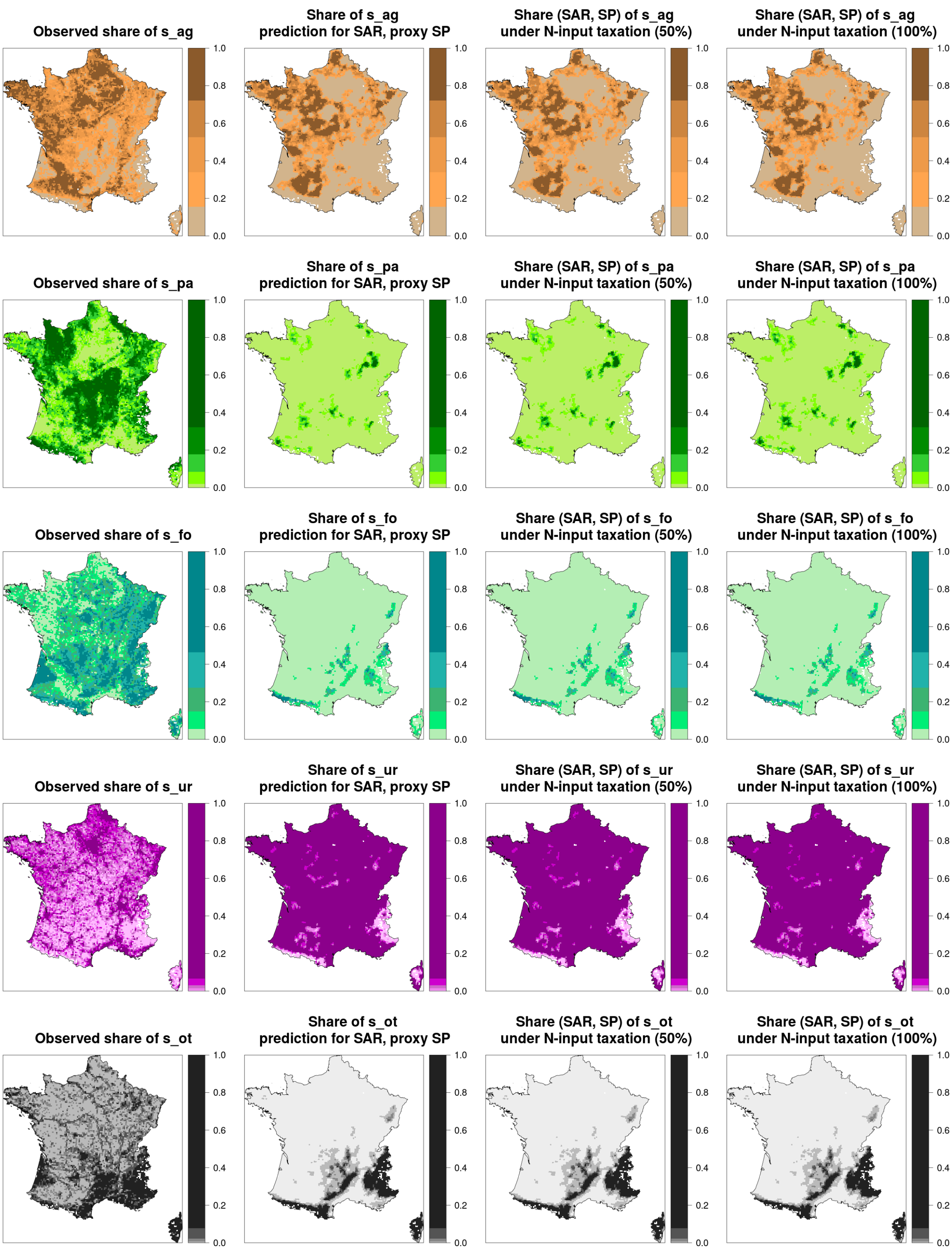

Figure 9 – Observed and predicted (via SAR) land use without tax (left), with 50% and with 100% tax (right) on the mineral nitrogen input. Proxy for the agricultural rent: shadow price.

F Land use shares under CC A2, B1

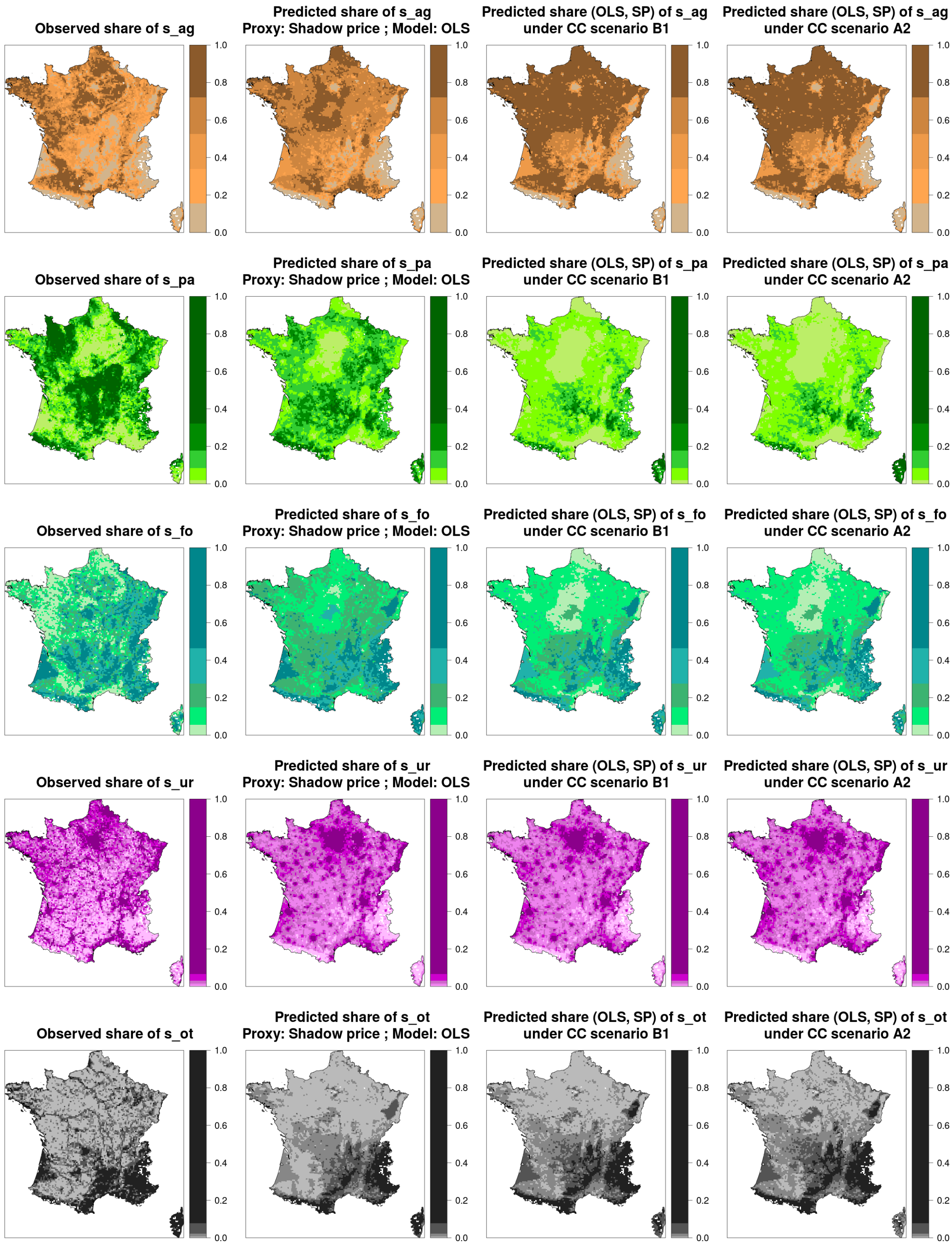

Figure 10 – Observed and predicted (via OLS) land use under CC scenarios B1 and A2. Proxy for the agricultural rent: shadow price.

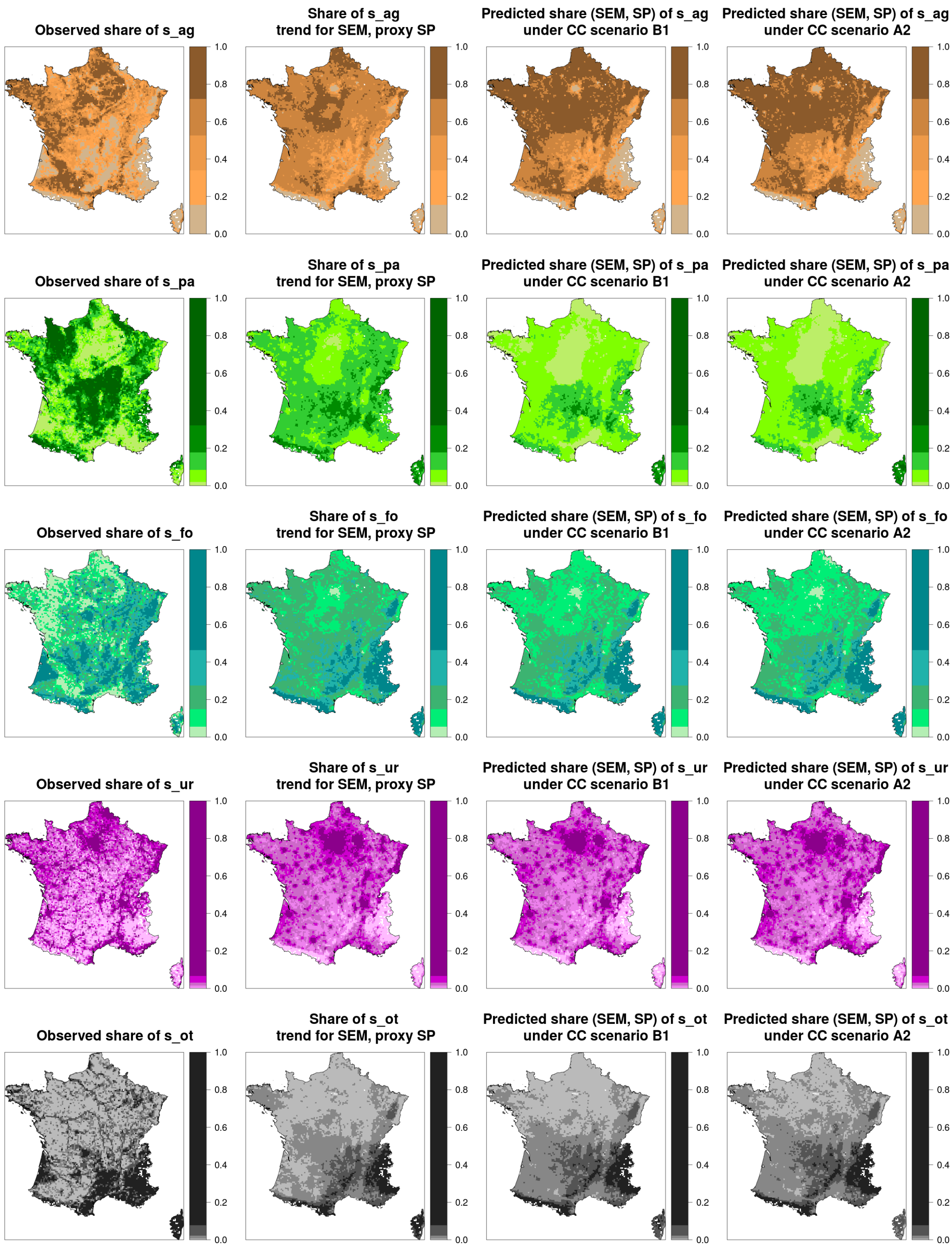

Figure 11 – Observed and predicted (via SEM) land use under CC scenarios B1 and A2. Proxy for the agricultural rent: shadow price.

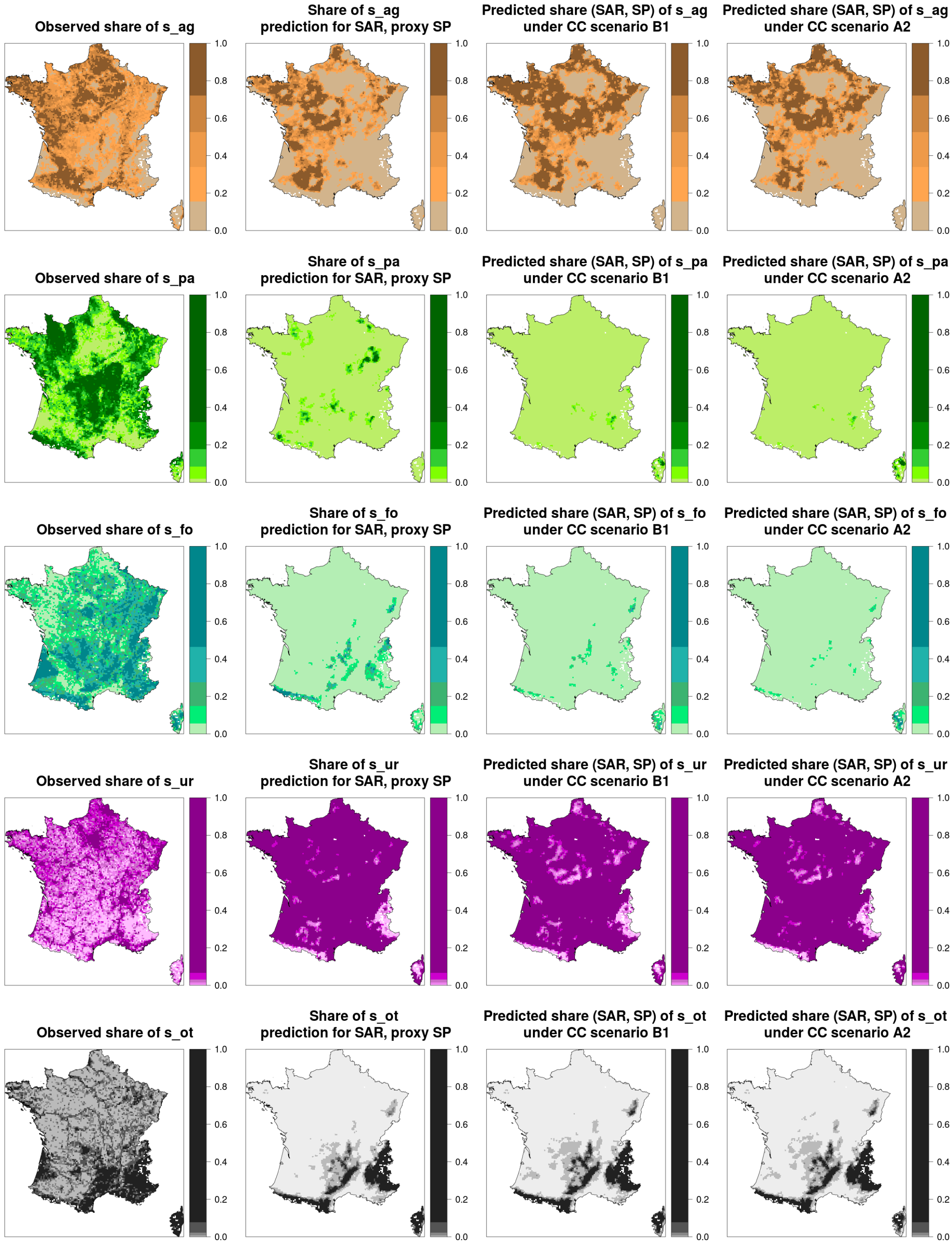

Figure 12 – Observed and predicted (via SAR) land use under CC scenarios B1 and A2. Proxy for the agricultural rent: shadow price.

G Land use shares under CC A2, B1, N tax

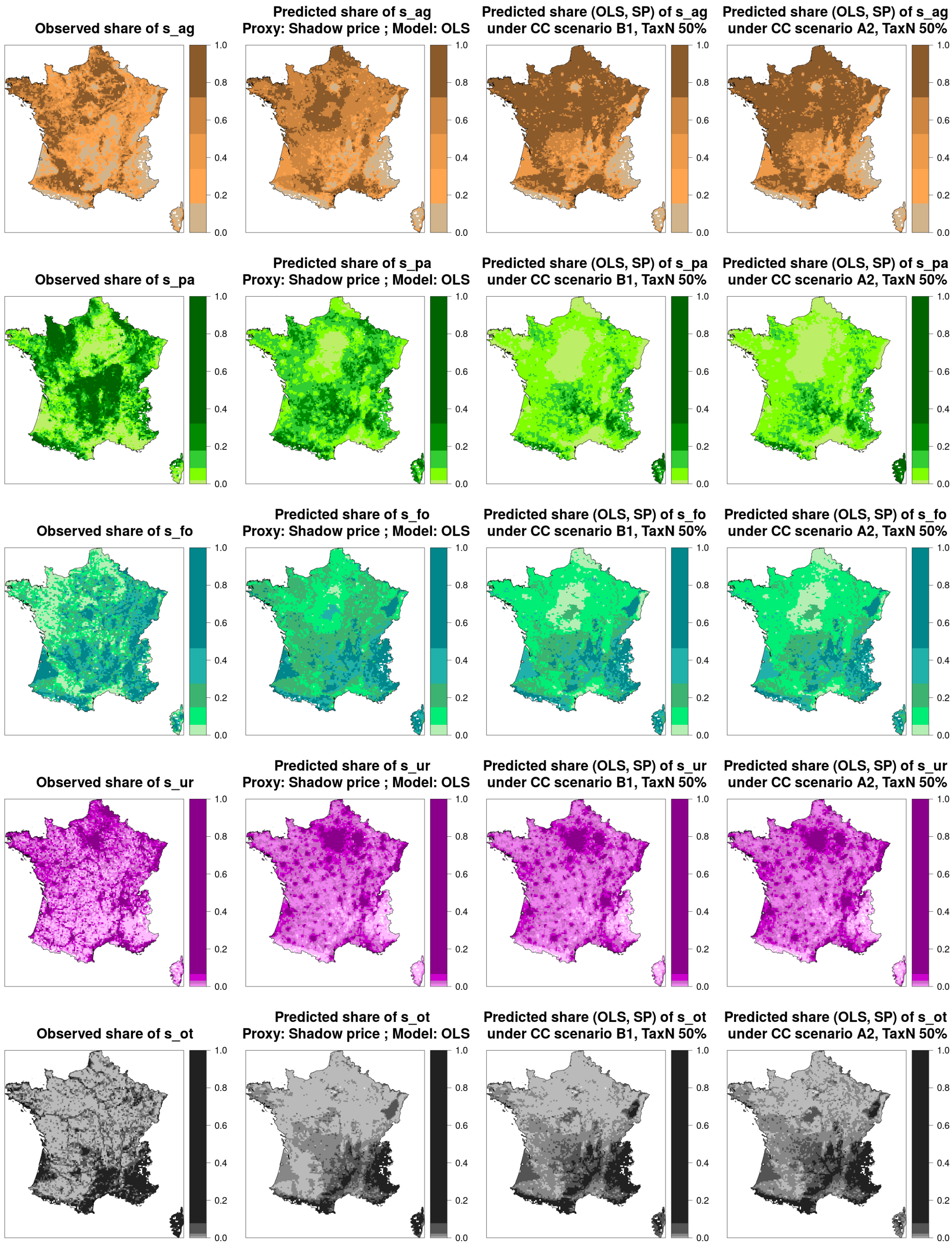

Figure 13 – Observed and predicted (via OLS) land use under CC scenarios B1 and A2 and 50% N tax. Proxy for the agricultural rent: shadow price.

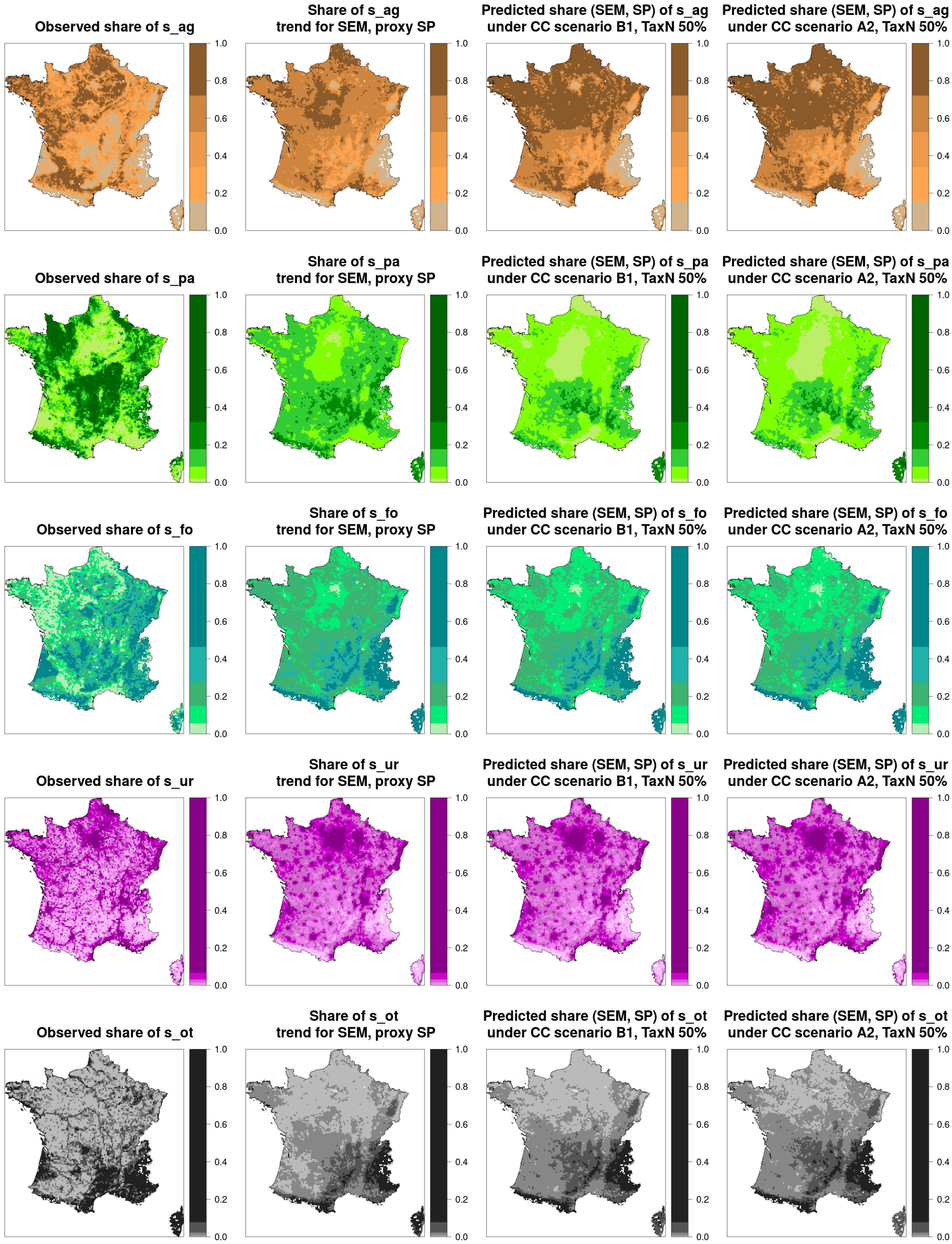

Figure 14 – Predicted (via SEM) land use under CC scenarios B1 and A2 and 50% N tax. Proxy for the agricultural rent: shadow price.

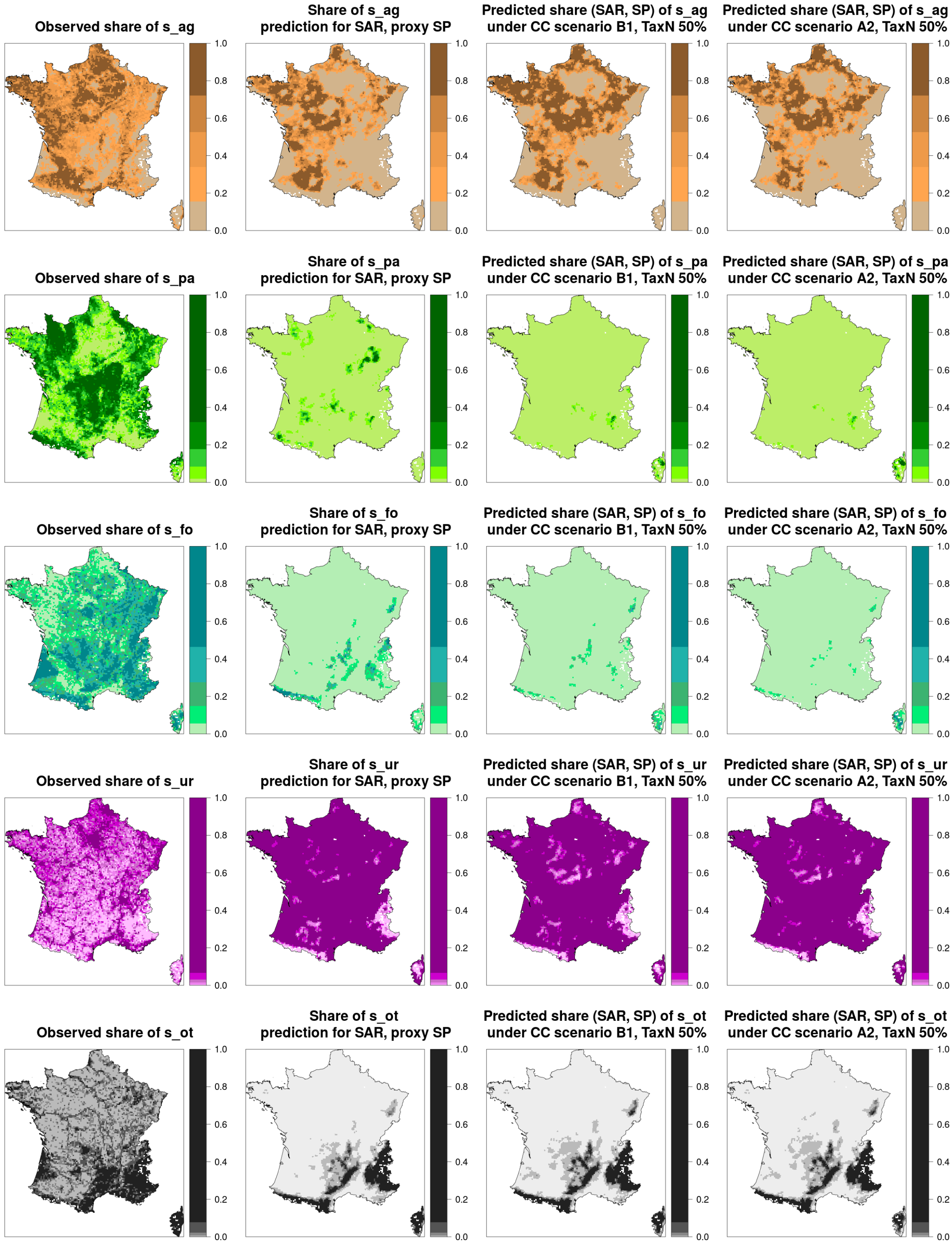

Figure 15 – Observed and predicted (via SAR) land use under CC scenarios B1 and A2 and 50% N tax. Proxy for the agricultural rent: shadow price.

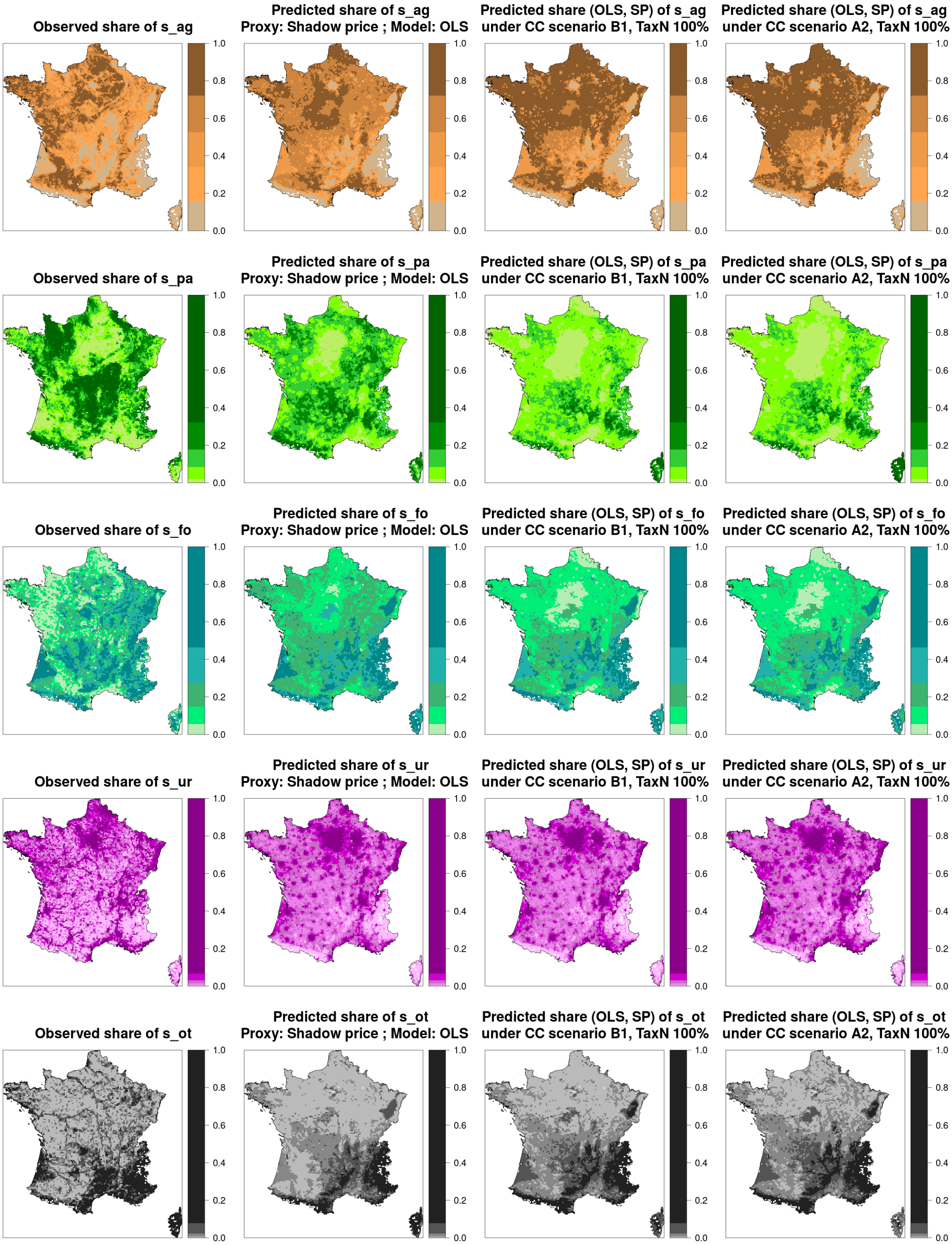

Figure 16 – Observed and predicted (via OLS) land use under CC scenarios B1 and A2 and 100% N tax. Proxy for the agricultural rent: shadow price.

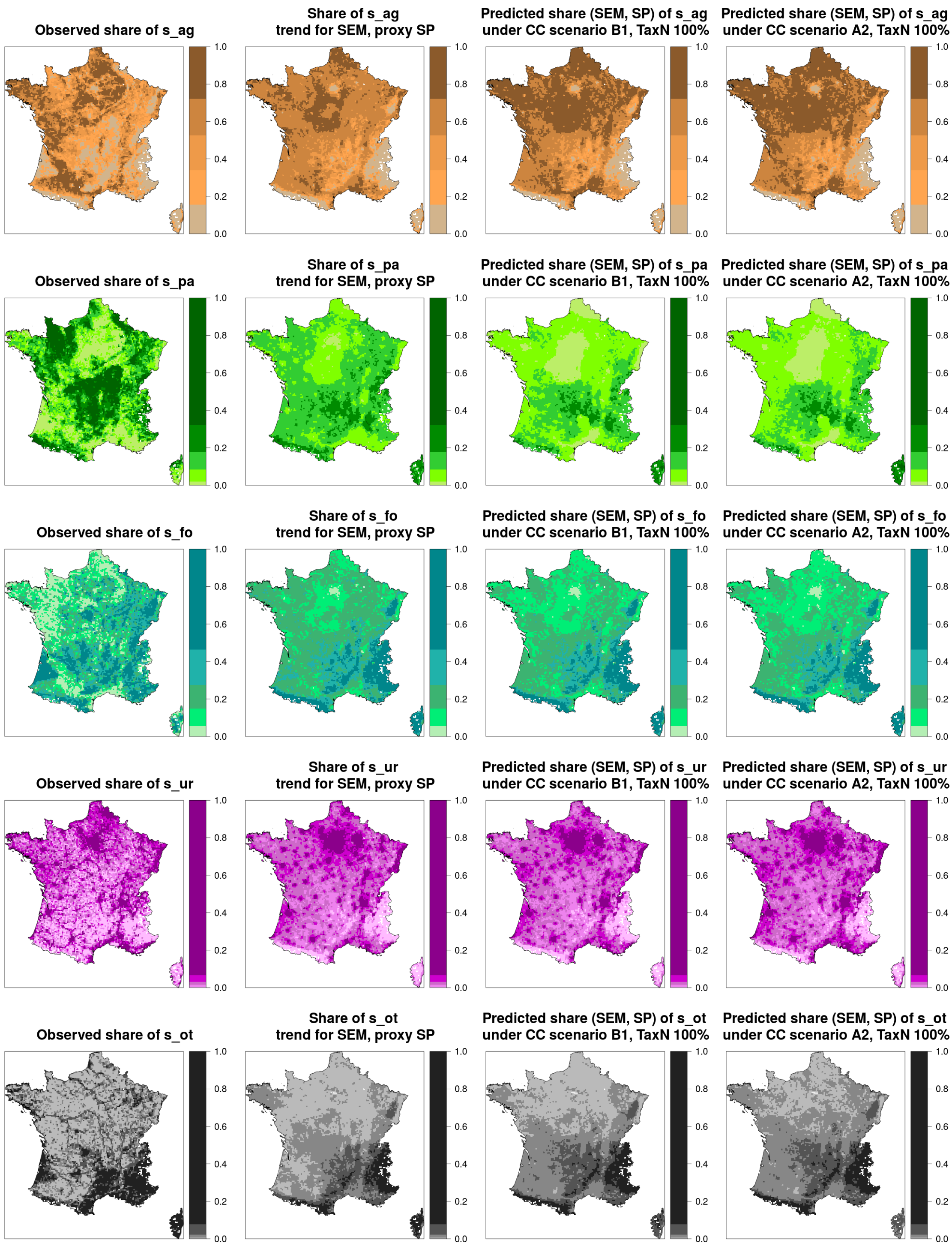

Figure 17 – Observed and predicted (via SEM) land use under CC scenarios B1 and A2 and 100% N tax. Proxy for the agricultural rent: shadow price.

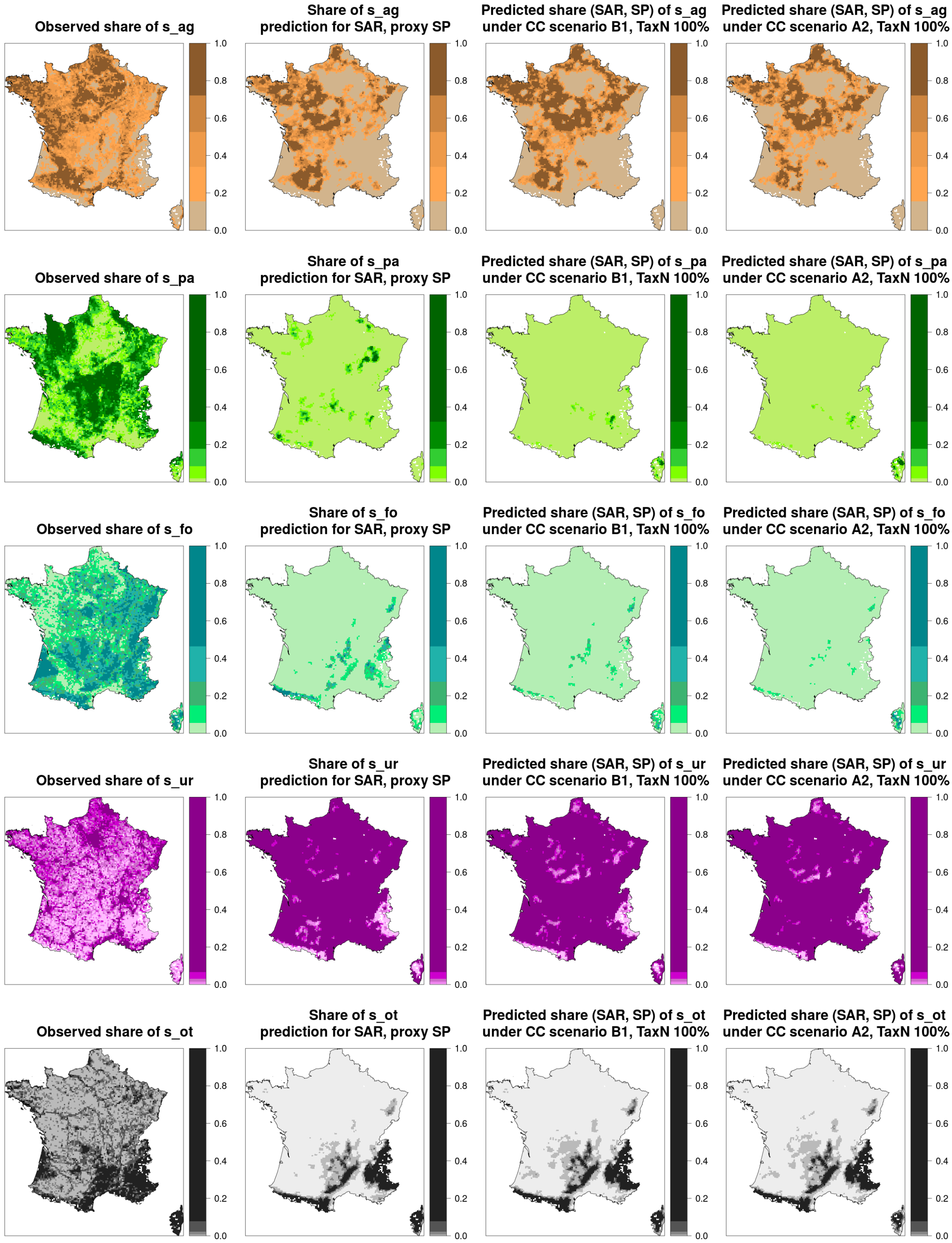

Figure 18 – Observed and predicted (via SAR) land use under CC scenarios B1 and A2 and 100% N tax. Proxy for the agricultural rent: shadow price.

Conclusion générale

Le changement d'allocation des sols est un processus global dont les implications sont multiples et diverses. Les effets des décisions prises à l'échelle de la parcelle se font ressentir à la fois au niveau local et mondial. Par exemple, le choix de la culture et de la quantité d'engrais appliquée au champ a des conséquences sur le changement climatique mais aussi sur l'état écologique des masses d'eaux associées. L'agriculture et le secteur forestier sont impliqués dans des actions de réduction des gaz à effet de serre et, simultanément, ils doivent s'adapter aux nouvelles conditions météorologiques. Les politiques publiques visant une de ces deux dimensions ont forcément un impact sur l'autre. Comme les travaux présentés dans cette thèse de doctorat le démontrent, une taxe sur les engrais azotés entraînerait non seulement l'abattement des émissions polluantes à l'hectare, mais également la baisse de la surface agricole totale suite à la diminution de sa rentabilité. Néanmoins, pour bien appréhender les conséquences d'une telle politique, il est nécessaire de mener les évaluations aux échelles auxquelles ses effets peuvent se manifester. Ceci permettrait d'identifier de possibles synergies entre les différents objectifs des politiques publiques ainsi que de potentiels effets négatifs jusqu'alors méconnus.

Dans les dernières décennies, des disparités régionales dans le changement d'usage des sols se sont révélées. Alors que les surfaces agricoles ont diminué en Amérique du Nord et d'Europe occidentale, les pays tropicaux ont vu leurs forêts progressivement disparaître (Forster et al., 2007). En France par exemple, la surface agricole a été réduite de 20% au cours des 50 dernières années, dont 7 points au profit de l'urbain et de l'infrastructure et 13 au profit de la forêt (SAFER, 2013a). À l'opposé, entre 1990 et 2012, la déforestation au Brésil a consommé presque 60 millions km^2 de la sylve, soit plus que la superficie de la France métropolitaine (FAO, 2015). Les préoccupations qui en découlent sont, pour les uns, l'affaiblissement de la sécurité alimentaire et, pour les autres, la menace de disparition des écosystèmes parmi les plus divers au monde. Ces disparités régionales s'expliquent par de nombreux facteurs socio-économiques : la concurrence internationale sur les marchés des matières premières, la croissance économique des pays émergents, les changements structurels dans les économies nationales et dans le paysage réglementaire, etc. Les perspectives d'évolution de ces différents déterminants restent incertaines, sans même parler de la difficulté à prévoir le progrès technique.

L'approche pour la modélisation de l'allocation des terres proposée ici se concentre sur les effets du changement climatique à l'horizon 2100 en prenant en compte un certain nombre d'options d'adaptation des secteurs agricoles et forestiers ainsi que la répartition actuelle de l'espace entre les principaux usages des sols. Inévitablement, beaucoup d'incertitudes et de biais sont introduits lorsque des prévisions sont réalisées à un horizon aussi lointain. Les modèles climatiques, les deux modèles biophysiques (cultures et forêts), les deux modèles économiques sectoriels, l'estimation économétrique des usages, ont tous leurs imperfections. Cependant, ce sont des modèles spécialisés, développés en adéquation avec l'état de l'art de leur domaine respectif. Ils offrent de larges possibilités de simulations dans lesquelles des facteurs tels que les évolutions en matière de variétés culturales ou de pratiques agricoles peuvent être intégrées. Les modèles économiques, au vu de leur paramétrisation avancée, permettent de tester les effets de changements conjoncturels comme les évolutions de la demande à travers des signaux prix ou des politiques publiques relatives aux secteurs.

Les travaux présentés dans cette thèse de doctorat confirment la nécessité de revisiter les théories économiques fondatrices au vu des nouveaux enjeux. Par exemple, la notion de rente agricole est largement exploitée dans la littérature sur le changement climatique alors que sa définition et sa détermination dans le présent restent ambiguës. Cela ouvre des perspectives en recherche théorique sur une bonne définition de la rente agricole dans les études empiriques des effets du changement climatique sur les activités humaines.

Le travail au sein du projet ORACLE a ouvert de nombreuses possibilités de collaborations interdisciplinaires sur le changement d'usage des sols. La future allocation des terres est une des variables d'intérêt pour les modèles climatiques et hydrologiques. L'interaction entre les modèles économiques AROPAj et FFSM++

peut également être améliorée grâce au modèle économétrique d'usage des sols. Ainsi, la réalisation de plusieurs simulations à un pas de temps donné permettrait de mieux tenir compte des effets du changement d'usage des sols sur la profitabilité des deux secteurs. Par ailleurs, de nombreux scénarios de politiques publiques concernant les deux secteurs sont intéressants à évaluer. Dans un premier temps, nous avons fait le choix d'étudier les pâturages comme un usage à part. Cependant, les systèmes d'élevage sont intimement liés aux autres activités agricoles. AROPAj nous permet de mener l'analyse sur l'allocation des terres entre cultures et prairies en tenant compte de cette interdépendance. Une autre voie d'exploration est l'étude avancée des liens entre agriculture et ressource en eau dans le contexte du changement climatique.

Les sciences actuelles ne fournissent pas et ne fourniront sûrement jamais les réponses à toutes les questions qui interpellent l'humanité. Le monde est fait de telle sorte que les solutions Pareto optimales sont rares voire non-existantes, surtout lorsque le genre humain dans son ensemble est impliqué. Certes, une meilleure compréhension des processus naturels et sociaux est extrêmement utile comme appui à la prise de décision publique. Néanmoins, l'arbitrage entre intérêts divergents ne sera jamais facile politiquement. Pour assurer le futur de notre monde au vu des nombreux défis environnementaux, une forte volonté de la part des décideurs et des institutions publics est nécessaire. Elle va de pair avec un large consensus autour d'une vision commune du monde dans lequel nous souhaitons vivre demain.

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Abstract The research work presented in this doctoral thesis is devoted to the study of land use. The question is examined from two angles : i) land use within the farm (the choice of crops, pastures), and ii) land use between economic sectors (forests, urban, agriculture, etc.). Two methods were employed : mathematical programming models for the agriculture and forestry sectors and econometric methods. The supply-side agricultural model, AROPAj, allows us to model farmers' decision in terms of crops and nitrogen input quantities. Since its economic agents are profit maximizers, the results from the model are directly forwarded to an econometric land use model integrating the returns for the other land demanding sectors. Three case studies are proposed. In the first one, we prove that the choice of crops should be taken into account when evaluating the economic and environmental impacts of an input tax on fertilizers. The second case study is focused on agricultural land prices as the result from the competition among the different land uses, namely field farming and pastures, viticulture, urban and tourism. As land prices and agricultural revenues are often used as proxies for the agricultural rent in the econometric land use models, in the third case study we compare the results of econometric models when these values and the estimates from the agricultural model are employed as explanatory variables. The combined use of these two modeling methods can be valuable for the study of the climate induced land use change. Furthermore, the agricultural model allows us to simulate multiple public policy scenarios.

Résumé Cette thèse de doctorat met au centre de la recherche le problème de l'allocation des terres selon deux perspectives : i) au sein de l'usage agricole, entre différentes activités (cultures, prairies), et ii) entre secteurs économiques (agriculture, forêt, urbain, etc.). Deux méthodes sont mobilisées : des modèles de programmation mathématique sectoriels pour la forêt et l'agriculture et des méthodes économétriques. Le modèle d'offre agricole européenne, AROPAj, permet d'étudier d'une manière très fine les décisions des agriculteurs en matière de choix des cultures et d'intrants (intrants azotés principalement). Puisque les agents économiques pris en compte dans le modèle cherchent à maximiser leur profit, les résultats obtenus sont directement utilisables dans des modèles économétriques d'allocation des terres intégrant la rentabilité des autres secteurs demandeurs en terres. Trois cas d'étude sont proposés. Dans le premier cas, on démontre l'importance de la prise en compte du choix des cultures lorsqu'une taxe sur l'apport azoté est introduite. Le deuxième cas d'étude est centré sur les prix des terres agricoles résultant de la concurrence entre les différents usages, à savoir les grandes cultures et les prairies, la viticulture, l'urbain et le tourisme. Comme les prix de la terre et les revenus agricoles sont souvent utilisés pour approximer la rente agricole dans les modèles économétriques d'allocation des terres, le troisième cas d'étude porte sur la comparaison des modèles économétriques dans lesquels ces données et les sorties du modèle économique servent de variables explicatives. L'emploi combiné de ces deux méthodes peut être utilisé pour l'étude des effets du changement climatique sur l'allocation des terres. Par ailleurs, le modèle économique permet aussi de tester des différents scénarios de politiques publiques.