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Les politiques publiques au défi de la biodiversité : modèles et scénarios bio-économiques pour une agriculture durable

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Production scientifique

1. Articles de recherche

Articles publiés dans des revues internationales avec des comités de lecture

Mouysset L, Doyen L, Jiguet F, 2012. Different policy scenarios to promote various targets of biodiversity. Ecological Indicators 14, 209-221.

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Articles en révision

Mouysset L, Miglianico M, Makowski D, Jiguet F, Doyen L, Revision. Dynamic models for bird community in farming landscapes. Ecological Modelling.

Articles soumis

Mouysset L, Doyen L, Jiguet F, Submitted in Ecological Economics. A double dividend of biodiversity in agriculture.

Mouysset L, Doyen L, Jiguet F, Submitted in Ecological Applications. How do the economic risk aversion affect the biodiversity ? Working Paper dans Les cahiers du GREThA 03, http ://cahiersdugretha.u-bordeaux4.fr/2012/2012-03.pdf.

Articles en préparation

Mouysset L, Doyen L, Jiguet F. In prep. Co-viability of farmland biodiversity and agriculture.

Articles publiés dans les actes de conférences (proceedings)

Mouysset L, Doyen L, Jiguet F, 2010. Risk aversion impact on bio-economic performances in an agricultural public policy context : a modeling approach. The 11th Biennial Conference of the International Society of Ecological Economics (ISEE), Breme-Oldenburg.

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2. Articles à vocation de transfert

Mouysset L, Doyen L, Jiguet F, Allaire G, Léger F, Juin 2011. Partnering biodiversity and income on French Farmlands. Article de communication par le Forum Suisse de la Biodiversité IBS Science and Policy.

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3. Participation à des congrès

Mouysset L, Doyen L, Jiguet F, Ecological responses to economic public agricultural policies in the French farmlands, The 25th International Congress of Conservation Biology, Auckland, Décembre 2011. Regular talk.

Mouysset L, Doyen L, Jiguet F, Ecological responses to economic public agricultural policies in the French farmlands, The 5th World Congress for Conservation Agriculture et The 3rd Farming System Design Congress, Brisbane, Septembre 2011. Regular talk.

Mouysset L, Doyen L, Jiguet F, How do the farmer risk preference affect the biodiversity ?, The 13th International BIOECON Conference, Genève, Septembre 2011. Regular talk.

Mouysset L, Doyen L, Jiguet F, Réponses écologiques aux scénarios de politiques publiques en milieu agricole : une approche bio-économique. Congrès Ecologisation des politiques publiques agricoles, Avignon, Mars 2011 (organisé par l'unité Ecodéveloppement, INRA). Regular talk.

Mouysset L, Doyen L, Jiguet F, Risk aversion impact on bio-economic performances in an agricultural public policy context : a modeling approach. The 11th Biennial Conference of the International Society of Ecological Economics, Breme-Oldenburg, Août 2010. Regular talk.

Mouysset L, Doyen L, Jiguet F, Allaire G, Léger F, Innovation rigidity and ecological-economic reconciliation in agriculture. Conference for the Innovation and Sustainable Development in Agriculture and Food (organisé par le CIRAD), Montpellier, Juillet 2010. Short talk.

Mouysset L, Doyen L, Jiguet F, Allaire G, Léger F, Bio-economic modeling for sustainable management of biodiversity and agriculture. Congress for Ecological Engineering from Concepts to Applications (organisé par Gaié, groupe d'application d'ingénierie pour les écosystèmes), Paris, Décembre 2009. Poster.

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Chapitre I

Présentation d'ensemble

Préambule

Le manuscrit s'organise autour de sept chapitres. Le chapitre I, en français, constitue une synthèse du travail mené au cours de ce doctorat. Il est composé d'une introduction générale, d'éléments méthodologiques (modèles et cas d'étude), d'une synthèse des principaux apports de la thèse issus des articles et d'une discussion générale. Les autres chapitres et les annexes, en anglais, constituent chacun un article. Leur lecture apportera davantage de détails méthodologiques, de résultats et de discussion spécifique.

1. Cadre et enjeux de l'étude

"*La biodiversité, une assurance-vie pour notre monde en changement*", tel est le slogan de la Journée internationale de la biodiversité mise en place par les Nations Unies le 22 mai 2005. Cela fait maintenant quelques années que la biodiversité n'est plus uniquement perçue sous le prisme de la conservation de la Nature pour elle-même ou de la protection de certaines espèces emblématiques. Les sociétés humaines ont pris conscience de l'importance des interactions existant entre leur bien-être et la biodiversité. L'expansion du concept de service écosystémique dans le vocabulaire scientifique aussi bien que dans les débats sur l'environnement est un signe manifeste de cette prise de conscience. La création de l'Intergovernmental Science Policy Plateform on Biodiversity and Ecosystem Services (IPBES) en 2010 confirme cette tendance.

1.1. De la conservation à la gestion durable de la biodiversité

1.1.1. La biodiversité : un enjeu écologique

La définition et la compréhension de la biodiversité représentent des enjeux majeurs conditionnant les travaux et les réflexions menés sur le sujet (Golley, 1991). Réfléchir à cette définition est d'autant plus nécessaire que la perception de la diversité du vivant a largement évolué au cours du temps. Initialement, elle n'était évaluée qu'au travers du nombre d'espèces. Cette approche fut cependant rapidement confrontée au caractère inaccessible que constitue le projet d'un grand inventaire tel que le pensait Linné. Cette difficulté permit de faire émerger un premier défi relatif à la conservation de la biodiversité, encore d'actualité : la description et la gestion de la biodiversité doivent se faire à partir d'une connaissance inévitablement partielle de sa réalité.

L'apparition de la pensée évolutionniste avec Charles Darwin (Darwin, 1872) a généré une avancée significative pour la compréhension de la biodiversité en mettant en évidence l'aspect dynamique du vivant. Cela a notamment permis de comprendre qu'il existait une diversité intra-spécifique dont découle la diversité inter-spécifique. Plus tard, la découverte des gènes éclaira cette diversité intra-spécifique avec la mise en évidence de la diversité génétique. Par conséquent, l'inventaire des espèces comme unique mesure de la diversité n'était plus satisfaisant. Le problème de la métrique de la biodiversité ainsi posé constitua alors un deuxième défi, auquel nous sommes toujours confrontés aujourd'hui.

De manière assez indépendante de la pensée évolutionniste, l'émergence de l'écologie, et plus spécifiquement de l'écologie fonctionnelle, proposa la notion d'écosystème (Tansley, 1935). L'ensemble des populations de différentes espèces vivant au même endroit constitue une biocénose, qui, associée au milieu physique, forme un écosystème. Ce courant souligna l'importance de la fonction des espèces et de leurs interactions au sein de l'écosystème (Lindeman, 1991). Ainsi, en plus du nombre d'espèces, d'autres caractéristiques comme le nombre d'individus par espèces, la place dans la chaine alimentaire ou le degré d'inter-dépendance entre deux espèces apparaissent essentielles pour décrire l'état de santé de l'écosystème, considéré maintenant comme la nouvelle dimension clef de la biodiversité. C'est ce système biologique intégré et original, formé d'espèces adaptées et inter-dépendantes, qu'il apparaît pertinent de conserver.

C'est dans ce cadre écosystémique que se place notre travail, notamment sous la définition de la biodiversité proposée par la Convention de la Diversité Biologique en 1992 (Article 2) : "*La variabilité des organismes vivants de toute origine y compris, entre autres, les écosystèmes terrestres, marins et autres écosystèmes aquatiques et les complexes écologiques dont ils font partie ; cela comprend la diversité au sein des espèces et entre espèces ainsi que celle des écosystèmes.*"

1.1.2. La biodiversité : un enjeu pour les sociétés humaines

Ces dernières décennies, les milieux naturels ont été largement modifiés à l'échelle mondiale. Le développement des zones urbaines, l'intensification de l'agriculture, la sur-pêche, les pollutions, le réchauffement climatique, le développement d'espèces invasives sont autant de perturbations environnementales auxquelles la biodiversité doit faire face. La vitesse à laquelle ont lieu ces changements, entraînant une mise en péril rapide d'habitats spécifiques, ne laisse guère le temps aux espèces de s'adapter. Ces changements globaux ont donc un impact direct sur la biodiversité. Le nombre d'espèces actuellement en déclin ou déjà disparues à cause de la pression exercée par l'Homme est grandissant : Flowerdew & Kirkwood (1997) l'illustrent pour les mammifères, Sotherton & Self (2000) pour les arthropodes et les plantes, Donald *et al.* (2001) pour les oiseaux. L'indice de la planète vivante développé par le World Wildlife Fund (McRae *et al.*, 2008) résume cette érosion et montre une diminution moyenne de 30% en 35 ans sur l'ensemble des espèces (fig. I.1).

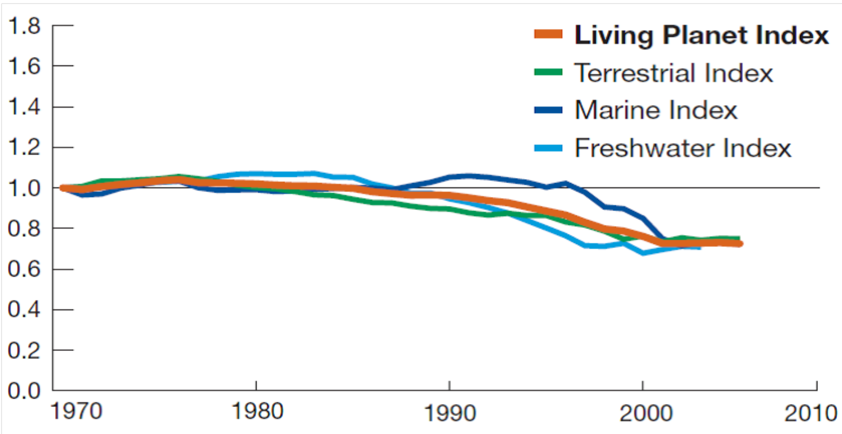


FIGURE I.1 – Indices de la planète vivante (Living Planet Index) sur la période 1970-2005 agrégés pour l'ensemble des espèces (orange), les espèces terrestres (vert), les espèces marines (bleu foncé) et les espèces d'eau douce (bleu clair) (source McRae *et al.* (2008)).

Or, nous savons maintenant que cette érosion a d'importantes conséquences sur les sociétés humaines puisque celles-ci en sont dépendantes à bien des niveaux (Pearce & Moran, 1994) :

- Une partie des activités humaines découle en effet de manière directe de la biodiversité (valeur d'usage direct), c'est le cas de la pêche ou de l'exploitation du bois.
- Mais la biodiversité peut aussi protéger ou entretenir des activités humaines de manière indirecte (valeur d'usage indirect) (Tilman *et al.*, 2005, Tilman, 1999) : la régulation des nuisibles des cultures par certains oiseaux, ou le renouvellement des sols par la micro-faune contribuent à ces valeurs.
- De futures découvertes réalisées par notre génération (valeur d'option) (Béné & Doyen, 2008, Arrow & Fisher, 1974, Henry, 1974, Weisbrod, 1964) ou par les prochaines générations (valeur d'héritage) (Krutilla, 1967) pourraient renforcer l'importance pour l'Homme de certaines espèces aujourd'hui mal connues voire inconnues. La découverte permanente de nouveaux médicaments dérivés des plantes ou des insectes de la forêt équatoriale illustre la valeur potentielle de la biodiversité. Les fonctions culturelles de la biodiversité (beauté d'un paysage, espèce emblématique) font également partie de ces valeurs.

Ainsi, même d'un point de vue uniquement utilitariste, la protection de la biodiversité est directement nécessaire au bien-être des sociétés humaines et représente un enjeu majeur. Les points de vues non utilitaristes ou attribuant directement une valeur intrinsèque à la biodiversité, conduisent à la même conclusion.

1.1.3. La biodiversité : un enjeu économique

Alors que la protection de la biodiversité apparaît aujourd'hui nécessaire, elle n'a généralement pas lieu spontanément. En effet, la biodiversité est un bien économique particulier dont l'utilisation peut relever du bien commun ou du bien public. Certaines de ses fonctionnalités, comme la fixation du carbone par la forêt ou la pollinisation par les insectes, la classent dans la catégorie des biens publics : quand le bien (la forêt par exemple) est présent, tout le monde profite de sa présence au travers de ses fonctions (fixation du CO2 atmosphérique) sans qu'il y ait de compétition entre les agents. Ce bien est donc non exclusif et sans rivalité. D'autres de ses fonctionnalités (comme la disponibilité en poissons) font davantage partie de la classe des biens communs. A nouveau, quand le bien (le banc de poissons) est présent, tous les agents (pêcheurs) peuvent l'utiliser mais de manière compétitive. Ce bien présente les deux caractéristiques de non exclusion et de rivalité.

Ces deux types de biens sont soumis respectivement à deux problèmes : le passager clandestin et la tragédie des communs. Le passager clandestin (Olson, 1935), concept issu de la théorie des jeux, illustre l'idée que puisque tous les agents peuvent profiter du bien lorsqu'il est présent, il est plus intéressant pour l'agent de ne pas payer pour sa mise en place. Il peut donc jouir du bien sans avoir de sur-coût. Ce comportement conduit à une carence d'investissements et donc à une sous-production du bien. La tragédie des communs (Hardin, 1968) repose sur l'idée que comme il y a compétition entre les agents pour l'utilisation du bien commun, chacun a intérêt à maximiser son exploitation pour se constituer des réserves, entraînant alors sa sur-exploitation. Ainsi, en étant à la fois sous-produite et sur-exploitée, la biodiversité ne peut pas être maintenue durablement sans régulation extérieure.

Nous pouvons traduire cette conclusion d'un point de vue micro-économique par l'assimilation de la biodiversité à une externalité (Meale, 1952). Autrement dit, la vérité des coûts ne s'impose pas aux agents, qui ne paient pas leur consommation de biodiversité. Il est alors nécessaire d'internaliser cette externalité, c'est-à-dire d'intégrer cette biodiversité dans les prises de décision en corrigeant la distorsion entre le coût supporté par la biodiversité et le coût payé par le consommateur. Généralement l'intervention de l'agence de régulation est requise (Pearce, 1976). Elle dispose de trois grands types d'instruments (Weitzman, 1974) :

- Les instruments prix sont des taxes ou des subventions qui vont changer artificiellement la valeur marchande des biens (Pigou, 1920).
- Les instruments réglementaires correspondent à une norme exogène fixée par l'État qu'il est interdit de dépasser sous peine de sanction (exemple des quotas de pêche).
- L'instrument informatif, au travers de labels certifiés par l'État, informe le consommateur sur la qualité des biens mis à sa disposition, justifiant ainsi d'éventuelles différences de prix (cas des éco-labels).

Ces trois outils ont pour but d'orienter le choix des agents en fonction d'objectifs ou de contraintes que l'agence s'est fixés.

Cependant quel que soit l'outil choisi, l'établissement d'une politique visant à protéger la biodiversité engendre des coûts : coût de l'élaboration et de la mise en place de la politique, coût provenant du manque à gagner des agents économiques soumis à cette politique et potentiellement limités dans leur production. Si la protection de la biodiversité est nécessaire, une réflexion reste alors à mener sur l'élaboration de la politique publique pour en limiter les coûts, en augmenter l'efficacité et favoriser son acceptation par les agents. La protection de la biodiversité sur le long terme nécessite donc de concilier objectifs environnementaux et objectifs économiques.

1.1.4. La biodiversité : un enjeu politique

C'est dans cette perspective qu'a été signée la Convention pour la Diversité Biologique (CDB) au Sommet de la Terre de Rio organisé par le Programme des Nations Unies pour l'Environnement (PNUE) en

1992. La biodiversité a été déclarée "*préoccupation commune de l'humanité*". L'article premier comporte trois objectifs : sa conservation, son usage durable et le partage juste et équitable des avantages tirés de l'exploitation des ressources génétiques. Une large réflexion à l'échelle mondiale a été menée avec l'élaboration du Millenium Ecosystem Assessment (MEA, 2005) afin de comprendre l'impact de l'Homme sur les écosystèmes et de prioriser les actions à entreprendre pour leur conservation. Cette étude multi-disciplinaire cherche à coupler questions écologiques, économiques et sociales. Cependant, la dixième Conférence des Parties s'est ouverte à Nagoya en octobre 2010 sur une inquiétude grandissante quant à la poursuite de l'érosion de la biodiversité. Les dernières Perspectives mondiales de la biodiversité (GBO3, 2010) publiées par le PNUE en 2010 ont en effet prévenu : "*la perte massive de la biodiversité est de plus en plus probable, et entraînerait avec elle une forte réduction du nombre de services essentiels fournis aux sociétés humaines puisque plusieurs points de basculement sont près d'être franchis, conduisant les écosystèmes vers des états moins productifs desquels il pourrait être difficile voire impossible de revenir*". C'est ainsi que, pour stimuler l'intégration d'objectifs de biodiversité dans l'élaboration des politiques publiques et conduire les politiques de conservation vers des réflexions bio-économiques, a été mise en place la plate-forme intergouvernementale IPBES (Intergovernmental Science Policy Plateform on Biodiversity and Ecosystem Services) à l'image du GIEC (Groupe Intergouvernemental d'experts sur l'Evolution du Climat), approuvée par une résolution des Nations Unies en décembre 2010.

1.1.5. La biodiversité : un enjeu de modélisation

Cependant, la conception et la mise en oeuvre de telles politiques s'avèrent difficiles. L'intégration d'objectifs d'ordres différents (économiques, écologiques, sociaux), la gestion de l'incertitude (des marchés, écologique et climatique), le risque d'irréversibilités écologiques ainsi que des verrouillages sociaux plaident dès lors en faveur d'une large utilisation de modèles et de scénarios. Ces scenarios ne sont pas des prévisions mais des gammes de futurs possibles prenant en compte les diverses incertitudes sous-jacentes.

Les récents travaux de Pereira *et al.* (2010) mettent en évidence l'importance de cette approche par la modélisation pour une gestion durable de la biodiversité. Ils montrent notamment que le développement économique actuel devrait mener à un devenir catastrophique de la biodiversité, même dans le cas des scénarios les plus optimistes. Ils soulignent cependant qu'il existe de vraies opportunités d'intervention bien que les marges de manoeuvre diminuent rapidement. A l'instar des climatologues, l'enjeu réside donc dans l'élaboration de scénarios d'évolution de la biodiversité en réponse à différentes pressions anthropiques (changement climatique, occupation des sols, pollutions) en se fondant sur des modèles socio-économiques (Brook *et al.*, 2008). Ces prospectives imposent alors un effort de recherche inter-disciplinaire. En organisant la coopération scientifique pour proposer des définitions, des indicateurs et des scénarios issus du consensus scientifique et utilisables par les décideurs, la plateforme IPBES devrait jouer un rôle majeur en ce sens. La Fondation pour la Recherche sur la Biodiversité (FRB) stimule, en France, la recherche dans cette direction.

Cette approche par la modélisation se plaçant clairement dans une perspective d'aide ou d'accompagnement de la décision, il existe un vrai enjeu quant à l'utilisation finale des modèles. S'agit-il de décrire un phénomène observé actuellement ou de projeter des tendances futures ? Dans le cadre de scénarisation, la projection est une composante prioritaire. L'enjeu principal est donc la construction d'un modèle suffisamment complexe, pour décrire les systèmes socio-économiques et écologiques étudiés, et calibré avec des données historiques, pour s'ancrer dans le réel, mais dont les capacités de prédiction hors du domaine de calibration sont avérées (Wallach, 2006). Finalement, ces scénarios sont développés pour communiquer sur le devenir de la biodiversité auprès de différents acteurs (scientifiques de différentes disciplines, politiques, industriels, grand public). Pour que le message soit correctement perçu, il est alors impératif de considérer dans les modèles des indicateurs intelligibles par les acteurs impliqués. De nombreux défis méthodologiques conditionnent le développement des modèles bio-économiques comme outil de gestion de la biodiversité.

1.2. Les modèles bio-économiques, un outil pour gérer la biodiversité

Depuis les premiers travaux développés par Clark (1976), les modèles bio-économiques sont confrontés à plusieurs défis et enjeux méthodologiques. Notamment, une vraie réflexion inter-disciplinaire doit être menée (Wätzold *et al.*, 2006) pour comprendre le système dans son ensemble et connecter correctement les processus écologiques et socio-économiques (Perrings, 2002).

1.2.1. Un cahier des charges pour la modélisation

L'élaboration des modèles bio-économiques peut s'articuler alors autour d'un cahier des charges, que nous structurons ici selon quatre grandes caractéristiques :

a - Une approche systémique

L'approche systémique est utilisée pour étudier un système complexe dans sa globalité, notamment en considérant simultanément les éléments de ce système, leurs interactions et leurs dynamiques. Elle se fonde sur des modèles mécanistes, qui reposent sur des hypothèses scientifiques a priori, mobilisant des connaissances sur le phénomène étudié (exemple avec le modèle simple proie-prédateur de Lotka-Volterra ou des modèles de communautés (Ferrière *et al.*, 1996)). Ils modélisent des mécanismes de causalité, à la différence des modèles statistiques qui établissent des corrélations entre différentes variables (exemple avec les modèles d'évolution de niches climatiques (Heikkinen *et al.*, 2006)). L'approche systémique inclut l'intégration de processus dynamiques. Les modèles statiques, qui sont utilisés pour déterminer un ensemble d'actions optimales ou la répartition spatiale optimale de ces actions (Polasky *et al.*, 2008), peuvent être combinés pour étudier des évolutions temporelles, où chaque pas de temps est caractérisé par un modèle statique. Cependant, ils ne permettent pas de caractériser des transitoires et des boucles de rétro-actions. L'utilisation de modèles dynamiques, définis par des équations dont des variables peuvent être elles-mêmes des fonctions explicites du temps, apparaît donc plus cohérente avec une approche systémique (De Lara & Doyen, 2008). La validation statistique et la calibration de ces modèles mécanistes et dynamiques ne doivent cependant pas être négligées.

b - Un système spatialisé

Nous distinguons deux éléments relatifs aux caractéristiques spatiales du modèle. D'une part, la littérature a souligné l'importance de la spatialisation dans les problèmes bio-économiques (Polasky *et al.*, 2008, Naidoo *et al.*, 2006). Il est alors nécessaire de capturer l'hétérogénéité des contextes économiques et environnementaux (au travers de données spatialisées par exemple) et d'expliciter des relations entre deux entités spatiales comme avec des méta-populations (Gill, 1978). D'autre part, le choix des échelles spatiales auxquelles sont réalisés les modèles est déterminant. En effet, les échelles décisionnelles ou économiques sont souvent plus larges que les échelles écologiquement pertinentes. Dans le cas de l'agriculture par exemple, l'échelle régionale apparaît comme une échelle de décision réaliste alors que les travaux en agro-écologie (Tscharntke *et al.*, 2005) ont souligné l'importance de l'échelle paysagère pour gérer la biodiversité. Une approche multi-échelle combinant différentes échelles emboîtées est donc un critère essentiel pour l'élaboration d'un modèle bio-économique global.

c - Un environnement incertain

Les incertitudes sur les états et les mécanismes sont nombreuses dans les écosystèmes. Sous couvert de manque de connaissances et de certitudes scientifiques, elles justifient dans de nombreux cas l'absence ou le retard dans la prise de décision ("wait and see"). Il existe deux types d'incertitude dans la connaissance scientifique : certaines incertitudes, relatives aux données et aux mesures, sont de nature probabiliste, aléatoire ou Bayesienne (c'est le cas de des incertitudes démographiques et environnementales), alors que d'autres (parfois qualifiées de Knightiennes) sont liées à des controverses ou des scénarios (croissance économique, dynamique des prix, scénario climatique). Cela conduit à distinguer incertitude stochastique (ou risque) et ambiguïté (Henry, 1981; 1974). Intégrer de l'ambiguïté dans le modèle est intéressant pour balayer l'ensemble des contextes et pressions pouvant s'exercer sur la biodiversité. Mais prendre en compte l'incertitude stochastique est tout aussi essentiel pour identifier des critères de gestion du risque et assurer une gestion durable de la biodiversité évitant les situations de crise.

d - Un modèle de décision

La gestion de la biodiversité est dépendante de décisions de régulation et d'actions anthropiques, qui sont interprétées comme des variables de contrôle dans les modèles bio-économiques. Typiquement, les niveaux de quota de pêches, les niveaux d'effort de chasse, les subventions agro-environnementales, et la taille des aires protégées peuvent jouer ce rôle dans les dynamiques de la biodiversité. Le choix d'un cadre formel et générique permettant de gérer correctement ces variables de contrôle est donc déterminant pour la pertinence du modèle. La théorie du contrôle des systèmes dynamiques (Clark, 1976) propose un cadre adapté et, via des éléments de théorie des jeux, permet également d'étudier les interactions stratégiques entre agents (coalition, compétition).

Un modèle bio-économique combinant l'ensemble de ces critères nous apparaît être un outil intéressant pour représenter le système complexe qu'est un écosystème, scénariser des tendances d'évolution de la biodiversité en réponse à des pressions anthropiques, et identifier des stratégies de gestion durable.

1.2.2. L'évaluation de la biodiversité

a - De la biodiversité remarquable à la biodiversité ordinaire

Comme l'illustre la définition donnée par la CDB, le terme biodiversité englobe de nombreuses composantes. Nous pouvons notamment distinguer la biodiversité ordinaire de la biodiversité remarquable. Cette dichotomie peut se traduire à l'échelle génétique, spécifique ou écosystémique. A l'échelle génétique, une population locale peut présenter des caractéristiques originales, mais existe aussi la diversité génétique ordinaire entre individus, qui permet globalement l'évolution de l'espèce. Au niveau spécifique, certaines espèces, dites emblématiques, sont associées à un patrimoine culturel fort et de ce fait inscrites sur des listes de protection (liste de l'UICN). Elles sont souvent rares. Inversement, les espèces ordinaires forment un ensemble beaucoup plus abondant et contribuent, par leurs interactions, au maintien des écosystèmes. Enfin, au niveau du paysage, certains territoires peuvent être distingués pour des habitats particulièrement originaux et rares à l'échelle régionale ou nationale (cas des tourbières), ou des paysages remarquables (sites classés). D'un autre côté, une mosaïque d'habitats ordinaires constitue l'essentiel des paysages et est responsable de leurs propriétés fonctionnelles. La biodiversité remarquable étant clairement identifiée, sa protection peut faire l'objet de mesures ciblées (aires de protection, espèces protégées). Alors que les premiers travaux ont naturellement porté sur des espèces emblématiques en danger (c'est notamment le cas de l'Outarde canepetière *Tetrax tetrax* (Bretagnolle *et al.*, 2011, Silva *et al.*, 2004, Wolff *et al.*, 2001)), l'importance de la biodiversité ordinaire est aujourd'hui de plus en plus soulignée pour mettre en place une gestion durable et globale de la biodiversité et des services écosystémiques (Chevassus-Au-Louis, 2009). C'est dans ce cadre de l'évaluation au travers de la biodiversité ordinaire que nous nous positionnons, notamment en développant une approche multi-espèce.

b - Des indicateurs de biodiversité

De nombreux indicateurs de biodiversité sont disponibles dans la littérature (Levrel, 2007, Magguran, 1988). Afin de bien comprendre l'impact des décisions économiques sur la biodiversité, le jeu d'indicateurs choisis doit balayer diverses caractéristiques de la communauté, comme par exemple sa structure, sa qualité et sa taille (Barbault & Chevassus-Au-Louis, 2004). Cependant, pour faciliter l'analyse, il est nécessaire de réduire au minimum le nombre d'indicateurs, tout en évitant les indicateurs trop composites qui ont tendance à moyenner (une même valeur de l'indicateur peut provenir de la somme de deux combinaisons très différentes de variables). Par ailleurs, certains indicateurs présentent une légitimité institutionnelle : ils ont été choisis comme indicateurs de référence par une agence publique et sont donc maintenant largement renseignés dans différents contextes et/ou depuis plusieurs années, ce qui offre de nombreux points de comparaison. C'est par exemple le cas du Farmland Bird Index choisi par l'Union Européenne pour étudier les changements structurels de la biodiversité en réponse aux pressions de l'agriculture (Balmford *et al.*, 2003). Ces indicateurs officiels ont un fort pouvoir communicatif. Il nous apparaît alors nécessaire de réfléchir à une combinaison adéquate d'indicateurs officiels et d'indicateurs non institutionnalisés pour décrire efficacement la biodiversité tout en gardant un pouvoir communicatif.

c - Un prix pour la biodiversité ?

Une approche classiquement utilisée en économie de l'environnement consiste à monétariser la biodiversité afin de l'intégrer dans les décisions économiques et internaliser l'externalité. Il existe plusieurs techniques de monétarisation (Pearce, 1976) comme la méthode des coûts de transport (Maille & Mendelsohn, 1993, Smith & Kaoru, 1990) ou l'évaluation contingente (Hanemann, 1994, Davis, 1963). Cependant, cette approche de monétarisation n'est pas satisfaisante dans le cas de la biodiversité. Elle se heurte à de nombreux problèmes à la fois techniques et conceptuels (Chevassus-Au-Louis, 2009). Aucun consensus sur l'évaluation monétaire de la biodiversité non exploitée (situation différente pour les ressources exploitées comme de nombreuses espèces de poissons) n'a émergé à l'heure actuelle. Face à l'absence de prix et de marché, nous n'avons donc pas choisi la direction de monétarisation pour aborder le problème de la gestion de la biodiversité.

1.2.3. L'évaluation de la durabilité

a - Via des équilibres

Dans la mesure où l'objectif de ces modèles bio-économiques est la gestion durable de la biodiversité, il est essentiel de réfléchir à un cadre formel permettant de représenter et de gérer cette durabilité de manière pertinente. Les approches d'équilibres et des états stationnaires constituent une démarche importante et historique pour traiter la durabilité en bio-économie. Notamment, le MSY (Maximum Sustainable Yield) et le MEY (Maximum Economic Yield) sont des concepts de référence pour la gestion des ressources renouvelables (Murphy & Smith, 1991, Beddington & May, 1980, Anderson, 1975, Schaefer, 1954). L'idée sous-jacente est la mise en équilibre des ressources en égalisant les prélèvements avec le taux de croissance de la population. Le MSY maximise les prélèvements à l'équilibre tandis que le MEY optimise les profits à l'équilibre. En intégrant des données monétaires relatives aux prix et aux coûts de l'effort, le MEY conduit à des prélèvements plus faibles. Ces concepts sont encore abondamment mobilisés pour la gestion des pêches (Christensen, 2010, Dichmont *et al.*, 2010, Grafton *et al.*, 2010, Kompas *et al.*, 2010). Néanmoins, ces approches stationnaires ne permettent pas l'intégration de processus dynamiques et s'étendent difficilement au cadre incertain.

b - Via l'optimalité

D'autres approches ont alors été développées pour permettre l'évaluation de critères inter-temporels hors équilibres. C'est notamment le cas des méthodes coût-bénéfice et coût-efficacité, qui font respectivement appel au contrôle optimal et contrôle optimal sous contraintes (Clark, 1976). Dans la méthode coût-bénéfice, comme dans Holzkamper & Seppelt (2007), le bien commun est monétarisé et intégré à la fonction d'utilité. La solution optimale maximise cette fonction. En plus des problèmes liés à la monétarisation, cette méthode n'est pas nécessairement compatible avec une perspective de conservation des ressources naturelles (Clark, 1990), puisque l'extinction peut être optimale dans certaines situations bio-économiques. La méthode coût-efficacité, comme dans Rashford *et al.* (2008), conduit à choisir la stratégie maximisant des objectifs économiques sous contraintes écologiques. En découplant les objectifs économiques et écologiques, ces contraintes empêchent la substitution de la biodiversité par d'autres biens économiques (Gatto & De Leo, 2000, Diamond & Hausman, 1994). Cette méthode a notamment permis de mettre en évidence des frontières d'efficience (formées par l'ensemble des optimums de Pareto [1]) entre critères économique et écologique (Polasky *et al.*, 2008, Drechsler *et al.*, 2007). Autrement dit, augmenter la contrainte écologique conduit nécessairement à un profit économique réduit. Même si cette approche de contrôle optimal propose un cadre d'étude intéressant, notamment avec sa variante "maximin" prenant en compte la génération la plus démunie (Heal, 1998) ou la mise en évidence de politique au coût minimal parmi celles réalisant les objectifs de conservation, nous pensons que ces études basées sur une recherche d'optimalité gagneraient à être complétées par d'autres approches.

c - Via la viabilité

En effet, nous pouvons distinguer deux grands enjeux entourant la notion de durabilité : l'évaluation multi-critère, et l'équité inter-générationnelle (Rawls, 1971). En d'autres termes, une politique publique durable doit satisfaire un ensemble de contraintes (qui peuvent être de natures différentes) à chaque pas de temps afin d'offrir les mêmes potentialités à chaque génération. La théorie du contrôle viable propose un cadre intéressant pour intégrer ces éléments de durabilité. Introduite par Aubin (1991), elle vise à traiter, à chaque pas de temps, un ensemble de contraintes sans hiérarchisation a priori. Cela ouvre ainsi les portes d'une analyse multi-critère. Doyen *et al.* (2012), De Lara *et al.* (2007), Tichit *et al.* (2007) ont montré que l'approche "Population Viability Analysis" (PVA) (Beissinger & McCullough, 2002) développée en biologie de la conservation conduit à des sorties comparables à celles obtenues avec une apporche de viabilité. Son lien direct avec les approches maximin (Doyen & Martinet, 2012) souligne son intérêt pour traiter l'équité inter-générationnelle. Cette méthode repose généralement sur l'obtention d'un noyau de viabilité, au lieu d'une trajectoire optimale unique (Baumgärtner & Quaas, 2009). Il caractérise l'ensemble des trajectoires respectant le jeu de contraintes, permettant alors une gestion du risque plus intégrée. La théorie du contrôle viable permet ainsi de dépasser l'antagonisme apparent entre l'écologie, souvent préoccupée pour des questions de survie et de conservation, et l'économie, plutôt attachée à la recherche d'efficience et d'optimalité. Elle est déjà utilisée dans de nombreux modèles de gestion de ressources renouvelables (Doyen *et al.*, 2012, Péreau *et al.*, 2012, Martinet *et al.*, 2007, Cury *et al.*, 2005, Eisenack *et al.*, 2005, Béné *et al.*, 2001) afin de concilier les objectifs environnementaux et économiques. Nous suggérons donc la combinaison d'approches coût-efficacité et de viabilité pour approfondir le débat des politiques publiques au défi de la biodiversité.

1.3. La réconciliation agriculture - biodiversité

1.3.1. Agriculture et biodiversité en France

Plusieurs types de biodiversité sont présentes dans le milieu agricole. D'une part, l'agro-biodiversité qualifie la diversité des espèces cultivées ou élevées par les agriculteurs. La diversité des cultivars et la diversité génétique au sein d'un troupeau par exemple font partie de l'agro-biodiversité. Cette agro-biodiversité est contrôlée par les agriculteurs et fait partie intégrante de la stratégie de production. D'autre part, la diversité agricole est formée par les espèces vivant en relation avec le milieu agricole exploité par les agriculteurs, mais qui ne font pas directement partie de la stratégie productive. C'est notamment le cas des pollinisateurs, des oiseaux, de la faune du sol, ou des plantes sauvages en périphérie des champs. Par les services qu'elle fournit, cette diversité agricole joue néanmoins un rôle essentiel dans la production. Notons par exemple la fixation de l'azote facilitée par la diversité végétale dans les prairies, la stimulation de l'appétit et de la digestion des herbivores domestiques par la diversité botanique des parcours, la pollinisation par les abeilles et les papillons, l'aération et le renouvellement des sols par la micro-faune du sol, et le contrôle des ravageurs

1. états d'où il est impossible d'améliorer l'état d'un agent sans diminuer l'état d'un autre agent

de cultures par des espèces insectivores. Dans ce travail, nous nous intéressons plus spécifiquement à cette diversité agricole.

La France métropolitaine occupe une place de choix pour la biodiversité agricole en Europe. En effet, sa position au carrefour de quatre zones biogéographiques majeures (méditerranéenne, alpine, océanique et continentale) en fait un pays particulièrement riche pour la biodiversité. Sur moins de 12% du territoire européen, il abrite 57% des types d'habitats d'intérêt communautaire listés dans le cadre de la directive Habitats et 40% de la flore d'Europe, avec une forte proportion d'espèces endémiques, surtout dans ses parties méditerranéenne et alpine (IFEN, 2006, UICN, 2005). Mais le patrimoine de la France en espèces vivantes est vulnérable puisqu'il se situe au 4e rang mondial pour les espèces animales menacées et au 9e rang pour les espèces végétales selon la liste rouge de l'IUCN. Les deux-tiers des habitats et des espèces sauvages du territoire se trouvent sur le territoire rural, où s'exercent notamment les activités agricoles, pastorales et forestières. L'imbrication entre agriculture et biodiversité agricole présente donc un enjeu particulièrement important en France.

1.3.2. Contexte politique de l'agriculture française

L'agriculture française a, de longue date, été influencée par les politiques publiques. Cette influence est devenue encore plus forte et visible avec l'instauration de la Politique Agricole Commune (PAC) en 1962 à l'échelle européenne (Bureau, 2007). Cette importante politique, qui recueille plus de 40% du budget européen (cadre financier 2007-2013), avait pour finalités initiales :

- de garantir la sécurité des approvisionnements,
- de stabiliser les marchés,
- d'assurer des prix raisonnables aux consommateurs,
- d'assurer un niveau de vie équitable aux agriculteurs.

La politique productiviste et protectionniste mise en place a été une réussite d'un point de vue ces finalités : modernisation de l'agriculture, développement de la production, gains de productivité considérables et auto-suffisance alimentaire. Cependant, l'intensification des cultures et des élevages, l'homogénéisation des paysages, la disparition des éléments semi-naturels (haies, arbres, bosquets), la mécanisation et l'utilisation systématique d'intrants (pesticides, insecticides, engrais) sous-jacents à ce productivisme ont eu de forts impacts sur l'environnement et la biodiversité (Griffon, 2006) : pollution des eaux et des sols, érosion des sols, disparition d'espèces, diminution de la taille des communautés, homogénéisation biotique.

Dans la lignée des mesures d'accompagnement de la PAC visant à la protection de l'environnement et des ressources naturelles mises en place en 1992, le conseil Européen de Berlin a alors instauré en 1999 un second pilier, complétant le premier pilier portant sur le soutien des revenus agricoles. Ce second pilier, consacré au développement rural, intègre notamment la promotion de l'environnement. Différentes Mesures Agro-Environnementales (MAE) ont été proposées aux agriculteurs (comme la mise en place de bandes enherbées le long de cours d'eau, l'interdiction d'apports d'engrais sur certaines prairies naturelles, la restauration et maintien des haies et du bocage, et la prime herbagère agri-environnementale). Cependant, de nombreuses études (Kleijn *et al.*, 2006, Vickery *et al.*, 2004) montrent que les mesures adoptées sur la période 1992-1999 présentent une efficacité limitée. Le rapport du Plan de Développement Rural Hexagonal (PDRH, 2008) conduit aux mêmes conclusions pour les MAE du second pilier. Notamment, un faible taux de contractualisation, par ailleurs biaisé en faveur des MAE peu contraignantes, limite leur impact. Le manque de coordination spatiale entre agriculteurs voisins pour adopter des MAE similaires ne permet pas de changements cohérents au niveau du paysage. Enfin, la base économique des MAE, fondée sur le manque à gagner des agriculteurs, n'amène pas de valorisation de l'environnement, sa protection restant considérée comme une contrainte pesant sur la production, qui demeure la raison d'être de l'agriculture. Face à l'ampleur de l'érosion de la biodiversité, la mise en place d'une stratégie d'ensemble intégrée apparaît indispensable.

Dans la lignée des réflexions menées pour la PAC d'après 2013 (période budgétaire 2014-2020), il existe un réel enjeu quant à l'élaboration de nouvelles politiques agricoles à visée environnementale afin d'améliorer leur efficacité. Du point de vue de la recherche scientifique, cela se traduit par une réflexion sur des scénarios de politiques publiques conciliant de manière simultanée des objectifs économiques et écologiques et intégrant une gestion du risque économique et environnemental. Ils doivent être conçus à un maillage pertinent tant agro-économiquement, qu'écologiquement et politiquement. Compléter des scénarios d'occupations des sols par une réflexion en termes d'outils politiques (scénarios de politiques publiques) représente une perspective intéressante pour identifier des leviers plus opérationnels. Enfin d'un point vue écologique, l'intégration de plusieurs espèces à la réflexion permet de gagner en généricité ainsi que d'aborder la biodiversité dans sa complexité.

1.3.3. Les oiseaux communs comme métrique de la biodiversité

Comme nous le soulignions dans la première partie de l'introduction, les problèmes de la gestion de la biodiversité à partir d'une connaissance partielle de sa réalité et de la métrique de la biodiversité sont deux questions auxquelles nous sommes encore confrontés aujourd'hui. L'enjeu réside donc dans le choix du groupe d'espèces qui sera étudié et considéré comme représentant de la biodiversité agricole. Des stratégies de gestion seront en effet mises en place dans l'objectif de conserver la biodiversité dans son ensemble à partir des seules conclusions obtenues avec ce groupe.

Plusieurs groupes fonctionnels (comme les pollinisateurs et la faune du sol) sont particulièrement intéressants pour représenter la biodiversité agricole au regard de leur fonction dans les agro-écosystèmes (pollinisation et aération de sols). Cependant, le manque de données constitue un obstacle majeur à leur intégration dans un modèle. A l'inverse, les oiseaux communs sont depuis longtemps suivis en Europe (37 pays d'Europe ont mis en place des suivis, dont le Suivi Temporel des Oiseaux Communs en France, le Breeding Bird Survey en Angleterre) et ont fait l'objet de nombreuses études en milieu agricole (voir encadré 1). Dans ce contexte, nous avons choisi d'appréhender la biodiversité au travers d'une communauté d'oiseaux communs nicheurs. Les espèces communes présentent les avantages d'avoir une large aire de répartition, une abondance locale forte et/ou une large amplitude d'habitats, permettant ainsi la construction de modèles à large échelle balayant de larges gradients spatiaux. Le choix des oiseaux comme représentant de la biodiversité présente plusieurs intérêts :

- de nombreuses études montrent une sensibilité de ce taxon aux évolutions de l'agriculture (Doxa *et al.*, 2010, Gregory *et al.*, 2004, Donald *et al.*, 2001),
- à leur place relativement haute dans la chaine alimentaire, ils intègrent beaucoup de variations de cette chaine,
- ils sont largement dépendants de la composition du paysage (Weibull *et al.*, 2003),
- ils offrent de larges services écosystémiques comme le contrôle des rongeurs par les rapaces, le contrôle des espèces nuisibles par les insectivores et la dispersion des graines par les frugivores (Sekercioglu *et al.*, 2004),
- ils constituent un indicateur visible du grand public par lequel il est facile de faire passer un message (Ormerod & Watkinson, 2000),
- ce taxon est fréquemment utilisé dans les modèles bio-économiques (Rashford *et al.*, 2008, Holzkamper & Seppelt, 2007, Doherty *et al.*, 1999).

ENCADRÉ 1 - L'agriculture et les oiseaux

Les oiseaux ont fait l'objet de nombreux suivis en milieu agricole. Ces bases de données ont généré un ensemble de travaux dans toute l'Europe, témoignant du déclin des oiseaux communs en milieu agricole, que Krebs *et al.* (1999) nomment *The Second Silent Spring* en référence à l'ouvrage de Rachel Carson (Carson, 1962). Ainsi, Siriwardena *et al.* (1998) montrent, sur la base d'un échantillon de 42 espèces d'oiseaux communs des milieux agricoles anglais, que 13 espèces spécialistes des milieux agricoles déclinent en moyenne de 30%, alors que 29 espèces généralistes progressent de 23%. Au Royaume-Uni, 10 espèces de milieux agricoles ont connu un déclin de 10 millions d'individus en 20 ans (Krebs *et al.*, 1999). En Suède, Wretenberg *et al.* (2007) montrent, pour 7 espèces spécialistes des milieux agricoles, des déclins significatifs dans au moins une des 3 régions considérées. En France, sur la base des données du STOC, Julliard *et al.* (2004) montrent un déclin des abondances de 14% entre 1989 et 2001 pour 65 espèces d'oiseaux communs. Dans l'ensemble des pays européens, la tendance moyenne entre 1970 et 1990 des abondances d'oiseaux en milieu agricole est à la baisse.

L'enjeu pour les écologues est de déterminer les causes de ce déclin à large échelle. Et la littérature s'accorde pour donner les transformations du monde agricole comme principal responsable. En effet, de nombreuses corrélations spatiales et temporelles plaident en faveur de cette hypothèse. D'un point de vue temporel, les perturbations de l'évolution de l'avifaune sont contemporaines aux transformations du secteur agricole. En Angleterre par exemple, le déclin s'accentue à partir des années 70, période où commence l'intensification de l'agriculture (Donald *et al.*, 2001, Chamberlain *et al.*, 2000, Siriwardena *et al.*, 1998). En Suède, sur la période 1976-2003, les ruptures de l'évolution de quatre espèces migratrices coïncident avec les ruptures des politiques agricoles (intensification en 87, gel des terres sur 87-95, et PAC après 95) (Wretenberg *et al.*, 2007). D'un point de vue spatial, Donald *et al.* (2001) montrent que les pays de l'UE où l'intensification de l'agriculture a été la plus forte (pays de l'Ouest de l'Europe), sont les plus touchés par le déclin des espèces.

L'agriculture affecte les populations d'oiseaux via différents effets. L'empoisonnement des individus par les pesticides (dont le DDT ou dichloro-diphényl-trichloro-éthane) a été l'un des premiers effets identifié (Carson, 1962). Mais de manière plus générale, toutes les perturbations environnementales affectant la qualité de l'habitat et la disponibilité des ressources jouent sur la démographie des oiseaux (Benton *et al.*, 2003).

Différentes solutions ont été alors proposées et étudiées dans la littérature pour réconcilier agriculture et biodiversité aviaire. Une stratégie consiste en la réintroduction de éléments semi-naturels au sein du territoire agricole. Les mesures environnementales, dont les MAE, ont en effet été construites dans cette perspective. En introduisant des zones de nidification, des corridors écologiques, des ressources alimentaires plus diversifiées et plus importantes, l'objectif est de stopper le déclin des espèces spécialistes des milieux agricoles. Cependant, comme nous le mentionnions précédemment, les effets de ces mesures restent limités (PDRH, 2008, Kleijn *et al.*, 2006, Vickery *et al.*, 2004). Une autre stratégie consiste à réflechir à la gestion de l'intensité, puisque celle-ci a été identifiée comme un moteur majeur du déclin des oiseaux. Schématiquement, deux solutions sont disponibles : soit l'intensité est réduite sur l'ensemble du territoire légérement ("landsharing"), soit une partie du territoire est maintenue à un fort niveau d'intensité et l'autre partie forme une réserve naturelle sur laquelle il n'y a pas d'activités agricoles ("landsparing"). Green *et al.* (2005) tranchent le débat en illustrant que les deux solutions sont intéressantes et que le choix dépend de la réponse fonctionnelle des oiseaux à l'intensité. Leur travail reste cependant purement théorique. La réconciliation agriculture-oiseau forme donc un sujet de recherche majeur dans les travaux en agro-écologie. C'est dans ce contexte que se place notre étude.

1.3.4. Problématique générale

Cette thèse contribue au débat sur les politiques publiques face au défi d'une gestion durable de l'agriculture et de la biodiversité en France métropolitaine. Réalisée dans le cadre du contrat de recherche ANR FARM-BIRD, elle est le fruit de collaborations inter-disciplinaires entre le CNRS, l'INRA et le MNHN. Cette thèse propose une approche fondée sur une modélisation bio-économique calibrée avec des données écologiques, agronomiques et économiques. Elle articule des scénarios de politiques publiques et d'occupation des sols, des décisions économiques et des dynamiques écologiques à différentes échelles spatiales, pertinentes d'un point de vue politique, agro-économique et écologique. Afin de s'inscrire dans la problématique de durabilité, le modèle s'efforce d'en intégrer ses caractéristiques : objectifs multi-critères, évaluation dynamique et inter-temporelle des performances, gestion de l'incertitude économique et écologique.

Cette thèse répond à deux objectifs :

1. **Développer un modèle bio-économique**. Cet enjeu méthodologique demande de conduire simultanément des réflexions sur le plan conceptuel et le plan technique pour modéliser les différentes dynamiques du système, coupler les compartiments, choisir les données, réaliser la calibration, et évaluer les scénarios. Elles sont sous-jacentes à l'ensemble des articles. Cependant, dans certains articles, nous avons exploré plus spécifiquement trois points relatifs à :
   - La validité de la calibration
   - La validité des indicateurs
   - La gestion de l'incertitude

2. **Réfléchir à la réconciliation agriculture-biodiversité**, en identifiant des leviers clefs pour l'élaboration d'une politique agricole conduisant à une agriculture multi-fonctionnelle et durable. Différents volets de cette réconciliation sont étudiés grâce à la combinaison des deux méthodes coût-efficacité et viabilité :
   - Quelle politique publique permet d'améliorer les performances économiques et écologiques ?
   - Quel est l'effet de paramètres micro-économiques sur les performances bio-économiques ?
   - La réconciliation peut-elle être optimale sur les deux critères économiques et écologiques ?
   - Comment dépasser l'antagonisme apparent entre performances économiques et écologiques ?

1.3.5. Structure du manuscrit

Cette thèse est construite sur des articles qui sont disponibles dans les chapitres suivants (voir la liste des articles en détail avec le tableau I.1). Cependant, nous avons synthétisé les principaux apports scientifiques de la thèse dans les sections suivantes de ce chapitre I. La section 2 présente le modèle générique développé dans cette thèse et proposé pour étudier les politiques publiques au défi de la biodiversité. Son application à l'agriculture en France et au cas d'étude (bases de données, échelles spatio-temporelles, scénarios testés) est spécifiée à la section 3. Une synthèse des résultats issus des articles est présentée à la section 4. Enfin, des éléments de discussion générale sur l'ensemble du travail sont proposés à la section 5. La lecture des articles apportera davantage de détails méthodologiques, de résultats et de discussion spécifique. Pour faciliter la lecture, le tableau I.1 propose une récapitulation des questions soulevées dans la thèse et des réponses apportées par les différents articles proposés aux chapitres suivants.

Questions	Ch. II	Ch. III	Ch. IV	Ch. V	Ch. VI	Ch. VII	An. A	An. B	An. C
Validation de la calibration						X			
Validation des indicateurs	X		X						
La gestion de l'incertitude		X			X		X		
Méthode coût-efficacité				X					
Méthode viabilité					X				
Rôle des politiques publiques	X		X	X	X		X	X	X
Effet des paramètres micro-économiques		X					X		
Optimalité de la réconciliation bio-économique			X	X					
Dépasser l'antagonisme bio-économique	X			X		X			X

avec

Chapitre II : Bio economic modeling for a sustainable management of biodiversity in agricultural lands (publié dans Ecological Economic, 2011)
Chapitre III : How does economic risk aversion affect biodiversity ? (Working paper dans les Cahiers du GREThA, soumis dans Ecological Applications)
Chapitre IV : Different policy scenarios to promote various targets of biodiversity (publié dans Ecological Indicators, 2012)
Chapitre V : A double dividend of biodiversity in agriculture (soumis dans Ecological Economics)
Chapitre VI : Co-viability of farmland biodiversity and agriculture (en préparation)
Chapitre VII : Dynamic models for bird community in farming landscapes (révision dans Ecological Modelling)
Annexe A : Risk aversion impact on bio-economic performances in an agricultural public policy context : a modeling approach (Proceedings ISEE, 2010)
Annexe B : Innovation rigidity and ecological-economic reconciliation in agriculture (Proceedings ISDA, 2010)
Annexe C : Partnering biodiversity and income in French farmlands (Note de la Commission Européenne, 2011)

Table I.1 – Tableau récapitulatif des questions soulevées dans la thèse et des réponses apportées par les différents articles.

2. Le modèle bio-économique

2.1. Organisation générale du modèle

Le modèle bio-économique développé dans cette thèse est résumé par le schéma fonctionnel présenté dans la figure I.2. Il est composé de trois sous-modèles, organisés autour de trois éléments : le décideur public,

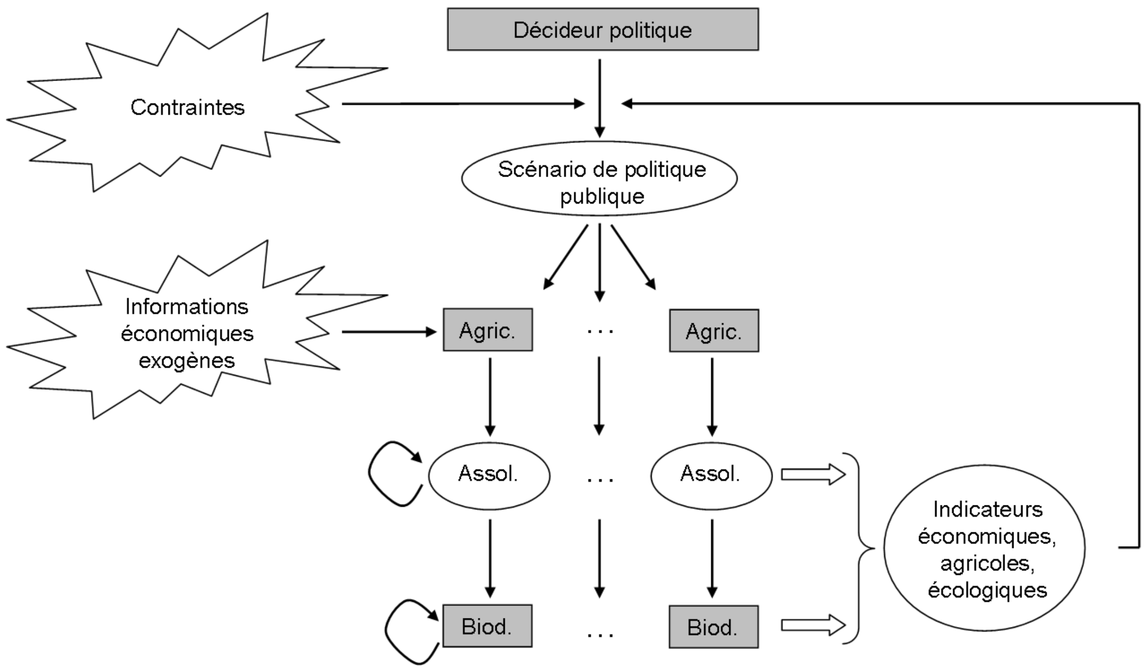


FIGURE I.2 – Schéma d'ensemble du modèle (avec Agric. pour agriculteur représentatif, Assol. pour assolements et plus généralement occupation des sols, et Biod. pour biodiversité)

l'agriculteur et la biodiversité. Le décideur politique définit à une échelle macro un scénario de politique publique en tenant compte de contraintes extérieures (contrainte budgétaire) et des objectifs (économiques, écologiques, productifs, sociaux) qu'il souhaite atteindre. Les agriculteurs représentatifs, à une échelle micro, déterminent leur stratégie agricole future compte tenu de la politique publique mise en place et d'informations économiques contextuelles (évolution des prix, incertitude). La qualité des habitats disponibles pour la biodiversité est dépendante des stratégies agricoles mises en place. L'évolution de la biodiversité est ainsi impactée par les décisions prises par les agriculteurs. Le scénario de politique publique testé est évalué au travers de différents indicateurs économiques, agricoles et écologiques, permettant de vérifier si les objectifs ont été atteints.

2.2. Le modèle agro-écologique micro

Le modèle écologique représente la dynamique de la taille de la population dans son ensemble en fonction des variables agricoles à une échelle micro. Nous n'explicitons pas les mécanismes sous-jacents à cette dynamique comme le succès reproducteur, la survie juvénile, ou la sénescence. La population est considérée dans son ensemble sans découpage en classes d'âge, notamment en raison de la structure des données utilisées pour la calibration (voir section 3.3).

La dynamique de population intègre néanmoins une compétition intra-spécifique fondée sur une capacité de charge du milieu. On peut définir cette dynamique dans sa forme générale de la manière suivante :

$$N_{s,r}(t+1) = f(N_{s,r}(t), M_{s,r}(t)) \tag{I.1}$$

où $N_{s,r}(t)$ représente l'abondance de l'espèce s dans la région r à l'année t et $M_{s,r}(t)$ la qualité de l'habitat la région r pour l'espèce s. Plusieurs formes fonctionnelles f ont été testées (chapitre VII (article en révision dans Ecological Modelling)) dont la forme fonctionnelle de Beverton-Holt (Beverton & Holt, 1957) :

$$f(N_{s,r}(t), M_{s,r}(t)) = N_{s,r}(t).(1+R_{s,r})\left(1+\frac{N_{s,r}(t)}{M_{s,r}(t)}\right)^{-1} \tag{I.2}$$

Le paramètre $R_{s,r}$ correspond au taux de croissance intrinsèque de l'espèce s dans la région r. Il intègre par exemple le nombre de portées par an et le nombre moyen de jeunes par portée. La capacité de charge de la région r est définie par le produit $M_{s,r}(t) * R_{s,r}$. Elle dépend de la qualité de l'habitat $M_{s,r}(t)$ dans la région r pour l'espèce s, supposée linéairement dépendante des proportions $A_{r,k}(t)$ de la Surface Agricole Utile (SAU) dédiées aux différentes activités agricoles k (comme par exemple les céréales et les prairies) :

$$M_{s,r}(t) = \beta_{s,r} + \sum_k \alpha_{s,r,k}.A_{r,k}(t) \tag{I.3}$$

Le coefficient $\alpha_{s,r,k}$ caractérise la réponse de l'espèce s à l'activité agricole k dans la région r. Le coefficient $\beta_{s,r}$ peut être interprété comme la qualité moyenne de l'habitat de la région r pour l'espèce s, intégrant par exemple la proportion de zones forestière, urbaine et aquatique, qui ont montré leur importance pour les dynamiques de populations (Devictor & Jiguet, 2007).

La taille de la population $N_{s,r}(t+1)$ peut également être modélisée dans un contexte d'incertitude (cas du chapitre VI) où elle est définit comme la somme de la taille de population issue du modèle $N_{s,r}(t+1) = f(N_{s,r}(t), M_{s,r}(t))$ et d'un coefficient d'incertitude $\vartheta_{s,r}(t)$ tiré dans une loi normale calibrée à partir des données historiques.

$$N_{s,r}(t+1) = f(N_{s,r}(t), M_{s,r}(t)) + \vartheta_{s,r}(t) \tag{I.4}$$

Le modèle ci-dessus est développé pour chaque espèce, la communauté formée par l'ensemble de ces espèces est alors appréhendée au moyen d'indicateurs de biodiversité notés *Biod*. De nombreux indicateurs sont disponibles dans la littérature, comme par exemple la diversité spécifique, la variation d'abondance ou la distance génétique. Leur pertinence dépendant des espèces étudiées et de la question posée, nous motivons notre choix dans la section 3. Cependant, nous pouvons définir ici ces indicateurs de manière formelle comme une fonction h des abondances $N_{s,r}(t)$ des espèces présentes dans la communauté :

$$Biod_r(t) = h\big(N_{1,r}(t), \ldots, N_{S,r}(t)\big) \tag{I.5}$$

2.3. Le modèle agro-économique micro

Chaque région étudiée est représentée par un agriculteur représentatif. Le revenu de l'agriculteur représentatif de la région r à l'année t est noté $Inc_r(t)$. Il dépend des bénéfices de la production d'une part et des subventions et taxes potentiellement distribuées par l'État d'autre part :

$$Inc_r(t) = \sum_k gm_{r,k}(t).A_{r,k}(t).(1+\tau_k) \tag{I.6}$$

La partie vente est calculée à partir des marges brutes par hectare $gm_{r,k}(t)$ des activités k présentes dans la région r et des proportions $A_{r,k}(t)$ de la Surface Agricole Utile (SAU) dédiées aux différentes activités

k. Les marges brutes $gm_{r,k}(t)$ peuvent être supposées incertaines (incertitudes de marché, de production et climatique). Des lois gaussiennes paramétrées à partir des moyennes et variances des données historiques ont été choisies pour capturer ces incertitudes. Les incitations publiques macro τ_k sont exprimées comme un pourcentage de la marge brute de l'activité k dans la région r et peuvent prendre la forme de taxes ($\tau_k < 0$) ou de subventions ($\tau_k > 0$).

Chaque année, l'agriculteur représentatif détermine l'allocation de sa SAU aux différentes activités k en maximisant sa fonction d'utilité dans un contexte d'incertitude économique et en respectant des contraintes techniques. La fonction d'utilité est une forme quadratique du risque basée sur une expression moyenne-variance :

$$Util_r(t) \quad = \quad \mathbb{E}[Inc_r(t)] - a.\mathbb{V}ar[Inc_r(t)] \tag{I.7}$$

où

$$\begin{aligned} \mathbb{E}[Inc_r(t)] &= \sum_k \overline{gm}_{r,k}.A_{r,k}(t).(1+\tau_k) && \text{(I.8)} \\ \mathbb{V}ar[Inc_r(t)] &= \sum_k \sum_{k'} \sigma_{r,k,k'}(t).A_{r,k}(t).A_{r,k'}(t).(1+\tau_k).(1+\tau_{k'}) && \text{(I.9)} \end{aligned}$$

Le premier terme $\mathbb{E}[Inc_r(t)]$ représente le revenu espéré de l'agriculteur. Il est exprimé à partir des marges brutes espérées $\overline{gm}_{r,k}$, moyennes des marges brutes de la Série Historique notée SH[1]. Le second terme caractérise le risque. Le paramètre a représente le coefficient absolu d'aversion au risque. Plus a est élevé, plus l'agent est averse au risque. Le cas $a = 0$ correspond à un comportement neutre au risque, donc à une situation déterministe. Le paramètre $\sigma_{r,k,k'}$ représente la covariance entre les activités k et k' dans la région r calculée au sein de la série historique[2].

Le programme de maximisation de l'agriculteur est alors défini ainsi :

$$\max_{A_{r,k}} \; Util_r(t) \tag{I.10}$$

sous les contraintes :

$$|A_{r,k}(t) - A_{r,k}(t-1)| \leq \varepsilon.A_{r,k}(t-1) \tag{I.11}$$

$$\sum_k A_{r,k}(t) = SAU_r(t_0) \tag{I.12}$$

La première contrainte (eq. I.11) est une contrainte technique. Le paramètre ε correspond à la rigidité du changement. A chaque pas de temps, l'agriculteur ne peut changer qu'une proportion ε de la superficie de l'année précédente dédiée à chacune des activités. Plus ε est grand, plus la marge de manoeuvre offerte à l'agriculteur est importante. Cette contrainte, en limitant la vitesse de changement, remplace en partie l'intégration de coûts de changement dans la fonction d'utilité. La seconde contrainte (eq. I.12) est une contrainte de capital, assurant la stabilité de la SAU au cours du temps. La valeur de la SAU de la dernière année t_0 de la série temporelle est choisie comme référence.

Les performances économiques régionales sont évaluées au travers du revenu $Inc_r(t)$ (eq. I.6). D'autres indicateurs évaluent directement les activités agricoles (proportions de certains assolements, diversité des assolements ...). De manière générale, ils sont définis comme une fonction g des assolements $A_{r,k}(t)$:

$$Agri_r(t) = g(A_{r,1}(t), \ldots, A_{r,K}(t)) \tag{I.13}$$

1. $\overline{gm}_{r,k} = \frac{1}{Card(SH)} \sum_{t \in SH} gm_{r,k}(t)$.
2. $\sigma_{r,k,k'} = \frac{1}{Card(SH)} \sum_{t \in SH} (gm_{r,k}(t) - \overline{gm}_{r,k}(t)).(gm_{r,k'}(t) - \overline{gm}_{r,k'}(t))$.

2.4. Le modèle de décision publique macro

2.4.1. Les indicateurs agrégés

D'un point de vue écologique, des indicateurs agrégés au niveau national $Biod(t)$ sont calculés à partir des indicateurs régionaux $Biod_r(t)$. Leurs calculs, dépendants des indicateurs choisis, sont détaillés dans la section 3.

D'un point de vue économique, les performances économiques nationales sont estimées en termes monétaires directement via le revenu à l'année t :

$$Inc(t) = \sum_r SAU_r \, . \, Inc_r(t) \tag{I.14}$$

ou au travers de la valeur inter-temporelle actualisée (Present Value - PV) :

$$PV(\tau) = \sum_{t=t_1}^{T} \rho^{t-t_1} \, . \, Inc(t) \tag{I.15}$$

Elle est définie comme la somme inter-temporelle du revenu national $Inc(t)$ actualisé par le facteur d'escompte ρ sur l'ensemble de la projection, depuis la première année t_1 jusqu'à l'horizon temporel T. Le revenu $Inc(t)$ étant dépendant des incitations τ_k mises en place (eq. I.6), la valeur actualisée est bien dépendante de τ, matrice des τ_k caractérisant la politique appliquée.

Enfin, le budget national $Budg(t)$ dépensé à l'année t est calculé à partir du montant des incitations distribuées à l'ensemble des régions r et des activités agricoles k :

$$Budg(t) = \sum_r SAU_r \, . \sum_k gm_{r,k}(t) \, . \, A_{r,k}(t) \, . \, \tau_k \tag{I.16}$$

2.4.2. Des scénarios de référence

Les scénarios de référence sont construits afin de poursuivre les tendances actuelles. Ils permettent d'une part de comprendre l'impact des politiques actuelles sur les performances bio-économiques si aucune nouvelle décision n'est prise et d'autre part d'analyser l'impact marginal de différents scénarios potentiels de politiques publiques par rapport à la tendance actuelle. Deux scénarios tendanciels ont été utilisés :

- le scénario Statu Quo (SQ), où l'occupation des sols est figée à celle de t_0.

$$A_{r,k}(t) = A_{r,k}(t_0) \;\; pour\ tous\ r,\ k\ et\ t = t_1, \ldots, T \tag{I.17}$$

- le scénario Laissez-Faire (LF), où l'agriculteur fait ses choix sans intervention politique additionnelle. Il y a donc maximisation de la fonction d'utilité de la part des agriculteurs dans ce scénario.

$$\tau_k = 0 \;\; pour\ tous\ k \tag{I.18}$$

2.4.3. Des scénarios exogènes

Les scénarios exogènes représentent des politiques publiques contrastées qui permettent d'estimer les évolutions possibles du système bio-économique en réponse à des incitations économiques. Les niveaux des incitations sont déterminés de manière exogène mais respectent néanmoins une contrainte budgétaire globale, où le budget annuel national $Budg(t)$ ne doit pas dépasser un montant préalablement fixé $Budg_0$:

$$Budg(t) \leq Budg_0 \tag{I.19}$$

L'objectif est de réfléchir globalement aux grandes orientations éventuelles de la PAC. L'opposition entre premier et second pilier (Gray, 2002) peut se traduire très schématiquement par la dichotomie grandes cultures - prairies (non intensives). Nous avons donc développé quatre scénarios exogènes ayant pour objectif d'explorer quatre situations contrastées de cette dichotomie :

- le scénario Crop (CR), avec des subventions distribuées aux grandes cultures de céréales-oléagineux-protéagineux (COP). Le scénario stylise une politique stimulant la culture des céréales, comme dans le cas d'une politique de développement des biocarburants.

$$\tau_{cop} > 0\,, \quad \tau_k = 0 \;\; \textit{pour tous}\; k \neq cop \tag{I.20}$$

- le scénario Grassland (GL) représente, à l'inverse, une politique favorisant un retour à l'herbe. Ce type de scénario caractérise une politique intégrant mieux les considérations environnementales développées dans le 2nd pilier. Un développement des prairies (grassland) non intensives, ayant un impact globalement positif sur la biodiversité (Batary *et al.*, 2011, Laiolo, 2005, Bignal & McCracken, 2000), est préconisé. Ce scénario est donc basé sur des subventions aux prairies non intensives.

$$\tau_{grass} > 0, \quad \tau_k = 0 \;\; \textit{pour tous}\; k \neq grass \tag{I.21}$$

- le scénario Double Subsidies (DS) couple des subventions à la fois aux grandes cultures et aux prairies non intensives. Il représente une politique proche de la tendance actuelle où les deux piliers coexistent et orientent les décisions agricoles dans des directions opposées. Le levier de cette politique réside dans les proportions de chacune des deux incitations.

$$\tau_{cop} > 0, \quad \tau_{grass} > 0, \quad \tau_k = 0 \;\; \textit{pour tous}\; k \neq \{cop, grass\} \tag{I.22}$$

- le scénario High Quality Environment (HQE) associe quant à lui des taxes aux grandes cultures et des subventions aux prairies non intensives. Ce scénario correspond à une politique fortement en faveur du retour à l'herbe. Le couplage avec des taxes sur les grandes cultures permet de toucher davantage les régions très spécialisées en grandes cultures. En effet, la rentabilité des COP y est largement dominante et une simple subvention aux prairies ne suffit pas pour inciter les agriculteurs à développer cette activité.

$$\tau_{cop} < 0, \quad \tau_{grass} > 0, \quad \tau_k = 0 \;\; \textit{pour tous}\; k \neq \{cop, grass\} \tag{I.23}$$

2.4.4. Des scénarios endogènes optimaux

Les scénarios endogènes représentent des politiques déterminées par la maximisation de l'espérance (la moyenne) de la valeur actualisée PV (définie eq. I.15) sous contraintes, ils s'inscrivent donc dans une approche coût-efficacité (Naidoo *et al.*, 2006) :

$$\max_{\tau} \; \mathbb{E}[PV(\tau)] \tag{I.24}$$

sous contraintes :

$$Budg(t) \leq Budg_0 \tag{I.25}$$

$$Biod(T) \geq B_{lim} \tag{I.26}$$

Comme pour les scénarios exogènes, la contrainte budgétaire (eq. I.25) limite le budget annuel national $Budg(t)$ à un un budget préalablement fixé $Budg_0$. La contrainte de biodiversité (eq. I.26) assure un objectif de conservation imposé à l'horizon temporel T. Différentes contraintes B_{lim} peuvent être testées entre le critère le moins contraignant $B_{lim} = 0$ et le critère maximal $B_{lim} = B^*$, obtenu par la maximisation du critère de biodiversité sous contrainte budgétaire :

$$B^* = \max_{\tau} Biod(\tau) \tag{I.27}$$

$$Budg(t) \leq Budg_0 \tag{I.28}$$

Les politiques coût-efficaces optimales, résultats du problème de maximisation (I.24)-(I.25)-(I.26), sont notées :

$$\tau^*(B_{lim}) = \text{Argmax}\, PV(\tau) \tag{I.29}$$

2.4.5. Des scénarios endogènes viables

Les scénarios viables sont définis comme l'ensemble des politiques publiques permettant de satisfaire un ensemble de contraintes bio-économiques à chaque pas de temps. Les contraintes sont ici d'ordres budgétaire (eq. I.30), économique (eq. I.31) et écologique (I.32).

$$Budg(t) \leq Budg_0 \tag{I.30}$$

$$Inc(t \geq \lambda.Inc^{SQ}(t) \tag{I.31}$$

$$Biod(t) \geq \lambda.Biod^{SQ}(t) \tag{I.32}$$

La contrainte budgétaire (eq. I.30) est identique à celles des scénarios exogènes (eq. I.25) et endogènes optimaux (eq. I.28). Les contraintes économiques et écologiques sont construites à partir de seuils, fondés sur les performances obtenues avec le scénario Statu Quo. Les contraintes (I.31) et (I.32) assurent que les performances économiques et écologiques sont supérieures ou égales à un pourcentage des performances obtenues avec le scénario Statu Quo à chaque pas de temps. Le paramètre λ représente la force de la contrainte. Plus λ est élevé, plus la contrainte est limitante.

Dans un contexte d'incertitude, on évalue la probabilité d'un scénario de politique publique τ à satisfaire les contraintes sur l'ensemble de la trajectoire. Ces probabilités sont notées CVA en référence à la "Co-Viability Analysis" :

$$\text{CVA}(\tau) = \mathbb{P}\Big(\text{ Constraints } (I.30), (I.31) \text{ and } (I.32), t = t_1...T \Big) \tag{I.33}$$

Un scénario de politique publique est dit viable si la probabilité qu'il satisfasse ses contraintes à chaque pas de temps est supérieure à un seuil de confiance δ, où la valeur $1 - \delta$ peut être interprétée comme le risque de la politique publique :

$$\text{CVA}(\tau) \geq \delta \tag{I.34}$$

L'ensemble des scénarios viables forment un noyau de viabilité associé à une valeur de risque $1 - \delta$ (Doyen *et al.*, 2012) :

$$\mathcal{T}^{\delta}_{viab} = \{\tau,\ \text{CVA}(\tau) \geq \delta\} \tag{I.35}$$

2.4.6. Le coût des politiques

Ces différentes politiques publiques peuvent être évaluées en termes de coût. Trois types de coûts sont distingués (Semaan *et al.*, 2007) : le coût public $PuC(\tau)$, le coût privé $PrC(\tau)$ et le coût social total

$SoC(\tau)$. Le coût public correspond au budget total inter-temporel escompté dépensé au cours du temps (de t_1 à T) lors de la mise en place du scénario τ étudié (eq. I.36). La contrainte budgétaire n'étant pas toujours saturée, le budget $Budg(t)$ peut être inférieur au budget total disponible $Budg_0$.

$$PuC(\tau) = \sum_{t=t_1}^{T} \rho^{t-t_1} \, . \, Budg(t) \tag{I.36}$$

Le coût privé ($PrC(\tau)$) correspond au manque à gagner des agriculteurs. Il est calculé comme la différence entre le revenu inter-temporel maximum (obtenu avec la politique maximisant la PV sans contrainte biodiversité, c'est-à-dire le scénario $\tau^*(0)$) et le revenu inter-temporel obtenu avec le scénario τ étudié :

$$PrC(\tau) = PV(\tau^*(0)) - PV(\tau) \tag{I.37}$$

Enfin, le coût total social $SoC(\tau)$ correspond à la somme du coût public et du coût social :

$$SoC(\tau) = PuC(\tau) + PrC(\tau) \tag{I.38}$$

Ces coûts sont définis similairement à l'échelle régionale avec les budgets et valeurs actualisées régionaux.

3. Application à l'agriculture en France métropolitaine

3.1. Contexte

Le modèle bio-économique est appliqué à la France métropolitaine. Le premier modèle, présenté au chapitre II (article publié dans Ecological Economics, 2011), est construit avec un maillage régional (ie région administrative). En revanche, tous les autres modèles sont développés à l'échelle de la Petite Région Agricole (PRA). La France est découpée en 714 petites régions agricoles [1] (fig. I.3). Ces PRA sont l'intersection entre les régions administratives (qui ont une pertinence décisionnelle) et les régions agricoles (qui présentent une homogénéité agro-écologique). Elles présentent donc une cohérence décisionnelle et agro-écologique. Les superficies des PRA sont variables, avec une moyenne de 500 km^2 pour un écart-type de 180 km^2.

FIGURE I.3 – Découpage de la France métropolitaine en 714 PRA.

1. Cependant la disponibilité des données nous permettra d'analyser seulement 620 PRA.

Les 14 occupations des sols (OTEA) k
(1) Céréales, Oléagineux, Protéagineux (COP)
(2) Cultures industrielles
(3) Bovin lait (élevage laitier)
(4) Bovin lait-viande (élevage mixte)
(5) Bovin viande (élevage allaitant)
(6) Polyculture-élevage avec orientation herbivore
(7) Élevage d'autres herbivores
(8) Polyculture-élevage avec une orientation granivore
(9) Polyculture-élevage avec une autre orientation
(10) Élevage de granivores
(11) Cultures pérennes
(12) Horticulture
(13) Viticulture (AOC et autres)
(14) Autres associations

TABLE I.2 – Liste des 14 occupations des sols (OTEA), notées k dans le modèle.

Les données agro-économiques et écologiques sont disponibles de 2002 à 2008 ($t_0 = 2008$). L'horizon temporel des scénarios est 2050, soit des projections de $t_1 = 2009$ à $T = 2050$. Un horizon temporel plus court (2025 par exemple) ne nous permettrait pas de mettre en évidence des transitoires et des états stationnaires notamment sur les dynamiques de biodiversité. En revanche, un horizon temporel trop long (2100) impliquerait des écarts trop importants aux hypothèses sous-jacentes à la structure des modèles. De plus, cet horizon temporel à 2050 est cohérent avec d'autres études (rapport Stern (2007)).

3.2. Les données agro-économiques d'occupation du sol

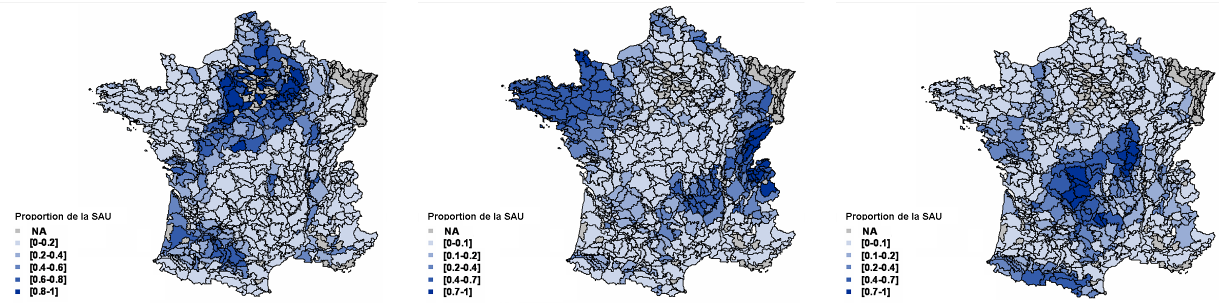
Les données agricoles sont issues de l'Observatoire du Développement Rural[1] (ODR) de l'INRA. De nombreuses données sont disponibles grâce au Réseau d'Information Comptable Agricole (RICA), cependant ces données ne sont pertinentes qu'à l'échelle de la région administrative. Afin d'obtenir des données au niveau PRA, nous avons, avec la collaboration de l'ODR, extrapolé des informations relatives à quatorze classes d'activités agricoles (OTEA) à partir de la base de données de la Mutuelle Sociale Agricole (MSA). Ces classes (tab. I.2) sont inspirées de la classification Orientation Technico-économique (OTEX) utilisée par le RICA. Ces données étant disponibles chaque année, nous avons pu élaborer une série temporelle continue entre 2002 et 2008. La figure I.4 illustre la réparition spatiale de trois de ces OTEA en 2008.

Les données économiques relatives aux OTEA ont été extrapolées à partir des assiettes fiscales disponibles dans la MSA. Ces données sont des proxys des marges brutes réalisées par les agriculteurs. La valeur du facteur d'escompte ρ est étudiée dans les rapports Lebègue (2005) et Chevassus-Au-Louis (2009). Il est proposé de prendre pour le taux d'actualisation une valeur de 4% pour les 30 premières années puis de diminuer linéairement jusqu'à atteindre 3% sur les 30 années suivantes. Pour simplifier, nous avons choisi ici un taux d'actualisation constant à 4% sur les 42 années de le projection. En d'autres termes, le facteur d'escompte ρ vaut $\rho = \frac{1}{1+0.04}$.

Enfin, le budget $Budg_0$ est calibré sur le budget du deuxième pilier de la PAC (montant $1,4.10^9$ euros en 2008), pilier sur lequel repose actuellement une grande partie des enjeux environnementaux de la PAC.

Une analyse de sensibilité a été explorée sur la contrainte de rigidité ϵ du modèle micro-économique (qui peut être interprétée comme une forme de capacité d'innovation) dans l'annexe B (Proceedings ISDA, 2010). Concernant la stabilité de la SAU, bien que les tendances actuelles montrent une tendance à une diminution, les données utilisées ne nous ont pas permis pas d'identifier d'évolutions majeures de la SAU totale.

1. https ://esrcarto.supagro.inra.fr/intranet/



(a) OTEA COP (b) OTEA Bovin lait (c) OTEA Bovin viande

FIGURE I.4 – Répartition spatiale de trois OTEA en 2008.

3.3. Les données de biodiversité

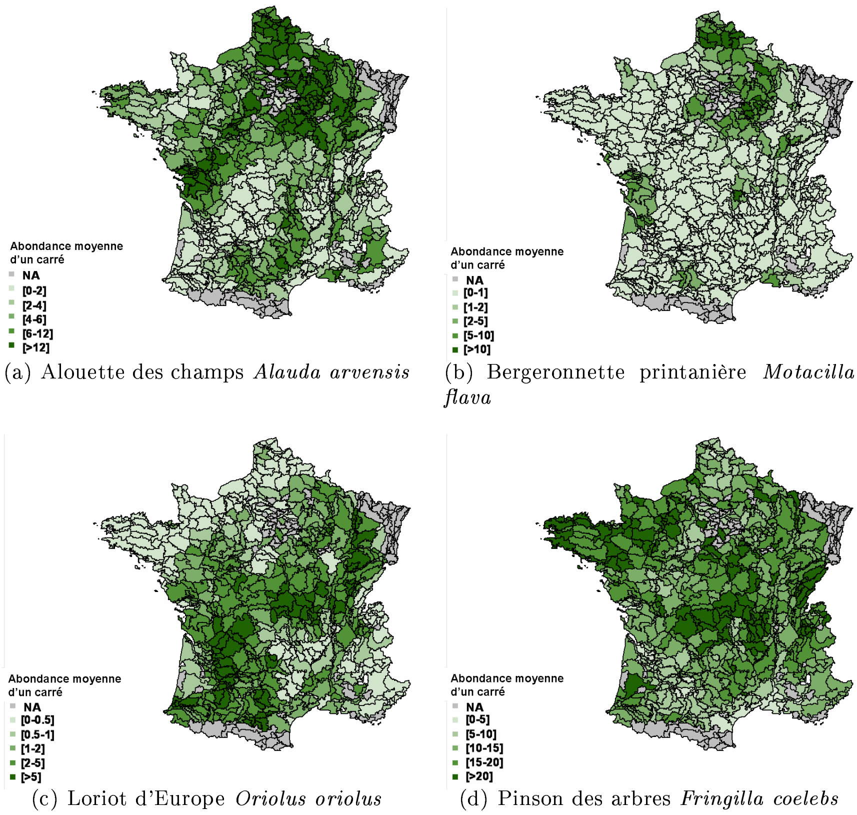
Comme nous l'avons mentionné dans l'introduction, le modèle est appliqué aux oiseaux communs nicheurs. Ce travail se concentre sur des espèces présentes significativement dans le milieu agricole (tab. I.3) : 14 espèces généralistes et 20 espèces spécialistes des milieux agricoles qui ont été identifiées par l'Union Européenne comme marqueurs de l'évolution de la biodiversité en réponse à des changements agricoles (Balmford *et al.*, 2003). La notion de spécialisation est liée au biais de répartition de l'espèce entre les trois grands types d'habitat forestier - agricole - urbain (Julliard *et al.*, 2006). Une espèce généraliste ne présente aucun biais de répartition entre les trois habitats.

20 espèces spécialistes	14 espèces généralistes
(1) Alouette des champs *Alauda arvensis*	(1) Accenteur mouchet *Prunella modularis*
(2) Alouette lulu *Lullula arborea*	(2) Corneille noire *Corvus corone*
(3) Bergeronnette printanière *Motacilla flava*	(3) Coucou gris *Cuculus canorus*
(4) Bruant jaune *Emberiza citrinella*	(4) Fauvette à tête noire *Sylvia atricapilla*
(5) Bruant proyer *Emberiza calandra*	(5) Geai des chênes *Garrulus glandarius*
(6) Bruant zizi *Emberiza cirlus*	(6) Hypolais polyglotte *Hippolais polyglotta*
(7) Buse variable *Buteo buteo*	(7) Loriot d'Europe *Oriolus oriolus*
(8) Caille des blés *Coturnix coturnix*	(8) Merle noir *Turdus merula*
(9) Corbeau freux *Corvus frugilegus*	(9) Mésange bleue *Cyanistes caeruleus*
(10) Faucon crécerelle *Falco tinnunculus*[1]	(10) Mésange charbonnière *Parus major*
(11) Fauvette grisette *Sylvia communis*	(11) Pic-vert *Picus viridis*
(12) Huppe fasciée *Upupa epops*	(12) Pigeon ramier *Columba palumbus*
(13) Linotte mélodieuse *Carduelis cannabina*	(13) Pinson des arbres *Fringilla coelebs*
(14) Pie-grièche écorcheur *Lanius collurio*	(14) Rossignol philomèle *Luscinia megarhynchos*
(15) Pipit farlouse *Anthus pratensis*	
(16) Perdrix grise *Perdix perdix*	
(17) Perdrix rouge *Alectoris rufa*	
(18) Tarier des près *Saxicola rubetra*	
(19) Tarier pâtre *Saxicola rubicola*	
(20) Vanneau huppé *Vanellus vanellus*	

TABLE I.3 – Liste des 20 espèces spécialistes des milieux agricoles et des 14 espèces généralistes étudiées, notées s dans le modèle.

Les données de biodiversité sont fournies par la base de données Suivi Temporel des Oiseaux Communs (STOC) mise en place par le Centre de Recherches par le Baguage des Populations d'Oiseaux (CRBPO) du Muséum National d'Histoire Naturelle. Elles caractérisent des abondances de la population considérée dans son ensemble (pas de structuration en classes d'âge) au moment de la période de reproduction. Les

1. Dont je partage le nom languedocien Mouisset (Dictionnaire de la langue d'Oc, par S-J Honnorat)



(a) Alouette des champs *Alauda arvensis* (b) Bergeronnette printanière *Motacilla flava*

(c) Loriot d'Europe *Oriolus oriolus* (d) Pinson des arbres *Fringilla coelebs*

FIGURE I.5 – Exemples d'aire de répartition pour 4 espèces en 2008.

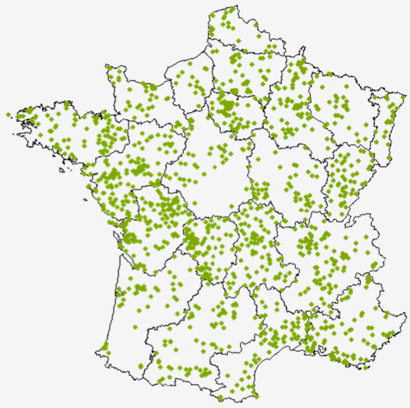
abondances sont disponibles tous les ans entre 2002 et 2009 et sont issues d'une extrapolation spatiale des carrés STOC enquêtés (plus de détails dans l'encadré 2). La figure I.5 présente des exemples d'aire de répartition à l'échelle de la PRA pour trois espèces : l'Alouette des champs *Alauda arvensis*, spécialiste des grandes cultures, la Bergeronnette printanière *Motacilla flava*, spécialiste des zones prairiales qui a colonisé les cultures de colza, ainsi que le Loriot d'Europe *Oriolus oriolus* et le Pinson des arbres *Fringilla coelebs*, deux espèces généralistes.

FIGURE I.6 – Evolution de l'indicateur de variation d'abondance en France, agrégé pour l'ensemble des espèces en noir et les 20 espèces spécialistes en vert (source Jiguet (2010))
.

L'étude de ces données confirment une érosion de la biodiversité. En effet, l'indicateur de variation d'abondance construit par le CRBPO montre une diminution moyenne de 14 % de la taille de la communauté sur l'ensemble des 65 espèces considérées entre 1989 et 2009 (fig. I.6). Ce déclin est plus particulièrement marqué chez les espèces spécialistes pour lesquelles on observe une perte de 25%.

Encadré 2 - La base de données STOC

Nous nous appuyons sur la série temporelle 2002-2009 basée sur la méthode des points d'écoute (STOC-EPS) (Jiguet *et al.*, 2011). Cette base renseigne sur 175 espèces d'oiseaux communs, couvrant ainsi 99% des individus présents en France durant la période de reproduction (Devictor *et al.*, 2008). L'abondance de ces espèces est évaluée au sein de carrés de 2km*2km tirés aléatoirement, grâce à l'observation d'ornithologues amateurs. Le tirage des carrés au hasard permet de balayer un ensemble d'habitats très variés. La série temporelle comporte ainsi 1747 carrés :



Pour chaque carré, l'ornithologue effectue deux visites espacées de 4-6 semaines pendant le printemps (avant et après le 8 mai) au cours desquelles il note toutes les espèces vues et entendues ainsi qu'un descriptif de l'habitat. Chaque carré est découpé régulièrement en 10 points d'écoute qui sont chacun explorés pendant 5 min exactement. Pour chaque espèce, nous sélectionnons l'abondance maximale parmi les deux visites pour chaque point de chaque carré. Nous calculons une abondance moyenne par carré (pas la somme car le nombre de points peut varier entre les carrés). Nous effectuons ensuite une interpolation spatiale (ou krigeage) afin d'estimer les abondances de tous les carrés (environ 136 000 carrés). Cette interpolation est basée sur des modèles spatiaux et une fonction exponentielle. Nous déterminons ensuite l'abondance moyenne d'un carré représentatif de chaque PRA en moyennant les abondances des carrés inclus dans une PRA (pas de somme car pas le même nombre de carrés par PRA). L'intérêt du krigeage est de compléter la base de données et donc d'avoir des informations pour des PRA où aucun carré n'a été exploré. Cela permet aussi de lisser les données et donc de diminuer les spécificités locales et les biais d'observation (lié à l'observateur et au moment de l'observation). Le lissage des spécificités locales (à l'échelle du carré) n'est pas pour nous un gros problème dans la mesure où notre étude porte sur des tendances générales, à large échelle, agrégeant plusieurs espèces. Cela impose cependant une limite à notre travail : il apparaît délicat de tirer des conclusions précises à une échelle locale. La principale limite de la méthode de krigeage est la pression de l'échantillonnage. La répartition des carrés observés n'est pas homogène spatialement, avec des zones très bien couvertes (Poitou-Charente) et d'autres plus vides (Midi Pyrénées). La qualité de l'extrapolation n'est donc pas homogène sur toute la France.

3.4. La calibration des modèles

Les modèles économiques et écologiques font intervenir des paramètres qui nécessitent d'être calibrés à partir des données historiques disponibles. Comme présentés dans l'ensemble des articles, les paramètres de calibration sont déterminés par minimisation des moindres carrés.

Dans le modèle écologique, le taux de croissance $R_{s,r}$ et les réponses $\beta_{s,r,k}$ et $\alpha_{s,r}$ des espèces aux différentes activités agricoles sont déterminés en maximisant l'ajustement entre abondances réelles $N_{s,r}^{Data}(t)$ et abondances prédites $N_{s,r}$:

$$\min_{R,\alpha,\beta} \sum_{s,r,t} (N_{s,r}^{Data}(t) - N_{s,r}(t))^2 \tag{I.39}$$

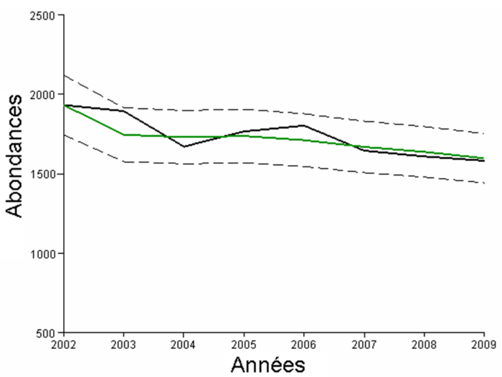
Le contraintes de calibration et les algorithmes choisis sont davantage détaillés dans l'ensemble des articles. L'article développé au chapitre VII (en révision dans Ecological Modelling) apporte davantage de précisions sur la calibration du modèle écologique et explicite notamment différentes dynamiques de population (Ricker, Logistique, Beverton-Holt, Gompertz).

Dans le modèle économique, les paramètres à calibrer sont la rigidité ε et l'aversion au risque a. Ils sont obtenus en minimisant l'écart entre les assolements historiques $A_{r,k}^{Data}(t)$ et les assolements prédits $A_{r,k}(t)$:

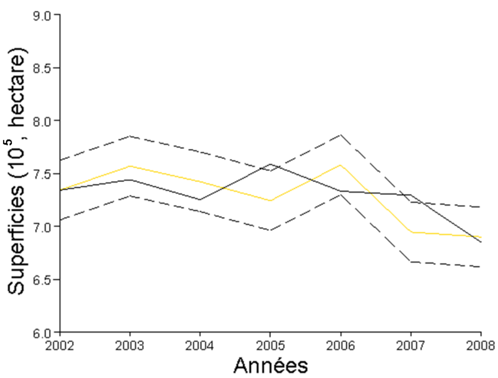
$$\min_{a,\varepsilon} \sum_{r,k,t} (A_{r,k}^{Data}(t) - A_{r,k}(t))^2 \tag{I.40}$$

Une analyse de sensibilité est menée sur les paramètres ε et a. Les valeurs de ε utilisées pour l'analyse de sensibilité sont choisies autour de la valeur obtenue par la calibration (annexe B (Proceedings ISDA, 2010)). Les valeurs de a testées dans l'analyse de sensibilité sont prises dans l'intervalle proposé par Lien (2002), dans lequel se situe aussi la valeur calibrée (chapitre III (Working paper GREThA soumis dans Ecological Applications) et annexe A (Proceedings ISEE, 2010)).

De nombreuses illustrations de ces calibrations comparant les données historiques aux données estimées par les modèles avec un intervalle à 99% sont disponibles dans l'ensemble des articles présentés aux chapitres suivants. Un exemple pour la calibration écologique et un exemple pour la calibration économique sont cependant repris ci-dessous avec la figure I.7. Les données issues des calibrations sont largement cohérentes avec les données historiques. Nous pouvons cependant observer un léger lissage des données, en particulier pour la calibration écologique.



(a) L'abondance nationale du Bruant jaune *Emberiza citrinella*



(b) L'assolement OTEA COP dans une PRA

FIGURE I.7 – Illustrations de la calibration pour le modèle écologique et économique avec en couleur les données réelles et en noir les données prédites avec leur intervalle de confiance à 99% en pointillés.

3.5. Les indicateurs de biodiversité

Deux grands types d'indicateurs de biodiversité *Biod*, comme définis eq. (I.5), sont utilisés dans ce travail : les indicateurs de variation d'abondance de communauté et les indicateurs relatifs à la structure de la communauté. Une différence importante entre ces deux types d'indicateurs est leur construction au niveau national. Comme nous le décrivons dans les équations suivantes, les indicateurs de variation d'abondance sont une agrégation d'indices spécifiques nationaux tandis que les indicateurs de structure sont des moyennes d'indices locaux. La prise en compte des déséquilibres locaux est donc plus importante dans les indicateurs de structures.

Concernant les indicateurs de variation d'abondance, un premier indicateur est disponible pour les espèces spécialistes des milieux agricoles. Le Farmland Bird Index (FBI) a montré dans de nombreuses études sa pertinence pour comprendre l'impact des changements agricoles sur la biodiversité et a été choisi par l'Union Européenne comme indicateur officiel (Balmford *et al.*, 2003). Il est fondé sur une moyenne géométrique des taux de croissance des populations des différentes espèces présentes dans la communauté. Il peut être construit à l'échelle de la PRA (eq. I.41) ou directement au niveau national (eq. I.42), où $N_{s,Nat}(t) = \sum_r N_{s,r}(t)$.

$$FBI_r(t) = \prod_{s=1}^{20} \left(\frac{N_{s,r}(t)}{N_{s,r}(2008)} \right)^{1/20} \tag{I.41}$$

$$FBI(t) = \prod_{s=1}^{20} \left(\frac{N_{s,Nat}(t)}{N_{s,Nat}(2008)} \right)^{1/20} \tag{I.42}$$

Un indicateur relatif aux espèces généralistes a été construit sur le même principe. Le Generalist Bird Index (GBI) est ainsi également défini au niveau de la PRA et national :

$$GBI_r(t) = \prod_{s=1}^{14} \left(\frac{N_{s,r}(t)}{N_{s,r}(2008)} \right)^{1/14} \tag{I.43}$$

$$GBI(t) = \prod_{s=1}^{14} \left(\frac{N_{s,Nat}(t)}{N_{s,Nat}(2008)} \right)^{1/14} \tag{I.44}$$

Trois indicateurs de structure de la communauté sont développés ici :
(i) L'indicateur de de Shannon-Wiener (ShI) mesure le biais par rapport à l'équi-répartition de le communauté entre les 34 espèces présentes (Shannon, 1948). Pour un nombre constant d'espèces, il est maximal quand les espèces sont équi-réparties. Il est défini au niveau de la PRA :

$$ShI_r(t) = -\sum_{s=1}^{34} \frac{N_{s,r}(t)}{N_{tot,r}(t)} \, . \log \left(\frac{\mathrm{N_{s,r}(t)}}{\mathrm{N_{tot,r}(t)}} \right) \tag{I.45}$$

où $N_{tot,r}(t) = \sum_s N_{s,r}(t)$ puis agrégé au niveau national :

$$ShI(t) = \frac{1}{620} . \sum_{r=1}^{620} ShI_r(t) \tag{I.46}$$

(ii) L'indicateur de spécialisation des communautés (CSI) évalue la dépendance de la communauté à un habitat spécifique (Barnagaud *et al.*, 2011). Il est basé sur l'indice de spécialisation des espèces (SSI) (tableau des SSI disponible dans le chapitre IV (article publié dans Ecological Indicators)). Plus le SSI est élevé, plus l'espèce est spécialiste et dépendante d'un milieu spécifique (grandes cultures ou prairies par exemple). Le CSI est construit comme une somme des proportions des différentes espèces présentes dans la communauté pondérées par le SSI :

$$CSI_r(t) = \sum_{s=1}^{34} \frac{N_{s,r}(t)}{N_{tot,r}(t)} \, . \, SSI_s \tag{I.47}$$

Puis il est agrégé au niveau national :

$$CSI(t) = \frac{1}{620} . \sum_{r=1}^{620} CSI_r(t) \tag{I.48}$$

(iii) L'indicateur de niveau trophique des communautés (CTI) caractérise le niveau trophique moyen de la communauté. Il est construit comme la somme des proportions des différentes espèces présentes dans la communauté pondérées par le niveau trophique de l'espèce (STI) (eq. I.49, tableau des SSI disponibles dans le chapitre IV (article publié dans Ecological Indicators)). Ces niveaux trophiques spécifiques sont déterminés en fonction du régime alimentaire de l'espèce (BWPi, 2006). Cet indicateur est une adaptation du Trophic Marine Index (Pauly *et al.*, 1998) à la biodiversité terrestre.

$$CTI_r(t) = \sum_{s=1}^{34} \frac{N_{s,r}(t)}{N_{tot,r}(t)} \, . \, exp(STI_s) \tag{I.49}$$

Il peut ensuite être agrégé au niveau national :

$$CTI(t) = \frac{1}{620} . \sum_{r=1}^{620} CTI_r(t) \tag{I.50}$$

3.6. Un indicateur de diversité agricole

Un indicateur agricole complémentaire aux indicateurs économiques ($Inc(t)$ et PV) est développé afin de mieux comprendre l'organisation de la SAU entre les différentes activités agricoles. Cet indicateur de diversité agricole $Hdiv_r(t)$ est construit comme un indicateur de Simpson des activités agricoles k :

$$Hdiv_r(t) = \left(\sum_{k=1}^{14} \left(\frac{A_{r,k}(t)}{A_r(t)} \right)^2 \right)^{-1} \tag{I.51}$$

Il évalue le bais par rapport à une répartition équi-repartie de la SAU régionale entre les 14 activités agricoles. Un indicateur de diversité agricole moyenne $Hdiv_{Nat}(t)$ est ensuite estimé au niveau national à partir d'une moyenne arithmétique des indicateurs régionaux :

$$Hdiv(t) = \frac{1}{620} \sum_{r=1}^{620} Hdiv_r(t) \tag{I.52}$$

3.7. Les scénarios

Comme nous l'avons décrit à la section 2, nous avons choisi de fonder les scénarios exogènes sur deux types d'incitations. Les incitations pour les grandes cultures τ_{cop} correspondent ici à des incitations à l'OTEA Céréales-Oléagineux-Protéagineux (k=1 dans le tab. I.2). Les incitations aux prairies τ_{grass} sont relatives aux OTEA caractérisant globalement des prairies non intensives (k=4,5,6,7 dans le tab. I.2). Le tableau I.4 décrit les subventions choisies pour les scénarios exogènes, compte tenu de la contrainte budgétaire.

Scénario	CR	GL	DS	HQE
τ_{cop}	0.65	0	0.3	-0.3
τ_{grass}	0	0.55	0.5	0.6

TABLE I.4 – Incitations τ_{cop} et τ_{grass} pour les différents scénarios exogènes Crop (CR), Grassland (GL), Double Subsidies (DS), High Quality Environment (HQE).

Afin de garder une cohérence avec les scénarios exogènes, les scénarios optimaux endogènes et viables sont fondés sur les deux mêmes leviers d'incitation τ_{cop} et τ_{grass}. Le niveau des incitations est déterminé dans l'intervalle [-1,1] selon les modèles décrits précédemment (voir section 2).

Pour les scénarios de viabilité, trois valeurs λ de la force des contraintes eq. (I.30)-(I.31)-(I.32) ont été testées (0.95, 0.97, 1). En d'autres termes pour $\lambda = 1$, le scénario est viable s'il obtient des performances au moins équivalentes à celles obtenues avec le scénario Statu Quo. Avec $\lambda = 0.95$, une dégradation de 5% par rapport au Statu Quo est tolérée.

4. Synthèse des résultats discutés

4.1. Une réconciliation agriculture-biodiversité possible ?

4.1.1. Un choix judicieux de politiques publiques réconcilie agriculture et biodiversité

Nous considérons tout d'abord la politique publique comme outil d'internalisation d'une externalité (Pascual & Perrings, 2007, Pacini, 2004). Cet outil macro-économique est largement utilisé par les décideurs. La question soulevée est la suivante : est-il possible d'orienter les performances bio-économiques par une distorsion des revenus au moyen d'un instrument prix (taxe-subvention) ? Cette question est plus spécifiquement explorée dans le chapitre II (article publié dans Ecological Economics, 2011) et l'annexe B (Proceedings ISDA, 2010).

La figure I.8 [1] présente les performances bio-économiques obtenues avec les différents scénarios de référence et exogènes dans un cadre déterministe (l'aversion au risque a est nulle) au travers du revenu inter-temporel escompté (PV) et du Farmland Bird Index (FBI) en 2050. La comparaison des scénarios Statu Quo (SQ) (scénario où l'occupation des sols est figée à celle de 2008) et Laissez Faire (LF) (scénario où l'ensemble des incitations τ sont nulles) illustre une dégradation des performances écologiques en l'absence de politiques publiques. En d'autres termes, sans intervention politique spécifique, les décisions économiques ne permettent pas le maintien de la biodiversité. Nous retrouvons ainsi, dans le cadre de l'agriculture, la conclusion qu'Alavalapati *et al.* (2002) avait illustrée pour l'exploitation forestière.

La figure I.8 confronte également les performances obtenues avec les scénario de référence (SQ et LF) avec celles obtenues avec des scénarios de politiques publiques exogènes (Crop CR, Grassland GL, Double Subsidies DS, et High quality Environment HQE). Nous observons alors que la présence de politiques publiques modifient les résultats bio-économiques. Nous illustrons donc qu'une politique publique fondée sur un instrument prix (taxe-subvention) est un outil efficace pour modifier le choix des agents dans un cadre détermiste. Nous étendons ainsi les travaux de Shi & Gill (2005), qui, dans le cadre de l'agriculture chinoise, ont souligné l'importance de 'Etat pour modifier les décisions des agents économiques au travers d'autres mesures (faible taux d'intérêt sur les investissements, soutien du gouvernement pour l'apprentissage).

1. La figure I.8 propose une nouvelle interprétation des résultats de l'article du chapitre II, notamment au travers des Present Value. De plus, afin de garder une cohérence au sein de ce chapitre I, les valeurs ont été recalculées avec le modèle construit à l'échelle de la PRA et fondé sur les bases de données utilisées dans les chapitres III à VI.

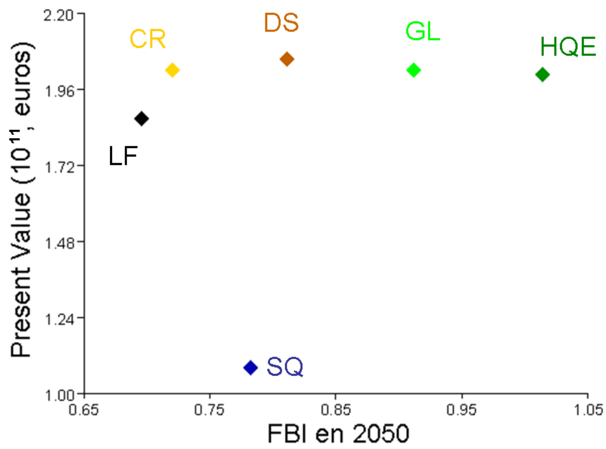


FIGURE I.8 – Comparaison des performances bio-économiques obtenus avec les différents scénarios de références et exogènes. La performance écologique est évaluée au travers du FBI en 2050 et la performance économique au travers du revenu inter-temporel escompté (Present Value PV). Les scénarios de référence sont le scénario Laissez-Faire (LF) et le scénario Statu Quo (SQ) et les 4 scénarios exogènes sont les scénarios Crop (CR), Double Subsidies (DS), Grassland (GL) et High Quality Environment (HQE).

Notre travail nous permet d'approfondir ces conclusions, en illustrant que certains scénarios (DS, GL, HQE) conduisent à une amélioration simultanée des performances économiques et écologiques par rapport à celles obtenues avec les scénarios de référence (scénarios LF et SQ). En ce sens, ils permettent une réconciliation bio-économique de l'agriculture et de la biodiversité. Ce sont les politiques de retour à l'herbe qui conduisent à cette réconciliation et non les scénarios de développement des grandes cultures. Cette conclusion est cohérente avec l'ensemble des travaux illustrant l'importance des prairies extensives pour la biodiversité (Batary *et al.*, 2011, Laiolo, 2005, Bignal & McCracken, 2000, Potter & Goodwin, 1998). Le choix de l'orientation générale de la politique est donc déterminant dans la réconciliation. Cependant, aucun scénario parmi les scénarios DS-GL-HQE n'apparaît optimal simultanément sur les deux critères. En effet, plus le scénario stimule ce retour à l'herbe, plus l'effet sur la biodiversité est positif mais la performance économique est moindre. La détermination de la politique publique capable de réconcilier de manière optimale agriculture et biodiversité n'est donc pas triviale. Nous revenons sur cette idée dans la section 4.2.

4.1.2. L'aversion au risque stimule la diversification et limite la destruction de la biodiversité

Dans le chapitre III (Working paper GREThA, soumis dans Ecological Applications) et l'annexe A (Proceedings ISEE, 2010), nous nous sommes intéressés à un paramètre micro-économique : l'aversion au risque (notée a dans le modèle). Il ne s'agit pas à proprement parler d'un outil de décision mais plutôt d'une caractéristique intrinsèque des agents. Cependant, le contexte économique, notamment le niveau de protection contre le risque découlant d'une politique plus ou moins protectionniste, peut modifier les choix des agriculteurs : par exemple, dans un contexte extrêmement protégé (comme c'est le cas avec la PAC historique), l'agriculteur est peu soumis au risque et peut donc prendre des décisions proches de celles d'un comportement "neutre au risque". Il nous a semblé intéressant d'explorer cette caractéristique micro-économique d'aversion au risque, et de comprendre comment elle modifie les performances bio-économiques.

La figure I.9 compare les deux critères économiques et écologiques pour le scénario Statu Quo et le scénario Laissez-Faire avec différents niveaux d'aversion au risque. Nous observons un frontière d'efficience convexe entre les performances économiques et écologiques selon le gradient d'aversion au risque. Ceci illustre que

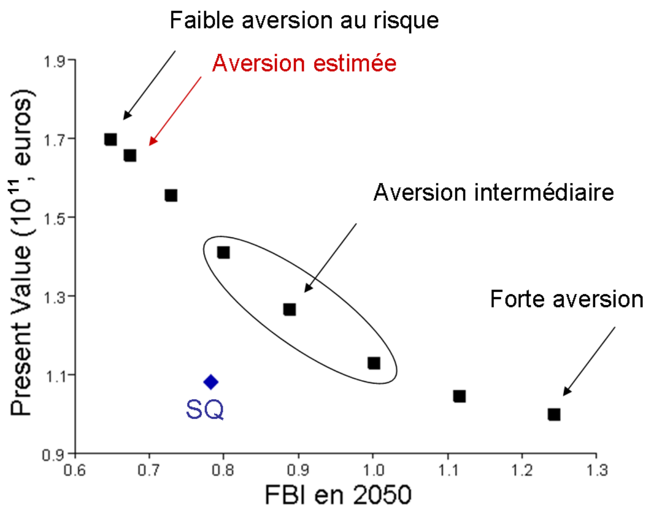
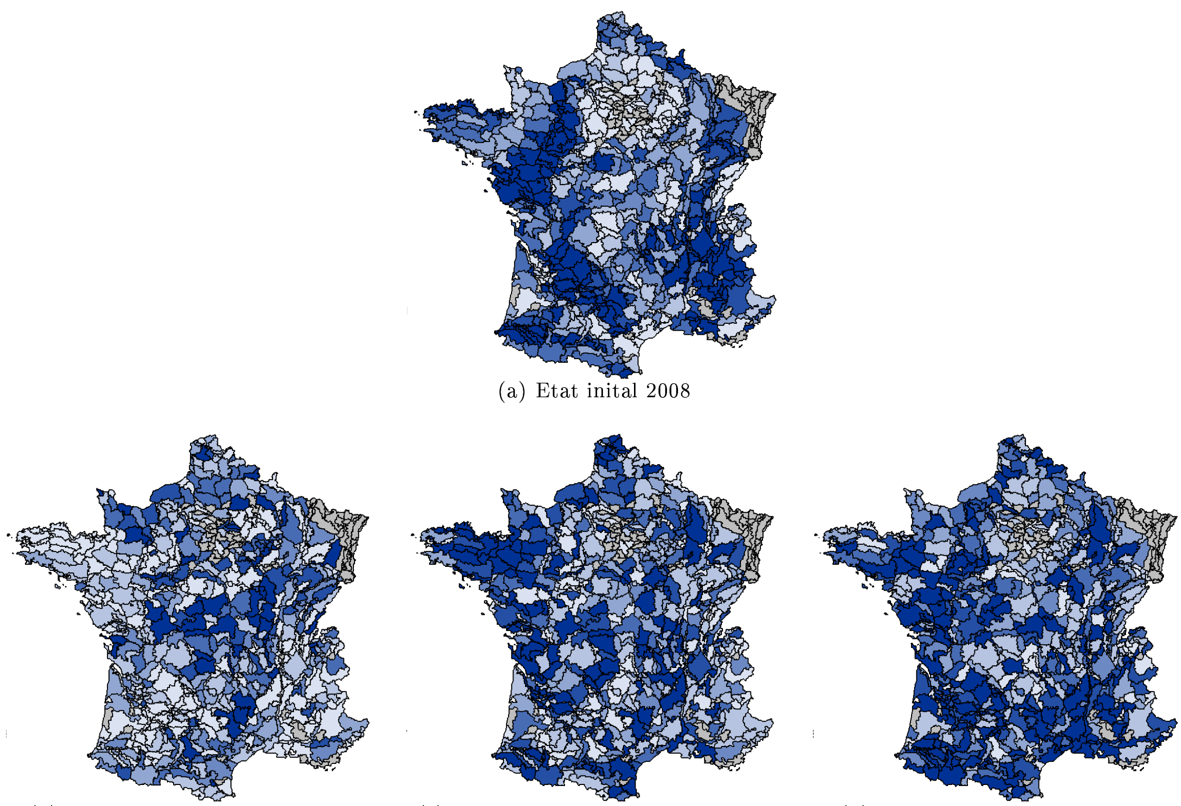


FIGURE I.9 – Impact de l'aversion au risque sur les performances bio-économiques dans le cas du scénario Laissez-Faire (LF). La performance écologique est évaluée au travers du FBI en 2050 et la performance économique au travers du revenu inter-temporel escompté (Present Value PV). Huit scénarios LF en noir avec différents niveaux d'aversion au risque sont comparés avec le scénario Statu Quo (SQ) en bleu.

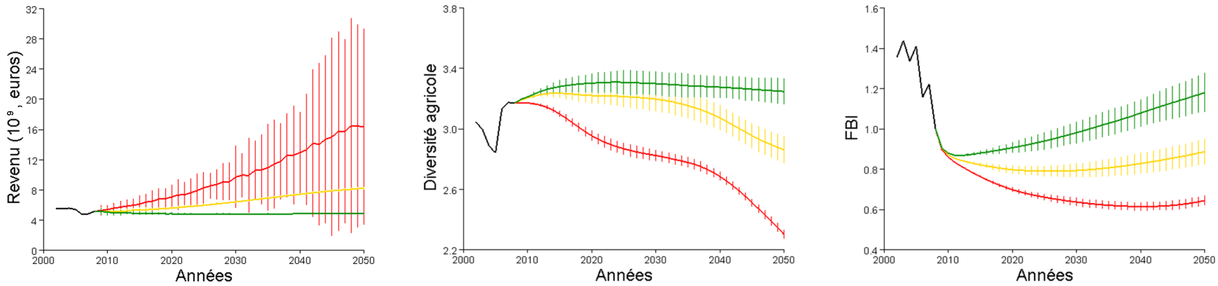
l'aversion au risque est un paramètre ayant un réel impact sur les performances bio-économiques. Nous étendons ainsi les résultats de Quaas *et al.* (2007) obtenus dans des zones semi-arides, illustrés ici dans le cas d'une agriculture en zones tempérées.



(a) Etat inital 2008

(b) 2050 avec une faible aversion (c) 2050 avec une aversion moyenne (d) 2050 avec une aversion forte

FIGURE I.10 – Comparaison de l'indicateur de diversité agricole à l'échelle de la PRA $Hdiv_r(t)$ en 2008 et 2050 selon le niveau d'aversion au risque (bleu foncé : diversité forte, bleu pâle : faible diversité, gris : NA).



(a) Revenu $Inc(t)$ (b) Diversité agricole $Hdiv(t)$ (c) Farmland Bird Index $FBI(t)$

FIGURE I.11 – Trajectoires moyennes des performances obtenues avec une aversion faible (en rouge), intermédiaire (en jaune), forte (en vert) en contexte incertain. Les barres verticales représentent les intervalles de confiance à 99%.

Le mécanisme sous-jacent est largement développé dans l'article du chapitre III. En résumé, cette frontière est pilotée par le niveau de diversité des cultures choisi par l'agriculteur. Comme l'illustre la figure I.10, la diversité des activités croît avec l'aversion au risque. En accord avec la théorie de gestion de portefeuilles, un agent très averse au risque diversifie ses activités afin de minimiser le risque au lieu de développer au maximum l'activité la plus rentable. La performance économique obtenue en moyenne sera donc plus faible, mais avec un risque réduit. Nous l'illustrons avec la figure I.11. Nous généralisons ainsi les travaux de Baumgärtner & Quaas (2010), Di Falco & Perrings (2003) et Schläpfer *et al.* (2002), qui ont illustré un phénomène similaire avec l'agrobiodiversité -diversité génétique- au sein d'une même culture. En favorisant l'hétérogénéité des habitats (Benton *et al.*, 2003), cette diversification des activités a un impact positif sur la biodiversité sans l'intervention directe d'une politique publique.

La comparaison du scénario SQ et du set de scénarios LF montre que, pour des valeurs intermédiaires d'aversion au risque, il est possible d'améliorer simultanément les performances économiques et écologiques par rapport au maintien de la situation actuelle (SQ). En ce sens, l'aversion au risque est un paramètre jouant favorablement sur la réconciliation agriculture-biodiversité. Le déplacement de la valeur actuelle d'aversion au risque (relativement faible puisque politique protectionniste) vers des valeurs intermédiaires grâce à un contexte politique adapté pourrait être un levier intéressant pour la réconciliation agriculture-biodiversité.

4.1.3. Les politiques publiques adaptées améliorent les performances bio-économiques quel que soit le niveau d'aversion au risque

Nous avons ensuite étudié l'effet des politiques publiques sur les performances bio-économiques pour plusieurs valeurs d'aversion au risque (annexe A , Proceedings ISEE, 2010). Cette étude confronte les deux leviers (macro-économique et micro-économique) afin d'identifier le plus déterminant dans la réconciliation entre agriculture et biodiversité. Dans cette synthèse, nous focalisons notre attention sur les niveaux intermédiaires d'aversion au risque qui ont permis l'amélioration des performances bio-économiques.

La figure I.12 montre que les politiques publiques ont un effet sur les performances bio-économiques par rapport au Statu Quo quel que soit le niveau d'aversion au risque. Une politique de retour à l'herbe aura donc systématiquement un impact positif pour la réconciliation agriculture-biodiversité. Cependant, le bénéfice marginal apporté par les politiques publiques s'atténue avec le niveau d'aversion au risque. L'intuition sous-jacente, développée dans l'article en annexe A, est que lorsque l'aversion au risque augmente, les agents sont de moins en moins sensibles aux rentabilités relatives des différentes activités pour privilégier de plus en plus la diversité des cultures (mécanisme de gestion du risque). Les politiques, qui jouent sur les rentabilités relatives des différentes activités, ont donc naturellement moins d'impact. Cependant, en illustrant l'effet

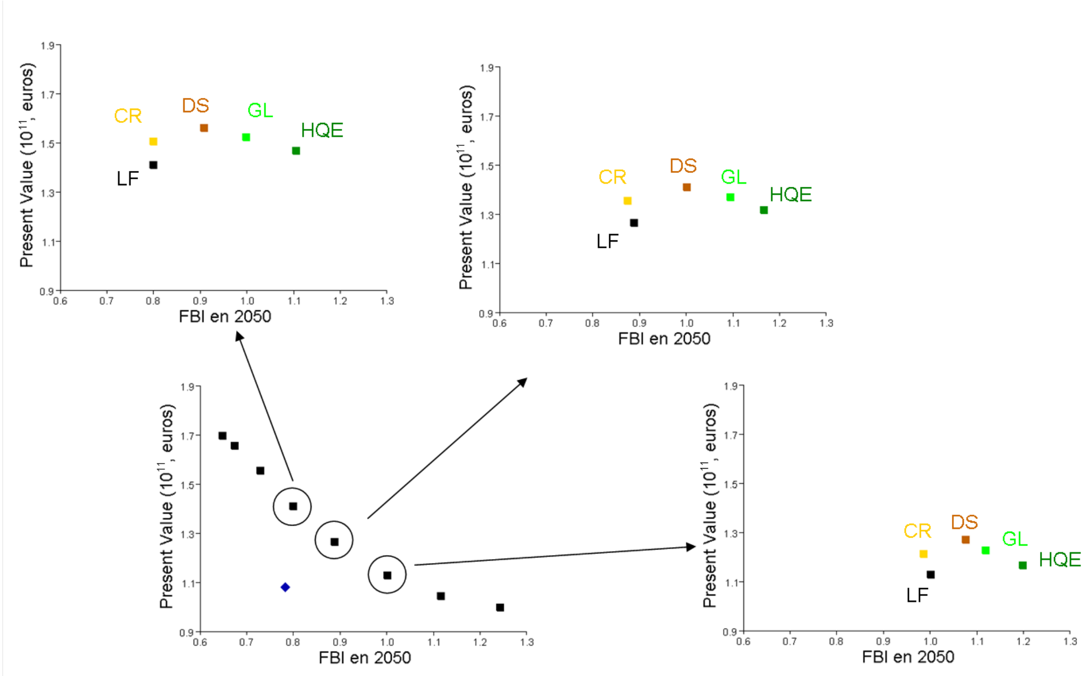


FIGURE I.12 – Effet des scénarios exogènes sur les performances bio-économiques en fonction de 3 niveaux d'aversion au risque intermédiaires. La performance écologique est évaluée au travers du FBI en 2050 et la performance économique au travers du revenu inte-temporel escompté (Present Value PV). Le scénario de base (Laissez-Faire LF) est comparé aux 4 scénarios exogènes (Crop CR, Double Subsidies DS, Grassland GL et High Quality Environment HQE).

systématique des politiques publiques, nous justifions leur utilisation dans la réconciliation agriculture-biodiversité. La confrontation des deux effets micro et macro-économiques nous permet de renforcer les conclusions de Shi & Gill (2005) et Alavalapati *et al.* (2002) sur la nécessité des politiques pour mettre en place une agriculture durable. Par ailleurs, un travail complémentaire (voir encadré 3) illustre que, même s'il est possible de réduire en partie le niveau des incitations, leur utilisation reste indispensable pour maintenir un impact sur la biodiversité. Ces différents résultats confirment que la poursuite des réflexions sur la construction de politiques viables demeure une perspective stimulante.

ENCADRÉ 3 - Une diminution modérée des incitations ne modifie pas l'effet sur la biodiversité
Nous avons étudié un cas particulier où les incitations étaient décroissantes au cours du temps. Cette diminution permet d'étudier la durabilité de l'agro-écosystème dans le cadre d'un désengagement progressif de la PAC. Nous avons choisi une fonction simple : une diminution linéaire en fonction du temps où l'incitation est maximale au début de la projection t_0 et atteint zéro à l'horizon temporel T :

$$\tau_k(t) = \tau_{r,k}(t_1)(1 - \frac{t - t_1}{T - t_1}) \tag{I.53}$$

L'article du chapitre IV (publié dans Ecological Indicators, 2012) se concentre sur l'impact de cette diminution sur les performances écologiques. Cette étude a permis de mettre en évidence que les scénarios qui présentaient des incitations diminuant au cours du temps pouvaient conserver un effet suffisant sur les agriculteurs pour impacter les communautés d'oiseaux. Cependant, une diminution trop importante entraîne un retour vers les comportements du scénario Statu Quo et le bénéfice sur la biodiversité est perdu. Cela suggère donc qu'une réduction modérée du volume des incitations est possible, permettant de conserver l'impact sur la biodiversité tout en libérant un budget qui pourra être réinvesti différemment.

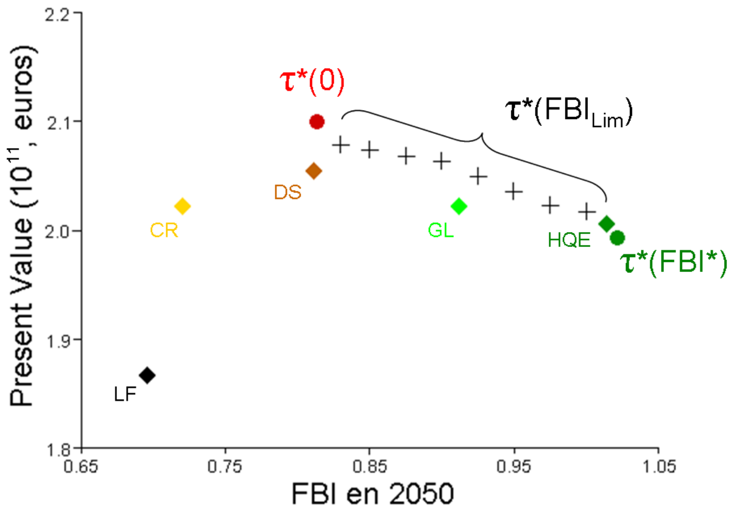


FIGURE I.13 – Comparaison des performances bio-économiques obtenues avec les scénarios de référence Laissez-Faire (LF), exogènes et optimaux endogènes. La performance écologique est évaluée au travers du FBI en 2050 et la performance économique au travers du revenu inter-temporel escompté (Present Value PV). Les 4 scénarios exogènes sont les scénarios Crop (CR), Double Subsidies (DS), Grassland (GL) et High Quality Environment (HQE). Les scénarios optimaux endogènes $\tau^*(FBI_{Lim})$ (croix noires) sont issus de la maximisation de la PV sous contrainte FBI. Le scénario endogène $\tau^*(0)$ (rond rouge) correspond à la maximisation de la PV sans contrainte FBI, et le scénario endogène $\tau^*(FBI^*)$ (rond vert) est déterminé par l'optimisation du FBI.

4.2. Peut-on optimiser la réconciliation ?

Nous avons vu qu'il était possible d'obtenir simultanément de meilleures performances économiques et écologiques par rapport à des situations de référence. En revanche est-il possible d'optimiser simultanément ces deux performances ? En d'autres termes, peut-on définir un scénario de politique publique permettant d'avoir à la fois la meilleure performance économique et la meilleure performance écologique ?

4.2.1. Il n'est pas possible de maximiser simultanément objectifs économiques et écologiques

Cette question de l'optimisation de la réconciliation au travers des scénarios endogènes est plus spécifiquement explorée dans le chapitre V (article soumis dans Ecological Economics).

Cette étude permet de retrouver, dans le cadre d'un cas d'étude fondé sur un modèle calibré, la frontière d'efficience entre performances économiques et écologiques décrites dans les travaux théoriques de Barraquand & Martinet (2011) et Polasky *et al.* (2005). La figure I.13 illustre que l'intégration d'une contrainte de biodiversité puis son augmentation entrainent inévitablement une diminution de la performance économique optimale, c'est-à-dire une perte économique par rapport à une situation sans contrainte de biodiversité (scénario $\tau^*(0)$). En ce sens, il n'est donc pas possible d'optimiser la réconciliation bio-économique.

FBI_{lim}	0	0.825	0.85	0.875	0.9	0.925	0.95	0.975	1	FBI*
τ^*_{cop}	0.47	0.27	0.23	0.23	0.14	0.02	-0.06	-0.19	-0.25	-0.54
τ^*_{grass}	0.52	0.58	0.59	0.58	0.61	0.62	0.61	0.62	0.62	0.63

TABLE I.5 – Incitations optimales τ^*_{cop} et τ^*_{grass} pour différentes contraintes de biodiversité FBI_{lim}.

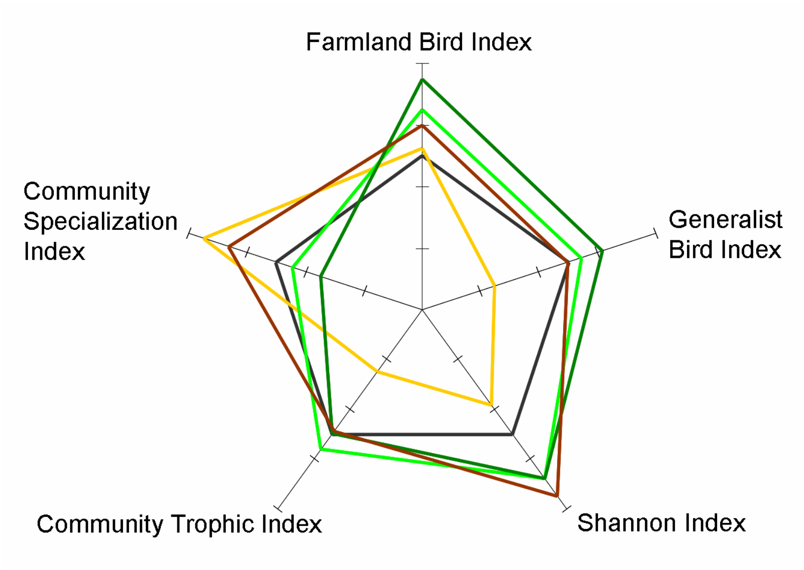


FIGURE I.14 – Performances écologiques obtenues en 2050 en réponse au scénario de référence (Laissez-Faire en noir) et aux 4 scénarios exogènes (Crop en jaune, Double Subsidies en marron, Grassland en vert clair et High Quality Environment en vert foncé), évaluées au travers de 5 indicateurs différents.

L'article du chapitre V étudie les mécanismes sous-jacents à cette frontière d'efficience. Elle s'explique par une évolution des incitations (tab. I.5) : une quasi-stabilité des subventions aux prairies non intensives à des niveaux élevés associée à une diminution nette des subventions distribuées aux COP jusqu'à une taxation quand la contrainte biodiversité augmente. Nous retrouvons ici la conclusion obtenue avec les scénarios exogènes : un retour à l'herbe poussé est favorable aux oiseaux mais moins intéressant au regard du revenu. Du point de vue de l'occupation des sols, cela se traduit par le développement des prairies non intensives au détriment des COP, ces dernières ayant un impact globalement négatif sur la communauté.

Cet article nous permet d'étendre l'ensemble de ces conclusions, en illustrant cette frontière d'efficience avec un autre critère écologique, le CTI (Community Trophic Index). Bien que la forme soit légèrement différente (davantage coudée), la conclusion reste la même : il n'est pas possible de maximiser simultanément performances économiques et écologiques.

4.2.2. Les indicateurs écologiques répondent différemment aux scénarios

Un focus sur les indicateurs écologiques et leurs évolutions en réponse à différents scénarios a été mené dans le chapitre IV (article publié dans Ecological Indicators, 2012). Ce travail a notamment permis de mettre en évidence que, même au sein des indicateurs écologiques, les réponses aux différents scénarios peuvent être antagonistes (fig. I.14).

En prenant l'exemple des quatre scénarios exogènes, cette étude montre que chaque indicateur classe les scénarios de manière différente (à part le FBI et le GBI). Il n'est donc pas possible de maximiser simultanément tous les critères écologiques. La croissance de la communauté (FBI et GBI) est stimulée par des politiques poussées de retour à l'herbe (scenario HQE). Une politique de retour à l'herbe plus modérée conduit à une population au niveau trophique le plus fort (scénario GL). L'équi-répartition des espèces est améliorée par des politiques mixtes proches des politiques actuelles (scénario DS). Enfin, la protection des spécialistes est maximale avec des politiques favorisant le développement des grandes cultures (scénario CR). Ce dernier point est cohérent avec les travaux de Doxa *et al.* (2010) et se justifie par le développement de communautés largement dominées par des espèces granivores très spécialisées, comme la perdrix grise *Perdix perdix*. En d'autres termes, le scénario Grassland favorise le développement de prairies, donc de fleurs, impliquant la présence d'insectes. La proportion d'espèces insectivores, au niveau trophique élevé et au degré de spécialisation faible, est donc important dans les communautés soumises à ce scénario. Inversement, le scénario Crop stimule le développement de grandes cultures et donc la présence de graines. Les espèces granivores, au niveau trophique faible et très spécialisées, envahissent la population. Cette conclusion conduit également à soulever le débat de la pertinence de l'indicateur CSI pour évaluer l'état de santé des communautés d'oiseaux en réponse à des pressions agricoles (voir encadré 4).

Ces différents arbitrages (écologie-économie et écologie-écologie) illustrent que, si plusieurs critères peuvent être améliorés simultanément par rapport aux scénarios de référence avec une politique de retour à l'herbe, ils ne peuvent pas forcément être maximisés en même temps. Déterminer des critères "prioritaires" pour décrire l'écosystème semble être une première étape importante pour limiter ces arbitrages. Le chapitre IV a permis de réfléchir plus précisément à l'arbitrage entre différents indicateurs écologiques. Cependant, face à la persistance de certains antagonismes, réfléchir à des approches permettant de passer outre ces antagonismes apparaît comme une perspective nécessaire pour la réconciliation agriculture-biodiversité.

Encadré 4 - Combiner le FBI, le CTI et le CSI pour évaluer les politiques publiques
Comme nous le mentionnions dans l'introdution, le problème de l'évaluation de la biodiversité constitue encore aujourd'hui un enjeu majeur. Le chapitre IV (article publié dans Ecological Indicators, 2012) a permis de mettre en évidence que le combinaison des trois indicateurs FBI, CTI et CSI est intéressante pour étudier l'impact des politiques publiques sur les communautés d'oiseaux. Cette combinaison permet de décrire à la fois la croissance et la structure de la population. Concernant la croissance, le FBI classe les scénarios de la même manière que le GBI mais avec une plus grande amplitude. Concernant la structure, l'association du CTI et du CSI permet de balayer un grand gradient de paysages agricoles. Le CSI renseigne correctement sur l'état de santé d'une communauté dans les milieux prairiaux (prairies) ou mixtes. Cependant, dans les milieux ouverts (milieux céréaliers), son estimation n'est plus représentative de la qualité de la communauté car il augmente sous l'effet de quelques espèces granivores très spécialistes. Cette conclusion est confirmée par l'analyse du CTI, qui indique alors un très faible niveau trophique moyen. Le CTI est donc un complément intéressant au CSI pour repérer les zones dans lesquelles l'analyse du CSI n'est plus pertinente. De plus, du point de vue de la communication, le choix d'un petit groupe d'indicateurs facilite la compréhension du message et l'intégration d'un indicateur institutionnalisé (FBI) renforce sa crédibilité politique. L'objectif d'une politique publique serait alors de conduire à une communauté en croissance (FBI) et diversifiée en termes de niveau trophique (CTI) et de spécialisation (CSI). Une telle communauté est robuste face aux changements globaux (Keesing *et al.*, 2010) et cohérente avec une perspective de durabilité.

4.3. Comment dépasser l'antagonisme entre performances économiques et performances écologiques ?

4.3.1. La contrainte biodiversité génère un double dividende pour l'ensemble de la société

La première approche pour dépasser cet antagonisme, proposée dans le chapitre V (article soumis dans Ecological Economics), consiste à changer le référent économique. Au lieu de se focaliser sur les agriculteurs et leur manque à gagner suite à l'introduction de la contrainte de biodiversité, nous considérons ici les coûts monétaires subis par l'ensemble de la société, i.e. les agriculteurs et l'État. Les politiques intégrant des contraintes de biodiversité croissantes sont alors analysées selon le coût privé (manque à gagner de l'agriculteur) mais aussi le coût public (budget dépensé pour la politique) et enfin le coût social total défini comme la somme des deux coûts précédents comme le proposent Naidoo *et al.* (2006).

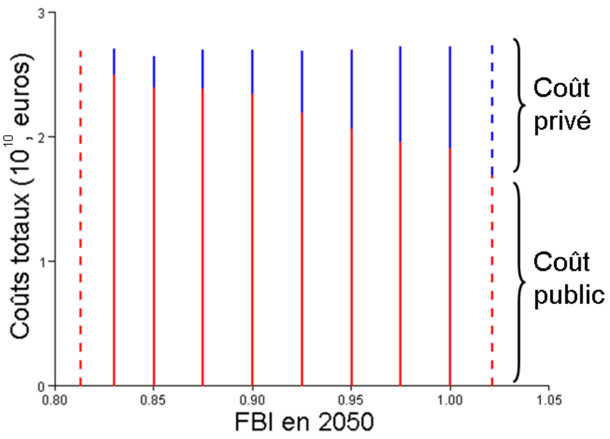
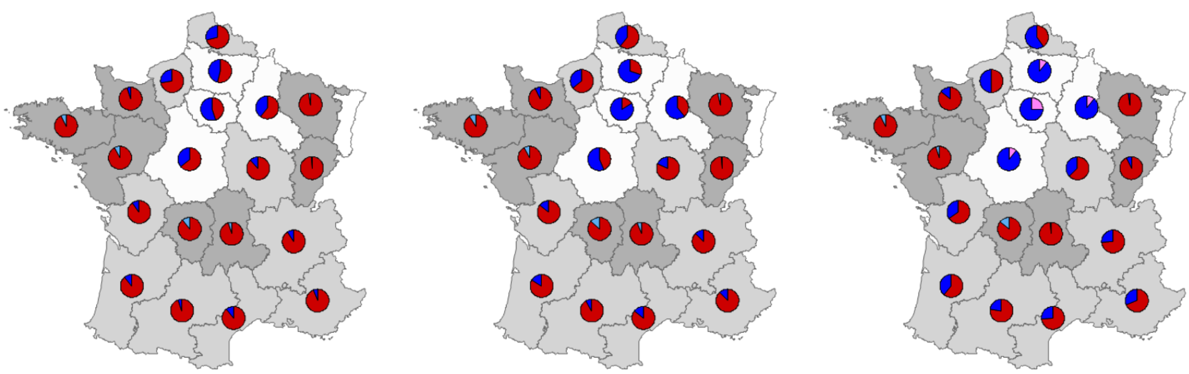


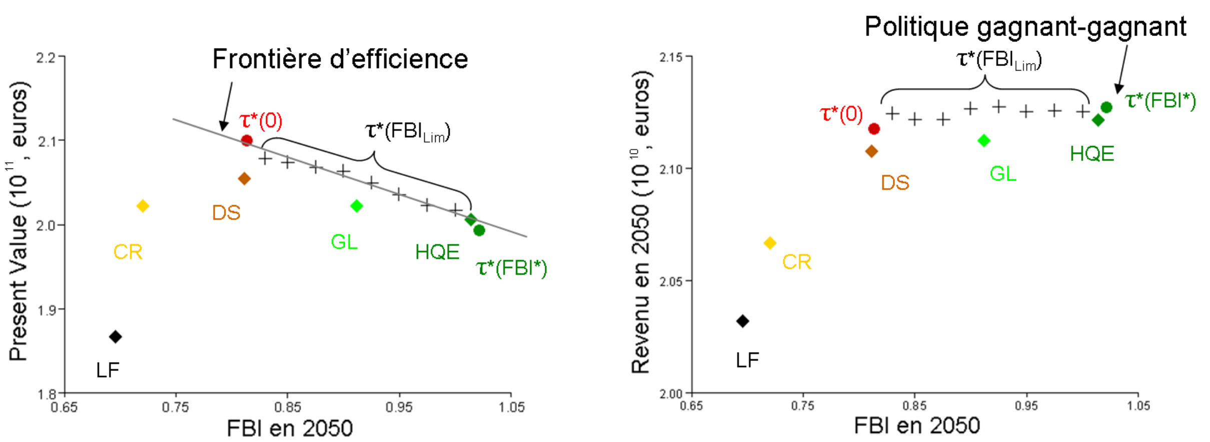
FIGURE I.15 – Comparaison des coûts privés (en bleu), publics (en rouge) et totaux (somme du rouge et bleu) pour les scénarios endogènes avec une contrainte biodiversité FBI croissante. En pointillés à gauche, scénario maximisant la Present Value sans contrainte écologique ($\tau^*(0)$). En pointillés à droite, scénario permettant la maximisation de l'indicateur écologique ($\tau^*(FBI^*)$).

Nous montrons avec la figure I.15 que le coût total est quasi-stable avec l'augmentation de la contrainte de biodiversité. Ainsi, la politique avec la contrainte de biodiversité la plus forte, qui a le même coût que les autres mais une performance écologique optimisée, apparaît comme une politique optimale (gagnant-gagnant) d'un point de vue bio-économique : elle optimise la performance écologique sans affecter la performance économique (coût total). Cette stabilité s'explique par une complémentarité forte (mais non exacte) entre coûts privés et coûts publics. Comme le décrit l'article du chapitre V, lorsque la contrainte de biodiversité augmente, nous observons que les subventions distribuées aux COP diminuent puis prennent la forme de taxe. Si cela induit un manque à gagner pour les agriculteurs (i.e. augmentation du coût privé), cela réduit également le volume du budget distribué par l'État (i.e. diminution du coût public). Ce phénomène se retrouve aussi au niveau régional (région administrative) (fig. I.16). En effet, les régions les plus touchées par l'augmentation du coût privé sont les régions historiquement orientées vers les grandes cultures (Picardie, Ile de France, Centre) et actuellement les plus grandes bénéficiaires de la PAC. Suite à l'intégration



(a) scénario $\tau^*(FBI > 0.85)$ (b) scénario $\tau^*(FBI > 0.925)$ (c) scénario $\tau^*(FBI^*)$

FIGURE I.16 – Représentation régionale des coûts privés (en bleu), publics (en rouge) pour trois scénarios endogènes. Le rose et le bleu pâle caractérisent des coûts négatifs, ie des bénéfices. Les régions en blanc (resp. gris clair, gris foncé) sont celles qui présentent une forte (resp. moyenne, faible) évolution des coûts en réponse à l'augmentation de la contrainte biodiversité.



(a) avec le revenu escompté

(b) avec le revenu en 2050

FIGURE I.17 – Comparaison des classements des scénarios de référence (Laissez-Faire), exogènes et endogènes en fonction de deux indicateurs économiques. La performance écologique est évaluée au travers du FBI en 2050 et la performance économique au travers du revenu inter-temporel escompté (Present Value PV) sur la fig. (a) et au travers du revenu en 2050 sur la fig. (b). Les 4 scénarios exogènes sont les scénarios Crop (CR), Double Subsidies (DS), Grassland (GL) et High Quality Environment (HQE). Les scénarios endogènes $\tau^*(FBI_{Lim})$ (croix noires) sont issus de la maximisation de la PV sous contrainte FBI. Le scénario endogène $\tau^*(0)$ (rond rouge) correspond à la maximisation de la PV sans contrainte FBI, et le scénario endogène $\tau^*(FBI^*)$ (rond vert) est déterminé par l'optimisation du FBI.

de la contrainte biodiversité, ce sont elles qui permettent une diminution du budget (coût public) voire un bénéfice public (le volume des taxes est supérieur au volume des incitations). Ainsi, une redistribution de ces gains pourrait être envisagée au niveau régional pour réduire le manque à gagner des agriculteurs les plus touchés et augmenter l'acceptation de la politique la plus contraignante. Ce système de redistribution maintiendrait la stabilité des coûts totaux et ne modifierait donc pas la conclusion initiale : la politique la plus contraignante écologiquement est optimale. Le mécanisme comprendrait donc deux étapes : un outil prix mixte subvention-taxe puis une subvention récompensant le changement (aux agriculteurs effectuant le changement). Ce système serait plus efficace économiquement qu'une augmentation des subventions aux prairies puisque la seconde subvention serait distribuée spécifiquement aux agriculteurs ayant un manque à gagner, évitant ainsi l'effet d'aubaine des agriculteurs qui ont déjà adopté l'activité subventionnée. En revanche, cette proposition de redistribution, telle qu'elle est proposée ici, ne permet pas de réduire les disparités régionales, ni d'aller vers une réduction de l'intervention politique. Les modalités de cette redistribution restent encore à définir, mais l'échelle régionale semble être une échelle intéressante permettant de cibler les régions en fonction de leurs caractéristiques.

Le même phénomène a été retrouvé avec un autre indicateur de biodiversité, le CTI (article du chapitre V), laissant présager ainsi d'un certain niveau de généricité.

4.3.2. Changer de préférence temporelle permet d'identifier des politiques "gagnant-gagnant"

Pour des raisons sociales et politiques, il y a, dans les prises de décisions, une large prépondérance du présent. Nous illustrons avec le chapitre II (article publié dans Ecological Economics, 2011) que les politiques optimales à long terme ne sont pas les plus intéressantes à court terme. Cette préférence pour le présent limite donc l'élaboration de politiques optimales à long terme.

Nous montrons sur la figure I.17 [1] que la comparaison des performances économiques escomptées (Revenu

1. A nouveau, cette figure est une nouvelle interprétation des résultats obtenus dans l'article du chapitre II. Des valeurs sont également re-calculées avec le modèle construit à l'échelle de la PRA.

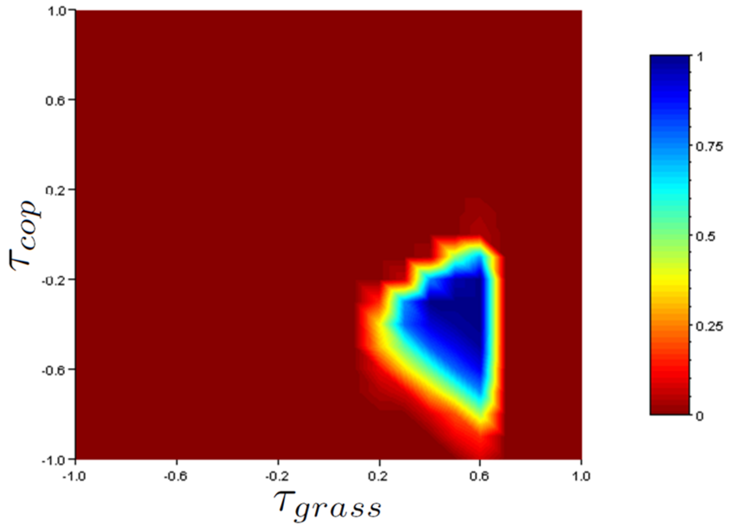


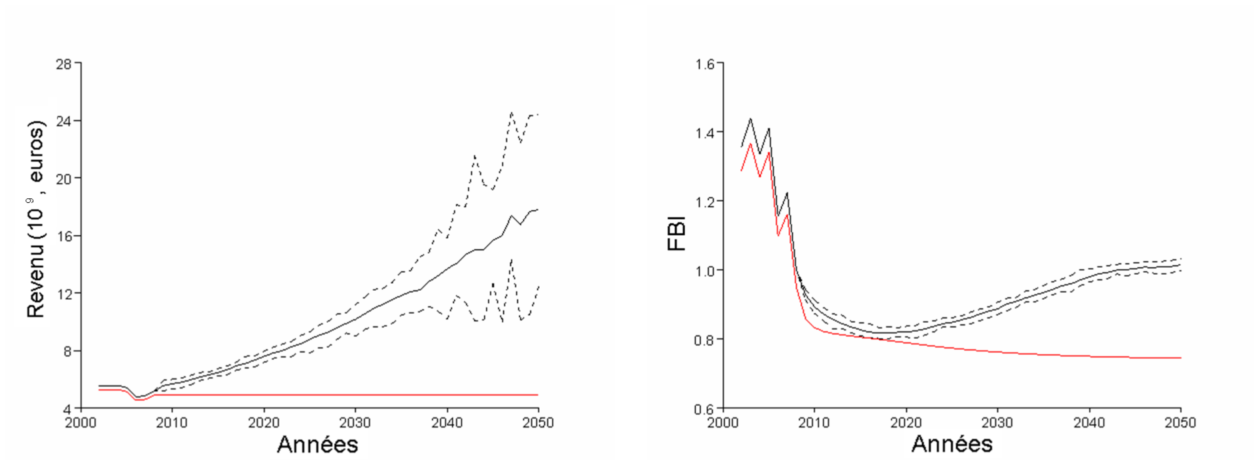
FIGURE I.18 – Probabilité des différents scénarios de satisfaire simultanément à chaque pas de temps la contrainte de budget, la contrainte économique à 95% et les trois contraintes écologiques (FBI, CTI, CSI) à 95 %. En bleu foncé, la probabilité est maximale (CVA=1), en rouge foncé elle est minimale (CVA=0).

intertemporel escompté) et des performances économiques à long terme de différents scénarios (Revenu en 2050) ne conduit pas aux mêmes conclusions. Si l'on s'intéresse au revenu en 2050 au lieu du revenu actualisé, comme le propose Solow (1956), les politiques sous contraintes de biodiversité croissantes présentent des performances économiques quasi-stables. La politique la plus performante en termes de biodiversité apparaît alors comme une politique gagnant-gagnant, optimisant simultanément performances économiques et écologiques. En effet, elle présente la meilleure performance écologique pour une performance économique similaire à celle des autres politiques. Cette observation est prometteuse pour le long terme mais pose néanmoins la question des transitoires : dans quelles proportions les états transitoires de la politique "gagnant-gagnant" sont-ils sous-optimaux ? Comment faire face à l'obstacle constitué par des états transitoires sous-optimaux ?

4.3.3. Il existe un noyau de politiques publiques viables

Enfin, une dernière approche est explorée afin de dépasser les antagonismes entre les différents objectifs. Elle apporte aussi une solution au problème des transitoires souligné précédemment. Au lieu de se focaliser sur la recherche d'un optimum, l'article présenté au chapitre VI (en préparation) propose de se centrer sur les politiques viables. Ces politiques forment un noyau de scénarios validant un ensemble de contraintes bio-économiques à chaque pas de temps.

La figure I.18 présente la probabilité de chaque scénario $\tau = (\tau_{grass}, \tau_{cop})$ de satisfaire simultanément et à chaque pas de temps la contrainte de budget, ainsi que la contrainte économique et les trois contraintes écologiques (FBI, CTI, CSI) à 95 % ($\lambda = 0.95$). Nous illustrons ainsi qu'il existe un ensemble de politiques permettant d'assurer simultanément la contrainte économique de revenu et les trois contraintes de biodiversité (FBI, CTI et CSI) à chaque pas de temps avec une tolérance de 5% par rapport aux performances du scénario Statu Quo. Certains scénarios (en bleu foncé) vérifient ces objectifs dans un contexte d'incertitude stochastique avec une excellente probabilité (CVA=0.98, c'est-à-dire avec un risque de violation des contraintes de 2%). La figure I.19 montre un exemple de trajectoires pour deux indicateurs (le revenu $Inc(t)$ et le FBI) avec un scénario dont la CVA est à 0.98. La confrontation de la contrainte (en rouge) avec les performances les plus faibles (pointillés inférieurs) montre que la contrainte de revenu est toujours satisfaite et que la contrainte FBI peut être violée légérement vers 2015 (dans 2% des cas).



(a) avec le revenu escompté

(b) avec le revenu en 2050

FIGURE I.19 – Performances économiques (revenu $Inc(t)$) et écologiques ($FBI(t)$) pour un scénario viable (CVA=0.98), $\tau = (0.6, -0.2)$. En noir, les trajectoires moyennes associées aux maximums et minimums en pointillés. En rouge, les contraintes économiques et écologiques avec $\lambda = 0.95$.

Alors que les approches fondées sur une recherche d'optimum suggèrent une opposition entre les objectifs économiques et écologiques (Barraquand & Martinet, 2011, Polasky *et al.*, 2005), ce travail nous permet d'illustrer une réconciliation possible entre ces différents objectifs. En appliquant la théorie du contrôle viable à une problématique agricole, nous confirmons son utilisation comme outil intéressant pour réconcilier différents objectifs apparemment opposés, jusque là utilisé seulement dans le cadre des pêcheries (Doyen *et al.*, 2012, Martinet *et al.*, 2007, Béné *et al.*, 2001). De plus, en identifiant des politiques publiques qui évitent des crises futures sans pénaliser la génération actuelle, cette approche est en parfaite adéquation avec la définition de la durabilité.

Par ailleurs, ce noyau de viabilité, globalement caractérisé par des scénarios de retour à l'herbe, confirme l'importance des prairies (Laiolo, 2005, Bignal & McCracken, 2000) pour mettre en place une agriculture durable conciliant performances économiques et environnementales. Les scénarios viables combinent des incitations pour prairies non intensives et des taxes sur les grandes cultures. Cependant, les incitations peuvent varier au sein du noyau de viabilité tout en gardant une gestion du risque satisfaisante : les incitations viables prairies τ_{grass} variant de 0.3 à 0.6 et les incitations viables cop τ_{cop} variant de -0.3 à -0.6. Cela offre une flexibilité supplémentaire aux décideurs politiques, qui peuvent choisir entre ces différentes politiques publiques en intégrant d'autres critères comme des objectifs sociaux ou politiques.

5. Discussion générale

5.1. Les apports de la thèse

5.1.1. Un modèle bio-économique générique

En satisfaisant une grande partie du cahier des charges développé dans l'introduction, les modèles présentés ici répondent au premier objectif de la thèse et offrent un cadre théorique pertinent pour examiner la durabilité de l'agriculture.

- Ils se fondent en effet sur une approche systémique. Les modèle mécanistes développés dans ce travail sont organisés autour des trois composants de l'écosystème étudié (Etat, agriculteurs et biodiversité) et explicitent leurs relations. Ils permettent en effet de traiter des questions bio-économiques en couplant deux modèles, économique et écologique, au travers d'une variable commune d'habitat agricole. Les dynamiques de ces relations sont également prises en compte, notamment dans le modèle agro-écologique.

- Le système étudié est spatialisé. En effet, la spatialisation des données et la calibration des modèles à l'échelle micro permettent l'intégration de spécificités régionales et capturent ainsi une partie de l'hétérogénéité des contextes économiques et écologiques, comme le recommande Polasky *et al.* (2008). A la différence de travaux aux échelles méso (Holzkamper & Seppelt, 2007, Shi & Gill, 2005) ou micro (Münier *et al.*, 2004), ces modèles proposent une approche globale en menant une étude à large échelle. L'articulation de plusieurs échelles (macro et micro) permet alors de représenter chaque composante de l'écosystème à une échelle adaptée.
- L'incertitude est intégrée au travers de sa composante stochastique (appelée aussi risque) à la fois dans le modèle économique et dans le modèle écologique. Cela permet alors de réflechir à la gestion du risque pour éviter les situations de crise.
- Les modèles permettent une identification des décisions anthropiques, caractérisées comme des variables de contrôle. Pour gérer cette structure, le cadre de la théorie du contrôle a été sollicité à différentes reprises et sous différentes formes : théorie du contrôle optimal sous contrainte pour la méthode coût-efficacité et théorie du contrôle viable pour la méthode de viabilité.

Le cadre de modélisation mécaniste, dynamique et multi-échelle développé ici permet donc de représenter de manière stylisée le territoire national, afin de déterminer ses tendances futures en réponse à différents scénarios macros. De plus, en utilisant des scénarios de politiques publiques en complément de scénarios d'occupation des sols, il s'inscrit directement dans une perspective de politique économique.

Par ailleurs, les modèles étudient une communauté d'oiseaux. Cette approche multi-espèce, fondée sur des espèces communes, s'inscrit parfaitement dans le développement des travaux sur la biodiversité ordinaire mentionnés en introduction. Cela conduit à une vision plus générale de la communauté, ouvrant vers une gestion durable et globale de la biodiversité et des services écosystémiques (Chevassus-Au-Louis, 2009). Grâce aux indicateurs de biodiversité, ils évitent l'écueil de la monétarisation (Rees, 1998). Plusieurs indicateurs de biodiversité sont développés et comparés. Ces modèles permettent ainsi de mener une réflexion sur le choix des indicateurs, leur interprétation en termes d'objectifs et leur sensibilité aux scénarios.

Finalement, deux cadres théoriques (théorie du contrôle optimal sous contrainte et théorie du contrôle viable) ont été confrontés pour pour évaluer les scénarios d'un point de vue bio-économique et étudier leur durabilité. Ils ont conduit à l'identification d'une part de stratégies optimales (optimums de pareto et frontières d'efficience) et d'autre part de stratégies viables (noyau de viabilité). En utilisant de manière complémentaire les deux approches de coût-efficacité et de viabilité, davantage de facettes de la durabilité agro-écologique ont été explorées.

Mais le cadre théorique développé ici contribue plus globalement à l'ensemble des problématiques sur le maintien de la biodiversité en milieu agricole. Sa structure stylisée est facilement adaptable à d'autres zones géographiques, y compris si l'agriculture y est différente. La calibration des dynamiques de populations d'oiseaux peut se faire avec les variables agricoles qui apparaîtront pertinentes dans le nouveau cadre d'étude. Le choix des variables peut être fait en suivant la démarche proposée dans l'article du chapitre VII. De même, il est envisageable de transposer ce modèle à d'autres taxa classiquement étudiés en bio-économie comme les papillons (Drechsler *et al.*, 2007). Leur déplacement aérien, comme pour les oiseaux, engendrera peu de modifications dans la structure du modèle.

5.1.2. La réconciliation agriculture-biodiversité

Le second objectif de cette thèse était d'apporter des éléments nouveaux pour la réconciliation agriculture-biodiversité par l'élaboration de politiques publiques.

a - L'importance des politiques publiques

La première contribution relative à cet objectif a été de mettre en évidence et de quantifier l'importance des politiques publiques pour améliorer la situation bio-économique par rapport aux tendances actuelles (sections 4.1.1 et 4.2.1). De manière indépendante, notre travail illustre que des paramètres micro-économiques, comme l'aversion au risque, ont une influence positive sur les performances bio-économiques via des mécanismes de diversification (section 4.1.2). Cependant, l'effet systématique des politiques publiques, quel soit l'aversion au risque, confirme l'importance de leur utilisation dans la mise en oeuvre d'une agriculture durable (section 4.1.3). Cette conclusion légitime l'approfondissement de la réflexion sur la conception de nouvelles politiques publiques. Au travers de ses diverses études, notre travail rappelle que l'enjeu porte à la fois sur le choix des instruments (taxe ou subvention), l'objet de l'incitation (quelle activité ?) et sur le niveau de ces incitations.

Notre travail montre que des scénarios de retour à l'herbe via un développement de prairies non intensives, au détriment des grandes cultures, apparaissent comme un chemin vers la réconciliation entre les objectifs économiques et écologiques de l'agriculture (sections 4.1.1, 4.2.1 et 4.3.3). L'originalité de notre approche est de pouvoir identifier les politiques publiques sous-jacentes à ces scénarios. Nous montrons que ces politiques publiques sont fondées sur des subventions fortes aux prairies non intensives. Cette conclusion plaide en faveur du maintien et du développement de la composante environnementale de la PAC, même si sa structure actuelle (notamment au travers des MAE du second pilier) n'est pas entièrement satisfaisante. Notre travail permet de compléter cette conclusion. En effet, contrairement aux subventions aux prairies indépendantes de la contrainte de biodiversité, nous identifions également des incitations attribuées aux céréales fortement corrélées à cette contrainte. Elles constituent donc un levier important pour les performances environnementales. Notre travail illustre ainsi un fort enjeu environnemental sur le premier pilier de la PAC (qui caractérise aujourd'hui les incitations céréalières) pour la réconciliation agriculture-biodiversité.

Notre étude montre également une forte hétérogénéité régionale (section 4.3.1). En effet, les régions présentant des habitudes agricoles différentes, leurs besoins et leurs enjeux présentent une forte variabilité. Ainsi une régionalisation au moins partielle de la PAC, au travers d'une gestion locale (régionale ou plus fine), pourrait permettre d'améliorer son efficacité économique et écologique. Au système général d'incitations, pourraient être rajoutées des subventions plus spécifiques, s'adressant aux agriculteurs ayant modifié leurs pratiques vers des pratiques davantage "eco-friendly" et présentant alors un manque à gagner par rapport à leur situation historique (section 4.3.1). Cette conclusion peut être interprétée comme un élément de validation de la proposition actuelle de la Commission Européenne à la conditionnalité d'une partie des aides à des normes environnementales.

Enfin, notre travail illustre qu'il est possible d'avoir une gestion largement satisfaisante de l'incertitude stochastique, ce qui ouvre une perspective féconde pour la conception d'une agriculture durable évitant les situations de crise (section 4.3.3).

b - Les paradigmes sous-jacents aux politiques publiques

La seconde contribution concerne les paradigmes sous-jacents à l'élaboration des politiques publiques. Cette thèse suggère que la résolution du problème de la durabilité de l'agriculture passe nécessairement par une réflexion de fond sur ses paradigmes. En effet, l'apparente antinomie entre objectifs économiques et écologiques provient en partie de politiques mal conçues, mais aussi des schémas de pensée dans lesquels nous nous plaçons lors de la conception de ces politiques. Notre travail a permis de mettre en évidence trois d'entre eux :

- Le paradigme du payeur : du consommateur-payeur vers un pollueur-payeur

Si la taxe et la subvention agissent sur les rentabilités de manière symétrique d'un point de vue mathématique, elles impliquent des considérations morales et juridiques complètement différentes. L'utilisation de subventions suggère que le bien commun (constitué par la biodiversité résidant sur les terres de l'agriculteur) est la propriété de l'agriculteur et que la société doit payer si elle veut qu'il fasse des efforts pour sa conservation : c'est le principe du consommateur-payeur. Inversement, la taxe implique que le bien commun appartient à la société et que l'agriculteur doit payer s'il le dégrade : principe du pollueur-payeur (Tobey & Smets, 1996, Baldock, 1992). Or, notre étude montre une diminution progressive des subventions attribuées aux grandes cultures jusqu'à devenir des taxes lorsque la contrainte de biodiversité augmente (sections 4.1.1 et 4.2.1). L'intégration d'objectifs environnementaux nécessite donc un glissement de la conception de consommateur-payeur à celle de pollueur-payeur. En d'autres termes, cela pose la question de qui supporte le coût de la biodiversité au sein d'une société.

D'autres résultats dans cette thèse, comme le choix du sujet économique (section 4.3.1) et l'évolution temporelle des incitations (section 4.3.2), rejoignent cette problématique. En effet, considérer la société dans son ensemble plutôt que l'agriculteur comme seul agent économique nous permet de mettre en évidence que l'intégration d'objectifs environnementaux n'engendre pas en réalité de poids économique supplémentaire. Cependant, ce changement de sujet économique implique de ne plus considérer la biodiversité comme la propriété des agriculteurs, et donc la PAC comme un dû de la société envers les agriculteurs si celle-ci veut voir son environnement protégé. La biodiversité est au contraire considérée comme un bien de la société dans son ensemble. De même, il semble possible de réduire progressivement en partie le volume des incitations tout en gardant un effet positif sur la biodiversité, libérant ainsi une partie du budget qui peut être ré-allouée à d'autres enjeux (encadré 3). Cela demande à nouveau de ne pas considérer la PAC comme un dû. L'ensemble de ces résultats souligne l'importance de l'évolution de ce paradigme pour pouvoir construire des politiques de réconciliation.

- Le paradigme du patrimoine naturel : du propriétaire vers l'usufruitier

La réflexion porte ici sur le poids attribué au présent dans la prise de décision. Dans le courant de la "régle d'or" (Solow, 1956) et de "l'overtaking" [1], notre étude montre qu'il existe des politiques de réconciliation à long terme mais que celles-ci ne sont pas optimales économiquement à court terme, expliquant ainsi pourquoi elles sont actuellement rarement choisies par les décideurs politiques (sections 4.1.1 et 4.3.2). Cependant, passer d'un objectif lourdement pondéré par le court terme à un objectif de long terme nécessite une évolution de la perception du patrimoine naturel : la Nature n'appartient plus à la génération présente. On peut alors la considérer comme la propriété de l'ensemble des générations. La génération actuelle, qui en a l'usufruit, a comme devoir sa protection, même si cela implique un coût. En d'autres termes, ce paradigme soulève la question de la propriété inter-temporelle de la biodiversité. L'évolution de ce paradigme permet de rejoindre la définition de développement durable proposée en 1987 dans le rapport Brundtland de la Commission Mondiale sur l'Environnement et le Développement de l'ONU (CMED, 1988) et utilisée comme base au Sommet de la Terre de Rio en 1992 : il est défini comme "*un développement qui répond aux besoins des générations du présent sans compromettre la capacité des générations futures à répondre aux leurs*". Même si cela est difficile à mettre en oeuvre [2], éviter la dictature du présent et revaloriser le futur semble indispensable pour pouvoir mettre en place une agriculture durable.

- Le paradigme de la performance : d'un maximum théorique à des seuils acceptables

Ce dernier levier, plus fondamental, concerne la perception de la performance économique en elle-même. Le contexte économique actuel de compétitivité dans lequel nous évoluons aujourd'hui est caractérisé par une recherche permanente d'optimalité. Or un assouplissement de ce contexte en direction de la viabilité pourrait être une perspective intéressante pour la durabilité : il ne s'agirait plus de chercher *la* politique

1. La règle d'or développée par Solow (1956) et Phelps (1961) consiste à maximiser le niveau d'utilité à long terme. Le critère de dominance à long terme ("overtaking") conduit à la sélection de trajectoires maximisant les utilités au delà d'une date T.

2. Le critère évitant la dictature du présent et du futur proposé par Chichilnisky (2000) (basé sur la maximisation d'une combinaison convexe de l'utilité actualisée et du niveau d'utilité à très long terme) semble présenter quelques difficultés dans sa construction.

optimale mais plutôt d'exclure les politiques *insoutenables*, en d'autres termes d'éviter une dégradation de la situation actuelle. Cette approche, basée sur des conditions de soutenabilité a minima, conduit à un nouveau paradigme qui fait écho à une définition négative de la soutenabilité. Cette thèse montre que cette approche a minima était pertinente dans le cas de la réconciliation agriculture-biodiversité puisqu'il existe bien un noyau de politiques publiques dites viables réalisant simultanément des objectifs environnementaux et économiques (section 4.3.3). Ce paradigme plus souple offre ainsi une flexibilité à la gouvernance nécessaire au contexte d'incertitude (économique, écologique, climatique, productif, technologique) dans lequel nous nous trouvons.

5.2. Les perspectives de la thèse

5.2.1. Vers un approfondissement des relations habitat-biodiversité

Comme nous le mentionnions dans l'introduction, cette thèse s'inscrit plus globalement dans la démarche de l'IPBES, qui vise à décrire les grandes tendances de la biodiversité en réponse à des pressions anthropiques, à l'image des scénarios du GIEC. Pour asseoir leur crédibilité, l'ancrage des modèles dans la réalité historique est alors indispensable. Cela passe notamment par leur calibration. Le renforcement de ces calibrations constitue donc un enjeu et une perspective majeure.

Une des originalités du modèle développé ici est l'explicitation du lien entre la taille de la population et les variables agricoles (via une compétition intra-spécifique et une capacité de charge du milieu). Renforcer ce mécanisme et plus généralement la calibration du modèle agro-écologique apparaît alors comme une perspective stimulante.

D'un point de vue agronomique, il pourrait être intéressant d'intégrer d'autres composantes agricoles ayant montré un impact sur les dynamiques écologiques et économiques et renseignées dans des bases de données :

- C'est notamment le cas d'éléments relatifs à l'intensité des activités. En effet, selon son niveau d'intensité, une même activité peut générer une qualité d'habitat complètement différente (disponibilité en insectes, destruction des nids) et donc avoir un impact différent sur les populations (Kleijn *et al.*, 2009, Billeter *et al.*, 2008). De plus, les coûts et les rendements qui lui sont associés varient, rendant sa mise en place sur le long terme rentable ou non. Des données relatives à cette intensité ont été récemment élaborées à une échelle fine, compatible avec notre modèle (Teillard *et al.*, 2012).
- Les éléments semi-naturels (linéaires de haies, bandes enherbées), qui ont montré leur importance dans les dynamiques des oiseaux (Billeter *et al.*, 2008), notamment pour la nidification, pourraient être intégrés. Des impacts positifs sur les performances économiques ont aussi été identifiés (protection du vent et limitation du déplacement des nuisibles des cultures par les haies).

D'un point de vue écologique, la pérennisation des processus d'acquisition de données, comme le programme STOC, est un élément déterminant pour maintenir et affiner la calibration. L'utilisation de séries temporelles longues au cours de la calibration renforce sa robustesse. De plus, le large gradient d'habitats fourni par une collecte de données à large échelle permet de gagner en généricité et constitue donc un point clef pour mettre en évidence des tendances générales. Enfin, pour avoir une vision plus globale des tendances de la biodiversité agricole, l'intégration d'autres taxa, comme les insectes pollinisateurs ou la faune du sol, serait particulièrement intéressante compte tenu de leurs nombreuses interactions avec l'agriculture. Développer des suivis relatifs à ces autres groupes constitue donc un développement indispensable pour approfondir les relations habitat-biodiversité et évaluer des tendances plus générales de la biodiversité en milieu agricole.

5.2.2. Vers une approche écosystémique de l'agriculture

Le positionnement écosystémique, en couplant dynamiques écologiques et socio-économiques, du modèle constitue une caractéristique clef de la démarche développée ici. L'explicitation des mécanismes et la prise en compte des aspects dynamiques, spatialisés et incertains du système étudié sont essentiels. Dans cette perspective, d'autres mécanismes pourraient être considérés pour renforcer le modèle.

D'un point de vue écologique, l'approche communauté pourrait être approfondie, notamment en complétant la compétition intra-spécifique par une compétition inter-spécifique, généralisant ainsi les interactions entre les individus. Dans le cas de notre étude, la construction d'une méta-communauté à l'échelle de la France pourrait mettre en évidence des régions sources et des régions puits en fonction du type d'agriculture qui y est menée, avec pour effet de renforcer les tendances actuelles : les régions avec une agriculture défavorable, qui présente par exemple une homogénéisation biotique, verraient cet effet accentué par la fuite des espèces non adaptées à cette agriculture vers les régions voisines. Cette spatialisation des dynamiques écologiques s'inscrit parfaitement dans le second point du cahier des charges, relatif à l'explicitation spatiale des éléments constituant le système. Notons cependant que la calibration d'une telle méta-communauté reste très difficile, notamment à cause de la rareté des informations relatives à la dispersion.

D'un point de vue macro-économique, la stabilité du contexte macro-économique représente une hypothèse majeure du modèle. L'évolution tendancielle des prix, la volatilité des marchés, le développement des marchés des technologies et l'évolution du marché de la main d'oeuvre constituent des moteurs essentiels pour la décision économique. Notre modèle se place dans une perspective d'équilibre de ce point de vue. Notamment, il n'intègre pas d'évolutions tendancielles des marges brutes. Or les prix à la vente (via l'évolution des habitudes alimentaires et le croissance démographique) et les coûts de production (via la hausse du prix des carburants, des intrants et de la main d'oeuvre) présentent des tendances, y compris à court terme. Cette stabilité des marges brutes suppose aussi un équilibre du marché des technologies, peu crédible à moyen terme. Considérer des scénarios de prix ou d'évolutions technologiques dans notre modèle permettrait d'intégrer de l'ambiguité, élement caractéristique d'un environnement incertain (point trois du cahier des charges). L'intégration de variables de niveau technologique serait d'autant plus intéressante qu'elles constitueraient un nouveau levier sur lequel l'État pourrait intervenir afin de promouvoir des "technologies propres", en diminuant leur coût d'investissement et en rendant attractive des technologies peu rentables à court terme. Par ailleurs, des rétro-actions de la production française sur les prix (ie les agriculteurs ne sont plus preneurs de prix) pourraient être intéressantes, notamment en relation avec le questionnement de la répartition mondiale de l'effort de production et de l'impact sur l'environnement.

Enfin, d'un point de vue micro-économique, l'hypothèse de rationalité [1] est une hypothèse classique mais qui impose un certain nombre de limites, dont l'absence de verrouillages sociaux. Ces verrouillages ont pourtant un effet essentiel puisqu'ils conduisent à la non-adoption par les agents d'une activité ou technologie économiquement rentable. Ils ont plusieurs sources : manque d'information, incertitude sur la nouveauté, problème de coordination entre agents. Concernant les problématiques agricoles, Vanloqueren & Baret (2008; 2009) les ont illustrés dans les cas de l'agro-biodiversité et de l'agro-écologie. Intégrer ces verrouillages sociaux en spatialisant les agriculteurs constitue donc un développement intéressant au regard du second point du cahier des charges. Cela rendrait certainement plus inertiel l'effet des politiques publiques très différentes des politiques actuelles ou demanderait des niveaux de subventions plus importantes pour influencer les décisions des agriculteurs.

5.2.3. Vers une exploration plus vaste des politiques publiques

Face aux différentes difficultés mentionnées dans l'introduction (intégration d'objectifs d'ordres différents, présence d'incertitudes ...), nous soulignions l'importance des démarches de modélisation et de scénarisation pour stimuler l'intégration d'objectifs de biodiversité dans l'élaboration des politiques publiques et conduire les politiques de conservation vers des réflexions bio-économiques. L'objectif de ces démarches est bien

1. Nous considérons que l'agent économique prend les décisions en maximisant strictement son utilité et qu'il est capable d'ordonner les différents choix qui s'offrent à lui.

l'exploration de diverses politiques publiques (pouvant largement différer des tendances historiques) et l'évaluation de leurs performances.

Le modèle peut être utilisé pour étudier la réaction de différents indicateurs aux scénarios, afin d'identifier un set d'indicateurs permettant de décrire correctement l'état de l'écosystème en fonction des questions posées. Notre étude a en effet permis de mieux comprendre les gammes de paysages dans lesquels les indicateurs FBI, CTI et CSI étaient intéressants. D'autres indicateurs peuvent être explorés de manière similaire. Par exemple, l'écart-type inter-régional du revenu permettrait d'évaluer l'équité inter-régionale et de mener une réflexion sur la répartition spatiale des richesses. De manière intuitive, l'évaluation des différents scénarios avec cet indicateur irait dans le sens des conclusions obtenues dans notre étude, en discréditant le scénario Crop qui subventionne les régions céréalières, régions les plus riches. Nous pourrions aussi étudier le minimum régional, rejoignant ainsi les approches maximin (Solow, 1974) appliquées spatialement. Du point de vue écologique, raffiner les classes déjà existantes, comme distinguer les spécialistes des milieux céréaliers et des spécialistes milieux prairiaux, permettrait de confirmer les domaines d'interprétation du CTI et du CSI : avec une politique stimulant le développement de biocarburants (scénario Crop), l'indicateur des spécialistes des céréales (granivores et très spécialistes) augmenterait alors que celui des spécialistes des prairies (niveau trophique plus haut et spécialiste) diminuerait. D'autres indicateurs, en lien avec des classifications officielles, pourraient complétés les indicateurs fonctionnels déjà étudiés. Par exemple, utiliser un indicateur fondé sur le niveau d'abondance, rejoignant ainsi le classement de l'IUCN du risque d'extinction, permettrait de réfléchir au maintien d'espèces spécifiques en danger et ayant une valeur patrimoniale forte. Cela conduirait à souligner la valeur culturelle de la biodiversité, en plus de sa valeur fonctionnelle, et ainsi d'avoir une vision plus globale des valeurs de la biodiversité.

Le modèle peut aussi être utilisé pour étudier d'autres scénarios ou d'autres instruments de politiques publiques. Par exemple, tester des politiques plus adaptatives (ré-actualisées tous les 10 ans) amènerait certainement à un gain d'efficacité. Ce développement est directement réalisable avec la structure actuelle du modèle. De même, l'intégration de nouvelles contraintes, notamment politiques (présence de résultats à court terme ou limitation de l'instrument taxe mal accepté socialement), rendrait plus difficile l'élaboration d'une politique de réconciliation. L'intégration de telles considérations est d'autant plus stimulant que ce travail a souligné l'importance des taxes pour les grandes cultures et la valorisation du long terme. Les instruments réglementaires (normes, zones protégées) apparaissent alors comme des instruments alternatifs et complémentaires. Coupler instruments prix et instruments réglementaires offrirait ainsi une plus grande flexibilité. La structure du modèle permet tout à fait ce développement en traitant la norme comme une contrainte supplémentaire dans l'optimisation de l'utilité des agriculteurs.

5.3. Les nouveaux défis

5.3.1. Un contexte de changement global

Les travaux sur l'évolution de la biodiversité en réponse à des pressions anthropiques s'inscrivent plus généralement dans le contexte des changements globaux. Intégrer une évolution plus générale de l'environnement est donc une perspective naturelle. Dans le contexte de l'agriculture, nous pouvons notamment souligner l'importance des territoires non agricoles (forêts, zones urbaines). En effet, Devictor & Jiguet (2007) ont montré un effet positif de la diversité des éléments environnants sur les dynamiques de populations d'oiseaux. Si les zones urbaines ont un effet globalement négatif, l'impact des forets est plus variable selon les espèces. Par ailleurs, la pression de l'immobilier sur les territoires agricoles conduit à une diminution globale de la surface agricole utile en France. Cela pose alors la question de la production : comment produire autant sur une superficie moindre sans nécessairement passer par une intensification systématique fondée des intrants ?

L'intégration des dynamiques des zones forestières et urbaines, toutes deux en expansion, et de leur impact sur la biodiversité forme donc une perspective intéressante d'un point de vue des changements globaux.

Par ailleurs, de récents travaux sur l'évolution des niches écologiques en réponse au changement climatique illustrent une tendance à la diminution des populations d'oiseaux nicheurs (Jiguet *et al.*, 2010). Les espèces nichant à des températures maximales moins élevées, en d'autres termes les espèces les plus septentrionales, sont les plus affectées. Cela concerne par exemple le Pipit farlouse *Anthus pratensis*, le Tarier des près *Saxicola rubetra*, le Bruant jaune *Emberiza citrinella* ou le Corbeau freux *Corvus frugilegus*. Intégrer différents scénarios de changement climatique (ambiguité) nous semble donc être un développement stimulant pour intégrer les évolutions plus générales de l'habitat.

5.3.2. Une gouvernance de la biodiversité

L'exploration de différents scénarios et indicateurs s'inscrit dans la démarche de l'IPBES, qui vise à l'élaboration d'une connaissance scientifique commune intelligible par les différents acteurs de la décision. Par la centralisation de la connaissance scientifique, la finalité de plateformes comme l'IPBES est bien d'enrichir le débat collectif sur la gestion de la biodiversité. Les modèles bio-économiques et les scénarios qui y sont rassemblés présentent donc une dimension heuristique forte.

En effet, même si des raffinements sont à envisager, l'intérêt porté par les décisionnaires pour ce type d'approche est croissant. La note de la Commission Européenne relative à nos travaux (annexe C) et son positionnement dans le top 10 des notes publiées en 2011 confirme cette tendance. L'intégration de plusieurs échelles et de différentes disciplines permet notamment de tenir compte des points de vue d'acteurs variés. Le développement de jeux de rôle fondés sur ces modèles bio-économiques constitue alors une application naturelle. En stimulant le débat entre plusieurs acteurs, ces jeux permettraient de faire émerger des processus de négociation, mettant notamment en évidence des freins à l'établissement de certaines politiques publiques, et de développer ainsi une gouvernance de la biodiversité.

5.3.3. Les services écosystémiques

Ces dernières années, la biodiversité a été de plus en plus valorisée au travers de sa traduction en services écosystémiques (SE). De nombreux travaux dans la communauté scientifique s'y intéressent (Foley *et al.*, 2005, Chapin *et al.*, 2000, Costanza *et al.*, 1997) et son intégration dans plusieurs rapports à destination des décideurs (Chevassus-Au-Louis, 2009) en font un objet au coeur du débat sur la préservation de la biodiversité. Nous considérons ici la définition des SE donnée par le MEA (MEA, 2005) : "*les bénéfices directs ou indirects que retire l'Homme des écosystèmes*". Quatre grandes classes de SE ont été identifiées selon leur fonction :

- fonctions de base non directement utilisées par l'Homme mais qui participent au bon fonctionnement de l'écosystème (cycle du carbone ou de l'azote),
- services d'approvisionnement qui conduisent à des biens appropriables (bois, pêche),
- services de régulation c'est-à-dire la capacité à moduler des phénomènes dans un sens favorable à l'Homme (contrôle des ravageurs, pollinisation),
- services culturels représentant l'utilisation de l'écosystème à des fins récréatives (beauté des paysages, héritage culturel ou religieux).

Les SE sont aujourd'hui considérés comme des outils prometteurs pour la conservation de la biodiversité et sa prise en compte dans la décision. En effet, la notion de service fait appel à une vision utilitariste de la biodiversité qui, si elle est quelque peu réductrice, peut faciliter la prise de conscience. Se focaliser sur un service et le risque de le perdre conduit à la conservation du faisceau d'espèces qui y participent. Cette approche s'inscrit donc une vision globale et systémique de la biodiversité.

Cependant, le champ de recherche ouvert par les services écosystémiques ne présente pas encore de cadre précis. En effet, de nombreuses incertitudes portent sur les liens entre biodiversité et fonctions des écosystèmes, base des services écosystémiques (Danovaro *et al.*, 2008, Lanta & Leps, 2008, Worm *et al.*, 2006, Hooper *et al.*, 2005, Loreau *et al.*, 2001). La réflexion bio-économique, à laquelle participent les travaux présentés dans ce document, apporte une contribution pour la mesure et la quantification de ces services. En effet, en explorant des relations et des mécanismes entre décisions anthropiques, environnement et biodiversité, nos travaux permettent d'expliciter l'impact de l'Homme sur un échantillon de la biodiversité. Bien qu'ils ne se focalisent que sur un seul taxon, ils participent à la clarification de l'impact de décisions économiques sur son maintien ou sa destruction. L'analyse de la communauté au travers de différents indicateurs renseignent sur le devenir de différents SE. Par exemple, l'indicateur de variation d'abondance de la communauté des oiseaux granivores informe sur le service de dissémination des plantes, et l'indicateur de variation d'abondance de la communauté des oiseaux insectivores renseigne sur le service de contrôle de ravageurs de cultures. Les démarches de modélisation explicite entre décisions anthropiques et biodiversité, dans lesquelles nous nous inscrivons, sont essentielles pour estimer le bénéfice et l'efficacité des actions mises en oeuvre sur l'ensemble de l'écosystème et donc sur un groupe de services écosystémiques.

Afin d'élargir les connaissances et d'avoir une vision plus globale des SE, il serait intéressant de développer ces démarches de modélisation sur d'autres taxa. Dans le cas de l'agriculture, l'aération des sols et la pollinisation sont par exemple deux SE essentiels fournis par la biodiversité agricole. En appliquant ces démarches aux insectes pollinisateurs et à la faune de sol, nous pourrions ainsi approfondir le lien entre les décisions agricoles et les SE. Dans la mesure où notre étude a montré que les réponses pouvaient être contrastées y compris au sein d'une même groupe, la généralisation à d'autres taxa permettrait de mettre en évidence et d'étudier plus précisément la complexité de l'impact des décisions anthropiques sur la biodiversité et les services écosystémiques dont elle est le support.

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Chapitre II

Bio economic modeling for a sustainable management of biodiversity in agricultural lands

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Abstract

For several decades, significant changes in farmland biodiversity have been reported in Europe. Agriculture is a major driver of these modifications. Taking into account these environmental impacts, agriculture nowadays aims at a more sustainable way of producing which would reconcile its economic and ecological functions. The objective of this paper is to give insights into the impact of public policies on both conservation of biodiversity and farming production. We develop a macro-regional model combining community dynamics of 34 bird species impacted by agricultural land-uses and an economic decision model. The ecological dynamic model is calibrated with the STOC (French Breeding Bird Survey) and AGRESTE (French land-uses) databases while the economic model relies on the gross margins of the FADN (Farm Accountancy Data Network). We investigate scenarios based on subsidies and taxes. We show that simple economic instruments could be used to establish scenarios promoting economic performances and bird populations. It is pointed out how the sustainability of the policies is sensitive to the ecological and economic indicators used by the planner. The bio-economical analysis shows several solutions for the ecology-economy trade-off. These results suggest that many possibilities are available to develop multi-functional sustainable agriculture.

Keywords : Biodiversity, Agriculture, Bioeconomic modelling, Sustainability, Bird, Land-use

1. Introduction

We observe in recent decades a biodiversity decline in Europe. The pressure is particularly strong on bird populations which have undergone severe and widespread decline (Jiguet, 2009, Chamberlain *et al.*, 2000, Krebs *et al.*, 1999). Such erosion is mainly due to a combination of habitat loss and degradation of habitat quality altering the nesting success and/or survival rates (Benton *et al.*, 2003). Global changes in European agriculture, including intensification and land abandonment, have significantly modified farmland bird (Donald *et al.*, 2006; 2001). In this context, the European Union, aiming at halting biodiversity loss, has adopted the Farmland Bird Index as an indicator of structural changes in biodiversity (Balmford *et al.*, 2003). In this perspective, the need to reconcile agricultural production and biodiversity is of particular interest (Jackson *et al.*, 2005).

Since the early 90's, several public policies have been developed to limit the negative impacts and externalities of agriculture on biodiversity. Typically, agri-environment schemes have been introduced in which farmers receive support for adopting environmentally friendly agricultural practices. There is an extensive and increasing volume of literature concerning agri-environmental schemes and policies for multi-functional agriculture (Drechsler & Watzold, 2007, Shi & Gill, 2005, Dobbs & Pretty, 2004, Münier *et al.*, 2004, Pacini *et al.*, 2004, Alavalapati *et al.*, 2002). However, after 15 years of implementation of such instruments, the question whether providing habitat quality conflicts with management for agricultural production remains controversial (Butler *et al.*, 2007, Kleijn *et al.*, 2006, Vickery *et al.*, 2004). To address agro-environmental sustainability, both economic and ecological criteria must be considered. As pointed out by Hughey *et al.* (2003) and (Perrings *et al.*, 2006), there is an urgent need for approaches that integrate economic criteria in conservation problems. Reinforcing such analyses and examining forms of farming allowing for the joint sustainability of biodiversity and agricultural production requires interdisciplinary research. Such work relies upon the development of interdisciplinary concepts, quantitative methods and integrated models that adequately incorporate the complex interdependencies between farmland ecosystems and economic systems.

The present paper deals with such modelling issues regarding agro-environmental sustainability. A bioeconomical model is developed to study the joint sustainability of agricultural land-use and bird biodiversity. To address agro-environmental sustainability, numerous modelling frameworks are proposed in the literature. They include Cost-Benefit (Rashford *et al.*, 2008, Drechsler, 2001) and Cost-Effectiveness (Holzkamper & Seppelt, 2007) approaches. A major criticism of Cost-Benefit analysis for conservation issues is that benefits related to biodiversity and habitat quality are usually non-market goods, and by definition difficult to quantify in monetary terms (Rees, 1998). Although pricing techniques such as contingent valuation are available, their suitability for complex biodiversity issues is disputed, notably in anthropogenic systems (Diamond & Hausman, 1994). Cost-Effectiveness analysis can be used to reveal a minimal cost policy among those satisfying the given goals of conservation and production (Macmillan *et al.*, 1998). This approach, based on optimization under constraints, avoids monetary evaluation of environmental goods (Gatto & De Leo, 2000). As far as agricultural economics is concerned, most models rely on mathematical programming and optimization under constraints. The joint production of an agricultural commodity and grassland biodiversity is examined for instance in Havlik *et al.* (2005) where the environmental service is approximated by the number of hectares managed in a prescribed environmentally friendly way. The real impacts on biodiversity and ecological services are more explicitly considered in van Wenum *et al.* (2004) or Polasky *et al.* (2005). To deal with sustainability, approaches such as ecological economics (Drechsler & Watzold, 2007) suggest studying environmental and economic effectiveness simultaneously, stressing the relevance of multi-criteria approaches. However, few economic studies cope with the spatial and temporal dynamics of biodiversity in this context (Hammack & Brown, 1974). In this vein, a range of spatially explicit models exist that aim at assessing consequences of different land use patterns for various environmental and economic criteria (Swihart *et al.*, 2003, Irwin & Geoghegan, 2001). Nevertheless, most of these models are static, which restricts the ecological processes taken into account. Moreover, they usually do not incorporate important economic drivers (e.g. agricultural prices, subsidies) that affect the returns of different land-use patterns. Some recent approaches overcome these limitations by integrating biological and economic models that demonstrate (Pareto-) efficient land-use patterns (Groot *et al.*, 2007, Polasky *et al.*, 2005).

The bio-economic model proposed in the present paper is in direct line with these considerations. First the model is dynamic. Furtermore it articulates ecological and economic compartments and adopts a multi-criteria perspective. Moreover, it offers a spatialized perspective as it is built up at a macro-regional scale and its calibration relies on French regional data of both land-use and bird abundance. The model allows to analyze how we can significantly drive the bio-economic performances with financial incentives. This model questions the way to evaluate the ecological and economic dimensions and to rank habitat management decisions in order to assess the relevance of different policies, notably with respect to sustainability. In particular we study the influence of different ecological and economic indicators on the sustainability of scenarios.

The paper is organized as follows. The second section presents our bio-economical model and its calibration. The third section describes the results. The fourth section is devoted to the discussion.

2. The bio-economic model

2.1. The ecological model

Regarding the model for bird populations, we choose a dynamic framework. We adopt the Beverton-Holt model (Beverton & Holt, 1957) which accounts for the intra-specific competition for the resources and the density dependance as follows :

$$\mathrm{N_{s,r}(t+1) = N_{s,r}(t).\frac{1+R_s}{1+\frac{N_{s,r}(t)}{M_{s,r}(t)}}} \tag{II.1}$$

where $N_{s,r}(t)$ stands for the bird abundance of species s in region r at year t. The R_s coefficient corresponds to the intrinsic growth rate specific to a given species s which is assumed to remain stable over France. This parameter takes into account the characteristics of each species such as the clutch size, mean reproductive success, or the number of clutches per year. The product $M_{s,r}(t)$*R_s represents the carrying capacity of the habitat r and the value $M_{s,r}$ captures the ability of the habitat to host the species. For computing the abundance at year $t+1$, we have chosen to involve the $M_{s,r}$ coefficient at year t, which shows a delay between the time when the habitat is modified and the time when the change affects the species. Habitat index $M_{s,r}$ is assumed to depend on land-uses as follows :

$$\mathrm{M_{s,r}(t)} = \beta_{\mathrm{s,r}} + \sum_{\mathrm{k}} \alpha_{\mathrm{s,r,k}}.\mathrm{A_{r,k}(t)} \tag{II.2}$$

where $A_{r,k}(t)$ represents the share of the region r dedicated to land use k at time t. The α and β coefficients, specific to each species, show how such a species s responds to the various agricultural uses in a given region r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a region r. Hence, one feature of this ecological model is that the abundances depend on the land-uses. It is noteworthy that the model is not explicitly spatialized. In terms of birds biodiversity, this is not a too restrictive hypothesis : their repartition areas are wider than those of other taxa and so they are less sensitive to spatial arrangements.

2.2. The economic model of the farmer

We consider the 21 regions of metropolitan France. Each region r is managed by a representative farmer who selects farming land-uses along time. It is assumed that forests and no agricultural land-uses are kept fixed. The farmers make their choice in order to maximize their income given rigidity and technical constraints. Thus this income depends on two economic parameters -unit gross margin and public incentives- and the current land-uses. The program of the regional agent is defined by :

$$\max_{A_{r,k}} \mathrm{Income_r(t)} = \sum_{farming\ k} \overline{\mathrm{mb}}_{\mathrm{r,k}}.\mathrm{A_{r,k}(t)}.(1+\tau_k(t)) \tag{II.3}$$

$$|\mathrm{A_{r,k}(t)} - \mathrm{A_{r,k}(t-1)}| \leq \varepsilon.\mathrm{A_{r,k}(t-1)} \tag{II.4}$$

$$\sum_k \mathrm{A_{r,k}(t)} = \mathrm{A_r} \tag{II.5}$$

For each use k, the farmers get a direct income derived by the mean gross margin per surface unit $\widehat{mb}_{r,k}$ of this use k in region r and the surface dedicated to such a use $A_{r,k}(t)$. This income is affected by public incentives τ_k on different uses k which take the form of taxes ($\tau_k < 0$) or subsidies ($\tau_k > 0$). It is computed as a rate τ of mean gross margin per surface unit. This rate represents the economic lever of public decision makers to steer the uses. We choose gross margins as economic data because it was the only one available for each region. In addition, this kind of indicator have been already used in bio-economic modelling (ten Berge et al., 2000, Pacini et al., 2004). Implicitly the gross margin is computed from the regional output of each activity and its sale price. When maximizing their income, the standard agents must comply with two constraints at every time. The first constraint (eq. (II.4)) corresponds to a technical constraint where the coefficient ϵ stands for the rigidity in changes (for example, $\epsilon = 0$ means the surfaces remain constant). The second constraint (eq. (II.5)) ensures merely that the total surface per region is kept fixed.

For any region, representative farmers define the share of their farming land which they dedicate to the various practices relying on a linear optimization under constraints. Certain hypotheses underlie this model. We assume that the economic system is at equilibrium and that the farmer's choice does not alter such equilibrium. First, we consider the farmers as price-takers. Second, we admit that the food demand remains constant. Third, the technological level does not evolve : there is neither improvement from research nor the quest for improved productivity from the farmers. The mean yield (which this income per surface unit depends on) is kept flat. Finally, the agricultural surface is assumed constant.

2.3. Model coupling and public decisions

Ecological and economic models described previously are linked by the land-uses as depicted by figure II.1. With the objective of maximizing incomes under technical and inertia constraints, the representative farmer exhibits pattern of land-uses $A_{r,k}(t)$ which are injected into the ecological model through the habitat $M_{s,r}(t)$: the agricultural states are the outputs of the economic model and the inputs of the ecological model. The farmer's economic choices thus condition bird abundances $N_{s,r}(t)$ associated with the habitats.

The third element of our modelling is the public stakeholder. The decision-makers impact the bio-economic system through an economic instrument : they use a set of incentives τ_k which affects land-uses, by modifying their profitability. Thanks to their economical model, the farmers shape their land-use patterns in order to maximize their income. These land-use rearrangements improve the global wealth while perturbing the evolution of the ecological model and bird community dynamics. Decision-makers define their incentive/tax politics depending on their ecological objectives and economic planning. For this purpose, the regulating agency must be able to evaluate the economic wealth and the biodiversity of the system that is governed. We

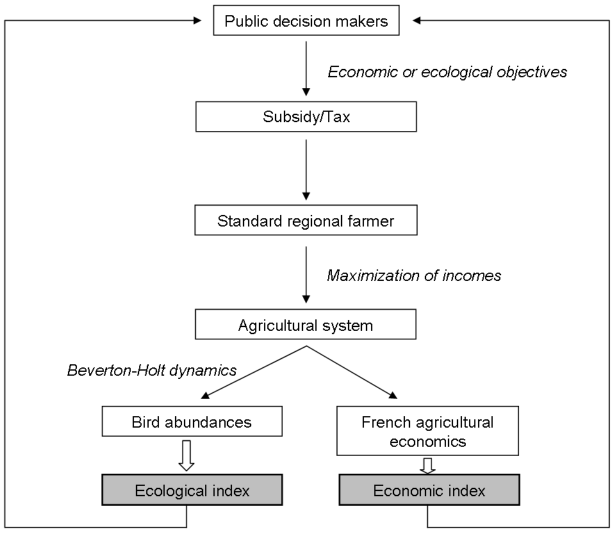


FIGURE II.1 – Model coupling : Farmers maximize their income and adjust theirs land-uses pending on subsidies. These choices affect french agricultural economics and bird's community dynamics.

assume that there is no holistic criterium, representing all dimensions of the system. So various indicators are used.

From an economic viewpoint, we focus on two indicators :

- the national mean income per unit surface (eq. (II.6)). It is computed from the mean gross margin of the 21 regions $Income_r(t)$ and the surfaces S_r. This represents a mean approach of the problem.

$$\overline{\text{income}}_{\text{France}} = \frac{\sum_{r=1}^{21} \text{S}_\text{r}.\text{Income}_\text{r}(\text{t})}{\text{S}_{\text{France}}} \tag{II.6}$$

- the regional minimum income per unit surface (eq. (II.7)). This indicator reflects the maximin analysis developed in economic studies (Solow, 1974).

$$\text{income}^{-}_{\text{France}} = \min_r \left(\frac{\text{Income}_\text{r}(\text{t})}{\text{S}_\text{r}} \right) \tag{II.7}$$

For sake of clarity, we represent these two criteria after normalisation by their current value (2008).

From an ecological viewpoint, we have selected the FBI index, provided by the Vigie-Nature website [1]. We focus on this indicator which has been adopted as the Farmland Bird Index to analyze structural changes in biodiversity (Balmford *et al.*, 2003). This is a variation index of abundances with respect to the reference year 2005. An aggregated STOC indicator is built for two bird groups : the generalist species and the farmland specialist species (Julliard *et al.*, 2004). It is computed as the geometric mean of the indices of the species considered in the class exposed by equation (II.8). In these aggregated indices, the abundance variation of each species is taken into account similarly, independently from the abundance value :

$$\text{STOC}_{\text{r,class}}(\text{t}) = \prod_{\text{s} \in \text{class}} \left(\frac{\text{N}_{\text{s,r}}(\text{t})}{\text{N}_{\text{s,r}}(2005)} \right)^{1/\text{Card(class)}} \tag{II.8}$$

1. http ://www2.mnhn.fr/vigie-nature/

2.4. Scenarios

Once the ecological and economic models have been calibrated, we can use them to analyse the impact of public policies. The selected timeframe runs up from 2008 to 2050, i.e a 43-year forecast. Selecting a shorter timeframe could consequently hide interesting long-term effects due to the inertia of the models.

We define scenarios for various incentive policies aimed at analysing the impact of public decisions on both the economy and agricultural biodiversity. In all scenarios described, the surfaces allocated to the forest and non-farming area remain steady in all times : we focus only on the evolution of the farmland uses. This approach highlights the impact of the composition of farmland uses on biodiversity, the global surface remaining constant. However, we integrate the surrounding habitats (with the forest and non-farming agricultural surfaces) to compute the $M_{s,r}$ in bird dynamics (Devictor & Jiguet, 2007).

The key parameter which characterizes each scenario is the vector τ representative of the subsidy or the tax. We have developed 4 scenarios :

- Crop scenario : subsidies for cereal, oleaginous, proteaginous (COP) with $\tau_{cop} > 0$.
- Grassland scenario : subsidies for the permanent grassland (GRASS) with $\tau_{grass} > 0$.
- Double subsidy (DS) scenario : subsidies for COP, and subsidies for permanent grassland with $\tau_{cop} > 0$ and $\tau_{grass} > 0$.
- High Quality Environmental (HQE) scenario : taxes on the COP, redistributed to the permanent grassland with $\tau_{cop} < 0$ and $\tau_{grass} > 0$.

The first two scenarios are very simplified variants of current policies. The first scenario represents policies which support COP, for example with the objective of developing bioenergies. The second scenario corresponds to a policy of extensification by the development of permanent grasslands. The DS scenario is closer to the current situation of the Common Agricultural Policy giving subsidies both to COP and grasslands. The fourth scenario, the HQE scenario, slightly more complex, plays at two levels : the tax on the COP and the incentive for permanent meadows. The interest of two last scenarios is to analyze possible synergies between two incentives : have the incentive on grasslands the same effectiveness alone or combined with a COP incentive ? The fourth scenario is of specific interest for the planner : the required budget is lower than for the three first scenarios, as the incentives for the permanent meadows are compensated by the taxes on the COP. This scenario is the most realistic from an economic perspective in the sense that it is partially self-funded for the public stakeholder. In all four cases, this is a simplified model since the same policy is applied to all regions, whatever the economic or habitat features.

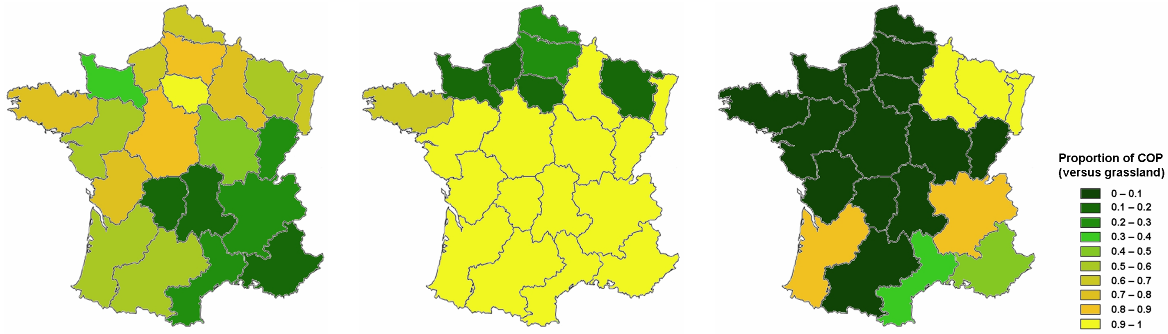
Furthermore we test two types of incentives : incentives can be constant for the projection or can decrease over the time. Incentives decrease linearly from previous levels in 2008 to achieve 0 in 2050 (eq. (II.9)). The decreasing incentives correspond to the current trend of the CAP.

$$\tau_k(t) = \tau_k(2008)(1 - \frac{t - 2008}{2050 - 2008}) \tag{II.9}$$

We study these scenarios from 3 points of view : a purely economic approach focusing on economic outcomes, a purely ecological approach studying ecological indicators and, finally, a combined bio-economic approach coupling both economic and ecological outputs. After the 43 years, the four scenarios lead to very contrasted situations as depicted by figure II.2.

At the first stage we noted that the rigidity parameter ϵ of the economic model plays qualitatively a similar role on all the results. It does not modify the qualitative nature of trajectories, but only speeds up the process : dynamics accelerate when ϵ increases.

Figures II.4, II.6, II.7 and II.8 display the results similarly. Each trajectory is composed of 43 points, corresponding to the years of the timeframe $t_0 = 2008$ to $t = 2050$. All trajectories start from the same



(a) Land use in 2008

(b) Land use in 2050 with the Crop scenario

(c) Land use in 2050 with the HQE scenario

FIGURE II.2 – Comparison of land uses before and after the projections, with the Crop and HQE scenarios. The projections are calculated with a rigidy parameter $\epsilon = 10\%$ and a level of incentives $\tau = 0.1$.

point at the lower left corner of the figure.

2.5. Model calibration

Data

We considered the 21 regions of metropolitan France as the unit of spatial scale. To assess the ecological performance, we here chose to focus on common bird populations and related indicators (Gregory *et al.*, 2004). Although the metric and the characterization of biodiversity remain an open debate (Le Roux, 2008, MEA, 2005), such a choice is justified for several reasons (Ormerod & Watkinson, 2000) : (i) Birds lie at a high level in the trophic food chains and thus capture the variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation and viability (Sekercioglu *et al.*, 2004). (iii) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens.

The STOC database managed by the Museum National d'Histoire Naturelle provides the data related to the bird abundances (details in Jiguet *et al.* (2010)). Among the species monitored by this program, we have selected 34 species which have been classified as farmland and habitat generalist species according to their habitat requirements at a Europe-wide scale European Bird Census Council (2007). Table II.1 displays the 14 habitat generalist species and the 20 species used as a reference for the European Farmland Bird Index FBI (Gregory *et al.*, 2004). Previous analyses have shown the relevance of the national FBI to reflect the response of farmland biodiversity to agriculture intensification (Doxa *et al.*, 2010). The regional abundances for the years 2001 to 2007 are available for each of these species.

Agronomical data measuring the surfaces of the various agricultural practices are published by the French Statistics Service of the Department of Agriculture [1] for the years 2002 to 2007. The agricultural surface is divided into 10 classes of land-uses as captured by Table II.2. Finally the economic data relating to the gross margins are derived from the national FADN (Farm Accountancy Data Network) [2] and the European FADN [3] for France. They are computed for each activity in each region. To insure consistency between the two databases, we balanced the economic data (coming from the European FADN) by the ratio of the regional Utilized Agricultural Areas UAA(national FADN)/UAA(European FADN).

1. http ://agreste.agriculture.gouv.fr/
2. http ://agreste.agriculture.gouv.fr/
3. http ://ec.europa.eu/agriculture/rica/

20 farmland bird species	14 generalist bird species
(1) Buzzard *Buteo buteo*	(1) Blackbird *Turdus merula*
(2) Cirl Bunting *Emberiza cirlus*	(2) Blackcap *Sylvia atricapilla*
(3) Corn Bunting *Emberiza calandra*	(3) Blue Tit *Parus caeruleus*
(4) Grey Partridge *Perdix perdix*	(4) Carrion crow *Corvus corone*
(5) Hoopoe *Upupa epops*	(5) Chaffinch *Fringilla coelebs*
(6) Kestrel *Falco tinnunculus*	(6) Cuckoo *Cuculus canorus*
(7) Lapwing *Vanellus vanellus*	(7) Dunnock *Prunella modularis*
(8) Linnet *Carduelis cannabina*	(8) Great Tit *Parus major*
(9) Meadow Pipit *Anthus pratensis*	(9) Green Woodpecker *Picus viridis*
(10) Quail *Coturnix coturnix*	(10) Golden oriole *Oriolus oriolus*
(11) Red-backed Shrike *Lanius collurio*	(11) Jay *Garrulus glandarius*
(12) Red-legged Partridge *Alectoris rufa*	(12) Melodius Warbler *Hippolais polyglotta*
(13) Rook *Corvus frugilegus*	(13) Nightingale *Luscinia megarhynchos*
(14) Skylark *Alauda arvensis*	(14) Wood Pigeon *Columba palumbus*
(15) Stonechat *Saxicola torquatus*	
(16) Whinchat *Saxicola rubetra*	
(17) Whitethroat *Sylvia communis*	
(18) Wood Lark *Lullula arborea*	
(19) Yellowhammer *Emberiza citrinella*	
(20) Yellow Wagtail *Motacilla flava*	

TABLE II.1 – List of the 20 farmland and 14 generalist bird species

Calibration of the bio-economic model

This step consists in determining the Beverton-Holt parameters through a calibration. For each of the 34 species, we must estimate the growth parameter R constant over the region as well as the α and β parameters specific to each region. We use a least square method to minimize errors between the observed abundances $N_{s,r}^{Data}$ as issued from STOC and the values derived from the model $N_{s,r}$:

$$\min_{\mathrm{R},\alpha,\beta} \sum_{\mathrm{s,r}} \|\mathrm{N}_{\mathrm{s,r}}^{\mathrm{Data}} - \mathrm{N}_{\mathrm{s,r}}\|^2 \tag{II.10}$$

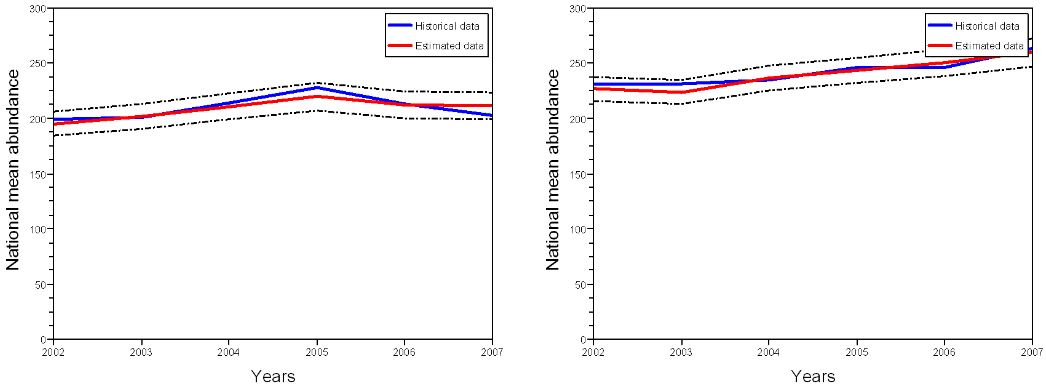
Figure II.3 illustrates the results of this calibration with two species : the Sky Lark *Alauda arvensis* (farmland specialist) and the Chaffinch *Fringilla coelebs* (generalist species). The errors of estimation are small (between 4% and 6% for the illustrated species) and the historical data do not go beyond the confidence interval (coming from the least square standard errors of calibration). Comparing the historical data with the model-generated data, we note that the model tends to smooth the variations of the observed data.

For the economic part, we computed the mean gross margin per surface $\overline{mb}_{r,k}$ for each land use and each region as the arithmetic mean of the historical data between 2002 and 2007. We therefore use an average market situation for our simulations.

$$\overline{mb}_{r,k} = \frac{\sum_{t=2002}^{2007} mb_{r,k}(t)}{6} \tag{II.11}$$

Hay production	No hay production	Others
Annual hay	Cereals-Oleaginous-Proteaginous	Fallow
Temporary grassland	Industrial crop	Forest
Permanent grassland	Horticultural crop	No agricultural surface
	Permanent crop	

TABLE II.2 – Distribution of the 10 land uses considered by the economic model



(a) Sky Lark (b) Chaffinch

FIGURE II.3 – Comparison of historical N_s^{Data} and estimated N_s abundances with the confidence interval (coming from the least square standard errors of calibration)

3. Results

3.1. Economic analysis

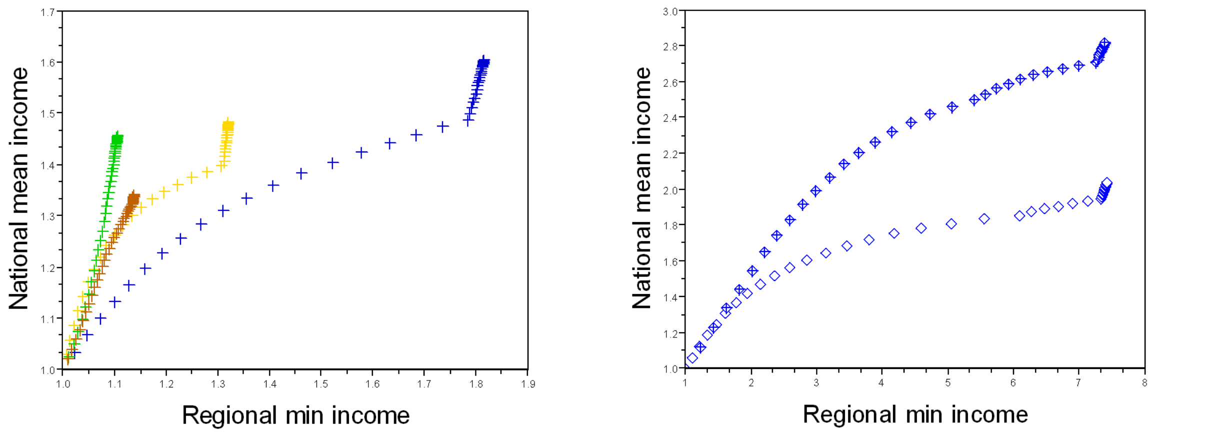
Figure II.4(a) allows for a comparison of the 4 scenarios based on identical rigidity ϵ and constant incentive τ parameters, set respectively at 10% and 0.5 : we use the indicators measuring the regional min income and the national mean income defined in equation (II.6) and (II.7). It turns out that the economic value always increases since the farmers are maximizing their income. To illustrate this point, in figure II.5 we represent the evolution of the regional income for one French region, Alsace. On the graph II.4(a), we note that the HQE scenario is the one which generates the best results on both indicators. The Grassland and DS scenarios are the least efficient. However, the differences between the policies are sensitive to the chosen index. The min regional income is more sensitive to the scenarios than the national mean gross margin.

For the graph II.4(b), we focus on the effect of τ parameter on the HQE scenario which was the more efficient. We compare the outcomes for two constant values of τ (1 and 1.8). By contrast to the figure II.4(a), the min regional income shows similar performances for the two taxation levels while the national mean income emphasizes their differences.

3.2. Ecological analysis

The figure II.6(a) allows for a comparison between the four scenarios thanks to two ecological indicators with same values for ϵ and τ, set respectively at 10% and 0.5, which we keep constant. We note that the HQE policy is the most efficient scenario from the point of view of both ecological indicators. Although the Crop scenario was quite efficient on the economic side, it appears as the least ecological efficient from an ecological point of view. The two indicators do not distinguish the scenario with the same sensitivity. The levels reached at the end of the trajectories are rather similar for the four policies regarding the generalist species viewpoint, while their differences are significantly highlighted by the farming specialist species. Figure II.5 illustrates this difference of growth with two species in Alsace.

The graph II.6(b) focuses on the HQE policy and exhibits four levels of taxation. It shows that the STOC indicator for generalists does not distinguish the various levels of taxation. The STOC farmland bird index (FBI) is more sensitive. In addition, the figure II.6(b) captures a new effect that did not appear with the economic analysis : a non-monotonicity of the impact of taxation level on the FBI. For its low value,



(a) Economic trade-off for the 4 scenarios (yellow : Crop scenario, green : Grassland scenario, brown : DS scenario, blue : HQE scenario) with the same rigidity and incentive level parameters $\epsilon = 10\%$ and $\tau = 0.5$

(b) Impact of incentive level τ (diamond : $\tau = 1$, crossed diamond : $\tau = 1.8$) on the economic trade off for the HQE scenario, with the rigidity parameter $\epsilon = 10\%$

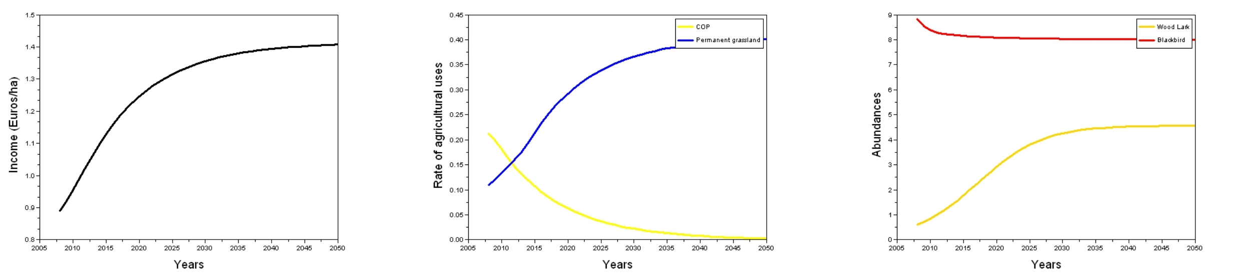
FIGURE II.4 – Impact of scenarios and incentive level τ on the economic trade-off between the national mean income ($\overline{\text{income}}_{\text{France}}$) and the regional min income ($\underline{\text{income}}_{\text{France}}$)

an increase of τ improves the ecological efficiency of the trajectory (with $\tau = 0.1$ and $\tau = 0.5$). On the contrary, for its high values, an increase of the taxation level has a negative marginal effect on the indicator as emphasized by the trajectory with $\tau = 1$ and $\tau = 1.8$.

3.3. Bio-economic analysis

The most effective scenario for both ecological and economic dimensions is the HQE scenario as depicted on figure II.7(a). Figure II.7(b) displays the trajectories of the HQE scenario for 4 constant levels of taxation. With respect to ecological and economic indicators, such a figure provides insights into the relationships and trade-offs between ecological and economic requirements. We remark that depending on the value given to τ, we get a full range of trajectories covering the set of possibilities. No trajectory exhibits an improvement of the system for both ecological and economic dimensions : some trajectories are more ecological efficent (for example with $\tau = 0.1$ and 0.5), while others show a better economic efficiency ($\tau = 1$ and 1.8).

We note that some values of the national mean income can be obtained for all trajectories (for example

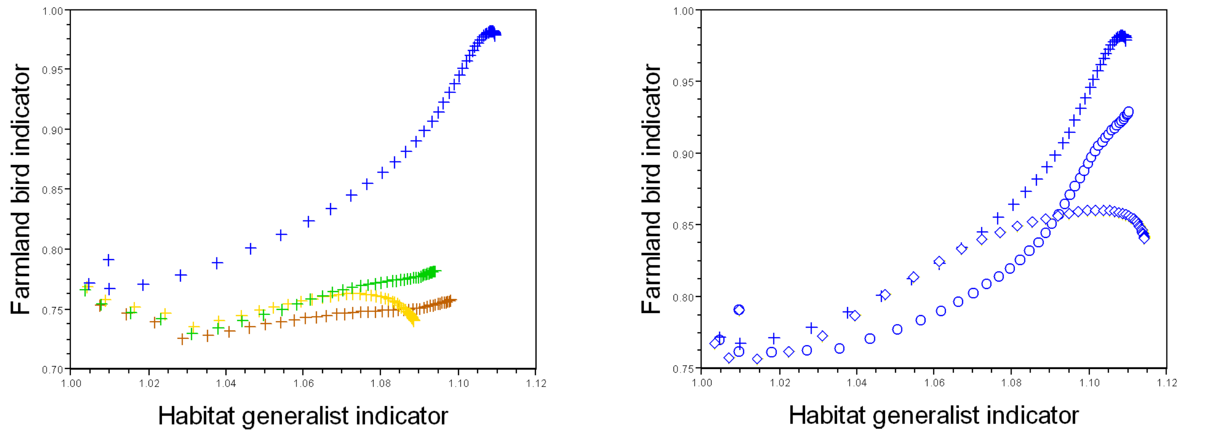


(a) Economic evolution

(b) Agronomic evolutions : percentages of COP and permanent grassland areas in total cultivated area

(c) Ecological evolutions : abundances of a generalist species (Blackbird) and a farmland specialist species (Wood Lark)

FIGURE II.5 – One example of economic, agronomic ans ecological trajectories in the Alsace region for the HQE scenario. The rigidity parameter is $\epsilon = 10\%$ and the incentive level $\tau = 0.5$



(a) Ecological trade-off for the 4 scenarios (yellow : Crop scenario, green : Grassland scenario, brown : DS scenario, blue : HQE scenario) with the same rigidity and incentive level parameters $\epsilon = 10\%$ and $\tau = 0.5$

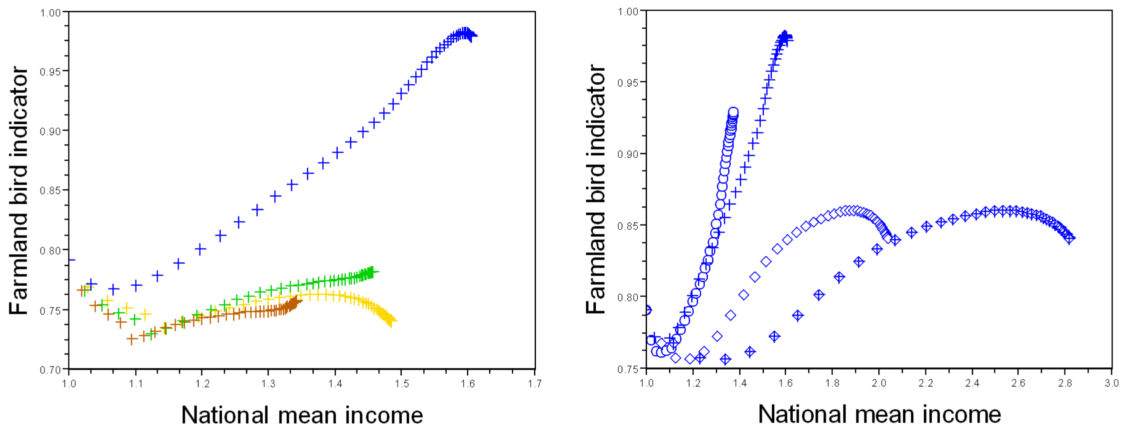
(b) Impact of the incentive level τ (circle : $\tau = 0.1$, plus : $\tau = 0.5$, diamond : $\tau = 1$, crossed diamond : $\tau = 1.8$) on the ecological trade off for the HQE scenario with the rigidity parameter $\epsilon = 10\%$. Diamond and crossed diamond trajectories are overlaying.

FIGURE II.6 – Impact of scenarios and the incentive level τ on the ecological trade-off between the farmland bird indicator ($STOC_{France,farmland}$) and the habitat generalist indicator ($STOC_{France,generalist}$)

margin $= 1.3$). However this value is not reached at the same speed for the four trajectories : with $\tau = 1.8$ (1, 0.5, 0.1 respectively), this income is obtained after 4 years (6 years, 11 years, 40 years respectively). The more stringent are the policies (higher level of incentive), the faster the income is reached. However, for a given income, the slowest trajectory is the most ecological efficient : for a national mean income of 1.3, the STOC indicator of farmland specialists provides a level of 0.93 with $\tau = 0.1$, against only 0.77 for $\tau = 1$.

3.4. Analysis of decreasing incentives

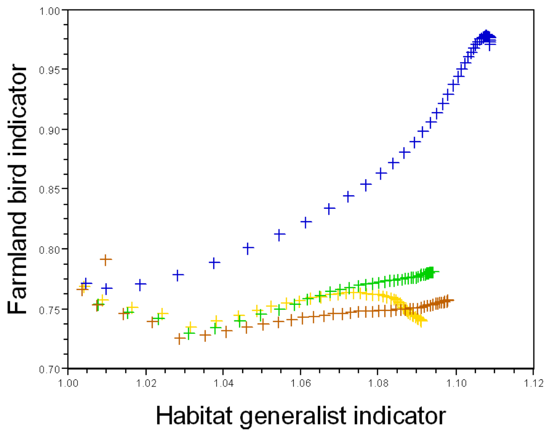
Figure II.8 illustrates the ecological and economic performances for the four scenarios with decreasing incentives (incentives start at $\tau = 0.5$ and go linearly to $\tau = 0$ at 2050 as defined in equation (II.9)). We



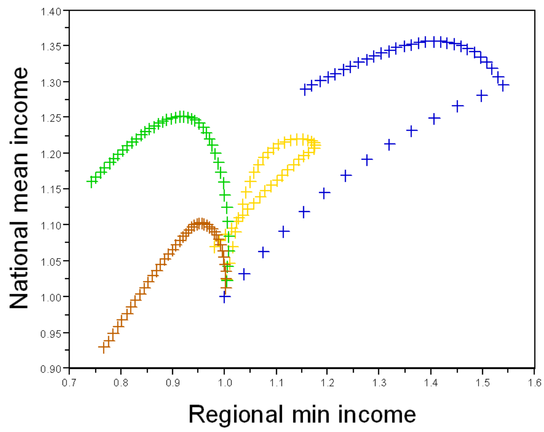
(a) Bio-economic trade-off for the 4 scenarios (yellow : Crop scenario, green : Grassland scenario, brown : DS scenario, blue : HQE scenario) with the same rigidity and incentive level parameters $\epsilon = 10\%$ and $\tau = 0.5$

(b) Impact of incentive level τ (circle : $\tau = 0.1$, plus : $\tau = 0.5$, diamond : $\tau = 1$, crossed diamond : $\tau = 1.8$) on the bio-economic trade off for the HQE scenario, with the rigidity parameter $\epsilon = 10\%$

FIGURE II.7 – Impact of scenarios and the incentive level τ on the bio-economic trade-off between the farmland bird indicator ($STOC_{France,farmland}$) and the national mean income ($\overline{income}_{France}$)



(a) Ecological trade-off for the 4 scenarios (yellow : Crop scenario, green : Grassland scenario, brown : DS scenario, blue : HQE scenario) with the rigidity parameter $\epsilon = 10\%$ and decreasing incentive level



(b) Economic trade-off for the 4 scenarios (yellow : Crop scenario, green : Grassland scenario, brown : Double subsidy scenario, blue : HQE scenario) with the rigidity parameter $\epsilon = 10\%$ and decreasing incentive level

FIGURE II.8 – Impact of decreasing incentives on ecological trade-off between the farmland bird indicator ($\text{STOC}_{\text{France,farmland}}$) and the habitat generalist indicator ($\text{STOC}_{\text{France,generalist}}$) and the economic trade-off between the national mean income ($\overline{\text{income}}_{\text{France}}$) and the regional min income ($\text{income}^{-}_{\text{France}}$)

observe on the graph II.8(a) that the ecological outcomes are quite similar to those with trajectories with constant τ (graph II.6(a)). By contrast, economic performances differ. The beginning of economic trajectories are similar to those of the figure II.6(b) but around 2020 both national mean income and regional min income start to decrease : agricultural choices done under strong subsidies become not profitable with low subsidies. For Crop and DS scenarios, the national mean income is the most affected. But for HQE and Grassland scenarios, it is the regional min income which is the most sensitive to decreasing incentives.

4. Discussion

4.1. Model construction

This paper presents an inter-disciplinary approach which is needed (Perrings *et al.*, 2006) to develop sustainable management of biodiversity and agriculture. Despite divergences between economic and ecological disciplines and approaches (Drechsler *et al.*, 2007), our model couples economic and ecological modelling to analyze bio-economic performances of French agricultural public policies at the national scale. This approach avoids the biodiversity monetary evaluation which is controversial (Rees, 1998). The coupling of economic (gross margins), agronomical (land uses) and ecological (bird abundances) data gives a strong realism to the modelling. Integrating these many data allows to obtain robust and informative results. With the integration of regional economic and ecological features, the model is spatialized, which reinforces its relevance (Polasky *et al.*, 2005). The choice to focus on common birds rather than one or two species leads to obtain more general results and constitutes a major step towards biodiversity analysis. Finally, the dynamic aspect which allows for an adjustment of carrying capacity through land uses lead to a precise representation of impacts of land-uses on avifauna evolution and transient dynamics. The initial integration of spatio-temporal elements and many databases creates a multi-criteria framework suggesting many developments without any modification of the basic modelling structures.

4.2. Ecological-economic reconciliation

With this bio-economic prototype, we have shown that both the ecological and economic performance are impacted by the public policies for agriculture and land-use. A basic economic instrument (subsidy/tax) separates policies according to the two criteria. It suggests that managing the agricultural practices in bio-economic terms is possible thanks to a simple economic distortion of the marginal incomes as argued by Alavalapati *et al.* (2002) and Shi & Gill (2005). The model illustrates that it is possible to build scenarios favourable on the long term to both ecological and economic criteria. It should therefore be possible to design public strategies improving both farmer incomes and the bird biodiversity.

4.3. Index selection

The evaluation of the various policies is highly dependent on the selected criterion. On the ecological side, the situation is clear : the STOC indicator for the generalist community is not significant enough to distinguish the policies while on the contrary the STOC indicator of the farmland specialists is sensitive to the proposed scenarios. This results is in direct line with the historical abundances (Julliard *et al.*, 2006). The generalist species are less farming-dependent since they can adapt, at least on the short term, to these perturbations by switching to other habitats. The farmland specialist species, the adaptation capability of which is lower, are more affected by the various scenarios. This result leads better understandings of the ecosystem, which is essential for the success of a sustainable policy (Drechsler & Watzold, 2007, Hein & Ierland, 2006). Hence, the STOC criterium of the farming species turns out as a relevant ecological metric for bird diversity to analyze impacts of public policies on biodiversity.

On the economic side, the two selected indicators relate to different methodologies : a global approach based on the national mean income, and a worst case approach based on the min regional income. Both indicators are sensitive to the policies but marginal impacts of the different policies depend on the economic criterion. The planner must choose the most adequate indicator for its target, keeping in mind the sensitivity of the two indicators to the scenario and to the level of incentives. Selection of the economic indicator is consequently a crucial stage before analyzing impacts of public policies on economy, since it partially influences the final conclusion.

4.4. The ecology-economy trade-off

The conciliation of ecological and economic performances under public policies is possible but not straightforward. Many points of this trade-off have to be discussed. This study focuses on three of them. Our study first suggests that synergies between adequate political incentives occur. Acting simultaneously on various incentives can improve performances from both the ecological and economic points of view as shown with the HQE scenario.

The second point is the time horizon of the incentives. Short-term incentives are sufficient to drive farmer choices and thus ecological performances but are not sustainable for the economic part. Farmers initially adopt activities which alter their rentability when the incentive level decrease.

The third point regards the priority between ecological and economic objectives. As depicted by graphs of the bio-economical approach (fig. II.7), no unique pareto optimum emerges : even if both criteria are improved with this scenario, it is always necessary to prioritise to ecological and economical objectives. Consequently, a set of admissible strategies is available to bring together ecological and economic performances. The challenge consists in selecting which farming activities should be subsidised or taxed and which magnitude of subsidy/tax is the most adequate in order to optimise trajectories for the set of ecological and economic criteria.

Others points of the ecological-economic reconciliation are underlying our study. Along the bio-economic trajectories, we have seen that the speed of change is very fluctuating. This variation gives another level of trade-off in terms of timeframe : how fast does the public agency want to reach the objectives ? The total budget of the regulating agency is another key element of the strategy. In our model, we have not imposed budgetary constraint. However, in a larger perspective, decision-making support requires the integration of this budgetary limitation in the model. Some policies may be attractive from ecological and economic viewpoints but not feasible in terms of public balance. Considering this global budget limitation raises the question of budget allocation to the regions. Contrary to the micro-scale studies (Münier *et al.*, 2004) and meso-scale studies (Holzkamper & Seppelt, 2007, Shi & Gill, 2005), our modelling at the national scale leads to study the budget regional allocation. This question is highly dependent on the selection of the economic indicator. Does the objective consist in reaching a maximal national mean, a maximal level for the poorest region or a minimal variability over the regions ? This spatial share of the global budget highly conditions the economical and ecological performances of each region, as well as for the whole country.

5. Conclusion

This work explores joint managements of agriculture and biodiversity. A macro-regional model has been developed and calibrated by coupling several ecological and agricultural databases monitored at the French scale. The model articulates the dynamic of large community of bird species impacted by agricultural land-use and an economic decision model based on farming gross margins for each French region. Through different scenarios relying on national subsidies and taxes, we have examined and assessed the impact of public policies on both biodiversity, regional land-use and farming economic outcomes. It has been shown how simple economic instruments could be used to simultaneously promote economic performance and bird populations. The bio-economical analysis shows several solutions for such ecology-economy trade-off. These results suggest that different options are available to develop multi-functional sustainable agriculture. It is, as well, pointed out to what extent the sustainability of the different policies is sensitive to the ecological and economic indicators used by the decision maker.

As suggested by the results, this model contributes to evaluations of public decisions according to several dimensions : conservation performances, economic performances, public budget, synergies between incentives, timing for objectives, incentives territoriality. The structure of this model allows for many developments. For the economic part, we can add risk aversion behavior (Quaas *et al.*, 2007) and uncertainty on gross margins due to climatic or market uncertainty. On the ecological side, it is possible to add migrations between regions and evolutions of ecological niches in response to climatic change (Brotons & Jiguet, 2010), or test other abundances indicators. From the public policy point of view, many scenarios can be tested : dynamic evolution of incentives, budgetary constraint, optimal policy. Finally, the use of inverse approaches including optimality in particular multi-criteria (Groot *et al.*, 2007) or viability (constraints) methods (Tichit *et al.*, 2007, Baumgärtner & Quaas, 2009) instead of exogeneous incentive-based scenarios should be fruitful for the search of relevant incentive strategies.

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Chapitre III

How does economic risk aversion affect biodiversity ?

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Abstract

Significant decline of biodiversity in farmlands has been reported for several decades. To limit the negative impact of agriculture, many agri-environmental schemes have been implemented but their effectiveness remains controversial. In this context, the study of economic drivers is helpful to understand the role played by farming on biodiversity. The present paper analyzes the impact of the risk aversion on the farmland biodiversity. A bio-economic model which articulates bird community dynamics and representative farmers selecting land uses within an uncertain macro-economic context is developed. It is specialized and calibrated at a regional scale for France through national databases. The influence of risk aversion is assessed on ecological, agricultural and economic outputs through projections at the 2050 horizon. A high enough aversion appears sufficient to both manage economic risk and promote ecological performance. This occurs through a diversification mechanism on regional land-uses. However economic calibration leads to a weak risk aversion parameter, which is consistent with the current decline of farmland birds. Two hypothesis are proposed to explain this low aversion and could appear as interesting leverages to favor farmer diversification behavior. Spatial disparities however suggest that public incentives could be necessary to reinforce the diversification and bio-economic effectiveness.

Keywords : Farmland Bird, Agriculture, Bio-economic modelling, Diversification

1. Introduction

Significant decline of biodiversity in European farmlands has been reported for several decades. Numerous studies point out spatial and temporal correlations between farmland biodiversity and agricultural changes (Chamberlain *et al.* 2000, Donald *et al.* 2001, Wretenberg *et al.* 2007). Modern agriculture and associated intensification of practices have been identified as major drivers of this erosion in farmland biodiversity. The breeding bird populations are particularly vulnerable to global agricultural change (Krebs *et al.* 1999, Chamberlain *et al.* 2000). Such a negative effect is due mainly to a degradation in habitat quality altering nesting success and survival (Benton *et al.* 2003). In this context, the European Union has formally adopted the Farmland Bird Index (FBI) as an indicator of structural changes in biodiversity (Balmford *et al.* 2003).

A challenge to reach sustainability for agricultural land-use is therefore to reconcile farming production and farmland biodiversity. Usual approaches to achieve such multifunctional goals for farming rely on public policies (Pacini *et al.* 2004) or economic incentives (Drechsler *et al.* 2007a, Mouysset *et al.* 2011). For Alavalapati *et al.* (2002) and Shi & Gill (2005), financial incentives are essential to convincing farmers to adopt eco-friendly activities. These policies modify the farmer's choices and thus impact both the habitat and the dynamics of biodiversity (Doherty *et al.* 1999, Holzkamper & Seppelt 2007, Rashford *et al.* 2008). In this perspective, many public policies including agri-environmental schemes have been developed by decision makers. However, fifteen years after the initial implementation of such instruments at a large scale, their ability to enhance biodiversity remains controversial (Vickery *et al.* 2004, Kleijn *et al.* 2006, Butler *et al.* 2009) . In this context, explore some micro-economic characteristics could be helpful to understand the impact of the farmers behaviors on biodiversity and eventually improve the effectiveness of the public policies. In particular, some studies focus on the farmer's microeconomic features, treating them as forms of risk aversion (Hardaker 2000, Lien 2002). Hence theoretical models (Quaas *et al.* 2007) suggest that an adequate risk aversion may bring farmers to adopt sustainables choices in the case of rangelands. The underlying mechanism is that risk averse farmers maintain an important agrobiodiversity (i.e. a genetic diversity) in their farming system as a way of managing increasing economic risk (Di Falco & Perrings 2003 in croplands, Schläpfer *et al.* 2002 in grasslands, Baumgärtner & Quaas 2010). Furthermore, strong agrobiodiversity also has a positive impact on farmland biodiversity (Robinson & Sutherland 2002, Laiolo 2005).

The objective of this work is to examine how does the farmer risk aversion affect the farmland biodiversity. For this purpose, a bio-economic dynamic model is developed for metropolitan France spatialized at the regional scale. By comparing the role played by the degrees of farmer risk aversion on bio-economic outcomes, it aims at quantifying the impact of aversion on the agro-ecosystem for both private (income) and public (farmland birds) goods.

To address such agro-environmental issues, different bio-economic modelling frameworks are proposed in the literature. Cost-Benefit methods require quantification of biodiversity in monetary terms (Drechsler 2001, Rashford *et al.* 2008). Although pricing techniques such as contingent valuation are available, their suitability for the complex issues of biodiversity is disputed, notably in anthropogenic systems (Diamond & Hausman 1994). In this context, cost-effectiveness is an interesting alternative to avoid monetary evaluation of environmental goods (Gatto & De Leo 2000). Approaches such as ecological economics suggest studying environmental and economic performances simultaneously, stressing the relevance of multi-criteria approaches (Drechsler *et al.* 2007a, Mouysset *et al.* 2011). However the metrics to adopt for evaluating biodiversity are not self-apparent and indicators used to assess biodiversity and environmental services are highly diverse (van Wenum *et al.* 2004, Havlik *et al.* 2005,Polasky *et al.* 2005). Moreover, numerous models emphasize spatial dimensions in dealing with agro-ecological issues. Such spatially explicit models aim at assessing consequences of different land use patterns for various environmental and economic criteria (Irwin

& Geoghegan 2001, Polasky *et al.* 2005, Groot *et al.* 2009). Nevertheless, most of these models are static, restricting the potential ecological processes accounted for. In the same vein, most of these models are deterministic and do not take into account the various uncertainties involved in the ecological and economic processes at play.

The bio-economic model proposed in this article is in direct line with these considerations. The model integrates representative rational agents selecting farming land-uses in an uncertain economic context through some expected utility and bird community dynamics driven by these land uses. The model is thus dynamic. Furthermore, it articulates ecological and economic compartments and adopts a multi-criteria perspective. Moreover, it offers a spatialized perspective as it is built up at a macro-regional scale and its calibration relies on French regional data of both land-use and bird abundance. Biodiversity is measured through the European Farmland Bird Index (FBI) which has already shown its relevance to reflect the response of farmland biodiversity to intensification of agriculture (Doxa *et al.* 2010, Mouysset *et al.* 2011, Mouysset *et al.* 2012). Moreover, the model accounts for economic uncertainties through gross margins. In this context, different projections and scenarios at the 2050 horizon give insights into the positive influence of economic risk aversion for reconciling agricultural income and biodiversity. It is shown how such multi-functionality is related to the heterogeneity of farming habitats and land-uses.

The paper is organized as follows. The second section describes the spatialized dynamic model and the bio-economic indicators. The third section presents the results regarding the influence of risk aversion on bio-economic performances. The fourth section is devoted to the discussion of these results.

2. Material and method

2.1. Context and data

Metropolitan France is split into 620 small agricultural regions (PRA for Petites Regions Agricoles). A PRA is part of a department (a major French administrative entity) which exhibits an agro-ecological homogeneity. This consistency from both the ecological and economic points of view makes the PRA scale well suited for our bio-economic modelling. The model described below is built for each PRA.

To assess the ecological performance, we here choose to focus on common bird populations and related indicators (Gregory *et al.* 2004). Although the metric and the characterization of biodiversity remain an open debate (MEA 2005), such a choice is justified for several reasons (Ormerod & Watkinson 2000) : (i) Birds lie at a high level in the trophic food chains and thus capture the variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation and viability (Sekercioglu et al., 2004). (iii) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens.

The STOC (French Bird Breeding Survey) database[1] provided the informations related to the bird abundances across the whole country. Abundance values for each species were available for the period 2001-2008. For each species, we further performed a spatial interpolation of these abundance data to obtain relative abundance values for each possible square in the country (e.g. 136 000 squares) using kriging models based on spatial autocorrelation and the exponential function (Doxa *et al.* 2010). We then averaged the abundance values at the PRA scale. Among the species monitored by this large-scale long-term survey, we selected 20 species which have been classified as farmland specialists according to their habitat requirements at a Europe scale (European Bird Census Council 2007). Table III.1 lists the 20 species used as a reference for the

1. See the Vigie-Nature website http ://www2.mnhn.fr/vigie-nature/. Standardized monitoring of spring-breeding birds at 1747 $2 * 2\,km^2$ plots across the whole country. Details of the monitoring method and sampling design can be found in Jiguet 2009.

20 farmland bird species s
(1) Buzzard *Buteo buteo*
(2) Cirl Bunting *Emberiza cirlus*
(3) Corn Bunting *Emberiza calandra*
(4) Grey Partridge *Perdix perdix*
(5) Hoopoe *Upupa epops*
(6) Kestrel *Falco tinnunculus*
(7) Lapwing *Vanellus vanellus*
(8) Linnet *Carduelis cannabina*
(9) Meadow Pipit *Anthus pratensis*
(10) Quail *Coturnix coturnix*
(11) Red-backed Shrike *Lanius collurio*
(12) Red-legged Partridge *Alectoris rufa*
(13) Rook *Corvus frugilegus*
(14) Skylark *Alauda arvensis*
(15) Stonechat *Saxicola torquatus*
(16) Whinchat *Saxicola rubetra*
(17) Whitethroat *Sylvia communis*
(18) Wood Lark *Lullula arborea*
(19) Yellowhammer *Emberiza citrinella*
(20) Yellow Wagtail *Motacilla flava*

TABLE III.1 – List of the 20 farmland bird species used in the model

European Farmland Bird Index FBI (Gregory *et al.* 2004). Previous analyses have shown the relevance of the national FBI to reflect the response of farmland biodiversity to agricultural intensification (Doxa *et al.* 2010, Mouysset *et al.* 2012).

For agro-economic data, we use the French agro-economic classification OTEX developed by the French Farm Accounting Data Network (FADN) [1] and the Observatory of Rural Development (ODR) [2]. This organization distinguishes 14 classes of land-uses denoted by OTEA (see tab. III.2). Each PRA is a specific combination of these OTEA. The surfaces dedicated to each of the 14 OTEA and the associated gross margins relying on tax return, for the years 2001 to 2008 are available on the ODR website under a private request. Gross margin is an economic index broadly used in agricultural economics (for instance Lien 2002).

2.2. The ecological model

Regarding bird populations, we chose a dynamic framework. We here adopt the Beverton-Holt model which accounts for the intra-specific competition and the density dependence as follows :

$$N_{s,r}(t+1) = N_{s,r}(t).\frac{1+R_{s,r}}{1+\dfrac{N_{s,r}(t)}{M_{s,r}(t)}} \tag{III.1}$$

where $N_{s,r}(t)$ stands for the bird abundance of species s in a PRA r at year t. The $R_{s,r}$ coefficient corresponds to the intrinsic growth rate specific to each species s in a region r (Smith *et al.* 2009). This parameter takes into account the characteristics of each species such as clutch size, mean reproductive success, number of clutches per year. The variable $M_{s,r}$ captures the ability of the habitat to host the species and the product $M_{s,r}(t) * R_s$ represents the carrying capacity of the habitat r.

1. http ://ec.europa.eu/agriculture/rica/
2. https ://esrcarto.supagro.inra.fr/intranet/

The 14 land-uses (OTEA) k
(1) Cereal, Oleaginous, Proteaginous (COP)
(2) Variegated crops
(3) Intensive bovine livestock breeding
(4) Medium bovine livestock breeding
(5) Extensive bovine livestock breeding
(6) Mixed crop-livestock farming with herbivorous direction
(7) Other herbivorous livestock breeding
(8) Mixed crop-livestock farming with granivorous direction
(9) Mixed crop-livestock farming with other direction
(10) Granivorous livestock breeding
(11) Permanent farming
(12) Flower farming
(13) Viticulture
(14) Others associations

TABLE III.2 – List of the 14 farming land-uses (OTEA)

The habitat variable $M_{s,r}(t)$ is assumed to depend linearly on land-uses (OTEA) as follows :

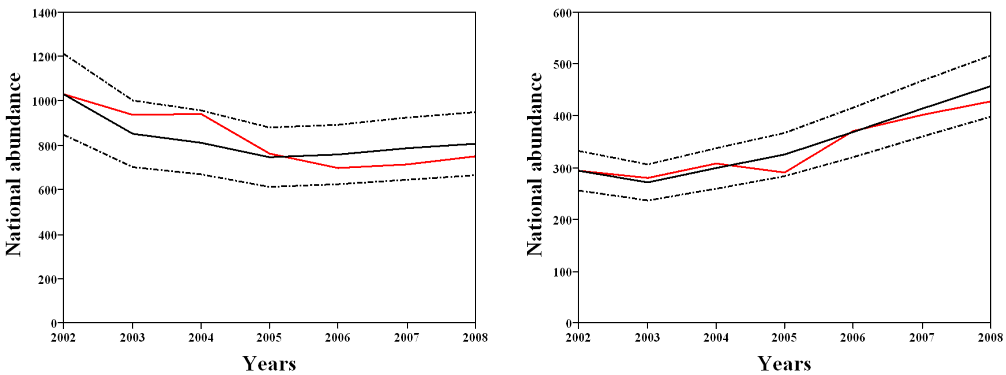
$$M_{s,r}(t) = \beta_{s,r} + \sum_{k} \alpha_{s,r,k}.A_{r,k}(t) \tag{III.2}$$

where $A_{r,k}(t)$ represents the share of the PRA r dedicated to OTEA k at time t. The $\alpha_{s,r,k}$ and $\beta_{s,r}$ coefficients, specific to each species, quantify how the species s responds to the various OTEA k in a given region r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a PRA r.

To estimate these different parameters, we use a least square method to minimize errors between the observed abundances $N_{s,r}^{Data}$ as issued from STOC survey and the values derived from the model $N_{s,r}$:

$$\min_{R,\alpha,\beta} \sum_{s,r,t} (N_{s,r}^{Data}(t) - N_{s,r}(t))^2 \tag{III.3}$$

Figure III.1 illustrates the results of this calibration for national abundances of two species : the Stonechat *Saxicola torquatus* and the Red-backed Shrike *Lanius collurio*. More globally, the mean errors of estimation per PRA are about 0.08%. Comparing the historical abundances with the model-generated ones, we note that the model tends to smooth the variations of the observed data.



(a) The Stonechat *Saxicola torquatus* (b) The Red-backed Shrike *Lanius collurio*

FIGURE III.1 – Comparison between historical $N_{s,r}^{Data}(t)$ (red) and estimated $N_{s,r}(t)$ (black) national abundances with the least square standard errors of calibration at 95% (dashed lines) for two of the species considered, the Stonechat *Saxicola torquatus* and the Red-backed Shrike *Lanius collurio*.

2.3. The economic model of the farmer

Each PRA r is assumed to be managed by a representative farmer who selects land-uses (OTEA) along time. The farmer determines the surfaces $A_{r,k}(t)$ of each OTEA k in a PRA r in order to maximize some expected utility depending on mean and dispersion of incomes together with risk aversion. The income $\mathrm{Inc}_r(t)$ is the sum of the incomes generated by the agricultural activities k through the unit gross margins $\mathrm{gm}_{r,k}(t)$:

$$\mathrm{Inc}_r(t) = \sum_k \mathrm{gm}_{r,k}(t).A_{r,k}(t) \tag{III.4}$$

Gross margins $\mathrm{gm}_{r,k}(t)$ are supposed to be uncertain. The variability on gross margins includes both market, production and climate uncertainties. A Gaussian distribution parametrized with the mean and the covariance matrix of the historical data is chosen to capture such uncertainties. Also assumed is a quadratic form for the utility function of the representative agent (Lien 2002). Even if the mean-variance preference can be critized (Gollier 2001), it is a convenient function from a modelling viewpoint. Hence, the utility $U_r(t)$ for the representative farmer corresponds to the difference between an expected income $\mathbb{E}[\mathrm{Inc}_r(t)]$ and its risky part $\mathbb{V}ar[\mathrm{Inc}_r(t)]$:

$$\begin{aligned} U_r(t) &= \mathbb{E}[\mathrm{Inc}_r(t)] - a.\mathbb{V}ar[\mathrm{Inc}_r(t)] && \text{(III.5)} \\ &= \sum_k \overline{\mathrm{gm}}_{r,k}.A_{r,k} \;-\; a \,.\, \sum_k \sum_{k'} \sigma_{r,k,k'}(t).A_{r,k}(t).A_{r,k'}(t) && \text{(III.6)} \end{aligned}$$

Expected gross margins $\overline{\mathrm{gm}}_{r,k}$ are the mean of the 7 historical years [1]. The coefficient a represents the risk aversion level of the farmer : the higher the a, more risk-averse the farmer. In particular $a = 0$ means farmers are risk neutral, they make their choices only focusing on the expected income. The risky term is computed with the covariance [2] $\sigma_{r,k,k'}$ between margins of land-uses k and k' in region r. The maximizing program of farmer's utility in an uncertain context is defined as follows :

$$\max_{A_{r,1};\ldots;A_{r,14}} U_r(t) \tag{III.7}$$

Furthermore, when maximizing the utility, the standard agent must comply with two constraints at every point in time :

$$|A_{r,k}(t) - A_{r,k}(t-1)| \leq \varepsilon.A_{r,k}(t-1) \tag{III.8}$$

$$\sum_k A_{r,k}(t) = A_r \tag{III.9}$$

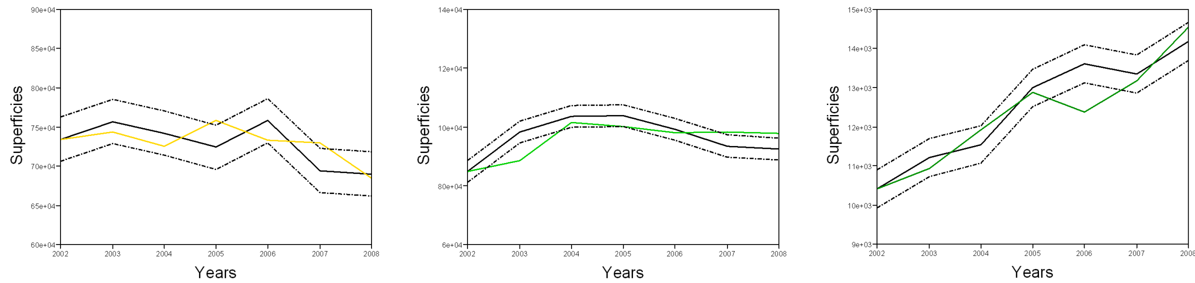
The first constraint (eq. III.8) corresponds to a technical constraint where the coefficient ε stands for the rigidity in changes. For example, the case where $\varepsilon = 0$ means that the land uses remain constant. The second constraint (eq. III.9) merely ensures that the total agricultural surface A_r per PRA remains constant. Typically, forest and urban areas are assumed to be steady.

To estimate the parameters a and ε, we use a least square method to minimize errors between the observed superficies $A^{Data}_{r,k}(t)$ dedicated to each OTEA as issued from the databases and the values derived from the model $A_{r,k}(t)$:

$$\min_{a,\varepsilon} \sum_{r,k,t} (A^{Data}_{r,k}(t) - A_{r,k}(t))^2 \tag{III.10}$$

1. $\overline{\mathrm{gm}}_{r,k} = \frac{1}{7}\sum_{t=1}^{t=7} \mathrm{gm}_{r,k}(t)$.

2. $\sigma_{r,k,k'} = \frac{1}{7}\sum_{t=1}^{t=7} (\mathrm{gm}_{r,k}(t) - \overline{\mathrm{gm}}_{r,k}(t)).(gm_{r,k'}(t) - \overline{\mathrm{gm}}_{r,k'}(t))$.



(a) COP ($k = 1$) (b) Extensive bovine livestock breeding ($k = 5$) (c) Other herbivorous livestock breeding ($k = 7$)

FIGURE III.2 – Three examples at PRA scale of comparison between historical $A^{Data}_{.,k}(t)$ (color) and estimated $A_{.,k}(t)$ (black) land-uses areas with the least square standard errors of calibration at 99% (dashed lines).

Figure III.2 illustrates the results of this calibration for three examples superficies at PRA scale : the COP (OTEA 1 in tab. (III.2)), the Extensive bovine livestock breeding (OTEA 5 in tab. (III.2)) and the Other herbivorous livestock breeding (OTEX 7 in tab. (III.2)). More globally, the average error of estimation at the national scale is about 1.3% per hectare. The calibration leads to $\varepsilon = 10\%$ and $a = 10^{-7}$.

2.4. Projections and indicators

Ecological and economic models described previously are thus linked through the agricultural system's OTEA as depicted by figure III.3. With the objective of maximizing incomes under technical and inertia constraints, the representative farmer in each PRA selects a pattern of OTEA $A_{r,k}(t)$ which impacts the ecological dynamics through the habitat $M_{s,r}(t)$. The farming land-uses are outputs of the economic model and inputs of the ecological model. The economic choices thus condition bird abundances $N_{s,r}(t)$ associated with the habitats.

We made different projections to analyze possible future trends of agriculture and biodiversity according to risk aversion of farmers involved in utility defined in eq. (III.6). We test eight absolute risk aversion levels a between 10^{-4} and 10^{-8} as suggested by Lien 2002 in a same economic context. For the projections, we do not add public policies in contrast to Mouysset *et al.* 2011. In other words, the case studied here corresponds to a Statu Quo scenario in the sense that it is assumed that the farmers evolve under the current policy context. To focus on the effect of the only aversion parameter, we consider the farmers as price takers since their choices do not affect temporally the gross margins. The selected time frame runs from 2009 to 2050, i.e a 42-year forecast. Selecting a shorter time frame could consequently hide interesting long-term effects due to the inertia of the models.

To analyze bio-economic performances, we focus on both ecological, land-uses and economic performances at national and regional scales.

2.5. Biodiversity index

From an ecological viewpoint, we have selected the Farmland Bird Index (FBI). We focus on this indicator which has been adopted by the European Community as the official environmental index especially to analyze structural changes in biodiversity (Balmford *et al.* 2003). FBI is an index of variation in abundances here computed with respect to the reference year 2008. We first estimated a regional FBI with 20 farmland

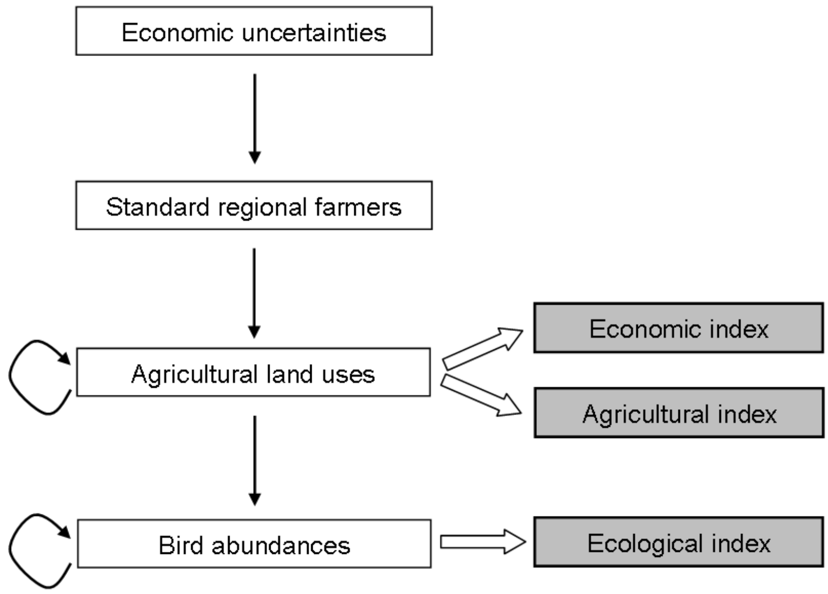


FIGURE III.3 – Model coupling : Farmers maximize their utility function and adjust their land uses depending on economic uncertainty and their risk aversion. These choices affect bird community dynamics.

specialist species (tab. III.1) for each PRA r as follows :

$$\mathrm{FBI}_r(t) = \prod_{s\,\in\,\mathrm{Specialist}} \left(\frac{N_{s,r}(t)}{N_{s,r}(2008)}\right)^{1/20} \tag{III.11}$$

Then at the national scale for France, we considered the aggregated indicator $\mathrm{FBI}_{\mathrm{nat}}$

$$\mathrm{FBI}_{\mathrm{nat}}(t) = \prod_{s\,\in\,\mathrm{Specialist}} \left(\frac{N_{s,\mathrm{nat}}(t)}{N_{s,\mathrm{nat}}(2008)}\right)^{1/20} \tag{III.12}$$

where $N_{s,\mathrm{nat}}(t)$ stands for the total abundance of species s over all PRA r.

2.6. Economic index

From an economic viewpoint, we use the regional income per income $\mathrm{Inc}_r(t)$ defined in equation (I.6) and the national mean income per hectare $\overline{\mathrm{Inc}}_{\mathrm{nat}}(t)$ defined in eq. (III.13). The national income is computed from the mean gross margin of the 620 PRA :

$$\overline{\mathrm{Inc}}_{\mathrm{nat}}(t) = \frac{1}{A_{\mathrm{nat}}} \sum_{r=1}^{620} A_r.\mathrm{Inc}_r(t) \tag{III.13}$$

where $A_{\mathrm{nat}} = \sum_{r=1}^{620} A_r$ is the total surface of PRA over France. For sake of clarity, we will represent this criterion on figures III.4 and III.5 after a normalization by their current value (2008).

2.7. Habitat heterogeneity index

To analyze farming habitat heterogeneity, we use a habitat heterogeity index denoted by $\mathrm{Hdiv}_r(t)$ which corresponds to an agricultural diversification index. In the same vein than Di Falco & Perrings (2003), it is here computed as the Simpson Index of land-uses $A_{r,k}(t)$:

$$\mathrm{Hdiv}_r(t) = \left(\sum_{k=1}^{14} \left(\frac{A_{r,k}(t)}{A_r(t)} \right)^2 \right)^{-1} \tag{III.14}$$

This indicator evaluates the bias compared to the equi-distribution. Its maximum is achieved when the agricultural area is divided equally between the 14 OTEA.

We also estimate an average heterogeneity indicator over France $\mathrm{Hdiv}_{\mathrm{nat}}(t)$ as an arithmetic mean of the 620 indicators at the PRA scale :

$$\mathrm{Hdiv}_{\mathrm{nat}}(t) = \frac{1}{620} \sum_{r=1}^{620} \mathrm{Hdiv}_r(t) \tag{III.15}$$

3. Results

As the modelling is realized in an uncertain context, we run one hundred simulations with different random Gaussian gross margins $\mathrm{gm}_{r,k}(t)$ from $t = 2009$ to $T = 2050$. Then both at PRA and national scales, we compute at any time the mean of the simulations for ecological, economic and habitat heterogeneity indices $\mathrm{FBI}(t)$, $\mathrm{Inc}(t)$ and $\mathrm{Hdiv}(t)$ along with their 99% confident interval.

3.1. Bioeconomic performances depending on risk aversion

We first compare the bio-economic performances $\mathrm{FBI}(t)$ and $\mathrm{Inc}(t)$ for the various levels of risk aversion at the national scale. Figure III.4 represents the mean of the 100 simulations for different risk aversion a. We observe a set of contrasted trajectories : Those with strong risk aversion levels are beneficial to the biodiversity while those with weak risk aversion levels promote the economic indicator. In other words, the risk aversion plays a significant role in the bio-economic performances achieved along time. However the ecological and economic performances are negatively correlated and thus different trade-offs can occur : there is no path optimizing both the economic and the biodiversity criteria compared to the initial situation.

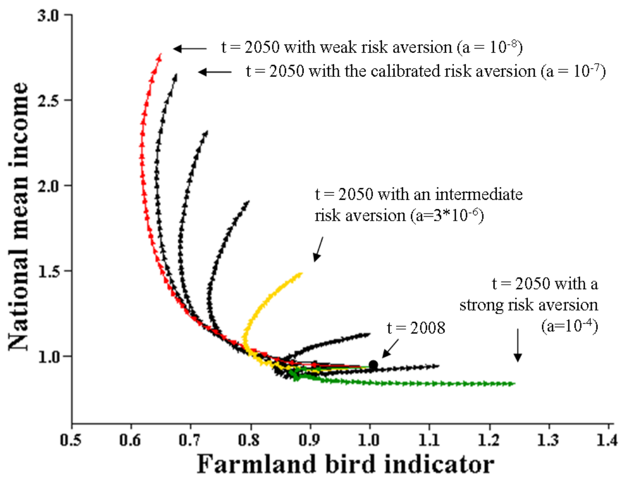
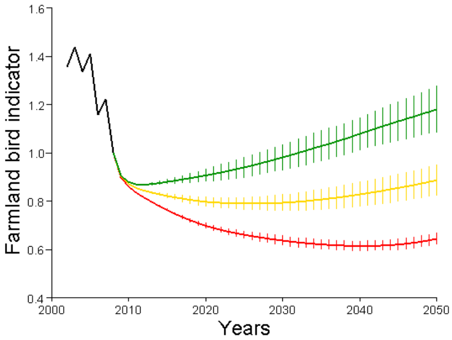
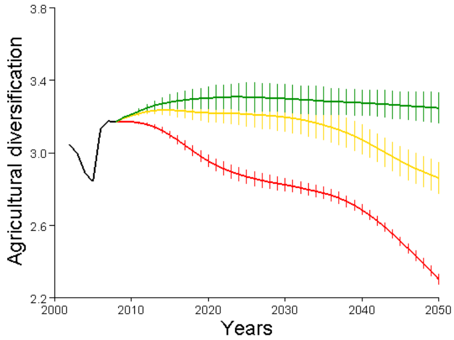


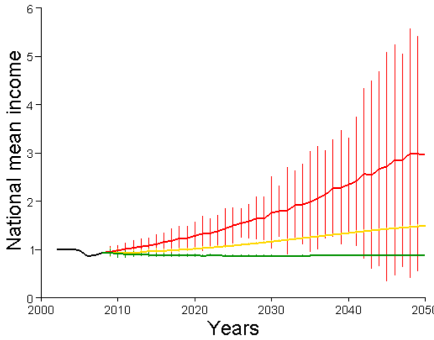
FIGURE III.4 – Mean bio-economic performances and trade-off between $\overline{\mathrm{Inc}}_{\mathrm{nat}}(t)$ and $\mathrm{FBI}_{\mathrm{nat}}(t)$ in a context of economic uncertainty for different levels of risk aversion. All trajectories start at the same black point : t=2008.



(a) Farmland bird indicator $\mathrm{FBI}_{\mathrm{nat}}(t)$



(b) Habitat heterogeneity $\mathrm{Hdiv}_{\mathrm{nat}}(t)$



(c) National mean income $\overline{\mathrm{Inc}}_{\mathrm{nat}}(t)$

FIGURE III.5 – Bio-economic performance and habitat heterogeneity for three contrasted risk aversion levels with 99 % confidence interval (red : weak risk aversion, yellow : intermediate risk aversion, green : strong risk aversion, black : historical data).

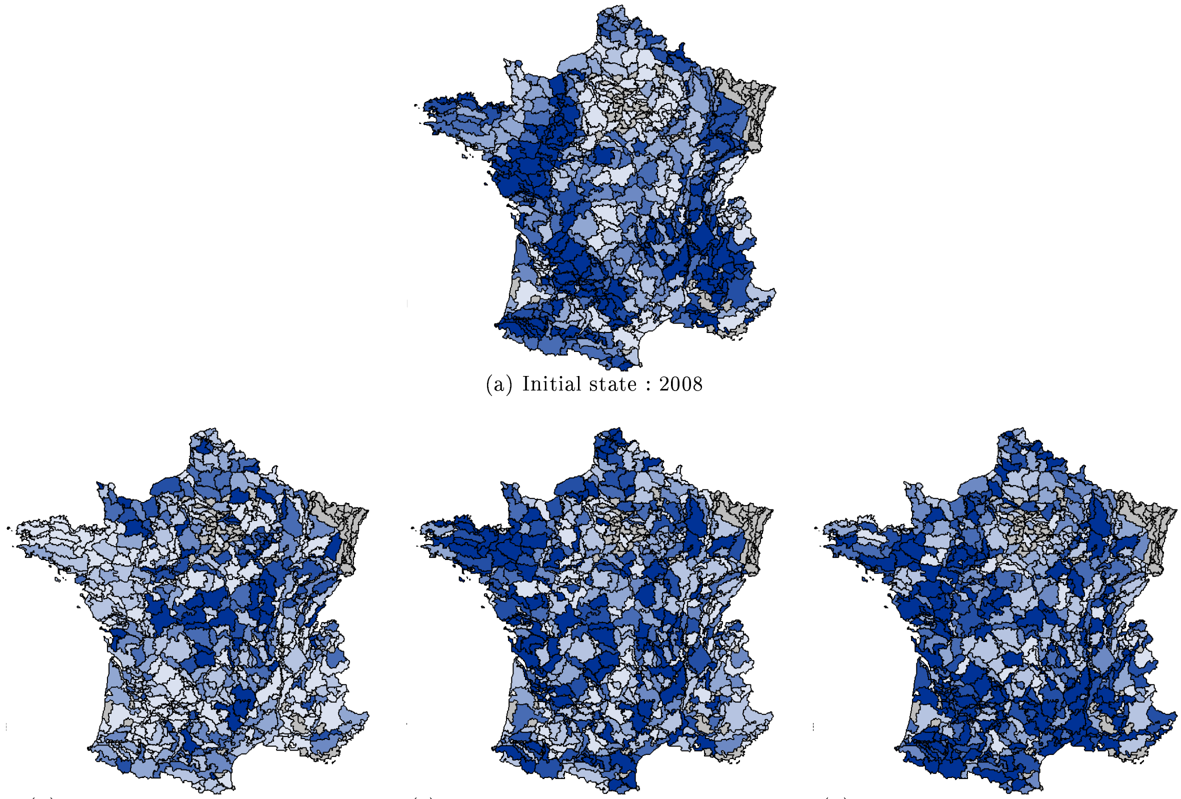
3.2. Performances and volatilities

Figure III.5 compares the national bio-economic performances and agricultural diversification by displaying the means with 99% confident interval for three contrasted levels of risk aversion a. We observe a positive correlation between FBI and agricultural diversification. This is clearly confirmed by a statistical analysis ($R^2 = 57\%$, P-value $\leq 2.2e^{-16}$) : positive ecological performances are associated with the stronger habitat heterogeneity index.

Concerning the dispersion of the outputs, the national income is the most stongly affected indicator. The lowest risk aversion allows for a better growth of national income in the mean but with the largest deviation. By contrast, with the strongest risk aversion, the national income is just stabilized but the volatility vanishes. The intermediate risk aversion leads to a moderate income growth with reduced volatility. On the ecological side, we note that economic risk aversion does not strongly affect the dispersion of farmland bird indicators. The standard deviation is about 3% for all cases. Similarly the habitat heterogeneity index dispersion is not deeply impacted by risk aversion as it ranges from 2% to 3 %.

3.3. Performances at the PRA scale

Figure III.6 displays the habitat diversification indicator $\mathrm{Hdiv}_r(t)$ at the PRA scale in 2008 (t=0) and in 2050, with three contrasted risk strategies. Risk aversion plays qualitatively the same role for the broad majority of regions. With a strong risk aversion, habitat heterogeneity occurs. Conversely, with a weak risk



(a) Initial state : 2008

(b) 2050 with a low risk aversion (c) 2050 with a medium risk aversion (d) 2050 with a strong risk aversion

FIGURE III.6 – Comparison of habitat heterogeneity index at the PRA scale $\mathrm{Hdiv}_r(t)$ in 2008 and in 2050 according to the risk aversion level (dark blue : strong diversity, pale blue : weak diversity, gray : NA).

aversion, we observe a specialization for most regions : the heterogeneity index decreases in comparison with 2008.

Figure III.7 compares the $\mathrm{FBI}_r(t)$ at the PRA scale in 2008 and in 2050 for the three risk aversion levels a. It turns out that the effect of risk aversion on ecological performances at the PRA scale is more reduced than on the agricultural heterogeneity. We observe a global enhancement of regional FBI for the strongest risk aversion levels. Still, contrary to the habitat heterogeneity maps of figure III.6, figure III.7 captures many regional differences : some PRA have a significant FBI improvement and others exhibit a steady FBI.

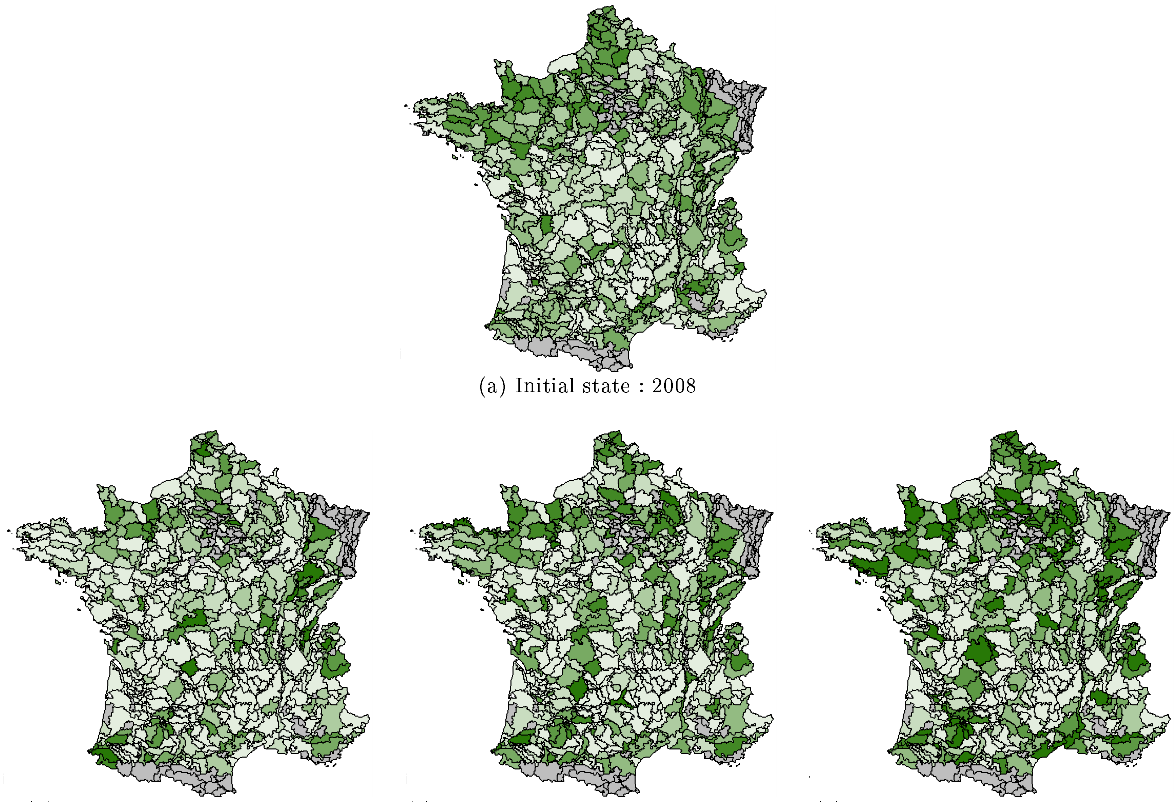
Statistical analysis strongly emphasizes significant correlations between habitat heterogeneity index and FBI for all PRA (P-value $\leq$ 2.2e-16). Nevertheless the quality of the fitness varies among the PRA : R2 varies between 3% and 94%, with a mean at 20%.

Finally, figure III.8 compares the mean income $\mathrm{Inc}_r(t)$ at the PRA scale in 2008 and in 2050 for the three risk aversions. Although risk aversion globally lessens the incomes, many regional discrepancies emerge similarly to bird biodiversity.

4. Discussion

4.1. Spatio-temporal bio-economic models to manage biodiversity

This paper presents an inter-disciplinary approach which is needed (Polasky *et al.* 2005, Perrings *et al.* 2006, Mouysset *et al.* 2011) to operationalize a sustainable management of biodiversity and agriculture. Despite divergences between economic and ecological disciplines (Drechsler *et al.* 2007b), our model couples



(a) Initial state : 2008

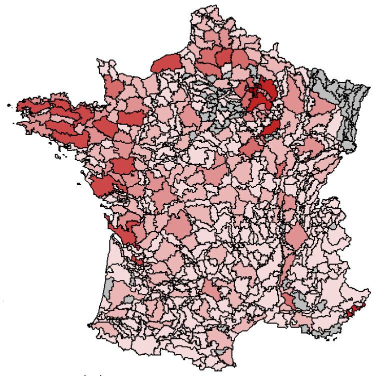
(b) 2050 with a low risk aversion (c) 2050 with a medium risk aversion (d) 2050 with a strong risk aversion

FIGURE III.7 – Comparison of Farmland Bird Index at the PRA scale $\mathrm{FBI}_r(t)$ in 2008 and in 2050 according to the risk aversion level (dark green : strong FBI, pale green : weak FBI, gray : NA).

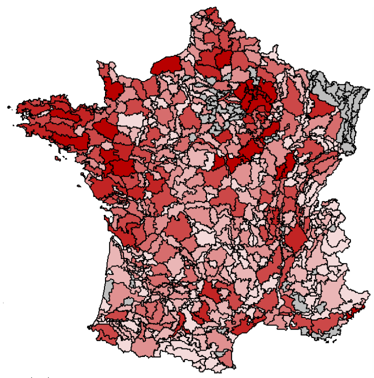
economic and ecological dynamics to analyze bio-economic performances of French agriculture at the national scale. This approach avoids the monetary evaluation of biodiversity which is controversial. The coupling of ecological (bird abundances), land-use and economic (gross margins) data gives strong realism to the modelling. The precision to integrate these data compensates for the simple formalism of the model and makes it possible to obtain robust and informative results. With the account of regional economic and ecological specificities, the model is spatialized at the landscape level, which reinforces its relevance (Polasky *et al.* 2005). Taking account of economic uncertainties through gross margins also reinforces its credibility. The choice to focus on a bird taxa rather than one or two emblematic species makes it possible to obtain more general results regarding biodiversity. Finally, the explicit and mechanistic modelling of the ecological process (with intra-specific competition) and its dynamic perspective (with an adjustment of the carrying capacity function of land uses) lead to a precise representation of impact of land uses on avifauna evolution and transient dynamics. The integration of these uncertain spatio-temporal components, multi-scale data and the multi-criteria viewpoint creates a flexible modelling framework allowing for many developments and refinements.

4.2. Risk aversion to reconcile biodiversity and economic scores

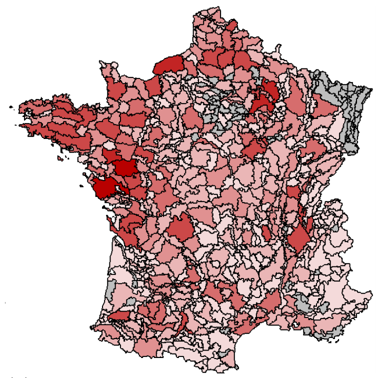
Some studies have already stressed the positive impact of the genetic diversity (ie agrobiodiversity) on the management of the economic risk. In this perspective, Schläpfer *et al.* (2002) for the grasslands, Di Falco & Perrings (2003) and Di Falco *et al.* (2007) for the croplands or Baumgärtner & Quaas (2010) in a theorical approach need to be mentionned. Although keeping a similar viewpoint, the present study suggests a more general mechanism based on the global diversification of the land uses : thanks to a portfolio effect, risk-averse farmers diversify their agricultural activities in order to dampen the uncertainties on expected



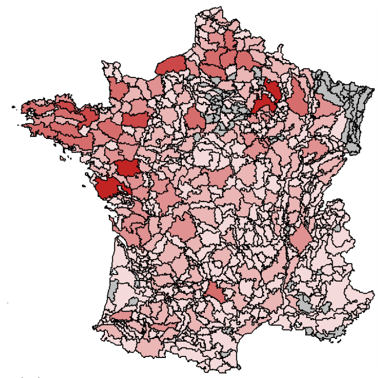
(a) Initial state : 2008



(b) 2050 with a low risk aversion



(c) 2050 with a medium risk aversion



(d) 2050 with a strong risk aversion

FIGURE III.8 – Comparison of income at the PRA scale $\mathrm{Inc}_r(t)$ in 2008 and in 2050 according to the risk aversion level (dark red : strong income, pale red : weak income, gray : NA).

incomes and manage their economic risk. With these insurance effects, the diversification has a positive effect on private goods (income). Moreover this diversification also has a strong positive impact on the production of public goods (biodiversity). Indeed, agricultural diversification creates heterogeneity of habitats and available resources which are essential for birds as stressed by Benton *et al.* (2003). This positive effect of diversification on biodiversity has been experimentally and separately identified for different land uses in Laiolo (2005) for crop landscapes and in Robinson & Sutherland (2002) for the grasslands. A strong risk-averse behavior leads to a simultaneous land-use diversity, which improves the dynamics of bird communities. Land use heterogenity induced by risk-averse farmers thus seems an efficient way of promoting both private and public values.

More globally, farming diversification is positive for the functioning of the agro-ecosystem. According to the insurance hypothesis (Yachi & Loreau 1999), an increasing biodiversity insures ecosystems against declines in their functioning caused by environmental fluctuations. Such an effect is expected because different species can adapt differently to environmental changes (Doak *et al.* 1998, Ives *et al.* 1999). Communities with strong biodiversity are more stable and more productive in the ecological sense than those with poor biological diversity (Caldeira *et al.* 2005). Hence, agricultural habitat diversity acts as a public natural insurance. Moreover, a larger and more diverse community provides the agrosystems with various ecosystem services such as pest control, pollination and decomposition processes (Altieri 1999, Schläpfer *et al.* 1999, Tilman *et al.* 2002) and consequently induces a stronger ecosystem viability. These services should indirectly contribute to the farming production and to its sustainability. In this context, agricultural diversification developped by risk-averse farmers could itself be identified as an ecosystem service yielding both private and public insurance effects for the agro-ecosystem.

4.3. Confrontation to the current biodiversity decline

This study suggests that biodiversity management and conservation is achievable through economic risk aversion. This aversion is sufficient for the farmers to select their land uses in an eco-friendly manner. Consequently the result presented here, based on utility maximization without any ecological awareness or goals, has sound connections with the theoretical statement of Quaas *et al.* (2007) : farmers do not necessarily need to have environmental preferences or to receive monetary benefits from ecosystem services to favor a land-use strategy allowing for a sustainable path for biodiversity. While the economic agents are generally considered risk averse, our calibration leads to a weak risk aversion parameter. This conclusion is consistent with the current decline of farmland birds. Two hypothesis could explain this weak risk aversion. Firstly, the agricultural economics was very protected by the Common Agricultural Policy. This policy has been built to protect the farmers against the market volatilities and the prices fluctuations by distributing economic compensations. In this context, in spite of a risk averse behavior, the farmers are encouraged to specialize their activities (as observed with risk neutral behavior), which has negative impact on the farmland bird. The second hypothesis concerns the financial insurance. To limit the economic risk, the risk averse farmers can opt for either a natural insurance (ie agricultural diversification of land uses and genetic varieties) or financial insurance associated to a specialized activity (Quaas & Baumgärtner 2008). This monetary insurance has shown detrimental effects for ecological performances by promoting more risky production (Horowitz & Lichtenberg 1993, Mahul 2001). In the context, public policies and financial insurance could appear as two interesting leverages to lead farmers to adopt an agricultural diversification by playing on their risk aversion behavior.

4.4. Strong spatial disparities

The previous conclusions at the national scale are intuitive and consistent with the literature. However another important contribution of this study is to provide a multi-scale analysis and complete the average approach with a spatial viewpoint of regional scores. This spatial distribution highlights an interesting effect : trends at the national scale hide many disparities between PRA. Economic risk aversion is sufficient to promote globally biodiversity at the national scale, but it is not enough for every PRA. Hence, some regions which enhance agricultural heterogeneity in a context of strong risk aversion do not always exhibit a strong FBI. This result suggests that other mechanisms govern bird dynamics. In particular, the quality of this diversification could be important : some agricultural systems such as extensive farming are decisive for specific bird populations. In this context, public policies could be developed to favor some agricultural systems and reinforce the diversification mechanisms mentioned above. Strong income differences between PRA are also in favor of public policies to reduce economic disparities. Alavalapati *et al.* (2002), Shi & Gill (2005) and Mouysset *et al.* (2011) have shown the effectiveness of some public policies for both ecological and economic criteria. In a regional perspective, the public policies seem to be essential to manage biodiversity and mitigate economic regional differences.

5. Conclusion

The present work shows how risk aversion directly entails an agricultural diversification which has strong positive impacts for the agro-ecosystem. This diversification plays on economic performances by mitigating income volatility and by promoting numerous and stable ecosystem services which can be used by the farmers. From the ecological point of view, it promotes biodiversity through broader and stabilized farmland bird communities. However, the effect of this risk aversion differs among the PRA. In this context, the public incentives appear essential to drive this diversification in order to reduce both regional ecological and economic disparities. To improve their effectiveness, such policies should account for risk aversion to foster spontaneous diversification.

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Chapitre IV

Different policy scenarios to promote various targets of biodiversity

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Abstract

Biodiversity loss in farmlands is widely documented, and agriculture intensification has been identified as a main driver of this decline. Numerous agri-environmental policies have been implemented to assess the negative impacts of agricultural intensification on biodiversity. However most published studies focus on land-use scenarios, thus neglecting the economic dimension. We develop a bio-economic spatially-explicit modelling across 620 small French agricultural areas, which couples a public decision maker under budgetary constraint, regional economic agents in a context of uncertainty and breeding bird dynamics. Using dynamic models, we analyse the direct impacts of several current economic scenarios of the Common Agriculture Policy on common bird communities through five ecological indicators, all related to breeding populations of birds in farmlands : the Farmland Bird Index (FBI), a Generalist Bird Index (GBI), the Shannon diversity index, a Community Specialization Index (CSI) and a Community Trophic Index (CTI). We consider these indicators to scan various functional traits of bird communities. Trends in the different indicators are significantly contrasted pending on economic policy scenarios. Scenarios promoting intensive crops lead to small but specialized communities with more granivorous species, hence a low trophic level for the community. By contrast, promoting extensive grasslands increases the population size, enhances high trophic level but decreases community specialisation. Evaluation of agricultural policies should not rely on a single indicator per taxonomic group. In the context of potential reversal of current bird declines, bio-economic modelling, involving farmland incomes, is proposed as a relevant support for decision making about sustainable agri-environmental policies. Promoting extensive grasslands is essential for the sustainable management of bird communities and agriculture. We however reveal more complex economic effects and synergies between public incentives which appear to give interesting leverage for enhancing the bio-economic effectiveness of agricultural policies.

Keywords : Breeding Bird Survey, Farmland Bird Index, Indicators, Biodiversity, Agriculture, Bio-economic modelling

1. Introduction

Numerous monitoring programs have reported farmland biodiversity losses across continents in recent decades with special focus on bird declines (Butchart *et al.*, 2010, Donald *et al.*, 2001, Sotherton & Self, 2000, Flowerdew & Kirkwood, 1997). The impacts of agricultural intensification on biodiversity are particularly strong on bird populations Jiguet (2009), Chamberlain *et al.* (2000), Krebs *et al.* (1999). Global changes in European agriculture, including intensification and land abandonment, have significantly modified farmland bird communities (Devictor *et al.*, 2008, Donald *et al.*, 2006; 2001). Such erosion is mainly induced by a combination of habitat loss and fragmentation and of degraded habitat quality altering the reproductive success and/or survival of individuals (Benton *et al.*, 2003). In this context, the need to reconcile agricultural production and biodiversity is of particular concern (Jackson *et al.*, 2005). In the European Union, since the 1990s, several public policies have been dedicated to limiting the negative impacts and externalities of agriculture on biodiversity. Typically, agri-environmental schemes have been implemented so that farmers receive financial support for adopting environment-friendly agricultural practices (Kleijn & van Zuijlen, 2004). There is an extensive and increasing literature concerning agri-environmental schemes and policies for multi-functional agriculture (Taylor & Morecroft, 2009, Albrecht *et al.*, 2007, Batary *et al.*, 2007, Kleijn & van Zuijlen, 2004, Münier *et al.*, 2004). Still, fifteen years after the implementation of such instruments, whether providing habitat quality conflicts with management for agricultural production remains controversial (Butler *et al.*, 2007, Kleijn *et al.*, 2006, Vickery *et al.*, 2004). One limit of the evaluations performed is the focus on land-use scenarios, ignoring the economic behaviour of farmers facing public incentives (Scholefield *et al.*, 2009). As agricultural policies are mainly proposed in economic terms, the introduction of economic dimensions appears as essential to define sustainable management of both agriculture and biodiversity (Mouysset *et al.*, 2011). As pointed out by Hughey *et al.* (2003) and Perrings *et al.* (2006), there is a need for approaches integrating economic criteria in conservation problems.

In order to analyse potential trends of different agricultural policies on biodiversity, the present paper developes a bio-economic model which articulates a national decision maker, regional farmers and biodiversity dynamics for France. To give strong realism to the scenarios, we integrate a national budgetary constraint and calibrate the model with ecological, farming land-use and economic databases. To characterize biodiversity, we focus on breeding birds which are largely recognized as a representative biodiversity compartment highly sensitive to agricultural practices (Gregory *et al.*, 2004), although the metrics and the characterization of biodiversity remain an open debate (Le Roux, 2008, MEA, 2005). Focus on breeding birds is further justified because (i) birds lie at a high level in the trophic food chains and thus capture the variations in the chains ; (ii) birds provide ecological services, such as the regulation of invertebrate and rodent populations and pest control (Sekercioglu *et al.*, 2004) ; (iii) their close vicinity to humans makes them a simple and comprehensive biodiversity index for a large audience of citizens (Ormerod & Watkinson, 2000).

In direct line with these considerations, the European Union has adopted the Farmland Bird Index FBI (Gregory *et al.*, 2004) as an indicator of structural changes in biodiversity (Balmford *et al.*, 2003). However, beyond changes in bird abundances, community traits and functions are only vaguely summarized by a single indicator (Barbault & Chevassus-Au-Louis, 2004), while various indicators are available in the literature to describe and analyse bird communities. State indicators such as the Shannon-Wiener diversity index or the EU Farmland Bird Index (FBI) are widely used to quantify biological diversity and associated trends in farmlands. Doxa *et al.* (2010) reports the relevance of the FBI to reflect the response of farmland biodiversity to agriculture intensification. Other trait- or function-based indicators referring to community specialization (Julliard *et al.*, 2006) or trophic level (Pauly *et al.*, 1998) explore functional characteristics of the communities. Bird communities are more specialized in unaltered and non-fragmented habitats, including farmlands (Devictor *et al.*, 2008), while higher trophic levels should testify to unaltered food chains and therefore communities ensuring more ecological functions. In this perspective, we used different ecological indicators to analyse the performance of our bio-economic modelling.

2. Material and methods

We developed a spatialized bio-economic model over the 620 small French metropolitan agricultural areas (PRA, "petite région agricole"). Their consistency at both agro-ecological and economic levels makes them particularly well-suited for our modelling and analysis. As in Mouysset *et al.* (2011), three compartments are linked : national public decision maker, regional economic agent and bird community (fig. IV.1) .

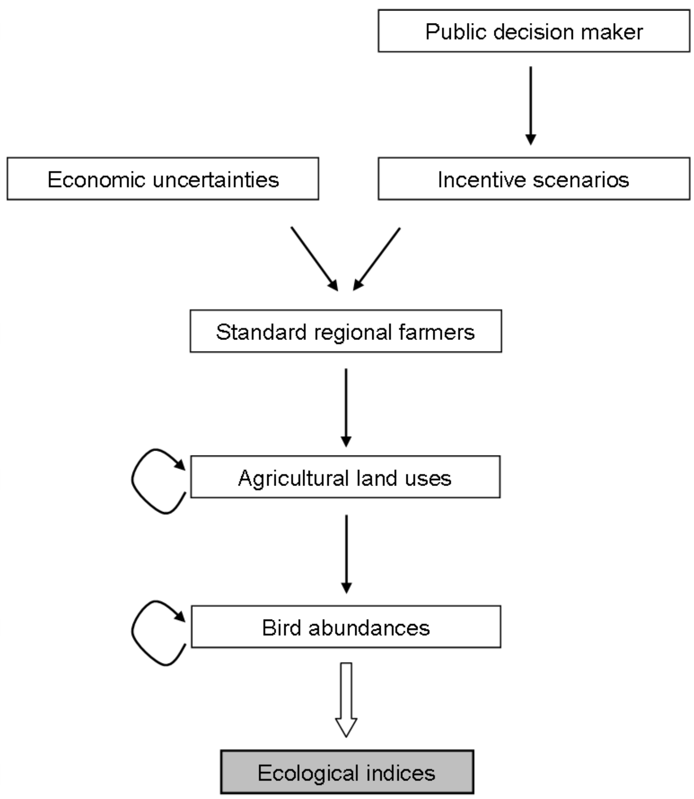


FIGURE IV.1 – Model coupling : Farmers adjust their agricultural systems pending on economic uncertainty and incentives. These choices affect bird community dynamics.

2.1. The ecological model

To assess ecological performance, we focus on common farmland birds (Ormerod & Watkinson, 2000). Bird populations are driven by Beverton-Holt dynamics (Beverton & Holt, 1957) which capture intra-specific competition through the carrying capacity parameter :

$$N_{s,r}(t+1) = N_{s,r}(t).\frac{1+R_{s,r}}{1+\frac{N_{s,r}(t)}{M_{s,r}(t)}} \tag{IV.1}$$

where $N_{s,k}(t)$ stands for the bird abundance of species s in PRA r at year t. The $R_{s,r}$ coefficient corresponds to the intrinsic growth rate specific to a given species s. The product $M_{s,k}(t) * R_{s,r}$ represents the carrying capacity of the habitat r and the value $M_{s,k}(t)$ captures the ability of the habitat to host the species.

This habitat parameter depends on the agricultural land-uses chosen by the farmers (eq. IV.2) as follows :

$$\mathrm{M_{s,r}(t)} = \beta_{\mathrm{s,r}} + \sum_{\mathrm{k}} \alpha_{\mathrm{s,r,k}}.\mathrm{A_{r,k}(t)} \tag{IV.2}$$

where surfaces $\mathrm{A_{r,k}(t)}$ including crop or grasslands are detailed in Table IV.1. Consequently, the α and β coefficients, specific to each species, inform on how such species s respond to various agricultural systems k in a PRA r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a PRA r. The ecological model is calibrated for each PRA in order to integrate regional agro-environmental features.

2.2. The micro-economic model

We considered 620 PRA of metropolitan France, so we have 620 regional economic standard agents with 620 representative farms. A representative farm does not really exist and represents an 'average' farm for the PRA. We compute these historical characteristics by averaging those of all the real farms of the PRA. As PRA has an agricultural and ecological homogeneity, all real farms in a PRA have similar characteristics and joining them in a 'mean' farm makes sense. The regional economic standard agents select their agricultural land-uses in order to maximize their utility under technical constraints. These choices, made in an uncertainty context, depend on expected gross margins, financial incentives specified by the public (national) decision maker and current land use areas. This approach refers to stochastic maximisation under constraints, usual in bio-economic modelling (Lien, 2002).

Farmer's income in PRA r at year t denoted by $\mathrm{Income_r(t)}$ relies on the gross margin $\mathrm{gm_{r,k}(t)}$ for the year t, current agricultural activities $\mathrm{A_{r,k}(t)}$ and incentives $\tau_{\mathrm{k}}(\mathrm{t})$ which take the form of taxes ($\tau_{\mathrm{k}} < 0$) or subsidies ($\tau_{\mathrm{k}} > 0$) (eq. IV.3). Gross margins $\mathrm{gm_{r,k}(t)}$ are taken to be uncertain. The variability on gross margins includes both market, production and climate uncertainties. A Gaussian distribution parametrized with the mean and the covariance matrix of the historical data is chosen to capture such uncertainties.

$$\mathrm{Income_r(t)} = \sum_{\mathrm{k}} \mathrm{gm_{r,k}(t).A_{r,k}(t).(1 + \tau_k(t))} \tag{IV.3}$$

For each year t, the regional standard agents chose their agricultural systems $\mathrm{A_{r,k}(t)}$ in order to maximise their utility in an uncertain context (eq. IV.4). This utility corresponds to the expected Income, which depends on the expected gross margins $\overline{\mathrm{gm}}_{\mathrm{r,k}}(\mathrm{t})$ computed with the 7 historical gross margins (2002-2008) (eq. IV.5).

$$\max_{\mathrm{A_{r,k}}} \ \mathrm{Utility_r(t)} = \max_{\mathrm{A_{r,k}}} \ \sum_{k} \overline{\mathrm{gm}}_{\mathrm{r,k}}\mathrm{(t).A_{r,k}(t).(1 + \tau_k(t))} \tag{IV.4}$$

$$\overline{\mathrm{gm}}_{\mathrm{r,k}} = \frac{1}{7} \sum_{\mathrm{t=2008}}^{\mathrm{t=2002}} \mathrm{gm_{r,k}(t)} \tag{IV.5}$$

The agricultural choices are limited at every time t by capital and rigidity constraints :

$$|\mathrm{A_{r,k}(t)} - \mathrm{A_{r,k}(t-1)}| \leq \varepsilon.\mathrm{A_{r,k}(t-1)} \tag{IV.6}$$

$$\sum_{k} \mathrm{A_{r,k}(t)} = \mathrm{A_r} \tag{IV.7}$$

The rigidity constraint (eq. IV.6) limits the area that the farmer can change at each time for each agricultural system. It captures change costs and limits change speed quantified by the parameter ε. The constraint (eq. IV.7) ensures that the total agricultural surface A_r in a PRA is kept fixed.

2.3. The public policy model

The national decision maker selects economic incentives scenarios $\tau_k(t)$. It defines taxes and/or subsidies for different agricultural activities k according to specific objectives and a budgetary constraint. For all scenarios, incentives $\tau_k(t)$ are assumed to be linearly decreasing with time, from 2009 to achieve 0 in 2050 (eq. IV.8). Such decreasing incentives capture the current trend of Common Agricultural Policy (CAP) perspectives.

$$\tau_k(t) = \tau_k(2009)(1 - \frac{t - 2009}{2050 - 2009}) \tag{IV.8}$$

We develop five scenarios to test different perspectives of CAP :

- A Laissez-Faire (LF) scenario, with no further tax or subsidy. It prolongs the current trend and leads to defining marginal effects of the other policies compared to the current evolution.
- A Crop (CR) scenario, which promotes Cereal-Oleaginous-Proteaginous crops (COP). We here test a pattern of intensification typically associated with the development of bioenergy.
- A Grassland (GL) scenario with subsidies to non-intensive grasslands. It corresponds to the opposite pattern of the intensification scenario and promotes non-intensive agricultural systems with low intensification, small fields and many linear elements.
- A Double Subsidies (DS) scenario with subsidies to both COP and non-intensive grasslands. This scenario is the closest to the current situation.
- A High Quality Environment (HQE) scenario with taxes on COP and subsidies to non-intensive grasslands. As with the Grassland scenario, the objective of this scenario is to favor extensive agriculture but with a more intensive policy.

Two scenarios make it possible to study potential synergies (HQE scenario) or antagonisms (DS scenario) between incentives.

To define the initial level of incentives $\tau_k(2009)$ for the agricultural systems underlying the scenarios, it is supposed that the public stakeholder complies with a budgetary constraint. We define the national budget as the sum of public incentives distributed over the 620 PRA (eq. IV.9).

$$\mathrm{Budget}_{Nat}\ (t) = \sum_r \sum_k gm_{r,k}(t)\ .\ A_{r,k}(t)\ .\ \tau_k(t) \tag{IV.9}$$

Incentives $\tau_k(2009)$ are such that the national budget required by the decision maker has to remain lower than the current budget $\mathrm{Budget}_{Nat}(2008)$ at each year t (eq. IV.10).

$$\mathrm{Budget}_{Nat}(t)\ \leq\ \mathrm{Budget}_{Nat}(2008) \tag{IV.10}$$

2.4. Data

To calibrate this model, several databases have been articulated : the Farm Accountancy Data Network[1] (FADN) and the Observatory of Rural Development[2] (ODR). Fourteen classes of agricultural systems, named OTEA (Orientation Technico-Economique) and displayed in table IV.1 are distinguished in this

1. http ://ec.europa.eu/agriculture/rica/
2. https ://esrcarto.supagro.inra.fr/intranet/

	Initial states	Statu quo scenario		Trend
	2008	2030	2050	
(1) Cereal, oleaginous, proteaginous	25.8%	13.3%	4.37%	↘
(2) Variegated crops	0.500%	1.88%	4.19%	↗
(3) Intensive bovine livestock breeding	17.2%	22.7%	9.79%	↘
(4) Medium bovine livestock breeding	5.75%	6.02%	3.00%	↘
(5) Extensive bovine livestock breeding	15.5%	5.13%	1.53%	↘
(6) Mixed crop-livestock farming with herbivorous direction	0.860%	3.56%	4.61%	↗
(7) Other herbivorous livestock breeding	5.07%	4.03%	3.01%	→
(8) Mixed crop-livestock farming with granivorous direction	0.01%	0.04%	0.22%	→
(9) Mixed crop-livestock farming with other direction	20.4%	11.4%	3.08%	↘
(10) Granivorous livestock breeding	2.46%	15.0%	33.0%	↗
(11) Permanent farming	1.05%	1.81%	1.36%	→
(12) Flower farming	0.971%	6.41%	21.8%	↗
(13) Viticulture	4.19%	8.10%	8.11%	↗
(14) Others associations	0.005%	0.007%	0.01%	→

Table IV.1 – Proportion of the French agricultural area dedicated to the 14 agricultural systems named OTEA in the initial state 2008 and under the Laissez-Faire scenario in 2030 and 2050.

manner. Each PRA is a specific combination of these OTEA. The surfaces dedicated to each of the 14 OTEA and the gross margins associated, for the years 2001 to 2008 are available on the ODR website. Gross margin is an economic indicator broadly used in bio-economic modelling (Pacini *et al.*, 2004, ten Berge *et al.*, 2000) and agricultural economics (Lien, 2002). The budgetary constraint was calibrated with the current French CAP budget.

For the ecological part of the model, we used data provided by the national Breeding Bird Survey (BBS) implemented in France since 2001. Among the common breeding species monitored by this scheme, we focused on those 34 species classified as farmland specialist and habitat generalist species, according to their habitat requirements (Julliard *et al.*, 2006). The list of these species is presented in table IV.2. Abundance values for each species were available for the period 2002-2008 for 1747 squares (a square is 2*2km in size) (as detailed in Jiguet *et al.* (2010). For each species, we further performed a spatial interpolation of these abundance data to obtain relative abundance values for each possible square in the country (e.g. 136 000 squares) using kriging models based on spatial autocorrelation and an exponential function. We then averaged the abundance values at the PRA scale to calibrate the ecological model. Figure IV.2 illustrates the results of this calibration with one species, the Wood Lark *Lullula arborea*. Comparing the historical data with the model-generated data, we note that the model tends to smooth the variations of the observed data.

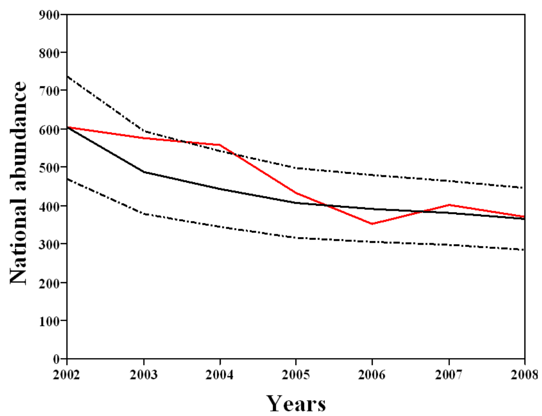


Figure IV.2 – Comparison between historical (red) and estimated (black) national abundances with the least square standard errors of calibration (dashed lines) for one of the species considered, the Wood Lark *Lullula arborea*.

Species	Habitat specialization	Specialization index	Trophic index
Buzzard *Buteo buteo*	Farmland	0.49	2.90
Cirl Bunting *Emberiza cirlus*	Farmland	0.59	1.30
Corn Bunting *Emberiza calandra*	Farmland	1.46	1.28
Grey Partridge *Perdix perdix*	Farmland	2.11	1.10
Hoopoe *Upupa epops*	Farmland	0.61	2.00
Kestrel *Falco tinnunculus*	Farmland	0.68	2.85
Lapwing *Vanellus vanellus*	Farmland	2.23	1.90
Linnet *Carduelis cannabina*	Farmland	0.70	1.05
Meadow Pipit *Anthus pratensis*	Farmland	1.37	1.75
Quail *Coturnix coturnix*	Farmland	1.52	1.22
Red-backed Shrike *Lanius collurio*	Farmland	1.14	2.15
Red-legged Partridge *Alectoris rufa*	Farmland	1.10	1.10
Rook *Corvus frugilegus*	Farmland	0.84	1.63
Skylark *Alauda arvensis*	Farmland	1,16	1.25
Stonechat *Saxicola torquatus*	Farmland	0.78	2.00
Whinchat *Saxicola rubetra*	Farmland	1.46	2.00
Whitethroat *Sylvia communis*	Farmland	0.65	1.60
Wood Lark *Lullula arborea*	Farmland	0.90	1.50
Yellowhammer *Emberiza citrinella*	Farmland	0.71	1.30
Yellow Wagtail *Motacillajlava*	Farmland	2.09	2.00
Blackbird *Turdus merula*	Generalist	0.23	1.60
Blackcap *Sylvia atricapilla*	Generalist	0.32	1.60
Blue Tit *Cyanistes caeruleus*	Generalist	0.35	1.80
Carrion Crow *Corvus corone*	Generalist	0.28	1.51
Chaffinch *Fringilla coelebs*	Generalist	0.27	1.10
Cuckoo *Cuculus canorus*	Generalist	0.43	2.00
Dunnock *Prunella modularis*	Generalist	0.50	1.50
Golden Oriole *Oriolus oriolus*	Generalist	0.47	1.95
Great Tit *Parus major*	Generalist	0.29	1.85
Green Woodpecker *Picus viridis*	Generalist	0.38	2.00
Jay *Garrulus glandarius*	Generalist	0.44	1.72
Melodious Warbler *Hippolais polyglotta*	Generalist	0.70	1.95
Nightingale *Luscinia megarhynchos*	Generalist	0.47	2.00
Wood Pigeon *Columba palumbus*	Generalist	0.30	1.01

Table IV.2 – The 34 bird species considered in this study, with reference to their habitat specialisation, and values of their species specialisation index and species trophic index.

We use this bio-economic model to assess the impact of public economic policies on bird communities. The selected timeframe runs up from 2009 to 2050, i.e a 42-year forecast. Adopting a shorter timeframe could consequently hide interesting medium-term effects due to the inertia of the model. Relevant $\tau_k(2009)$ for each scenario are described in table IV.3. To clarify the impacts of policy scenarios, table IV.1 presents the allocation of the agriculturally- utilized area among the 14 OTEA at the national scale before the projection (initial state : 2008) and under the Laissez-Faire scenario. Table IV.4 illustrates the variations in the areas allocated to the various OTEA under the 4 policy scenario compared to the Laissez-Faire scenario. To study the community obtained after the different public policies, we develop various and complementary indicators to capture state, functional and pressure responses as described in the Biodiversity Indicators section.

2.5. Biodiversity indicators

2.5.1. Abundance indices

To analyse predicted trends in population abundances, we first focused on the national Farmland Bird Index (FBI) to study the structural changes in biodiversity (Balmford *et al.*, 2003). Previous analyses have shown the relevance of the national FBI to reflect the response of farmland biodiversity to agriculture intensification

	ST scenario	CR scenario	GL scenario	DS scenario	HQE scenario
(1) Cereal, oleaginous, proteaginous	0%	+65%	–	+30%	−30%
(4) Medium bovine livestock breeding	0%	–	+55%	+50%	+60%
(5) Extensive bovine livestock breeding	0%	–	+55%	+50%	+60%
(6) Mixed crop-livestock farming with herbivorous direction	0%	–	+55%	+50%	+60%
(7) Other herbivorous livestock breeding	0%	–	+55%	+50%	+60%

Table IV.3 – Initial incentives $\tau_k(2009)$ for the 4 policy scenarios.

	CR scenario			GL scenario			DS scenario			HQE scenario		
	2030	2050		2030	2050		2030	2050		2030	2050	
(1) Cereal, oleaginous, proteaginous	**+20 %**	+9.3%	↗	−3.44%	−2%	↗	**+6.3 %**	+2.09%	↗	**−7.55%**	−3.19%	↘
(2) Variegated crops	−0.34%	−0.5%		−0.1%	−0.22%		−0.16%	−0.31%		−0.01%	−0.21%	
(3) Intensive bovine livestock breeding	−8.7%	−3.63%		−13.43%	−5.43%		−15.9%	−6.2%		−13.27%	−5.47%	
(4) Medium bovine livestock breeding	−1.8%	−0.49%		**+11.9 %**	+1.5%	↗	**+10.58 %**	+5.02%	↗	**+12.28 %**	+6.39%	↗
(5) Extensive bovine livestock breeding	−0.97%	−0.44%		**+8.47 %**	+2.05%	↗	**+6.37 %**	+1.43%	↗	**+8.97 %**	+2.22%	↗
(6) Mixed crop-livestock farming with herbivorous direction	−0.24%	−0.24%		+0.93%	+1.42%		+0.79%	+1.11%		+0.97%	+1.55%	
(7) Other herbivorous livestock breeding	−0.84%	−0.79%		+2.32%	+5.89%		+1.87%	+1.18%		+2.6%	+1.67%	
(8) Mixed crop-livestock farming with granivorous direction	+0%	+0%		+0%	+0%		+0%	+0%		+0%	+0.01%	
(9) Mixed crop-livestock farming with other direction	−6.19%	−1.51%		−5.16%	−1.31%		−8.14%	−2.21%		−1.26%	−1.34%	
(10) Granivorous livestock breeding	−0.3%	−0.3%		−0.8%	−0.9%		−0.7%	−0.9%		−0.8%	−1%	
(11) Permanent farming	+0%	−0.01%		−0.05%	−0.02%		−0.04%	−0.02%		−0.07%	−0.02%	
(12) Flower farming	−0.31%	−0.8%		−0.31%	−0.6%		−0.45%	−0.7%		−0.33%	−0.7%	
(13) Viticulture	−0.81%	−0.19%		−0.33%	−0.13%		−0.39%	−0.17%		−0.2%	−0.13%	
(14) Others associations	−0.001%	+0%		+0%	+0%		+0%	+0.005%		+0%	+0%	

TABLE IV.4 – Variations of the proportions of the French agricultural area dedicated to the 14 agricultural systems named OTEA with the 4 policy scenario in 2030 and 2050.

(Doxa *et al.*, 2010). This indicator reports the variation in the abundances of 20 habitat specialists distinctive of farmland habitats. A similar indicator is proposed here for 14 habitat generalists, namely a Generalist Bird Index (GBI), similarly reporting the variations of abundances of these species (Julliard *et al.*, 2004), with the aim of comparing the response of the two groups (tab. IV.2). These multiple-species indicators are computed as the geometric mean of the yearly indices of the species considered in the group. In theses aggregated indices, the abundances variation of each species is taken into account similarly, independently from the abundance value. We first estimated a national population index for each species from the abundances values of all PRA r (eq. IV.11), then we calculated the aggregated indicators $\mathrm{FBI_{Nat}}$ and $\mathrm{GBI_{Nat}}$ (eq. IV.12 and IV.13).

$$\mathrm{N_{s,Nat}(t)} = \sum_{\mathrm{r}} \mathrm{N_{s,r}(t)} \tag{IV.11}$$

$$\mathrm{FBI_{Nat}(t)} = \prod_{\mathrm{s} \,\in\, \mathrm{Specialist}} \left(\frac{\mathrm{N_{s,Nat}(t)}}{\mathrm{N_{s,Nat}(2008)}} \right)^{1/20} \tag{IV.12}$$

$$\mathrm{GBI_{Nat}(t)} = \prod_{\mathrm{s} \,\in\, \mathrm{Generalist}} \left(\frac{\mathrm{N_{s,Nat}(t)}}{\mathrm{N_{s,Nat}(2008)}} \right)^{1/14} \tag{IV.13}$$

2.5.2. Shannon index

We computed the Shannon-Wiener diversity index for the whole community of the 34 species (including both habitat generalists and farmland specialists). This index informs about the repartition of individual birds within the different species in the community (eq. IV.14 and IV.15). A larger value of this index means a more balanced repartition of individuals between species. The National Shannon Index is the arithmetic mean of the regional Shannon Indices (eq. IV.16).

$$\mathrm{N_{tot,r}(t)} = \sum_{\mathrm{s}} \mathrm{N_{s,r}(t)} \tag{IV.14}$$

$$\text{Shannon Index}_{\mathrm{r}}(\mathrm{t}) = -\sum_{\mathrm{s}} \frac{\mathrm{N_{s,r}(t)}}{\mathrm{N_{tot,r}(t)}} \,.\, \log\left(\frac{\mathrm{N_{s,r}(t)}}{\mathrm{N_{tot,r}(t)}} \right) \tag{IV.15}$$

$$\text{Shannon Index}_{\mathrm{Nat}}(\mathrm{t}) = \frac{1}{620} . \sum_{\mathrm{r}} \text{Shannon Index}_{\mathrm{r}}(\mathrm{t}) \tag{IV.16}$$

2.5.3. Community specialization index

We considered an indicator of pressure : the Community Specialisation Index (CSI). The objective of this indicator is to interpret the response of the composition of local bird communities to agricultural pressures. A habitat specialisation species index (SSI) has been computed for each species, reporting the coefficient of variation of the abundance of a species across 18 habitat categories (see Julliard *et al.* (2006) ; Tab. IV.2 [1]). For each square, the local CSI_r is then calculated as the arithmetic mean of the species specialisation index weighted by the abundances (eq. IV.14 and IV.17). This index measures the average degree of habitat specialisation among the individuals of the community. It leads to discriminating the ordinary community of generalist species, which are more resilient to perturbation, from the specialized communities with more specialist species, which are especially sensitive to global change (Julliard *et al.*, 2006). National CSI_{Nat} is the arithmetic mean of the 620 regional CSI_r (eq. IV.18).

$$CSI_r(t) = \sum_s \frac{N_{s,r}(t)}{N_{tot,r}(t)} \, . \, SSI_s \tag{IV.17}$$

$$CSI_{Nat}(t) = \frac{1}{620} . \sum_r CSI_r(t) \tag{IV.18}$$

2.5.4. Community trophic index

To consider a functional dimension of bird communities, we supplemented the previous indices with a Community Trophic Index (CTI) (Pauly *et al.*, 1998). The position of each species within the trophic chain was computed from information on specific diets as available in BWPi (2006), defining the proportion of each species diet made of vegetables, invertebrates and vertebrates, then estimating an average species trophic index by computing a weighted mean of the 3 diet proportions (weighting coefficients being 1 for vegetables, 2 for invertebrates, 3 for vertebrates). Specific trophic indices (STI) are described table IV.2 for the 34 studied species. The CTI_r reports on the average trophic level of the community. It is computed as the weighted arithmetic mean of the exponential of the species trophic level balanced by the abundances (eq. IV.14 and IV.19). An exponential function is used to better contrast communities with or without bird individuals of the higher trophic levels. This indicator discriminates the communities with more granivorous species (e.g. low trophic level) against the communities with more insectivorous and carnivorous species (e.g. high trophic level). National CTI_{Nat} is the arithmetic mean of the 620 regional CTI_r (eq. IV.20).

$$CTI_r(t) = \sum_s \frac{N_{s,r}(t)}{N_{tot,r}(t)} \, . \, \exp(STI_s) \tag{IV.19}$$

$$CTI_{Nat}(t) = \frac{1}{620} . \sum_r CTI_r(t) \tag{IV.20}$$

2.5.5. Rate index

To clarify CSI and CTI analyses, we complement these indicators with species indices. They measure the proportion of one species among the community (eq. IV.14 and IV.21). We can compare the proportions of different species in the bird population according to their functional characteristics. The National species index is the arithmetic mean of the 620 regional Rate $Index_{s,r}$ (eq. IV.22).

1. Although the Species Specialisation Index for Buzzard is smaller than for some of generalist species, we kept it in Farmland to conserve the European classification.

$$\text{Rate Index}_{s,r}(t) = \frac{N_{s,r}(t)}{N_{tot,r}(t)} \tag{IV.21}$$

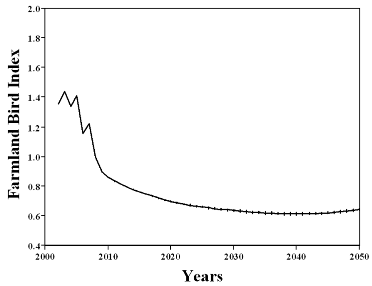
$$\text{Rate Index}_{s,Nat}(t) = \frac{1}{620} \cdot \sum_{r} \text{Species Index}_{s,r}(t) \tag{IV.22}$$

3. Results

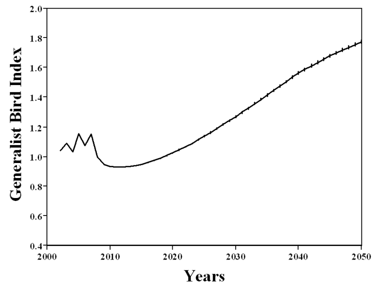
As the farmer choices occur in uncertainty contexts, we ran 100 simulations for each scenario with different Gaussian gross margins $gm_{r,k}(t)$ to estimate the means of community outcomes and their 95% confident interval. Species indices are illustrated for one run. We present the outcomes of the Laissez-Faire (LF) scenario (fig. IV.3) then the outcomes of the Crop (CR), Grassland (GL), Double Subsidies (DS) and High Quality Environmental (HQE) scenarios normalized by the outcomes of the Laissez-Faire scenario (fig. IV.4). We can thus analyse the marginal benefits of the different economic policies compared to current policy trends. To clarify the ecological effects of the various scenarios, table IV.1 describes the land uses allocation at the initial states and under the Laissez-Faire scenario. Table IV.4 illustrates the changes in land uses compared to the Laissez-Faire for the four policy scenarios and figure IV.5 presents the proportion of the utilized agricultural area dedicated to the non-intensive activities (OTEA 4, 5, 6, 7) under the five scenarios.

3.1. Global outcomes

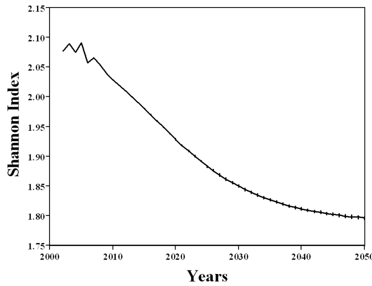
The fig. IV.3 shows that with the LF scenario the quantities of farmland bird species decrease while those of generalist species increase. This leads to a decreasing specialization level and a loss of diversity for the community. The community trophic index follows the historical trend. Fig. IV.4 illustrates non linear and non monotonous trajectories among the policy scenarios. Indicators have contrasted trends in function of the



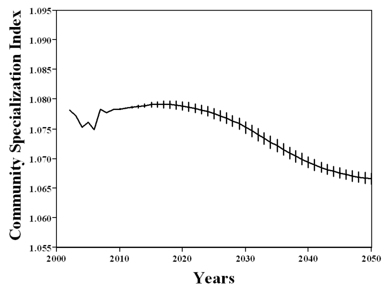
(a) Farmland Bird Index $FBI_{Nat}(t)$



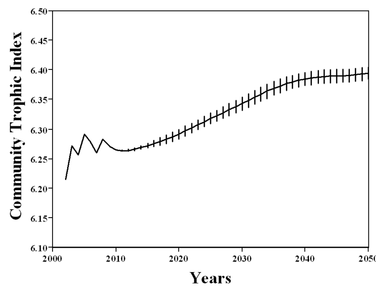
(b) Generalist Bird Index $GBI_{Nat}(t)$



(c) Shannon index $Shannon_{Nat}(t)$

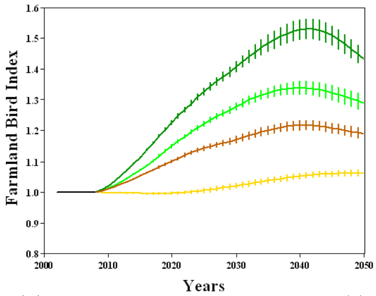


(d) Community Specialization Index $CSI_{Nat}(t)$

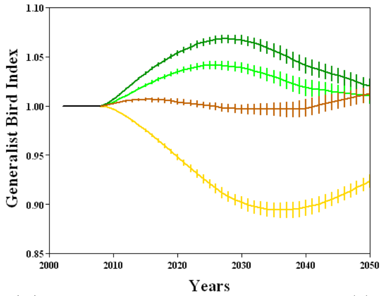


(e) Community Trophic Index $CTI_{Nat}(t)$

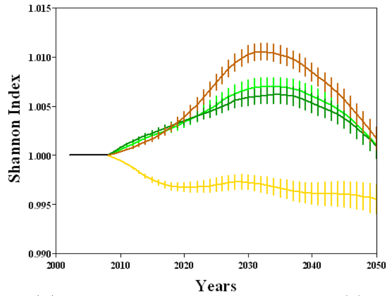
FIGURE IV.3 – Ecological indicators evolutions (mean outcomes and 95% confident interval) with the Laissez-Faire scenario.



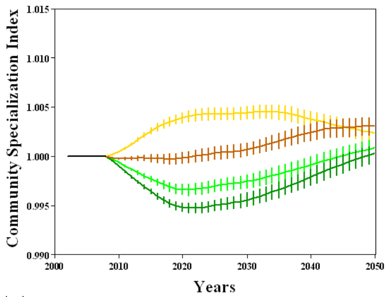
(a) Farmland Bird Index $FBI_{Nat}(t)$



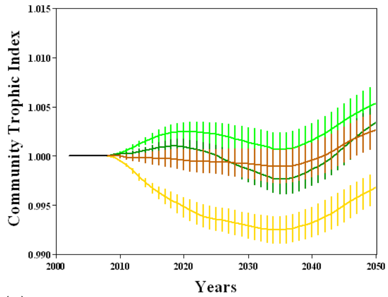
(b) Generalist Bird Index $GBI_{Nat}(t)$



(c) Shannon index $Shannon_{Nat}(t)$



(d) Community Specialization Index $CSI_{Nat}(t)$



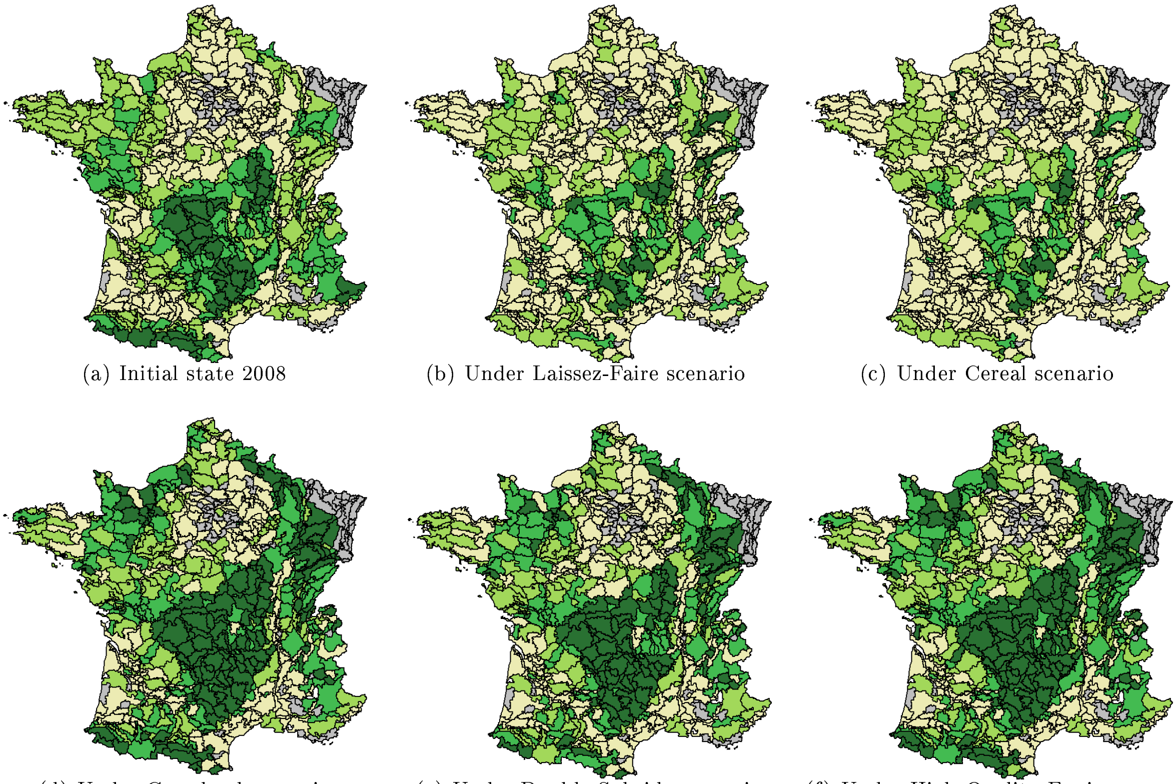
(e) Community Trophic Index $CTI_{Nat}(t)$

FIGURE IV.4 – Ecological indicators evolutions (mean outcomes and 95% confident interval) with the Crop (yellow), Grassland (light-green), High Quality Environmental (dark-green), and Double Subsidy (brown) scenarios, normalized by the Laissez-Faire evolutions.

scenarios for 30-40 years, then they start to return to a baseline value (of 1) around 2030-2040. Table IV.4 illustrates that changes in agricultural systems within the four policy scenarios compared to the Laissez-Faire are broader in 2030 than in 2050. The farmers modify their agricultural systems according to the incentives as long as they are sufficient to significantly impact the rentabilities of the OTEA. When incentives become too small with time, representative farmers face a decision problem similar to the LF scenario although the rigidity constraint slows down such a pattern. Hence, at the end of the projections, we obtain bio-economic performances close to those obtained with the LF scenario, which is represented in fig. IV.4 with indicators converging to one. However the year of the optimum can vary according to the indicators (around 2030 for the Shannon index versus 2040 for the FBI), or on the scenarios for a given indicator (2028 with the HQE scenario and 2035 with the CR scenario for the Generalist Bird Index). The marginal effects of the different scenarios compared to the current trend are strong only for abundance indices (GBI and FBI) and more particularly for the FBI. For the other indicators, the maximal variations remain around 1%. However analysing dispersion of outcomes, we can distinguish significative differences in trajectories for the other indicators and thus significative marginal effects of economic public policies.

3.2. Abundance of habitat generalist and farmland specialist species

We note clear differences between the four scenarios with regards to both abundance indicators FBI and GBI (figs. IV.4(a) et IV.4(b)). Both indicators classify the scenarios in the same order. If evaluating the efficiency of the scenarios to enhance bird numbers, the most effective scenario is the HQE scenario, then the GL scenario, the DS scenario and finally the least effective one is the CR scenario. The FBI is very sensitive and we observe a stronger population increase, with a maximum improvement of 60% with HQE against 8% with the same scenario for the GBI. Although the farmland bird populations increase in the four considered scenarios, compared to a Laissez-Faire trend, populations of habitat generalists show more



(a) Initial state 2008 (b) Under Laissez-Faire scenario (c) Under Cereal scenario

(d) Under Grassland scenario (e) Under Double Subsidy scenario (f) Under High Quality Environmental scenario

FIGURE IV.5 – Proportions of non-intensive grassland activities at the initial state (2008) and at 2030 under the five scenarios (yellow (resp. yellowgreen, green, darkgreen) when the ratio is between 0 and 0.1 (0.1 and 0.35, 0.35 and 0.7, 0.7 and 1))

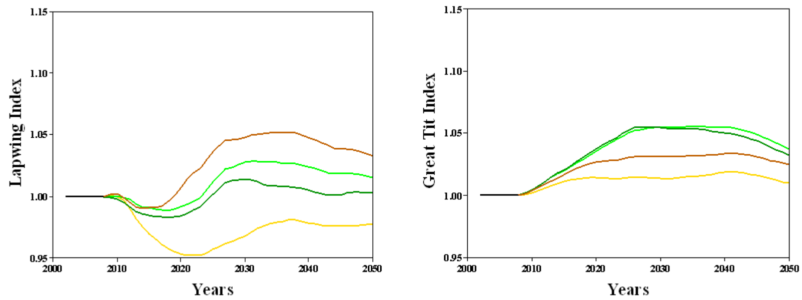
variable responses and can increase (with the HQE and GL scenarios), remain stable (with the DS scenario) or decrease (with the CR scenario).

3.3. Shannon diversity

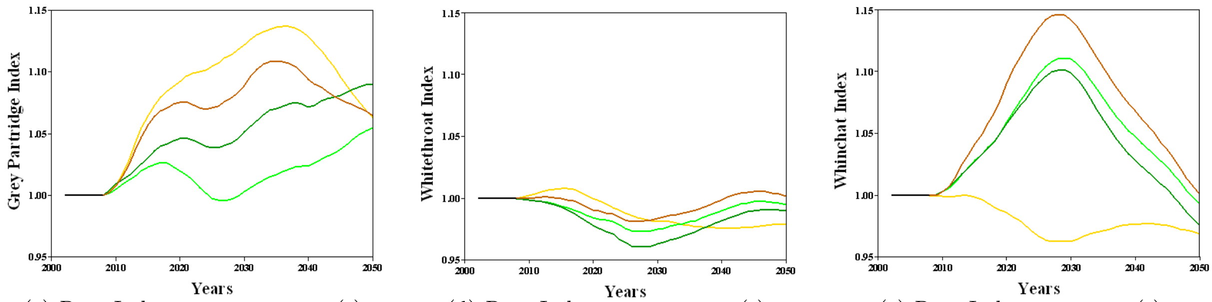
We observe two general trends for the Shannon index on fig. IV.4(c). With the CR scenario, the index decreases compared to the current trend, while it increases with the three other scenarios. As a consequence, bird communities ensuing from scenarios promoting grasslands display a better balanced composition than would be observed by maintaining the current trends. The best level is reached with the DS scenario.

3.4. Community specialization

The four scenarios are different with regards to the Community Specialization Index (fig. IV.4(d)). Compared to the ongoing trend, both scenarios with subsidies for COP (CR and DS scenarios) enhance the specialisation level of bird communities. The other two scenarios (GL and HQE) have lower CSI with respect to the current trend. As with the abundance indices, the extreme situations are obtained with the extreme scenarios : the CR scenario for the highest CSI (the most specialized community) and the HQE for the lowest CSI (the least specialized community). Figures IV.6(a) and IV.6(b) report the modeled trends for two species of similar trophic level but contrasted habitat specialisation. The specialized Lapwing *Vanellus vanellus* is more sensitive to scenarios than the generalist Great Tit *Parus major*, the numbers of which



(a) Rate Index$_{\text{Lapwing,Nat}}(t)$ (b) Rate Index$_{\text{Great Tit,Nat}}(t)$



(c) Rate Index$_{\text{Grey Partridge,Nat}}(t)$ (d) Rate Index$_{\text{Whitethroat,Nat}}(t)$ (e) Rate Index$_{\text{Winchat,Nat}}(t)$

FIGURE IV.6 – Rate indices with the Crop (yellow), Grassland (light-green), High Quality Environmental (dark-green) and Double Subsidy (brown) scenarios normalized by the Laissez-Faire evolutions for very specialized Lapwing (SSI=2.23, STI=1.90) and generalist Great Tit (SSI=0.29, STI=1.85) ; specialized Grey Partridge, intermediate landscape specialized Whitethroat and Grassland specialized Winchat.

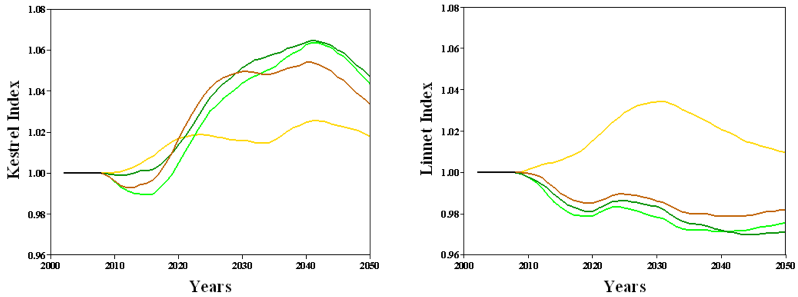
always increase within the community. Figures IV.6(c), IV.6(d) and IV.6(e) report the trends for three farmland specialists, though specialized to different types of farmland. The abundances of Grey Partridge *Perdix perdix* and Whinchat *Saxicola rubetra*, being more specialized in one farmland habitat type (openfield versus extensive grasslands, respectively) respond more strongly to scenarios. For the Whinchat *Saxicola rubetra*, the progressive reduction in incentives and taxes has an obvious effect and the population size decreases substantially after an initial increase in the scenarios favoring grasslands. The Common Whitethroat *Sylvia communis*, favoring more mixed farmed landscapes, is less affected by variations between scenarios.

3.5. Community trophic level

As observed for the Shannon index, the CR scenario is the only one clearly discriminated compared to the other three (fig. IV.4(e)) : the average trophic level of the bird community is lower for the CR scenario than the current trend and the other scenarios. Differences between the three scenarios promoting grasslands are not obvious though the GL scenario seems to display slightly better trophic levels than the DS and HQE scenarios. Fig. IV.7 compares responses of two species with similar habitat specialisation levels but contrasted trophic indices. Scenarios promoting grasslands lead to communities with larger proportions of Kestrel *Falco tinnunculus* (a bird of prey of high trophic level) and a smaller proportion of Linnet *Carduelis cannabina* (a granivorous passerine of lower trophic level).

3.6. Indicators comparison

In order to compare the indicators, fig. IV.8 synthesizes the different ecological performances among the scenarios at year 2040. Both population indicators FBI and GBI rank scenarios in the same way. However



(a) Rate $\text{Index}_{\text{Kestrel,Nat}}(t)$ (b) Rate $\text{Index}_{\text{Linnet,Nat}}(t)$

FIGURE IV.7 – Rate indices with the Crop (yellow), Grassland (light-green), High Quality Environmental (dark-green) and Double Subsidy brown) scenarios, normalized by the Laissez-Faire evolutions for Kestrel (SSI=0.98, STI=2.85) and Linnet (SSI=0.70, STI=1.05).

the responses of the three structure indicators, the Shannon Index, the CTI and the CSI, show considerable variation. The Shannon Index and the CTI, which have similarities, are the complete opposite of the CSI.

4. Discussion

By developing dynamic models coupling economic agricultural policies and biodiversity dynamics, we intended to evaluate the potential impacts of incentives or taxes dedicated to crops or grasslands on various ecological indicators related to bird populations and communities. Overall, the five bird-related indicators behaved differently according to the incentive scenarios, and can be interpreted in light of their ecological meanings for bird communities (Couvet *et al.*, 2008).

4.1. Incentives to drive ecological performance

The contrasted responses of the five indicators to the various tested economic public policies emphasize that economic incentives can be an adequate driver for bird biodiversity. The improvment of 60% in FBI with HQE scenario confirms that the current decline of farmland biodiversity is potentially reversible (Mouysset *et al.*, 2011), which is consistent with the recent increase of the FBI in French High Nature Value farmlands (Doxa *et al.*, 2010). The impacts of public incentives on bird communities justify their use by the Common Agricultural Policy (CAP).

However indicators show a trend to go back to their baseline value (e.g. 1) by 2030-2040. This means that incentives become too small to influence the farmer's decisions and a return to the initial land uses occurs in the long run. This result suggests that contrary to the current CAP trend, it is important to maintain incentives to obtain sustainability for the bird community. Positive indicator evolutions for the first 20-30 years of the projections in spite of already decreasing incentives suggest that decision-makers can initially use decreasing incentives but then have to stabilize them. However the "optimal" times differ between the indicators as well as between scenarios for a same indicator. Determining the incentive stabilization requires a specific study and will depend on ecological indicators chosen to evaluate the bird communities. However, reducing current incentives, while keeping beneficial effects on bird communities, opens many possibilities for a budget re-allocation to other environmental options. Moreover, all public policies presented here are compatible with the current decision-maker budget and in this sense are sustainable.

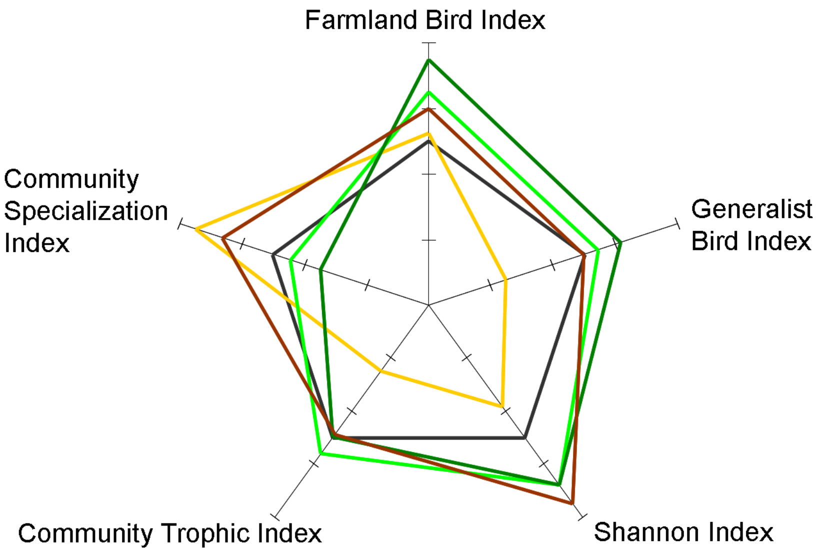


FIGURE IV.8 – Comparison of the 5 ecological indicators in 2040 with the Crop scenario (yellow), Grassland scenario (light-green), High Quality Environmental scenario (dark-green) and Double Subsidy scenario (brown), compared to the Laissez-Faire evolutions (represented by the black pentagon). The indicator value enhances when it gets further from the center.

4.2. Contrasted populations among the scenarios

With the CR scenario, we illustrate that public policies in favour of crops, for example dedicated to bioenergy developments, could be catastrophic for bird communities. With this kind of scenario, we obtain very limited improvements of species abundances, more specialized communities (Doxa *et al.*, 2010) and a strong decrease in the average trophic level of communities, with more granivorous species such as the partridges and the linnet. In contrast, promoting non-intensive grasslands appears beneficial for breeding birds when compared to the current situation. Communities are larger in size, are more diversified (with farmland and generalist species), and of higher average trophic level (with granivorous, insectivorous and carnivorous species, a more complete and balanced food chain). Species with higher positions in the food chain, such as the Red-backed Shrike, the Buzzard and the Krestel, are more abundant when non-intensive grasslands are economically promoted. Larger communities composed of more diverse species and spanning the complete food chain, certainly present advantages in terms of ecological services and sustainability. The sensitivity of such communities to disturbances is lower and the sustainability of the whole community improves (Ives & Carpenter, 2007, McCann, 2000). This kind of community has a strong interest for agricultural activities through ecosystem services. Large and diverse communities are more resilient to global changes (Keesing *et al.*, 2010), and provide more diversified and sustainable ecosystem services, such as pest control, pollination and decomposition processes (Tilman *et al.*, 2002, Wilby & Thomas, 2002, Altieri, 1999, Schläpfer *et al.*, 1999).

However, none of the three scenarios promoting non-intensive grasslands (GL, DS and HQE) are the most effective for all studied ecological dimensions. The DS scenario is particularly interesting for the balance between the different species as well as for the stability of the community. The GL scenario performs better at enhancing the average trophic level of bird communities, potentially maximising the associated provision of ecosystem services. Finally, the HQE scenario induces the largest population increases, ensuring a larger bird biomass but less specialized community. Promoting non-intensive grasslands appears essential for the management of the bird communities, and of agriculture, even if it induces more complex economic effects.

Adding other incentives to subsidies for non-intensive grasslands seems to favor some functional features of bird communities. Before selecting a policy scenario, the decision-maker has to prioritize the biodiversity metrics to be targeted.

4.3. Indicator revelance

In order to describe the bird communities, we developed two kinds of indicators : population size indicators (FBI and GBI) which aggregate annual indices of species population trends at the national scale, and community structure indicators (Shannon Index, CSI and CTI) averaging local diversity/functional indices. These two kinds of indicators are complementary and combining them globally informs on the community dynamics. Concerning population sizes, FBI and GBI behave similarly but with a higher sensibility for the FBI. FBI therefore appears more relevant to discriminate scenarios if considering population size indices (Mouysset *et al.*, 2011). Concerning community indicators, the Shannon diversity, CSI and CTI discriminate scenarios differently. CSI and CTI are built up from functional traits of species and are thus more informative about the community functioning than Shannon diversity. Moreover, the relevance of CSI and CTI depends on the agricultural farming system. CSI discriminates very well communities in mixed farmland and grassland landscapes, where more diverse communities will have higher CSI. However, in openfield landscapes, e.g. in more intensive cropping systems, poorly-diverse communities also have a high CSI. Indeed, such communities contain few species and those they have are almost all crop-specialists along with very low numbers of habitat generalists. In this context, CTI turns out to be a better functional indicator as it classifies openfield communities within those of low trophic level. However, in grassland landscapes, CTI cannot clearly rank communities. A sustainable landscape will lead to both high CSI and high CTI for bird communities. Combining FBI, CSI and CTI makes it possible to describe communities with relevance and thus specify sustainable scenarios. We advocate their use as an appropriate support for policy decisions and adaptive management of farmland biodiversity.

4.4. Bio-economic model for decision support

We here argue that several characteristics of the proposed bio-economic model make it relevant for decision support in agriculture and biodiversity management. First, in complement to studies focusing on farmland use scenarios (Taylor & Morecroft, 2009, Albrecht *et al.*, 2007, Batary *et al.*, 2007, Kleijn & van Zuijlen, 2004, Münier *et al.*, 2004), the bio-economic approach relies on economic scenarios, financial incentives and policies which constitute major inputs taken into account by stakeholders in reality. Similarly, the compatibility of the policies with the current CAP budget also gives realism to such a study. Moreover, the national scale leads us to think more about global policies and general directions than about local management, the effectiveness of which is controversial (Le Roux, 2008, Kleijn & van Zuijlen, 2004). Second, another important characteristic of this bio-economic modelling regarding decision support is its the prospective dimension. With the description of different future scenarios beyond the Laissez-Faire scenario, this work is complementary with the numerous studies which focus on the impact of current policies (Taylor & Morecroft, 2009, Vickery *et al.*, 2004). These simulations are useful for public policies in agriculture and biodiversity conservation where experimental schemes are complicated to establish. In particular, our approach stresses some non-linearities between incentives and ecological indicators according to the scenarios. If outcomes with the DS scenario are often intermediate between those issued from the CR scenario and the GL/HQE scenarios, such is not the case with the Shannon index where the DS scenario leads to an extreme trajectory. As previously mentioned, synergies between incentives do not affect bird communities similarly, depending on the same functional traits considered, and can represent an interesting leverage to enhance the effectiveness of agricultural policies on different criteria. Finally focusing on one general taxon (birds) rather than on one or two emblematic species makes it possible to adopt a broad viewpoint for biodiversity and potential associated ecosystem services. Consequently, the genericity of the results is reinforced as regard biodiversity management and agri-environmental policies.

4.5. Perspectives

The present work advocates the use of bio-economic models for public farming policies and terrestrial biodiversity conservation. A major trend of the French CAP which has not been tested in this study is the regionalisation of incentives. With the national scale decomposed into economic-ecological homogeneous areas (PRA), our conceptual framework could be a fruitful instrument to test several policies of incentive regionalisation in consistency with the national budget.

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Chapitre V

A double dividend of biodiversity in agriculture

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Abstract

The objective of this paper is to contribute to the accounting for biodiversity goals in the design of agricultural policies. A bio-economic dynamic model is developed with a multi-scale perspective. It couples biodiversity dynamics, farming land-uses selected at micro level and public policies at macro level based on financial incentives for land-uses. The public decision maker provides optimal incentives respecting both biodiversity and budgetary constraints. These optimal policies are then analyzed through their private, public and total costs. The model is calibrated and applied to metropolitan France at PRA ("petite région agricole") scale using the common birds as biodiversity metrics. The study first shows that the efficiency curves display decreasing concavities for different biodiversity indicator pointing out the trade-off occuring between biodiversity and economic scores. However, the total and public costs suggests that the account for biodiversity can generate a second benefit in terms of public incomes. It is argued how a regional redistribution of this public earning to the farmers could promote the acceptability of biodiversity goals in agricultural policies.

Keywords : Biodiversity, Land-use, Bio-economic modelling, Cost-effectiveness, Birds

1. Introduction

In many European countries, a strong decline of biodiversity is observed in agricultural landscapes. This is especially documented for mammals in Flowerdew & Kirkwood (1997), for arthropods and plants in Sotherton & Self (2000) or for birds in Donald *et al.* (2001). Numerous studies (Chamberlain *et al.*, 2000, Wretenberg *et al.*, 2007) identify the changes in agricultural systems over the last decades and especially the intensification processes at play as major drivers of this erosion. The breeding bird populations are particularly vulnerable to global agricultural change (Jiguet *et al.*, 2010, Krebs *et al.*, 1999). Such a negative effect is due mainly to a degradation in habitat quality altering nesting success and survival (Benton *et al.*, 2003). In this context, the European Union has formally adopted the Farmland Bird Index (FBI) as an indicator of structural changes in biodiversity (Balmford *et al.*, 2003).

A challenge to reach sustainability for agricultural land-use is therefore to reconcile farming production and farmland biodiversity. Usual approaches to achieve such multifunctional goals for farming rely on public policies (Pacini *et al.*, 2004) or economic incentives (Drechsler *et al.*, 2007, Mouysset *et al.*, 2011). For Alavalapati *et al.* (2002) and Shi & Gill (2005), financial incentives are essential for convincing farmers to adopt eco-friendly activities. These policies modify the farmer's choices and thus impact both the habitat and the dynamics of biodiversity (Doherty *et al.*, 1999, Holzkamper & Seppelt, 2007, Rashford *et al.*, 2008). In this perspective, many public policies including agri-environmental schemes have been proposed by decision makers. However, fifteen years after the initial implementation of such instruments at a large scale, their ability to enhance biodiversity remains controversial (Butler *et al.*, 2009, Kleijn *et al.*, 2006, Vickery *et al.*, 2004). These policies face different difficulties. From the ecological viewpoint, the insufficient knowledge about the agro-ecological processes at play and the focus on few emblematic species limits the results. From the economic viewpoint, the weak acceptability by the farmers constitutes a major obstacle for the effectiveness of these policies. In this context, testing the efficiency of the different agricultural policy scenarios through quantitative methods and models is useful. The Cost-Benefit method (Boardman *et al.* (2005)) compares the costs and the benefits of a policy using monetary values. However, quantifying the economic benefits of agricultural policy is particularly difficult for complex biodiversity (Diamond & Hausman, 1994). The Cost-Effectiveness analysis, which avoids the monetary evaluation, appears as a relevant alternative. This method, based on optimization under constraint, leads to define either the less expensive policy satisfying a biodiversity goal or the policy with the best biodiversity performances under a budgetary constraint (Naidoo *et al.*, 2006). Many authors (Polasky *et al.*, 2008, Drechsler *et al.*, 2007, Polasky *et al.*, 2005) using this method for agricultural policy issues exhibit a pareto-efficient frontier of optimal policies. As in Green *et al.* (2005) this frontier is generally concave pointing out a trade-off occuring between biodiversity and economic scores. In other words, it is possible to moderatly improve biodiversity performances with weak income losses (Barraquand & Martinet, 2011, Lewis *et al.*, 2011, Polasky *et al.*, 2005).

The objective of this paper is to contribute to the accounting for biodiversity goals in the design of agricultural policies. More specificaly, cost-effective policies are designed and analyzed through different costs to identify potential ways to reduce the trade-off and improve the acceptability of such policies. The study relies on a spatio-temporal bio-economic model which articulates farming land-uses selected by rational agents, biodiversity community dynamics at micro (landscape) level and macro (typically national) financial incentives associated with land-uses. The public policies are computed at the macro scale through a cost-effectiveness method which maximize a present value of incomes under different biodiversity targets and a budgetary constraint. Then the study focuses on the public and total social costs associated with each cost-effective policies as in Semaan *et al.* (2007). The method is applied to the metropolitan France case study. The calibration relies on French time series of 34 birds abundance and 14 farming land-uses over years 2001-2009 and 620 "small" regions (PRA) in metropolitan France. Two indicators, the Farmland Bird Index (FBI) which has been adopted by the European Union (Balmford *et al.*, 2003), and the Commu-

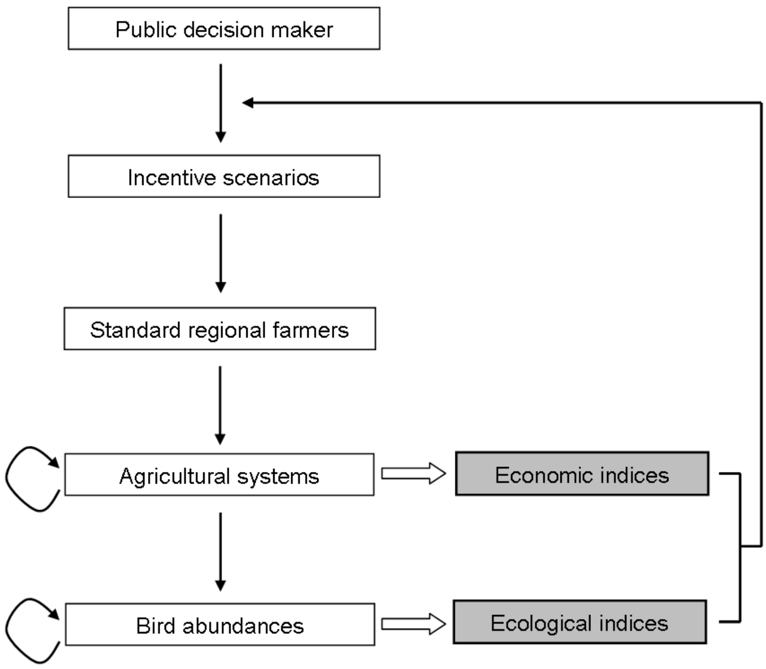


FIGURE V.1 – Bio-economic model coupling. The decision maker determines an incentive scenario according a bio-economic optimization. The farmers choose their agricultural systems by maximizing their income under technical constraints. These choices affect the habitat and the bird communities.

nity Trophic Index (CTI) which informs on a functional feature of the community (Mouysset *et al.*, 2012, Pauly *et al.*, 1998) capture the biodiversity scores. The study illustrates that the efficiency curves of the agricultural policies with biodiversity constraints have different qualitative shape according to the ecological indicator. The analysis of the total and public costs shows that the integration of biodiversity goals is not detrimental to the whole society in the sense that it can generate a benefit in terms of public budget. In other words, the biodiversity-oriented policy yields a double benefit. We suggest that the redistribution of the induced earnings to the farmers could compensate their private loss and so increase their acceptance of biodiversity objectives in the design of agricultural policy. A first strategy is proposed through the regional redistribution.

The paper is organized as follows. The second section describes the bio-economic model. The third section presents the case study. The fourth and fifth sections are devoted to the results and their discussion respectively.

2. The bio-economic modelling

As depicted by figure V.1, the bio-economic model is composed of three compartements with a multi-scale pespective as in Mouysset *et al.* (2011) : the public policy at macro (national) scale interacts with the farming land-uses and biodiversity dynamics at micro scale.

2.1. The micro-economic model

Each region is represented by a standard agent. Agent income in region r at year t denoted by $Inc_r(t)$ relies on the expected gross margin per unit of scale $\overline{gm}_{r,k}$, current proportions of the Utilized Agricultural Area

(UAA) dedicated to the agricultural land-uses $A_{r,k}(t)$ and incentives τ_k (taxes with $\tau_k < 0$ or subsidies with $\tau_k > 0$) which takes form of a percentage of gross margins as follows :

$$Inc_r(t) = \sum_k \overline{gm}_{r,k}.A_{r,k}(t).(1+\tau_k) \tag{V.1}$$

For each year t, the regional standard agents choose their agricultural land-uses $A_{r,k}(t)$ in order to maximize their income $Inc_r(t)$ according to capital and rigidity constraints :

$$\max_{A_{r,k}} Inc_r(t) = \max_{A_{r,k}} \sum_k \overline{gm}_{r,k}.A_{r,k}(t).(1+\tau_k) \tag{V.2}$$

under the constraints

$$|A_{r,k}(t) - A_{r,k}(t-1)| \leq \varepsilon.A_{r,k}(t-1) \tag{V.3}$$

$$\sum_k A_{r,k}(t) = UAA_r(t_0) \tag{V.4}$$

The rigidity constraint (V.3) restricts the area that the farmer can modify at each time for each agricultural system k. The parameter ε captures change costs or inertia. The constraint (V.4) ensures that the total utilized agricultural area (UAA) is kept fixed.

2.2. The biodiversity model

The biodiversity model deals with a community of species instead of focusing on emblematic species. It is based on population dynamics with intra-specific competition depending on habitat and especially on farming land-use. A Beverton-Holt function is selected for sake of simplicity. It captures intra-specific competition through a carrying capacity parameter as follows :

$$N_{s,r}(t+1) = N_{s,r}(t).(1+R_{s,r}) \left(1 + \frac{N_{s,r}(t)}{M_{s,r}(t)}\right)^{-1} \tag{V.5}$$

where $N_{s,r}(t)$ stands for the abundance of species s in region r at year t. The $R_{s,r}$ coefficient corresponds to the intrinsic growth rate specific to a given species s in region r. The product $M_{s,k}(t) * R_{s,r}$ represents the carrying capacity of the habitat r and the value $M_{s,k}(t)$ captures the ability of the habitat r to host the species s. The habitat parameter $M_{s,r}(t)$ is assumed to depend on the farming land-uses $A_{r,k}(t)$ as follows :

$$M_{s,r}(t) = \beta_{s,r} + \sum_k \alpha_{s,r,k}.A_{r,k}(t) \tag{V.6}$$

Consequently, the $\alpha_{s,r,k}$ and $\beta_{s,r}$ coefficients, specific to each species, inform on how such species s responds to agricultural land-use k in a region r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a region r and integrates others factors such as the proportion of forests or urban areas.

The indicators used to assess the ecological performances are computed through the abundances $N_{s,r}(t)$ of the species at play. We denote the biodiversity index by *Biod* without specifying it at this stage. Such formulation includes usual biodiversity indices such as species richness, simpson or trophic indices. In each region, it is defined as follows :

$$Biod_r(t) = h\big(N_{1,r}(t), \ldots, N_{S,r}(t)\big) \tag{V.7}$$

2.3. The public policy model

To analyze at macro scale the intertemporal economic performance, we use the net present value of incomes. The macro (national) income $Inc(t)$ depends on the micro (regional) incomes $Inc_r(t)$ and the superficy S_r of the UAA in every region r as follows

$$Inc(t) = \sum_r S_r \, . \, Inc_r(t) \tag{V.8}$$

The present value $PV(\tau)$ is defined as the intertemporal sum of the national incomes $Inc(t)$ associated with a discount rate ρ from the first year of the projection t_1 to the time horizon T.

$$PV(\tau) = \sum_{t=t_1}^{T} \rho^{t-t_1} \, . \, Inc(t) \tag{V.9}$$

This present value is contingent to the public incentives τ_k through relation (V.1). The decision maker at macro scale selects the vector τ of the economic incentives τ_k, defined as percentages of the gross margins $\overline{gm}_{r,k}$. The decision maker optimaly chooses taxes and/or subsidies for different agricultural land-uses k by maximizing the present value $PV(\tau)$ according to a budgetary and a biodiversity constraints :

$$\max_{\tau} PV(\tau) \tag{V.10}$$

under the constraints

$$Budg(t) \ \leq \ Budg(t_0) \tag{V.11}$$

$$Biod(T) \ \geq \ B_{lim} \tag{V.12}$$

The budgetary constraint (V.11) ensures that the public budget at each time t does not exceed the current budget at time t_0. The budget $Budg(t)$ is computed according to the different incentives τ_k as follows :

$$Budg(t) = \sum_r \sum_k S_r \, . \, \overline{gm}_{r,k} \, . \, A_{r,k}(t) \, . \, \tau_k \tag{V.13}$$

The ecological target (V.12) is based on a conservation limit B_{lim} for the biodiversity goal only imposed at the temporal horizon T. Different values of B_{lim} can be tested between the maximal feasible biodiversity [1] $B_{lim} = B^*$ and the lowest value $B_{lim} = 0$.

The cost-effective incentives, optimal solution of the problem (V.10)-(V.11)-(V.12), are denoted by :

$$\tau^*(B_{lim}) = \underset{\tau \text{ admissible}}{\text{Argmax}} \quad PV(\tau) \tag{V.15}$$

2.4. Public, private and social biodiversity costs

The public policies induce two kinds of cost as proposed by Semaan *et al.* (2007) : public and private costs. The analyze of such costs is helpful to evaluate the price of the different policies for the entire society and the weight dedicated to each part (public and private agents). The public cost denoted by $PuC(B_{lim})$ corresponds to the public budget of an optimal policy allocated to the agents at each time t. It depends on

1. This maximum B^* is defined by a biodiversity maximisation under the budgetary constraint :

$$B^* \ = \ \max_{\begin{cases} -1 \ \leq \ \tau_k \ \leq \ 1 \\ Budg(t) \ \leq \ Budg(t_0) \end{cases}} \quad Biod(\tau) \tag{V.14}$$

the biodiversity target B_{lim} as the budget is itself function of the optimal incentives $\tau^*(B_{lim})$. The public cost reads as follows :

$$PuC(B_{lim}) = \sum_{t=t_1}^{T} \rho^{t-t_1} \, . \, Budg^*(t) \tag{V.16}$$

where $Budg^*$ stands for the cost-effective budget in the following sense :

$$Budg^*(t) = \sum_{k} S_r.\overline{gm}_{r,k} \, . \, A_{r,k}(t).\tau^*(B_{lim}) \tag{V.17}$$

By contrast, the private cost $PrC(B_{lim})$, based on the loss of farmer's income due to biodiversity requirements, is computed as the difference between the maximum feasible present value $PV(\tau^*(0))$ without biodiversity target and the present value $PV(\tau^*(B_{lim}))$ under biodiversity goal B_{lim} :

$$PrC(B_{lim}) = PV(\tau^*(0)) - PV(\tau^*(B_{lim})) \tag{V.18}$$

The total social cost $SoC(B_{lim})$ is defined as the sum of the public and the private costs :

$$SoC(B_{lim}) = PuC(B_{lim}) + PrC(B_{lim}) \tag{V.19}$$

The question whether these costs are positive or not is decisive for the acceptability of biodiversity requirements and the adoption of eco-friendly agricultural policies.

2.5. Costs at regional scale

The different costs are computed at micro scale in a similar way. The cost-effective budget $Budg_r^*(t)$ is defined by :

$$Budg_r^*(t) = \sum_{k} \mathrm{S_r} \, . \, \overline{\mathrm{gm}}_{\mathrm{r,k}} \, . \, \mathrm{A_{r,k}(t)} \, .\tau^*(\mathrm{B_{lim}}) \tag{V.20}$$

while the micro public cost $PuC_r(B_{lim})$ corresponds to :

$$PuC_r(B_{lim}) = \sum_{t=t_1}^{T} \rho^{t-t_1} \, . \, Budg_r^*(t) \tag{V.21}$$

The micro (regional) private cost $PrC_r(B_{lim})$, based on the regional present value $PV_r(\tau)$, evaluates the loss of earnings due to the ecological objective :

$$PrC_r(B_{lim}) = PV_r(\tau^*(0)) - PV_r(\tau^*(B_{lim})) \tag{V.22}$$

where

$$PV_r(\tau) = \sum_{t=t_1}^{T} \rho^{t-t_1} \, . \, Inc_r(t) \tag{V.23}$$

Finally the regional total social cost is the sum between the regional public and private costs :

$$SoC_r(B_{lim}) = PuC_r(B_{lim}) + PrC_r(B_{lim}) \tag{V.24}$$

The 14 land-uses (OTEA) k
(1) Cereal, Oleaginous, Proteaginous (COP)
(2) Variegated crops
(3) Intensive bovine livestock breeding
(4) Medium bovine livestock breeding
(5) Extensive bovine livestock breeding
(6) Mixed crop-livestock farming with herbivorous direction
(7) Other herbivorous livestock breeding
(8) Mixed crop-livestock farming with granivorous direction
(9) Mixed crop-livestock farming with other direction
(10) Granivorous livestock breeding
(11) Permanent farming
(12) Flower farming
(13) Viticulture
(14) Others associations

TABLE V.1 – List of the 14 farming land-uses (OTEA)

3. The French case study

3.1. Context

We apply this bio-economic modelling framework to metropolitan France. France is split into 620 small agricultural regions (PRA for Petites Regions Agricoles). A PRA is part of a department (a major French administrative entity) which exhibits an agro-ecological homogeneity. This consistency from both the ecological and economic points of view makes the PRA the relevant regional scale for economic and biodiversity models. Ecological and economic data are available from 2001 to 2008 (t_0). The policy scenarios are tested between $t_1 = 2009$ and $T = 2050$. Selecting a shorter timeframe could consequently hide interesting long-term effects due to the inertia of the models.

3.2. Economic data

For agro-economic data, we use the French agro-economic classification OTEX (orientation technico-economique) developed by the French Farm Accounting Data Network (FADN) [1] and the Observatory of Rural Development (ODR) [2]. This organization distinguishes 14 classes of land-use named OTEA detailed in table V.1. Each PRA is a specific combination of these OTEA. The surfaces dedicated to the 14 land-uses OTEA and the associated fiscal bases (tax return) used as a proxy of gross margins for the years 2001 to 2008 are available on the ODR website under a private request. Gross margin is an economic index broadly used in agricultural economics (Lien, 2002). For accelerating the numerical computations, the public decision variables τ_k are restricted to only two incentives : the cereal incentive τ_{cop} is dedicated to arable lands (Otea (1) in table V.1) and the grassland incentive τ_{grass} is applied to non-intensive grassland systems (Otea (4), (5), (6), (7) in table V.1). The gross margins $\overline{gm}_{r,k}$ are computed as the temporal mean of the historical gross margins :

$$\overline{gm}_{r,k} = \frac{1}{8} \sum_{t=2001}^{2008} gm_{r,k}(t) \qquad \text{(V.25)}$$

The budgetary constraint is calibrated with the current French CAP budget.

1. http ://ec.europa.eu/agriculture/rica/
2. https ://esrcarto.supagro.inra.fr/intranet/

20 farmland bird species	14 generalist bird species
(1) Buzzard *Buteo buteo*	(1) Blackbird *Turdus merula*
(2) Cirl Bunting *Emberiza cirlus*	(2) Blackcap *Sylvia atricapilla*
(3) Corn Bunting *Emberiza calandra*	(3) Blue Tit *Parus caeruleus*
(4) Grey Partridge *Perdix perdix*	(4) Carrion crow *Corvus corone*
(5) Hoopoe *Upupa epops*	(5) Chaffinch *Fringilla coelebs*
(6) Kestrel *Falco tinnunculus*	(6) Cuckoo *Cuculus canorus*
(7) Lapwing *Vanellus vanellus*	(7) Dunnock *Prunella modularis*
(8) Linnet *Carduelis cannabina*	(8) Great Tit *Parus major*
(9) Meadow Pipit *Anthus pratensis*	(9) Green Woodpecker *Picus viridis*
(10) Quail *Coturnix coturnix*	(10) Golden oriole *Oriolus oriolus*
(11) Red-backed Shrike *Lanius collurio*	(11) Jay *Garrulus glandarius*
(12) Red-legged Partridge *Alectoris rufa*	(12) Melodius Warbler *Hippolais polyglotta*
(13) Rook *Corvus frugilegus*	(13) Nightingale *Luscinia megarhynchos*
(14) Skylark *Alauda arvensis*	(14) Wood Pigeon *Columba palumbus*
(15) Stonechat *Saxicola torquatus*	
(16) Whinchat *Saxicola rubetra*	
(17) Whitethroat *Sylvia communis*	
(18) Wood Lark *Lullula arborea*	
(19) Yellowhammer *Emberiza citrinella*	
(20) Yellow Wagtail *Motacilla flava*	

Table V.2 – List of the 20 farmland and 14 generalist bird species s

3.3. Biodiversity data

As regards the biodiversity, we focus on common bird populations and related indicators (Gregory *et al.*, 2004). Although the metric and the characterization of biodiversity remain an open debate (MEA, 2005), such a choice is justified for several reasons (Ormerod & Watkinson, 2000) : (i) Birds lie at a high level in the trophic food chains and thus capture the variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation and viability (Sekercioglu *et al.*, 2004). (iii) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens.

The STOC (French Bird Breeding Survey) database [1] provides the informations related to the bird abundances across the whole country. Abundance values for each species are available [2] for the period 2001-2008. Among the species monitored by this large-scale long-term survey, we selected 34 species which have been classified according to their habitat requirements at a Europe scale (European Bird Census Council, 2007). Table V.2 lists the 14 habitat generalist species and the 20 farmland specialist species used as a reference by the European Union (Gregory *et al.*, 2004).

3.4. Model calibration

The agro-ecological parameters $R_{s,r}$, $\alpha_{s,r,k}$ and $\beta_{s,r}$ introduced in equations (V.5)-(V.6) and the economic parameter ϵ of equation (V.3) are determined by a calibration based on a least square method. Hence are minimized errors between the observed outputs and the outputs derived from the model. The considered outputs of the model are the land-use values $A_{s,r,k}(t)$ for the economic model and the bird abundances $N_{s,r}(t)$ for the ecological model as detailed in Mouysset *et al.* (2012; 2011). The discount rate is set to $\rho = 4\%$.

1. See the Vigie-Nature website http ://www2.mnhn.fr/vigie-nature/. Standardized monitoring of spring-breeding birds at 1747 $2 * 2\, km^2$ plots across the whole country. Details of the monitoring method and sampling design in Jiguet (2009).
2. For each species, a spatial interpolation of these abundance data is performed to obtain relative abundance values for each possible square in the country (Doxa *et al.*, 2010). We then average the abundance values at the PRA scale.

3.5. Biodiversity indicators

The biodiversity indicators used in this study are the Farmland Bird Index (FBI) and the Community Trophic Index (CTI) both evaluated in final year $T = 2050$. The Farmland Bird Index has been adopted by the European Community as the official environmental index especially to analyze structural changes in biodiversity (Balmford *et al.*, 2003). The relevance of the FBI to reflect the response of farmland biodiversity to agriculture intensification has been shown in Doxa *et al.* (2010), Mouysset *et al.* (2012). We compute the FBI at the national scale with 20 farmland specialist species for each PRA :

$$FBI(t) = \prod_{s \,\in\, \text{Specialist}} \left(\frac{N_{s,nat}(t)}{N_{s,nat}(2008)} \right)^{1/20} \tag{V.26}$$

where $N_{s,nat}(t) = \sum_{r=1}^{620} N_{s,r}(t)$ stands for the total abundance of species s over the 620 PRA r.

The Community Trophic Index (CTI) informs on the average trophic level of a community as in Mouysset *et al.* (2012), Pauly *et al.* (1998). The CTI here integrates both the 14 generalist species and the 20 farmland specialist species (table V.2). It is computed as the arithmetic mean of the exponential of the species trophic level [1] weighted by the relative abundances :

$$CTI_r(t) = \sum_s \frac{N_{s,r}(t)}{N_{tot,r}(t)} \,.\exp(STI_s) \tag{V.27}$$

where $N_{tot,r} = \sum_{s=1}^{34} N_{s,r}(t)$ represents the total abundance of birds in a PRA r. The exponential function is used to better contrast communities with or without bird individuals of the higher trophic levels as in Mouysset *et al.* (2012). This indicator classifies the communities with more granivorous species (e.g. low trophic level) against the communities with more insectivorous and carnivorous species (e.g. high trophic level).

National CTI is the arithmetic mean of the 620 regional CTI_r :

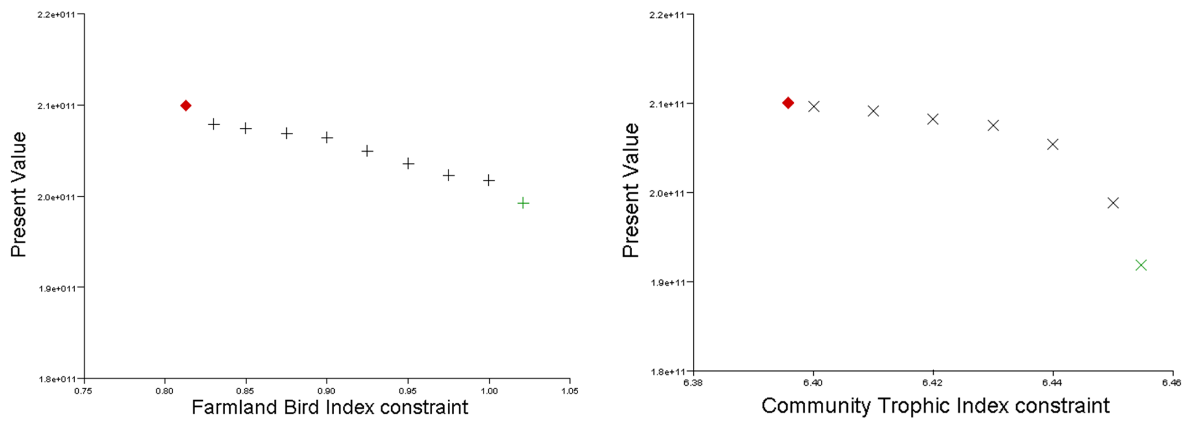
$$CTI(t) = \frac{1}{620} . \sum_r CTI_r(t) \tag{V.28}$$

4. Results

4.1. Efficiency curves

The figure V.2 illustrates the bio-economic performances of the optimized present values under biodiversity and budgetary constraints. The red diamond corresponds to the policy $\tau^*(0)$ without biodiversity constraint and the green plus in fig. V.2(a) (cross in fig. V.2(b) resp.) to the $\tau^*(FBI^*)$ policy ($\tau^*(CTI^*)$ resp.). The black plus (crosses resp.) represent the $\tau^*(FBI_{lim})$ policies (the $\tau^*(CTI_{lim})$ policies resp.). Their projection on the x-axis illustrates the level of the biodiversity constraint B_{lim} and their projection on the y-axis shows the associated present value $PV(\tau^*)$. We observe two efficiency curves which are both decreasing but with different shapes. The curve obtained with the FBI constraint in fig. V.2(a) is almost linear. Thus, the increase of the FBI constraint leads to regular losses on the economic indicator. By contrast, the curve obtained with the CTI constraint in fig. V.2(b) displays a concavity especially strong for large biodiversity level B_{lim}. Hence the increase of the CTI constraint has slight impacts on the economic indicator for CTI levels lower than 6.43. After this threshold, the economic loss becomes major.

1. See in Mouysset *et al.* (2012) for the Species Trophic Indices



(a) (b)

FIGURE V.2 – Optimal present values $PV(\tau^*(B_{lim})$ with respect to the biodiversity constraint B_{lim}. In (a) with the $FBI(2050)$ biodiversity indicator and in (b) with the $CTI(2050)$ biodiversity indicator. The extreme policies $\tau^*(0)$ and $\tau^*(B^*)$ are in red and green respectively.

4.2. Optimal public incentives

The tables V.3 and V.4 depict the optimal incentives with increasing biodiversity goals. For both ecological indices, we observe a decrease of the cereal subsidies τ_{cop} with biodiversity objective B_{lim}. In particular, for the strongest biodiversity targets, the incentive becomes a tax. In contrast, the incentive for non-intensive grasslands τ_{grass} remains globally stable with a high value except for the policy with the more stringent CTI constraint namely CTI^*. Globally, these observations highlight the need to promote the grassland at the expense of the crops to satisfy biodiversity objectives. According to the selected ecological indicator, this pattern is more or less emphasized.

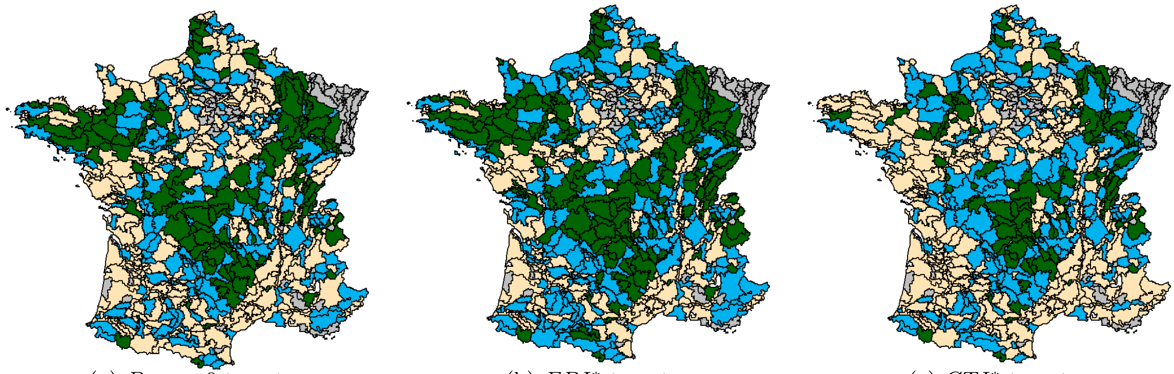
The figure V.3 illustrates the proportions of UAA dedicated to the non-intensive grassland systems for the three extreme policies : the $\tau^*(0)$ policy in figure V.3(a), the $\tau^*(FBI^*)$ policy in figure V.3(b) and the $\tau^*(CTI^*)$ policy in figure V.3(c). The $\tau^*(FBI^*)$ strategy promotes the grassland activities through an increase of PRA with important grassland proportions. The $\tau^*(CTI^*)$ incentives induce a developement of PRA with moderate grassland proportions on contrary to the $\tau^*(0)$ option where the rate of intermediate PRA declines.

FBI_{lim}	0	0.825	0.85	0.875	0.9	0.925	0.95	0.975	1	FBI*
τ^*_{cop}	0.47	0.27	0.23	0.23	0.14	0.02	-0.06	-0.19	-0.25	-0.54
τ^*_{grass}	0.52	0.58	0.59	0.58	0.61	0.62	0.61	0.62	0.62	0.63

TABLE V.3 – Optimal cereal incentives τ^*_{cop} and grassland incentives τ^*_{grass} for different biodiversity target B_{lim} using the FBI as biodiversity index.

CTI_{lim}	0	6.40	6.41	6.42	6.43	6.44	6.45	CTI*
τ^*_{cop}	0.47	0.42	0.37	0.33	0.23	0.20	0.34	-0.02
τ^*_{grass}	0.52	0.54	0.56	0.56	0.59	0.54	0.23	0.23

TABLE V.4 – Optimal cereal incentives τ^*_{cop} and grassland incentives τ^*_{grass} for different biodiversity target B_{lim} using the CTI as biodiversity index.



(a) $B_{lim} = 0$ target (b) FBI^* target (c) CTI^* target

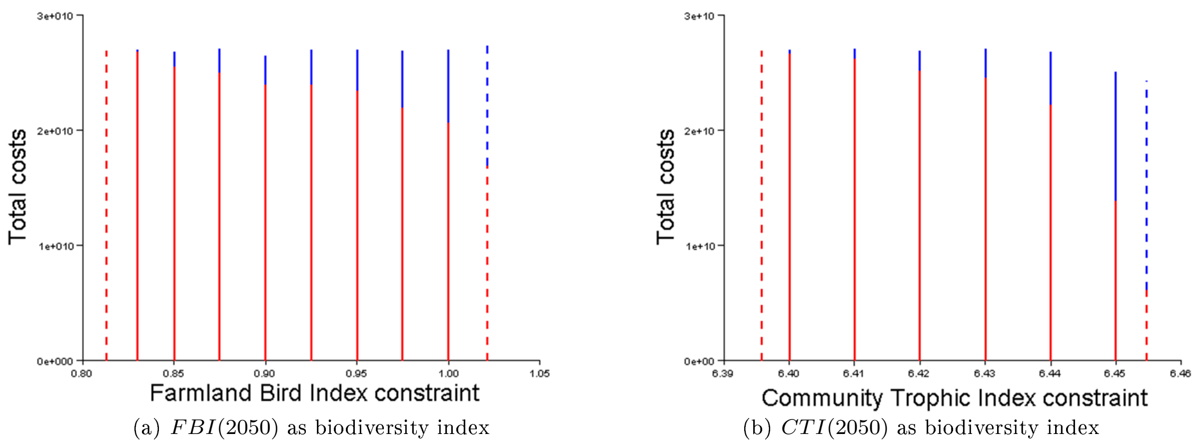
FIGURE V.3 – Proportions of the non-intensive grassland land-use (OTEA) ($\sum_{k=4}^{7} \frac{A_{r,k}(2050)}{UAA_r}$) at the PRA scale for optimal policies under several biodiversity target B_{lim}. In green : 100-45%, in blue : 45-10%, in yellow : 10-0%.

4.3. National costs

The figure V.4 plots the total social costs $SoC(B_{lim})$ by detailing the public $PuC(B_{lim})$ (in red) and the private $PrC(B_{lim})$ (in blue) costs for the different optimal solutions. The dotted lines on the left correspond to the $\tau^*(0)$ policy (without biodiversity) and on the right to the $\tau^*(B^*)$ policy (biodiversity oriented). We observe that the total social cost is similar with the FBI or CTI constraints. Morever it is globally steady although we note a slight decrease for the highest CTI constraints. The figure V.4 highlights the fact that the repartition between public and pivate costs changes in the same way with FBI and CTI indicators when the ecological requirement is more demanding : the public cost decreases while the private cost increases. These patterns are more contrasted with the CTI index than with the FBI.

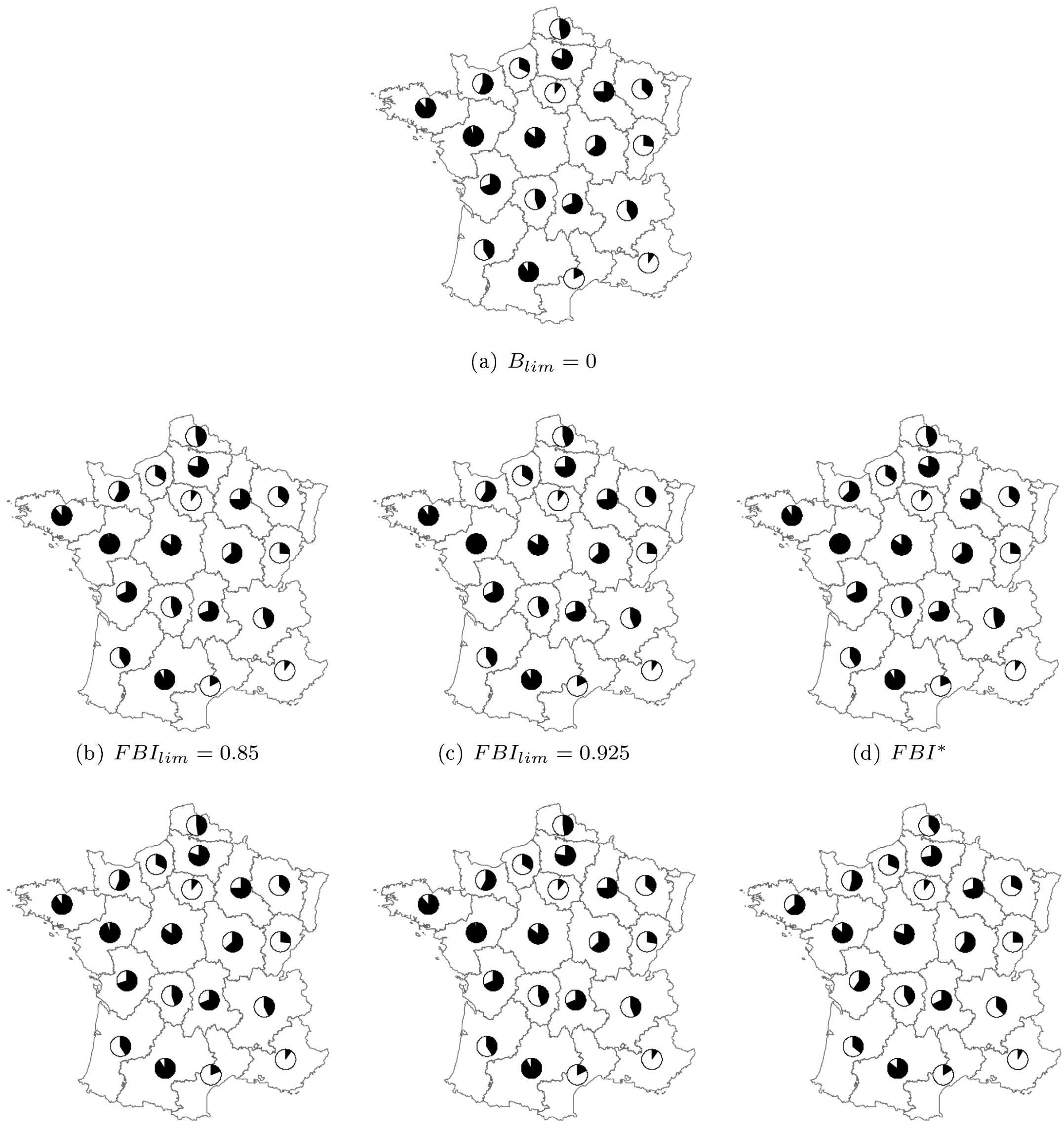
4.4. Regional costs

The figure V.5 details the regional total social costs $SoC_r(B_{lim})$ at the regional scale for several public policies $\tau^*(B_{lim})$. The figure V.5(a) stands for the $\tau^*(0)$ policy (without biodiversity target). The figures



(a) $FBI(2050)$ as biodiversity index (b) $CTI(2050)$ as biodiversity index

FIGURE V.4 – Total social costs $SoC(B_{lim})$ separated between the public costs $PuC(B_{lim})$ in red and the private costs $PrC(B_{lim})$ in blue for different biodiversity targets B_{lim}. Dashed lines stands for the extreme cases $B_{lim} = 0$ on the left and $B_{lim} = B^*$ on the right.



(a) $B_{lim} = 0$

(b) $FBI_{lim} = 0.85$ (c) $FBI_{lim} = 0.925$ (d) FBI^*

(e) $CTI_{lim} = 6.40$ (f) $CTI_{lim} = 6.43$ (g) CTI^*

FIGURE V.5 – Regional total social costs $SoC_r(B_{lim})$ in black under several biodiversity target B_{lim}. On the left the FBI and on the right the CTI for the biodiversity index.

V.5(b), V.5(c), V.5(e) and V.5(f) represent several $\tau^*(FBI_{lim})$ policies with two intermediate B_{lim} for each biodiversity indicator. Finally the figures V.5(d) and V.5(g) depict the $\tau^*(B^*)$ policies. A complete pie-chart represents the maximum regional total costs (i.e. $2,5$ millions Euros). We observe that the regional social total costs are also very stable among the optimal policies. In other words, the biodiversity constraint does not affect the social cost even at the micro level. We note also that the repartition of the national cost is not equally distributed between the regions.

The figure V.6 presents the distribution of the regional total social costs $SoC_r(B_{lim})$ between the regional public costs $PuC_r(B_{lim})$ (in red) and the regional private costs $PrC_r(B_{lim})$ (in blue) . According to the equation (V.18), there is no private biodiversity costs for the $\tau^*(0)$ policy. So we start directly with the $\tau^*(FBI_{lim})$ policies with two medium B_{lim} for each indicator. Pink represents negative public costs, where taxes exceed subsidies. Pale blue represents negative private costs, i.e. the regional farmer income is larger than under the $\tau^*(0)$ policy without biodiversity requirement. Finally, strong grey (pale grey, white resp.) regions which have very stationary (intermediary stationary, instable resp.) costs among the cost-effective strategies.

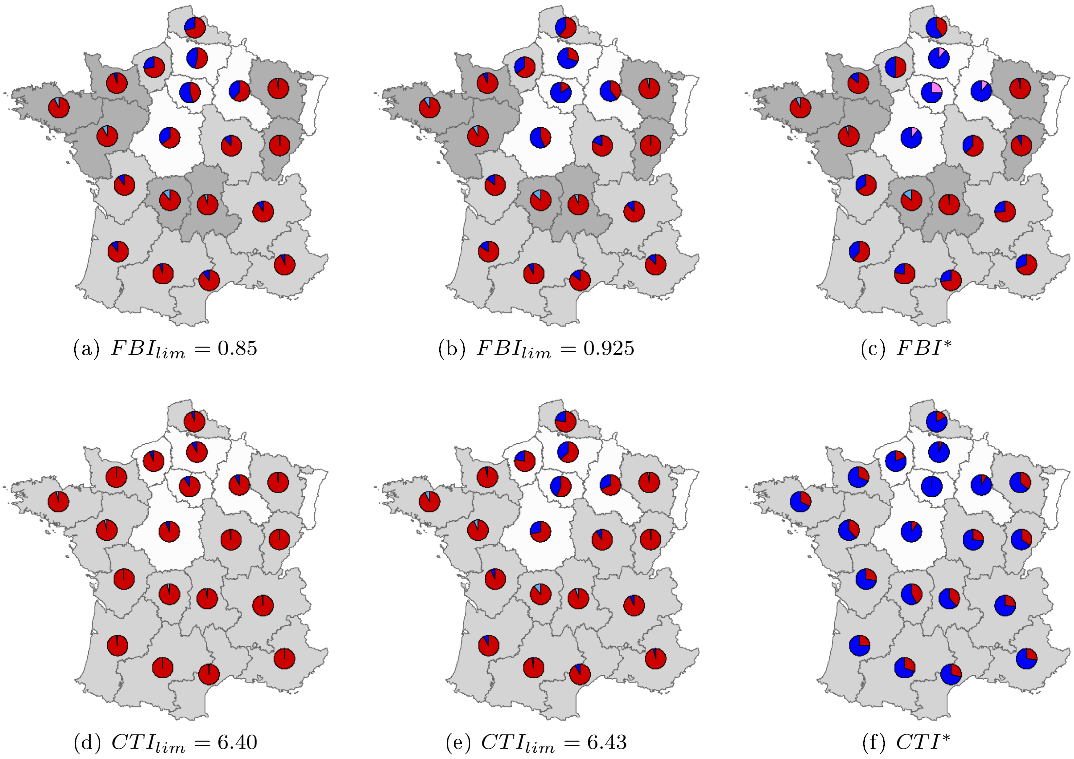


FIGURE V.6 – Regional public $PuC_r(B_{lim})$ (in red) and private $PrC_r(B_{lim})$ (in blue) costs under several biodiversity target B_{lim}. Pink stands for negative public costs and pale blue negative private costs. Grey (resp. pale grey, white) regions present stable (resp. intermediary stable, instable) costs.

We observe that the policies differently affect the regions. But the patterns are similar for the two indicator and all the regions : when the biodiversity constraint is more stringent, the public cost decreases and the private cost increases. As suggested by the figure V.4, there is a strong complementarity between the two costs : regions where the public cost strongly decreases are those where the private cost strongly growths. Typically, the four regions (in white on figure V.6) which have historically a strong specialization in arable lands are the more affected by the "green" policies. Hence, for the strongest biodiversity targets, they generate a public benefit.

5. Discussion

5.1. The bio-economic trade-off

The bio-economic model developed in this study leads to design optimal policies with respect to budgetary and biodiversity constraints. The optimal strategies maximize the aggregated intertemporal farming income or equivalently minimize the (global) private cost under a biodiversity target with a non increasing budget. The cost-effective analysis of the policies with different objectives of biodiversity provides bio-economic efficiency curves. As stressed by figure V.2 for the tested biodiversity indicators, the bio-economic trade-off is strictly negative. This suggests that integrating biodiversity goals in agricultural policies entails a loss of earnings for the farmers as in Barraquand & Martinet (2011), Drechsler *et al.* (2007), Lewis *et al.* (2011), Polasky *et al.* (2005).

However according to the biodiversity indicator, the shape of the efficiency curve slightly differs (fig. V.2). The curves displayed in the litterature (Barraquand & Martinet, 2011, Polasky *et al.*, 2005) are concave with a change of slope for high levels of the ecological score. We recover this pattern for the Community Trophic Index. In this context, it is so possible to moderately improve the CTI without strong private costs for farming. The strongest biodiversity requirements implies a major decrease of the farmer's incomes. Such a change is explained by a switch in the incentives (as exhibited in tab. V.4) : the strongest CTI goals impose a change in the optimum incentive set with smaller subsidies As regards the FBI, the trade-off is clearly more linear. This is explained by the improvement of the FBI with a continous decrease of crop incentives (as detailed in tab. V.3). With the second shape, it is not possible to improve the biodiversity performance, even moderatly, without strongly affect the farmers. The diversity of these efficiency curves stresses the difficulty to select a policy among the optimal ones and get out from the bio-economic trade-off.

5.2. A second dividend of policies with biodiversity goals

The first dividend of policies with biodiversity goals is obviously the improvment of biodiversity performances. But public and social costs gives insight on a second dividend. First, it turns out that the total cost does not rise in response to biodiversity requirements. This suggests that the biodiversity is not penalizing for the whole (macro) economic performances. Second, such an assertion is reinforced with the study of the public cost. We observe that, for both biodiversity indicators, the increase of biodiversity objectives leads to a decrease of the farming public budget. Therefore, it is possible to improve biodiversity performances without altering the public budget. In other words, the policies with demanding biodiversity goals yield a budgetary benefit which can be interpreted as a second dividend. This budgetary margin could be redistributed to the agents (farmers) in order to improve their private income. By reducing their private cost, their acceptability for adopting biodiversity goals in agricultural policies should be favored.

5.3. Regional redistribution of the second dividend

However, this financial redistribution of the public margin questions the equity between the agents, or the spatial scale of the redistribution. The regional analysis of the different costs provides a first answer to the second dividend redistribution. Indeed, the study shows that the stability of the total cost with respect to the biodiversity target also occurs at the regional scale. The policies do not affect all the regions with the same intensity but a gain between public and private costs is obtained for each region. As the regions with private losses are also those where the public cost decreases, a first redistribution mechanism emerges at the regional scale.

5.4. Perspectives and limitations

The objective of this study is to examine to role played by biodiversity goals on agricultural policies and symetrically to help conservation biology to take into account socio-economic issues. In this vein, ecological-economic modelling is a fruitful framework to bring together social and natural sciences in order to tackle biodiversity management issues (Cooke *et al.*, 2009) especially within agro-ecological and terrestrial context. By stylizing the agro-ecological system, this kind of modelling leads both to improve its understanding and to reinforce decison making support by fostering the policy effectiveness. The integration of dynamics and spatialization of the processes taken into account stresses the relevance of their use. Moreover, the relative simplicity of the initial mechanisms underlying the model together with its multi-scale perspective should make it easily transferable to other case-studies and other biodiversity taxa.

However, the results presented in this paper should be viewed as suggestive rather than predictive elements. Some improvements could have a positive impact on the design of relevant policies and should be integrated in future developments. The account of more explicit spatial processes within the bio-economic model should reinforce the derived assertions. For example, accounting for the level of landscape fragmentation which affects both biodiversity dynamics (Tscharntke *et al.*, 2005) and agricultural and-use policies (Hartig & Drechsler, 2009, Polasky *et al.*, 2008) should be a fruitful task. From the economic viewpoint, it would be accurate to account for price mechanisms. Typically, future profitabilities of the agricultural activites can vary according to the influence of fuel price or technical progress. Finally, allowing for dynamic incentives instead of fixed incentives could be a relevant way to improve the effectiveness of agricultural strategies as in Hartig & Drechsler (2009).

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Chapitre VI

Co-viability of farmland biodiversity and agriculture

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In prep for Biological Conservation

Abstract

Significant declines of farmland biodiversity have been reported in Europe since several decades. Agricultural changes have been identified as a main driver of this erosion. Although different agri-environmental schemes have been implemented, their positive role on biodiversity remain controversial. This questions the way to reconcile farming production and biodiversity conservation in order to operationalize a sustainable and multifunctional agriculture. To deal with such issues, the present paper proposes a bio-economic model and an analysis based on a co-viability perspective. The model couples stochastic dynamics of both biodiversity and farming land-uses selected at micro level with public policies at macro level based on financial incentives (taxes or subsidies) for land-uses. The co-viability approach allows to evaluate bio-economic risks for these public incentives through the probability to satisfy a mix of biodiversity and economic constraints throughout time. The model is calibrated and applied to metropolitan France at PRA ("petite région agricole") scale using a community of 34 common birds. The viable kernel allows to identify different public policies and scenarios with tolerable agro-ecological risk. It suggests how some combinations of taxes on cereals and subsidies on grasslands could be relevant. Moreover, the flexibility and multi-criteria viewpoint underlying the approach can be fruitful for decision makers in the perspective of adaptive management.

Keywords : Bio-economics, modeling, Viability, Bird, Farming, Land-use.

1. Introduction

Significant declines of biodiversity have been reported worldwide for several decades (Butchart *et al.*, 2010). This is especially documented in Europe for mammals in Flowerdew & Kirkwood (1997), for arthropods and plants in Sotherton & Self (2000) or for birds in Donald *et al.* (2001). Such erosion is mainly due to a combination of habitat loss and degradation of habitat quality altering the nesting success and/or survival rates (Benton *et al.*, 2002). Modern agriculture and associated intensification of practices have been identified as major drivers of this erosion in farmland biodiversity (Krebs *et al.*, 1999, Jiguet, 2010). In this context, the need to reconcile in a sustainable way economic and conservation objectives for agriculture is a major issue. The importance of the public policies to achieve this goal has been highlighted in Alavalapati *et al.* (2002), Shi & Gill (2005), Mouysset *et al.* (2011). The public policies can potentially modify the farmer's choices in terms of land-uses and practices and thus impact both the habitat and the dynamics of biodiversity (Doherty *et al.*, 1999, Holzkamper & Seppelt, 2007, Rashford *et al.*, 2008). In this perspective, many public policies including agri-environmental schemes have been proposed by decision makers. However, fifteen years after the initial implementation of such instruments at a large scale, their ability to enhance biodiversity remains controversial (Vickery *et al.*, 2004, Kleijn *et al.*, 2006, Butler *et al.*, 2009). Thus the conservation and the sustainable management of farmland biodiversity still constitute difficult scientific challenges.

To address such agro-ecological issues, the use of bio-economic models can be fruitful. Different bio-economic modeling frameworks are proposed in the literature. Many rely on the optimal-control theory (Polasky *et al.*, 2005, Drechsler *et al.*, 2007, Holzkamper & Seppelt, 2007). In particular, cost-benefit methods require quantification of biodiversity in monetary terms (Drechsler, 2001, Rashford *et al.*, 2008). Although pricing techniques such as contingent valuation are available, their suitability for biodiversity is disputed (Diamond & Hausman, 1994). In this context, cost-effectiveness is an interesting alternative to avoid monetary evaluation of environmental goods (Gatto & De Leo, 2000). Approaches such as ecological economics suggest studying environmental and economic performances simultaneously, stressing the relevance of multi-criteria approaches (Drechsler *et al.*, 2007, Mouysset *et al.*, 2011). However the choice of metrics for evaluating biodiversity is not straightforward and indicators used to assess biodiversity and environmental services are highly diverse (van Wenum *et al.*, 2004, Havlik *et al.*, 2005, Polasky *et al.*, 2005, Mouysset *et al.*, 2012a). Moreover, numerous models emphasize spatial dimensions in dealing with agro-ecological issues. Such spatially explicit models aim at assessing consequences of different land use patterns for various environmental and economic criteria (Irwin & Geoghegan, 2001, Swihart *et al.*, 2003, Polasky *et al.*, 2005, Groot *et al.*, 2007). Nevertheless, most of these models are static, restricting the potential ecological processes accounted for. In the same vein, most of these models are deterministic and do not take into account the various uncertainties involved in the ecological and economic processes at play.

The bio-economic model proposed in this article is in direct line with these considerations. First, adopting a spatio-temporal and multi-scale viewpoint, it articulates biodiversity community dynamics with farming land-uses selected by heteregenous agents at micro (landscape) level and macro (typically national) financial incentives associated with land-uses. Moreover the model accounts for bio-economic uncertainties through stochasticity both on community dynamics and gross margins. Second a viability or co-viability approach is applied to the bio-economic model to identify public policies promoting a multifunctional and sustainable agriculture.

The viability (or viable control) approach aims at identifying desirable combinations of states and associated controls that ensure the 'good health', safety or effectiveness of the system (Aubin, 1991, Béné *et al.*, 2001). This approach focuses on feasible paths within a set of desirable objectives or constraints. This framework has been applied to renewable resources management in Béné *et al.* (2001), Martinet *et al.* (2007), De Lara *et al.* (2007), Péreau *et al.* (2012), as well as to broader (eco)-system dynamics (Cury *et al.*, 2005, Doyen *et al.*, 2007; 2012). Specific focus on agro-ecological issues can be found in Tichit *et al.* (2007), Sabatier *et al.* (2012). In bio-economic contexts, viability does not aim at identifying optimal or steady state paths for the co-dynamics of resources and exploitation but, instead, provides acceptable trajectories and controls satisfying both socio-economic and ecological constraints. In this respect, it is a multi-criteria approach (Baumgärtner & Quaas, 2009). Moreover, by identifying the conditions that allow desirable objectives to be fulfilled over time, considering both present and future states of a system, the viability approach conveys information on sustainability. In particular, the approach is also closely related to the maximin, or Rawlsian, approach with respect to intergenerational equity (Martinet & Doyen, 2007). Furthermore, Tichit *et al.* (2007), De Lara *et al.* (2007), De Lara & Doyen (2008), Doyen *et al.* (2012) shows how the so-called Population Viability Analysis (PVA) developed in conservation biology addresses issues comparable to those of the viability approach. Cury *et al.* (2005), Doyen *et al.* (2012) illustrate how the viability approach can potentially be useful to integrate ecosystem considerations into management.

This viability or co-viability approach is here applied to a complex agro-ecological model in order to assess the possible reconciliation between conservation goals and economic requirements. The co-viability approach allows to evaluate the bio-economic risks for the public incentives through the probability to satisfy a set of ecological and economic constraints throughout time. By ecological constraint is meant guaranteed levels for different biodiversity indicators. The method is applied to the metropolitan France case study. The calibration relies on French time series of 34 birds abundance and 14 farming land-uses over years 2002-2009 and 620 small agricultural regions in metropolitan France. Three indicators which has been identified in (Mouysset *et al.*, 2012a) as relevant to characterize the state of bird community in response of agricultural public policies capture the biodiversity scores : the Farmland Bird Index which has been adopted by the European Union (Balmford *et al.*, 2003, Gregory *et al.*, 2004), the Community Specialization Index which evaluates the dependence of the community of a habitat (Barnagaud *et al.*, 2011) and the Community Trophic Index based on the species diets (Pauly *et al.*, 1998).

2. The bio-economic modeling

2.1. Context and data

Metropolitan France is split into 620 small agricultural regions (PRA for Petites Regions Agricoles). A PRA is part of a department (a major French administrative entity) which exhibits an agro-ecological homogeneity. This consistency from both the ecological and economic points of view makes the PRA scale well suited for our bio-economic modeling. The model described below is built for each PRA.

To assess the ecological performance, we here choose to focus on common bird populations and related indicators (Gregory *et al.*, 2004). Although the metric and the characterization of biodiversity remain an open debate (MEA, 2005), such a choice is justified for several reasons (Ormerod & Watkinson, 2000) : (i) Birds lie at a high level in the trophic food chains and thus capture the variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation and viability (Sekercioglu *et al.*, 2004). (iii) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens.

The STOC (French Bird Breeding Survey) database [1] provided the informations related to the bird abundances across the whole country. Abundance values for each species were available for the period 2002-2008.

1. See the Vigie-Nature website http ://www2.mnhn.fr/vigie-nature/. Standardized monitoring of spring-breeding birds at 1747 $2 * 2\, km^2$ plots across the whole country. Details of the monitoring method and sampling design can be found in Jiguet (2009).

20 farmland bird species	14 generalist bird species
(1) Buzzard *Buteo buteo*	(1) Blackbird *Turdus merula*
(2) Cirl Bunting *Emberiza cirlus*	(2) Blackcap *Sylvia atricapilla*
(3) Corn Bunting *Emberiza calandra*	(3) Blue Tit *Parus caeruleus*
(4) Grey Partridge *Perdix perdix*	(4) Carrion crow *Corvus corone*
(5) Hoopoe *Upupa epops*	(5) Chaffinch *Fringilla coelebs*
(6) Kestrel *Falco tinnunculus*	(6) Cuckoo *Cuculus canorus*
(7) Lapwing *Vanellus vanellus*	(7) Dunnock *Prunella modularis*
(8) Linnet *Carduelis cannabina*	(8) Great Tit *Parus major*
(9) Meadow Pipit *Anthus pratensis*	(9) Green Woodpecker *Picus viridis*
(10) Quail *Coturnix coturnix*	(10) Golden oriole *Oriolus oriolus*
(11) Red-backed Shrike *Lanius collurio*	(11) Jay *Garrulus glandarius*
(12) Red-legged Partridge *Alectoris rufa*	(12) Melodius Warbler *Hippolais polyglotta*
(13) Rook *Corvus frugilegus*	(13) Nightingale *Luscinia megarhynchos*
(14) Skylark *Alauda arvensis*	(14) Wood Pigeon *Columba palumbus*
(15) Stonechat *Saxicola torquatus*	
(16) Whinchat *Saxicola rubetra*	
(17) Whitethroat *Sylvia communis*	
(18) Wood Lark *Lullula arborea*	
(19) Yellowhammer *Emberiza citrinella*	
(20) Yellow Wagtail *Motacilla flava*	

Table VI.1 – List of the 20 farmland and 14 generalist bird species s

For each species, we further performed a spatial interpolation of these abundance data to obtain relative abundance values for each possible square in the country (e.g. 136 000 squares) using kriging models based on spatial autocorrelation and the exponential function (Doxa *et al.*, 2010). We then averaged the abundance values at the PRA scale. Among the species monitored by this large-scale long-term survey, we selected 14 generalist species and 20 farmland specialist species which have been classified according to their habitat requirements at a Europe scale (European Bird Census Council, 2007). Table VI.1 lists the 14 generalist species and the 20 farmland specialist species used as a reference for the European Farmland Bird Index FBI (Gregory *et al.*, 2004). Previous analyses have shown the relevance of the national FBI to reflect the response of farmland biodiversity to agricultural intensification (Doxa *et al.*, 2010, Mouysset *et al.*, 2012a).

For agro-economic data, we use the French agro-economic classification OTEX (orientation technico-economique) developed by the French Farm Accounting Data Network (FADN) [1] and the Observatory of Rural Development (ODR) [2]. This organization distinguishes 14 classes of land-uses denoted by OTEA (see tab. VI.2). Each PRA is a specific combination of these OTEA. The surfaces dedicated to each of the 14 OTEA and the associated gross margins relying on tax return, for the years 2002 to 2008 are available on the ODR website. The budgetary constraint is calibrated with the current French Common Agricultural Policy budget.

As depicted by figure VI.1, the bio-economic model is composed of three compartments with a multi-scale perspective as in Mouysset *et al.* (2011) : the public policy at large (national) scale interacts with the farming land-uses and biodiversity dynamics at region (PRA) scale.

2.2. The biodiversity model

The biodiversity model deals with a community of species instead of focusing on emblematic species. It is based on population dynamics with intra-specific competition depending on habitat and especially on farming land-use. A Beverton-Holt function is selected for sake of simplicity. It captures intra-specific competition

1. http ://ec.europa.eu/agriculture/rica/
2. https ://esrcarto.supagro.inra.fr/intranet/

The 14 land-uses (OTEA) k
(1) Cereal, Oleaginous, Proteaginous (COP)
(2) Variegated crops
(3) Intensive bovine livestock breeding
(4) Medium bovine livestock breeding
(5) Extensive bovine livestock breeding
(6) Mixed crop-livestock farming with herbivorous direction
(7) Other herbivorous livestock breeding
(8) Mixed crop-livestock farming with granivorous direction
(9) Mixed crop-livestock farming with other direction
(10) Granivorous livestock breeding
(11) Permanent farming
(12) Flower farming
(13) Viticulture
(14) Others associations

Table VI.2 – List of the 14 farming land-uses (OTEA)

through a carrying capacity parameter as follows :

$$N_{s,r}(t+1) = N_{s,r}(t).(1+R_{s,r})\left(1+\frac{N_{s,r}(t)}{M_{s,r}(t)}\right)^{-1} \tag{VI.1}$$

where $N_{s,r}(t)$ stands for the abundance of species s in region r at year t. The $R_{s,r}$ coefficient corresponds to the intrinsic growth rate specific to a given species s in region r. The product $M_{s,k}(t) * R_{s,r}$ represents the carrying capacity of the habitat r and the value $M_{s,k}(t)$ captures the ability of the habitat r to host the species s. The habitat parameter $M_{s,r}(t)$ is assumed to depend on the farming land-uses $A_{r,k}(t)$ as follows :

$$M_{s,r}(t) = \beta_{s,r} + \sum_k \alpha_{s,r,k}.A_{r,k}(t) \tag{VI.2}$$

where $A_{r,k}(t)$ represents the shares of the Utilized Agricultural Areas of the region r dedicated to the agricultural system k. Consequently, the $\alpha_{s,r}$ and $\beta_{s,r}$ coefficients, specific to each species, inform on how such species s responds to agricultural land-use k in a region r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a region r and integrates others factors such as the proportion of forests or urban areas. The coefficients $\alpha_{s,r}$ and $\beta_{s,r}$ are calibrated according to a least-square method (see Mouysset *et al.* (2012b)).

The population size $\tilde{N}_{s,r}(t+1)$ is estimated with the population size computed by the model $N_{s,r}(t+1) = f(N_{s,r}(t))$ and a uncertainty coefficient $\vartheta_{s,r}(t)$:

$$\tilde{N}_{s,r}(t+1) = f(N_{s,r}(t)) + \vartheta_{s,r}(t) \tag{VI.3}$$

The random variables $\vartheta_{s,r}(t)$ captures the ecological stochasticy affecting the dynamics. The variables $\vartheta_{s,r}(t)$ are coming from Gaussian distributions calibrated with the variance of the historical abundances compared to the computed abundances in the time series :

2.3. The economic model

Each region is represented by a standard agent. Agent income in region r at year t denoted by $Inc_r(t)$ relies on the gross margin per unit of scale $gm_{r,k}(t)$, current proportions of the Utilized Agricultural Area (UAA) dedicated to the agricultural land-uses $A_{r,k}(t)$ and incentives τ_k (taxes with $\tau_k < 0$ or subsidies with $\tau_k > 0$) which takes form of a percentage of gross margins as follows :

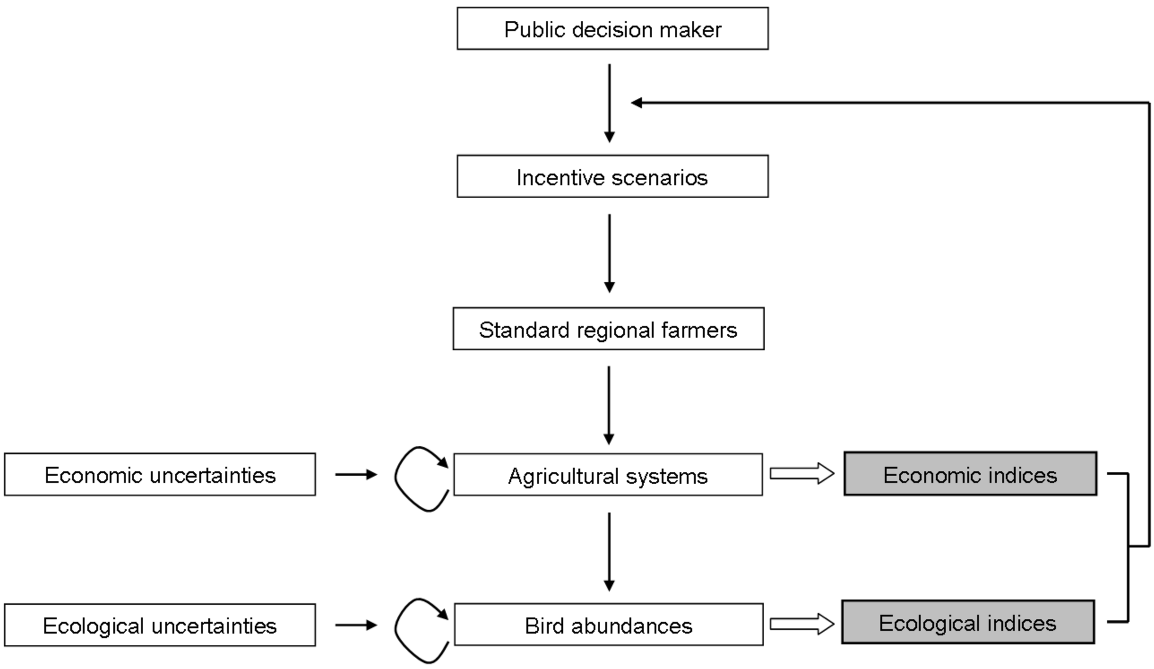


FIGURE VI.1 – Bio-economic model coupling. The decision maker determines an incentive scenarios which affect farmer's decisions. The farmers choose their agricultural systems by maximizing their utility function under technical constraints. These choices affect the habitat and the bird communities.

$$Inc_r(t) = \sum_k gm_{r,k}(t).A_{r,k}(t).(1+\tau_k) \tag{VI.4}$$

Gross margins $gm_{r,k}(t)$ are supposed to be uncertain. The variability on gross margins includes both market, production and climate uncertainties. A Gaussian distribution parametrized with the mean and the covariance matrix of the historical data is chosen to capture such uncertainties. Also assumed is a quadratic form for the utility function of the representative agent (Lien 2002). Hence, the utility $U_r(t)$ for the representative farmer corresponds to the difference between an expected income $\mathbb{E}[Inc_r(t)]$ and its risky part $\mathbb{V}ar[Inc_r(t)]$:

$$U_r(t) \quad = \quad \mathbb{E}[Inc_r(t)] - a.\mathbb{V}ar[Inc_r(t)] \tag{VI.5}$$

with

$$\mathbb{E}[Inc_r(t)] \quad = \quad \sum_k \overline{gm}_{r,k}.A_{r,k}(t).(1+\tau_k) \tag{VI.6}$$

$$\mathbb{V}ar[Inc_r(t)] \quad = \quad \sum_k \sum_{k'} \sigma_{r,k,k'}(t).A_{r,k}(t).A_{r,k'}(t).(1+\tau_k).(1+\tau_{k'}) \tag{VI.7}$$

Expected gross margins $\overline{gm}_{r,k}$ are the mean of the 7 historical years [1]. The coefficient a represents the risk aversion level of the farmer : the higher the a, more risk-averse the farmer. The risky term is computed with the covariance [2] $\sigma_{r,k,k'}$ between margins of land-uses k and k' in region r. For each year t, the regional

1. $\overline{gm}_{r,k} = \frac{1}{7}\sum_{t=1}^{t=7} gm_{r,k}(t)$.

2. $\sigma_{r,k,k'} = \frac{1}{7}\sum_{t=1}^{t=7}(gm_{r,k}(t) - \overline{gm}_{r,k}(t)).(gm_{r,k'}(t) - \overline{gm}_{r,k'}(t))$.

standard agents choose their agricultural land-uses $A_{r,k}(t)$ in order to maximize their income utility in an uncertain context (eq. VI.8) according to rigidity (eq. VI.9) and rigidity constraints (eq. VI.10). This approach refers to the framework of maximization under constraints as in Polasky *et al.* (2005), Pacini *et al.* (2004), Drechsler *et al.* (2007), Mouysset *et al.* (2011).

$$\max_{A_{r,1};\ldots;A_{r,14}} U_r(t) \tag{VI.8}$$

When maximizing the utility, the standard agent must comply with two constraints at every point in time :

$$|A_{r,k}(t) - A_{r,k}(t-1)| \leq \varepsilon.A_{r,k}(t-1) \tag{VI.9}$$

$$\sum_k A_{r,k}(t) = UAA_r(t_0) \tag{VI.10}$$

The rigidity constraint (eq. VI.9) restricts the area that the farmer can modify at each time for each agricultural system k. The parameter ε captures change costs or inertia. The constraint (eq. VI.10) ensures that the total utilized agricultural area (UAA) is kept fixed. The parameters a and ε are calibrated according to a least-square method (see Mouysset *et al.* (2012b)).

2.4. The bio-economic indicators

2.4.1. The ecological indicators

The indicators used to assess the ecological performances are computed through the abundances $\tilde{N}_{s,r}(t)$ of the species at play. As suggested by Mouysset *et al.* (2012a), the community are analyzed through the combination of the Farmland Bird Index (FBI), the Community Specialization Index (CSI) and the Community Trophic Index (CTI). The Farmland Bird Index, which measures the growth of the farmland specialist community, has been adopted by the European Community as the official environmental index especially to analyze structural changes in biodiversity (Balmford *et al.*, 2003). The relevance of the FBI to reflect the response of farmland biodiversity to agriculture intensification has been shown in Doxa *et al.* (2010), Mouysset *et al.* (2012a). In this aggregated index, the abundances variation of each species is taken into account similarly, independently from the abundance value. We first estimated a national population index for each species from the abundances values of all PRA r (eq. VI.11), then we calculated the aggregated indicator $\mathrm{FBI}_{\mathrm{Nat}}$ (eq. VI.12).

$$\tilde{N}_{s,Nat}(t) = \sum_r \tilde{N}_{s,r}(t) \tag{VI.11}$$

$$FBI_{Nat}(t) = \prod_{s\,\in\,Specialist} \left(\frac{\tilde{N}_{s,Nat}(t)}{\tilde{N}_{s,Nat}(2008)}\right)^{1/20} \tag{VI.12}$$

The Community Trophic Index (CTI) informs on the average trophic level of a community as in Pauly *et al.* (1998), Mouysset *et al.* (2012a). The CTI integrates both the generalist species and the farmland specialist species (table VI.1). This indicator classifies the communities with more granivorous species (e.g. low trophic level) against the communities with more insectivorous and carnivorous species (e.g. high trophic level) (Mouysset *et al.*, 2012a). It is computed as the weighted arithmetic mean of the exponential of the species trophic level balanced by the abundances (eq. VI.13 and VI.14). n exponential function is used to better contrast communities with or without bird individuals of the higher trophic levels. National $\mathrm{CTI}_{\mathrm{Nat}}$ is the arithmetic mean of the 620 regional $\mathrm{CTI}_{\mathrm{r}}$ (eq. VI.15).

$$\tilde{N}_{tot,r}(t) = \sum_{s} N_{s,r}(t) \tag{VI.13}$$

$$CTI_r(t) = \sum_{s} \frac{\tilde{N}_{s,r}(t)}{\tilde{N}_{tot,r}(t)} \, . \exp(\mathrm{STI_s}) \tag{VI.14}$$

$$CTI_{Nat}(t) = \frac{1}{620} . \sum_{r} CTI_r(t) \tag{VI.15}$$

Finally, the Community Specialization Index (CSI) leads to interpret the response of the composition of local bird communities to agricultural pressures. A habitat specialization species index (SSI) has been computed for each species, reporting the coefficient of variation of the abundance of a species across 18 habitat categories (see Julliard *et al.* (2006)). This index measures the average degree of habitat specialization among the individuals of the community. It leads to discriminating the ordinary community of generalist species, which are more resilient to perturbation, from the specialized communities with more specialist species, which are especially sensitive to global change (Julliard *et al.*, 2006). For each square, the local $\mathrm{CSI_r}$ is then calculated as the arithmetic mean of the species specialisation index weighted by the abundances (eq. VI.13 and VI.16). National $\mathrm{CSI_{Nat}}$ is the arithmetic mean of the 620 regional $\mathrm{CSI_r}$ (eq. VI.17).

$$CSI_r(t) = \sum_{s} \frac{\tilde{N}_{s,r}(t)}{\tilde{N}_{tot,r}(t)} \, . \, SSI_s \tag{VI.16}$$

$$CSI_{Nat}(t) = \frac{1}{620} . \sum_{r} CSI_r(t) \tag{VI.17}$$

2.4.2. The economic indicators

The economic performances of the farmers are measured with the national income which is computed as follows :

$$\overline{Inc}_{nat}(t) = \frac{1}{A_{nat}} \sum_{r=1}^{620} A_r . Inc_r(t) \tag{VI.18}$$

where $A_{nat} = \sum_{r=1}^{620} A_r$ is the total surface of PRA over France.

Hereafter, the public budget plays a major role for the viability analysis This budget $Budg(t)$ is computed according to the different incentives τ_k as follows :

$$Budg(t) = \sum_{r} \sum_{k} S_r \, . \, gm_{r,k} \, . \, A_{r,k}(t) \, . \, \tau_k \tag{VI.19}$$

3. The co-viability scenarios

We now examine the bio-economic sustainability of the agro-ecological system through a viability approach. The CVA approach (Co-Viability Analysis) considers both biodiversity and economic viability objectives through a large set of constraints that have to be satisfied. Here the constraints have to be satisfied in the probalistic sense as in De Lara & Doyen (2008), Doyen & De Lara (2010), Doyen *et al.* (2012). Given a confindent rate, the viable kernel allows to identify different public policies and scenarios with admissible agro-ecological risk. We now describe the different constraints taken into account hereafter. They basically relies on a comparaison with the Statu Quo (SQ) scenario which corresponds to the performances obtained if we fixed the land-uses to the 2008 land-uses. In others words :

$$A^{\text{SQ}}_{r,k}(t) = A_{r,k}(2008), \quad t = 2009, \dots, T \tag{VI.20}$$

3.1. Ecological constraints

By ecological constraints is meant guaranteed levels for different biodiversity indices. Three indicators which has been identified in (Mouysset *et al.*, 2012a) as relevant to characterize the state of bird community in response of agricultural public policies capture the biodiversity scores : the Farmland Bird Index $FBI(t)$, the Community Specialization Index $CSI(t)$ and the Community Trophic Index $CTI(t)$. The lower thresholds for the constraints are based on the performances $FBI^{SQ}(t)$, $CTI^{SQ}(t)$, $CSI^{SQ}(t)$ obtained with a Statu Quo (SQ) scenario as defined by relation (VI.20).

$$FBI(t) \geq \lambda * FBI^{SQ}(t) \tag{VI.21}$$
$$CTI(t) \geq \lambda * CTI^{SQ}(t) \tag{VI.22}$$
$$CSI(t) \geq \lambda * CSI^{SQ}(t) \tag{VI.23}$$

where the rate λ measures the strength of the constraint. Typically, three levels are tested among $\lambda = 0.95$, $\lambda = 0.97$ and $\lambda = 1$.

3.2. Economic constraints

Similarly, from the economic viewpoint, viability requires that the incomes $Inc(t)$ derived from farming activities are not worst than the current or SQ level $Inc^{SQ}(t)$ derived from relation (VI.20) :

$$Inc(t) \geq \lambda * Inc^{SQ}(t) \tag{VI.24}$$

It is also required that the public policy scenarios are always complying with a budgetary rule (eq. VI.25).

$$Budg(t) \leq Budg(2008) \tag{VI.25}$$

In other words, the spending public budget at each time t does not exceed the spending current budget at time 2008.

3.3. Viable incentives

The public policy in this study are based on incentives τ (subsidies and taxes) distributed to the different agricultural system k. Bio-economic performances in response to these scenarios are computed from 2009 to 2050 according to the ecological and economic models described above.

For accelerating the numerical computations, the public decision variables τ are restricted to only two incentives : the cereal incentive τ_{cop} is dedicated to arable lands (Otex (1) in table I.2) and the grassland incentive τ_{grass} is applied to non-intensive grassland systems (Otex (4), (5), (6), (7) in table VI.2).

We choose to deal with uncertainty in a probabilistic sense. We therefore perform a stochastic viability analysis. For this, we consider a probability $\mathbb{P}$ on scenarios $\omega(.) \in \Omega$. Then, for all public policy scenarios τ, we compute the probability to satisfy the different bio-economic constraints :

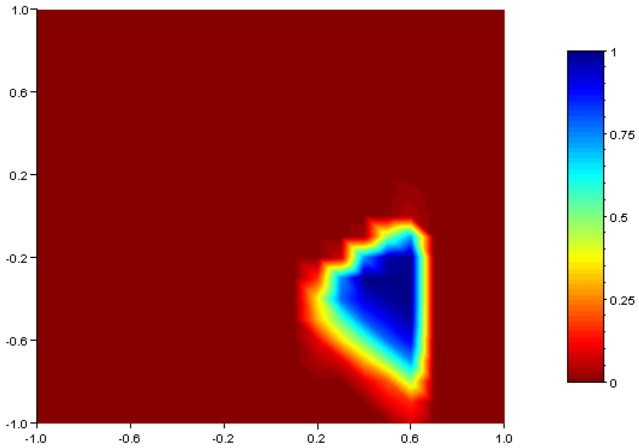


FIGURE VI.2 – Probability of the different public policies scenarios to satisfy the set of constraints (budgetary, income, FBI, CTI and CSI constraints) with $\lambda = 0.95$ in an uncertain context. The x-axis stands for the τ_{grass} incentives and the y-axis for the τ_{cop} incentives. In strong blue, the probability is 1, in dark red, the probability is 0.

$$\text{CVA}(\tau) = \mathbb{P}\Big(\text{ Constraints } (VI.21), (VI.22), (VI.23), VI.24) \text{ and } (VI.25), \quad \text{holds for } t = 2009...2050\Big) \tag{VI.26}$$

Given a confidence rate $\delta \in]0, 1]$, we aim at identifying the controls τ_{cop} and τ_{grass}) that satisfy the following condition [1] :

$$\text{CVA}(\tau) \geq \delta \tag{VI.27}$$

In terms of decision, given a level of risk $1-\delta$, we aim at identifying viable incentives, namely $\tau = (\tau_{cop}, \tau_{grass})$ that satisfy viability condition (VI.26). In this context, of particular interest are the controls that maximize the viability probabilities, that is $\max_\tau \text{CVA}(\tau)$. Given a level of risk $1 - \delta$, we define the viable decision kernel $\mathcal{T}^{\delta}_{viab}$ by :

$$\mathcal{T}^{\delta}_{viab} = \{\tau, \ \text{CVA}(\tau) \geq \delta\} \tag{VI.28}$$

As we are in uncertain context, the outcomes of every incentive scenario $\tau = (\tau_{grass},\tau_{cop})$ are generated 100 times to approximate its viability probability $\text{CVA}(\tau)$. Gaussian and i.i.d. assumptions are considered for random variables $\vartheta(t)$ and $gm(t)$ introduced in eq. (VI.4) and eq. (VI.1) respectively.

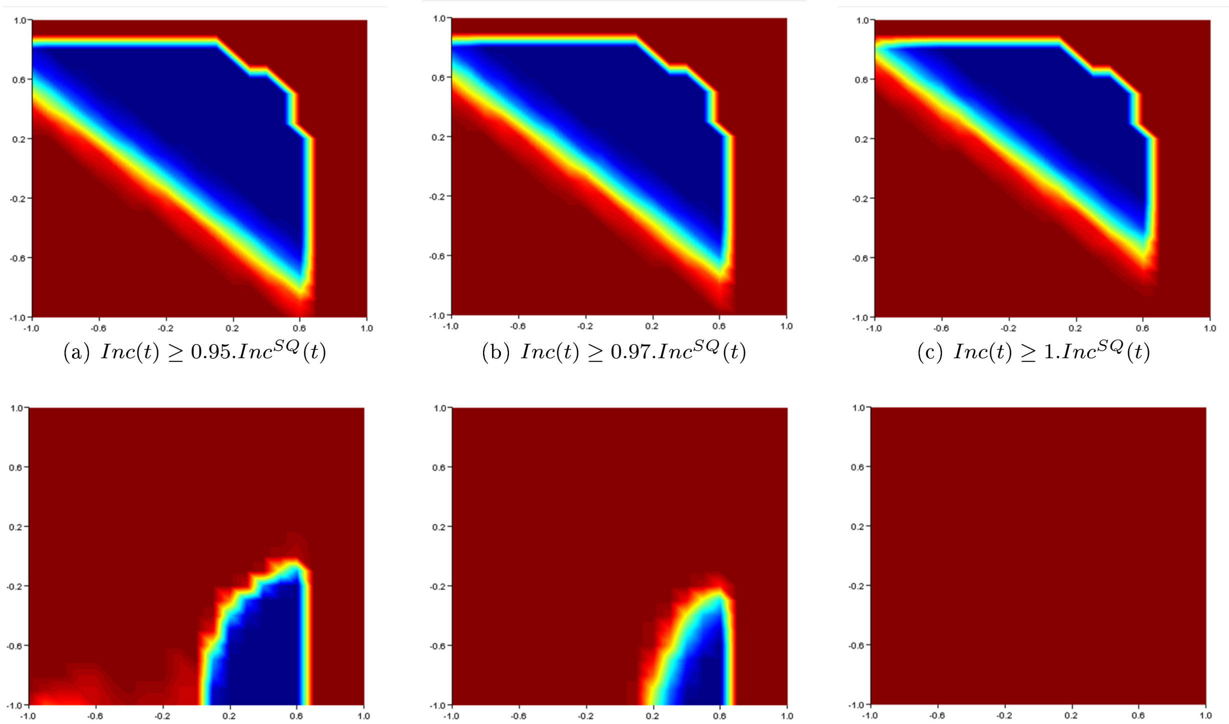
4. Results

4.1. Public policies combining bio-economic constraints

For each tested public policy $\tau = (\tau_{grass}, \tau_{cop})$, the figure VI.2 represents the probability ($\text{CVA}(\tau)$) to satisfy the constraint along the trajectories : the more blue the scenario is, the higher the probability to verify the

1. In a more formal way, stochastic viability analysis refers to the identification of the stochastic viability kernel $viab_\delta$ (De Lara & Doyen, 2008) defined as

$$viab_\delta(t_0) = \big\{N(t_0) \ \big| \ \exists \tau \ \ \text{CVA}(\tau) \geq \delta \ \ \big\} .$$



(a) $Inc(t) \geq 0.95.Inc^{SQ}(t)$ (b) $Inc(t) \geq 0.97.Inc^{SQ}(t)$ (c) $Inc(t) \geq 1.Inc^{SQ}(t)$

(d) $FBI(t) \geq 0.95.FBI^{SQ}(t)$ (e) $FBI(t) \geq 0.97.FBI^{SQ}(t)$ (f) $FBI(t) \geq 1.FBI^{SQ}(t)$

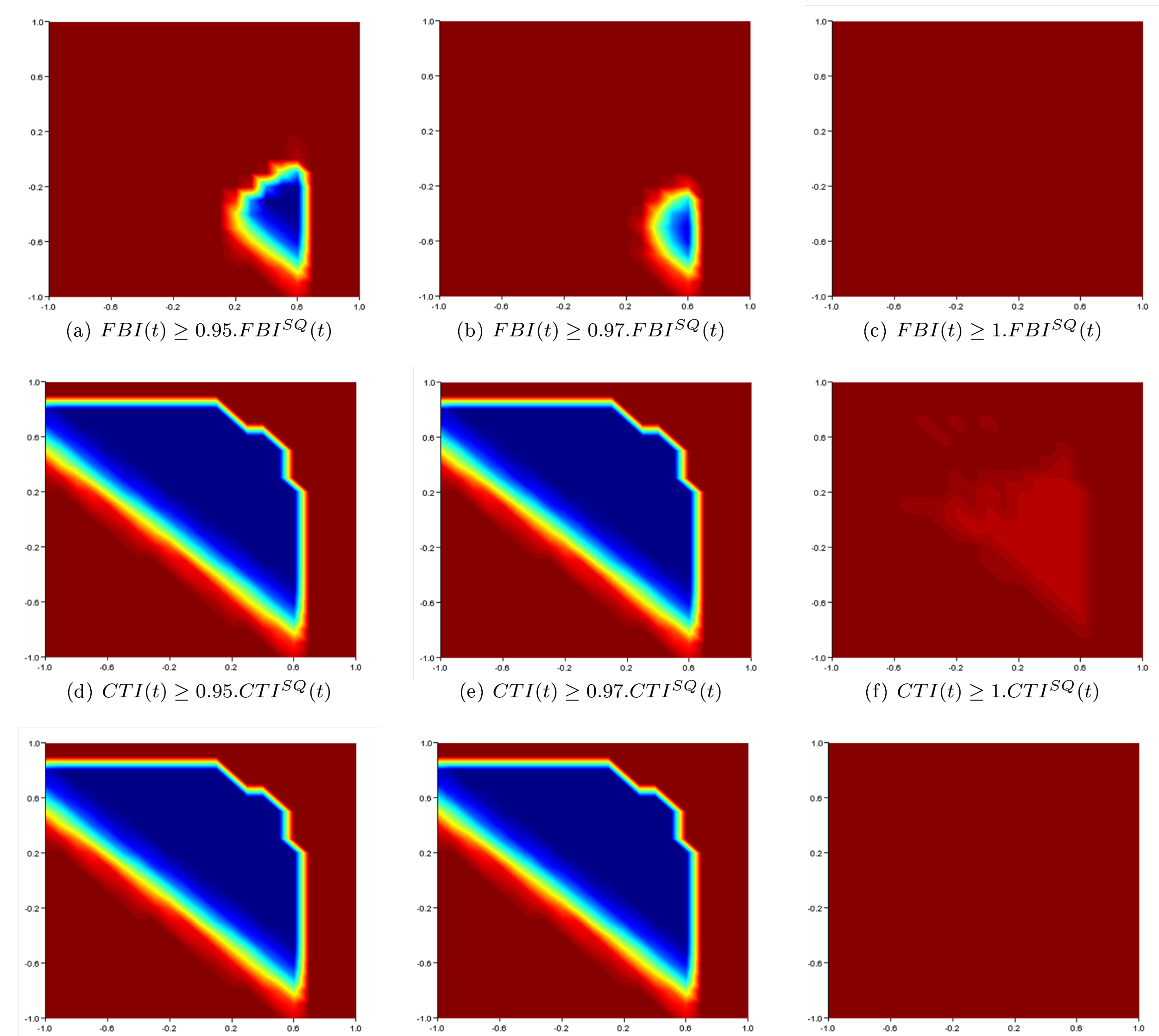
FIGURE VI.3 – Probability of the different public policies scenarios in an uncertain context to satisfy the set of constraints : the budgetary constraint and an additional constraint which differs according to the graph. The x-axis stands for the τ_{grass} incentives and the y-axis for the τ_{cop} incentives. In strong blue, the probability is 1, in dark red, the probability is 0.

constraints in an uncertain context. The figure VI.2 shows the kernel of public policies $\mathcal{T}_{viab}$ which insures the budgetary constraint and at least 0.95 % ($\lambda = 0.95$) of the bio-economic performances obtained with the Statu Quo scenario (according to the income, FBI, CTI and CSI constraints). This figure highlights that it exists a large viable set of public policies $\mathcal{T}_{viab}$ satisfying simultaneously economic and ecological performances. We observe that this viable kernel $\mathcal{T}_{viab}$ is roughly based on subsidies for non intensive grasslands ($\tau_{grass} \geq 0$) and taxes for crops ($\tau_{cop} \leq 0$). However, the incentives can relatively vary within this kernel ($0.3 \leq \tau_{grass} \leq 0.6$ and $-0.2 \leq \tau_{cop} \leq -0.6$) by keeping the same bio-economic risk. Moreover, the uncertainty management is satisfying since a part of this kernel (in strong blue) complies with the constraints with a high confidence rate ($\delta = 0.98\%$).

4.2. Bio-economic sensitivity

The figure VI.3 compares the kernel of public policies under increasing economic constraint in the one hand, and under increasing FBI constraint in the other hand. With the economic constraint only (eq. VI.24), we observe the viability kernel $\mathcal{T}_{viab}$ is the upper triangle beyond a negative trade-off between crop and grassland incentives. If the crop subsidies τ_{cop} decrease, the grassland incentives τ_{grass} have to be increased to maintain the same level of income. When the economic objective is more demanding (i.e. λ is increased in eq. VI.24), the trade-off is moved upward. This means that if we want to keep the same level of grassland subsidies, the crop incentives need to be increased to ensure the economic constraint.

With the FBI contrainst only (eq. VI.21), the set of viable public policies $\mathcal{T}_{viab}$ is broadly smaller and it corresponds to lower triangle under a positive trade-off between the two incentives (τ_{grass} and τ_{cop}) : if the crop subsidies decrease, the grassland subsidies can decrease. When the FBI constraint is more stringent



(a) $FBI(t) \geq 0.95.FBI^{SQ}(t)$ (b) $FBI(t) \geq 0.97.FBI^{SQ}(t)$ (c) $FBI(t) \geq 1.FBI^{SQ}(t)$

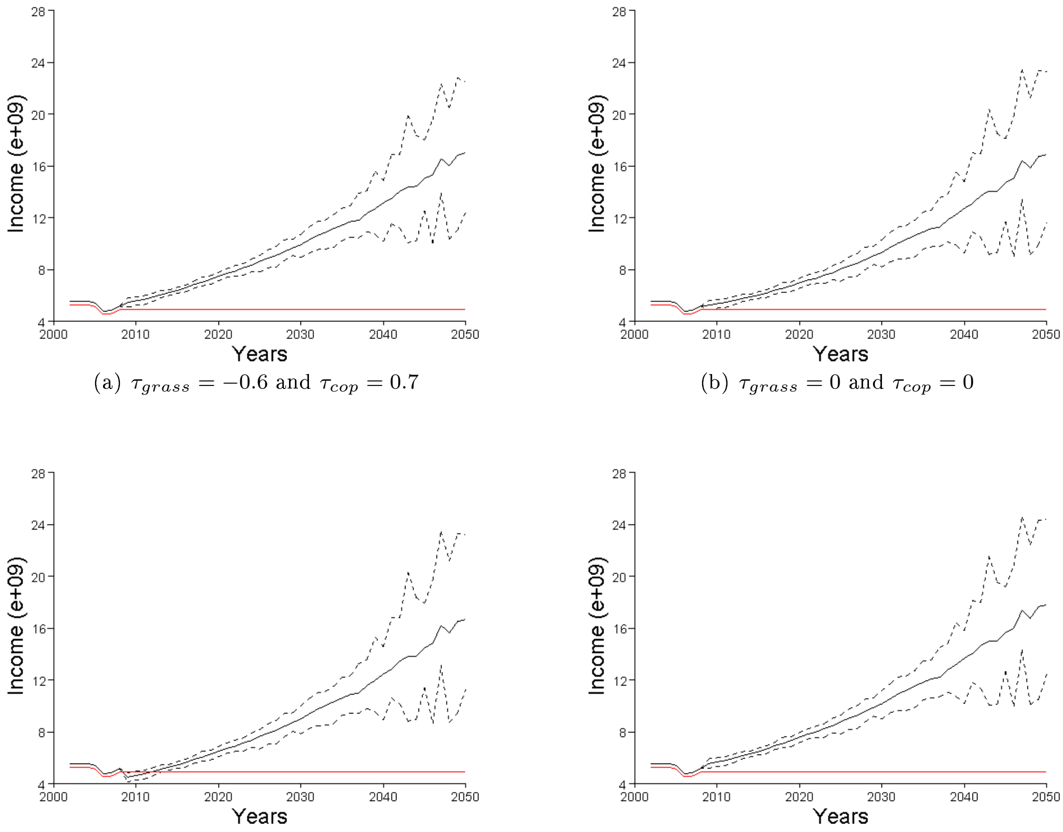
(d) $CTI(t) \geq 0.95.CTI^{SQ}(t)$ (e) $CTI(t) \geq 0.97.CTI^{SQ}(t)$ (f) $CTI(t) \geq 1.CTI^{SQ}(t)$

(g) $CSI(t) \geq 0.95.CSI^{SQ}(t)$ (h) $CSI(t) \geq 0.97.CSI^{SQ}(t)$ (i) $CSI(t) \geq 1.CSI^{SQ}(t)$

FIGURE VI.4 – Probability of the different public policies scenarios in an uncertain context to satisfy the set of constraints : the budgetary constraint, the economic constraint $Inc(t) \geq 0.95.Inc^{SQ}(t)$, and an additional ecological constraint, which differs according to the graph. The x-axis stands for the τ_{grass} incentives and the y-axis for the τ_{cop} incentives. In strong blue, the probability is 1, in dark red, the probability is 0.

(i.e. λ is increased in eq. VI.21), the positive trade-off is moved downward. In others words, to keep the same level of grassland subsidies with the FBI constraint, the crop subsidies have to be smaller (i.e. the crop taxes have to be higher).

More generally, the figure VI.3 shows that there exists viability kernels $\mathcal{T}_{viab}$ for both economic and ecological constraints. However, these kernels reduces with the viability requirements and constraint levels. These reductions are going in opposite way according to the economic or ecological constraint. Whatever the constraint, the management of risk is satisfying with the existence of public policies with a probability of success at $\lambda = 0.98$ (in strong blue) in the center of the viability kernel : the viable decision kernel $\mathcal{T}_{viab}^{0.98}$.



(a) $\tau_{grass} = -0.6$ and $\tau_{cop} = 0.7$

(b) $\tau_{grass} = 0$ and $\tau_{cop} = 0$

(c) $\tau_{grass} = -0.6$ and $\tau_{cop} = -0.2$

(d) $\tau_{grass} = 0.6$ and $\tau_{cop} = -0.2$

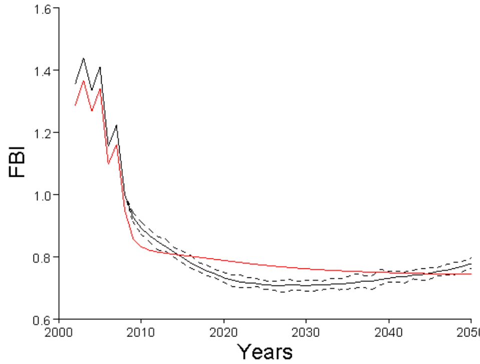
FIGURE VI.5 – Examples of income mean trajectories (black solid lines) and their min and max (black dashed lines) with four public policy scenarios. The red lines represent the constraints with $\lambda = 0.95$

4.3. Ecological sensitivity

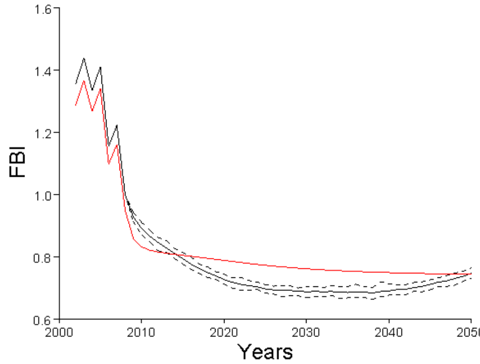
The figure VI.4 displays the viable public policies $\mathcal{T}_{viab}$ under increasing ecological constraints (i.e. λ is increased in eq. VI.21, VI.22, VI.23) keeping the same economic constraint (i.e. λ is constant in eq. VI.24). We clearly observe that the FBI constraint is the more restrictive ecological constraint. For the same strength λ of constraint, the kernel obtained with the FBI constraint is smaller than those obtained with the CTI and CSI constraints. For the intermediary constraint ($\lambda = 0.97$), the management of risk is decreased with the FBI constraint. Indeed if the viability kernel $\mathcal{T}_{viab}$ exists, the part in strong blue which insures a strong probability of success (δ=98%) is quite missing. With the more stringent constraints ($\lambda = 1$), the viability kernel $\mathcal{T}_{viab}$ is empty for the three ecological constraints. In other terms, it is not possible to maintain at least the inter-temporal bio-economic performances obtained with the Statu Quo scenario in this ecological-economic uncertain context.

4.4. Trajectories and land-uses

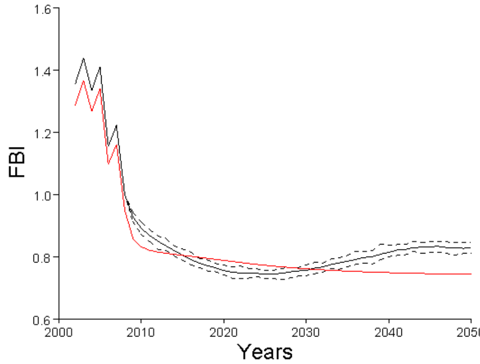
The figures VI.5 and VI.6 present four examples of trajectories with the associated constraint at 95% ($\lambda = 0.95$) for the two more stringent criteria : the income and the FBI. The income satisfies the constraint in three cases. Only the scenario with taxes on both grasslands and crops violates the constraint at the beginning of the trajectory. According to the FBI, only the scenario coupling taxes on crops and subsidies on grassland verifies the constraint. This scenario ($\tau_{grass} = 0.6$ and $\tau_{cop} = -0.2$) is the only viable public



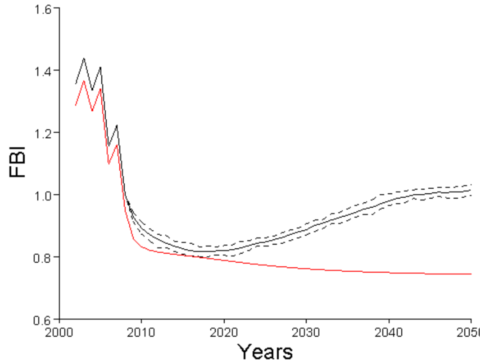
(a) $\tau_{grass} = -0.6$ and $\tau_{cop} = 0.7$



(b) $\tau_{grass} = 0$ and $\tau_{cop} = 0$



(c) $\tau_{grass} = -0.6$ and $\tau_{cop} = -0.2$



(d) $\tau_{grass} = 0.6$ and $\tau_{cop} = -0.2$

FIGURE VI.6 – Examples of FBI mean trajectories (black solid lines) and their min and max (black dashed lines) with four public policy scenarios. The red lines represent the constraints with $\lambda = 0.95$

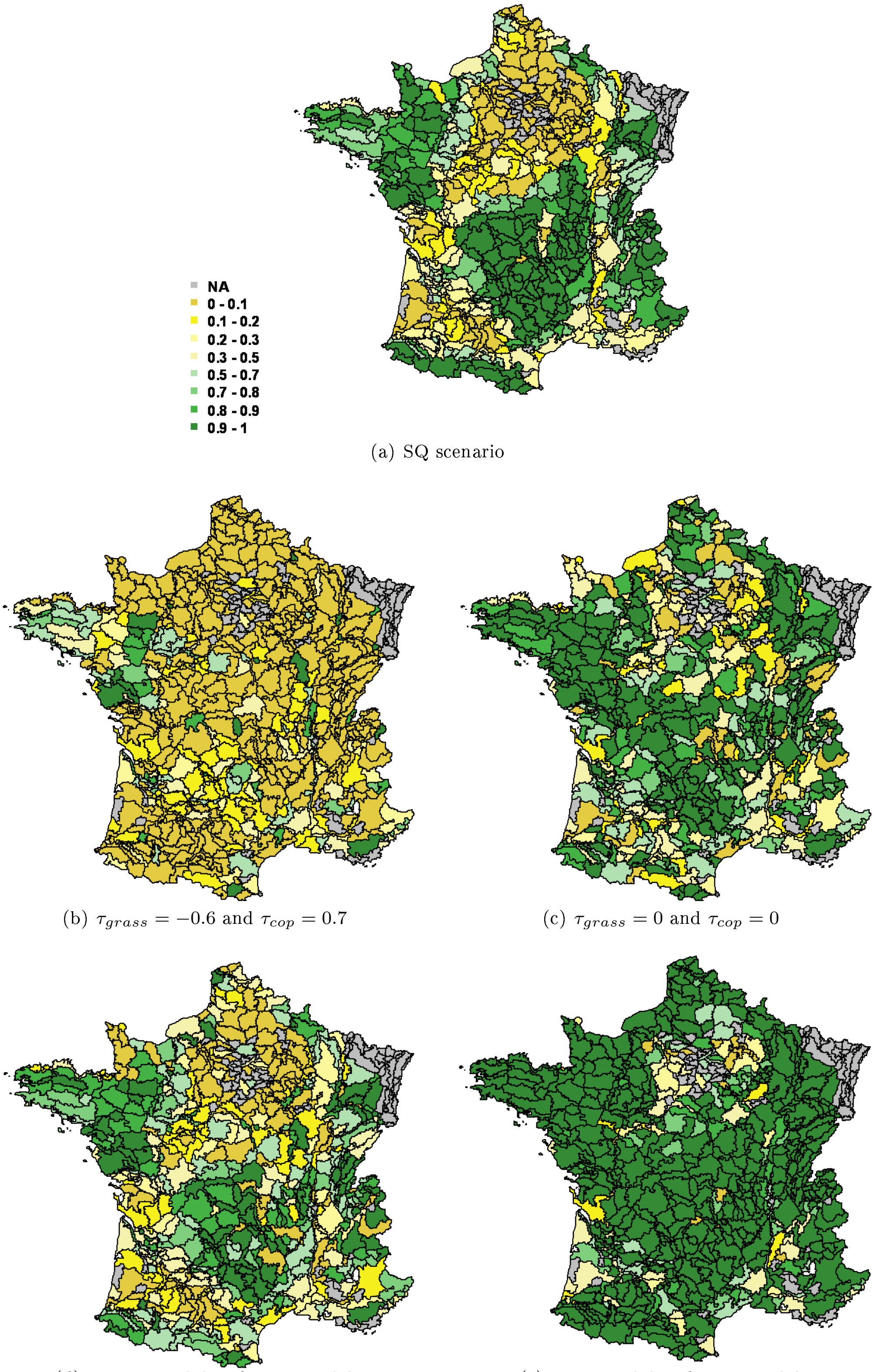
policy among the four presented scenarios. The figure VI.7 compares the land-uses for the four scenarios and the Statu Quo scenario. This figure illustrates the substitution of the crops by the non intensive grasslands in the viable scenario (fig. VI.7(e)).

5. Discussion

5.1. The co-viability approach for sustainability issues

Let us first analyze the whole set of results in terms of sustainability. We built in this study a dynamic bio-economic modeling to represent the agro-ecosystem. This model combines multi-species and multi-scale considerations. The study of this modeling through the viability approach (Aubin, 1991) has lead to identify public policies for multi-functional and sustainable agriculture.

Beyond the analysis on the case study, this work advocates an integrated and multi-criteria approach involving many scientific disciplines, in broad collaborative efforts. A wide range of stakeholders are involved in agro-ecological issues. Each of these groups has an interest in particular outcomes and the outcomes that are considered desirable by one stakeholder may be undesirable to another group. The consideration of this multi-dimensional nature of farming management is a way of guaranteeing a reasonable exploitation of terrestrial resources, allowing the creation of conditions for sustainability from economic, environmental and social viewpoints. The present work is in direct line with these considerations. First, of interest is the use



(a) SQ scenario

(b) $\tau_{grass} = -0.6$ and $\tau_{cop} = 0.7$

(c) $\tau_{grass} = 0$ and $\tau_{cop} = 0$

(d) $\tau_{grass} = -0.6$ and $\tau_{cop} = -0.2$

(e) $\tau_{grass} = 0.6$ and $\tau_{cop} = -0.2$

FIGURE VI.7 – Proportion of non intensive grasslands of the sum of crops and non intensive grasslands with the Statu Quo scenario (SQ) and four public policy scenarios in 2050.

of bio-economic models and assessments articulating ecological and socio-economic processes and goals as in Béné *et al.* (2001), Doyen *et al.* (2012), Péreau *et al.* (2012). More generally, this study confirms the use of this viability method as a convenient tool to reconcile apparently contradictory objectives. Indeed, many study based on optimum-control approaches had identified a Pareto-optimum frontier between ecological and economic performances of agriculture (Polasky *et al.*, 2005, Barraquand & Martinet, 2011). In others words, it does not exist a unique win-win scenario which maximizes simultaneously ecological and economic criteria. This viability approach offers a more flexible context of sustainability which lead to reconcile different objectives.

Moreover by focusing on sustainability and viability, the present model exhibits management strategies and scenarios which satisfy different constraints at each time. With these considerations, the model is taken accounts for intergenerational equity and allows a conciliation between the present and the future. By identifying current public policy decisions which avoids future crisis without penalizing the current generation, the viability approach is in direct line with definition of the sustainability. As emphasized in Martinet & Doyen (2007) and De Lara & Doyen (2008), viability is closely related to the maximin (Rawlsian) approach with respect to intergenerational equity. In this respect, the co-viability strategy turns out to be a promising approach.

5.2. Decision support for farmland biodiversity

This study highlights that it exists a set of sustainable public policy scenarios able to reconciliate biodiversity and agriculture. The management of uncertainty is satisfying since several policies comply with the set of inter-temporal constraints with a high probability. In terms of biodiversity, these public decisions warrant the persistence of the community. The viability of the community should sustain ecosystemic services, which should indirectly contribute to the farming production and to its sustainability.

We have showed that the viable policies are based on a combination between taxes on crops and subsidies on extensive and semi-extensive grasslands. This conclusion confirms the requirement to develop grasslands and reduce crops for a sustainable management of agriculture and biodiversity (Potter & Goodwin, 1998). Indeed, the development of grasslands is positive for biodiversity (Laiolo, 2005). And relevant subsidy scenarios can make this agriculture perspective economically viable (Mouysset *et al.*, 2011). However, the size of the kernel with various combinations between the two incentives offers an additional flexibility to integrate others constraints such as social objectives.

As suggested by Mouysset *et al.* (2012a), we integrate the three indicators FBI, CSI and CTI as ecological criteria of biodiversity. The sensitivity analysis shows that the FBI is the more sensitive and the more restrictive criteria. In others terms, by sustaining the FBI, the three ecological constraints are satisfied. This observation validates the choice of this indicator by the European Union as indicator of structural biodiversity changes in response to agricultural evolution even if this is not the more relevant in functional terms.

The absence of viable public policy for the more restrictive level of biodiversity constraints confirms the negative impact of the current evolution of agriculture. The evolution of agriculture does not lead to guarantee a community at least equivalent at the community obtained with steady land-uses (Statu Quo scenario). In particular, constraints can be violated in a short term due to the inertia of ecological system.

Acknowldgements

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Chapitre VII

Dynamic models for bird community in farming landscapes

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Revision in Ecological Modelling

Abstract

Agricultural changes have caused a severe decline in common bird community in Europe. Mitigating this loss requires to both understand and predict how agriculture affect biodiversity. The objective of this paper is to test different dynamic models coupling bird abundances and farming land-uses in the description and the prediction of birds to agricultural changes. The agro-ecological calibration relies on 2002-2009 data for 34 bird species and 14 farming land-uses in 620 regions of metropolitan France. The models are compared through an indicator of fitness and indicators of predictive quality. The results showed that models including variables related to farming land uses can predict bird abundances in agricultural landscapes.

Keywords : Agro-ecology, Model selection, Intra specific competition, Farming land-use, Adjustment, Prediction

1. Introduction

In many European countries, a strong decline of biodiversity is observed in agricultural landscapes. This is especially documented for mammals in Flowerdew & Kirkwood (1997), for arthropods and plants in Sotherton & Self (2000) or for birds in Donald *et al.* (2001). Common bird are particularly affected and bird breeding surveys report a significant loss of both abundance and diversity in many communities (Butchart *et al.*, 2010, Jiguet, 2010, Gregory *et al.*, 2004). Farmland specialist birds are the most affected species : typically, a 25%-loss of abundance in 20 years has been showed in France (Jiguet, 2010). The changes in agricultural practices over the last decades and especially the intensification processes at play are pointed out as major detrimental factors (Benton *et al.*, 2003). Hence, the mechanization spread, the hedge disappearance, the habitat homogenization and the increase use of pesticids all contribute to the disturbance of bird survival rate and breeding success (Devictor *et al.*, 2008, Gregory *et al.*, 2004). Meanwhile, no biodiversity loss has been outlined in High Nature Value farmlands (Doxa *et al.*, 2010). In the prospect of reconciling productivity and biodiversity conservation in farmlands, agricultural public policies now aim at developing agriculture both economically and ecologically viable (Sabatier *et al.*, 2010, De Lara & Doyen, 2008, Tichit *et al.*, 2007). For instance, the European agri-environmental scheme offers national support to the farmers who adopt environment-friendly practices. However, as most of these schemes are not assessed in a cost-effectiveness way, the question whether ecological, productive and economic performances can be balanced remains an open debate (Kleijn *et al.*, 2006, Perrings *et al.*, 2006).

The development of such a multi-functional agriculture requires the understanding of the link between agriculture and biodiversity. In this context, agro-ecological models linking agriculture and biodiversity dynamics can be fruitful. Such agro-ecological modelling can be used to either describe past or current evolutions of biodiversity (Etterson & Nagy, 2008) or predict future trends in response to different agricultural changes (Maxwell *et al.*, 2009, Reyjol *et al.*, 2001, Roche, 1999). In this vein, many studies focus on a reduced number of species or a particular landscape (Rashford *et al.*, 2008, Polasky *et al.*, 2005, Drechsler, 2001, Doherty *et al.*, 1999). By contrast, the use of larger spatial scale (ecosystem, regional, national, international) or multi-scale approaches leads to more informative results. Similarly, the use of a taxon instead of one or two emblematic species provides more robust and generic agro-ecological informations. In this context, birds are often used as a representative taxon (Mouysset *et al.*, 2012; 2011, Rashford *et al.*, 2008, Holzkamper & Seppelt, 2007, Polasky *et al.*, 2005). Indeed, birds are largely recognized as a biodiversity compartment highly sensitive to agricultural patterns (Gregory *et al.*, 2004). Besides, focusing on breeding birds is further justified because (i) birds lie at a high level in the trophic food chains and thus capture the variations in the chains ; (ii) birds provide ecological services, such as the regulation of invertebrate and rodent populations and pest control (Sekercioglu *et al.*, 2004) ; (iii) their close vicinity to humans makes them a simple and comprehensive biodiversity index for a large audience of citizens (Ormerod & Watkinson, 2000).

In line with this, the present paper develops dynamic bird community models impacted by farming land-uses. In order to capture the biodiversity variations in response to agricultural pressure, these models link the abundances of a large set (34) of common birds species and the proportions of the Utilized Agricultural Area (UAA) dedicated to 14 agricultural patterns such as the crop systems or the bovine livestock breeding. These models are spatialized and calibrated over the 620 metropolitan French small agricultural areas (PRA, "Petite Region Agricole"). Birds data 2002-2009 are provided by the national Breeding Bird Survey (BBS) while agricultural data are derived from by the Farm Accountancy Data Network (FADN) and the Observatory of Rural Development (ODR). We test different models of population dynamics with increasing complexity and different intra specific competitions. The comparison and selection of models rely on two criteria (Levins, 1966) . Firstly, the adjustment to the data (Etterson & Nagy, 2008, Williams *et al.*, 2002) : the model has to fit the past time series. Secondly, the predictive capacity (Geisser & Eddy, 1979, Geisser, 1975) : the model has to be able to predict trend of abundance with a moderate error as it will be used to

The 14 land-uses (OTEA) k
(1) Cereal, Oleaginous, Proteaginous (COP)
(2) Variegated crops
(3) Intensive bovine livestock breeding
(4) Medium bovine livestock breeding
(5) Extensive bovine livestock breeding
(6) Mixed crop-livestock farming with herbivorous direction
(7) Other herbivorous livestock breeding
(8) Mixed crop-livestock farming with granivorous direction
(9) Mixed crop-livestock farming with other direction
(10) Granivorous livestock breeding
(11) Permanent farming
(12) Flower farming
(13) Viticulture
(14) Others associations

Table VII.1 – List of the 14 farming land-uses (OTEA)

make long term projections and scenarios.

The paper is organized as follows. The second section describes the material and methods. The third section presents the results. The fourth is devoted to their discussion. Finally the conclusions and perspectives are depicted in the fifth section.

2. Material and methods

2.1. Data

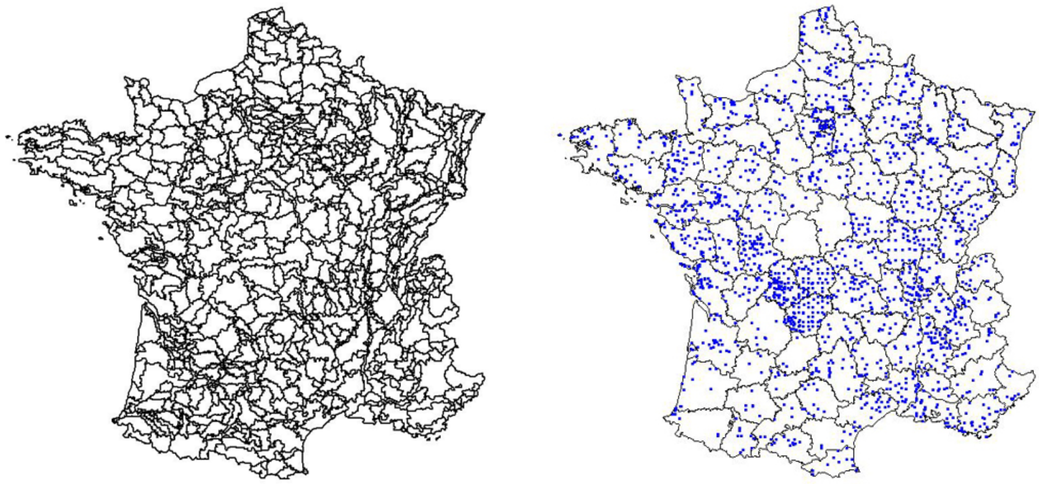
The models are spatialized over the 620 metropolitan French small agricultural regions (PRA, "Petite Region Agricole") (fig. VII.1(a)). The consistency of PRA at both agronomic and ecological levels makes them particularly well-suited for our modelling. The historical agricultural data of these regions are given by the Farm Accountancy Data Network[1] (FADN) and the Observatory of Rural Development[2] (ODR). They distinguish 14 agricultural land-use named OTEA detailed in Table VII.1. The surfaces dedicated to each OTEA in every PRA is known between 2001 and 2008.

For the ecological part of the model, we used 2002-2009 data provided by the national Breeding Bird Survey (BBS) implemented in France (fig. VII.1(b). Among the common breeding species monitored by BBS, we focus on those 34 species classified as farmland specialist and habitat generalist species (Tab VII.2, Julliard *et al.* 2006). Abundance values for each species were available for the period 2002-2009 as detailed in Jiguet 2009. For each species, we further performed a spatial interpolation of these abundance data to obtain relative abundance values for each possible square in the country using kriging models based on spatial autocorrelation (Mouysset *et al.*, 2012, Doxa *et al.*, 2010).

2.2. Bird dynamics

Ten dynamic models with increasing complexities (structure and number of parameters) are described in discrete time. Each model is built independently for each species s at the PRA scale r. The variable $N_{s,r}(t)$ stands for the abundance of species s in PRA r at year t.

1. http ://ec.europa.eu/agriculture/rica
2. https ://esrcarto.supagro.inra.fr/intranet/



(a) PRA map (b) French Bird Breeding Survey map

FIGURE VII.1 – Maps of the PRA and the observation squares of the French Bird Breeding Survey.

- **Model M0 :**

$$N_{s,r}(t+1) = N_{s,r}(t) \tag{VII.1}$$

Although this model is not very stimulating regarding agro-ecological issues, it hereafter plays the role of statistical benchmark model as shown by fig. VII.2(a).

- **Model M1 :**

$$N_{s,r}(t+1) = N_{s,r}(t).(1 + R_{s,r}) \tag{VII.2}$$

The second population model M1 based on linear dynamics of eq. (VII.2) involves the intrinsic growth rates $R_{s,r}$ for every species s in region r. It also plays the role of a benchmark model.

- **Model BH$_M$:**

$$N_{s,r}(t+1) = N_{s,r}(t).\frac{1 + R_{s,r}}{1 + \frac{N_{s,r}(t)}{\beta_{s,r}}} \tag{VII.3}$$

The third model BH_M in eq. (VII.3) corresponds to the Beverton-Holt dynamics (Beverton & Holt 1957). It includes an intra specific competition through a carrying capaticy computing as the product between the growth parameter $R_{s,r}$ and a second parameter $\beta_{s,r}$ (fig. VII.2(b)). This parameter $\beta_{s,r}$ can be interpretated as the quality of habitat in the region r for the species s. This Beverton-Holt model is an example of the contest competition, which implies that a part of the population dominates the other in the sense that only this part will eventually survive if resources are lacking (Hassel 1975). Thus, when the initial abundance is above the carrying capacity, the population declines gradually towards the equilibrium.

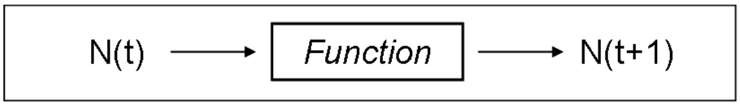
- **Model BH$_A$:**

$$N_{s,r}(t+1) = N_{s,r}(t).\frac{1 + R_{s,r}}{1 + \frac{N_{s,r}(t)}{M_{s,r,k}(t)}} \tag{VII.4}$$

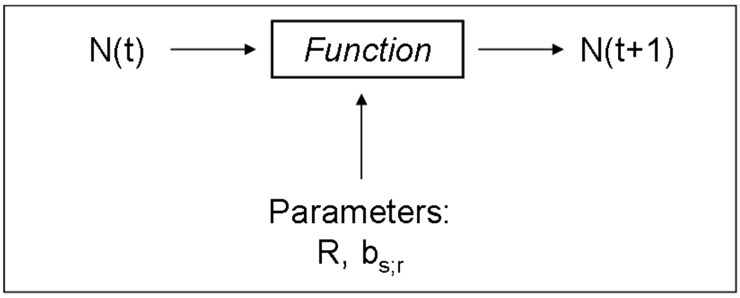
Now habitat quality $M_{s,r,k}(t)$ is depending on a specific land-use (OTEA) k as follows :

$$M_{s,r,k}(t) = \beta_{s,r} + \alpha_{s,r,k}.A_{r,k}(t) \tag{VII.5}$$

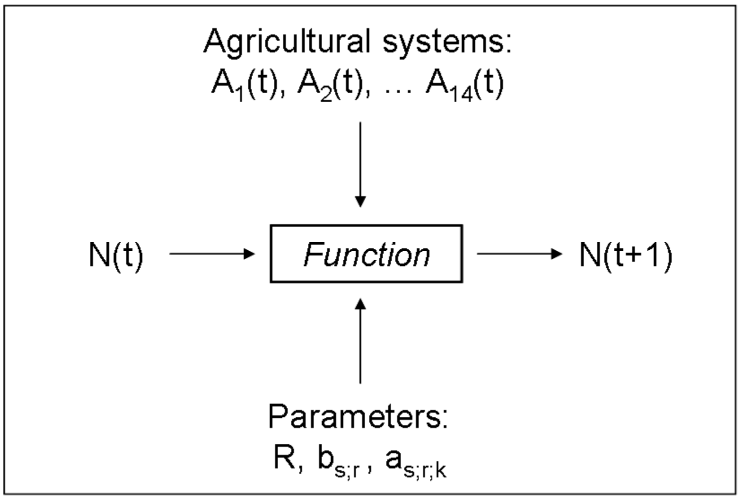
where $A_{r,k}(t)$ stands for the proportion of the Utilized Agricultural Area (UAA) dedicated to the agricultural OTEA k in the PRA r at time t. Two contrasted agricultural systems have been tested : the crop (k=1 in the table VII.1) and the extensive bovine livestock breeding (k=5 in the table VII.1). Thus this model BH_A also relies on a Beverton-Holt dynamic but the habitat value $M_{s,r,k}(t)$ is dynamically



(a) PRA map



(b) French Bird Breeding Survey map



(c) French Bird Breeding Survey map

FIGURE VII.2 – Model coupling for three levels of complexity of modelling.

related to the agricultural pattern and land-uses k (fig. VII.2(c)). The parameter $M_{s,r,k}(t)$ is built as an affine function of the proportion $A_{r,k}(t)$ of the farming OTEA k. The coefficient $\alpha_{s,r,k}$ corresponds to the response to the species s to this agricultural pattern k in the PRA r. The parameter $\beta_{s,r}$ captures the effect of all other habitats : the other agricultural systems and the non-agricultural lands of the region r (forests, cities...).

- **Model BH :**

$$N_{s,r}(t+1) = N_{s,r}(t).\frac{1+R_{s,r}}{1+\frac{N_{s,r}(t)}{M_{s,r}(t)}} \tag{VII.6}$$

where habitat quality $M_{s,r}(t)$ is now characterized by

$$M_{s,r}(t) = \beta_{s,r} + \sum_{k=1}^{14} \alpha_{s,r,k}.A_{r,k}(t) \tag{VII.7}$$

This fifth model BH also relying on the Beverton-Holt population dynamics, involves a broader complexity in the sense that the habitat variables $M_{s,r}(t)$ are now affected by the whole set of 14 agricultural land-uses k. This models are thus depending on 16 parameters through coefficients $R_{s,r}$, $\beta_{s,r}$ and the 14 $\alpha_{s,r,k}$.

- **Model GO :**

$$N_{s,r}(t+1) = N_{s,r}(t)\ .\ \exp\left[R_{s,r}\ .\ \left(1 - \frac{\log(N_{s,r}(t))}{\log(R_{s,r}\ .\ M_{s,r}(t))}\right)\right] \tag{VII.8}$$

with $M_{s,r}(t)$ is again defined in eq. (VII.7). A contest of competition also underlies this Gompertz GO model with the same account of complexity and impact from land-uses than the BH model but according to a different functional form.

- **Model RI :**

$$N_{s,r}(t+1) = N_{s,r}(t)\ .\ \exp\left[R_{s,r}\ .\ \left(1 - \frac{N_{s,r}(t)}{R_{s,r}\ .\ M_{s,r}(t)}\right)\right] \tag{VII.9}$$

with $M_{s,r}(t)$ is again defined as in eq. (VII.7). Such Ricker model RI (eq. VII.9) refers to the same level of complexity than the BH model but with a different density-dependence relation. The Ricker relation captures a scramble competition : resources are equally divided for all individuals (Hassel, 1975) and whenever the population is above carrying capacity, the abundance first collapses then grows back to equilibrium.

- **Model LO :**

$$N_{s,r}(t+1) = N_{s,r}(t) + N_{s,r}(t)\ .\ R_{s,r}\ .\ \left(1 - \frac{N_{s,r}(t)}{R_{s,r}\ .\ M_{s,r}(t)}\right) \tag{VII.10}$$

with $M_{s,r}(t)$ is again defined by eq. (VII.7). Finally the logistic model LO is a different illustration of the scramble competition. It keeps the same level of complexity with respect to farming land-uses than the RI model.

- **Model BH$_\theta$:**

$$N_{s,r}(t+1) = N_{s,r}(t) . \frac{1 + R_{s,r}}{\left(1 + \frac{N_{s,r}(t)}{M_{s,r}(t)}\right)^{\theta}} \tag{VII.11}$$

with $M_{s,r}(t)$ defined by eq. (VII.7). This model aims at reinforcing the role played by the density-dependence through the parameter θ (Ross, 2009, Polansky *et al.*, 2008). The new parameter θ characterizes the type and intensity of density dependence. In order to keep the same number of parameters to calibrate compared to previous models, the value of θ is set before calibration.

- **Model LO$_\theta$:**

$$N_{s,r}(t+1) = N_{s,r}(t) + N_{s,r}(t)\ .\ R_{s,r}\ .\ \left[1 - \left(\frac{N_{s,r}(t)}{R_{s,r}\ .\ M_{s,r}(t)}\right)^{\theta}\right] \tag{VII.12}$$

Based now on the logistic form, this model also fosters the role played by density-dependence in the similar way to the BH$_\theta$ model. The value of θ is also set before calibration.

2.3. Calibration

For each species, the ten previous models are calibrated through a least square method used to estimate the set of parameters that minimizes the Mean Square Error (MSE) of calibration :

$$\min_{R_{s,r};\ \beta_{s,r};\ \alpha_{s,r,k}} MSE_s^T \tag{VII.13}$$

This calibration error is computed in :

$$MSE_s^T = \frac{\sum_{r=1}^{620} \sum_{2002}^{T} (N_{s,r}^h(t) - N_{s,r}^T(t))^2}{\sum_{r=1}^{620} \sum_{2002}^{T} N_{s,r}^h(t)} \tag{VII.14}$$

$N^h_{s,r}$ represents the vector of historical abundances of species s in area r while $N^T_{s,r}$ represents the vector of abundances estimated by the model which the calibration is based on the time serie 2002 - T. The *datafit* function of Scilab software [1] is used. It relies on an optimization algorithm of conjugate gradients (Flechter & Reeves 1964). The growth rate parameters $R_{s,r}$ are chosen between bounds -1 and 1 while $\alpha_{s,r,k}$ and $\beta_{s,r}$ are selected within the interval $[-100, 100]$. [2] Values $\theta = 2, 3, 4, 5, 6$ are tested for models BH_θ and LO_θ. This implies that only scramble competition are represented by the θ-models (Brannstrom & Sumpter 2005) and that the θ-logistic equation has a convex density dependence relationship, which means that competition strength is limited for small populations and increase strongly with population size (Ross 2009).

2.4. Fitness

The quality of the calibration is directly measured through the Mean Square Error MSE on the entire time serie (T = 2009) for each species s as in eq. (VII.14) (Etterson & Nagy 2008). Mean MSE are computed for the both groups G (farmland specialists and generalists) as an arithmetic mean (eq. VII.15). This indicator tends to favor models including a high number of parameters (Wallach 2006).

$$MSE = \frac{1}{Card(G)} \sum_{s \in G} MSE_s^{2009} \tag{VII.15}$$

2.5. Prediction quality

Cross-validation is a widely used technique to assess the prediction ability of models (Hawkins *et al.* 2003). It compares the prediction quality of different models with eventually different complexity levels (Roche 1999). We develop two indicators of prediction quality to distinguish the short term and the long term predictive qualities. In the two cases, we evaluate a Mean Square Error of prediction $MSEp_s$ for each species s based on the errors between the estimated and the historical abundances for these 4 last years (from 2006 to 2009) :

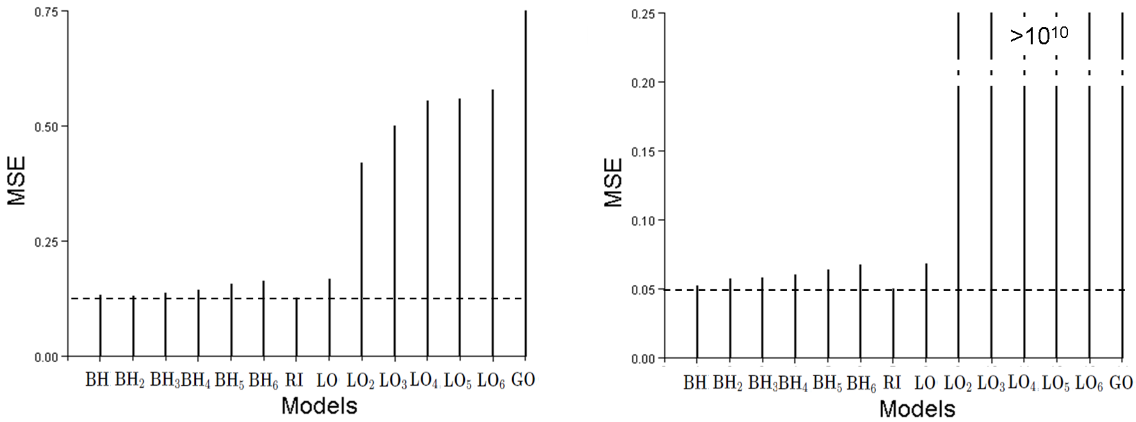
$$MSEp_s = \frac{\sum_{r=1}^{620} \sum_{t=2006}^{2009} (N^h_{s,r}(t) - N^T_{s,r}(t))^2}{\sum_{r=1}^{620} \sum_{t=2006}^{2009} N^h_{s,r}(t)} \tag{VII.16}$$

$N^h_{s,r}$ represents the vector of historical abundances of species s in area r while $N^T_{s,r}$ represents the vector of abundances estimated by the model which the calibration is based on the time serie 2002 - T. The difference betweeen the long-term and short-term indidicators is the time serie 2002-T on which is based the calibration used to compute the estimated abundances $N^T_{s,r}(t)$ from 2006 to 2009. For the long-term indicator, T is fixed with T = 2005, which means that the estimated abundances $N_{s,r}(t)$ from 2006 - 2009 are calculated with the same calibration based on the time serie 2002 - 2005. For the short-term indicator, T is dynamic with T = t - 1. The observations from 2002 to 2005 (2002 - 2006, 2002 - 2007, 2002 - 2008 resp.) are included in the calibration to compute the estimated abundance in 2006 (2007, 2008, 2009 resp.).

Finally, mean MSEp are computed for the both groups G (farmland specialists and generalists) as an arithmetic mean :

$$MSEp = \frac{1}{Card(G)} \sum_{s \in G} MSEp_s \tag{VII.17}$$

1. Free software for scientific computation : http ://www.scilab.org/
2. Such choice stems from the fact the maximum observed abundance of birds by PRA is close to 200. Therefore the carrying capacity $R_{s,r} * M_{s,r}(t)$ is bounded by 200, which leads to the specified bounds for a and b.



(a) The specialist group (b) The generalist group

FIGURE VII.3 – Valuation of the quality of the calibration through the Mean Square Error MSE for different models with the same level of complexity in the specialist and the generalist groups.

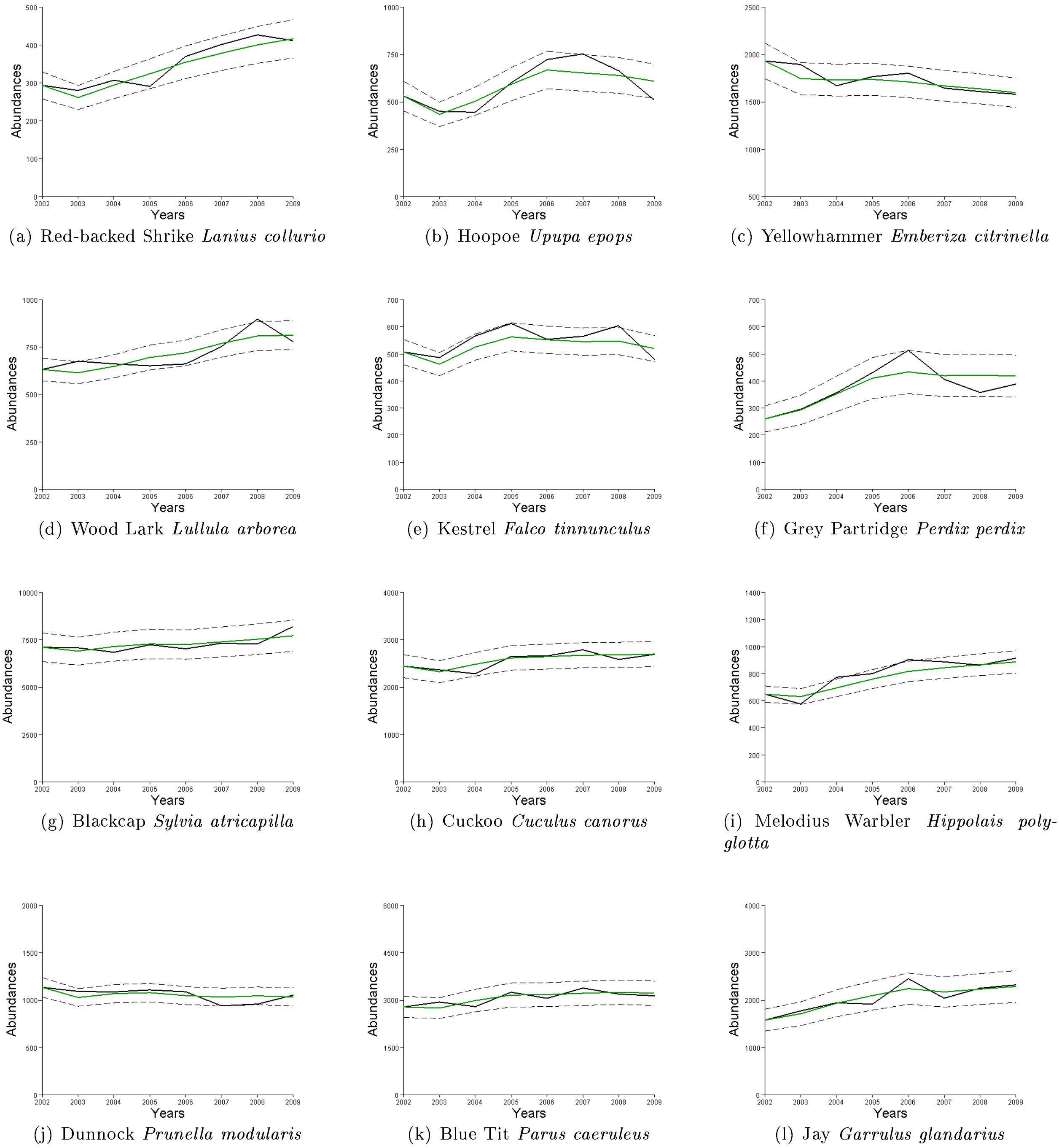
3. Results

3.1. Fitness

The figure VII.3 illustrates the fitness indicator MSE for 14 tested models with the same level of complexity in the two groups farmland specialist and generalist species. A set of models (BH, BH_θ, RI, LO) performs better than others with a small MSE. According to the set of tested models, these models are the more relevant to correctly describe the bird populations. The MSE are in the same order of magnitude with a slight advantage for the BH, BH_2 and RI models. If the classifications of the models are the same by the specialist and generalist species, we observe that the MSE is smallest with the generalist group than with the specialist one. The figure VII.4 depicts 12 examples of calibrations (6 specialist and 6 generalist species) with the historical abundances (in black solid line) and the estimated abundances with the optimal models (in green solid line) with the 99% interval confidence. These examples show that the evolutions integrate correctly the historical trends but by smoothing the inter-annual variations. In this context, the smallest inter-annual variations in the generalist rather than the specialist trends explain the smallest MSE obtained in the generalist groups.

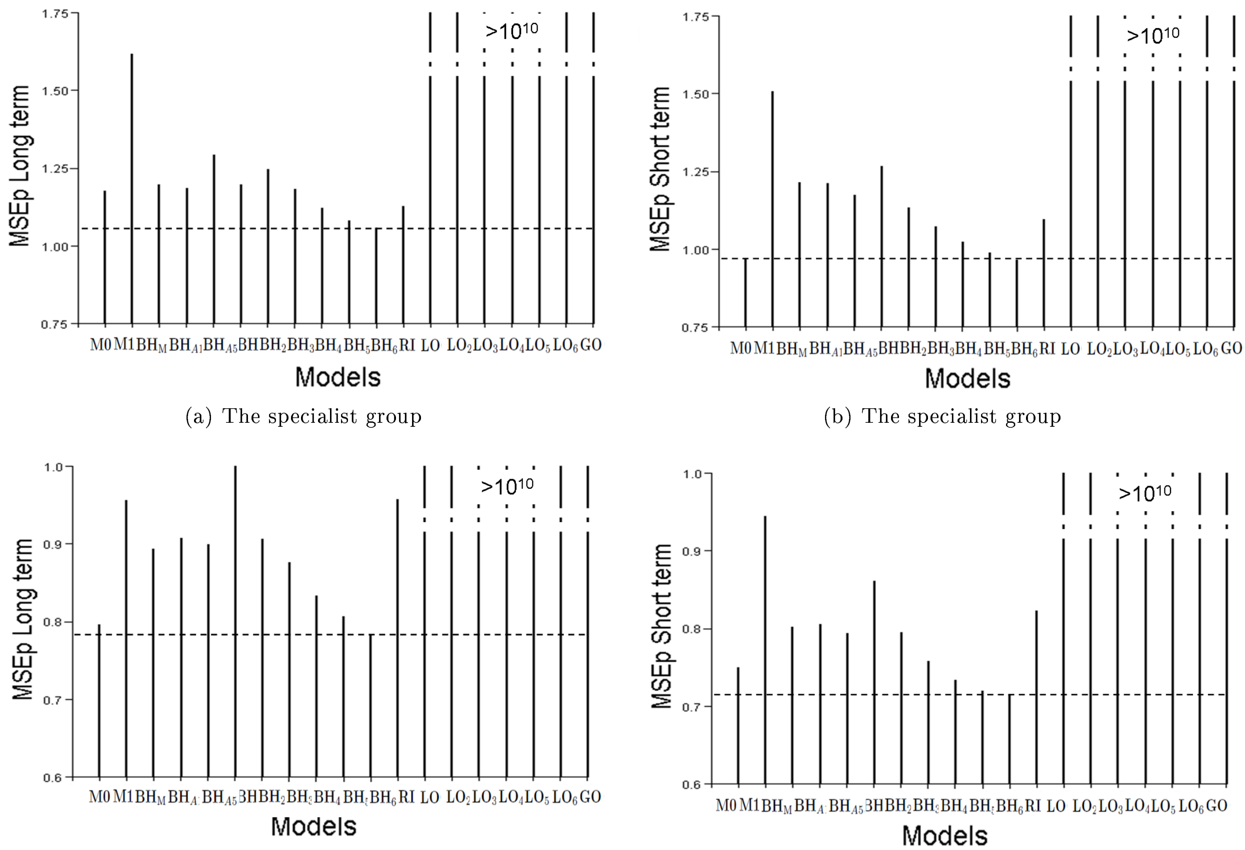
3.2. Prediction

The figure VII.5 describes the long term and short prediction errors MSEp for the 19 tested models in the two groups of species. As in figure VII.3, the errors are smaller in the generalist group rather than in the specialist one and the both groups present the same global conclusions : a set of models (LO, LO_θ, GO) are not relevant to predict the bird evolutions. Among the others models, the BH_5 and BH_6 models display the smallest errors of prediction at short and long terms for the specialist and the generalist species. However if the long and short term indicators classify the models in the same order, we can note an interesting result with the model M0. For the long-term predictions, the difference between the M0 and the BH_6 models is more contrasted in the specialist group rather than in the generalist group. This is explained by the relative stability of many generalist species. The complexification of the model and the integration of agricultural drivers significantly improve the specialist bird trajectories in response to agricultural changes. This is observed for every specialist species (fig. VII.6(a), VII.6(b), VII.6(c)) but only a few generalist species (fig. VII.6(f)). The fact that almost every generalist species follow a quasi stable evolution (fig.



(a) Red-backed Shrike *Lanius collurio*
(b) Hoopoe *Upupa epops*
(c) Yellowhammer *Emberiza citrinella*
(d) Wood Lark *Lullula arborea*
(e) Kestrel *Falco tinnunculus*
(f) Grey Partridge *Perdix perdix*
(g) Blackcap *Sylvia atricapilla*
(h) Cuckoo *Cuculus canorus*
(i) Melodius Warbler *Hippolais polyglotta*
(j) Dunnock *Prunella modularis*
(k) Blue Tit *Parus caeruleus*
(l) Jay *Garrulus glandarius*

FIGURE VII.4 – Examples of calibration comparing the historical (in black) and the estimated (in green) national abundances with the optimal model (see table VII.2) with the 99% confident interval (dashed lines) for 6 specialist species (a-f) and 6 generalist species (g-l).



(a) The specialist group

(b) The specialist group

(c) The generalist group

(d) The generalist group

FIGURE VII.5 – Valuation of the long and short terms predictions through the Mean Square Error of prediction $MSEp$ for all the tested models in the specialist and the generalist groups.

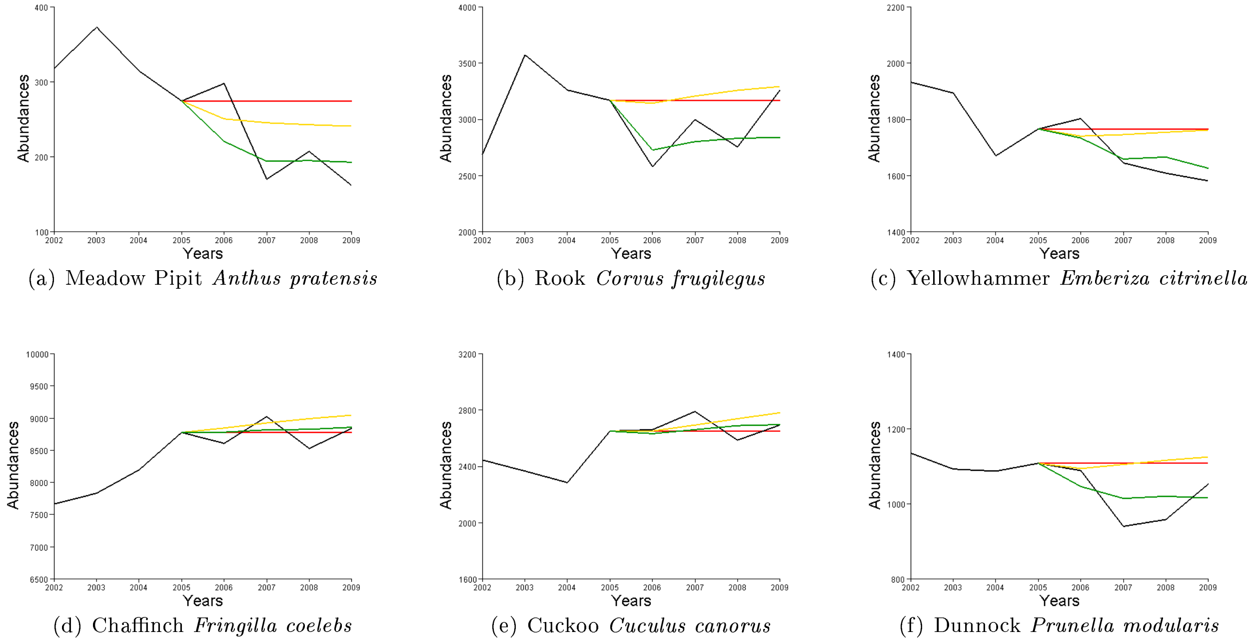
VII.6(d), VII.6(e)) suggests that the benefit of the complexification is small. For the short-term predictions, similar performances in the specialist group occur for the M0 and the BH$_6$ models. This loss of advantage in including agricultural variables at short-term (where the agricultural changes are small) confirms the importance of these agricultural variables to predict specialist bird abundances.

3.3. Model selection by species

The table VII.2 depicts the optimal models following the fitness error MSE and the prediction evaluation MSEp for each species. We observe that only four models (BH$_2$ BH$_5$, BH$_6$) turn out optimal according to the MSEp score. Moreover, the models BH$_5$ and BH$_6$, chosen for 74% of the species, are clearly prevailing. For the MSE, the result is more contrasted as the optimal models according to the MSE criterion vary among the species. Indeed, in addition to the four previous models, several models based on the logistic function (LO, LO$_2$, LO$_6$) and the BH model are also well-suited. However a difference between the specialist and the generalist groups can be emphasized at this stage. Within the generalist species group, the RI model is selected in most cases, while the BH$_2$ model is dominant in the specialist group. The specialist group is also characterized by a broader diversity of models than the generalist group.

Species	Habitat	lg MSEp	MSE
(1) Buzzard *Buteo buteo*	Specialist	$\boxed{BH_6}$	$\boxed{BH_2}$
(2) Cirl Bunting *Emberiza cirlus*	Specialist	$\boxed{BH_6}$	LO_2
(3) Corn Bunting *Emberiza calandra*	Specialist	$\boxed{BH_2}$	LO_6
(4) Grey Partridge *Perdix perdix*	Specialist	$\boxed{BH_6}$	LO_2
(5) Hoopoe *Upupa epops*	Specialist	$\boxed{BH_6}$	LO_2
(6) Kestrel *Falco tinnunculus*	Specialist	$\boxed{BH_6}$	$\boxed{BH_2}$
(7) Lapwing *Vanellus vanellus*	Specialist	$\boxed{RI}$	$\boxed{BH_2}$
(8) Linnet *Carduelis cannabina*	Specialist	$\boxed{BH_6}$	$\boxed{BH_2}$
(9) Meadow Pipit *Anthus pratensis*	Specialist	$\boxed{BH_6}$	$\boxed{BH_2}$
(10) Quail *Coturnix coturnix*	Specialist	$\boxed{BH_5}$	LO_6
(11) Red-backed Shrike *Lanius collurio*	Specialist	$\boxed{BH_6}$	$\boxed{RI}$
(12) Red-legged Partridge *Alectoris rufa*	Specialist	$\boxed{BH_5}$	$\boxed{BH_2}$
(13) Rook *Corvus frugilegus*	Specialist	$\boxed{RI}$	$\boxed{BH_5}$
(14) Skylark *Alauda arvensis*	Specialist	$\boxed{BH_6}$	$\boxed{BH_6}$
(15) Stonechat *Saxicola torquatus*	Specialist	$\boxed{BH_5}$	LO_6
(16) Whinchat *Saxicola rubetra*	Specialist	$\boxed{BH_2}$	BH
(17) Whitethroat *Sylvia communis*	Specialist	$\boxed{BH_5}$	$\boxed{BH_2}$
(18) Wood Lark *Lullula arborea*	Specialist	$\boxed{BH_6}$	$\boxed{BH_2}$
(19) Yellowhammer *Emberiza citrinella*	Specialist	$\boxed{RI}$	$\boxed{BH_2}$
(20) Yellow Wagtail *Motacilla flava*	Specialist	$\boxed{RI}$	$\boxed{BH_2}$
(21) Blackbird *Turdus merula*	Generalist	$\boxed{BH_6}$	$\boxed{RI}$
(22) Blackcap *Sylvia atricapilla*	Generalist	$\boxed{BH_6}$	$\boxed{RI}$
(23) Blue Tit *Parus caeruleus*	Generalist	$\boxed{BH_2}$	$\boxed{RI}$
(24) Carrion crow *Corvus corone*	Generalist	$\boxed{BH_6}$	$\boxed{RI}$
(25) Chaffinch *Fringilla coelebs*	Generalist	$\boxed{BH_6}$	LO
(26) Cuckoo *Cuculus canorus*	Generalist	$\boxed{BH_6}$	$\boxed{BH_5}$
(27) Dunnock *Prunella modularis*	Generalist	$\boxed{RI}$	$\boxed{BH_2}$
(28) Great Tit *Parus major*	Generalist	$\boxed{BH_6}$	$\boxed{RI}$
(29) Green Woodpecker *Picus viridis*	Generalist	$\boxed{BH_6}$	LO
(30) Golden oriole *Oriolus oriolus*	Generalist	$\boxed{BH_5}$	$\boxed{RI}$
(31) Jay *Garrulus glandarius*	Generalist	$\boxed{BH_6}$	$\boxed{RI}$
(32) Melodius Warbler *Hippolais polyglotta*	Generalist	$\boxed{BH_5}$	$\boxed{BH_2}$
(33) Nightingale *Luscinia megarhynchos*	Generalist	$\boxed{RI}$	LO
(34) Wood Pigeon *Columba palumbus*	Generalist	$\boxed{BH_6}$	BH

TABLE VII.2 – Optimal models according the both criteria long-term Mean Square Error MSE_p of prediction and Mean Square Error MSE for the 20 farmland specialist and the 14 habitat generalist species.



(a) Meadow Pipit *Anthus pratensis*
(b) Rook *Corvus frugilegus*
(c) Yellowhammer *Emberiza citrinella*
(d) Chaffinch *Fringilla coelebs*
(e) Cuckoo *Cuculus canorus*
(f) Dunnock *Prunella modularis*

FIGURE VII.6 – Ilustrations of long term prediction for 3 specialist species (a-c) and 3 generalist species (d-f) with the historical national abundances in black and the abundances estimated by the M0 model in red, the BH_M model in yellow and the optimal model (see table VII.2 column 1) in green.

4. Discussion

4.1. Fitness of bird population in farming landscapes

It is well-known that the quality of the adjusment between a model and data increases with the complexity of the model and the number of explicative variables. So we directly compared the performances of different models with similar complexity to identify the more relevant dynamics to depict the common bird abundance evolution in farmland landscapes. As captured by the table VII.2, two models (BH_2 and RI) are more often selected among the species and satisfying errors of calibration are obtained (10% for the specialists and 5% for the generalist group). The repartition of these two models in the species according to the habitat specialization confirms the relevance of this classification and a difference in their relation to the environment (Julliard *et al.*, 2006). However, in both cases, the selected dynamics is not associated with a strong parameter θ related to habitat strength. This highlights that a balance between growth R and habitat M parameters is needed to improve the description of the bird populations in farming landscapes. A fine adjustement between agricultural variables affecting habitat quality M and others traits through R is required to improve the calibration. The selection of a dynamic with habitat strength of $\theta = 2$ for the bird specialists compared with a strength of $\theta = 1$ for the generalists points out a more stringent relation to the habitat for the specialists rather than for the generalists (Julliard *et al.* 2006) as expected.

4.2. Prediction of bird population in farming landscapes

The results also illustrate that the choice of the best or a satisfying dynamic relation differs if the objective of the modelling becomes the prediction instead of fitness as above. By contrast to fitness analysis, prediction analysis first requires to confirm that the complexification of the function is useful for the prediction. Here by confronting the complex models BH, GO, RI, ... to simpler ones M0, M1, BH_A, we show that the complexification of the dynamics by the account for an intra-specific competition depending on agricultural land-uses is relevant to improve the prediction of bird abundances. Second, among the set of tested models, only two models (BH_5 and BH_6) emerge as the most efficient to predict the bird abundances in response to agricultural changes. These models are characterized by a strong habitat strength θ which points out that the balance between growth R and habitat M is clearly in favor of M. This emphasizes that the prediction of bird abundances is strongly driven by the agricultural variables. This conclusion is reinforced by the fact that these models are chosen for a large majority of the species independently of their habitat specialization.

4.3. Applications

This kind of models have different applications. First, the values of the parameters of the models selected by the fitness indicator can be analyzed to understand the relative impact of the different agricultural activities on the population sizes according the species and its functional traits. This can be helpful to identify positive and negative activites for the bird populations. Second, the selected models by the long-term prediction can be used to make projections of bird populations included in more global models (agri-ecological or bio-economic models). For example, this leads to understand the future trends of biodiversity in response of different public policies scenarios in order to choose the more suitable policy according biodiversity goals (Mouysset *et al.*, 2012).

5. Conclusion and Perspectives

This study highlights the role played by intra-specific competitions depending on agricultural variables to explain the bird populations trajectories in farming landscapes. However the paper emphasizes the importance of the modelling goal especially between adjustment and prediction. In the predictive context, the weight dedicated to the agricultural variables is increased and the differences among the species in their relation to farming habitat is reduced. In other words, the agriculture turns out as the main driver of the future trends of common birds in farmland. In this context, it should be fruitful to add other agricultural variables such as the heterogeneity of the landscape (Tscharntke *et al.*, 2005) to improve the prediction. In contrast, the more balanced situation between the growth and habitat traits in the descriptive and adjustment context suggests that other drivers should be tested. In particlar, community variables including inter-specific competition, regional dispersion or evolution of geographic range in response of climate change should be combined with new land-use as forest or urban areas.

Acknowldgements

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Annexes

Annexe A

Risk aversion impact on bio-economic performances in an agricultural public policy context : a modeling approach

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Abstract

For several decades, strong changes in bird biodiversity, especially in Europe, have been reported. Agriculture, and more specifically agricultural intensification, is a major driver of these modifications. Taking into account these environmental impacts, agriculture nowadays aims at a more sustainable way of producing, which would reconcile its economic and ecological functions. In this context, many bioeconomic studies analyse the impact of public policies and financial incentives on both biodiversity conservation and farming production. We want to add a essential economic element in this approach : the risk behaviour again income uncertainty. The question asked by this article is : how does the economic risk aversion impact biodiversity and bio-economic performances in a agricultural public policy context ? For this purpose, we chose the inter-discplinary modeling coupling agro-economic and ecological processes. We therefore developed a micro-regional model, which combines a community dynamics of 20 farmland bird species and an economic decision model for each French region. The ecological dynamic model is calibrated with the STOC (Time Survey of Common Birds) database while the economic model relies on the optimization of the gross margin of the RICA (Network of Agricultural Accountant Information). First we analyse the impact of risk aversion level on bio-economic performances and its variability without politic incentives. Then we investigate different scenarios based on subsidies and taxes to study the impact of public policies on both biodiversity depending on the risk aversion. We show that risk aversion is a major driver for bio-economic performances without public policies and can reverse the bio-economic trade-off. Adequate scenarios promote biodiversity, but marginal benefit is bigger when risk aversion is weak. This study illustrates the interest of agricultural public policies use to enhance bird conservation. Indeed the ecological indicator variability is independant on the risk management. These results suggest that many possibilities are available to develop multi-functional sustainable agriculture using public policies.

Keywords : Biodiversity, Agriculture, Bioeconomic modeling, Risk aversion, Sustainability, Bird

1. Introduction

Global changes in European agriculture in recent decades, including intensification and land abandonment, have significantly modified farmland biodiversity. The pressure is particularly strong on bird populations which have undergone severe and widespread decline (Donald *et al.*, 2006, Julliard *et al.*, 2004, Donald *et al.*, 2001, Chamberlain *et al.*, 2000, Krebs *et al.*, 1999). Such erosion is mainly due to a combination of habitat loss and degradation of habitat quality altering the nesting success and/or survival rates (Benton *et al.*, 2003). In this context, the European Union, aiming at halting biodiversity loss by 2010, has adopted the farmland bird index as an indicator of structural changes in biodiversity Balmford *et al.* (2003). In this perspective of particular interest is the need to reconcile agricultural production and biodiversity Jackson *et al.* (2005). A major driver of agricultural economic is the incentives : with Commune Agricultural Policy (CAP) incentives, governments are very implicated in this sector.

There is an extensive and increasing volume of literature concerning agri-environmental schemes and policies for multi-functional agriculture (Dobbs & Pretty, 2004). Approaches with politic scenarios (Shi & Gill, 2005, Pacini *et al.*, 2004) or economic incentives (Drechsler & Watzold, 2007) are developed to reconcile agricultural production and biodiversity. To analyse the sustainability of these scenarios, authors study their impacts on ecological and/or economic criteria (Rashford *et al.*, 2008, Holzkamper & Seppelt, 2007, Doherty *et al.*, 1999). Despite all these studies, the agricultural sustainability is not very-well understood. Two studies (Shi & Gill, 2005, Alavalapati *et al.*, 2002) show that finacial incentives are essential so that farmers adopt eco-friendly activities. On the other hand, Quaas *et al.* (2007) illustrate in a theoretical model that an adequate risk aversion brings farmers to adopt conservative choices. They introduce an essential factor of the economic theory, which was neglected by empirical bio-economic works, risk aversion, which is an important element in the decision-maker behavior and plays a major role in his choices.

The objective of this study is to insert the risk management in traditional bio-economic approaches with incentives scenarios and to analyse more precisely the impact of risk aversion on the agro-ecosystem sustainability function of scenarios. Asked question in this article concerns the intercation between the economic risk aversion and the multi-fonctional agriculture sustainability : how does the economic risk aversion impact biodiversity in a agricultural public policy context ? First we study the impact of risk aversion on bio-economic performance and its variability without public policies. Then we perturbate the agro-ecosystem evolution with incentive scenarios and we analyse the response of this perturbation fonction of the risk aversion.

To address agro-environmental sustainability, numerous modeling frameworks are proposed in the literature. They include Cost-Benefit and Cost-Effectiveness approaches. A major criticism of Cost-Benefit analysis for conservation issues is that benefits related to biodiversity and habitat quality are usually non-market goods, and by definition are difficult to quantify in monetary terms (Rees, 1998). Although pricing techniques such as contingent valuation are available, their suitability for complex biodiversity issues is disputed, notably in anthropogenic systems (Diamond & Hausman, 1994). Cost-Effectiveness analysis can be used to reveal a minimal cost policy among those satisfying the given goals of conservation and production (Macmillan *et al.*, 1998). This approach, based on optimization under constraints, avoids monetary evaluation of environmental goods (Gatto & De Leo, 2000). As far as agricultural economics is concerned, most models rely on mathematical programming and optimization under constraints (Havlik *et al.*, 2005, Polasky *et al.*, 2005, van Wenum *et al.*, 2004). To deal with sustainability, approaches such as ecological economics (Drechsler & Watzold, 2007) suggest studying environmental and economic effectiveness simultaneously, stressing the relevance of multi-criteria approaches. However, few economic studies cope with the spatial and temporal dynamics of biodiversity in this context (Hammack & Brown, 1974). In this vein, a range of spatially explicit models exist that aim at assessing consequences of different land use patterns for various environmental and economic criteria (Swihart *et al.*, 2003, Irwin & Geoghegan, 2001). Nevertheless, most of these models are static, which restricts the ecological processes taken into account. Moreover, they usually do not incorporate important and risky economic drivers (agricultural prices, subsidies) that affect the returns of different land-use patterns.

The bio-economic model proposed in the present paper is in direct line with these considerations. First the model is dynamic : it articulates ecological and economic compartments including several risk management and it adopts a multi-criteria perspective. Moreover, it offers a spatialized perspective as it is built up at an ecological and economic relevant regional scale. Finally its calibration relies on French regional data of both land-use and bird abundance, aimed at promoting an empirically rich and realistic modeling.

The paper is organized as follows : The second section presents data and our bio-economical model. The third section describes the scenarios which we study here and the results. The fourth section is devoted to the discussion.

2. The bio-economic model

2.1. Context and data

We split France in 620 little agricultural regions (PRA : Petites Régions Agricoles). This spatial scale is a cross between departements, which are the relevant economic entities, and agricultural regions, which show an agro-ecological unity. Thus the PRA scale is relevant in ecological-economic perspectives. A bio-economic model, as describe below, is built for each PRA.

Regarding the biodiversity, we focus on common bird populations and related indicators (Julliard *et al.*, 2006). Although the metric and the characterization of biodiversity remain an open debate (Le Roux, 2008, MEA, 2005), such a choice is justified for several reasons (Ormerod & Watkinson, 2000) : (i) Birds lie at a high level in the trophic food chains and thus capture the variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation and viability (Sekercioglu *et al.*, 2004). (iii) The availability of data and ecological knowledge : Birds belong to the most studied taxa and many databases are available. (iv) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens. The STOC database developed by the Museum National d'Histoire Naturelle provides the data related to the bird abundances. Among the 175 species monitored by this program, we have selected the 20 farmland bird species (see appendix A) used as a reference for the European FarmlandBird Index (European Bird Census Council, 2007). The abundances at PRA scale for the years 2001 to 2008 are available for each of these species.

For agro-economic data, we use the french agro-economic classification (OTEX : orientation technico-economique) developed by the RICA (Reseau d'Information Comptable Agricole) [1]. This organisation distinguishes 14 classes depending of main activities of farmers (see appendix B). Each PRA is a patchwork of many agricultural systems. For each agricultural system-region couple, we have the allocated surface and the gross margin for the years 2001 to 2008.

2.2. The ecological model

Regarding the model for bird populations, we have chosen a dynamic approach. We have adopted the Beverton-Holt model (Beverton & Holt, 1957) which accounts for the intra-specific competition for the

1. http ://ec.europa.eu/agriculture/rica/index-fr.cfm

resources and the density dependance as follows :

$$\mathrm{N_{s,r}(t+1) = N_{s,r}(t).\frac{R_{s,r}}{1+\frac{N_{s,r}(t)}{M_{s,r}(t)}}} \quad (A.1)$$

where s is the species and r the PRA. The state $N(t)$ stand for the bird abundances on year t. The $R_{s,r}-1$ coefficient corresponds to the growth rate. This parameter takes into account the characteristics specific to each species such as the clutch size, mean reproductive success, the number of clutches per year. The value $M_{s,r}(t)$ is related to the inverse of the intra-specific competition coefficient. In other words, the product $M_{s,r}(t)$*$(R_{s,r}-1)$ represents the carrying capacity of the habitat r and the value $M_{s,r}$ captures the ability of the habitat to host the species. For computing the abundance at year $t+1$, we have chosen to involve the $M_{s,r}$ coefficient at year t, which shows a delay between the time the habitat is modified and the time when change affects the species. Habitat index $M_{s,r}$ is assumed to depend on agricultural activities as follows :

$$\mathrm{M_{s,r}(t) = \beta_{s,r} + \sum_k \alpha_{s,r,k}.A_{r,k}(t)} \quad (A.2)$$

The k indices stand for the agricultural systems. $A_{r,k}(t)$ represents the share of the region r dedicated to agricultural system k. The α and β coefficients, specific to each species, show how such a species s respond to the various agricultural uses in a given PRA r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a PRA r. Hence, one feature of this ecological model is that the abundances depend on the agricultural systems. It is noteworthy that the model is implicitely spatially spatialized.

2.3. The economic model of the farmer

Each PRA r is managed by a representative farmer who selects activities along time. The farmer determines the surfaces $A_{r,k}(t)$ of each activities k in order to maximise his utility, computed by his expected income and a risk aversion term. The income depends on the direct income of the agricultural activity and public incentives (eq. I.6). For his utility fonction, the regional farmer defines an expected income $E(Income_r(t))$ and its risk $a.Var(Income_r(t))$ function of the agricultural activities repartition he choose (eq. A.4). The program of maximising of farmer's utility is defined by equation A.6.

$$\mathrm{Income_r(t) = \sum_k gm_{r,k}(t).A_{r,k}(t).(1+\tau_k)} \quad (A.3)$$

$$\mathrm{Utility_r(t) = E\ (\ Income_r(t)\ )\ -\ a\ .\ Var\ (\ Income_r(t)\ )} \quad (A.4)$$

$$\max_{A_{r,k}} (\ \mathrm{Utility_r(t)}\ ) \quad = \quad \max_{A_{r,k}} (\ \sum_k \mathrm{\widehat{gm}_{r,k}(t).A_{r,k}(t).(1+\tau_k)} \quad (A.5)$$

$$- \quad a\ .\ \sum_k \sum_{k'} \mathrm{cov_{k,k'}(t).A_{k,k'}(t).A_{k',k}(t).(1+\tau_k).(1+\tau_{k'})}\ ) \quad (A.6)$$

The income is affected by public incentives τ_k on different activities k which take the form of taxes ($\tau_k < 0$) or subsidies ($\tau_k > 0$). We have chosen to keep the incentive or tax stable along time in this modeling. It is computed as a rate τ of mean gross margin per surface unit. At any time, the farmer knows exactly the incentive levels τ_k. But we insert uncertainty on gross margin per hectare. We determine the economic volatility for each activity in any PRA with historical data. We draw gross margins from 2009 to 2050 in a normal distribution computed with the mean and the standard deviation of the temporal set. The variability on income represents both production and climate uncertainty ans market uncertainty. When the farmer makes his choice, he do not know gross margins at t time but he uses information about the 7 previous years ($\widehat{gm}_{r,k}$ and $cov_{k,k'}$) [1] . The coefficient a represents the aversion level of the farmer : the higher a is, more risk-averse the farmer is. In particular $a = 0$ means the farmer is risk neutral, he makes his choices taking into account the expected income only.

When maximizing his revenue, the standard agent must comply with two constraints at every time :

$$|\mathrm{A_{r,k}(t) - A_{r,k}(t-1)}| \leq \varepsilon.\mathrm{A_{r,k}(t-1)} \tag{A.7}$$

$$\sum_k \mathrm{A_{r,k}(t) = A_r} \tag{A.8}$$

The first constraint (eq. A.7) corresponds to a technical constraint where the coefficient ϵ stands for the rigidity in changes (for example, $\epsilon = 0$ means the surfaces remain constant). The second constraint (eq. A.8) ensures merely that the total surface per region remains constant.

For any region, the representative farmer defines the share of his land which he dedicates to the various practices relying on a linear optimization under constraints. Hypotheses underlie this model. We suppose that the system is at equilibrium in various dimensions and that the farmer's choice does not alter such equilibrium. First, we consider the farmers as price-takers. Second, we admit that food consumers have changing habits, though the demand remains constant. Third, the technological level does not evolve : there is neither improvement from research nor the quest for improved productivity from the farmers. The mean yield (which this revenue per surface unit depends on) is kept flat. The same is applied to the labor market : we do not study the number of farmers, which is assumed constant.

2.4. Model coupling and public decisions

Ecological and economic models described previously are linked by the agricultural activities as depicted by figure A.1. With the objective of maximising his utility, the representative farmer exhibits pattern of activities $A_{r,k}(t)$ which are injected into the ecological model through the habitat $M_{s,r}(t)$: the agricultural states are the outputs of the economic model and the inputs of the ecological model. The farmer's economic choices thus condition bird biodiversity $N_{s,r}(t)$ associated with the habitats.

We can therefore now add a new agent to our model : the public stakeholder. The decision-makers impact the bio-economic system through an economic instrument : they use a set of incentives and taxes which impacts the various agricultural practices, thus modifying their profitability. Thanks to their economical model, the farmers shape their land-use patterns in order to maximise their utility. These land-use rearrangements improve the global wealth while perturbing the evolution of ecological model and bird community dynamic. Decision-makers define their incentive/tax politcies depending on their ecological objectives and economic planning. For this purpose, the regulating agency must be able to evaluate the economic wealth and the biodiversity of the system he governs. It uses various indicators. We choose to focus on an economic indicator and ecological indicator to analyse the bio-economic trade-off.

From an economic standpoint, we use the national mean income per hectare (eq. A.9). It is computed from the mean gross margin of the 620 PRA. For sake of clarity, we represent this criterion after normalisation by their current value (2008).

$$\overline{\mathrm{income}}_{\mathrm{France}}(t) = \frac{\sum_r \mathrm{S_r}.\left[\sum_{\mathrm{k}} \mathrm{gm_{r,k}(t).A_{r,k}(t).(1+\tau_k)}\right]}{\mathrm{S_{France}}} \tag{A.9}$$

1. $\widehat{\mathrm{gm}}_{\mathrm{r,k}}(\mathrm{t}) = \frac{\sum_{t=t-8}^{t-1} \mathrm{gm_{r,k}(t)}}{7}$ and $cov_{k,k'}(t) = \frac{\sum_{t=t-8}^{t-1} (\mathrm{gm_{r,k}(t)} - \widehat{\mathrm{gm}}_{\mathrm{r,k}}(\mathrm{t})).(\mathrm{gm_{r,k'}(t)} - \widehat{\mathrm{gm}}_{\mathrm{r,k'}}(\mathrm{t}))}{7}$

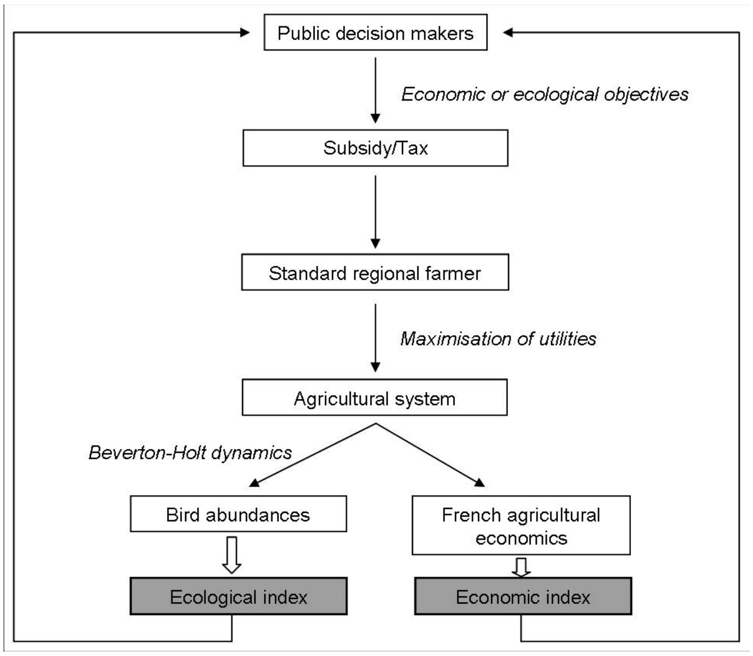


FIGURE A.1 – Model coupling : Farmers maximise their utility and adjust theirs activities pending on incentives. These choices affect french agricultural economics and bird's community dynamics

From an ecological standpoint, we have selected the STOC indicator provided by the Vigie-Nature website [1]. This is a variation indicator of abundances with respect to the reference year 2008. An aggregated STOC indicator is built for the farmland specialist species class (Julliard *et al.*, 2006). It is computed as the geometric mean of the indicators of the species considered in the class (eq. A.10). In these aggregated indicators, the abundance variation of each species is taken into account similarly, independently from the abundance value.

$$\mathrm{STOC}_{\mathrm{r,class}}(\mathrm{t}) = \prod_{\mathrm{s} \in \mathrm{class}} \left(\frac{\mathrm{N}_{\mathrm{s,r}}(\mathrm{t})}{\mathrm{N}_{\mathrm{s,r}}(2008)} \right)^{1/\mathrm{Card(class)}} \tag{A.10}$$

To understand land-uses organisation, we use a diversification indicator. It represents the minimal number of activities to obtain 80% of the agricultural surface (eq. A.11)

$$\mathrm{diversity}_{\mathrm{r}}(\mathrm{t}) = \min \left(\mathrm{Card(K)} \ , \ \sum_{\mathrm{k} \in \mathrm{K}} \mathrm{A}_{\mathrm{r,k}}(\mathrm{t}) \geq 0.8 * \sum_{\mathrm{k}=1}^{14} \mathrm{A}_{\mathrm{r,k}}(\mathrm{t}) \right) \tag{A.11}$$

3. Scenarios and results

3.1. Scenarios

The key parameter which characterizes each scenario is the vector τ representative of the subsidy or the tax. We have developed 3 scenarios :

- Statu quo scenario : no incentive with $\tau = 0$ for all activities
- Crop scenario : taxes for COP (class (1) in appendix B) with $\tau_{COP} = -50\%$.
- Grassland scenario : incentives for the non-intensive grasslands (classes (4), (5), (6), (7) in appendix B) with $\tau_{grassland} = 50\%$.

1. http ://www2.mnhn.fr/vigie-nature/

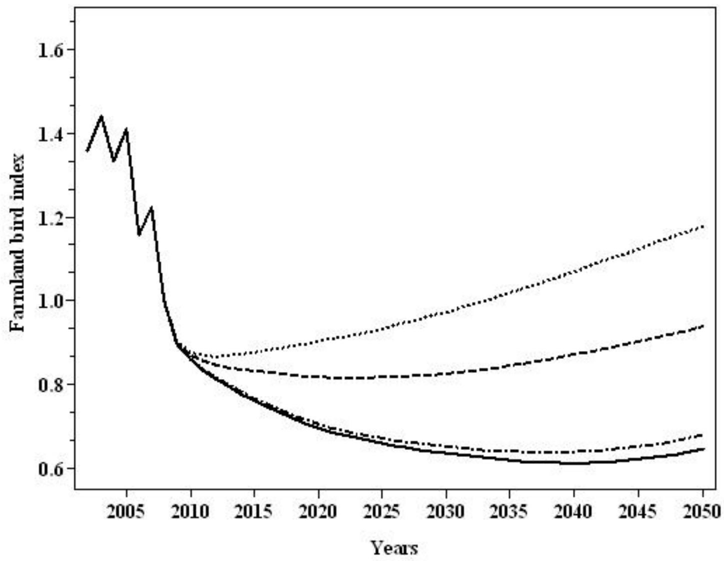


FIGURE A.2 – Evolution of biodiversity index under statu quo scenario with 4 risk aversion levels : full line risk neutral (a=0), alternative line weak aversion (a=10^{-7}), dashed line medium aversion (a=10^{-6}), dotted line strong aversion (a=3.10^{-5})

- High Environmental Quality (HQE) scenario : taxes on the COP, redistributed to the non-intensive grasslands with $\tau_{COP} = -50\%$ and $\tau_{grassland} = 50\%$.

When we study a scenario, we run 50 simulations with different random drawings of gross margins gm. Then we calculate at any time the mean of the 50 simulations for ecological and economic indicators and their variability. Each scenario can be tested for different risk aversion levels a.

3.2. Without public incentives, risk aversion impacts bio-economic performances

3.2.1. Possible ecological indicator improvement without public policies

The various forecasts of figure A.2 illustrate results with contrasted ecological performances. When the farmer is characterized by a weak aversion to risk, his selection of agricultural activities leads to a decreasing farmland bird index. Conversely, when he demonstrates a sufficient risk aversion, he selects a land-use sharing which recovers the farmland bird indicator. Thus it seems possible to concile ecological performance and agricultural activity without public intervention : the more averse to risk the standard agent is, the higher the ecological performance is.

3.2.2. Risk aversion drives bio-economic performances without public policies

Still within the statu quo scenario, we add the economic dimension with figure A.3 : we are monitoring the bio-economic performance at various levels of risk aversion. We observe a set of contrasted trajectories : some of them are beneficial to the biodiversity, while others promote the economic indicator. Thus the ecological and economic performances are adversely correlated : there is no trajectory optimizing both the economic development and the biodiversity. Without any public intervention, the risk aversion plays a significant role in the bio-economic performances the farmer achieves along time.

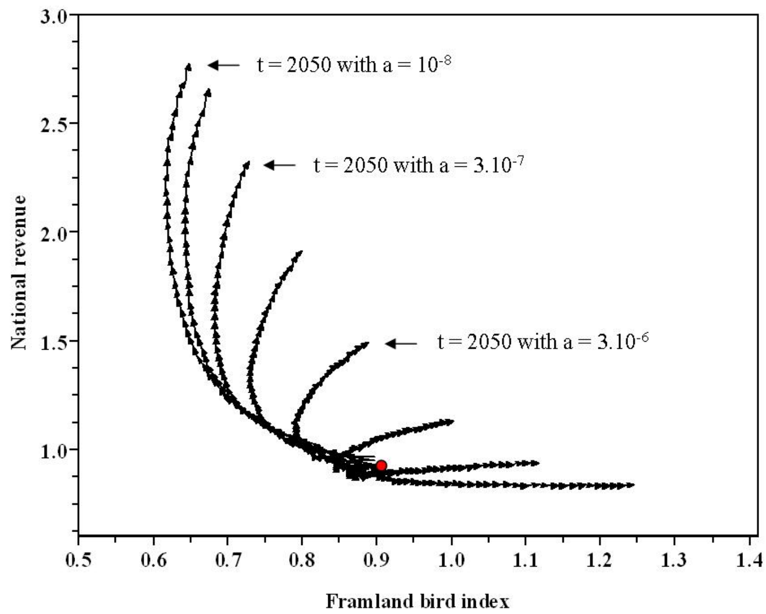
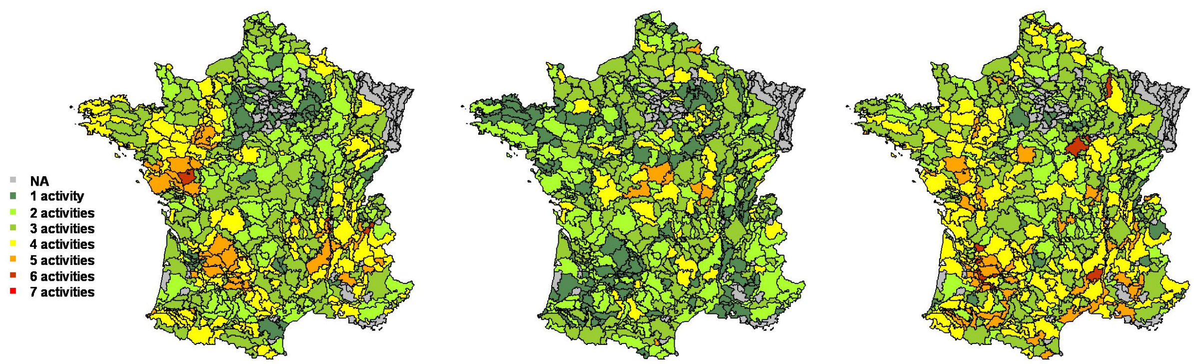


FIGURE A.3 – Bio-economic performances with statu quo scenario for several risk aversion levels (left to right : 10^{-8}, 10^{-7}, 3.10^{-7}, 10^{-6}, 3.10^{-6}, 10^{-5}, 3.10^{-5}, 10^{-4}). All the trajectories start in the same point (red) : Income = 0.92 and Framland bird index = 0.89.

3.2.3. Risk aversion modifies diversification level

In order to better understand the trade-off of figure A.3, we have illustrated on figure A.4 the diversification indicator in 2008 (T=0) and in 2050, with two different risk management strategies in the frame of the statu quo scenario. We observe that the good ecological performances, achieved with a high risk aversion on graphs A.2 and A.3, are related to a high diversification of agricultural practices/activities within the regions : for the great majority of regions, the diversification indicator has increased versus 2008. Conversely, with a weak risk aversion, we fully observe a territory specialization : the diversification indicator decreases in comparison to 2008.

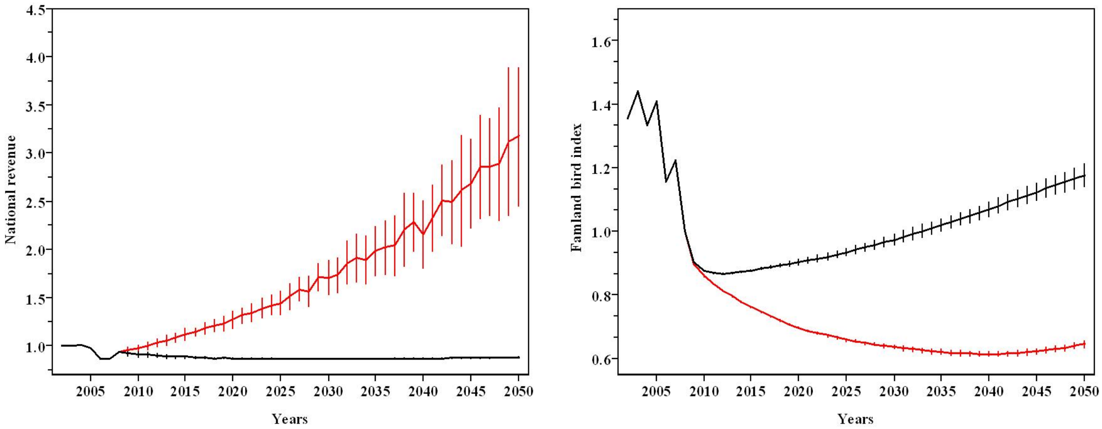


(a) Agricultural diversity en 2008

(b) Agricultural specialisation en 2050 with a weak risk aversion (a=10^{-7})

(c) Agricultural diversification en 2050 with a strong risk aversion (a=3.10^{-5})

FIGURE A.4 – Diversification indicators with statu quo scenario in 2008 and 2050 with different risk aversions



(a) National income evolution (b) Farmland bird index evolution

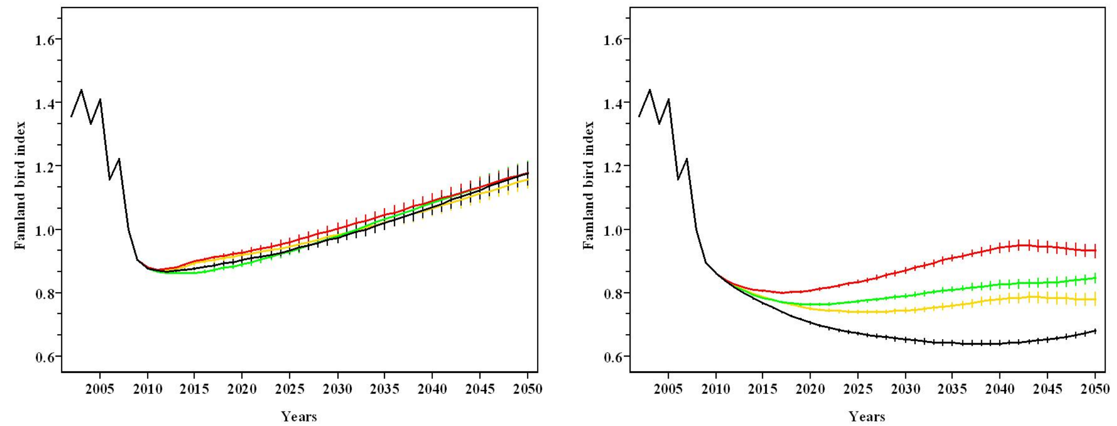
FIGURE A.5 – Risk management impact on bio-economic performances and their standard deviation for a strong risk aversion level (black, a=3.10^{-5}) and a no risk aversion (red, a=0) with the statu quo scenario.

3.3. Without public incentives, risk aversion impacts only economic performances volatility

Figure A.5 allows studying the impact of income risk management on the volatility of ecological and economic performances in the frame of the statu quo scenario. Such impact on ecological indicator differs from the one on economic indicator. As already mentioned on figure A.3, globally the mean income increases more when the risk aversion is weak. But the volatility of such performance increases as well when the aversion weakens. A low risk averse agent may expect globally a high income growth, but the uncertainty about his incomes is getting high too. Conversely, a very high aversion to risk shall not allow a high income growth, but the associated risk shall be very low. On the contrary the ecological indicator shows a similar dispersion (3%) for both levels of risk version. The volatility of the farmland bird indicator is not sensitive to the risk management strategy applied by the farmer. Thus, with regard to the biodiversity, the risk aversion shall impact the mean value of the index, but not its volatility. A high economic risk is not systematically correlated with an ecological risk. The aversion to market risk is necessary to limit the risk on incomes, but seems useless for limiting the variance of the ecological indicator.

3.4. Public incentives sensibility depends on risk aversion

After having analyzed the agro-ecosystem without public intervention, we add political incentives. Figure A.6 shows that public incentives alter the trajectories of the farmland bird indicator : the ecological performances achieved with the three incentive strategies are better than with the statu quo scenario. Political scenarios (based only on economic levers) have a sufficient impact on farmers that they adopt more eco-friendly practices. Nevertheless, the marginal benefit of public policies is higher with farmers who demonstrate a low aversion to risk. Moreover all scenarios do not show the same efficiency. The more interesting scenario from an ecological perspective is the HQE strategy : the combination of both incentives (taxes on COP and subsidies on extensive grasslands) is more favorable to the biodiversity than these separated strategies (Cereal and Grassland scenarios). Thus there is a true synergy between these public interventions. Regarding the dispersion of the ecological indicator, we observe that it is not affected by these strategies. The public incentive strategies do not alter the small volatility of the ecological indicator for both risk aversion level.



(a) Ecological performance and variability with a strong risk aversion level (a=3.10^{-5})

(b) Ecological performance and variability with a weak risk aversion level (a=10^{-7})

Figure A.6 – Ecological performances and their standard deviation with cereal scenario (yellow), grassland scenario (green), HQE scenario (red) and statu quo scenario (black)

4. Discussion

4.1. A risk averse farmer adopts spontaneously ecofriendly activities

The first part of the study shows that a conciliation of both biodiversity and agricultural activities is achievable without public intervention. An aversion to risk is sufficient for the farmer to organize spontaneously his activities in an eco-friendly manner. Indeed, a risk aversion leads to the diversification of the various agricultural activities aimed at mitigating the risk on revenues due market fluctuations. This activity diversification allows diversity of habitats and available resources (as highlighted by Benton *et al.* (2003)). The empirically rich model presented here, based on utility maximization without any ecological awareness, allows us to find again the theoretical result stated by Quaas *et al.* (2007) : the farmer does not need necessarily to be aware of environmental issues or to get economical benefits from ecosystemic services to select a land-use strategy allowing a sustainable development of the biodiversity . We complement this result by observing that in theory incentives seems not necessary as well to achieve a sustainable management of the agro-ecosystem. A high risk aversion shall lead to an activity diversification beneficial to the birds. The variety of observed bio-economic performances shows that, without any public intervention, the risk aversion is the main driver of the farmer strategy and that its impact is strong enough to reverse the bio-economic performances.

4.2. Actual farmer adopts a weak risk averse behavior, leading to agricultural specialisation

Today the biodiversity indices show a decline trend of farmland bird species (Julliard *et al.*, 2006). However, agents are rather averse to risk. Thus we could expect a better level of biodiversity. Nevertheless agriculture is an economic domain of its own that, because of its characteristics (instability due to the food demand rigidity, constraints on space utilization), cannot self-control. Early, public intervention has become necessary to ensuring an efficient agriculture. Consequently farmers have been working since 30 years in an economic system highly driven by public interventions, protecting them -among others- from the market fluctuations : potential losses due a financial issue in a high-specialization context are mitigated by revenues from incentives. This revenue insurance has led to a change of their activity management. This management is much more similar to what we observed with low-averse agents. Then the PRA tend to specialization : this explains today decline of the biodiversity indices. In the field, the farmer strategy does not lead them towards an eco-friendly management. These are the conclusions drawn by Alavalapati *et al.* (2002) and Shi & Gill (2005), whose studies were based directly on field observations.

4.3. In a specialization context, public policies improve ecological performances

It is important not to misinterpret these results out of their context. A premature conclusion from this study could promote stoppage of incentives. Indeed, farmers would come back to a management sensitive to the market risk aversion, leading to activity diversification, favorable to the biodiversity. However this very economical solution is out of scope from a State perspective ; not only it is almost impossible to cut off all incentives without social turmoil, but the rationale for having set these incentives has not disappeared : they are still indispensable. Against very low international prices for agricultural products, they allow to avoid a collapse of French agriculture, to maintain a competitive economy and agricultural revenues consistent with the other economic domains. The State is so committed to go with an agriculture support policy.

In the scope of this interventionist policy indispensable to the agricultural domain, the second part of the study shows that scenarios based on financial public interventions allow recovering ecological performance. These incentives drive the farmers to develop eco-friendly activities and to keep a better level of diversification. We show in this study that synergies exist across the incentive policies. Thus there are achievable incentive policies aimed at optimizing the ecological performance. Aligned with intuition, a very risk averse agent shall have already set up a strategy of activity diversification, and shall be consequently less sensitive to public incentive policies. In the much protected context of the agriculture domain, the agents show a strategy very similar to the low averse farmers, and shall consequently be very sensitive to the public incentive policies. This leads us to foresee good efficiency of public policies for agriculture. At the opposite of a premature conclusion, this study promotes the setting up of relevant public incentive policies, proving their efficiency on the biodiversity.

4.4. Public policies do not enhance ecological risk

Aware of the risk of irreversibility (Hein & Ierland, 2006) threatening the ecosystems, a last step before definitive validation of public policies for agriculture regards the volatility of ecological performance. Indeed if ecological variability enhances too much, a little part of set of possibilities could be a irreversible state of the ecosystem. Thus we would be restrained to eliminate this strategy, reducing the number of available public policies. The income risk alters the mean level of economic performance, but also its scattering : an improved mean performance is associated to a higher variance. But this correlation does not apply on the ecological index, which demonstrates in our study a steady volatility around 3%. The dynamic model of bird population that we have selected considers species sensitivity to the habitat through the carrying capacity. The agricultural activities do not affect bird populations directly ; the birds are essentially sensitive to the general trend of space organization selected by the farmer. But the yearly weather variations which do not affect this trend on the long term do not alter this dynamics. The income variability, mainly translated into yearly variations, is so erased and does not increase the variance of ecological performance. This effect exists independently from the selected public policy, and regardless of the level of risk aversion of the farmers. Whatever the economic risk management, the application of scenarios does not raise any risk of variance increase which could lead the agro-system into irreversible states. Thus it is not necessary that the political stakeholder estimates the mean risk aversion of the farmers and adapts his incentive scenario accordingly.

5. Conclusion

This study has enabled us to develop a robust bio-economic model, concerned about the consistency with terrain observations, aimed at evaluating the public incentive policies on agriculture and their impact on the agro-ecosystem. The objective of this publication consists in adding income uncertainty to the classic approach in bio-economics based on public policy scenarios. We observe that a risk-averse strategy is sufficient for the farmer to develop diversified agricultural activities favorable to the biodiversity. However in the agriculture domain highly driven by public incentives, the agents develop agricultural activities similar to the low risk-averse farmers, leading to an agricultural specialization and to the decline of the farmland bird species. We show that public scenarios based on economic incentives allow improving the mean performance of biodiversity indices without increasing the variance, and independently from the agent level of risk aversion. This pleads for a relevant use of public incentive scenarios aimed at developing a sustainable agriculture.

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Appendices

A - 20 farmland bird species

(1) Cirl bunting, (2) Corn bunting, (3) Common buzzard, (4) Common pheasant, (5) Common quail, (6) Grey partridge, (7) Common stonechat, (8) Common whitethroat, (9) Hoopoe, (10) Lapwing, (11) Linnet, (12) Meadow pipit, (13) Rad-backed shrike, (14) Red-legged partridge, (15) Rook, (16) Sky lark, (17) Whinchat, (18) Wood lark, (19) Yellow hammer, (20) Yellow wagtail.

B - 14 agricultural systems (Otex)

(1) Cereal, Oleaginous, Proteaginous (COP) ; (2) Variegated crops ; (3) Intensive bovine livestock breeding ; (4) Medium bovine livestock breeding ; (5) Extensive bovine livestock breeding ; (6) Mixed crop-livestock farming with herbivorous direction ; (7) Other herbivorous livestock breeding ; (8) Mixed crop-livestock farming with granivorous direction ; (9) Mixed crop-livestock farming with other direction ; (10) Granivorous livestock breeding ; (11) Permanent farming ; (12) Flower farming ; (13) Viticulture ; (14) Others associations

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Annexe B

Innovation rigidity and ecological-economic reconciliation in agriculture

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Abstract

For several decades, significant changes in bird biodiversity have been reported, especially in Europe. Agriculture, and more specifically agricultural intensification, is a major driver of these modifications. Taking into account these environmental impacts, agriculture nowadays aims at a more sustainable way of producing which would reconcile its economic and ecological functions. The objective of this paper is to give insights into the impact of public policies and financial incentives on both the conservation of biodiversity and farming production. We therefore develop a macro-regional model combining a community dynamics of 65 bird species impacted by agricultural land-use and an economic decision model for each French region. The ecological dynamic model is calibrated with the STOC (Time Survey of Common Birds) and AGRESTE (French land-uses) databases while the economic model relies on the optimization of the gross margin of the RICA (Network of Agricultural Accountant Information). We investigate different scenarios based on subsidies and taxes to study the impact of public policies on both biodiversity and agricultural economics. We show that simple economic instruments could be used to establish scenarios promoting economic performance and bird populations. The bio-economical analysis shows several solutions for the ecology-economy trade-off. These results suggest that many possibilities are available to develop multi-functional sustainable agriculture. We focus here on the impact of the innovation rigidity and we show that a too big innovation ability is not necessary favourable to the biodiversity because of the inertia of the biological systems.

Keywords : Common birds, Agriculture, Bio-economic modeling, Public policy, Scenario

1. Introduction

Global changes in European agriculture in recent decades, including intensification and land abandonment, have significantly modified farmland biodiversity. The pressure is particularly strong on bird populations which have undergone severe and widespread decline (Donald *et al.*, 2006, Julliard *et al.*, 2004, Donald *et al.*, 2001, Chamberlain *et al.*, 2000, Krebs *et al.*, 1999). Such erosion is mainly due to a combination of habitat loss and degradation of habitat quality altering the nesting success and/or survival rates (Benton *et al.*, 2003). In this context, the European Union, aiming at halting biodiversity loss by 2010, has adopted the farmland bird index as an indicator of structural changes in biodiversity (Balmford *et al.*, 2003). In this perspective, of particular interest is the need to reconcile agricultural production and biodiversity (Jackson *et al.*, 2005). There is an extensive and increasing volume of literature concerning agri-environmental schemes and policies for multi-functional agriculture (Dobbs & Pretty, 2004). However, after 15 years of implementation of such instruments, the question whether providing habitat quality conflicts with management for agricultural production remains controversial (Butler *et al.*, 2007, Kleijn *et al.*, 2006, Vickery *et al.*, 2004). To address agro-environmental sustainability, both economic and ecological criteria must be considered. As pointed out by Perrings *et al.* (2006) et Hughey *et al.* (2003), there is an urgent need for approaches that integrate economic criteria in conservation problems. Reinforcing such analyses and examining forms of farming allowing for the joint sustainability of biodiversity and agricultural production requires interdisciplinary research. Such work relies upon the development of interdisciplinary concepts, quantitative methods and integrated models that adequately incorporate the complex interdependencies between farmland ecosystems and economic systems.

The present paper deals with such modeling issues regarding agro-environmental sustainability. A bio-economical model is developed to study the joint sustainability of agricultural land-use and bird biodiversity. This model questions the way to evaluate the ecological and economical dimensions and to rank habitat management decisions in order to assess the relevance of different policies, notably with respect to sustainability. To deal with sustainability, approaches such as ecological economics (Drechsler & Watzold, 2007) suggest studying environmental and economic effectiveness simultaneously, stressing the relevance of multi-criteria approaches. However, few economic studies cope with the spatial and temporal dynamics of biodiversity in this context (Hammack & Brown, 1974). In this vein, a range of spatially explicit models exist that aim at assessing consequences of different land use patterns for various environmental and economic criteria (Swihart *et al.*, 2003, Irwin & Geoghegan, 2001). Nevertheless, most of these models are static, they restricts the ecological processes taken into account. Moreover, they usually do not incorporate important economic drivers (e.g. agricultural prices, subsidies) that affect the returns of different land-use patterns.

The bio-economic model proposed in the present paper is in direct line with these considerations. First the model is dynamic : it articulates ecological and economic compartments and it adopts a multi-criteria perspective. Moreover, it offers a spatialized perspective as it is built up at a macro-regional scale and its calibration relies on French regional data of both land-use and bird abundance. The objective of this study is to analyze how we can significantly drive the bio-economic model with financial incentives and have both interesting ecological and economic performances. We focus on the bio-economic trade-off. Then we study the impact of the innovation ability on this trade-off. The paper is organized as follows. The first section presents our bio-economical model. The second section describes the scenarios which we study here and the results. The third section is devoted to the discussion

2. The bio-economic model

2.1. The ecological model

To assess the ecological performance, we here chose to focus on common bird populations and related indicators (Julliard *et al.*, 2006). Although the metric and the characterization of biodiversity remain an open debate (Le Roux, 2008, MEA, 2005), such a choice is justified for several reasons (Ormerod & Watkinson, 2000) : (i) Birds lie at a high level in the trophic food chains and thus capture the variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation and viability (Sekercioglu *et al.*, 2004). (iii) The availability of data and ecological knowledge : Birds belong to the most studied taxa and many databases are available. (iv) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens.

Regarding the model for bird populations, we have chosen a dynamic approach. We have adopted the Beverton-Holt model (Beverton & Holt, 1957) which accounts for the intra-specific competition for the resources and the density dependence. The carrying capacity is depending on the quality of the habitat, computing by the land-uses. So bird dynamics are dynamic and evolve function of the evolution of land-uses. Bird populations are estimated for each region. The ecological model is so implicitly spatialized.

2.2. The economic model of the farmer

We consider 21 regions of France. Each region is managed by a representative farmer who selects land-uses along time. The farmer makes his choice in order to maximise his revenue given rigidity and technical constraints. His revenue depends on two parameters : unit gross margin and public incentives. Implicitly the gross margin is computed from the regional output of each activity and its sale price. Here the sole reason for the direct revenue to change along time comes from the variation of surface allocated to such use. This income is affected by public incentives τ on different uses which take the form of subsidies ($\tau>0$) or taxes ($\tau<0$). We have chosen to keep the incentive or tax stable along time in this simplified prototype. It is computed as a rate τ of mean gross margin per surface unit.

When maximizing his revenue, the standard agent must comply with two constraints at every time. The first constraint corresponds to a technical constraint, which drives the rigidity (with the parameter ϵ) in changes. The higher ϵ, the larger the changes performed by the farmer at any time. This parameter could be read as the farmer innovation ability. The second constraint ensures merely that the total surface per region remains constant.

For any region, the representative farmer defines the share of his land which he dedicates to the various practices relying on a linear optimization under constraints. Certain hypotheses underlie this model. We suppose that the system is at equilibrium in various dimensions and that the farmer's choice does not alter such equilibrium. First, we consider the farmers as price-takers. Second, we admit that food consumers have changing habits, though the demand remains constant. Third, the technological level does not evolve : there is neither improvement from research nor the quest for improved productivity from the farmers. The mean yield (which this revenue per surface unit depends on) is kept flat. The same is applied to the labor market : we do not study the number of farmers, which is assumed constant.

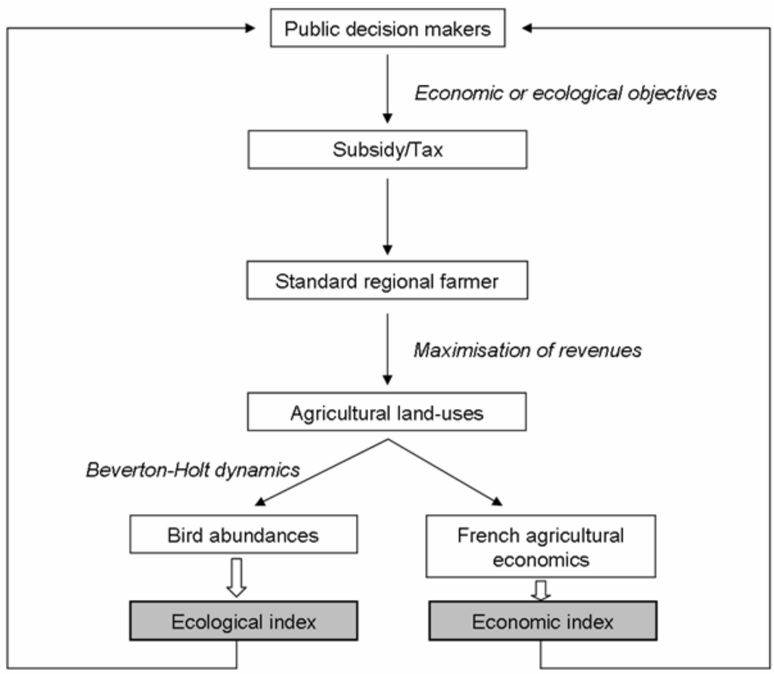

FIGURE B.1 – Model coupling : Farmers maximize their income and adjust theirs land-uses pending on subsidies. These choices affect french agricultural economics and bird's community dynamics.

2.3. Model coupling and public decisions

Ecological and economic models described previously are linked by the land-uses as depicted by figure B.1. With the objective of maximising his revenue, the representative farmer exhibits pattern of land-uses which are injected into the ecological model through the carrying capacity : the agricultural states are the outputs of the economic model and the inputs of the ecological model. The farmer's economic choices thus condition bird biodiversity associated with the habitats.

We can therefore now add a new agent to our model : the public stakeholder. The decision-makers impact the bio-economic system through an economic instrument : they use a set of incentives and taxes which impacts the various agricultural practices, thus modifying their profitability. Thanks to their economical model, the farmers shape their land-use patterns in order to maximise their revenue. These land-use rearrangements improve the global wealth while perturbing the evolution of ecological model and bird community dynamic. Decision-makers define their incentive/tax politics depending on their ecological objectives and economic planning. For this purpose, the regulating agency must be able to evaluate the economic wealth and the biodiversity of the system it governs. It uses various performance indicators of the system. However there is no holistic criterion, representing all dimensions of the system. So we choose to focus on an economic index and ecological index to analyse the trade-off between them.

From an economic perspective, we use the national mean income per unit surface. It is computed from the mean gross margin of the 21 regions and represents a mean approach of the problem. For sake of clarity, we represent this criterion after normalisation by their current value (2008) on the next graphs (fig. B.2 et B.4). From an ecological point of view, we have selected the STOC index provided by the Vigie-Nature website[1]. This is a variation index of abundances with respect to the reference year 2005. An aggregated STOC index is built for the farmland specialist species (Julliard *et al.*, 2006). Mouysset *et al.* (2011) show

1. http ://www2.mnhn.fr/vigie-nature/

these species are more relevant for this kind of study than the generalist habitat species. It is computed as the geometric mean of the indices of the species considered in the class. In these aggregated indices, the abundance variation of each species is taken into account similarly, independently from the abundance value.

2.4. Model calibration

We selected the metropolitan region as the unit of spatial scale. We split France into 21 regions (Corsica excluded). On the ecological side, the STOC database developed by the Museum National d'Histoire Naturelle provides the data related to the bird abundances. Among the 175 species monitored by this program, we have selected the 65 species used as a reference for the European Farmland Bird Index. The regional abundances for the years 2001 to 2007 are available for each of these species. According to their relation with the habitat, we can classify the 65 species into four main classes : generalists, farmland specialists, forest specialists, and urban specialists. Agronomical data measuring the surfaces of the various agricultural practices are published by Agreste (Statistics Service of the Department of Agriculture) for the years 2002 to 2007. Finally the economic data relating to the gross margins are derived from RICA (Réseau d'Information Comptable Agricole). We use 10 classes of land-uses. The first step consists in determining the Beverton-Holt parameters through a calibration. For each of the 65 species, we must estimate the growth parameter constant over the region as well as the carrying capacity specific to each region. We use a least square method to minimize errors between the observed abundances as issued from STOC and the values derived from the model. The errors of calibration are small (between 4% and 6% for the illustrated species) and the historical data do not go beyond the confidence interval (coming from the least square standard errors of calibration). Comparing the historical data with the model-generated data, we note that the model tends to smooth the variations of the observed data.

3. Scenarios et results

3.1. Scenarios

Once the ecological and economic models have been calibrated, we can use them to analyse the impact of public policies. The selected timeframe runs up to 2050, i.e a 43-year forecast. Selecting a shorter timeframe could consequently hide interesting long-term effects due to the inertia of the models. We define scenarios for various incentive/tax policies aimed at analysing the impact of governmental decisions on both the economy and agricultural biodiversity. In all scenarios described in this article, surfaces allocated to the forest and non-farming area remain steady in all times : we focus only on the evolution of the farmland use. This approach highlights the impact of the composition of farmland uses on biodiversity, the global surface remaining constant.

We have developed 3 scenarios :

- Crop scenario : incentives for COP (cereal, oleaginous, proteaginous).
- Grassland scenario : incentives for the extensive grasslands.
- High Environmental Quality (HQE) scenario : taxes on the COP, redistributed to the extensive grasslands.

The first two scenarios are very simplified variants of current policies. The first scenario represents policies which support COP, for example with the objective of developing bioenergies. The second scenario corresponds to a policy of extensification by the development of permanent grasslands. The third scenario, slightly more complex, plays at two levels : the tax on the COP and the incentive for permanent meadows. We study the synergy of these two levels. This scenario is of specific interest for the planner : the required budget

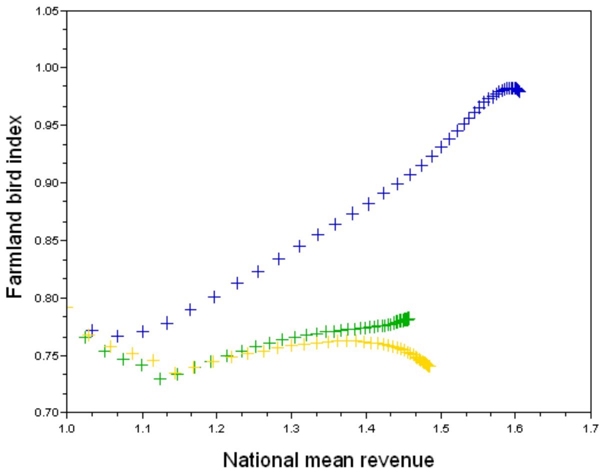

FIGURE B.2 – Impact of scenarios (yellow : Bioenergy, green : HQE, blue : Redistribution) on the bio-economic trade-off between the mean revenue and the farmland bird index (τ = 50%, ϵ = 10%).

is lower than for the two first scenarios, as the incentives for the permanent meadows are compensated by the taxes on the COP. This scenario is more realistic from an economical perspective in the sense that it is partially self-funded for the public stakeholder. In all three cases, this is a simplified model since the same policy is applied to all regions, whatever the economic or habitat features.

We study these 3 scenarios with a combined bio-economic approach coupling both economic and ecological outputs to better understand the trade-off. For this step, we keep the innovation ability constant. Then we analyse the impact of this parameter on the bio-economic trade-off. All graphs display the results similarly. Each trajectory is composed of 43 points, corresponding to each years of the timeframe from 2008 to 2050. All trajectories start from the same point at the lower left corner of the figure.

3.2. Impact of scenarios on the bio-economic trade-off

Figure B.2 allows for a comparison of the 3 scenarios based on identical ϵ and τ parameters, set respectively at 10% and 50%. We observe that we obtain contrasted results pending on the scenarios. If all of them improve the economic index, it is not the same situation with the ecological index. The Crop scenario seems to not to be favourable for the birds in a long term. From the economic perspective, the Grassland scenario is the least efficient. We note that the HQE scenario is the one which generates the best results on both index. The marginal effect of this two-actions scenario is positive for both indicators. But it is particularly interesting for the STOC index, which goes close to the reference value. The gain with the HQE scenario (compare to the others) is around 7% for the economic index while it is 30% for the ecological indicator.

3.3. Impact of incentive's level on the bio-economic trade-off

Figure B.3 displays the trajectories of the HQE scenario for 4 levels of incentive. We remark that depending on the value given to τ, we get a full range of trajectories covering the set of possibilities. No trajectory exhibits an improvement of the system for either ecological or economic dimensions. However, some trajectories are more eco-efficient (for example with τ = 0.1 and 0.5), while others show a better economic effectiveness (τ = 1 and 1.8). We note that certain values of the national mean income can be obtained for all trajectories (for example revenue = 1.3). This value is not reached at the same speed for the four

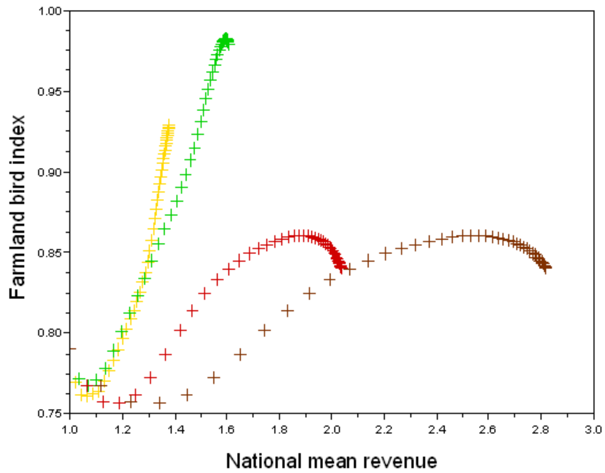

FIGURE B.3 – Impact incentive's level τ (yellow $\tau = 10\%$, green $\tau = 50\%$, red $\tau = 100\%$, brown $\tau = 180\%$) on the bio-economic trade-off between the mean revenue and the farmland bird index ($\epsilon = 10\%$).

trajectories : with $\tau = 1.8$ (1, 0.5, 0.1 respectively), this income is obtained respectively after 4 years (6 years, 11 years, 40 years). The more drastic the policies (higher level of tax and incentive), the faster the income. However, for a given income, the slowest trajectory is the most eco-effective : for a national mean income of 1.3, the STOC index of farmland specialists provides a level of 0.93 with $\tau = 0.1$, against only 0.77 for $\tau = 1$.

3.4. Impact of the innovation ability on the bio-economic trade-off

With the figure B.4, we illustrate trajectories of the HQE scenario for 4 levels of innovation ability for one level of incentive. We find exactly the same kind of results for the other levels of incentive. This graph shows an impact of farmer rigidity of changes on the bio-economic trade-off. On the economic side, the smaller is the ϵ parameter, the smaller is the speed of the revenue. But for all the cases, the model converges to the same long-term revenue. On the ecological side, the rigidity affects also the speed of growth of the STOC index, but there is a second effect, more interesting. The level of ϵ can not stop the bird decrease at the end of the trajectories, but still drives the ecological optimum. The smaller the rigidity, the higher is the optimum reached along the projection. To obtain good ecological results with a bird favourable scenario, it is not necessary to have too big innovation ability.

4. Discussion

4.1. Ecological-economic reconciliation

With this simplified bio-economic prototype, we have shown that both the ecological and economic performances are impacted by the public policies for agriculture and land-use. A basic economical instrument (incentive/tax) separates policies according to the two criteria. In line with the bio-economic literature (Drechsler & Watzold, 2007), it suggests that managing the agricultural practices in bio-economic terms is possible thanks to a simple economic distortion of the marginal revenues. The model illustrates how it is possible to build scenarios which appear favourable on the long term to both ecological and economic

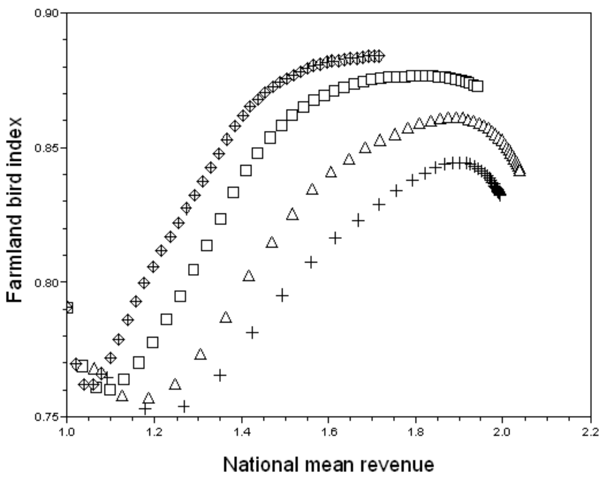

FIGURE B.4 – Impact of innovation ability parameter ϵ (diamond $\epsilon = 3\%$, square $\epsilon = 5\%$, triangle $\epsilon = 10\%$, plus $\epsilon = 15\%$) on the bio-economic trade-off between the farmland bird index and the national mean revenue for the Redistribution scenario ($\tau = 100\%$).

criteria. It should therefore be possible to define public strategies improving both farmer incomes and the avifauna. Our study suggests that the most favourable case occurs with the HQE scenario. This observation highlights that acting simultaneously on various incentives can improve performance from both the ecological and economic points of view.

4.2. The ecology-economy trade-off

As depicted by the figure B.3, no unique pareto optimum arises : even if both criteria are improved, it is always necessary to prioritise ecological and economical objectives. Consequently, a set of admissible strategies is available to bring together ecological and economic performances. The challenge consists in selecting which farming activities should be subsidised or taxed and which magnitude of incentive/tax is the most adequate in order to optimise trajectories for the set of selected ecological and economic criteria. However, along these trajectories, we have seen that the speed of change is very fluctuating. This variation gives another level of trade-off in terms of timeframe : how fast the public agency wants to reach the objectives. The growth rate is linked to the level changes requiring a larger budget. The total budget of the regulating agency is another key element of his strategy. In our model, we have not imposed budgetary constraint. However, in a larger perspective, decision-making support requires the integration of this budgetary limitation in the model. Indeed, some policies may be attractive from ecological and economic perspectives but not feasible in terms of public balance. Considering this global budget limitation raises the question of budget allocation to the regions. The answer to such a question is highly dependent on the selection of the economical indicator. Does the objective consist in reaching a maximal national mean, a maximal level for the poorest region or a minimal variability over the regions ? This spatial share of the global budget highly conditions the economical and ecological performance of each region, as well as for the whole country.

4.3. Importance of the innovation rigidity for the ecological performances

As expected, the innovation rigidity has an impact on the speed of changes for the ecological and economic performances. But a strong rigidity has also a positive effect on the ecological results. The farmer's changes in land-uses modify the carrying capacity in the Beverton-Holt dynamic, but do not directly alter the population size. Under relevant public incentives, farmers adjust their land-uses in an eco-friendly way. So the carrying capacity increases, as well as the populations with a delay. If farmers do not stop, they keep changing their activities over time : they reach the optimal land-use repartition but continue their changes to optimise their revenue. Over the optimal repartition, the carrying capacity decreases leading, with the delay, the decrease of the bird population. The faster the changes (low rigidity), the faster optimal repartition ; thus the shorter the growth period for bird populations, the lower the optimum reached by the ecosystem. Biological dynamics show stronger inertia than the economic system and we illustrate here that it is no useful to promote scenarios which develop a too big innovation rigidity. Biodiversity needs time to take benefits of eco-friendly activities. It will be particularly necessary to elaborate a dynamic policy, which changes when the optimum repartition is achieved. We can note that the optimal repartition is compiled with a set of land-uses : a bigger proportion is allocated to eco-friendly land-uses, but it is necessary to keep a diversification in the activities as shown by Benton *et al.* (2003).

5. Conclusion

This interdisciplinary model illustrates that reconciliation between agricultural production and conservation is possible. The research approaches using optimisation under constraints are more widely used in interdisciplinary problems, as the multi-functionality of agriculture. The objective of these methods is to build a plausible model to predict the impact of public policies on bio-economic performances. We develop a dynamic, spatialized and empirically rich model to study the links between bird biodiversity and agricultural policies. This kind of models can be an interesting aid to the decision, promoting interactions between research and society. The ex-ante analysis of public policies allows to test innovating scenarios, that we cannot directly test in reality, and to analyse strategic features of these policies to enhance their sustainability. We show that reconciliation between ecological and economic efficiency is possible using relevant public policies. We focus here on the impact of the innovation rigidity and we show that a too big innovation ability is not favourable to the biodiversity because of the inertia of the biological system.

Acknowledgements

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Annexe C

Note de la Commission Européenne

Science for Environment Policy
DG Environment News Alert Service
European Commission

7 April 2011

Partnering biodiversity and income on French farmlands

Striking a sustainable balance between making a profit and maintaining biodiversity on agricultural lands is challenging. A new French study has combined economic and ecological models and indicated that a simple combination of taxes and subsidies could promote economic performance on farms, whilst conserving bird populations.

Biodiversity has been declining in Europe, mainly as a result of habitat loss and a decline in habitat quality. Birds have been particularly affected, especially farmland species that have been affected by changes in European agriculture, such as intensified farming and land abandonment. To help identify vulnerable areas, the EU has adopted the Farmland Bird Index[1] as an indicator of biodiversity change. However, farmers also need to earn income and, in order to promote sustainable environmental schemes, both economic and ecological criteria must be considered.

The study proposed a bio-economic model to analyse possible policies to support biodiversity whilst ensuring a viable income for farmers. The ecological part of the model estimated the abundance of 34 bird species that live on French farmland. This is measured by the Farmland Bird Index that considers birds that can only live on farmland and an index used in the French Breeding Bird Survey (STOC) that considered both specialist farmland birds and more generalist birds.

The model considered the impact of four policy scenarios for the period up until the year 2050. These were a cereal scenario with subsidies for cereal crops, a grassland scenario with subsidies for permanent grassland, a double subsidy with subsidies for both cereals and grassland, and finally, a High Quality Environmental (HQE) scenario which taxes cereal crops and redistributes the taxes as subsidies for permanent grassland. This last scenario had the lowest cost of all the scenarios.

Economically speaking, the HQE scenario would provide the most economic benefit in 2050 at both a regional level (an increase in income of about 83 per cent from 2008) and a national level (an increase in income of 60 per cent from 2008). The grassland and the double subsidy scenarios are the least efficient. Ecologically speaking, the HQE scenario would be again more effective as measured by both indexes, whilst the Cereal scenario would be least effective.

The study analysed the HQE policy in more depth with four levels of taxation for cereal crops (and therefore subsidies for grassland). With greater taxes (and subsidies) the economic gains are always greater. In terms of biodiversity, it appears there is an improvement at low and medium levels of taxation, but at higher levels there is a negative ecological effect. This is because at high levels there will be a much greater amount of grassland than cropland and many farmland birds that are adapted to cropland would be adversely affected.

The results indicated that both ecological and economic benefits from farmland are affected by public policies for agriculture and land-use. The research only looked at basic policies, i.e. subsidies and taxes, but these appeared to be central to shaping both biodiversity and income.

The researchers highlighted that the results depend on the indicators used, for example, an indicator for general bird biodiversity (STOC) could not distinguish between all policy options, but an indicator for specialist farmland birds (Farmland Bird Index) could. They conclude that balancing ecological and economic performance is possible, but not straightforward. Issues that policy makers need to consider are the synergy between different policies, the time frame of the instruments, which indicators to use and how to prioritise between ecological and economic objectives.

1. See: http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&plugin=1&language=en&pcode=tsien170

Source: Mouysset, L., Doyen, L., Jiguet, F., *et al.* (2011) Bio economic modelling for a sustainable management of biodiversity in agricultural lands. *Ecological Economics*. 70:617-626.
Contact: mouysset@mnhn.fr
Theme(s): Agriculture, Biodiversity

Liste des figures

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Résumé - Les changements de l'agriculture, comme l'intensification et la déprise agricole, ont entraîné des modifications importantes de la biodiversité (déclins, extinctions et homogénéisation biotique). Dans ce contexte, la réconciliation des activités anthropiques avec le maintien de la biodiversité se révèle être un enjeu majeur. Cependant, les politiques agricoles mises en oeuvre n'ont pas permis à ce jour une gestion efficace de la biodiversité. En s'appuyant sur une démarche bio-économique, cette thèse cherche à contribuer au débat sur les politiques publiques au défi d'une gestion durable de l'agriculture et de la biodiversité. Les modèles systémiques et multi-échelles développés couplent des dynamiques écologiques et économiques au travers de variables d'occupation des sols à l'échelle de la petite région agricole pour l'ensemble de la France. La biodiversité est appréhendée au travers d'une communauté de 34 oiseaux communs, sans se restreindre à des espèces emblématiques. La calibration des modèles repose sur des séries temporelles agro-économiques et écologiques de 2001 à 2009. Différents scénarios sont simulés à l'horizon 2050 et leurs performances bio-économiques sont comparées grâce à un ensemble d'indicateurs, évitant l'écueil de la monétarisation de la biodiversité. Une réflexion est menée sur l'efficacité des indicateurs à caractériser correctement l'état de la communauté. Enfin, différentes facettes de la durabilité sont explorées grâce à la combinaison des approches coût-efficacité et de contrôle viable. Cette étude met en évidence qu'il est possible d'améliorer conjointement les performances écologiques et économiques par rapport aux tendances actuelles. D'une part, des incitations économiques à l'échelle macro-économique peuvent orienter directement les choix des agriculteurs vers des pratiques plus favorables à la biodiversité. D'autre part, des mécanismes de diversification, en réponse au risque économique, ont une influence positive sur les performances bio-économiques. En revanche, il apparaît difficile de maximiser simultanément objectifs économiques et écologiques. La thèse explore alors plusieurs options afin de sortir de cet arbitrage bio-économique. Notamment, une analyse des coûts publics suggère que l'intégration d'objectifs écologiques dans les politiques publiques produit un double dividende bio-économique. Enfin, dans un contexte d'incertitude, l'approche de co-viabilité permet d'identifier des scénarios conduisant à une agriculture multi-fonctionnelle et équitable entre les générations. L'extension de ces approches bio-économiques vers le concept de services écosystémiques devrait apporter des éclairages complémentaires pour la construction de politiques publiques relevant le défi de la biodiversité.

Mots clefs - Bio-économie, Biodiversité, Agriculture, Modélisation, Coût-efficacité, Viabilité, Oiseaux, Occupations des sols, Indicateur, Risque.

Abstract - Global changes such as the climate change, the agriculture evolution or the urbanization, have exerted significant pressures on biodiversity (declines, extinctions, and biotic homogenizations). In this context, reconciling human activities with a sustainable biodiversity turns out as a main issue. To respond to this objective, the development of bio-economic analysis appears as an interesting perspective for public policies facing biodiversity. However the agri-environmental policies have not yet been able to provide a relevant management of biodiversity. Based on a bio-economic process, this PhD intends to contribute to the debate on public policies facing the challenge of a sustainable management of agriculture and biodiversity. The systemic models developed in this work combine both ecologic and economic dynamics through land-use variables at the small agricultural area scale across the whole France. In our case, biodiversity is perceived as a community of 34 common birds, avoiding an emblematic species-based approach. Calibration of the models is based on agri-economic and ecological time series from 2001 to 2009. Different scenarios are generated up to 2050 and their bio-economic performances are compared through a set of indicators, avoiding the problem of biodiversity monetization. A focus on the choice of these indicators has been driven to characterize correctly the status of communities. In particular, different aspects of the sustainability are explored by combining cost-effectiveness and co-viability approaches. This study shows that it is possible to improve simultaneously ecological and economic performances in comparison to the current trends. In the one hand, economic incentives at the macro-economic scale guide directly the farmers' choices towards more biodiversity-friendly activities. In the other hand, diversification mechanisms, in response to economic risk, have also a positive influence on the bio-economic performances. However, it appears difficult to maximize simultaneously economic and ecological objectives. The PhD explores several options to overcome this bio-economic trade-off. In particular, an analysis based on public costs suggests that the integration of ecological objectives in the public policies generates a double dividend. Finally, in an uncertain context, the approach of co-viability allows us to identify scenarios leading to a multi-functional agriculture that remains fair through generations. Extending these bio-economic approaches towards the concept of ecosystemic services should bring further insight into the design of public policies achieving a sustainable biodiversity.

Keywords - Bio-economics, Biodiversity, Agriculture, Modelling, Cost-effectiveness, Viability, Bird, Land-use, Indicator, Risk.